Publication No. 03-149-238

WETLANDS ON CLAY SETTLING AREAS

FINAL REPORT

Prepared by

UNIVERSITY OF FLORIDA
The Howard T. Odum Center for Wetlands

under a grant sponsored by

Florida Institute of Phosphate Research

November 2010

The Florida Institute of Phosphate Research was created in 1978 by the Florida Legislature (Chapter 378.101, Florida Statutes) and empowered to conduct research supportive to the responsible development of the state's phosphate resources. The Institute has targeted areas of research responsibility. These are: reclamation alternatives in mining and processing, including wetlands reclamation, phosphogypsum storage areas and phosphatic clay containment areas; methods for more efficient, economical and environmentally balanced phosphate recovery and processing; disposal and utilization of phosphatic clay; and environmental effects involving the health and welfare of the people, including those effects related to radiation and water consumption.

FIPR is located in Polk County, in the heart of the Central Florida phosphate district. The Institute seeks to serve as an information center on phosphate-related topics and welcomes information requests made in person, or by mail, email, or telephone.

Florida Institute of Phosphate Research
1855 West Main Street
Bartow, Florida 33830
(863) 534-7160
Fax: (863) 534-7165
http://www.fipr.state.fl.us

WETLANDS ON CLAY SETTLING AREAS

FINAL REPORT

Mark T. Brown
Principal Investigator

with

Mary Boyd, Wesley Ingwersen, Sean King, and Daniel McLaughlin

The Howard T. Odum Center for Wetlands
UNIVERSITY OF FLORIDA
Gainesville, Florida 32611

Prepared for

FLORIDA INSTITUTE OF PHOSPHATE RESEARCH
1855 West Main Street
Bartow, Florida 33830 USA

Project Manager: Steven G. Richardson, Ph.D.
FIPR Contract Number 03-03-149

November 2010

DISCLAIMER

The contents of this report are reproduced herein as received from the contractor. The report may have been edited as to format in conformance with the FIPR *Style Manual*.

The opinions, findings and conclusions expressed herein are not necessarily those of the Florida Institute of Phosphate Research, nor does mention of company names or products constitute endorsement by the Florida Institute of Phosphate Research.

PERSPECTIVE

Clay settling areas (CSAs) have typically occupied 40% or more of the post-mining landscape. Virtually all have wet areas. Can wetlands be established on CSAs that have some or all of the ecological functions and values of unmined wetlands? Can wetlands on CSAs serve as mitigation for wetlands disturbed or destroyed by mining?

The phosphatic clays are very fertile, and a wide array of plants can grow on these clays. Hydrology is the key factor in supporting a wetland and is related to topography and the characteristics of the watershed feeding each wet depression. The topography of a CSA is related to the inflow and outflow locations plus the presence and configuration of old spoil piles remaining when a CSA is constructed on mined land. The coarser particles in the clay slurry tend to settle-out nearer the inflow, and the finer particles end up closer to the outflow. This results in a slight slope from inflow to outflow areas when the CSA is decommissioned and the clay consolidates. Internal spoil piles in the CSA affect depth of clay deposits and result in differential settling, which produces a complex surface topography with multiple depressions and multiple watersheds.

CSAs are usually elevated above natural grade and are somewhat hydrologically isolated from the surrounding terrain. Thus, the water available to support wetlands on a CSA comes from the precipitation falling on that CSA. Permeability of the surface soils is important. Large desiccation cracks that may fill with organic matter or sand (especially in sand-clay settling areas) or old root channels allow some lateral movement of water through the surface 0.5 to 1.0 meter of the clay. Some CSAs may be partially capped with sand tailings or overburden, which may allow a more sustained supply of moisture via groundwater seepage versus the flashy hydrologic nature when the watershed soils are all clay. For more information on CSA hydrology, in addition to this study, see reports for FIPR Project 03-03-150, "Hydrology of Clay Settling Areas."

Although hydrology is of prime importance, development of wetland vegetation communities and development of wetland soils are also important. We wished to learn from older planned and unplanned (volunteer) wetlands and also to develop new information that may guide us in establishing quality wetlands on CSAs and sand-clay mix settling areas. The goals of the project were to:

- Document the current status of wetlands on CSAs, including vegetation community structure, soils, microclimate, and hydrology
- Link vegetation communities to hydrology through on-site monitoring
- Document survival and growth of tree species that were planted on CSAs about twenty years ago
- Conduct field plantings to test and evaluate techniques for creating new wetlands and enhancing existing wetlands
- Develop temporal and spatial models that predict the depth, duration, and spatial extent of flooding on CSAs

Dr. Steven G. Richardson
Reclamation Research Director

ABSTRACT

This project was a five-year study of wetlands on Clay Settling Areas (CSAs) aimed at developing an understanding of their ecology and hydrology to apply when restoring functional wetland systems. Characterization of wetlands naturally occurring on CSAs found that wetland plant communities were associated most strongly with hydrology. Properties of typical wetland soils were developing with time on studied CSAs, although these wetlands were unlike reference wetlands in species composition, bathymetry, or hydrology. Multi-year monitoring of hydrology and collection of climatic data in order to create water budgets found unexplained losses that were later accounted for primarily through evapotranspiration, secondarily through infiltration at some sites, and negligibly through dikes. Spatial and temporal models of water features were created that performed best when considering CSAs as multiple watersheds with unique runoff and groundwater interactions. Ecohydrology studies revealed high transpiration and deep rooting across the hydrologic gradient by *Salix caroliniana* and documented the relationship between water availability and water table depths in different CSA substrates. Monitoring of old and new field trials yielded lists of wetland species appropriate for introducing to CSAs as well as recommendations for establishment in relation to hydrologic and biotic factors. This research produced knowledge, tools, and guidelines which can be used for wetland establishment on CSAs.

ACKNOWLEDGEMENTS

We would like to express our gratitude first to our field assistants Elliot Campbell, Lance Ingwersen, Tyler Hollingsworth, Shannon King, and Julie McLaughlin, who helped us in challenging conditions to collect the five years of data underlying this report. Thank you to the nursery owners, foremen, and their crews, in particular Jay Allen, who donated a large quantity of plants and assisted in planting efforts for the common interest of this project. Thank you to the mining company representatives who provided site information and access, data, and conveyed to us their significant reclamation experience with Clay Settling Areas; in particular we would like to mention Rosemarie Garcia and Keith Hancock at Mosaic, John Wester at PCS, Gary Blitch at CF Industries, and Tim King at the Tenoroc Fish Management Area. We thank the members of our Technical Advisory Committee for guidance as we progressed through the project. And finally we would like to thank FIPR, who funded this research and provided strong foundation of decades of phosphate mining reclamation research to build upon.

TABLE OF CONTENTS

TABLE OF CONTENTS (CONT.)

TABLE OF CONTENTS (CONT.)

TABLE OF CONTENTS (CONT.)

TABLE OF CONTENTS (CONT.)

TABLE OF CONTENTS (CONT.)

TABLE OF CONTENTS (CONT.)

TABLE OF CONTENTS (CONT.)

LIST OF FIGURES

LIST OF FIGURES (CONT.)

Figure **Page**

LIST OF FIGURES (CONT.)

Figure **Page**

LIST OF FIGURES (CONT.)

Figure **Page**

LIST OF FIGURES (CONT.)

Figure **Page**

LIST OF FIGURES (CONT.)

LIST OF FIGURES (CONT.)

Figure **Page**

LIST OF FIGURES (CONT.)

Figure **Page**

LIST OF FIGURES (CONT.)

Figure **Page**

LIST OF TABLES

LIST OF TABLES

Table **Page**

LIST OF TABLES

Table **Page**

LIST OF TABLES

Table **Page**

EXECUTIVE SUMMARY

This research project was a five-year investigation of wetlands on clay settling areas (CSAs) to develop knowledge and understanding of their ecology and hydrology for establishing functional wetlands and to suggest ways to enhance their creation. In short the research has:

- Collected field data documenting the current status and historical trends of wetlands naturally establishing on CSAs,
- Evaluated CSA hydrologic regimes and their major determinants,
- Evaluated the interrelationships between CSA vegetation, soil, and hydrology,
- Developed temporal and spatial models that predict the depth, duration, and spatial extent of flooding on CSAs,
- Documented survival and growth of wetland trees and ecosystem development on CSA sites planted over 20 years ago,
- Conducted field trials with revegetation of desirable species and subsequent monitoring of growth and survival, and
- Synthesized project findings into guidelines for enhancement of existing wetlands and the creation of new ones on CSAs.

The following provides a summary of the findings of the study:

- ***Documentation of the current status of wetlands naturally establishing on CSAs including vegetative community structure, soils, and hydrology***

Areas that meet hydrologic, soil, and plant community definitions for wetlands occupied substantial portions of abandoned CSAs. The majority of these wetland areas were dominated by a limited and predictable set of ubiquitous hydrophytic species organized along hydrologic gradients including: *Baccharis halimifolia*, *Ludwigia peruviana*, *Typha spp., and Salix caroliniana.* Topography and wetland bathymetry substantially varied among the monitored CSAs which, along with the presence or absence of an active outfall structure, determined the spatial extent and hydroperiod of wetland areas. At least in the initial few decades following decommissioning, the topography and resulting hydrologic regimes, in addition to seed sources in surrounding landscapes, were more important than available nutrients in determining wetland development. Water availability, abundant soil nutrients, and the presence of early successional species likely contributed to the high plant densities and biomass found in CSA wetlands. Indicators of wetland soil development were documented, such as soil pH shifting from alkaline to neutral, increasing organic matter and balancing N:P ratios with site age.

Monitoring of water levels and identification of seasonal high water level (SHWL) indicators along vegetation transects established across wetlands areas was conducted on 17 CSAs to establish water level ranges experienced by the dominant plant species. The results revealed that *Baccharis halimifolia* occurred in the more transitional areas and rarely experienced inundation. *Ludwigia peruviana* was typically observed in inundated conditions and with flooding depths ranging from 0 to 0.5 m. *Typha spp.* was found occurring in areas ranging from saturated conditions to inundation depths over a meter. *Salix caroliniana* was typically the dominant species and was found in the largest range of hydrologic regimes, ranging from unsaturated conditions with no signs of past inundation to areas flooded over a meter in depth. Wetland areas with seasonally fluctuating hydroperiods tended to have higher diversity compared to features experiencing permanent, deep inundation. The latter were often dominated by monotypic stands of *Salix caroliniana* along with a dense cover of floating aquatic species.

- ***Evaluation of the Major Determinants of CSA Hydrology***

Hydrologic evaluations of eight CSAs using surface water, groundwater, and climatic data were performed to calculate water budgets of potential wetland areas. Groundwater elevations relative to surface water and lateral hydraulic conductivities determined by slug tests were used to investigate the interactions between the surface water and local groundwater systems. Overland inflow following storm events was empirically related to rainfall intensity and antecedent moisture conditions. Climatic-based empirical models and analysis of continuous surface water data were used to estimate daily evapotranspiration (ET) losses. Groundwater profiles and lateral hydraulic conductivities within the dikes suggested negligible lateral outflow through the dikes. Diurnal analysis of surface water levels along with groundwater elevations indicated that connection between the local groundwater and surface water systems was variable among the monitored CSAs, and both groundwater inflow and outflow to wetland features was documented. Runoff analysis highlighted differences in characteristic runoff responses among and within CSAs and demonstrated that runoff could be accurately predicted solely as a function of rainfall. Runoff amounts were found to be strongly affected by surrounding upland soil type, with more conductive soils reducing responses to rain events. Water budget calculations resulted in significant residual losses that varied among and within CSAs, which suggested underestimation of summer ET when using traditional empirical models. Analysis of the diurnal surface water fluctuations supported these findings and resulted in ET rates as much as two times greater than both typical summer values in the region and rates estimated with climatic models, demonstrating the high productivity of these systems.

Topography and upland soil type determined the two major hydrologic regimes observed on the CSA study sites. In CSAs with steeper gradients across their lengths, water tended to accumulate mainly in depressions near outfall structures, where greater depths, less fluctuation, and larger spatial extent of flooding was experienced compared to flatter CSAs, which had many watersheds and thus multiple surface water features. The latter were characterized by flashy hydroperiods with greater response to rain events and faster rates of decline. Wetland features, even within the same CSA, often exhibited different hydroperiods depending on their surrounding upland soils. Features with sand tailings or overburden deposited in the surrounding uplands experienced much more buffered hydrologic regimes with less fluctuation and greater depths compared to features surrounded by pure clay. The sandier upland created less surface runoff and for at least one site studied, provided significant local groundwater flow into the surface water feature.

- ***Ecohydrology: relationships between CSA vegetation, soils, and hydrology***

To relate hydrology with the biota, transects were established across a hydrologic gradient from upland into the surface water features, where vegetation was monitored and a series of soil moisture probes were installed at different depths. Root biomass allocation with depth and transpiration of *Salix caroliniana* were measured along these transects to quantify effects of the hydrologic regime on plant behavior. Moisture release curves were developed with laboratory analysis to relate soil moisture to water potential and to determine capillary fringe heights, saturation values, and soil moisture levels in the clay soils that may induce permanent wilting. Saturation levels and capillary fringe heights were higher than those of typical clay soils. Soil moisture data demonstrated the large capillary forces of the clayey soils, with saturation levels occurring over a meter above the water table. Permanent wilting points, however, were often observed within the top 10-25 cm of the soil profile. Root biomass allocation was only slightly related to soil moisture levels, and roots were found at depths over one meter and into the water table with little preclusion from the clays. The results from the transpiration studies supported the evidence of high ET rates of these systems, and found that this was in large part due to the transpiration of *Salix caroliniana*. These results revealed attributes of *Salix caroliniana* that, along with high soil moisture levels through the soil profile, contributed to its success on CSAs.. Furthermore, these results demonstrated the presence of broad transitional (saturated) zones across clay uplands that are appropriate for wetland tree species planted with adequate initial rooting depth.

- ***Temporal and spatial models that predict the depth, duration, and spatial extent of flooding on CSAs***

The hydrologic evaluations and water budget analyses were used to create temporal hydrologic models that predict daily water levels with coefficients of determination exceeding 0.87. The models required high ET rates using applied seasonal coefficients to balance the water budget, which supported the results gleaned during the diurnal analysis of surface water data and transpiration studies. Furthermore, variable groundwater inflow and outflow rates were required, again highlighting the effect from surrounding upland soil type. The models can be applied as tools to evaluate design considerations of CSAs such as upland fill type, watershed configuration, and outfall elevations.

A spatial hydrologic model was developed using GIS software to simulate the hydrology of CSAs using high resolution topographic maps (1-2.3 m^2/cell) and local rainfall data. To accurately distribute water across a CSA, a multiple watershed approach was required due to the presence of multiple depressions within a CSA. The results from the spatial model included a time-series of maps of depth and spatial extent of surface water that were further synthesized into hydropattern maps which illustrated the frequency of inundation spatially. The model determined the locations and total area of a given CSA watershed that had wetland hydrologic characteristics. Since the model provides spatial data in terms of frequency and depth of inundation, hydropattern and water depth maps are easily extracted which can guide plans for wetland creation and/or enhancement by aiding in plant species selection and positioning. The spatial model can also be used at a larger scale for entire CSA planning if adequate long-term hydrologic data and accurate high resolution topographic maps are available.

- ***Survival and growth of wetland trees and ecosystem development following 20 years after planting***

Monitoring of sites planted with wetland trees over 20 years ago was conducted on five CSAs. Survival and growth of planted trees and ecosystem development in terms of soil condition, canopy structure, and diversity of both planted and volunteer species were documented. The results demonstrated that species including *Taxodium distichum*, *Fraxinus spp.*, and *Nyssa aquatica* had approximately 30% average survival and significant growth across many CSAs of different ages, construction, and fill type, with up to 80% survival of certain species in optimal conditions. Survival of species planted in shallower transitional areas was negligible, possibly due to increased vulnerability to disturbances such as fire and drought. Expansion of populations appeared to be limited by very low seedling recruitment rates, with evidence that wetlands had hydroperiods not amenable to seedling survival. This limited recruitment was found to be the driving factor in predicted long-term population stasis or decline based on a population model built for two sites. Planted trees formed dense canopies which enhanced the microclimate and added more structure to CSA canopies than provided by pioneer species. Planted trees moderated soil organic matter accumulation binding up a greater percent in biomass. These results indicated that the most significant factors in ecosystem development are hydrology and landscape level influences, such as lack of seed sources.

- ***Field trials of vegetation planting, monitoring growth and survival of desirable species***

Five field trials were designed and implemented with a high diversity of herbaceous and tree species on sites exhibiting a range of hydrologic conditions and existing vegetation. Depressional water features with no canopy were planted at two sites with herbaceous marsh species and a periphery of trees and shrubs. Wetland tree seedlings of 23 species were planted under an existing *Salix caroliniana*-dominated canopy at three sites. Species were positioned along the wetland features' hydrologic gradients based on species moisture tolerances, site topography, and hydrologic modeling. During three years of monitoring the field trial sites, certain species were observed that were suited for wetland revegetation in open depressional marsh features and/or underneath existing wetland forest canopies. Lack of success by other planted species, however, should not indicate inappropriateness of these species on CSA wetlands due to the severe drought conditions experienced over the period of record. Marsh features were affected by sudden increases in water depth and prolonged periods of drought, which caused the mortality of several planted species and allowed previously cleared herbaceous and woody volunteer species to heavily encroach at planting sites.

Different pre- and post- volunteer species management and more controllable hydrologic conditions could have resulted in better survival and growth of planted species at marsh sites and should be considered in future plantings. Tree species from a variety of wetland ecosystems were able to successfully establish at planting sites. For most species, seedling survival was higher under a stable canopy than in full sunlight, likely due to less competition from volunteer species and a more suitable microclimate. Appropriate planting positions, based on hydroperiod, were identified for a variety of wetland tree species on CSA wetland features.

- ***Enhancement of existing wetlands and the creation of new ones on CSAs***

While the development of restoration plans for wetlands on CSAs will need to be site-specific and will require a certain amount of detailed topography and hydrology data, some general principles are emerging. Existing vegetation on CSAs can provide references of longer term conditions and be used as guidance for site selection and planting, and may provide structure that facilitates establishment of planted species. On sites without active outfall structures, enhancement of existing wetlands will be restricted by the lack of controllable hydroperiods, although suitable hydrologic conditions exist on some older sites to support supplementing wetland communities. Suitable hydrologic conditions are characterized by fluctuating hydroperiods which aid in soil aeration and seedling recruitment which can be maintained in the long term with flexible outfall structures. Planting success in existing CSA wetlands has been demonstrated by long-term survival of trees, especially those adapted to longer periods of inundation. Current field trials have identified a wider range of species, including herbaceous, shrub and other tree species found in native Florida wetlands that are suitable for establishment on CSAs. A better understanding of interactions between root depths, transpiration, and soil water availability suggests the incorporation of wide transitional zones into restoration efforts that extend beyond surface water features where conditions are appropriate for wetland tree species. Designing CSAs soon after decommissioning, with the goal of creating wetlands communities, may offer new opportunities for wetland establishment with greater flexibility over plant community composition and hydrologic regime. Certain design tools and principles have been identified during this study which can be employed on older and more recently reclaimed CSAs, including:

- Site specific analysis of watershed configuration and upland soil types
- Overburden/sand positioning to buffer otherwise flashy hydroperiods
- Spatial and temporal models for prediction of hydrologic regimes and/or CSA design in terms of topographies and outfall elevations
- Plant species selection and positioning
- Recognition of broad transition zones appropriate for wetland tree species
- Inclusion of gradual gradients into wetland areas and microtopographic relief to enhance vegetation recruitment
- Utilization of canopy cover from existing vegetation to enhance restoration success

RECOMMENDATIONS FOR FURTHER RESEARCH

The project has been successful at increasing our understanding of the interplay between the ecology, hydrology and physical characteristics of CSA soils and synthesizing the findings into design knowledge. The design tools identified in this study need to be evaluated using a system-level approach on a site-scale and on more recently decommissioned CSAs. Design tools include positioning of overburden, installation of flexible outfall structures, grading to achieve optimal topographies, hydrologic modeling, and plant species selection and positioning. Additionally, continued monitoring of the field trials conducted in this study will provide more long-term data of survival and ecosystem development following revegetation with a wide range of wetland herbaceous and tree species. Finally, further research is needed to investigate ecohydrologic characteristics of desirable plant species to identify possible challenges or opportunities for their establishment on CSAs as well as any changes in CSA ecohydrology, particularly transpiration rates, that may be induced by changes in community assemblages.

INTRODUCTION

STATEMENT OF THE PROBLEM

An estimated 40% of the post mining landscape may consist of Clay Settling Areas (CSAs) (Richardson 2005). Given this large spatial footprint across the landscape, it is important to understand how these landforms might fulfill ecosystem and landscape functions. It has been suggested that functional wetlands might be established on CSAs, yet little is known about the current or long-term status of wetlands that have been created in the past or about those that have developed through natural processes. If functional wetlands are to be created on CSAs, it will be important to document, evaluate, and analyze their biophysical conditions, not only as they exist today, but also how they change over time. In addition, it will be necessary to document successful methods of creating wetlands on CSAs; including appropriate vegetation species, techniques of planting, hydrologic prediction, and maintenance.

This research addresses a number of questions that consider the possibility of fulfilling ecosystem and landscape function and the needs of society. First, can functional wetlands be established on CSAs? What is the current status of wetlands on CSAs including those established naturally and those where interventions occurred to enhance vegetation, hydrology, and/or soils? Should vegetation be cleared before planting wetlands on CSAs? What are the temporal and spatial characteristics of the hydroperiods of CSAs? What vegetation types and planting methods are appropriate for the hydroperiods exhibited by CSAs? What are the appropriate methods for enhancement of existing wetlands? What methods and techniques need to be implemented for the creation of new wetlands on CSAs?

The overall goal of this research is to evaluate wetland development on CSAs. Application of this research will be used to suggest methods and techniques of establishing and/or enhancing functional CSA wetlands. Several of the FIPR 1998-2003 Strategic Research Priorities are addressed, including objectives and approaches related to Environment as follows:

Approach 2. Further develop mapping, modeling and related visualization tools and databases to assist in the evaluation and implementation of ecological and hydrological system restoration.

Approach 3. Further develop techniques for reclaiming and restoring mined lands to improve their functioning and to facilitate their integration into larger landscapes and ecosystems, including habitat networks and greenways.

In the Reclamation/Restoration area this research project addresses the following approaches:

Approach 1. Further develop techniques and recommendations for reducing mining impacts and for reclaiming and restoring critical habitats and ecological systems, including wetlands, streams, lakes, xeric uplands, flatwoods, etc.

Approach 4. Further develop appropriate techniques for post planting vegetation management on uplands and wetlands, including weed control.

Approach 9. Further develop techniques and recommendations for reclaiming CSAs that will enhance their hydrologic functioning and increase their usefulness for wildlife habitat, forests, wetlands agriculture, etc.

PLAN OF STUDY

This research project was a four-year investigation of wetlands vegetation and hydrology of CSAs including sand-clay mix areas. The plan of study included the following six linked investigations of the physical, chemical, and biological aspects of CSAs:

- Document the current status of wetlands on CSAs including vegetative community structure, soils, micro-climate, and hydrology;
- Link vegetation communities to hydrology through on-site monitoring of hydrology;
- Document survival and growth of tree species that were planted on CSAs 20 years ago;
- Conduct field plantings to test and evaluate techniques for creating new wetlands and enhancing existing wetlands;
- Develop temporal and spatial models that predict the depth, duration, and spatial extent of flooding on CSAs.

The outcome of this investigation is to use the knowledge gained to facilitate long-term, functional wetlands as parts of functional landscapes following phosphate mining by providing recommendations for the enhancement of existing wetlands and the creation of new ones on CSAs.

REVIEW OF THE LITERATURE

Uses of Clay Settling Areas

Various uses for CSAs have been proposed and researched over the years including silviculture, grazing, and establishment of natural communities for wildlife and recreation. Feiertag, in Odum and others (1990), reported success in growing native Atlantic white cedar (*Chamaecyparis thyoides*) on a CSA. Growth rates in that study suggested that plantations of this species may potentially yield economic returns. However, as of 1988, most reclamation projects had converted CSAs to pasture (Rushton 1988). Creation of valuable wildlife habitat or recreational areas on CSAs has also been explored (King and others 1980), as have ideas relating to accelerating natural succession

(Butner and Best 1981; Kangas 1981). In fact, Schnoes and Humphrey (1980) noted that re-establishing natural communities may be a more logical and economically sound endeavor versus other possible approaches.

Early Studies of CSA Reclamation

Studies encompassing both natural and engineered succession on CSAs began with Farmer and Blue (1978) who found that reclamation presents difficult problems due to the unstable and highly colloidal qualities of clay. Furthermore, clay soil characteristics act to deter effective land recovery in the phosphate mining district (Lamont and others 1975). In the early 1980's, much research was devoted to methods for rapid dewatering of CSAs (Carrier 1982; Lamont and others 1983; Pittman and Sweeney 1983; Garlanger and Babcock 1987). However, due to the high water holding capacity and slow compressibility of clay soils, the most effective stabilizing factor is likely the passage of time. Reigner and Winkler (2001) suggest 90% of consolidation occurs within five years following clay deposition.

In studies of the Alderman's Ford Ranch site in the late 1980s and early 1990s, which is one of the oldest CSAs in central Florida that was abandoned in the early 1950's, Odum and others (1991) suggested the site exemplifies the result of natural succession on a CSA over many years. From observations of individual well records, profiles, surfaces, and aquifer characteristics several things were apparent. The site was drying out in the deep clay area near the high west dike. However, the water table in the clay soils behaved in a predictable manner. The overall groundwater flow pattern on the site reflected the original topography and groundwater flow of the area. The old clay-settling pond had become a part of the landscape, and did not function independently. Water table levels in many parts of the pond appeared to have stabilized, to the point where predictions could be made for vegetation development.

Reclamation processes can be extremely labor intensive and costly due to the perceived need to extirpate all volunteer vegetation and begin the reclamation process with brand new plantings on bare soil. Rushton (1988) found there were no discernible long-term benefits to the success of seedling establishment by removing herbaceous ground cover at the time of planting. Further, local site conditions may be more important in growth and survival than the effect of removing vegetation. Rushton also noted that small grazing mammals may work to retard the development of complex forests, allowing only fast growing *Salix caroliniana* (willows) to succeed, yet, a willow-dominated community may provide less suitable habitat for grazers and perhaps, in these willow communities natural succession may more readily proceed to a rich forest ecosystem. Overall, growth was better when planted under some tree canopy of willows and, in fact, clearing the plots and removing the canopy increased competition from the herb layer (1988).

Succession on Clay Settling Areas

Rushton (1983) observed that some sites showed rapid re-establishment of wetland vegetation comparable to succession on disturbed soils without mining. Furthermore, where hydrology was right, typical wetland hardwoods were found developing in 30 years. Usual natural succession follows the course of an initial cover of cattails (*Typha spp.*) and common water-hyacinth (*Eichhornia crassipes*) followed by primrose-willow (*Ludwigia peruviana*) and Carolina willow (*Salix caroliniana*). Wax myrtle (*Myrica cerifera*) and a profusion of vines come about as sites continue to dry (King and others 1980; Schnoes and Humphrey 1980; Zellar-Williams and Conservation Consultants 1980; Butner and Best 1981; Gilbert and others 1981; Rushton 1983). Without a close source of genetic material CSAs may remain in arrested successional states dominated by early successional herbaceous and shrub species. If there is a relatively close seed source, hardwood species such as red maple (*Acer rubrum*) and laurel oak (*Quercus laurifolia*) may colonize older CSAs (Zellars-Williams and Conservation Consultants 1980; Rushton 1983).

In an extensive study on the natural succession of CSAs, Rushton (Odum and others 1991) examined vegetation, hydrology and soils at several clay settling ponds of various ages and in various stages of ecological succession. Her study showed that marsh and willow ecosystems within CSAs were similar to marsh and early successional swamp ecosystems in unmined areas. Equally important was her documentation of species and ecosystem functional attributes found on CSAs that were similar to bottomland hardwood systems of lower floodplains. She concluded that reforestation to swamps appears to be a suitable alternative for reclamation of clay settling ponds to replace some wetlands destroyed by mining. Her observations that hydric hardwood communities can grow well on drier locations in clay settling ponds seems a fitting reclamation avenue to pursue, in combination with wetland reclamation, since the trend over time is that only a percentage of CSAs remain wet.

CSA Hydrology

Probably the most important question regarding creating wetlands on CSAs is predicting long-term hydrology. Researchers have periodically studied hydrology, dewatering and surface stabilization over the last 20 years. However, little research has focused specifically on establishing wetlands and their required hydrology. CSAs have been observed to be slow to dewater, compact, and stabilize, and it is not yet certain how to accurately predict when they will stabilize in terms of their hydrology and final ground surface elevation (Reigner and Winkler 2001). While methods to rapidly dewater CSAs have been explored in the past, questions surrounding when hydrology on CSAs stabilizes and how much of the area will retain characteristic wetland hydrology need be addressed. The need to dewater and stabilize the clays may be counter to achieving long-term viable wetlands on large areas of CSAs. Clearly, more research is needed.

A USGS study (which included four CSA basins) collected data at each basin using a streamflow-gaging station, a rainfall-recording gage, continuous water-level recorders at wells open to the surficial and intermediate aquifer systems, and periodic measurement of water levels in 10 to 13 shallow observation wells (Lewelling and Wylie 1993). They found that water levels did not fluctuate in response to variations in stream flow indicating that the surface-water and ground-water systems have little to no hydraulic connection (Lewelling and Wylie 1993). These findings support the notion that after mining and stage filling with clay waste, these tracts of land become isolated from existing groundwater networks.

Odum and others (1991) studied transpiration and modeled relationships between vegetation and hydrology. Transpiration studies were conducted using near infrared reflectance as an indicator to understand the role transpiration plays in a CSA water budget and how it relates to dewatering and vegetation restoration. Odum and McClanahan, in Odum and others (1991), suggested that different types of vegetation regulate their heat budget by varying their infrared reflectance. Plants adapted to low nutrient environments may have higher near infrared reflectance capabilities in response to low nutrients and thus low transpiration rates. The opposite may be true for plants adapted to high nutrient environments. Their studies may be important relative to long-term water budgets on CSAs and point to the fact that transpiration may be one of the most critical variables in CSA hydrology.

Several researchers have studied aspects of CSA hydrologic behavior after use has ceased. Some influencing factors on water budget of a CSA are precipitation, transpiration, evaporation, permeability and hydraulic conductivity of the soils, soil matrix composition, surface slope, runoff, and groundwater discharge (the first three being the major contributors). Bromwell and Carrier, Inc. (Reigner and Winkler 2001) conducted a two-year study collecting field data in order to develop a hydrologic model for assessing and predicting CSA behavior. The joint effort between the consulting company and USGS (whose 1996-97 data served to calibrate and verify hydrologic models) shed some light on factors involved in clay consolidation and hydrology prediction. A few of the major findings were that hydrologic predictability is largely dependent on the percent of solids in soils and that compressibility and permeability control the magnitude and rate of consolidation. The post-mining infiltration rate was found to be very low (~0.5 cm/yr). Therefore, all other things being equal, surface water might be expected to increase as CSA clays compress. Data collection in this study combined evaporation and transpiration so it is unclear how much each contributes to the water budget. If vegetation can be established successfully, then transpiration may be a factor in the removal of water from a CSA.

Further, cracks in the clay substrate may play an important role in CSA hydrology. If cracks extend to overburden piles left within the CSA, an increase in horizontal and vertical flow of surface waters may result. Reigner and Winkler (2001) found that actual CSA water storage capacity turned out to be greater than predicted. This may be partially attributable to that fact that the commonly used Bromwell and Carrier model does not account for cracks and internal pooling necessarily eliminated in the interest of experimental efficiency. In addition, a 0.9-1.5 m change in water level was observed over the two year observation period. The reason for declining water tables are unclear at present, but could be attributed to the combination of unaccounted flow via overburden piles and transpiration. Indications have been made that more field data are needed to verify model assumptions (Reigner and Winkler 2001) warranting further exploration into the true effects of cracking and transpiration on hydrology. Miller's study (Odum and others 1991) covering groundwater control on CSAs found that water tables were stable or dropped slightly in clay areas that were more internally drained, not adjacent to dikes, or lower areas. Many CSAs have spoil piles or dikes near where trees are planted, causing two-dimensional drainage occurring when spoil piles are left (Carrier 1982). This increased drainage could help improve consolidation. On the other hand, it may also increase draw-down rate. This would be especially true where there are interior spoil piles, pre-regulation dikes that allow seepage, and a regional water table below the surface of the pond (Odum and others 1991).

Planting of Wetland Trees on Clay

Planting species characteristic of mid to late succession is one method to direct the successional process (Brown and Tighe 1991). Monitored field trials on CSAs using wetland tree species began in the 1980s (Rushton 1988; Paulic and Rushton 1991a; Everett 1991), and tree survival and growth was documented during the initial years after planting. Water availability, species properties, tree size, and edaphic factors including soil age and nutrient levels have all been shown to affect tree survival on clay settling areas. The following list summarizes findings of earlier studies of wetland trees on CSAs.

- Hydrology was more important in determining tree survival than canopy or understory cover (Rushton 1998; Paulic and Rushton 1991b).
- Wetland trees typical of floodplain and backwater swamps of central and northern Florida have had greater than 50% survival after 1 year on clays, including *Acer rubrum*, *Betula nigra*, *Carya aquatica*, *Liquidambar styraciflua*, *Quercus laurifolia*, *Quercus lyrata*, *Quercus michauxii*, *Sabal palmetto*, and *Ulmus americana*. (Paulic and Rushton 1991b)
- *Fraxinus spp.* and *Taxodium spp.* had high (>80%) survival after 3 years (Paulic and Rushton 1991a; Everett 1991);
- Clay is a suitable medium for wetland species (Cates 2001);
- After three years, trees growing on a sand-clay mix and on sand had higher survival than those on clay. Trees in clay grew faster than trees in sand (Paulic and Rushton 1991a);
- Most major nutrients are available in sufficient quantities for tree growth. Nitrogen may be the limiting nutrient. N-fertilizer increased growth but had no effect on survival of *Acer rubrum* in a greenhouse experiment (Paulic 1991). Fertilizer enhanced growth of *Taxodium spp.* in clay both in the field and in the greenhouse (Everett 1991; Paulic 1991).

- Soil age was positively correlated with *Acer rubrum* growth in a greenhouse experiment (Paulic 1991);
- Animal grazing can reduce tree survival (Rushton 1988).

These earlier studies have censused planted and non-planted trees in a variety of hydrologic conditions, among different vegetation communities, and on a number of CSAs. However, these earlier studies did not monitor planted trees after more than a few years, and thus could not consider longer-term survival and growth, or the potential ecosystem function of more mature trees on CSAs. Time until maturity for forested swamps can be as long as 250 years in a natural environment. Long-term monitoring is necessary to understand the long-term dynamics of a restored forested system (Clewell 1999).

Marsh Restoration on CSAs

Another option for ecologically engineering the restoration of wetlands on CSAs may be to enhance herbaceous marsh features with greater species diversity and structure; however, little, if any, research has been devoted to revegetation of herbaceous wetland systems on CSAs. One known marsh restoration was installed in late 2001 at the Florida Power Corporation Hines Energy Complex in Polk County on a CSA; however, the project's methodology and evaluation have not been published. Although not typical on CSAs, herbaceous marsh restoration is a common goal on post-phosphate mined lands to mitigate for mining impacts. Brown and others (1997) analyzed vegetative cover data from 41 reclaimed herbaceous marsh systems on previously phosphate mined lands, excluding CSAs. Although variability in monitoring period, small sample size, and the quality of past data collected hampered a thorough statistical analysis, several findings emerged, including: (1) percent cover within the marsh systems seemed to increase and then level off after three to five years; (2) species richness, on average, was comparable with natural systems; (3) mulching tended to increase initial percent cover in herbaceous systems; and (4) initial water levels were most likely a major determinant in the success or failure of marsh vegetation restoration. The most common planted species included *Pontederia cordata* (pickerelweed), *Sagittaria lancifolia* (bulltongue arrowhead), *Spartina bakeri* (sand cordgrass), and the most commonly recruited, or "naturally occurring," were *Panicum hemitomon* (maidencane), *Pontederia cordata* (pickerelweed), and *Juncus effusus* (common rush).

Recruitment

An important ingredient for the sustainability of a constructed forested system and an indicator of the appropriateness of an environment for introduced species is the ability to propagate. Wetland trees have specific moisture requirements for successful reproduction (Mitsch and Gosselink 1993). These requirements can be important for seed set, germination, and establishment. Poor seed set may occur from pollen limitations (McLanahan 1986). Dispersal is important in order for fertilized seeds to find a viable location in which to germinate. Together water levels and microtopography are important in determining seed dispersal. Because some seeds float in water they tend to accumulate in greatest densities near the edge of water or near obstructions. Seeds of most wetland trees do not germinate in standing water. Thus areas of permanent standing water may preclude the emergence of new seedlings. In areas with infrequent drawdown, seed germination may still occur but viability of seeds may be decreased by long periods of inundation (Schneider and Sharitz 1986). If seeds are able to germinate, water conditions during the first few months can be critical to survival. Most wetland tree seedlings cannot survive extended periods of inundation.

The recruitment success of wetland trees characteristic of mid to late succession is unknown on CSAs. One direct seeding experiment on phosphate mined land was largely unsuccessful: 10 of 14 plots that were covered with litter collected from floodplains in the vicinity failed to produce seedlings (Rushton 1988). The quantity of viable seeds in the collected litter was unknown.

Ecosystem Development

A series of gradual changes in the dominant vegetation community toward a predictable climax state summarizes the traditional concept of succession. Numerous theories have emerged further elucidating the mechanisms of succession (Clements 1916, Egler 1954, Connell and Slayter 1977), and challenging its linearity and predictability (Anand and Desrochers 2004). Yet the changes in the composition of the vegetation community are just one aspect of alterations to both the abiotic and biotic environment that are associated with succession. In the context of the entire system, this dynamic process has been called ecosystem development (Odum 1969).

A key aspect in the development of an ecosystem is an increasing effect of the biotic components of the system on the modification of the environment and the selection of the biota. The increasing control exerted by the biotic components is a characteristic of self-organization (Odum 1989). The dynamics of self-organization in the "emerging" ecosystems on CSAs are unclear. Measures of the modifications that the biota are making to the environment and the changes in the community composition that may be resulting from those changes are potential indicators of ecosystem development.

In forested ecosystems, trees are key agents of influence over the local environment and thus the ecosystem. As trees mature and canopies develop, they reduce the quantity of light that is able to penetrate to the lower vertical strata of the forest. The reduction in light penetration alters the microclimate (notably temperature and humidity) underneath the tree canopy. These changes to the abiotic environment imparted by the trees may in turn cause changes in the cover and composition of the understory vegetation (Beatty 1984) and the rate of organic matter decomposition in the soil. Trees also contribute a substantial amount of the detritus that decomposes and becomes incorporated in soil organic matter (Rhoades and others 1998). In a study of carbon budgets in the Dismal Swamp, tree leaf litter and fine tree roots composed the largest annual input to the detritus pool in both cypress-dominated swamps and mixed forested wetlands (Megongial and Day 1988). All these effects are expected to be enhanced with increasing tree size and dominance in the landscape.

Planted wetland trees on CSAs may serve the role of directing ecosystem development. Restoration ecologists have traditionally looked at a spectrum of similar sites of different ages to study the dynamics of ecosystem development. A number of studies of the progress of restoration efforts in the phosphate mining districts have adopted this approach (Rushton 1983, Carstenn 2000), and identified trends in ecosystem development across sites. A potential drawback of this approach is that it overlooks the site-specific influences. The topography and its influence over the hydrology and the proximity to seed source are unique to a CSA and important external drivers of ecosystem development. These external factors may create challenges for cross-site comparison of CSAs.

Underplanting as a Restoration Technique

Underplanting, or use of a "nurse crop," (Matthews 1989) is a technique common to silviculture and forest ecosystem restoration used to encourage desired canopy species while avoiding competition from undesirable mid-canopy and understory species. This technique has been studied and recommended for the restoration of forest ecosystems on abandoned pasture and agricultural lands (McKevlin 1992) and has been suggested as a prescription for rehabilitating degraded bottomland forests (Clewell and Lea 1990; Stanturf and Meadows 1994). In addition to its functions as a pioneer species in reestablishing habitat complexity, improving soil structure and nutrient status, and benefiting wildlife in early successional environments, work has shown *Salix spp.* as an effective genus for restoration of structure and function within ecosystems, and it has been used as a "nurse crop" in wetland floodplain restoration (Kuzovkina and Quigley 2005). Clewell (1999) found several wetland tree species survived well eleven years after planting occurred under a *Salix caroliniana* canopy as part of a riverine headwater forest restoration. Dulroney and others (2000) showed a *Salix caroliniana* canopy to help facilitate wetland tree seedling establishment for four common wetland tree species, and ameliorate the effects of herbaceous species. Also, McLeod and others (2001) showed through controlled experiments that *Salix nigra* did not negatively affect the survival on the survival of four bottomland hardwood species. Thus, a goal of restoration on CSAs may be to transform the wetland site beyond willow dominance into an intact forested wetland by establishing later successional species that can function as a future seed source by correct placement of seedlings within the CSA wetland with regard to hydrology, inundation, topography, and light and nutrient availability. Since past research has shown several tree species to be successful when underplanted beneath a *Salix caroliniana* canopy on CSAs (Paulic and Rushton 1991b) this approach should continue to be tested, refined, and monitored.

METHODS

CHARACTERIZATION OF NATURALLY OCCURRING WETLANDS ON CSAs*

Site Selection and Data Collection Overview

Twenty-two CSAs of different ages/constructions and from both the north and central Florida phosphate districts with wetland features were selected. CSAs with both sand-clay mix fill and clay only fill were studied (Table 1). Only CSAs that had been abandoned (clays are no longer being pumped in), on which the surface had hardened into a crust, and on which wetland vegetation† was present were selected. All sites selected had been abandoned for at least 10 years. The right columns in Table 1 provide information on the type of data collection and analysis preformed on the CSA.

Plant Community Evaluation

Field data were collected along multiple transects established along the hydrological gradient of each CSA. Transects were designed to capture every unique wetland community on a CSA,‡ and ran from 30 to 100 meters, with a minimum of one transect per site. Length depended on the redundancy of either the gradient or the vegetation (shorter transects were run for areas with little (<1%) gradient or vegetation change) or to the end of the wetland feature. Transects always began in an ecotone area on the edge of the wetland, so that either standing water or obligate wetland vegetation was reached within 20 meters of the start. If standing water depths of greater than 1.5 m were reached, then the transect was ended at the nearest 10 m mark. Vegetation, elevation, water levels, and soils data were collected along each transect. Belted transects were used to facilitate sampling across the environmental gradient from ecotone to wetland.

Canopy and subcanopy trees, including *Salix caroliniana* (Carolina willow), that had at least one stem with a diameter at breast height (dbh) of 5 cm or greater, were

---

*The term 'wetland' used to describe areas under investigation on CSAs does not refer to 'wetlands' as defined by the National Science Foundation (NSF), United States Environmental Protection Agency (USEPA), United States Army Corps of Engineers (Corps), Florida Department of Environmental Protection (FDEP), or any other agency, but rather is used to describe the periodically inundated areas which are the subject of investigation in this report. Jurisdictional delineation was not completed as part of this research.

†Wetland vegetation determinations used in site selection were based on species with a wetlands indicator status of *obligate* (always occurring in wetlands) or *facultative wetland* (usually occurs in wetlands but occasionally found in non-wetlands) (Tobe 1998).

‡Unique wetland communities were identified with site aerial photos when available, or alternatively with visual inspection of the site.

Table 1. Sites with Selected Research Objectives.

No.	Proprietor	Site designation	Description	Research Type: Sand-Clay Mix	Character-ization	Hydrologic Evaluation	Spatial Modelling	Eco-hydrology	Eval. of Old Field Trials	New Field Trials
1	CF Industries	CFI R-6	Older sand/clay mix site (1994-95) appears that wetlands are enlarging.	Y	√					
2	CF Industries	CFI R-8	12 acre herbaceous mitigation wetland. Planted multiple times, first in 1995, most recently in 2001.	Y	√					
3	CF Industries	CFI SP-5	22 acre wetland mosaic of forested and herbaceous. Planted in 1989; herbicided in 1997.	Y	√					
4	CF Industries	CFI SP-1	24.5 acre wetland on W lobe; 20 acre wetland on East lobe. Sites were filled in 1983. Rushton planted on E lobe	Y	√	√	√	√	√	
5	DEP Homeland	HOM	Sand capped in 1979. 8 cypress-gum transects planted.	N[1]	√				√	
6	Mosaic	H1	Built in 1976. Ditched in the early 1980's, reclaimed in 1984-86. Willing to plant wetland species on wet areas.	N	√	√	√			√
7	Mosaic	HP-10	100 acre CSA capped with sand that was later removed which exposed small clay depressional features	N		√				
8	Mosaic	D	Currently being used for agricultural demonstration. Has wetlands in low areas intermixed with crops.	N	√					
9	Mosaic	OH Wright (OHW)	Adjacent to Whidden Creek flooplain. Clay backfilled into 10m mine cuts. 3 cypress-gum and 4 hydric plots planted.	N	√				√	
10	Mosaic	FG3	Aapproximately 17 years old. 1 square mile. Diverse topography with forested and herbaceous wetlands.	N	√					
11	Mosaic	FGH1A	In final dewatering stage;recently planted with pine and cypress in NW corner.	N	√					

Table 1 (Cont'd). Sites with Selected Research Objectives.

No.	Proprietor	Site designation	Description	Research Type						
				Sand-Clay Mix	Character-ization	Hydrologic Evaluation	Spatial Modelling	Eco-hydrology	Eval. of Old Field Trials	New Field Trials
12	Mosaic	K5	Older site that is a thriving willow forest. Reclaimed 10 yrs ago.	N	√	√	√	√		
13	Mosaic	F2B	In dewatering stage. High wall construction. Significant herbaceous wetland area.	N	√					
14	PCS	SA 10	Mandatory reclamation. 50 acre wetland. Started in 1991; 92-93 trees planted.	N	√	√	√			√
15	PCS	SA 3A	Non-mandatory. Hand planted in '96 with some aerial seeding, left canopy. 66 acre wetland area.	N	√					
16	PCS	SA 04	Non-mandatory. 850 acre total, about 200 acres of wetland.	N	√					
17	PCS	SA 01	12 acre reclaimed wetland on W side. Trees planted in 1987.	N	√	√	√			
18	Polk County Peace River Park	PRP/PWP	County park previously used for pasture, cogon dominated, small depressional and lake fringe wetlands	N	√				√	√
19	Teneroc Fish Management Area	Ten-1	Two main wetland areas divided by spoil pile. Tree planting in the eastern area.	N						√
20	Teneroc Fish Management Area	Ten-3	85 acre wetland. Nonmandatory reclaimation in 1986. Interior spoils contoured and planted with trees.	N	√					
21	Teneroc Fish Management Area	Ten-4	Unmanaged area with wetlands in sinks between spoil rows. E side was not mined; cypress domes still intact.	N	√	√			√	
22	Williams Company	AC-OP-06	Mined in the late 1950's. Active 1961 to 1973; nonmandatory reclamation completed in 1994.	N	√	√	√			

1 Typically clay fill capped with sand > 1 m in depth

Total Sites By Research Type

Character-ization	Hydrologic Evaluation	Spatial Modelling	Eco-hydrology	Eval. of Old Field Trials	New Field Trials
20	**8**	**6**	**2**	**5**	**4**

enumerated in a belt transect extending 3 m to each side of the line transect along the entire length of the transect. Shrubs and multi-stemmed subcanopy trees (e.g., *Myrica cerifera,* wax myrtle), including *Salix Caroliniana* with dbh less than 5 cm, were measured in 9 m^2 (3m x 3m) quadrats established randomly along each 10 m segment of transect. A random number between 0 and 9 was chosen uniquely for each transect which designated the meter within each 10 m segment that the quadrat was placed (e.g., a random number '8' would mean the quadrat was placed at 8,18,28, and 38 meters on a 40 m transect). For trees and shrubs that reached 1 m in height, the dbh of all stems was recorded and all individuals were identified to species.

Basal area was calculated based on the following formula:

$$\text{Basal Area} = \pi*(\text{dbh}/2)^2 \qquad [1]$$

In these calculations, basal areas of stems of trees within each 10 m interval were summed together. The same sum was calculated for all stems of shrubs.

All plants less than 1 m in height were identified to species in a 1 m^2 (1m x 1m) quadrat nested within the 9 m^2 quadrat along each 10 m segment of transect. Percent vegetative cover of the quadrat (cover abundance) of all vegetation less than 1 m was visually estimated from above and assigned a number based upon a rating scale (Table 2). All unknown species were harvested, labeled, and transported in a cooler until they could be pressed in the lab and taken to a professional botanist for identification.

Table 2. Cover Abundance Scale.

Rating	% Cover
5	76% to 100% cover
4	51% to 75% cover
3	6% to 50% cover
2	10% to 25% cover
1	< 10% cover

Zones of similar vegetation along the transects were delineated. In each zone, dominate ground level (also called herbaceous level), shrub level, and canopy species were listed for those levels where vegetation occurred. Ground level was defined as <1 m above the ground or at the waters surface in the case of standing water, shrub level as communities with plants with woody stems taller than 1 m but less than 5 cm in diameter, and canopy level as communities with plants with woody stems taller than 1 m and equal or greater than 5 cm in diameter. Plant communities imply those that shared the same dominant species for at least 2 m along the transect at one vertical strata (ground, shrub, or canopy). Mixed communities occur where multiple species occurred. Mixed wetlands communities were composed primarily of obligate and facultative wetland species, whereas facultative or transition communities were composed primarily of facultative, facultative upland, and facultative wetland species. The reference for wetland status for species was the Florida Department of Environmental Protection (Tobe 1998).

Transect Photos

Four digital photographs were taken of each transect. One was taken at the origin looking down the transect; a second was taken at the origin looking 180 degrees away from the transect; a third was taken at the end looking up the transect; a fourth was taken at the end looking 180 degrees away from the transect. These photos were used to verify that data collected were matched to the correct transect.

Soils

At the center of each 1 m^2 quadrat a 15 cm soil core was collected using a 7.62 cm diameter coring tube. Cores were placed in a sealed plastic bag, stored on ice, homogenized, and the wet weight of a 25 g sub-sample of the core was recorded in the lab. Each sub-sample was placed in a drying oven at 70° C until constant weight was achieved. The following formula was used to calculate percent soil moisture by weight:

$$\text{(wet weight – dry weight) / wet weight} = \text{\% moisture} \qquad [2]$$

Dried samples were ground with a mortar and pestle and 1 g of the ground soil was ashed in a muffle furnace for 6 hrs at 500° C. This relatively low temperature drives off the organic matter while leaving inorganic carbon ($CaCO_3$), which volatizes at approximately 540° C. The loss from ignition was reported as a rough estimate of organic matter. This method can over estimate organic carbon because inorganics can lose mass after heating. Specifically, clays can lose water bound within their chemical structure. The following formula was used to calculate percent organic matter:

$$\text{(dry weight – ashed weight) / dry weight} = \text{\% organic matter} \qquad [3]$$

Soil samples from zones of similar vegetation along individual transects were combined. These composite samples were analyzed for available phosphorus (P), ammonium (NH_4-N), nitrate (NO_3-N), nitrite (NO_2-N), total Kjeldahl nitrogen (TKN), and pH at the IFAS Analytical Research Laboratory at the University of Florida. Available P was estimated using the Mehlich III extractant (Mehlich 1978) and was analyzed by inductively coupled argon plasma (ICAP) spectroscopy. A KCl extractant (Bremner 1965; Kenney and Nelson 1982) was used to estimate available NO_x-N and NH_4-N. Results from nitrogen analyses were reported in mg NH_4-N, NO_3-N, and TKN as nitrogen per kg dry soil. Total nitrogen was defined with the following equation:

$$TN = TKN + NO_x\text{-}N \qquad [4]$$

Elevation Profiles and Water Levels

A relative elevation profile was created for each transect using a Topcon RL 20 laser level (Topcon Positioning Systems, Livermore, CA USA). The laser level was set at one or multiple positions along the transect to enable a clear line of sight along the transect. An elevation recording was taken at the transect origin and at each point of recognizable elevation change, x, along the transect or every 2 m, whichever occurred first. Data were transformed so that the transect origin had an arbitrary elevation of 0 m. Percent elevation change on the transect was calculated as:

$$\frac{\sum_{0}^{t}\left|y_{\text{n-1}} - y_{\text{n}}\right|}{\sum_{0}^{t}\left|x_{\text{n-1}} - x_{\text{n}}\right|} \times 100\,, \qquad [5]$$

where y_n is equal to the relative elevation in meters at sampling point n, x_n is equal to the distance along the transect from the origin, and t is the farthest point along the transect.

Water levels were measured to the nearest cm with point measurements taken at a known point of elevation wherever standing water occurred during initial visits. The elevation of historic seasonal high water levels (SHWL) were determined using the laser level during data collection for the elevation profile. SHWL indicators included lichen lines, water stains, and/or adventitious rooting (Tiner 1999). During each successive transect visit, water levels were recorded with a measuring tape at a point(s) where a laser level reading was previously taken.

The ecotone was defined as the area above the seasonal high water level. The wetland area was defined as the area along the transect at or below the seasonal high water mark. The wetland depth, when generally described, was calculated as the difference between the seasonal high water mark and the minimum elevation recorded along the transect.

Microclimate

To measure difference in light, temperature, and humidity in open areas and under canopies on CSAs, two HOBO® weather stations (Onset Computer Corporation, Bourne, MA) with PAR, temperature and humidity sensors were installed one site, SA 10, in north Florida. One weather station was installed in an open area of the site and one under a healthy *Salix caroliniana* dominated canopy 50 m from the forest edge. Both weather station sensors were programmed to record at one-hour intervals.

Vegetation Mapping of Aerial Photographs

On a representative number of sites, historic aerial photos were collected from industry partners to evaluate temporal changes in vegetation and hydrology. Current photos were interpreted and ground-truthed and vegetation signatures were developed that were used to interpret the historical aerials. Maps generated in this way were digitized and analyzed using GIS to determine changes in vegetation. Typical land cover types used in the vegetation mapping included upland, herbaceous wet, *Typha spp.* , *Salix* *caroliniana*, *Baccharis halimifolia, Lemna spp.* (duckweed), and sand tailings. Each vegetation zone was assigned relative water level ranges based on vegetation transects and observed minimum and maximum water levels. Vegetation maps were used in conjunction with observed water levels for each vegetation zone to create water depth maps.

HYDROLOGIC ANALYSIS AND MODELING

Hydrologic Monitoring

Hydrologic monitoring from the fall of 2004 through the summer of 2008 of eight CSAs, which ranged in size and age, was conducted to evaluate all inputs and losses to the surface water systems. The nature of fill also differs among the eight sites, with five sites having pure clay, two sites having sand tailings deposited over pure clay, and one site with sand-clay mix co-deposited (Table 1). The collected hydrologic data included precipitation from HOBO® data-logging tipping bucket gauges (Onset Computer Corporation, Bourne, MA) and continuous water levels from surface water and groundwater wells. Solinst® mini LT 15' continuous logging pressure transducers (Solinst Canada Ltd., Ontario, Canada; accuracy = 0.5 cm, resolution = 0.1 cm) were deployed in the wells to collect continuous water level data. Locations of all wells on the eight sites are shown in Figures 1-8. The pressure transducers measured total pressure, in equivalent height of water, and therefore needed to be corrected for barometric pressure. Solinst® Barologgers were installed at each site to provide barometric pressure data and were programmed to record simultaneously with the pressure transducers. The pressure transducers were originally programmed to record every hour, but were then set to record on 15-minute intervals to provide more detailed data.

Surface water wells were installed in the deepest part of the main surface water feature of each site and typically along previously established vegetation transects. Three sites had multiple surface water features instrumented. The wells were screened below ground to allow for belowground water table recordings in the event the water feature became dry. Direct measurements at the wells during site visits were conducted to calibrate the continuous data. One groundwater well was installed in an area just upland from and adjacent to the surface feature and typically along the vegetation transect. Another groundwater well was installed at a later date further from the feature and at a higher elevation than the first groundwater well to observe how groundwater behavior may change with distance from surface water. Three sites had an additional groundwater well later installed in an intermediate zone between the other two groundwater wells in terms of elevation. The ground elevations of all wells were used to represent groundwater and surface water levels relative to one benchmark, the ground surface of the surface water well. At three sites that had surface water in contact with surrounding dikes, wells were installed on the tops and downward slopes of the dikes where only discrete groundwater levels were recorded on site visits to provide data for dike seepage estimation.

HOBO® weather stations (Onset Computer Corporation, Bourne, MA) were installed on three of the eight hydrology sites, one in north Florida and two in central Florida. The weather stations were installed in an open area of each site with relative humidity, temperature, and PAR sensors along with an anemometer to measure average wind speed, wind gusts, and wind direction. All weather station sensors were programmed to record at one-hour intervals. The climatic data were used in estimating ET with various empirical models.

Light detection and ranging (LiDAR) technology was employed, along with subsequent analysis to correct for vegetation structure, to gather the most recent and accurate topographic information for the eight CSAs. The technology is not capable of penetrating water and only gives water surface elevation and, therefore, no topographic information was obtained for areas flooded during the collection period. LiDAR topography maps were generated for the CSAs by the National Center for Airborne Laser Mapping (NCALM) (Figures 1-8). These maps were estimated to be accurate to 15 cm vertically and 12 cm horizontally, with a cell resolution of 1 m. Due to potential error, the elevation data was rounded to the nearest tenth of a meter, and was then used as the input DEM for watershed delineation. Since some of the potential wetlands of interest in the CSAs have areas less than a hectare, high resolution DEMs were required in order to be able to model the hydrology of these water features.

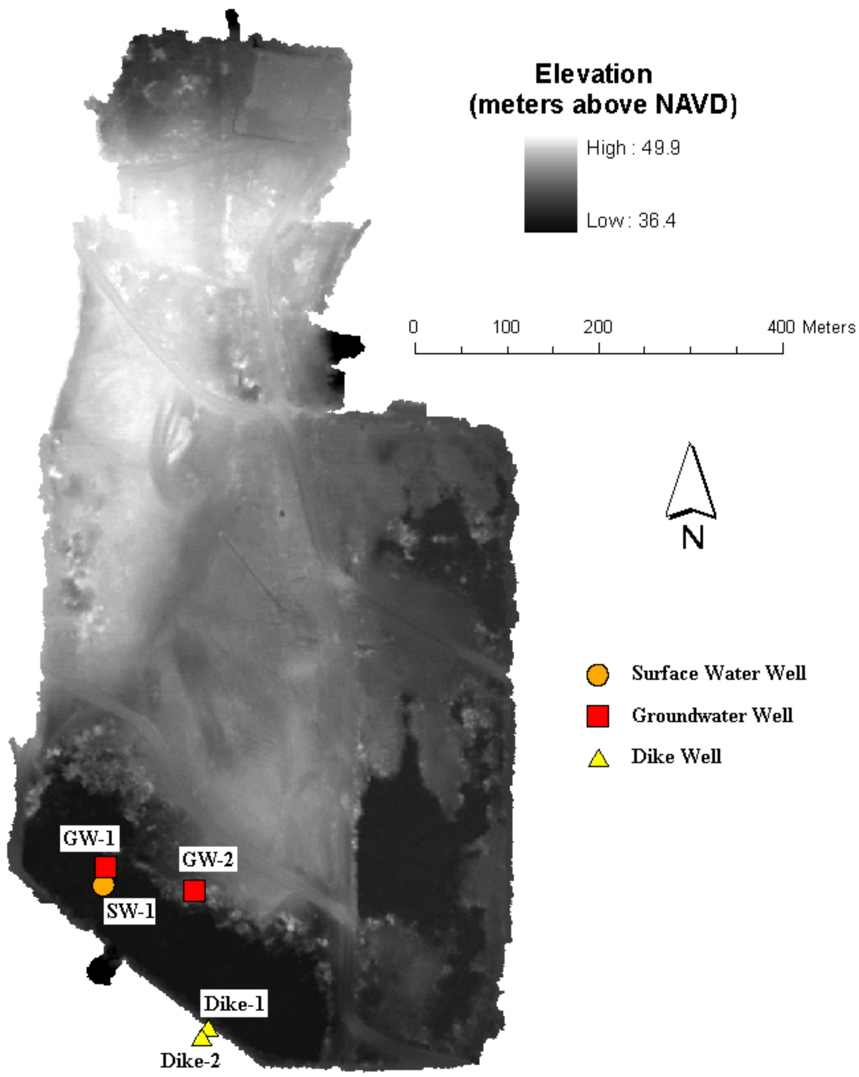


Figure 1. Well Locations and DEM of PCS SA 01.

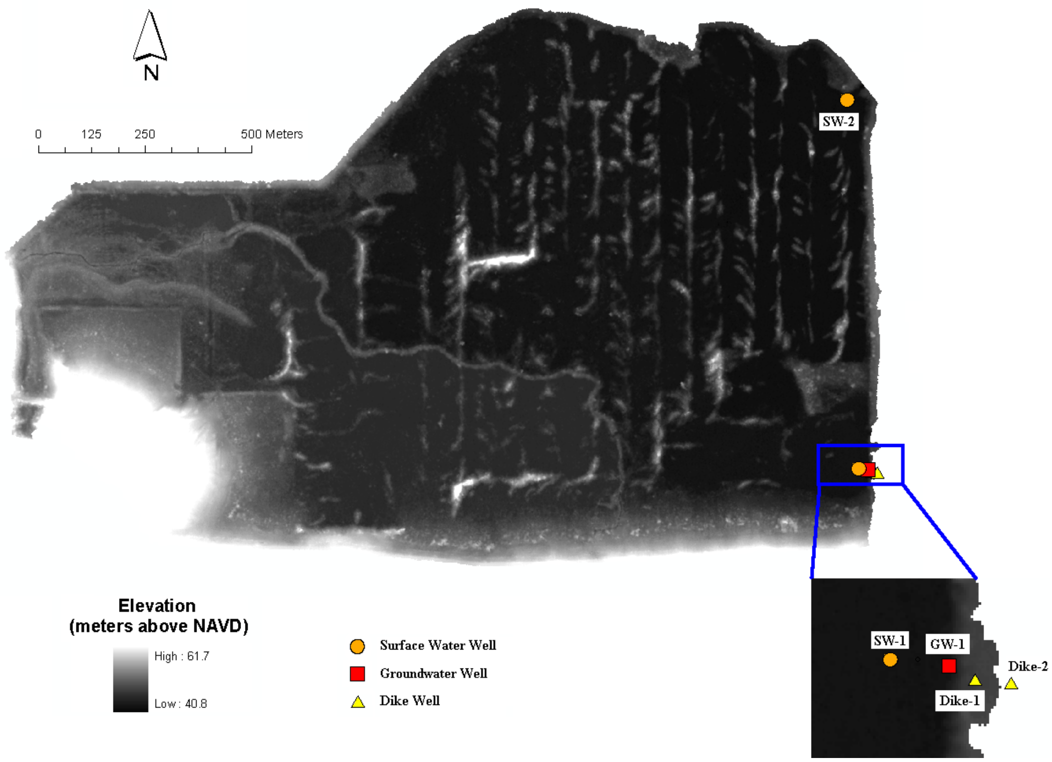


Figure 2. Well Locations and DEM of PCS SA 10.

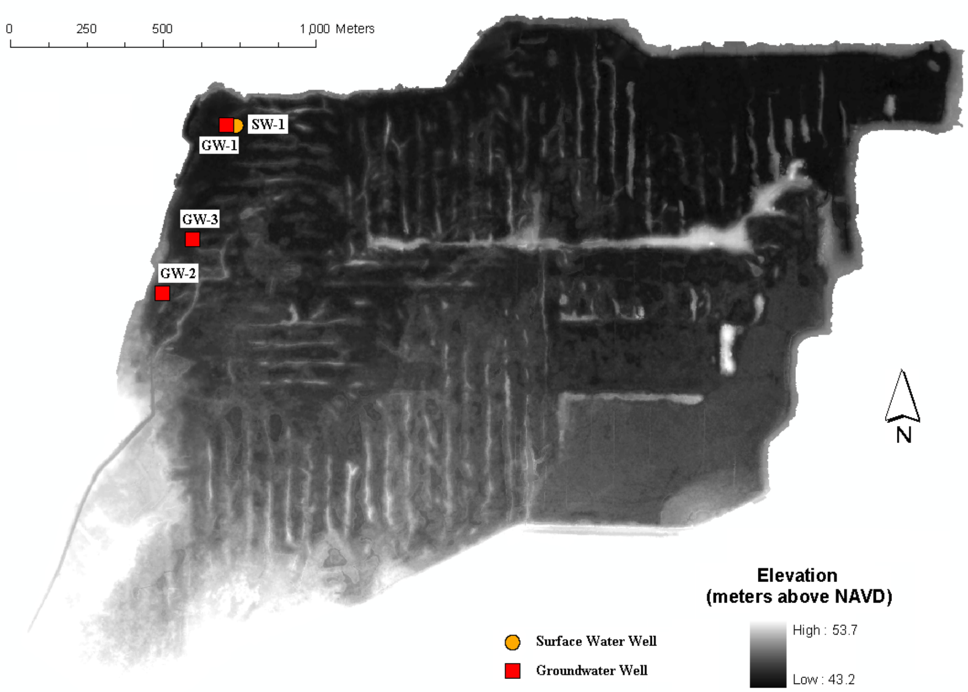


Figure 3. Well Locations and DEM of Williams Co.

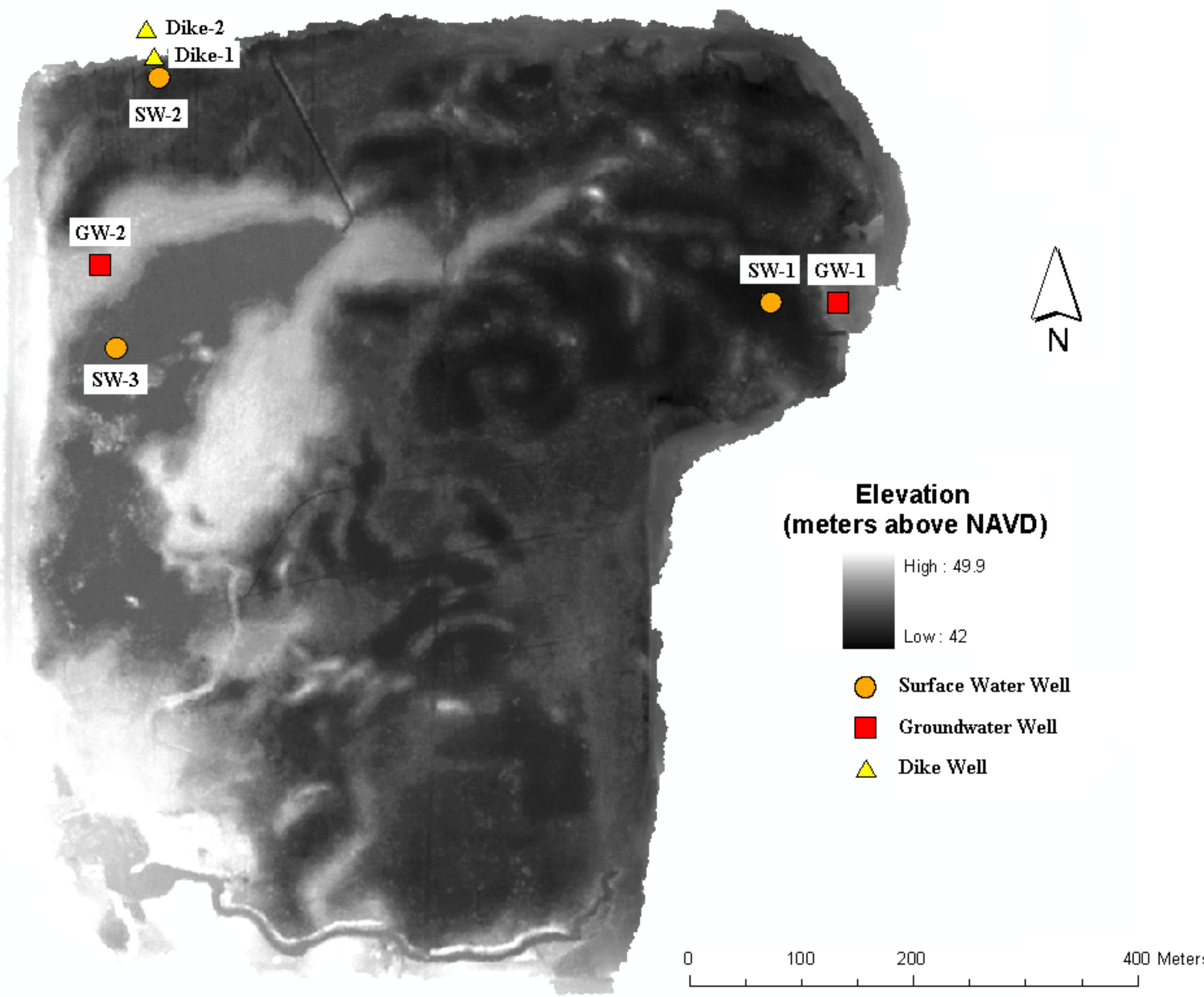


Figure 4. Well Locations and DEM of Mosaic H1.

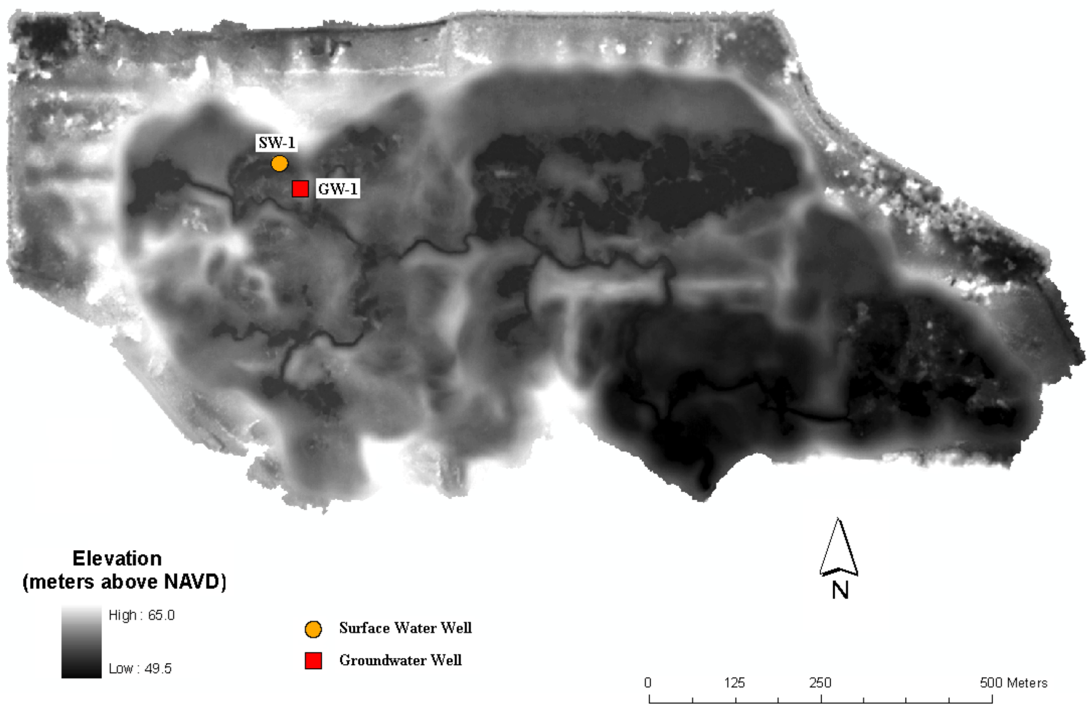


Figure 5. Well Locations and DEM of Mosaic HP-10.

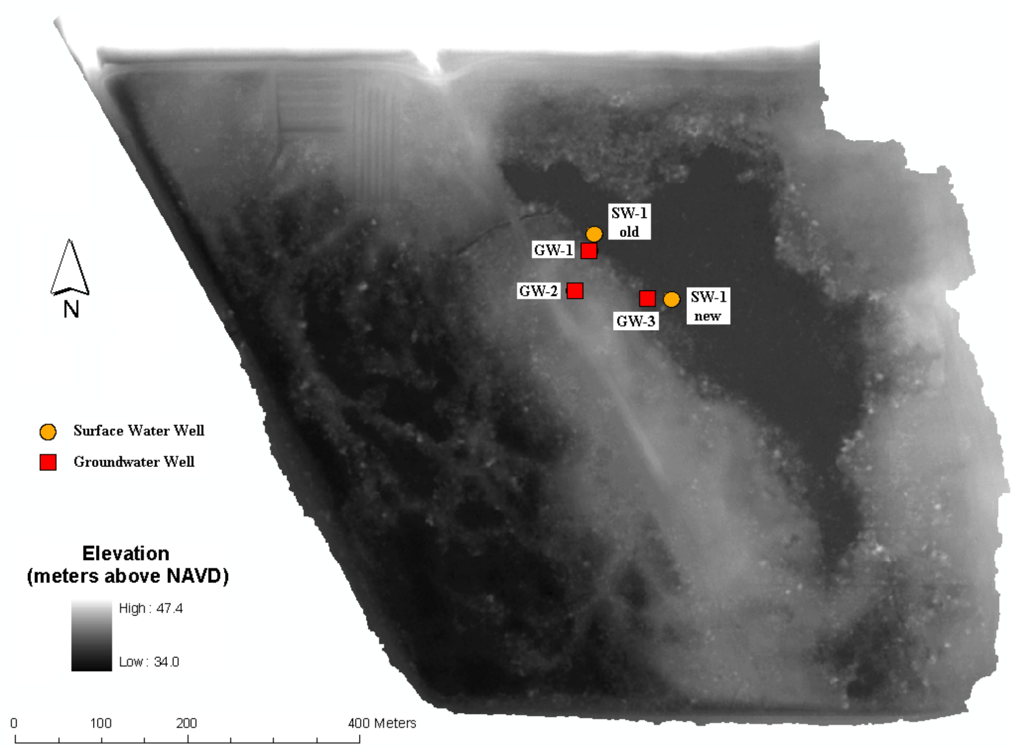


Figure 6. Well Locations and DEM of CFI SP-1.

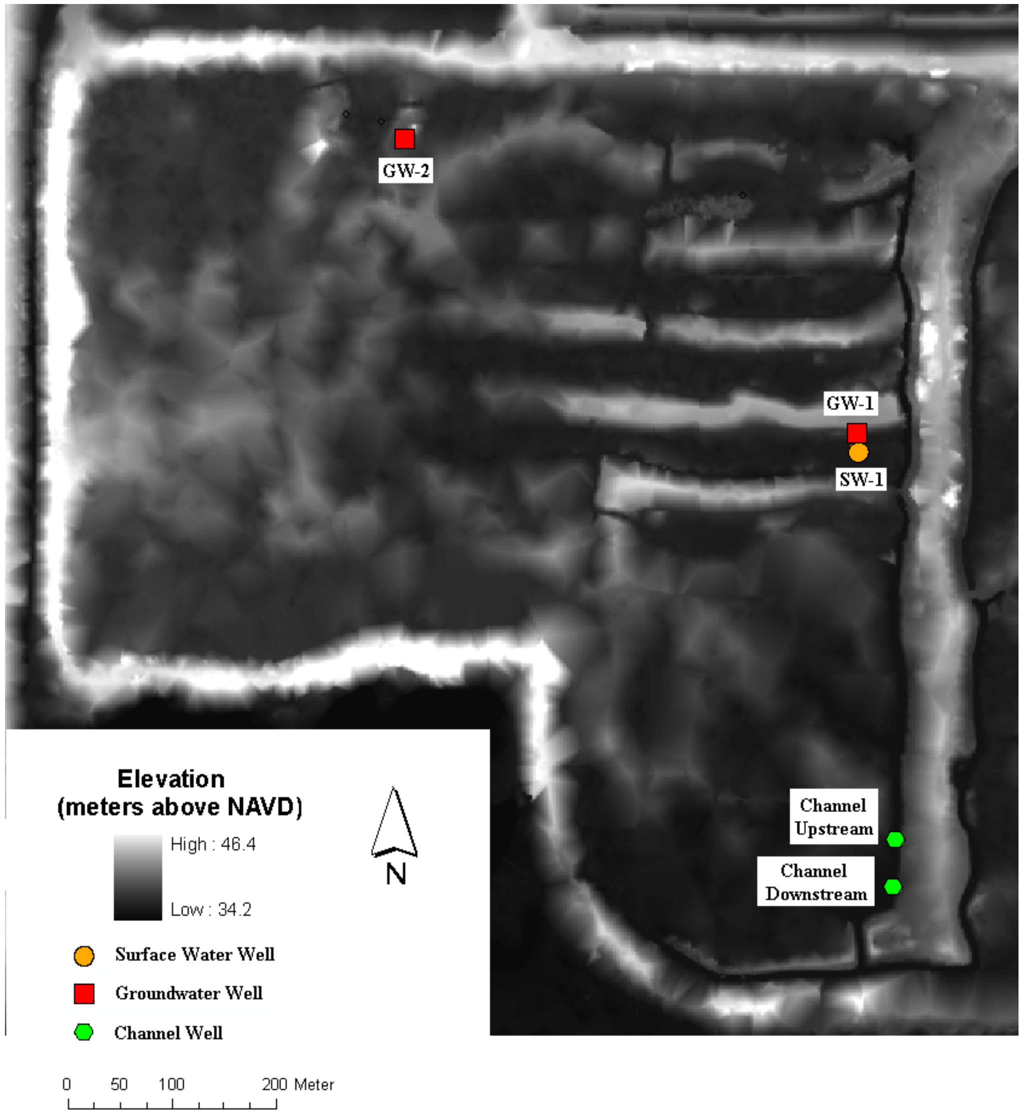


Figure 7. Well Locations and DEM of Tenoroc-4.

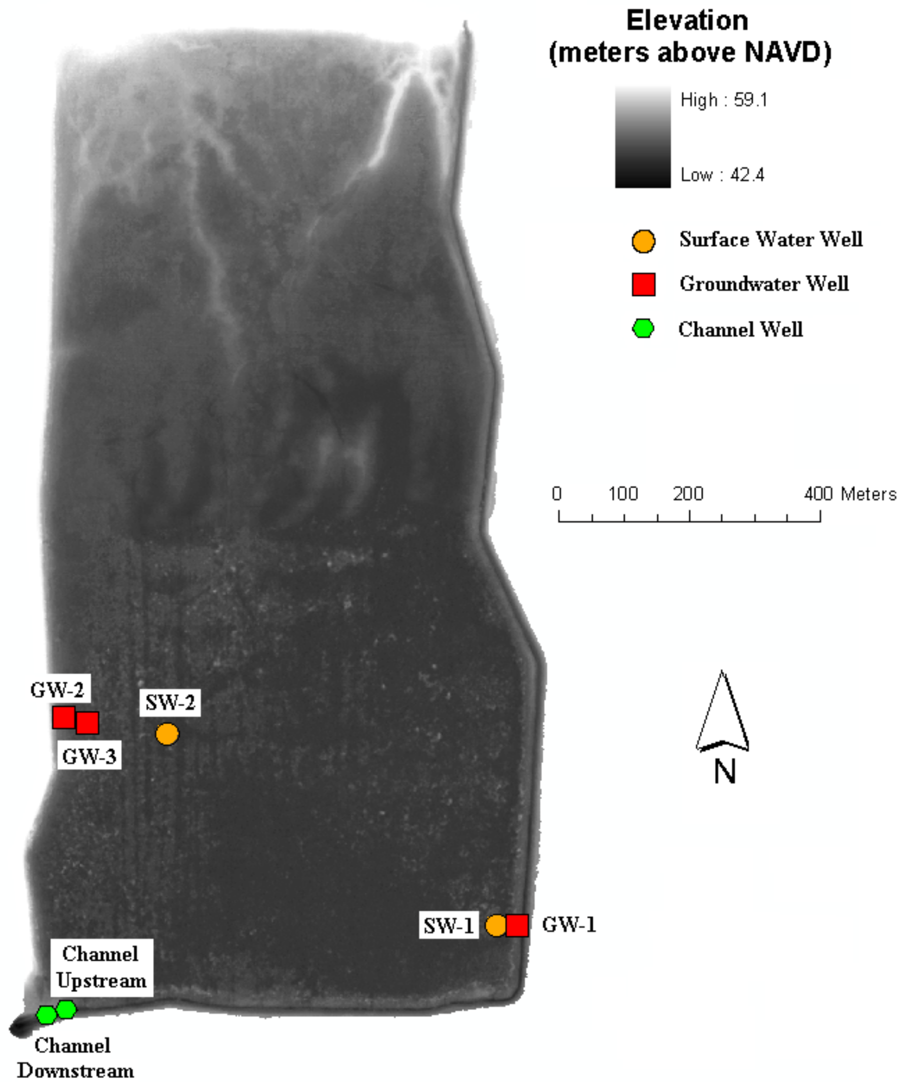


Figure 8. Well Locations and DEM of Mosaic K5.

Hydroperiod Analysis

Hydroperiods in terms of surface water fluctuations and depths, and frequency and duration of dry periods were compared among CSAs. Multiple surface water features, referred to as Surface Water-1 (SW-1) through Surface Water-3 (SW-3), instrumented on an individual site allowed these comparisons within a CSA. The water levels recorded in the features were calculated as water elevations relative to the elevation at the ground surface of the SW-1 feature. Comparisons of hydroperiod signatures and the relative water elevations of the multiple surface water features were performed to observe if and when features on one CSA were isolated from one another. The surface water elevations of features which were connected to an outfall system were evaluated in reference to the invert elevations of the outfall to determine depths at which surface water outflow occurred.

Surface water data collected at Ted's Marsh from 1994 to 2003 and at a *Taxodium spp.* forested wetland, Green Swamp #5, from 1981 to 2003 during a study by Bardi and others (2005) were averaged to develop average hydroperiods typical of central Florida wetland systems. The average hydroperiods were compared to surface water data collected at the CSAs. Surface water data collected at a marsh system within the Green Swamp were provided by the Southwest Florida Water Management District and compared to surface water data collected during the same period of time at the CSAs.

Spatial Modeling of CSA Hydrology

A spatial hydrology model was developed to better understand the spatial dimensions of hydrology on CSAs by simulating the extent, depth, and frequency of surface water on six sites. The spatial model was created using the GIS software ArcGIS® 9.2 (ESRI 2008) and was programmed in the VBA ArcObjects language included with the software package. The spatial model was based on a simple water budget including inflows of rain and surface runoff, and outflows of ET, groundwater infiltration, and surface outflow if applicable. Inputs to the model included daily rainfall data and high resolution (1-2.3 m^2/cell) topographic maps in the raster format generated from LiDAR (Light Detection And Ranging) data. Daily surface water data recorded in the main surface water feature of each CSA were used for model calibration. Once calibrated, the model generated a map for each day of the simulated period that showed the surface water depth throughout the modeled watershed. These daily water depth maps were then combined to generate a hydropattern map showing the frequency that each area of the map was inundated over the simulation period.

Spatial Model Methodology

The spatial model was based on the level-pool assumption which states that the water surface within a watershed is at a constant elevation. This assumption was used even when there were separate depressions within a watershed, and therefore the depth of water could vary between these depressions but the elevation of the water surface was equal (Figure 9). The model was run on a daily time step, and each day the water budget was used to calculate a volume of water that was then added to the watershed being modeled. Using GIS, the process of adding a volume of water to the watershed was conducted by starting at the absolute lowest elevation within the watershed and then raising the surface water level until the entire volume of water was accounted for. The model then output a map of each day showing the extent and depth of water throughout the watershed.

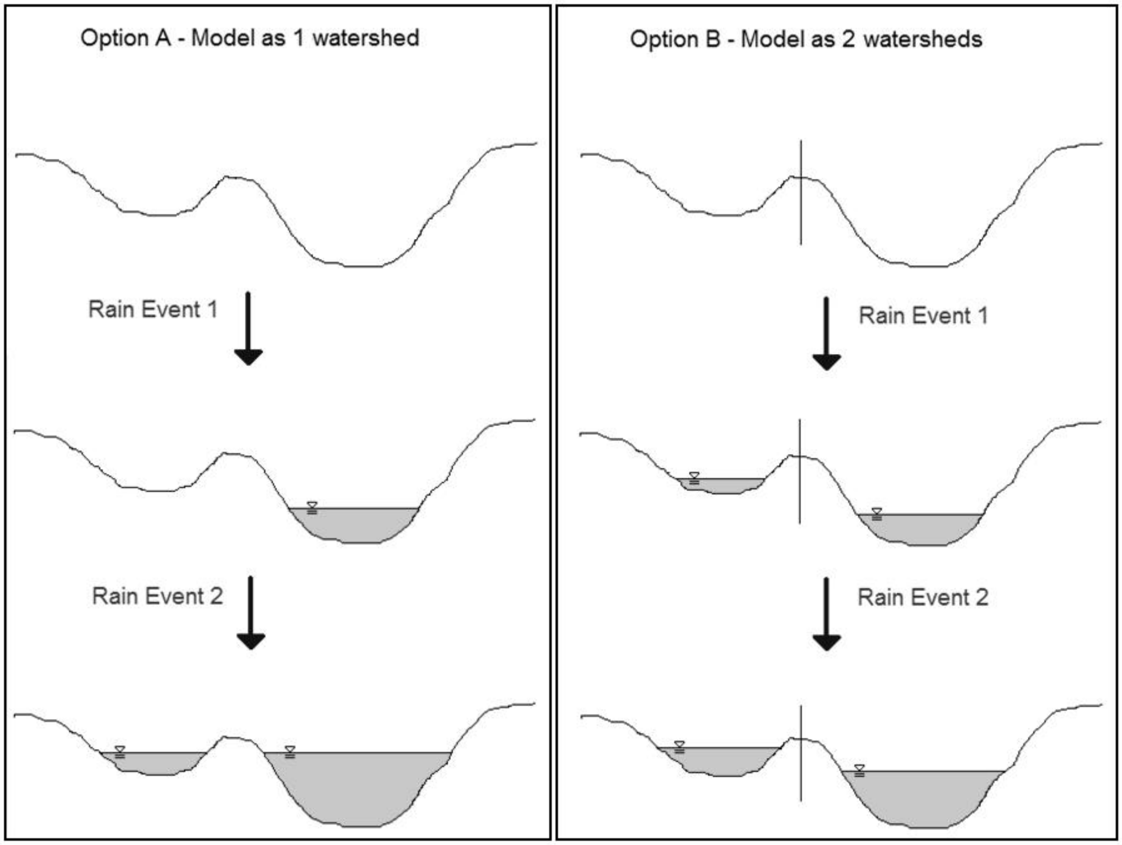


Figure 9. Spatial Model Results Using Different Watershed Boundaries.

Spatial Model Limitations

There were three issues with the spatial model that limited its application in this study: lack of multiple monitored water features for each site, standing water when the LiDAR was flown, and lack of below-ground hydrologic simulation. The first two issues were the result of insufficient data required for optimal model results, whereas the third issue was related to limitations of the model itself.

The first issue was that for most of the six CSAs modeled, only one water feature was monitored for surface water levels, which prevented calibration of the other watersheds within each CSA. Ideally, each water feature that is of significant size to be considered for active wetland enhancement (i.e., planting) would be monitored for surface water levels at least a few years prior to model simulation. This should provide sufficient data for model calibration, and therefore, accurate water depth and hydropattern maps for the main water features within each CSA. Based on evidence from three water features monitored at the site Mosaic H1, each feature in a CSA should be monitored separately because of significant differences in hydrologic parameters such as groundwater infiltration and surface runoff.

The second issue was that when the LiDAR maps were created, there was standing water on some portions of each CSA modeled. The LiDAR technology used in this study did not measure ground surface topography where standing water was present. Therefore, the model was only able to simulate periods when the water level was above the water level height on the date when the LiDAR was flown. In the model when the water level went below this height, simulation could not continue until the water level returned to a height greater than it was when the LiDAR was flown. Fortunately the water level when the LiDAR was flown was known for the main surface water feature at most sites, since the surface water wells had already been installed. For the Williams Co. CSA, the LiDAR maps had already been created prior to installation of the surface water well. The water level when the LiDAR was flown, however, was able to be estimated since ground elevations had been measured on a transect which contained the surface water well. Unfortunately for sites that had unmonitored water features, the water level when LiDAR was flown was unknown. The preferred solution to this problem is to ensure that sites have no standing water when the LiDAR is flown, since any methods used to edit the LiDAR maps to fill in this bathymetry would be time-consuming and may not represent the actual topography.

Finally, the model was limited to simulation of above-ground water levels, and terminated once water levels went below ground. When water levels rose above ground, then the model was restarted and simulation continued until either the model period was over or water levels receded below ground again. This limitation was similar to the case mentioned above in which water levels went below the water level at which the LiDAR was flown. Both of these instances can be handled in the model by restarting it once water levels rise again, but they may make simulation of future conditions difficult because it would be unknown at what time the model should be restarted since this would be dependent on the water levels below ground.

Watershed Delineation

Prior to running the spatial model, multiple watersheds were delineated within each CSA studied. In most spatial hydrology models a volume of water from a rain event is routed through a stream network to an outlet in order to predict flows at that point (Garbrecht and Martz 2000). For this study the goal was not to route water to watershed outlets but instead to model the water level fluctuations within watersheds located throughout each CSA. This approach was taken because the hydrology of wetlands is best characterized by looking at water depths and hydroperiods rather than flows at outlets. While this approach is not common in hydrological modeling, there are examples of similar models in the literature (Poiani and Johnson 1993, Kirk and others 2004).

Watershed delineation was an important part of the spatial model because it set the boundary for the water budget calculations. In this study, each CSA was considered to be the main watershed and subwatersheds were delineated within this main watershed. ArcGIS® 9.2 contains pre-made programs called "tools," and within the Hydrology subset of Spatial Analyst tools there are four tools that are widely used to delineate watersheds. These tools (Fill, Flow Direction, Sink, and Watershed) were used sequentially, and the only input required was a topographic map in the raster format, also known as a digital elevation model (DEM). For further information about this process, see the ArcGIS® Desktop Help documentation provided with ArcGIS® 9.2.

The amount of watersheds delineated using the ArcGIS® procedure was based on the z-limit feature within the Fill tool. The z-limit is a measure of the difference between the lowest point within a watershed and the lowest point on the watershed boundary; in other words, it is a measure of the depth of a watershed. If the Fill tool was used with a z-limit equal to zero, then every cell that was lower than its surrounding cells, also known as a sink, in the topographic map had a watershed delineated for it. However if the z-limit was set to 50 cm, for example, then every watershed with a depth less than 50 cm was filled in, which essentially combined shallower watersheds, so that ultimately fewer watersheds were generated. The z-limit is a useful feature because it allows for the creation of multiple scales of watershed delineation. Therefore, when using a high resolution topography map for a typical CSA, a z-limit of zero generated over a thousand subwatersheds, a z-limit of 20 cm generated hundreds of subwatersheds, and when a large enough z-limit was used then the entire CSA was delineated as one watershed (Figure 10).

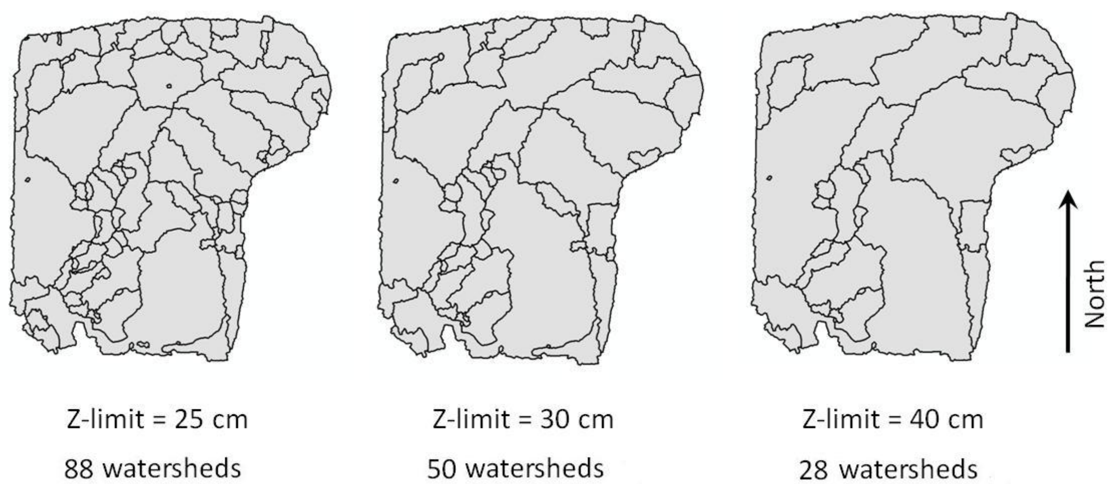


Figure 10. Watershed Delineation Results Using Three Different Z-limits.

Since the spatial model was based on the level-pool assumption for each watershed, the scale of watershed delineation was very important (Figure 9). The predominantly clay substrate within CSAs allows for water to accumulate in local depressions throughout a CSA. If the entire CSA was modeled as one watershed then water only accumulated in the overall lowest parts of the CSA, and depressions in the higher portions of the CSA were not included. However, by delineating watershed boundaries for each depression within a CSA, then depressions were subsequently modeled separately, and the hydrology of the entire CSA was more accurately simulated.

In this study, each CSA was modeled based on the scale of watershed delineation that seemed to best fit the observed presence of surface water features, some of which were perched at higher ground elevations within a CSA. The drawback of this method was that hydrologic data was necessary for each watershed modeled in order to achieve the most accurate results, however only one watershed was monitored for most sites. Also for some CSAs an optimal scale, or z-limit, was not readily identifiable. One reason for this was that some surface water features were separate at low water levels, but became connected at higher water levels. Therefore these water features essentially became one larger water feature at higher water levels and their boundaries combined. Despite these problems associated with delineating multiple watersheds within a CSA it was evident that treating each CSA as one watershed would not lead to an accurate portrayal of the hydrology of the entire CSA in most cases.

Spatial Model Calibration

The spatial model was calibrated by comparing the modeled water levels to measured water levels at the surface water well location. For each CSA that was monitored only one surface water feature had appropriate data to use in the spatial model (due to LiDAR limitations), except for H1, which had two such features. The model was calibrated by adjusting model parameters until a best fit was achieved between the modeled and measured water level data as calculated by a least sum of squares regression.

Three components of the model were adjusted in order to calibrate the model for each monitored watershed: the runoff coefficient, the daily loss rate due to a combination of ET and groundwater infiltration, and surface outflow. The runoff coefficient was adjusted so that the water level increase due to rainfall events was captured. A runoff coefficient that allowed a best fit between monitored and measured water levels over the entire simulation period was used and was held constant throughout that period. The only exception occurred for Mosaic H1 when large rain events caused additional water to flow into the modeled watershed from adjacent watersheds, requiring an increase in the runoff coefficient for those events.

Another component that was adjusted during calibration was the daily loss rate of water. ET and groundwater infiltration were lumped into a single daily loss rate. This value was determined for each month of the simulation period, assuming that each year should have relatively similar monthly values due to the seasonal nature of ET.

For watersheds that had surface water outflow through a weir or channel, the amount of outflow at certain water levels was also calibrated. The water level at which water began to flow out was determined for each watershed to be modeled, and then the amount of outflow was calibrated for water levels above the initial outflow level. This approach was simple compared to more detailed hydrologic models that use equations developed for outfall structures that can relate water level height above the structure to outflow. Due to the irregularity of the outfall structures (e.g., cracked weirs, periodically vegetated ditches) observed at some of the watersheds, this approach was deemed appropriate for the level of accuracy required. Also, traditional weir equations would not have worked since the model was based on a daily time step and was not able to capture changes in water level throughout a day.

Water Depth and Hydropattern Maps

Once the spatial model was calibrated for a watershed, the model was run for every day of the monitoring period that was able to be modeled, thus generating a water depth map for each day. It should be noted that the simulation period may be different from the monitoring period because some days were not able to be modeled due to water levels being below ground or below the water level when the LiDAR was flown.

Before the daily water depth maps were created an initial water depth map was generated, from which the other maps were successively created. This was done by calculating the volume of water that was necessary to achieve the initial water level within the monitored watershed, starting with a topography map without any surface water. This approach to generating a water depth map for a specific water level was also used to create maps that simulated the extent and depth of surface water for the maximum and average water depths measured. The maximum depth of surface water was based on the maximum depth measured throughout the monitoring period, whereas the average depth of surface water was an average of the measured depths throughout the monitoring period including below ground (negative) depths. For all CSAs except Mosaic H1, the highest water depths listed for both the maximum and average water depth maps were based on the depth relative to the LiDAR water level when flown, and therefore the water depths may be greater than what was listed in the map legend.

In order to create the hydropattern map, the daily water depth maps for the simulation period were then altered to indicate either presence (value = 1) or absence (value = 0) of surface water on a cell-by-cell basis. These maps were then added together and divided by the amount of days modeled in order to generate a map that showed the frequency that each cell contained surface water, also known as a hydropattern map.

Water Budgets

Overview and Definition of Terms

Detailed water budgets were performed using all inputs, outputs, and storage in terms of depth. These water budgets should not be confused with the simpler water budgets utilized in the spatial modeling. A systems diagram using diagramming language developed by H.T. Odum is shown in Figure 11, illustrating the major water inputs and outputs to a surface water feature of a CSA (Odum 1983). Inputs include direct rainfall and runoff from the surrounding watershed. Collected rainfall and surface water well data and watershed analysis were used in storm event analysis to evaluate the effect of rainfall intensities and antecedent conditions on stage responses.

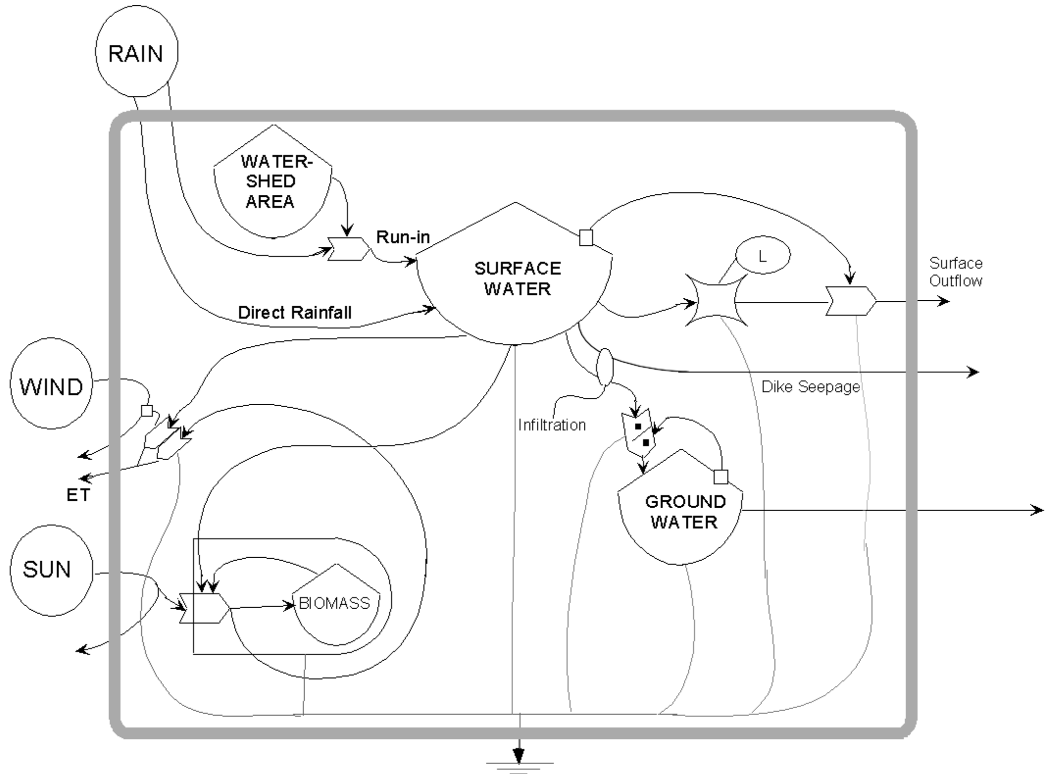


Figure 11. Systems Diagram of the Inflows and Outflows of a Surface Water Feature.

Outputs of the surface water include surface water outflow, ET, and infiltration. While most of these sites do not have active outfalls, several do, and thus, evaluations of outflow were performed for these CSAs. Figure 11 shows a switch controlling outflow, which represents the presence/absence of an active outfall structure, and an interaction symbol representing the relationship between outflow and stage. ET is shown in Figure 11 as the cumulative effect of loss from evaporation directly from the surface water and loss by transpiration of the plants. Different methods for estimating ET were explored, including the use of various climatic-based empirical models and continuous water level data. Infiltration is shown in Figure 11 as loss of surface water to the local groundwater of the system and as lateral seepage of surface water out of the dikes, and is defined here as the cumulative effect of these flows. Figure 11 also shows a loss of local groundwater out of the system, whose flow was not be directly evaluated in this study. This loss, however, affects the quantity of local groundwater which is inversely related to surface water infiltration as shown in Figure 11, thereby affecting the water exchange between the surface water and local groundwater systems. Water exchange between the local groundwater and surface water was evaluated using groundwater well data and lateral hydraulic conductivities. Networks of groundwater wells on the dikes and hydraulic conductivities were used to provide data to estimate dike seepage. Total infiltration losses of surface water were estimated using continuous surface water data.

Evapotranspiration Estimation

Potential evapotranspiration (PET) was estimated using seven empirically based models and climatic data from the weather stations established in areas free of canopy cover. Data from the closest weather station were used for sites with no weather station. Data from the nearest weather stations that are part of the Florida Automated Weather Network program operated by the University of Florida's Institute of Food and Agricultural Sciences were used for periods of data gaps due to equipment failure. The weather data used in the various models included daily photosynthetic active radiation (PAR), minimum, maximum and average daily temperature and relative humidity, and average daily wind speed. Total solar radiation (W/m^2) was estimated from PAR ($\mu mol/m^2/sec$) using a conversion of 0.435 (Meek and others 1984). The seven empirical models used to estimate ET included Priestly-Taylor, Penman-Monteith, Hargreaves-Samani, Hamon, Makkink, Turc, and Thornwaite, whose equations follow:

Priestly-Taylor Method

$$\lambda PET=\alpha*(R_n-G)*\Delta/(\Delta+\gamma) \quad [6]$$

Penman Method

$$PET= [\Delta*(Rn-G) + \gamma*K_E*\rho_w*\lambda*u_2*(e_s-e_a)]/ \rho_w* \lambda*(\Delta+\gamma) \quad [7]$$

Penman-Monteith Method (modified by the FAO)

$$PET=[0.408*\Delta*(R_n-G) + \gamma*(900/(T+273))*u_2*(e_s-e_a)]/(\Delta+\gamma*(1+0.34u_2)) \quad [8]$$

Hargreaves-Samani Method

$$\lambda PET=0.0023*R_a*TD^{0.5}*(T+17.8) \quad [9]$$

Hamon Method

$$PET=0.1651*L_d*RHOSAT*KPEC \quad [10]$$

Makkink Method

$$PET=0.61*(\Delta/(\Delta+\gamma))*(R_s/58.5)-0.12 \quad [11]$$

Turc Method

$$RH<50\% \quad [12]$$
$$PET=0.013*(T/(T+15))*(R_s+50)*(1+(50-RH)/70)$$

$$RH>50\%$$
$$PET=0.013*(T/(T+15))*(R_s+50),$$

where PET is the daily PET (mm/day); λ is the latent heat of vaporization (MJ/kg); α=1.26 for wet or humid conditions; R_n is the net radiation (MJ/m^2/day) and is calculated as the difference between net shortwave radiation and net longwave radiation where shortwave radiation is total radiation minus albedo of water (0.05) and longwave is a function of total incoming radiation, clear sky radiation, temperature, and relative humidity (equations not shown); G is the heat flux density to the ground which is assumed to be negligible when calculating daily PET; Δ is the slope of the saturation vapor pressure temperature curve (kPa/^oC); γ is the psychrometric constant (kPa/^oC); K_E is a coefficient reflecting efficiency of vertical transport of water vapor and is calculated as a function of wetland area(equation not shown); ρ_w is the density of water (kg/m^3); T is average daily air temperature (^oC); u_2 is the wind speed at 2m height (m/s); (e_s-e_a) is the saturation vapor pressure deficit (kPa); R_a is the extraterrestrial solar radiation (MJ/m^2/day); TD is the daily difference between the maximum and minimum air temperature (^oC); L_d is the daytime length in multiples of 12 hours; RHOSAT is the saturated vapor density (g/m^3); KPEC is a calibration coefficient = 1.2; R_s is the daily solar radiation (MJ/m^2/day); and RH is the relative humidity (%).

The Penman method is typically applied to open water systems (Dingman 2002) and wetland systems (Mitsch and Gosselink 2000); therefore, Penman-predicted ET rates were selected to use in the runoff analysis, water balance analysis, and temporal hydrologic modeling.

Runoff Analysis

Runoff analysis was performed for each instrumented surface water feature with all rain events when no surface outflow occurred to avoid inclusion of outflow estimation error in the analysis. Event response (m) for each rain event was calculated as the stage increase from midnight of the day it rained to midnight of the day after it rained plus the associated daily ET estimated with the Penman method that occurred during the day it rained. Therefore, event response included the increase in the surface water from direct rainfall and runoff along with the balance of the loss from ET. Runoff depths were determined by subtracting the rainfall amount from the event response. Regressions of event response versus magnitude of rain were performed to determine if runoff could simply be predicted from rainfall and independent of watershed size and antecedent conditions.

Ratios of upland to wetland area were calculated for different surface water levels at 10 cm increments and were modeled as a function of stage using a multiple watershed approach (See “Watershed Delineation,” page 33). Two sites had these ratios calculated using three different scales of watershed delineation to observe effects of scale during runoff analysis. The largest scale delineates the entire CSA as one watershed contributing to the surface water feature. The smallest scale of analysis produces the highest number of individual watersheds within the CSA and therefore the smallest size watershed contributing to the surface water feature (See “Watershed Delineation,” page 33). Upland-to-wetland ratios were determined for all other water features with the smallest scale of analysis while adding contributing areas in the case water features merged. The latter was necessary since some surface water features were separate at low water levels, but became connected at higher water levels. Upland-to-wetland ratios were calculated from the stage prior to a rain event. The ratios, along with rain and runoff depths, were used to calculate runoff coefficients, C. It should be noted that the runoff coefficients should not be considered analogous to the runoff coefficients determined in the spatial modeling as the coefficients were calculated and employed differently. The runoff coefficients calculated here represented the percentage of total rain on the watershed that contributes to the surface water feature through overland flow using the following equation;

$$C = (\text{wetland area}/\text{upland area})*(\text{runoff depth}/\text{total rain}) \qquad [13]$$

Runoff coefficients were calculated for each rain event and at the three scales of watershed delineation for two sites. The coefficients were averaged over all events and compared among the different scales of watershed analysis. Average coefficients with the smallest scale of delineation for all water features were compared to observe differences both among and within the sites.

As a constant runoff coefficient for a feature may not be applicable to all events or conditions, models for predicting the coefficient based on magnitude of event and antecedent conditions were explored. Antecedent conditions were represented with two different indices. The antecedent rain index following Woods and Rowe (1996) was related to the 14 day history of rainfall prior to the event being modeled with the following equation which weights recent rainfall more heavily:

$$\text{Antecedent rain index} = I_1/1 + I_2/2 + I_3/3 + \ldots\ldots + I_{13}/13 + I_{14}/14, \qquad [14]$$

where I_n is the rainfall(m) that occurred n days previous to the event being modeled. An index, antecedent ET index, represented the amount of ET that occurred previous to the event and was calculated by summing the ET (m/day) of the previous 14 days (Dingman 2002). Multiple regression with the calculated runoff coefficients was performed in Microsoft Excel® to determine predictive models for the coefficients using rainfall amount, antecedent rain indices, and antecedent ET indices. The resulting models were compared to the simpler models that predicted runoff with rainfall.

All observed positive event responses were summed over the period of analysis and compared to the runoff estimated with various runoff models plus rainfall to compare the models' efficiency in predicting runoff.

Surface Water Outflow

Two of the monitored surface water features have weirs at the outfalls that were intermittently active. The inverts of the weirs were surveyed to determine the elevation difference between the invert and the ground surface of the surface water well. The wells were installed at a distance away from the weir where there was no effect from the hydraulic slope induced from flow over the weir. The distance was at least the recommended value of twice the vertical dimension of the weir opening (Dingman 2002).

Two of the sites have channels that produced surface water outflow at certain stages. Two wells were installed with logging pressure transducers in the center of each channel and at least 50 m apart. The upstream and downstream wells were surveyed to determine relative elevation differences between the two. The channel cross section at the upstream well was surveyed to determine cross-sectional flow area and wetted perimeter as a function of surface water level at the upstream well. The elevation differences and continuous water level data from the set of wells were used to calculate daily hydraulic slope (m/m). The hydraulic slopes were used to determine outflow (m/day) using Manning's Equation:

$$Q_o/A_w=1.0*R^{2/3}*S^{1/2}*A_x/n \text{ ,} \qquad [15]$$

where Q_o/A_w is the surface water outflow (m/day), R is the wetted perimeter, S is the hydraulic slope (m/m), A_x is the cross-sectional area, and n is a Manning's roughness coefficient. A_w, the wetland area, was calculated using surface water data and upland-to-wetland ratios. The roughness coefficient was field-calibrated with velocity measurements using an Acoustic Doppler Velocity (ADV) meter and the velocity-area flow calculation technique, where velocities were measured at 6/10 depth and at 50 cm increments across the width of the channel (Dingman 2002).

Water Balances

Water budgets for the monitored surface water features were performed using rainfall, predicted runoff, Penman-estimated ET, and surface water outflow and the basic water balance equation:

$$\Delta S/\Delta t=P - Q_o/A_w + R + G_i/A_w - G_o/A_w - ET \qquad [16]$$

where S is stage (m), t is time (days), Q_o/A_w is surface water outflow (m/day), R is runoff into the wetland from the surrounding watershed (m/day), G_i/A_w and G_o/A_w are groundwater inflow and outflow (m/day), P is precipitation (m/day), and ET is ET (m/day). The balance between groundwater inflow and outflow to the surface feature was treated as a residual value (m/day), simplifying Equation 16:

$$\Delta S/\Delta t= -Q_o/A_w + R + P - ET - \text{Residual} \qquad [17]$$

Water balances were performed with data when no surface water outflow occurred to minimize error from that estimation, except in the case where channels were instrumented to provide more accurate outflow estimates. In such circumstances, a surface water outflow term was included in Equation 16. Again, these water balances should not be confused with the simpler ones used in development of the spatial models.

The resulting residual values represented infiltration/exfiltration plus any propagated errors from the calculation of the other flows. It was assumed that most of the error was associated with ET and, therefore, positive residual values represented a combination underestimation of ET and/or infiltration. Runoff as percentage of rain and the resulting residuals were compared among features to identify differences among and within the sites.

Dike Seepage

Slug tests were performed in the dike wells on three different occasions by removing a volume of water and using pressure transducers to collect data on the time of recovery to the initial static water level. The Hvorslev (1951) method was used to calculate lateral saturated hydraulic conductivities of the groundwater features and dikes using the following equation:

$$K_s=r^2\ln(L/R)/2*L*T_0\,, \qquad [18]$$

where K_s is saturated hydraulic conductivity, r is the radius of the well casing, R is the radius of the well screen, L is the length of the well screen below the initial water level, and T_0 is the time it takes for the water level to rise 63% of the initial displacement.

Discretely collected groundwater levels in each of the dike wells were used to develop groundwater profiles across the dikes. The hydraulic conductivities and groundwater profiles were used to calculate dike seepage with a derivation of the Darcy equation for bank loss:

$$q_b=[k_s*D*L_b(dh/dx)]/W_A\,, \qquad [19]$$

where q_b is dike seepage (m/day), k_s is lateral saturated hydraulic conductivities (m/day), D is the depth of surface water within the dike (m), L_b is the length of dike (m), dh/dx is the slope of the groundwater profile across the dike (m/m), and W_A is the wetland area (m^2). Dike length is defined as the length of dike in contact with surface water and was measured using ArcGIS®. Wetland area was determined with the upland-to-wetland ratio and total site area. Dike seepage was estimated at different water depths within the dike to observe sensitivity of that parameter. Performing this analysis only on sites that have surface water in contact with the dikes follows the definition of dike seepage as lateral seepage of surface water (Figure 11).

Local Groundwater Analysis

Depths to water table recorded at the topographical highest wells were compared among sites. Continuous water level data from the groundwater and surface water wells, along with topographic data either from surveying or LiDAR, were used to determine the groundwater elevations relative to surface water elevations. The elevation differences between the recorded groundwater levels and surface water levels were evaluated to observe hydraulic gradients and thus direction of potential groundwater flow. As the potential for groundwater flow is created by the hydraulic gradient but limited by the saturated lateral hydraulic conductivity, slug tests were performed in the groundwater wells to obtain estimates of conductivity. The same method of determining saturated hydraulic conductivities used in dike seepage was employed and on three different occasions. Conductivities and elevation differences between groundwater and surface water were compared to determine differences among sites in terms of potential groundwater flows.

Drawdown Analysis

Time periods where no rainfall or surface water outflow occurred were identified for each site. Linear regression was performed with surface water levels (15 minute to 1 hour increments) during these times to calculate average surface water loss in terms of depth per day. The daily decline rates were compared to the associated daily ET rates calculated by the Penman method to observe how well the methods predicted loss and if significant differences occurred, suggesting infiltration and/or exfiltration. Additionally, the decline rates were compared among water features to observe any differences.

At selected surface water features, the original Solinst® pressure transducers were replaced by more recently developed Solinst® transducers which are more accurate (0.25 cm) with a higher resolution (0.005 cm) to more accurately separate ET and groundwater flows. The resolution of the data collected by the new transducers allowed analysis of diurnal fluctuations in surface water. The White (1932) method was utilized to calculate groundwater flow and ET rates (cm/day) using the equation:

$$ET = S_y(24h \ {}^{+}/_{-}\ s) , \qquad [20]$$

where ET is ET (cm/day), S_y is the specific yield (dimensionless, equal to 1.0), h is the net groundwater inflow rate (cm/hr), and s is the net fall (+) or net rise (-) over one day. The method requires two assumptions: (1) ET ceases at night and is negligible which allows calculation of groundwater flow (h) as cm/hr and (2) the calculated hourly groundwater flow from the night is constant and can be applied throughout the day. Groundwater flow (h) was conservatively estimated by performing linear regression with surface water data from 10 pm to 6, thereby not including the hours soon after sundown where a quicker recovery may be possible. The White method has often been applied to water table fluctuations where knowledge of the specific yield of the soils is required, while a specific yield of one is typically used when applied to surface water (Hill and Neary 2007). At rather low surface water levels and non-cylindrical bathymetries, however, the specific yield of the surrounding soils may result in overestimation of ET (Hill and Neary 2007). With low surface water levels, a unit area drop in water levels may occur both within the ponded water and within the soil at the edges of the feature. The drop that occurs in the soils results in a composite specific yield less than one and failing to account for this overestimates ET. Hill and Neary (2007) found limited effects from this phenomenon with surface water levels above 15 cm. The analysis of surface water levels with the White method was performed with surface water levels above 15 cm to avoid overestimation of ET and groundwater rates.

The White method was used to calculate daily ET rates and groundwater flow as infiltration rates. A negative infiltration rate was defined as groundwater inflow to a surface water feature and thus exfiltration. Monthly averages of the calculated daily rates were determined and compared among sites and to monthly average daily Penman-predicted ET rates. Total daily declines were determined using linear regression, and the resulting slopes were also included in the comparison.

Temporal Hydrologic Modeling

Hydrologic models were developed that predict daily stage (m) for the instrumented surface water features using rainfall, estimated ET rates, the developed runoff models, and Equation 17 with the exclusion of the residual term. The temporal models were developed separately from the spatial models and using the more complex water budgets and runoff analysis. Daily ET was estimated with the daily climatic data and Penman method. Infiltration rates equivalent to the residuals from the water balances were included and the models' predictive abilities were evaluated by comparing to actual surface water data. Seasonal, multiplicative coefficients applied to Penman-estimated ET rates were explored and infiltration rates were adjusted in an iterative approach to produce the best fits. Seasonal coefficients refers to coefficients multiplied by Penman-estimated ET rates in the non-growing and growing seasons, with the latter including April through September. The seasonal ET coefficients are analogous to the crop coefficients used in the Canopy Cover Coefficient (CCC) method which is typically performed in conjunction with the Penman-Monteith method (Drexler and others 2004). The quality of fit between the modeled stage and actual stage was quantified using a correlation coefficient, R^2, value for nonlinear regressions with the equation

$$R^2=1.0-SS_{reg}/SS_{tot}\,, \qquad [21]$$

where SS_{reg} is the sum of the squares of the differences between predicted stage and actual stage and SS_{tot} is the sum of the squares of the differences between predicted stage and average actual stage.

Ecohydrology

Experimental Design

An ecohydrologic evaluation of a dominant CSA wetland tree species, *Salix caroliniana*, and CSA soils and hydrology was performed on Mosaic K5, CFI SP-1, and Williams during the summer of 2007. The study was conducted along transects from an upland area into a surface water feature. Groundwater wells already installed on the transect or newly installed groundwater wells provided water table depths along the transect, and surface water wells within the water feature provided water levels at the lowest point of the transects. The experimental setup for Mosaic K5 is shown in Figure 12. The elevations were surveyed and are presented as relative to the ground surface at the surface water well (Figure 12). Four ecohydrology stations were established across the transect at Mosaic K5. The highest area, referred to as upland zone, was established at a zero distance. The transitional zone station was established in an area 40 cm lower than and 41 m from the upland zone. The saturated zone was established 60 m from the upland station at an elevation 20 cm lower than the transition zone. The inundated zone station was established 200 m away from and at an elevation 1.5 m lower than the upland zone. Each of these stations, except for the saturated zone, had a well installed, shown in Figure 12 with a blue rectangle, which continually recorded levels with a Solinst® pressure transducer. At each location, except for the inundated zone, soil moisture probes were installed and soils were sampled to perform moisture release curve analysis in the laboratory. At each location, transpiration data was collected and root biomass was sampled.

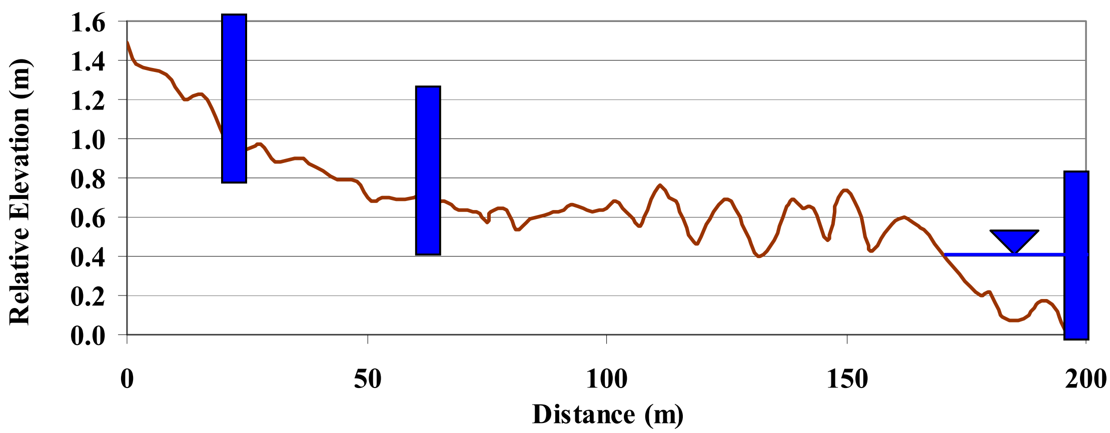


Figure 12. Ecohydrology Experimental Design at Mosaic K5.

A similar design and data collection scheme was implemented at CFI SP-1. The site was not inundated during the study and, therefore, only three stations were established which all had adjacent groundwater wells (Figure 13). The upland zone was established in sand-dominated soils and at an elevation 2 m higher than the saturated zone. The transitional zone was established 30 cm higher and 25 m away from the saturated zone. A similar setup to CFI SP-1 was employed at Williams Co., which also did not experience inundation during the study period and only had three stations established. While the same data was collected at Mosaic K5 and CFI SP-1, only sampling for moisture release curve analysis and installation of soil moisture probes were performed at Williams Co. Transpiration studies or root sampling were not conducted at Williams Co.

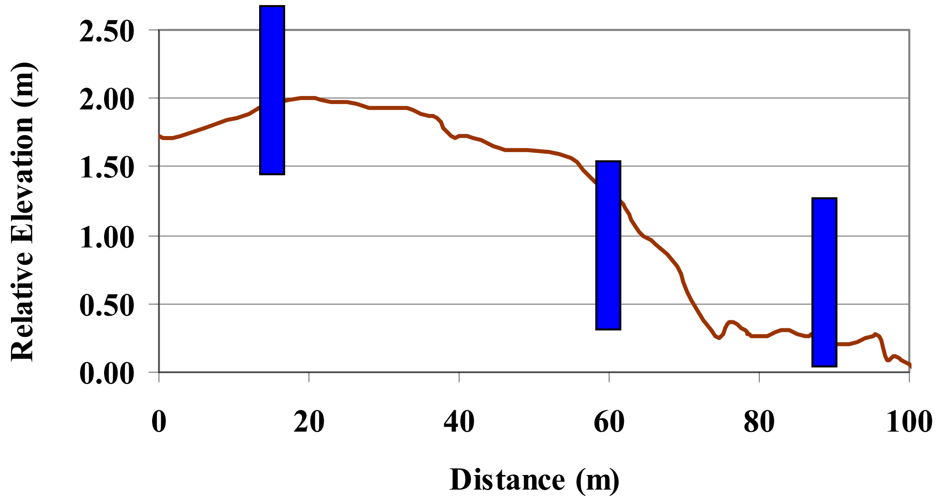


Figure 13. Ecohydrology Experimental Design at CFI SP-1.

Moisture Release Curves

Soils were sampled at each ecohydrology station and transported back to the laboratory where they were allowed to soak in water for 3 days. Hysteric behavior between moisture release curves of wetting and drying soils has often been observed (Dingman 2002). Wetting the samples first resulted in moisture release curves for drying soils. The samples were removed from the water and allowed to dry in constant conditions with no sun or direct heat. At 1-3 day increments during the drying period, sub samples (approximately 100 cm^3) were removed and placed into sealed bags where they were allowed to equilibrate for four days. Decagon Devices EC-5® dielectric soil moisture probes (accuracy = 3% VWC, resolution = 0.1% VWC; independent of soil texture and salinity) were inserted into the sub samples to obtain average volumetric water content (VWC) (m^3/m^3) for each sub sample (Decagon Devices, Pullman, WA, USA). Immediately after removing the soil moisture probes, sub samples were analyzed for water potential. A UMS T5® tensiometer (range = 0 to -850 hPa) was used on the wetter samples in which it was inserted to record negative water potential (hPa) (UMS, Munich, Germany). A UMS Infield 7® was used to obtain direct readings from the tensiometer. The tensiometer was checked daily by wetting the ceramic tip to obtain a measurement of 0.0 hPa (+/- 3 hPa). The instrument's range limited its application to only wetter samples so to avoid cavitation. The drier samples were analyzed with a Decagon Devices WP4T® to obtain water potential (MPA) with accuracy of 0.1 MPa from -0.0 to -10 MPa and 1.0% from -10 to -300 MPa (Decagon Devices, Pullman, WA, USA). The instrument utilizes the chilled mirror dew point technique and a dew point potentiometer to measure water potential (Scanlon and others 2002). The WP4T® was calibrated daily using the supplied 0.5 molal KCl calibration standard. The accuracy of the instrument limits confident results to a minimum water potential of -0.2 MPa, corresponding to 2000 hPa. Therefore, the possibility of samples having water potential values out of the ranges of both instruments and in between -850 and -2000 hPA was possible.

All water potentials were converted to negative water pressure (-cm H_20) and were related to the measured VWCs. The resulting relationships, referred to as moisture release curves, for each station were analyzed for critical values including saturation levels, permanent wilting points (PWP), and capillary fringes. Saturation levels corresponded to the VWCs that were related to water potentials near zero (Dingman 2002). VWCs that induced PWP, assuming permanent wilting occurs at -15,000 cm H_2O, were identified for each site (Dingman 2002). Capillary fringe heights were determined by observing the inflection point of the curves with decreasing VWCs. Capillary fringes heights were assumed to be equivalent to the absolute value of the water potential where VWCs begin to significantly change with changing water potential (Dingman 2002).

Soil Moisture Analysis

Decagon Devices EC-5® dielectric soil moisture probes were installed at various depths at each ecohydrology station, excluding the inundation zone at Mosaic K5. A 7.62 cm diameter corer was used to create a hole slightly over 1 m deep that was adjacent to the groundwater well and in an area within 5 cm of the surface elevation of the groundwater well. The soil moisture probes were installed at 10, 25, 40, 70, and 100 cm depths below the ground surface at the upland and transitional stations. Depths of 10, 20, 35, and 50 cm were used at the saturated stations. The probes, which consist of two prongs, were installed in the sidewall of the cored hole and perpendicular to the ground surface to avoid effects from infiltrating water pooling on the probes. The probes were inserted into the sidewall so that they were completely covered. The removed soil was backfilled and packed in best attempts to mimic the initial bulk density of the cored hole. The soil moisture probes were connected to a Decagon Devices Em5b® logger which was placed in a water resistant container. The logger was set to record soil moisture as VWC every hour. Soil moisture levels were compared to accompanying groundwater levels to evaluate relationships between the two.

Root Biomass Analysis

The coring method (Snowdon and others 2002) was used around *Salix caroliniana* trees to sample root biomass with depth at increments of 15 cm and to a total depth of 1.5 m. Sampling was performed near each ecohydrology station, including at the inundated station of Mosaic K5. Root biomass was sampled from three trees at each location and with three cores per tree. Trees were sampled at locations within 5 cm of the surface elevation of their respective soil moisture station. Sampled trees had dbhs ranging from 7 to 10 cm to minimize the effect of different tree age on root biomass allocation. Samples were taken to a lab where they were washed, sieved with 2 mm mesh, and dried in 70 ^{0}C oven until constant weight was achieved. Root biomass data were expressed as average dry biomass per sampled volume per depth and related to soil moisture data to observe how root allocation responded to soil moisture dynamics.

Transpiration Analysis

Stems of four *Salix caroliniana* trees surrounding each ecohydrology station were instrumented with Dynamax Sapflow® gauges, along with a Dynamax Flow4 DL® sapflow logger, to obtain stem flow rates as mass of water per hour using the stem heat balance technique (Dynamax, Houston, TX, USA). Sapflow® technology has been tested and proven to give accurate measurements of water flow within a stem (Akilan and others 1994; Brooks and others 2003; Gansart 2003). Four sapflow gauges were used, two that were installed on stems with diameters ranging from 24 to 32 cm and two installed on stems with diameters of 45 to 65 cm (Dynagage Models SGB 25 and SGB 50). A station was instrumented for at least 14 days before the sapflow system was moved to another station. All instructions provided by Dynamax® were followed including careful attention to provide the best contact between the stem and the sensor by sanding the stem and insulation of the sensor with provided heat shields and additional foil wrap. Diameter, leaf area index, and canopy size of instrumented stems were measured and recorded. Leaf area index was measured with a ceptometer (Decagon Devices, Pullman, WA, USA) and used along with measured canopy size to determine total leaf area of each instrumented stem.

Hourly stem flow rates were converted to daily flow rates, which were divided by the cross-sectional area of the instrumented stems to obtain stem flow per cross-sectional area (g H_2O/cm^2/day). These rates were then divided by the daily ET rates as estimated with the Penman method using on-site climatic data. Indexing with the daily Penman-estimated ET allowed comparison among stations and sites while excluding climatic effects. The indexed daily stem flow rates from all four sensors collected over the 14 day period were averaged to calculate average daily stem flow rate (g H_2O/cm^2/Penman ET) for each station. A 180 m^2 belted transect for each station was established with the soil moisture station as the center and where all *Salix caroliniana* trees were measured for dbh to calculate total basal area. Daily stem flow rates per stem cross-sectional area and total basal area were used to scale up rates and determine stand transpiration (cm/day) using the equation;

$$\text{Stand Transpiration} = (\text{stem flow}/\text{stem A})*(\text{total basal A}/\text{stand A})*(1/\rho) \qquad [22]$$

where stand transpiration is cm H_20/day, stem flow/stem A is the daily stem flow per cross-sectional area of the instrumented stem (g H_2O/cm^2/day), total basal A (cm^2) is the total basal area of *Salix caroliniana* measured on the belted transect, stand A is the area of the belted transect, and ρ is the density of water (0.998 g/cm^3). Such a calculation assumes that all *Salix caroliniana* trees in a belted transect have the same water availability and experience similar stem flow rates per stem area as the instrumented stems. The assumption of water availability is reasonable as the belted transects were established in fairly flat areas that immediately surrounded the soil moisture station. The plots were surveyed, and the relief in all plots was limited to 10 cm.

A daily stand transpiration rate was calculated for each stem flow rate and was indexed with daily Penman-estimated ET. The stand transpiration rates were averaged to obtain average daily stand transpiration per Penman-estimated ET (cm/cm) for each station. Both the daily stem flow rates and stand level transpiration rates were compared between stations and among sites.

EVALUATION OF B. RUSHTON FIELD TRIALS

Site and Plot Selection

Five CSAs where Betty Rushton conducted field trials were selected for study (Rushton 1988). Sites were chosen that were currently accessible and that had an average of at least 50% tree survival after one year. Table 3 presents a summary of the selected sites.

Table 3. B. Rushton Site Summary Table.

Site Name	Symbol	Years Abandoned (Estimated)	Type	No. of Cypress-gum Plots	No. of Hydric Swamp Plots
CFI SP-1	CFI	23	Sand-Clay	6	0
Homeland	HOM	46	Sand Cap	8	0
O.H. Wright	OHW	46	Clay	3	4
Peace River Park	PRP	38	Clay	0	4
Tenoroc-4	TEN	34	Clay	8	4

A total of 37 planted plots on 5 CSAs were selected for study. Selected plots were locatable in the field from site diagrams (Rushton 1988) and able to be matched to an original plot number. All selected plots had at least one surviving tree at the present time. Plots were representative of the two planting schemes used, referred to by Rushton as cypress-gum plots and hydric-swamp plots. Figures 14 and 15 depict planting schemes for these two types of plots. 25 cypress-gum plots and 12 hydric swamp plots were selected. Species planted in the two plots types are listed in Tables 4 and 5.

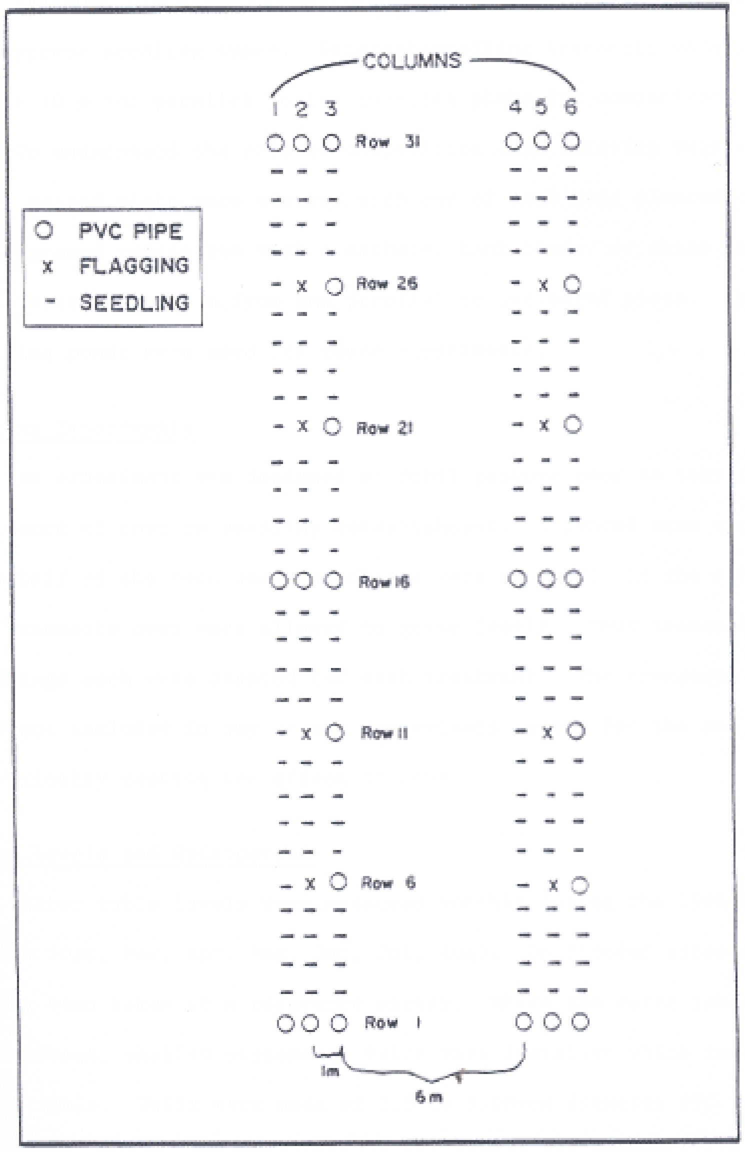
Table 4. Species List for Cypress-Gum Plots.

Species	Symbol
Fraxinus pennsylvanica	FRPA
Nyssa aquatica	NYAQ
Taxodium distichum	TADI

Table 5. Species List for "Wet" and "Transitional" Hydric Swamp Plots.

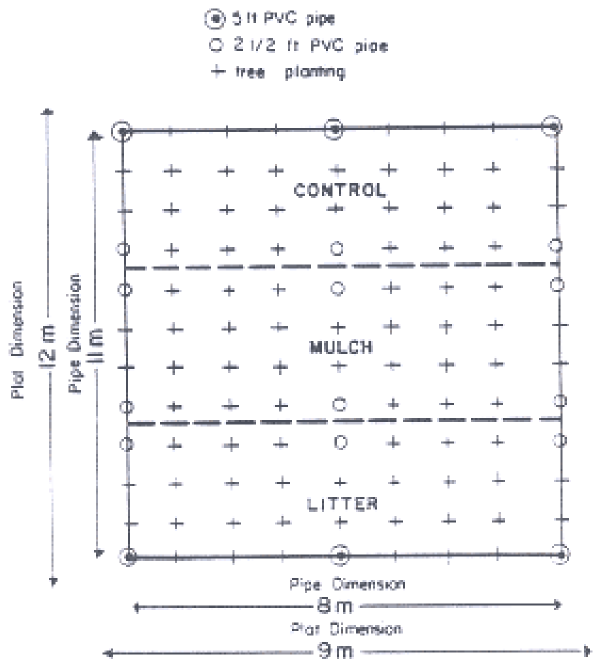
"Wet" Plots		Transitional Plots	
Species	Symbol	Species	Symbol
Fraxinus caroliniana	FRCA	*Acer rubrum*	ACRU
Nyssa sylvatica	NYSY	*Gordonia lasianthus*	GOLA
Persea palustris	PEPA	*Nyssa sylvatica*	NYSY
Quercus laurifolia	QULA	*Quercus laurifolia*	QULA
Taxodium distichum	TADI	*Sabal palmetto*	SAPA
Ulmus americana	ULAM	*Taxodium distichum*	TADI

Cypress-gum plots were planted with all three species except for 4 plots at Tenoroc-4 planted only with two species. Among the 12 hydric swamp plots, 8 were planted with species with a group of 'transitional' trees and four were planted with a group of 'wet' trees.



Note: Two plots are pictured. Each plot was planted with 93 seedlings.

Figure 14. Cypress-Gum Plot Layout from Rushton (1988).



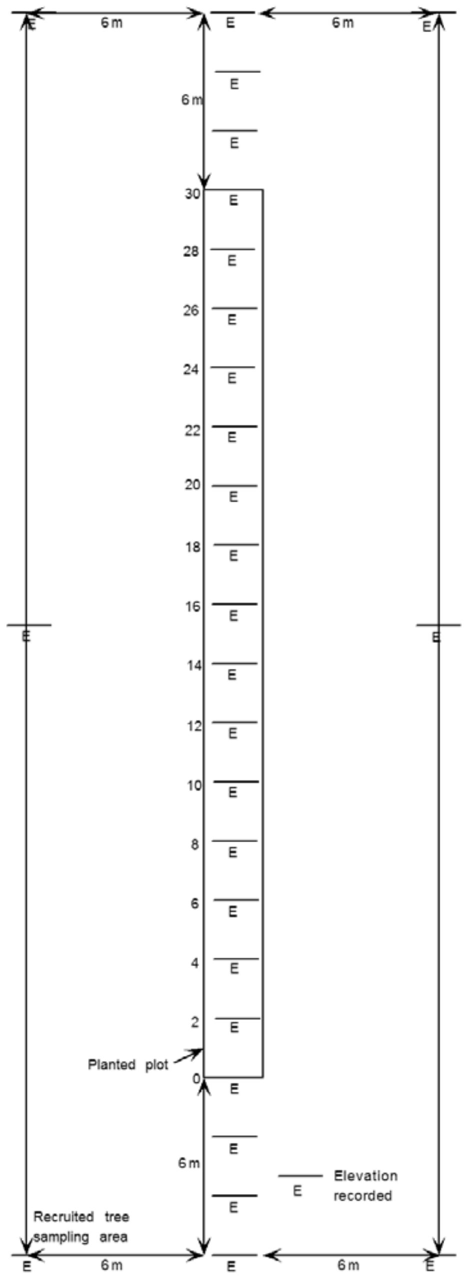
Note: Two plots are pictured. Each plot was planted with 108 seedlings.

Figure 15. Hydric Plot Layout from Rushton (1988).

In order to compare the ecosystem development on non-planted areas that were similar to the Rushton plots at the time of planting, "reference" plots were designated. Reference plots were of equal dimensions to the Rushton plots for which they were a reference. Reference plots had to be at least 25 meters away from Rushton plots. A single reference plot was designated for all plots that shared connection to a water feature. Reference plot selection was random provided that a plot met the conditions: (1) it was adjacent to the same water feature and (2) the topography was similar so that a similar hydrologic regime to the Rushton plot could be inferred. An exception to the first condition occurred at Homeland, where the reference plot was located in a pond fed by a ditch from the pond containing the Rushton plots, because not enough non-planted area within the pond with the Rushton plots was available. At all sites, one reference plot served as a reference condition for between 1 and 8 Rushton plots.

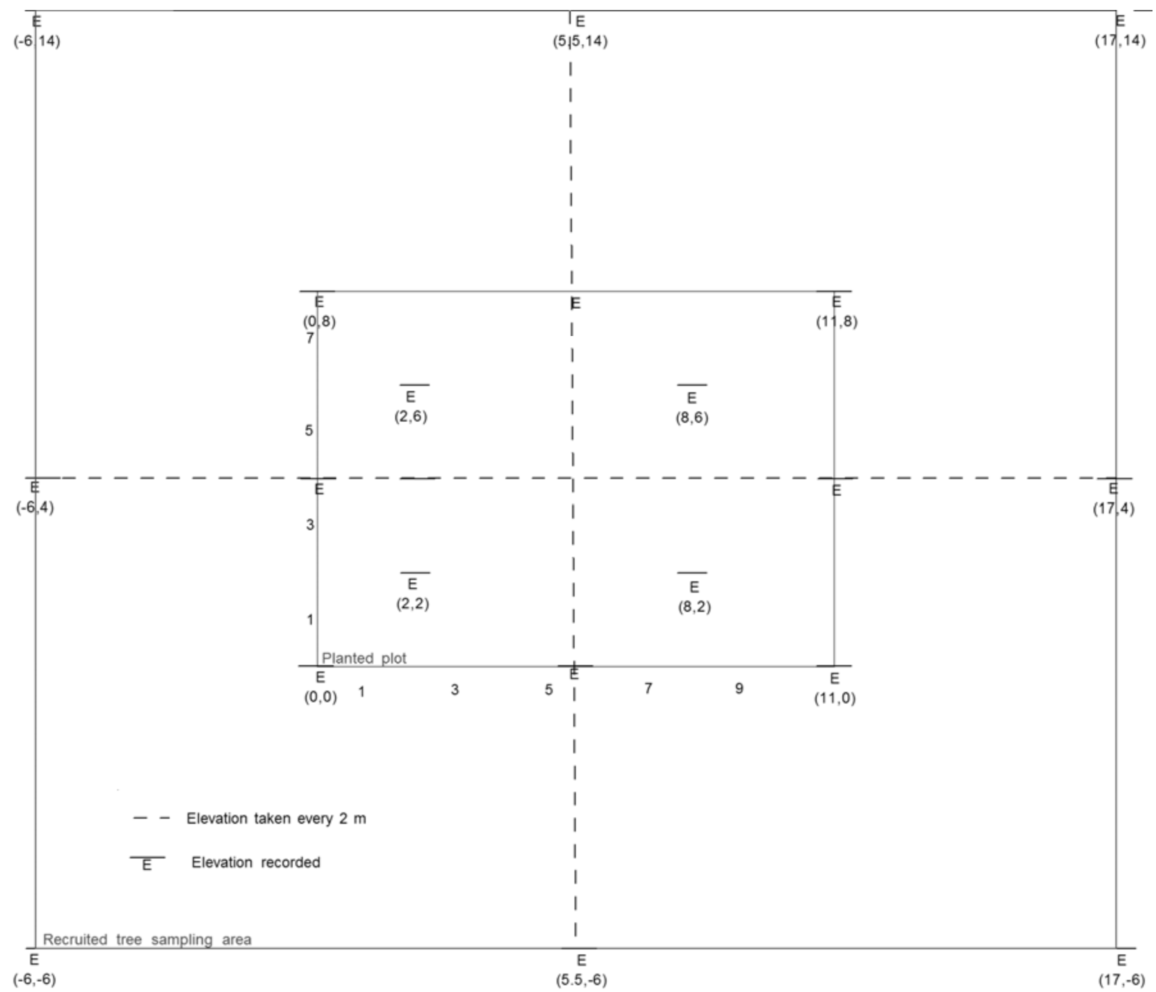
Data Collection

A laser level was used to determine elevations within the plot relative to the water level at the time of first visit. Figure 16 shows where data was collected in cypress-gum plots. For these plots, elevation was recorded every meter along a 42 m longitudinal axis which traversed the planted area as well as 6m in front and back of it. The plots were originally laid out such that this axis ran parallel to the elevation gradient. Additionally, elevation data was recorded from spots 6m to each side of the transect at the beginning, middle, and end of the longitudinal transect. Figure 17 shows where data were collected in hydric swamp plots. For hydric swamp plots, elevation data were recorded every two meters along two perpendicular axes crossing from the 6m away from the edge through the center of the plot to 6m beyond the far edge. In these plots elevation data were also collected at the soil and plant sample points within the planted plot, and at the four planted plot corners.



Note: Numbers are in meters.

Figure 16. Elevation Diagram for a Cypress-Gum Plot.



Note: Numbers are in meters.

Figure 17. Elevation Diagram for a Hydric Swamp Plot.

For reference plots, elevation data were collected in the same manner according to whether it matched the cypress-gum or hydric swamp dimensions, except in these plots only data within the plot boundary were collected.

Water levels at a point of recorded elevation were manually measured to the nearest centimeter each month from the initial visit to a plot in the spring or early summer of 2005 through October 2005.

On two sites, continuous digital data loggers were installed close to or within Rushton plots at CF SP-1 and Tenoroc-4. At these sites, one surface water well within the water feature and one ground water well 25m into the upland were equipped with loggers that recorded hourly water levels. The loggers were operational from the date of installation in the early part of the growing season through the end of October.

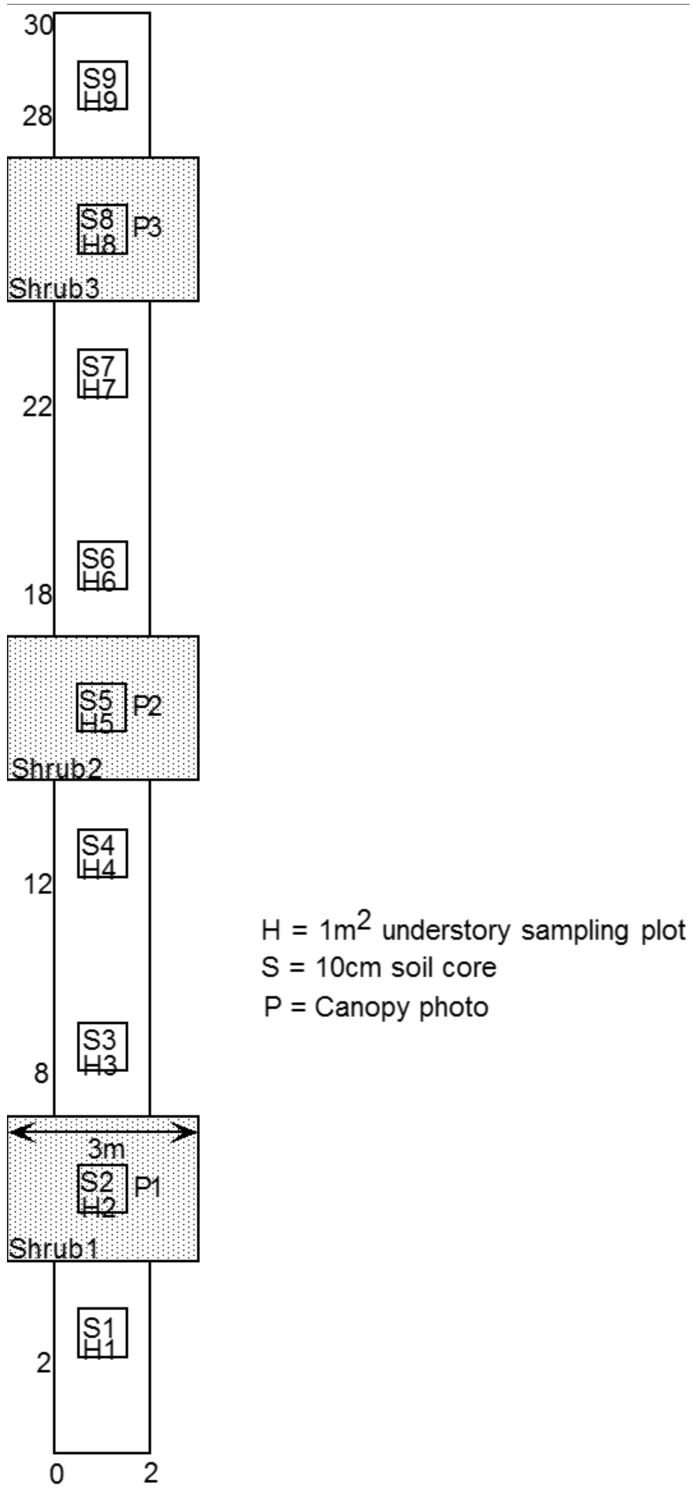
Planted and recruited trees were identified by location and species. X,Y plot location was recorded to the nearest meter for each of these trees. Recruited trees are defined in this study as individuals of the same species as planted trees not occurring in originally planted locations, irrespective of the size of the individuals. Diameter at 1.5 meters (DBH) to the nearest centimeter was recorded for all stems originating below that height. If no stems reached 1.5m, height of the tallest stem was recorded to the nearest centimeter.

All individuals of unplanted species found within the planted plot were identified to species and their DBH was recorded if they reached 1.5m in height. Woody plants were classified as trees or shrubs according to Tobe and others (1996). For *Salix caroliniana*, which is classified as a tree or shrub, individuals with at least one stem with a DBH $\geq$ 5cm were classified as trees. In cypress-gum plots, the 10m segment (0-10, 10-20, 20-30) that a tree was found in was noted.

In the plot where the greatest number of seedlings emerged, the seedlings were re-sampled at the end of the growing season to determine the survival rate.

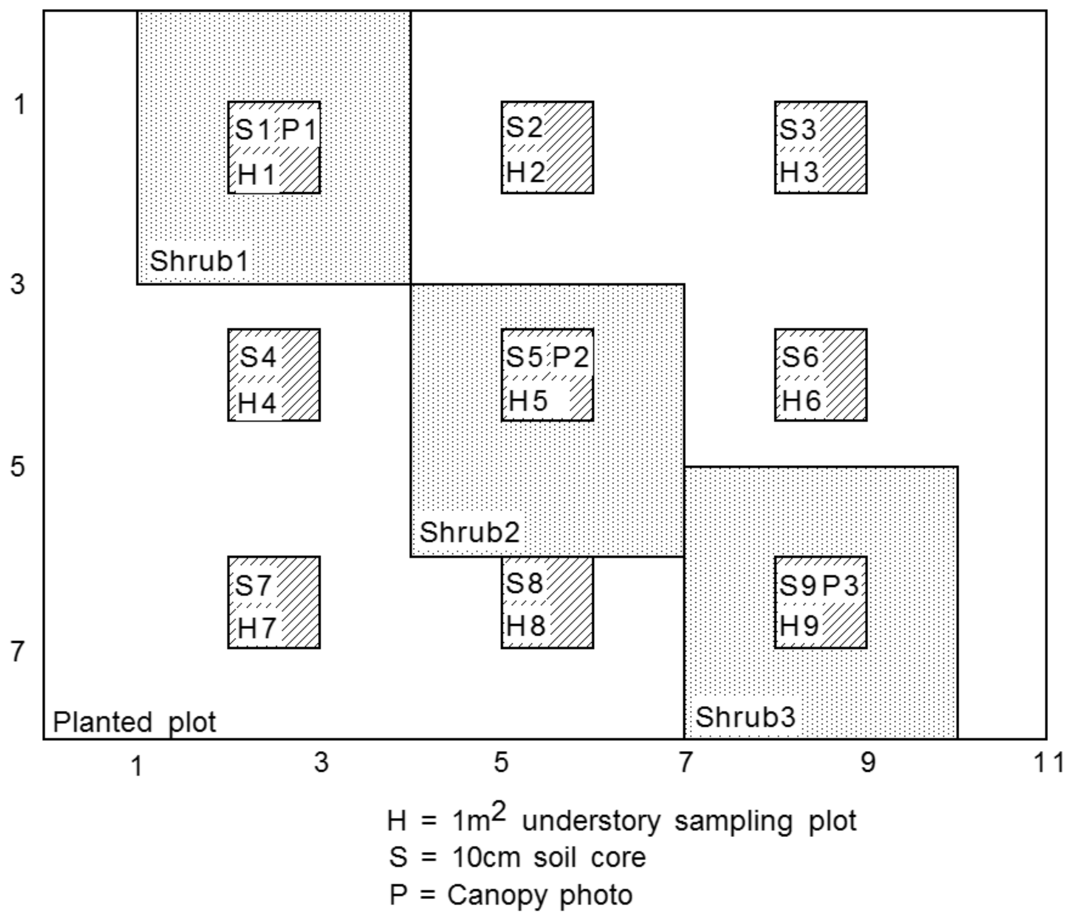
Additional Measures of Ecosystem Development

Figure 18 and 19 show the standardized sampling locations for shrubs, understory vegetation, soil, and canopy photos for cypress-gum and hydric swamp plots.



Note: Numbers are in meters.

Figure 18. Soil, Understory, Shrub, and Canopy Photo Sampling Scheme for Cypress-Gum Plots.



Note: Numbers are in meters.

Figure 19. Soil, Understory, Shrub, and Canopy Photo Sampling Scheme for Hydric Swamp Plots.

Three 3 × 3 m subplots within each plot were sampled for shrubs. DBH was recorded for all stems ≥1.5m in height and species was recorded for all individuals.

Nine 1 × 1 m subplots within each plot were used to sample all understory macrophytes with stem heights ≤1.5m. Each species occurring was identified and the coverage of each species was estimated into one of five possible coverage classes: 1: 1-10%, 2: 10-25%, 3: 25-50%, 4: 50-75%, 5:75-100%. Coverage was defined as the percentage of the 1 × 1 m horizontal plot area covered by the plant. In the case where different species occupied the same horizontal location but different vertical strata (overlapping), both species were counted.

Cores of the top 10 cm of the soil were collected with a 7.6 cm-diameter auger at within all 1x1m understory sampling plots.

To estimate canopy cover, hemispherical photographs were taken using a Nikon digital camera, with 180 degree "fish-eye" lens. Inside all plots, photos were taken in 3 equi-spaced understory subplots. For the Rushton plots, photos were also taken from the understory subplots outside of the canopy. The camera was placed on a tripod approximately 50 cm above the ground or slightly above the surface of the water, whichever was higher. The camera was then leveled with the lens pointing up, oriented so the back of the camera faced north, and zoomed out to 100%. When possible, photos were taken close to dawn or dusk or on overcast days to avoid distortion from direct sunlight.

Information about possible disturbance or site modification during the 20 year period since the trees were planted was collected from site managers, from the Rushton dissertation, through consultation with Betty Rushton, or through inference from evidence found in the plot in 2005.

Data Analysis

Topographic data collected was input in x,y,z form into Surfer surface mapping software, from which a kriging function was used to create a surface map. From this interpolated map, relative elevations were output for every square meter. Using this elevation data and the monthly water level data, water levels were calculated for the entire sampling area for every date water level was recorded. "Average water depth," as referred to in the remainder of the study, refers to the average of these monthly water levels.

On the two sites (CFI and TEN) where continuous data-logging water level recorders,. The average of the sampled monthly water levels was compared with the average of all the hourly water levels recorded by the data-loggers on each of these two sites.

A relative elevation at every tree along with monthly water level measurements allowed for determination of the average sampled depth of water for every tree and at every location where soils, shrubs, understory vegetation, and canopy photo sampling occurred.

Basal area, $BA(m^2)$, was calculated for trees and shrubs as the sum of the all stem area at 1.5m for an individual according to the following equation:

$$BA = \pi * dbh^2 \qquad [23]$$

Plot basal area (m^2/hec) was the sum of the tree and shrub basal area (m^2) divided by the plot area (hectares). Plot basal area was calculated for every 10m section of cypress-gum plots as well as for the entire plot, but only for the entire plot in hydric swamp plots because trees were not subsampled in these plots.

Soil cores were manually homogenized and three 40g samples of each core were dried a minimum of 48 hours at 30° C. The ignition method was then used to estimate % organic matter (% OM). Dried samples were ground with a mortar and pestle and three 1 g sub-samples were ashed in a muffle furnace for 6 hours at 450° C. This temperature was deemed appropriate for burning off the organic matter without removing inorganic carbon ($CaCO_3$). The loss from ignition is a rough estimate of organic matter. The following equation was used to calculate percent organic matter:

$$\text{(dry weight – ashed weight) / dry weight = \% organic matter} \qquad [24]$$

All surviving planted and recruited trees were classified into size classes that represented 5 or 10 cm DBH intervals (Table 6). However classification was done by basal area, to accommodate multiple stem trees where summation of DBH would have resulted in inflated values and inflated classification.* Classified trees were then grouped by species and by basins to define a population. Basins are defined as areas where multiple plots are adjacent to the same body of water and no plot is more than 50 meters away from its nearest neighbor. The sampled area of each basin represented the sum of the seedling sampling areas of every plot within the basin; not the area of the entire basin.

Table 6. Size Class Key Used in Tree Size Class Distributions.

Size Class	DBH (cm)	BA (cm^2)
0	NA	0
1	0.1-5	0.01-19.6
2	5-10	19.7-78.5
3	10-15	78.6-176.7
4	15-20	176.8-314.2
5	20-30	314.3-706.9
6	30-40	707-1256.6
7	>40	>1256.6

In order to predict the population trajectory of a planted tree population, a size class matrix population model was constructed for populations of planted *Taxodium distichum* at CFI and in one basin at OHW.

Size class matrix population models use principles of matrix algebra to estimate changes in population distribution over a time series as well as the steady-state population distribution and growth rate (Caswell 1989). Size class bins are determined and individuals are classified into size classes. A transition matrix (Figure 20), A, is constructed by determining probabilities after a year that a tree will remain in a size class, P_i, transition, G_i , and/or reproduce, F_i. This figure is a matrix for a population with four

*For example, a tree with two 5 cm DBH stems has less basal area (39.4 cm^2) than a tree with one 10 cm DBH stem (78.5 cm^2).

size classes (based on the number of columns). The *P* values along the diagonal represent probabilities of remaining in the same size class; the *G* values represent the probability of advancing into the next class, and the F values represent the probability of successful reproduction.

$$A_b = \begin{pmatrix} P_1 & F_2 & F_3 & F_4 \\ G_1 & P_2 & 0 & 0 \\ 0 & G_2 & P_3 & 0 \\ 0 & 0 & G_3 & P_4 \end{pmatrix}$$

Figure 20. The Format for the Transition, *A*, Matrix for a Matrix Population Model.

The transition matrix is multiplied by a vector of the number of individuals in each size class, N_t, to determine the number of individuals in each size class after one time increment, N_{t+1}. According to matrix theory the transition matrix alone determines the long-term population state. Mathematically decomposing the transition matrix, *A*, yields a vector of eigenvalues and their associated eigenvectors. The dominant eigenvalue of *A*, λ, gives the population growth rate at steady state and its associated right eigenvector is the stable state population distribution.

Customarily tracking the growth, survival, and seed production of a cohort of trees over a period of years provides the data from which transition probabilities are calculated. In this case, empirical time series data was not available for the entire period. Using data from the most current year and incorporating data on survival and growth after 1 and 3 years, survival and growth of individual tree histories were interpolated by fitting a curve based on the growth rate of other *Taxodium distichum* in the phosphate mining area. Reproductive probabilities were calculated based on the ratio of first-year seedlings to mature adults.

The matrix populations models were created in the Python 2.3 programming language. The model was programmed to estimate population change over a 50 year period. An elasticity analysis of the model was conducted to estimate the sensitivity of the model to the probability values in the transition matrix, A.

Models were not created for populations of other species for two reasons: 1) good basal area growth data from trees in central Florida from which to interpolate a growth curve were lacking, and (2) there were too few seedlings of these species found from which to calculate reproductive probabilities with confidence.

Canopy photos were analyzed in Adobe® Photoshop software. Photos were transformed into 2-color black and white images based on a threshold. The threshold was chosen to give the best conversion of vegetation pixels to black and all sky pixels to white. Before transformation, images were cleaned up with editing tools to remove shadows, clouds, sun spots, glare, or other aspects of the image that would have been incorrectly assigned to black or white. After transformation, the black and white pixels were counted in MFworks software. The percent canopy cover was then calculated as the sum of black pixels divided by the sum of black and white pixels.

Cover for all understory vegetation in a plot was estimated using the mean of the coverage class. The classes thus corresponded to the following percentages: Class 1: 5.0%; Class 2: 17.5%; Class 3: 37.5%; Class 4: 62.5%; Class 5: 87.5%.

Species richness was calculated for all plots as the sum of the unique species occurring. Shannon evenness, a measure of the evenness of the distribution of species, was calculated with the following formulas:

$$E = H / \ln(S) \qquad [25]$$

$$H = \sum_{i=1}^{s} (p_i * \ln(p_i)) \qquad [26]$$

where evenness, E, is equal to the Shannon-Weiner index, H, divided by the natural log of the total number of species, S. The Shannon-Weiner index was calculated as the sum over all the species of the average cover of a species, p_i, times the natural log of itself.

Prevalence (P) for species occurring in the understory were calculated using the following equation:

$$P_s = rf_s + rc_s \qquad [27]$$

where prevalence of a species, P_s, is equal to the sum of the relative frequency, rf_s, and relative cover, rc_s, of that species. Relative frequency was calculated using the following equations:

$$rf_s = f_s / \sum_{s=1}^{n} f_s \qquad [28]$$

$$f_s = o_s / q \qquad [29]$$

Relative frequency is equal to the frequency of a species, s, divided by the sum of the frequency of species encountered on a plot. The frequency of a species was calculated by the number of a 1m^2 quadrats in which species s occurred, o_s, divided by the number of 1m^2 quadrats, q, in a plot.

Relative cover was calculated using following equations:

$$rc_s = c_s / \sum_{s=1}^{n} c_s \qquad [30]$$

$$c_s = \sum_{i=1}^{q} c_{si}$$

[31]

where the relative cover of a species, rc_s, is the cover of a species divided by the sum of the cover all species, n, in a plot. The cover of a species, c_s, is equal to the sum of the mean cover of a species, s, in all $1m^2$ quadrats, q. Because a cover class was assigned to a species rather than a mean cover, each cover class was translated to a mean cover (reference on method) as follows: 1: 5%, 2: 17.5%, 3: 37.5%, 4: 62.5%, 5: 87.5%.

In order to visualize the differences in the cover of prevalent understory species between plots, an ordination technique called Nonmetric Multidimensional Scaling (NMDS) was applied. The prevalent understory species were those with a prevalence value of >0.10 (out of 1.0) for a plot. The NMDS method is regarded as the most appropriate ordination technique for ecology (Faith 1987; McCune 2002), as it does not require assumptions that the data fits a normal distribution nor that the data fits a linear pattern. The NDMS was run on a (nxp) contingency table of average species cover in a plot where the rows, n, were plots, and the columns, p, were species. The data was first standardized using a Wisconsin double standardization and then square-root transformed. A Bray-Curtis dissimilarity method was used as to create the dissimilarity matrix necessary to rank plots by dissimilarity and to position the point along the two principal component axes, so that the ordination could be shown in two-dimensional space.

To find patterns in the relationship between Rushton trees and total basal area, canopy cover, understory cover, understory species richness, understory species evenness, and soil organic matter, correlation matrices were created using R statistical software. Pearson's formula was the correlation method used to produce the matrices.

WETLAND REVEGETATION FIELD TRIALS

Location, bathymetry, and wetland flora, prior to planting efforts, are described for each revegetation site. Planting designs including species composition and placement for each wetland site are illustrated. Methods for monitoring and analyzing planted and volunteer vegetation, soil parameters, and wetland hydroperiod are summarized.

Site Description

Marsh Revegetation Sites

Two marsh sites, both located on Clay Settling Areas (CSAs) in Polk County Florida, were selected for revegetation and study. Average maximum and minimum annual temperatures for Polk County (Bartow, Florida) are 83.6° F and 61.6° F, and average annual rainfall is 136.5 cm (SERCC 2008). The first site, at Mosaic's Hooker's

Prairie 1 (H1) CSA is located southwest of the town of Bartow. Constructed in 1978, this 56 ha CSA was retired from filling, ditched, and reclaimed approximately 20 years ago. In 2004, vegetation was sampled across the environmental gradient, from upland to wetland area. *Imperata cylindrica* (cogon grass), *Ludwigia peruviana* (Peruvian primrose willow), and *Baccharis halimifolia* (eastern baccharis) were present in drier areas, while deeper wetland areas were dominated by *Typha latifolia* (cattail) and *Salix caroliniana* (Carolina willow). A 0.36 ha wetland area, located on the northeastern side of H1 was chosen for the revegetation (Figure 21). Prior to planting the wetland area was dominated by *Typha latifolia* and surrounded by *Salix caroliniana*, *Myrica cerifera* (wax myrtle), and *Momordica charantia* (balsam pear) on higher ground. The entire H1 CSA was treated with herbicide in April 2005, and burned under controlled conditions in July 2005. The planting area was manually cleared in July before planting occurred in October 2005 (Figure 22). No additional management of naturally recruiting vegetation was performed after manual clearing in 2005.

Figure 21. H1 Marsh Revegetation Site.

Figure 22. H1 Marsh Prior to Planting in 2005.

The second site, at IMC-Agrico Peace River Park, is a 161.87 ha CSA formerly owned by IMC-Agrico mining company that has been converted to a County Park. The site is located in Homeland, Florida, south of the town of Bartow. The CSA was decommissioned in 1968 and then leased for pasture until 1986. This CSA contains upland areas as well as small wetland depressions throughout. A 0.346 ha wetland area (PPW-3) located at the southeast corner the CSA was selected for revegetation (Figure 23). Upland trees and *Panicum hemitomum* (maidencane) were planted to the south and west of the marsh site as part of an unrelated study conducted by the Florida Institute of Phosphate Research (FIPR). Further south are two demonstration wetlands, PPW-1 and PPW-2. Existing vegetation at and around the revegetation site consisted of *Imperata cylindrica* (cogon grass) in upland and transition zones, and *Typha latifolia* (cattail), *Salix caroliniana* (Carolina willow), and *Ludwigia peruviana* (Peruvian primrose willow) in wet areas. Most existing vegetation within and adjacent to the revegetation site was eliminated through repetitive herbicidal treatments by FIPR staff prior to planting on August 2nd and 3rd, 2006 (Figure 24).

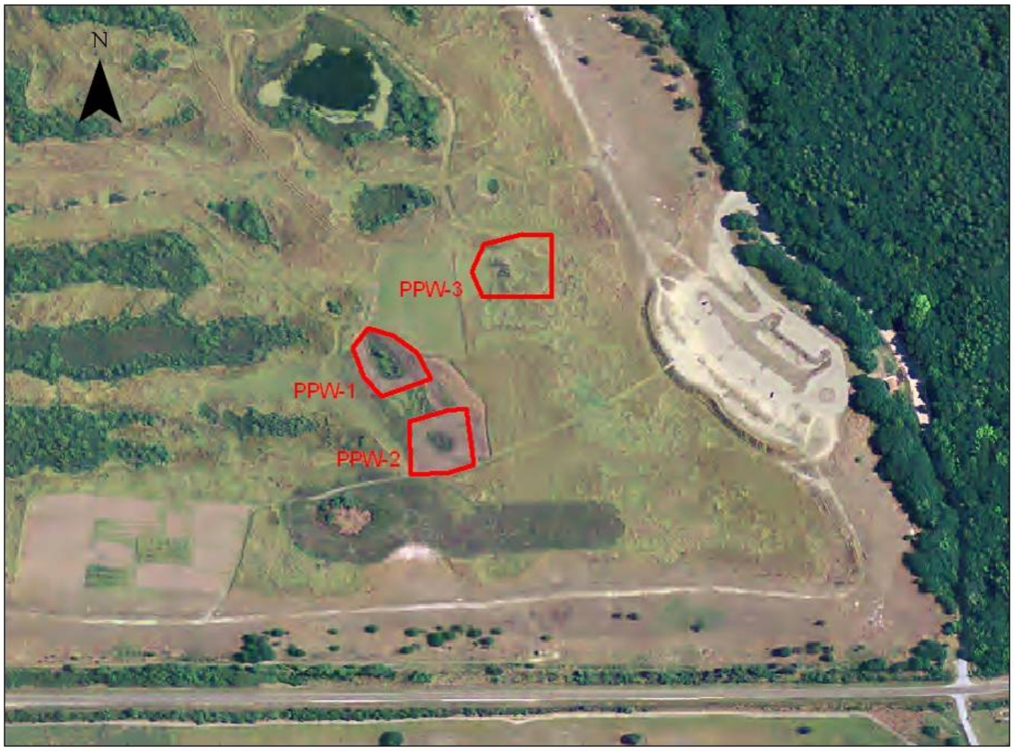


Figure 23. PPW-3 Marsh Revegetation Site and PPW-1, PPW-2 Demonstration Wetlands.

Figure 24. PPW-3 Marsh on 07/21/2006.

Wetland Tree Underplanting Sites

To evaluate growth rates and survival of tree seedlings planted under existing canopies on CSAs, three sites were chosen: one in the North Florida phosphate district and two in the Central Florida phosphate district. Aerial photographs, site visits, and personal communication with PCS, FIPR, and Mosaic staff were used to identify sites within CSA wetlands that provide (1) an existing canopy of early successional wetland tree species for underplanting of native wetland tree species and (2) an area with an appropriate slope to observe the effect of hydroperiod on survival and growth of planted seedlings.

The first, a CSA owned by the PCS Phosphate mining company, of PotashCorp, (PCS SA 10) is located in Hamilton Co. in the Northern Florida Phosphate district. Average maximum and minimum annual temperatures for Hamilton County (Jasper, Florida) are 79.8°F and 54.5°F, and average annual rainfall is 135.3 cm (SERCC 2008). This 162 ha CSA contains a 20.23 ha wetland consisting of temporarily and semi-permanently flooded, low slope features adjacent to a permanently ponded area. Water depth in this wetland is controlled by an active outfall at the northeast corner of the CSA. The underplanting site at SA 10 is located in the southern corner of the CSA (Figure 25), where a canopy of *Salix caroliniana* (Carolina willow) and *Acer rubrum* (red maple) is present at the temporarily flooded areas along the water's edge of the wetland (Figure 26). *Salix caroliniana* is pervasive throughout the semi-permanently and permanently inundated areas of the wetland. Planting occurred on July 17, 2006.

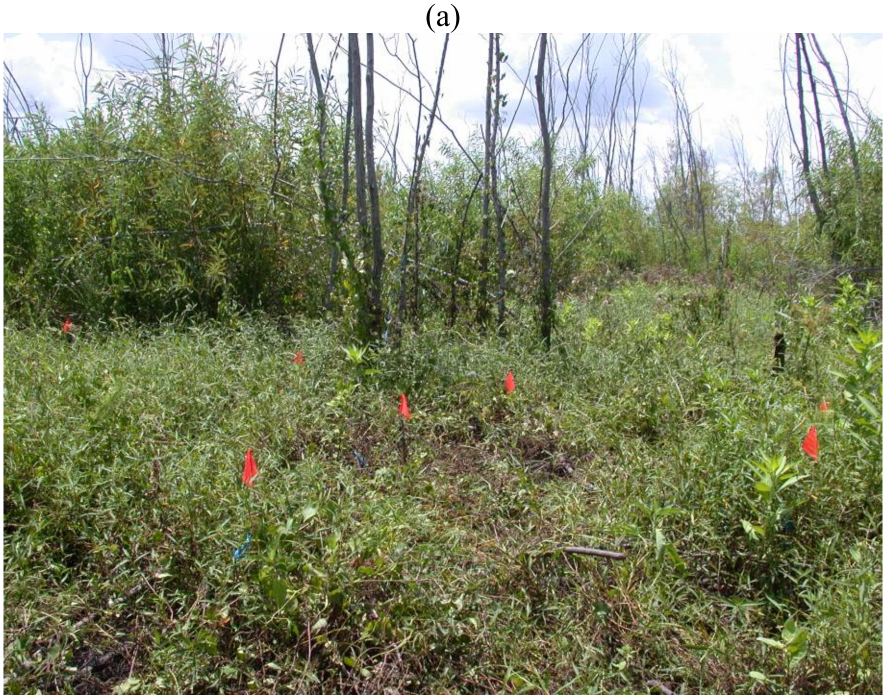
Figure 25. SA 10 Underplanting Site.

Figure 26. SA 10 Prior to Planting in 2006.

The second site, H1u, was a forested wetland area under a *Salix caroliniana* canopy on CSA H1. The planting area was located within a wetland swale present along the northern edge of the CSA (Figure 27). This site was selected to test planting under a dying canopy as much of the *Salix caroliniana* canopy had died due to burning and herbicidal treatments in 2005. Most dead trees remained standing, providing some shading (Figure 28). Planting occurred on July 11, 2006.

Figure 27. H1u Underplanting Site.

(a)



(b)

Figure 28. H1u Canopy and Understory (a) Prior to Planting and (b) in 2007.

The third site, a CSA at the Tenoroc Fish and Game Management Area (TEN-1), contained a large wetland area suitable for underplanting along the southeastern corner (Figure 29). The underplanting site (TEN-1) exists between a *Typha latifolia* marsh in the deepest part of the wetland and the CSA dike. The canopy was composed of mainly *Salix caroliniana* in wetter areas and *Sapium seriferum* (Chinese tallow), a non-native invasive tree species, along the dike slope (Figure 30). Planting occurred on August 16, 2006.

Figure 29. TEN-1 Underplanting Site.

Figure 30. TEN-1 Prior to Planting.

Demonstration Wetland Monitoring Sites

Two wetland areas, planted in September 2003 as part of earlier revegetation efforts were included in this study to evaluate more long-term wetland tree seedling survival. The wetland sites were located at the IMC-Agrico Peace River Park CSA, southwest of the PPW-3 planting (Figure 23). Both sites were approximately 0.2 ha in size and were planted with nine wetland tree species. Trees were planted from gallon pots (6" diameter). Planting design and species composition can be found in Table 7 and Figures 284 and 285. Figures 284 and 285 were provided by Kate Himel, a FIPR biologist, and included data from the 2005 monitoring. Planting was not random, with transitional wetland tree species concentrated at the edge of the wetlands, and more water tolerant, obligate wetland species toward the centers. Naturally recruiting vegetation surrounding each site was removed through mowing and herbicidal treatments over the entire period of record (2003-2007) since planting initially occurred. Trees were subject to flooding and high wind damaged from three hurricanes in 2004. Standing water occurred at the site in 2003, 2004, 2005, and part of 2006. No standing water occurred at the wetland sites in 2007 or 2008.

Table 7. Species Planted at PPW-1 and PPW-2.

Species	Common Name	PPW-1	PPW-2
		# Seedlings	# Seedlings
Quercus laurifolia	Laurel oak	137	219
Quercus virginiana	Live oak	163	193
Myrica cerifera	Wax myrtle	100	187
Carya aquatica	Water hickory	40	40
Liquidambar styraciflua	Sweet gum	40	40
Fraxinus caroliniana	Popash	39	40
Taxodium distichum	Bald cypress	51	50
Acer rubrum	Red maple	40	40
Quercus nigra	Water oak	3	4

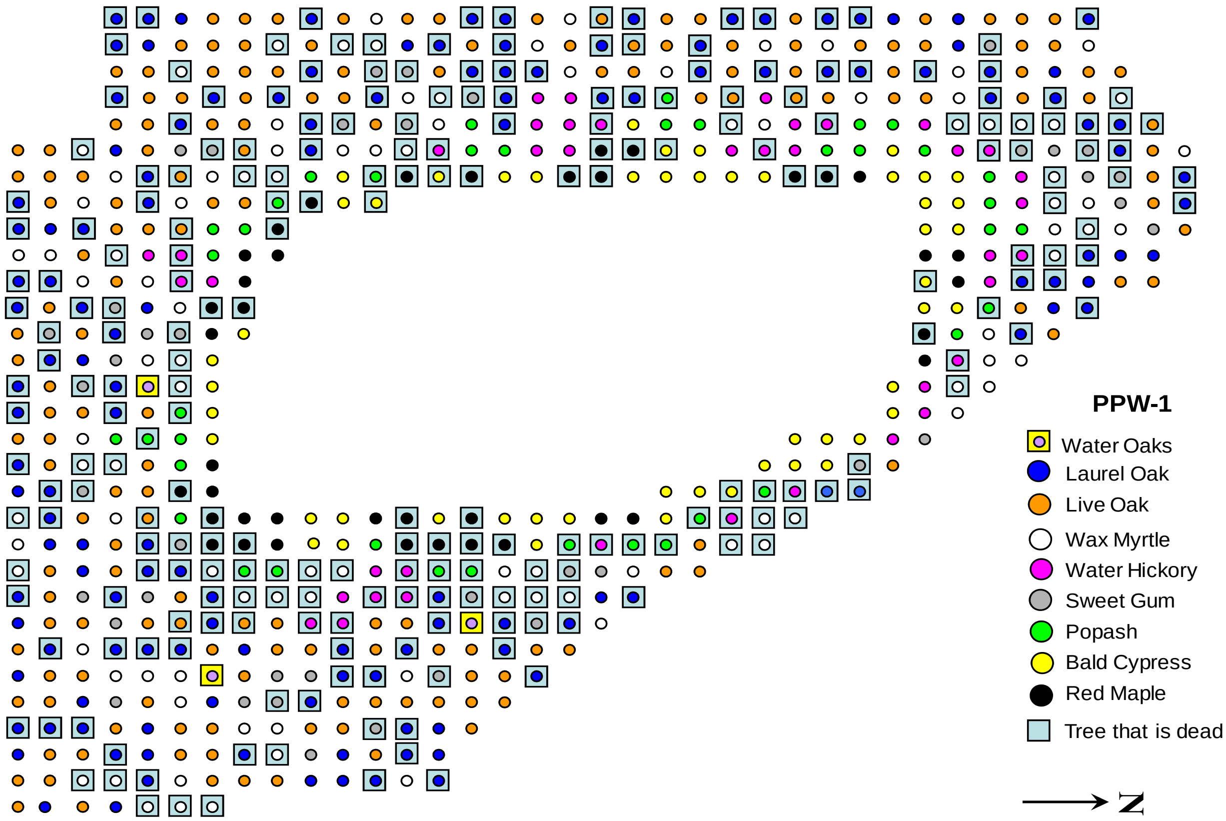


Figure 31. PPW-1 Demonstration Wetland.

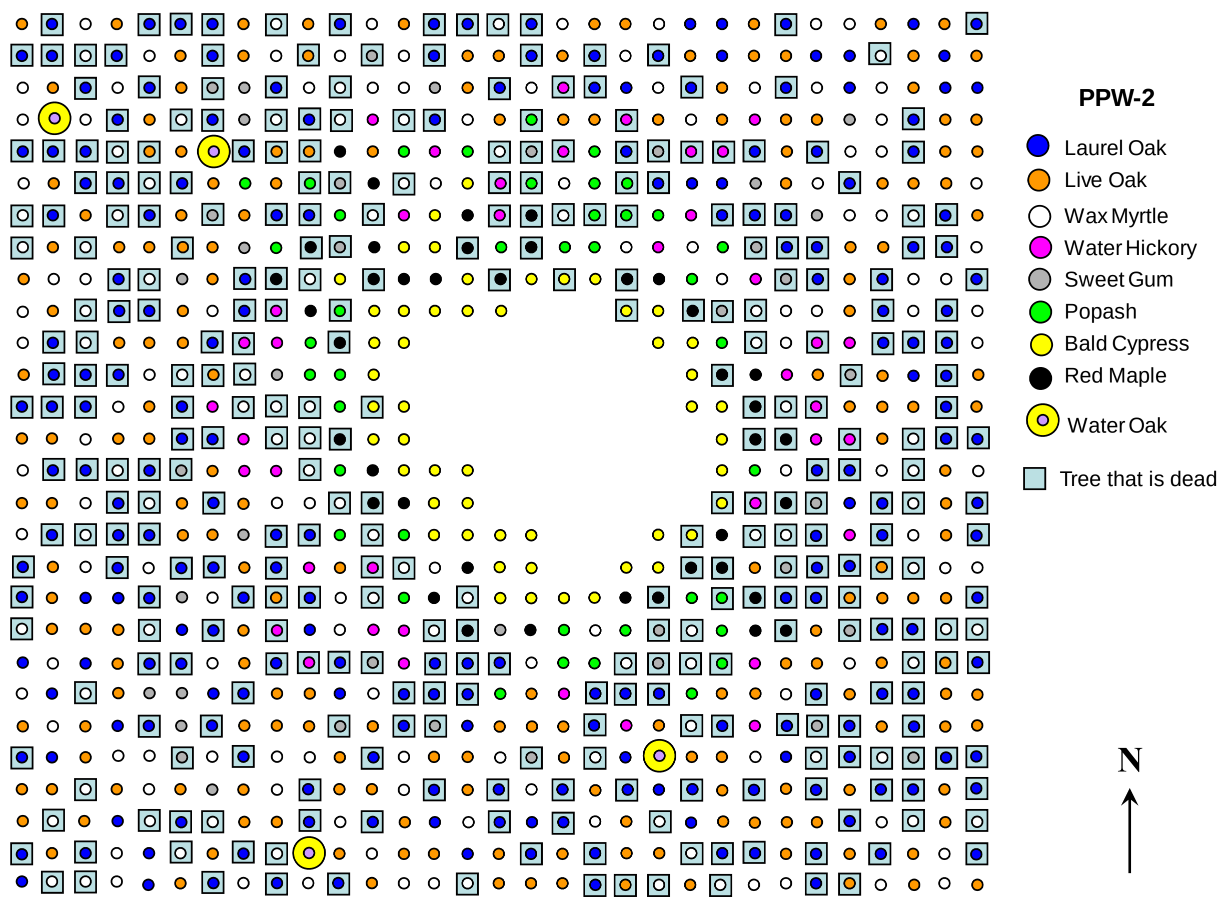


Figure 32. PPW-2 Demonstration Wetland.

Pre-Planting Topography

Planting at the marsh and underplanting sites was designed around each wetland's topography. Initially, a laser level was used to determine elevations along multiple permanently established transects at each marsh and underplanting site. ArcMap® Spatial Analyst was then used to interpolate ground elevation measurements (x-y-z) and create contour maps for each planting site. Contour maps for the H1 and PPW-3 marsh revegetation sites are shown in Figures 33 and 34. In 2006, LIDAR mapping was performed at CSAs H1, SA 10, and TEN-1 as part of another FIPR study. LIDAR is a remote sensing system used to collect topographic data with aircraft mounted lasers and yields topographic data on a one meter square basis with a vertical accuracy of less than 15 cm. Elevation data from these maps allowed the creation of more detailed contour maps for the three underplanting sites found in Figures 35, 36, and 37.

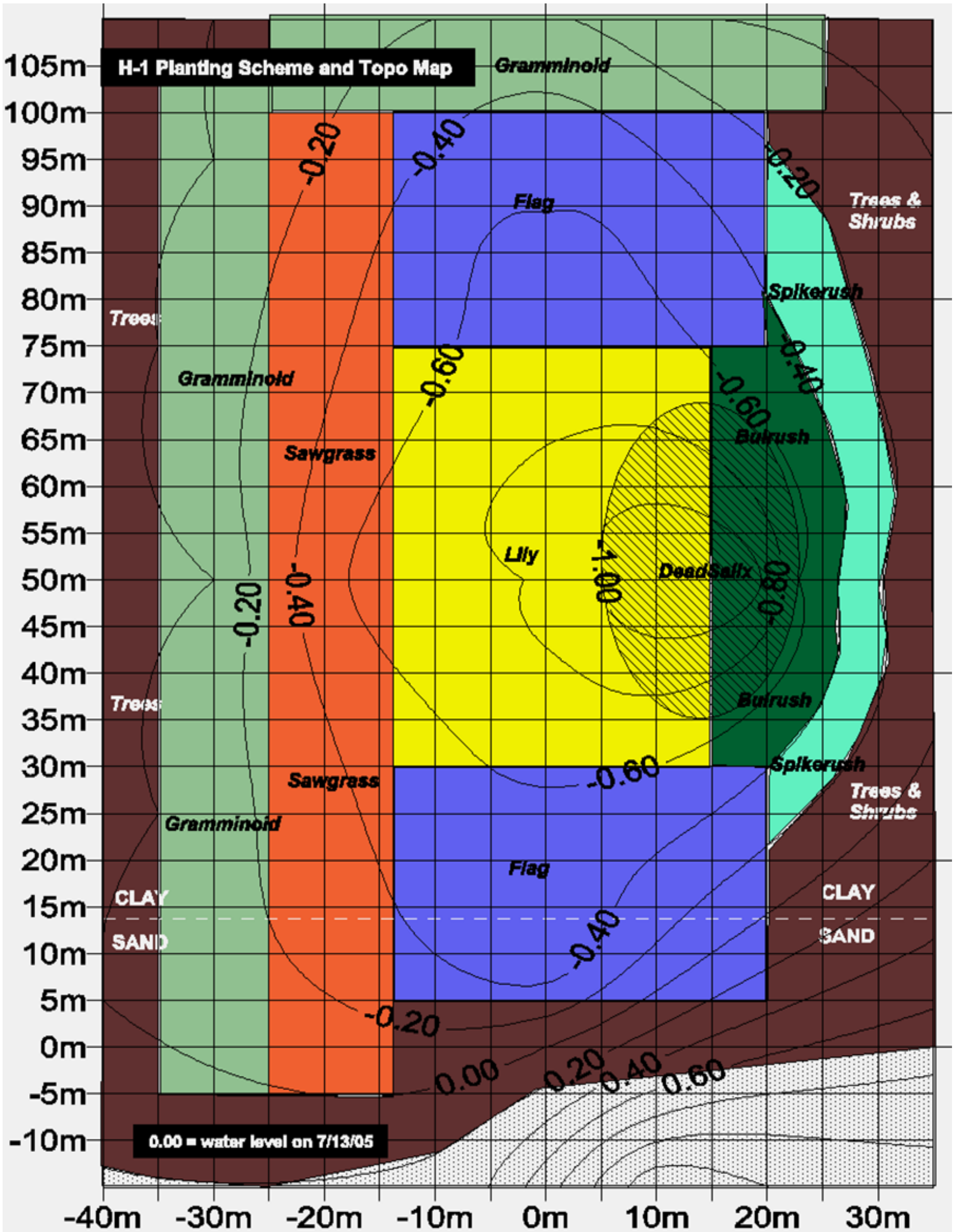


Figure 33. H1 Marsh Contour Map.

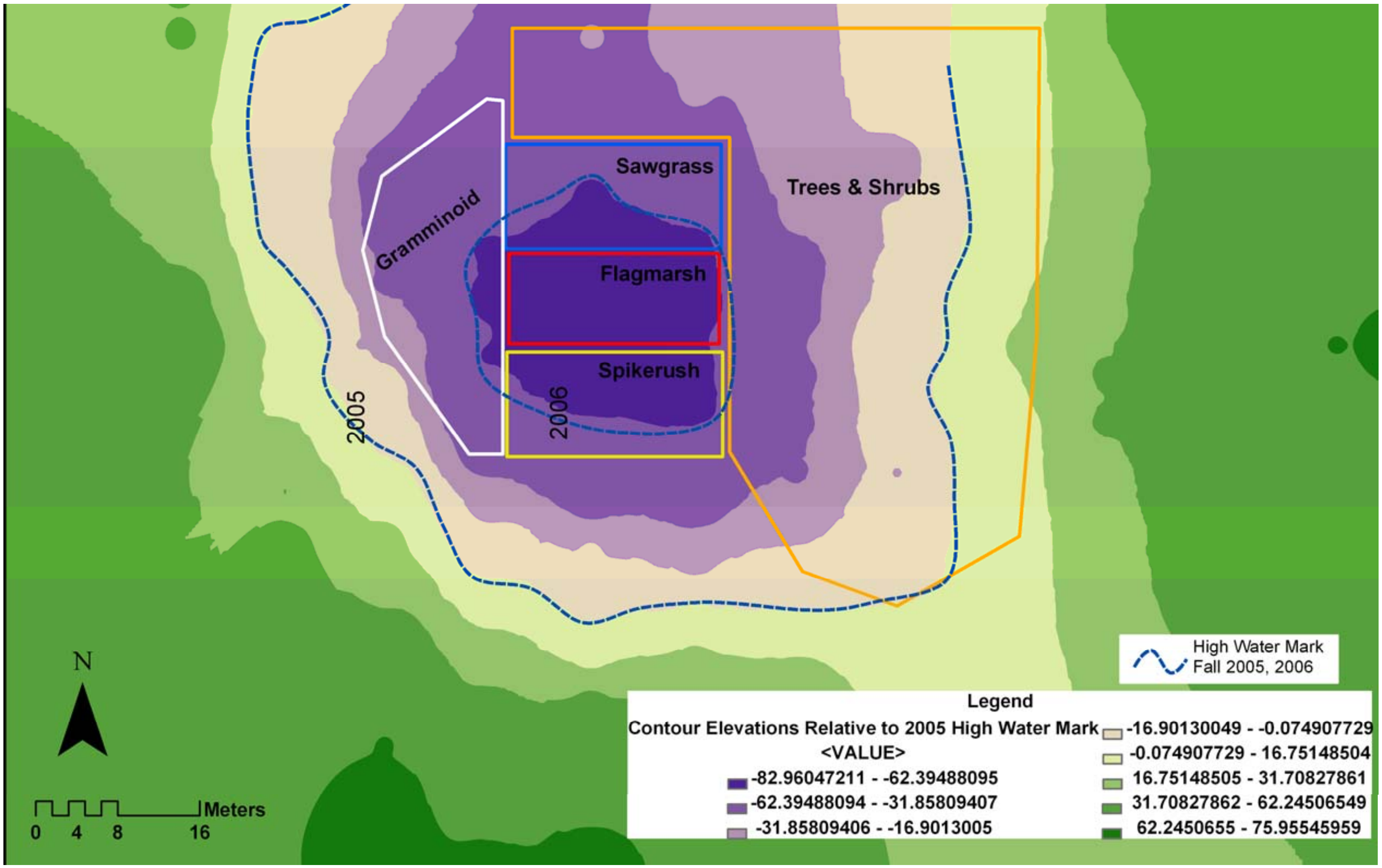


Figure 34. PPW-3 Contour Map.

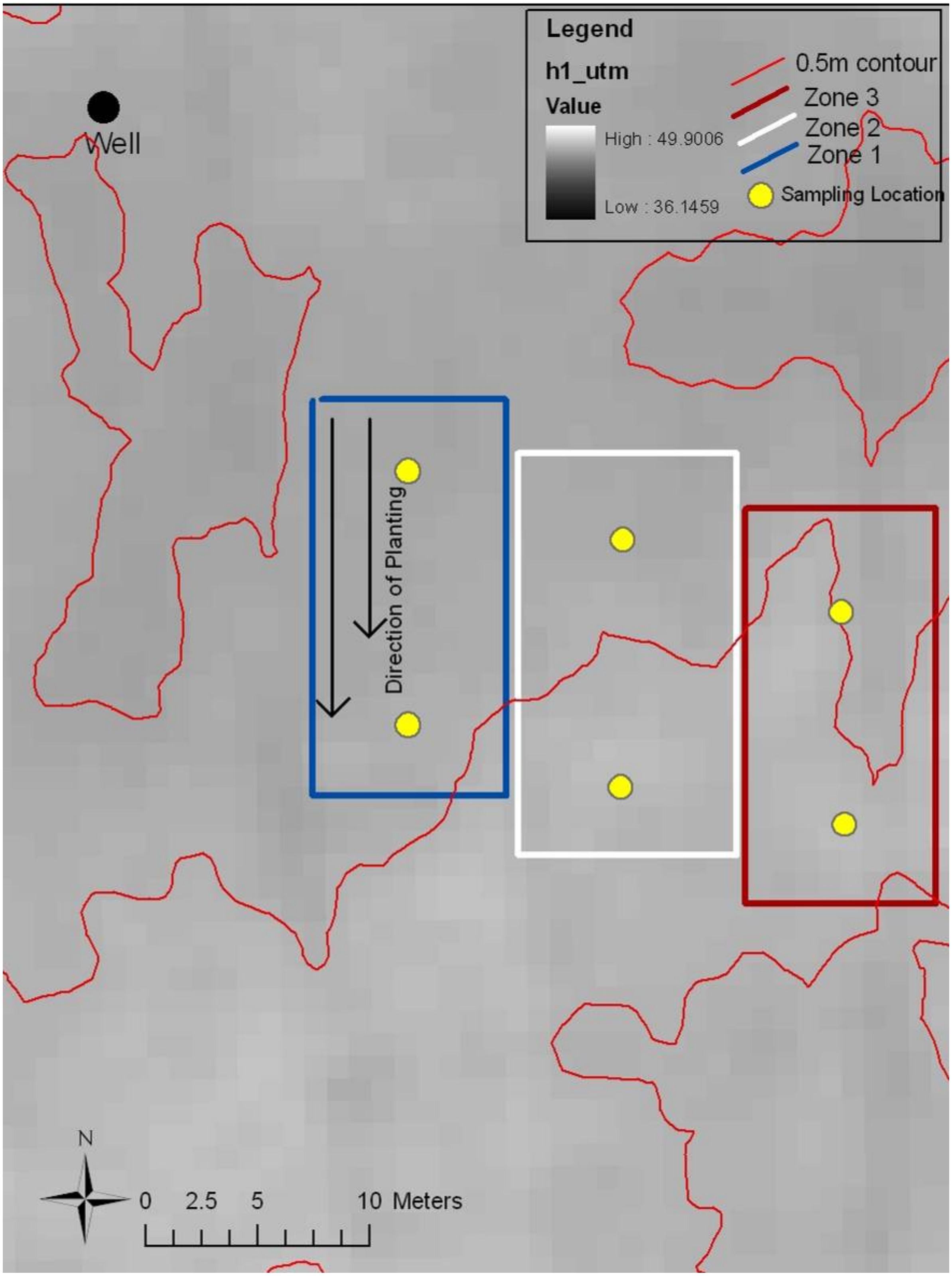


Figure 35. H1u Contour Map.

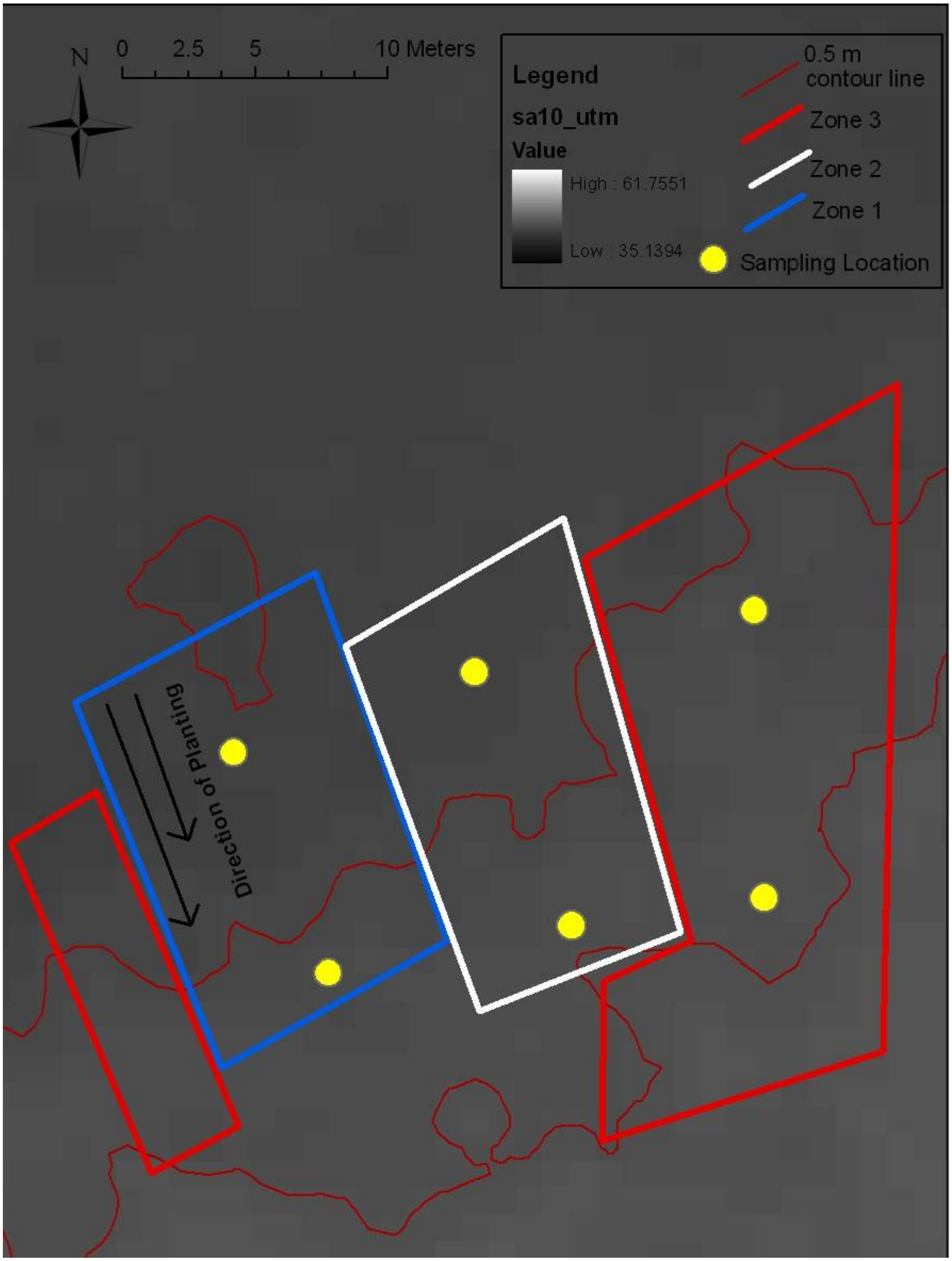


Figure 36. SA 10 Contour Map.

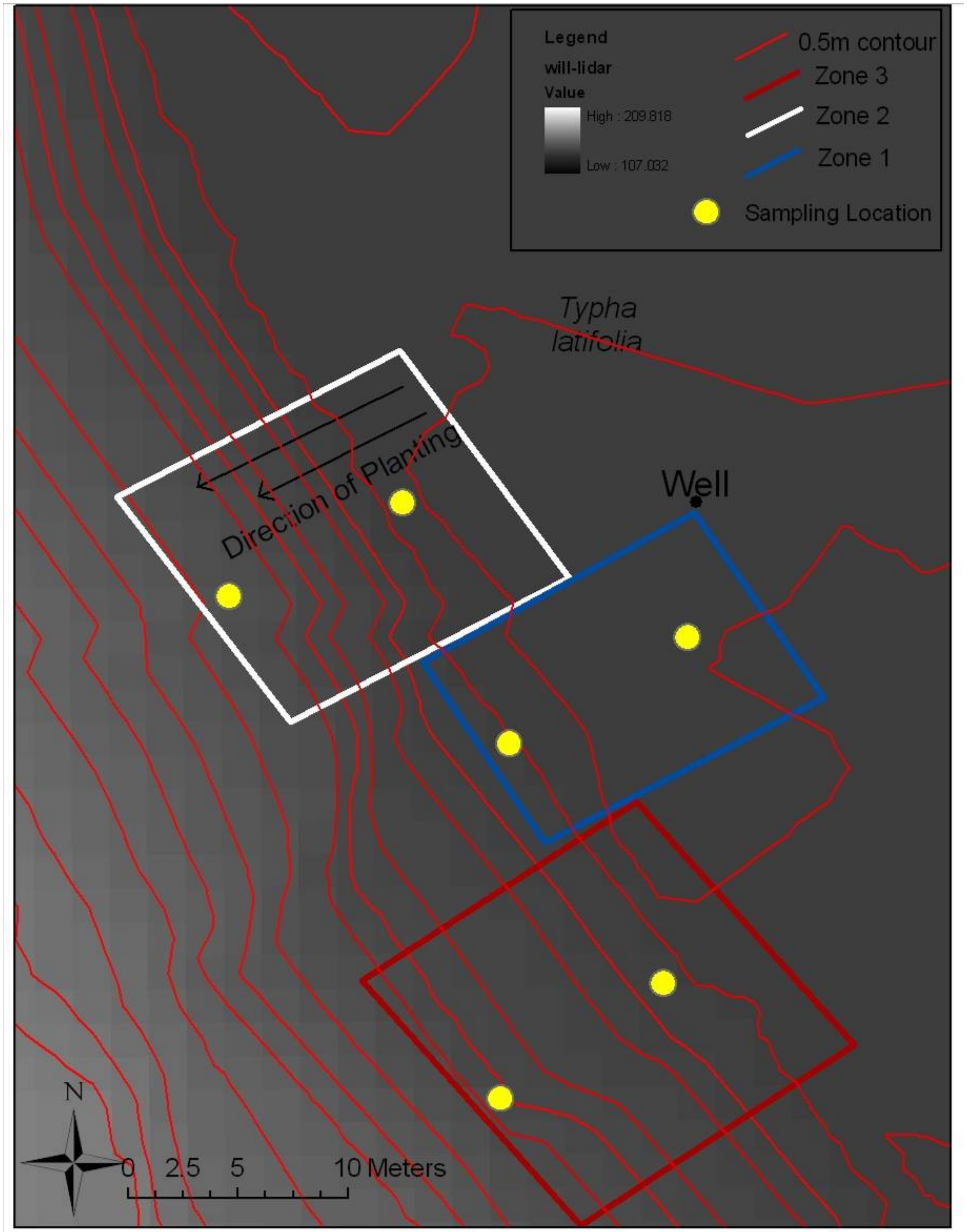


Figure 37. TEN-1 Contour Map.

Planting Design

Marsh Revegetation Sites

Planting zones, species composition, and planting densities at marsh sites are listed in Tables 8, 9, and 10. Final species planting zones and monitoring locations at H1 and PPW-3 are presented in Figures 38 and 39. Marsh revegetation sites were planted with herbaceous wetland species in the central portion of sites and wetland tree species at the periphery. In addition to the main tree planting zone at the PPW-3 periphery, *Taxodium distichum* (bald cypress) (n = 22) and *Annona glabra* (pond apple) (n = 21) were planted within the flag marsh and spike rush planting zones (Figure 39). These species naturally occur in wetter, more permanently flooded zones within wetlands (Godfrey and Wooten 1979) and were chosen, accordingly, for planting within the central portion of the marsh in addition to the larger wetland tree planting zone to compare with the species' survival in drier areas of the revegetation.

Wetland species appropriate for planting were chosen using FLDEP's wetland species status, wetland vegetation zones (DeLotelle and others 1981), Flood Tolerance Index values (Theriot 1993), the USDA plant database, and other pertinent literature and personal communication with Mosaic staff. Species' planting locations were selected using the aforementioned information in coordination with site contour maps. Some planting zones contained a single species while others were planted with a mixture of species. The lily marsh and bulrush planting zones were excluded from the PPW-3 revegetation, due to drier site conditions. One-gallon size (15-cm diameter container) tree seedlings were used for planting within the wetland tree planting zones. Seedlings were planted on 1-m centers.

Wetland Tree Underplanting Sites

Wetland tree species were grouped into three zones at each underplanting site (Tables 11, 12, 13). Groupings reflected species' water tolerance and their natural zonation within forested wetland ecosystems, from wettest (Zone 1) to driest (Zone 3). Species selection and grouping was based on FLDEP wetland species status, Flood Tolerance Index values (Theriot 1993), Water Logging Tolerance (Hook 1984), and the USDA plant database. Two parallel rows of each species were planted perpendicular to the elevation gradient, from wet to dry, at each underplanting site (Figures 35, 36, 37). One-gallon size (15cm diameter container) tree seedlings were used for planting on 1m centers.

Table 8. Species Planted at the H1 Marsh Revegetation (October 2005).

Zone of Planting	Abbreviation	Scientific Name	Common Name	FLDEP Classification[c]	Wetland Vegetation Zone[a]	# Planted	Density
Graminoid Marsh	BACM	*Bacopa caroliniana*	Lemon bacopa	OBL	Shallow marsh	250	1 plant/3.25 m^2
	JUNE	*Juncus effusus*	Soft rush	OBL	Fresh meadow	250	
	PANH	*Panicum hemotomum*	Maidencane	FACW	Shallow marsh	250	
	SPAB	*Spartina bakeri*	Spartina grass	FACW	Transition zone	250	
	MUHL	*Muhlenbergia capillaris*	Muhly grass	OBL	N/A	250	
	PELV	*Peltandra virginicum*	Green arrow-arrum	OBL	Shallow/deep marsh	250	
			TOTAL			1500	
Sawgrass Marsh	CLAJ	*Cladium jamaicense*	Saw-grass	OBL	Shallow marsh	1050	1 plant/0.718 m^2
			TOTAL			1050	
Scirpus Marsh	SCIC	*Scirpus californicus*	Giant bulrush	OBL	Shallow/deep marsh	200	1 plant/0.69 m^2
			TOTAL			200	
Spike Rush Marsh	ELEC	*Eleocharis cellulosa*	Club-rush	OBL	Deep marsh	375	1 plant/0.406 m^2
			TOTAL			375	
Flag Marsh	PONC	*Pontedaria cordata*	Pickerelweed	OBL	Shallow/deep marsh	600	1 plant/2.37 m^2
	SAGL	*Sagittaria lancifolia*	Bulltongue arrowhead	OBL	Shallow marsh	600	
	THAG	*Thalia geniculata*	Bent-alligator flag	OBL	Deep marsh	600	
			TOTAL			1800	
				FLDEP Classification[c]	Flood Tolerance Index[b]		
Trees and Shrubs	FRAC	*Fraxinus caroliniana*	Pop ash	OBL	N/A	200	1 plant/0.648 m^2
	NYSB	*Nyssa biflora*	Swamp tupelo	OBL	3.04	200	
	PERP	*Persea palustris*	Swamp bay	OBL	N/A	200	
	TAXD	*Taxodium distichum*	Bald cypress	OBL	2.97	200	
	GLEA	*Gleditsia aquatica*	Water locust	OBL	3.5	200	
	CEPO	*Cephalanthus occidentalis*	Buttonbush	OBL	2.83	200	
	ITEV	*Itea virginica*	Virginia willow	OBL	2.83	200	
	STYA	*Styrax americana*	American snowbell	OBL	3.41	200	
	HYPF	*Hypericum fasciculatum*	St. John's wort	OBL	N/A	200	
			TOTAL			1800	

[a]Vegetation Zones (DeLotelle and others 1981)
Transition zone: FACU-FACW. Duration of flooding: < 60 days with water levels less than 10 cm.
Shallow marsh zone: Depths to 100 cm from 60 to 365 days.
Deep marsh zone: Water depths to 130 cm for >6 months.
[b]FTI = Flood Tolerance Index (Theriot 1993).
[c]Wetland Status, Department of Environmental Protection (DEP): Source – Delineation of the Landward Extent of Wetland and Surface Waters, Chapter 62-340 Florida Administrative Code. 1994.

Table 9. Supplemental Planting at the H1 Revegetation (August 2006).

Zone of Planting	Abbreviation	Scientific Name	Common Name	FLDEP Classification[c]	Wetland Vegetation Zone[a]	# Planted	Density
Lily Marsh	NUPL	*Nuphar lutea*	Spatter-dock	OBL	Deep marsh	450	1 plant/1 m^2
	NYMO	*Nymphaea odorata*	Fragrant water lily	OBL	Shallow/deep marsh	450	
			TOTAL			900	
				FLDEP Classification[c]	Flood Tolerance Index[b]		
Trees	ANNG	*Annona glabra*	Pond apple	OBL	N/A	25	1 tree/1 m^2
			TOTAL			25	

[a]Vegetation Zones (DeLotelle and others 1981)
Transition zone: FACU-FACW. Duration of flooding: < 60 days with water levels less than 10 cm.
Shallow marsh zone: Depths to 100 cm from 60 to 365 days.
Deep marsh zone: Water depths to 130 cm for >6 months.
[b]FTI = Flood Tolerance Index (Theriot 1993).
[c]Wetland Status, Department of Environmental Protection (DEP): Source – Delineation of the Landward Extent of Wetland and Surface Waters, Chapter 62-340 Florida Administrative Code. 1994.

Table 10. Species Planted at the PPW-3 Revegetation (August 2006).

Zone of Planting	Abbreviation	Scientific Name	Common Name	FLDEP Classification[c]	Wetland Vegetation Zone[a]	# Planted	Density
Graminoid Marsh	BACM	*Bacopa caroliniana*	Lemon bacopa	OBL	Shallow marsh	30	1 plant/2.0 m^2
	SPAB	*Spartina bakeri*	Spartina grass	FACW	Transition zone	30	
	MUHC	*Muhlenbergia capillaris*	Muhly grass	OBL	N/A	30	
	JUNE	*Juncus effusus*	Soft rush	OBL	Fresh meadow	30	
			TOTAL			120	
Sawgrass Marsh	CLAJ	*Cladium jamaicense*	Saw-grass	OBL	Shallow marsh	280	1 plant/~0.718 m^2
			TOTAL			280	
Spike Rush Marsh	ELEC	*Eleocharis cellulosa*	Club-rush	OBL	Deep marsh	400	1 plant/0.5 m^2
			TOTAL			400	
Flag Marsh	PONC	*Pontedaria cordata*	Pickerelweed	OBL	Shallow/deep marsh	30	1 plant/1.5 m^2
	SAGL	*Sagittaria lancifolia*	Bulltongue arrowhead	OBL	Shallow marsh	30	
	THAG	*Thalia geniculata*	Bent-alligator flag	OBL	Deep marsh	30	
			TOTAL			90	
				FLDEP Classification[c]	Flood Tolerance Index[b]		
Trees and Shrubs	ANNG	*Annona glabra*	Pond apple	OBL	N/A	200	1 tree/1 m^2
	FRAC	*Fraxinus caroliniana*	Pop ash	OBL	N/A	200	
	NYSS	*Nyssa biflora*	Swamp tupelo	OBL	3.04	200	
	PERP	*Persea palustris*	Sweet bay	OBL	N/A	200	
	TAXD	*Taxodium distichum*	Bald cypress	OBL	2.97	200	
	GLEA	*Gleditsia aquatica*	Water locust	OBL	3.5	200	
	CEPO	*Cephalanthus occidentalis*	Buttonbush	OBL	2.83	200	
	ITEV	*Itea virginica*	Virginia willow	OBL	2.83	200	
	STYA	*Styrax americana*	American snowbell	OBL	3.41	200	
	HYPF	*Hypericum fasciculatum*	St. John's wort	OBL	N/A	200	
			TOTAL			2000	

[a]Vegetation Zones (DeLotelle and others 1981)
Transition zone: FACU-FACW. Duration of flooding: < 60 days with water levels less than 10 cm.
Shallow marsh zone: Depths to 100 cm from 60 to 365 days.
Deep marsh zone: Water depths to 130 cm for >6 months.
[b]FTI = Flood Tolerance Index (Theriot 1993).
[c]Wetland Status, Department of Environmental Protection (DEP): Source – Delineation of the Landward Extent of Wetland and Surface Waters, Chapter 62-340 Florida Administrative Code. 1994.

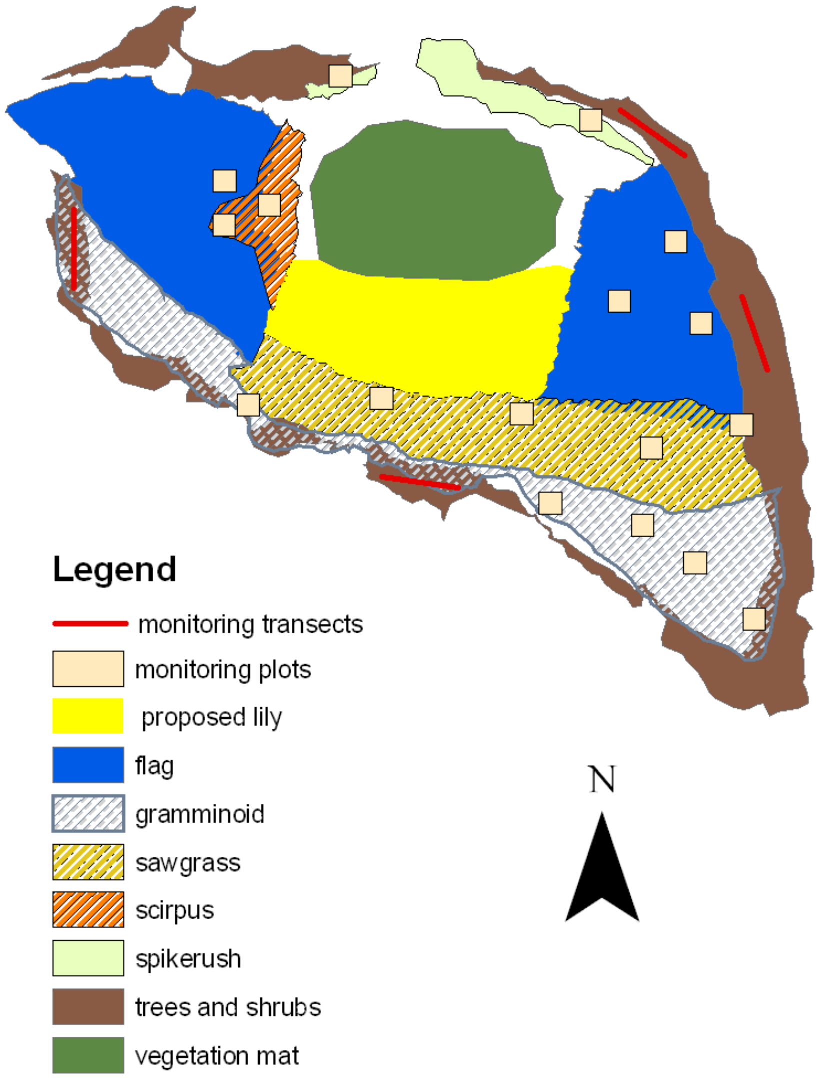


Figure 38. Planting Design and Monitoring Locations at the H1 Marsh.

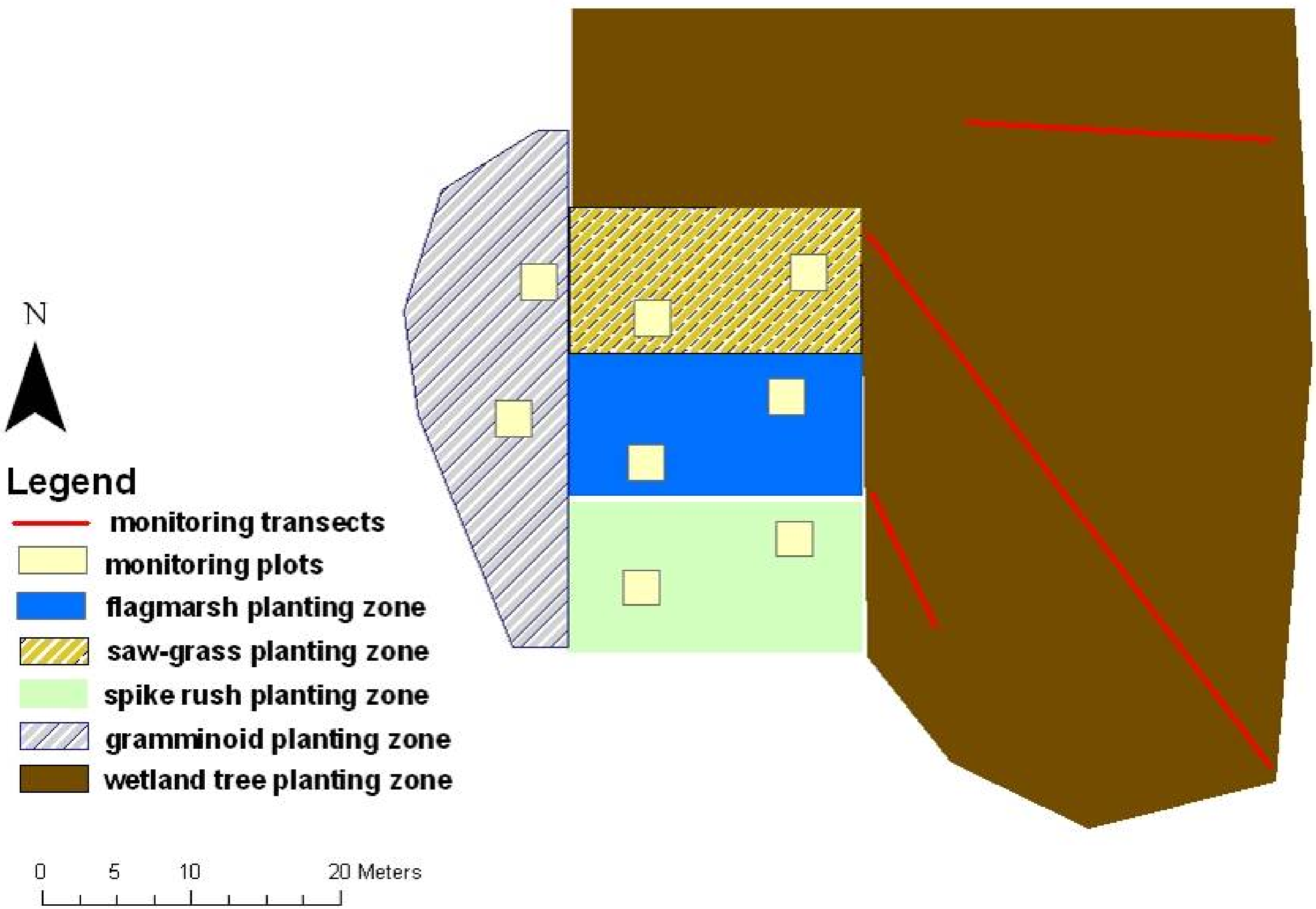


Figure 39. Planting Design and Monitoring Locations at the PPW-3 Marsh.

Table 11. Planted Species at SA 10 (July 2006).

Zone of Planting	Abbreviation	Scientific Name	Common Name	# Planted
1	ITEV	*Itea virginica*	Virginia willow	25
1	TAXD	*Taxodium distichum*	Bald cypress	30
1	LIQS	*Liquidambar styraciflua*	Sweetgum	25
1	TAXA	*Taxodium distichum var. ascendens*	Pond cypress	21
1	FRAP	*Fraxinus pennsylvanica*	Green ash	18
1	CEPO	*Cephalanthus occidentalis*	Buttonbush	22
1	NYSA	*Nyssa aquatica*	Water tupelo	3
2	ULMA	*Ulmus Americana var. floridana*	American elm	24
2	CARA	*Carya aquatica*	Water hickory	22
2	BETN	*Betula nigra*	River birch	24
2	QUEL	*Quercus lyrata*	Overcup oak	22
2	NYSS	*Nyssa sylvatica var. biflora*	Swamp tupelo	25
3	PLAO	*Plantanus occidentalis*	American sycamore	23
3	LIRT	*Liriodendron tulipifera*	Tulip poplar	23
3	CELL	*Celtis laevigata*	Hackberry	25
3	MAGV	*Magnolia virginiana*	Swamp bay	25
3	CORF	*Cornus foemina*	Swamp dogwood	25
3	QUEM	*Quercus michauxii*	Swamp chestnut oak	25
3	ILEC	*Ilex cassine*	Dahoon holly	22
3	NYSA	*Nyssa aquatica*	Water tupelo	25
3	QUEN	*Quercus nigra*	Water oak	25
			TOTAL	479

	FTI[a]	WLT[b]	DEP[c]
Zone 1	2-3.25	Most	OBL
Zone 2	3.25-4.5	Moderate	OBL/FACW
Zone 3	4.5->	Weak	FACW/FAC/FACU

[a]FTI = Flood Tolerance Index (Theriot 1993).
[b]WLT = Water Logging Tolerance Rating (Hook 1984).
[c]Wetland Status, Department of Environmental Protection (DEP); Source—Delineation of the Landward Extent of Wetlands and Surface Waters, Chapter 62-340, Florida Administrative Code. 1994.

Table 12. Planted Species at H1u (July 2006).

Zone of Planting	Abbreviation	Scientific Name	Common Name	# Planted
1	TAXD	*Taxodium distichum*	Bald cypress	24
1	NYSS	*Nyssa sylvatica var. biflora*	Swamp tupelo	24
1	FRAC	*Fraxinus caroliniana*	Popash	28
2	CELL	*Celtis laevigata*	Hackberry	25
2	ULMA	*Ulmus Americana var. floridana*	American elm	25
2	ILEC	*Ilex cassine*	Dahoon holly	25
3	CARA	*Carya aquatica*	Water hickory	22
3	MAGV	*Magnolia virginiana*	Swamp bay	24
3	LIQS	*Liquidambar styraciflua*	Sweetgum	19
3	QUEN	*Quercus nigra*	Water oak	24
3	SABP	*Sabal minor*	Dwarf palmetto	26
			TOTAL	266

	FTI[a]	WLT[b]	DEP[c]
Zone 1	2-3.25	Most	OBL
Zone 2	3.25-4.5	Moderate	OBL/FACW
Zone 3	4.5->	Weak	FACW/FAC/FACU

[a]FTI = Flood Tolerance Index (Theriot 1993).

[b]WLT = Water Logging Tolerance Rating (Hook 1984).

[c]Wetland Status, Department of Environmental Protection (DEP); Source—Delineation of the Landward Extent of Wetlands and Surface Waters, Chapter 62-340, Florida Administrative Code. 1994.

Table 13. Planted Species at TEN-1 (August 2006).

Zone of Planting	Abbreviation	Scientific Name	Common Name	# Planted
1	ITEV	*Itea virginica*	Virginia willow	25
1	TAXD	*Taxodium distichum*	Bald cypress	24
1	NYSS	*Nyssa sylvatica var. biflora*	Swamp tupelo	25
1	FRAC	*Fraxinus caroliniana*	Popash	24
1	TAXA	*Taxodium distichum var. ascendens*	Pond cypress	22
2	CELL	*Celtis laevigata*	Hackberry	24
2	ULMA	*Ulmus Americana var. floridana*	American elm	25
2	QUEL	*Quercus lyrata*	Overcup oak	25
2	CARA	*Carya aquatica*	Water hickory	24
2	ILEC	*Ilex cassine*	Dahoon holly	23
2	BETN	*Betula nigra*	River birch	26
3	QUEN	*Quercus nigra*	Water oak	27
3	SABP	*Sabal palmetto*	Cabbage palm	26
3	CORF	*Cornus foemina*	Swamp dogwood	26
3	QUEM	*Quercus michauxii*	Swamp chestnut oak	27
3	LIRT	*Liriodendron tulipifera*	Tulip poplar	26
3	LIQD	*Liquidambar styraciflua*	Sweetgum	25
3	MAGV	*Magnolia virginiana*	Swamp bay	23
			TOTAL	447

	FTI[a]	WLT[b]	DEP[c]
Zone 1	2-3.25	Most	OBL
Zone 2	3.25-4.5	Moderate	OBL/FACW
Zone 3	4.5->	Weak	FACW/FAC/FACU

[a]FTI = Flood Tolerance Index (Theriot 1993).
[b]WLT = Water Logging Tolerance Rating (Hook 1984).
[c]Wetland Status, Department of Environmental Protection (DEP); Source—Delineation of the Landward Extent of Wetlands and Surface Waters, Chapter 62-340, Florida Administrative Code. 1994.

Field Data Collection

Hydrologic Monitoring

A shallow piezometer was established in the deepest portion of each marsh and underplanting wetland site with Solinst® mini LT 15’ electronic water level recorders and Solinst® Barologgers correcting for barometric pressure. The location of piezometers were recorded using a Trimble® GPS unit and Topcon® RL 20 laser level, which allowed for determination of water depths and wetland hydroperiod at monitoring plots, using recorded plot locations and topographic data. Precipitation gauges were also installed at each site and recorded daily cumulative precipitation.

Revegetation Monitoring

Monitoring methods and sampling plot design for planted and volunteer vegetation at marsh revegetation sites (H1, PPW-3), seedling underplanting sites (H1, SA 10, TEN-1), and continued monitoring site (PPW-1, PPW-2) are presented. As well, the determination of hydrologic regime at monitoring areas for marsh and underplanting sites is described. Finally, methods are given for the characterization of revegetation sites in terms of their soils and pre-planting floral composition.

Marsh Revegetation Sites. In order to evaluate the success of herbaceous species plantings at H1 and PPW-3, species frequency was monitored using multiple 3 m × 3 m (9 m^2) permanent monitoring plots established along the hydrologic gradient within each planting zone (Figures 38, 39). Monitoring plots were permanently established with rebar anchored PVC piping at the plots’ NE and SW corners. The presence or absence of species, both planted and volunteer, was documented in nine, 1 m^2 quadrats within each plot immediately after planting occurred and after each subsequent growing season. Nested subquadrats within quadrats were used to observe individual plant growth over the period of record. Trees planted randomly at the periphery of marsh plantings were monitored for survival and growth using belt transects of sufficient length to sample planted populations. The entire population of seedlings planted in the central portion of PPW-3 marsh was evaluated for survival and growth.

Each monitoring plot and transect was documented using a Trimble GPS® unit, with horizontal sub-meter accuracy. GPS points were taken at the northwest and southeast corners of each monitoring plot and along the length and width of belted transects. The points were then overlain onto previously generated topographic maps using ArcMap to determine the corresponding elevations at each monitoring location. Elevation values were then used to generate hydrologic characteristics for each plot or transect by adjusting the surface and groundwater data at the piezometer for the elevation difference between the sites’ piezometer and monitoring location. At the H1 marsh, belted transects extend parallel to site contours and so elevation varied within belts.

Minimum and maximum elevations at belted transects were used to calculate the range of hydrologic conditions experienced within the belt. At PPW-3, the peripheral tree planting zone is much wider than at the H1 marsh, and so long belted transects could be established perpendicular to the elevation gradient, rather than parallel. Elevation for each meter of distance along each transect at PPW-3 was calculated by generating a slope for each transect.

Seedling Underplanting Sites. All trees planted at each site were monitored for survival and growth. In order to effectively monitor areas where cleared groundcover will overtake the height of the seedlings planted, permanent monitoring transects, with rebar and PVC, were established and the known location of each seedling were mapped. Growth was monitored using height, and was measured to the top of the main stem of each seedling, unless splitting occurred and then the tallest main stem was used. GPS points along each transect (every 1 m) were overlain on topographic maps to relate elevations within each planting zone to the elevation of the surface water well at each site. Hydrologic characteristics along transects were then calculated by adjusting the surface and groundwater data at the piezometer for the differences in elevation.

PPW-1 and PPW-2

Both populations of trees planted at PPW-1 and PPW-2 were monitored for survival at the end of the 2007 growing season using previously generated planting maps. To account for possible regrowth, the location of all seedlings recorded as absent in 2005 were monitored.

Site Characterization

Cores of the top 20 cm of soil were collected using a 7.6 cm diameter auger at all monitoring locations at marsh sites and each sampling location at understory sites shown in Figures 35, 36, and 37. Each sample core was placed on ice and returned to Phelps Laboratory for analysis. At seedling underplanting sites, canopy cover was captured at each sampling location with hemispherical photographs taken 50 cm above the ground surface, with a Nikon® digital camera and 180 degree "fish-eye" lens. The camera was secured to the end of the tripod and the picture was zoomed to 100%, with the camera oriented in the same direction at each site. Species composition of naturally recruiting understory vegetation was sampled using a 1 m^2 quadrat at each sampling location at seedling underplanting sites. Species that could not be identified in the field, were stored in a cooler and returned to the lab for correct identification.

Data Analysis

Methods for analysis of collected data are described below. Procedures for analysis of canopy photos were described earlier (see "Data Analysis" under "Evaluation of B. Rushton Field Trials," page 58).

Wetland Hydroperiod

Once elevation of each plot or transect relative to the surface water well was determined, water levels at each location of interest was calculated on a daily basis. Annual and growing season average water levels, percent (%) inundation, flooding frequency, and inundation of the herbaceous vegetation and seedling root zones were calculated for each 1 m^2 of interest.

Herbaceous Frequency

Species frequency within each monitoring quadrat was calculated for every planted and volunteer species at H1 and PPW-3. Frequency is the percentage of quadrats in which a species occurs at least once, and results in a calculated value between 0 and 1, or 0%-100%. Frequency within each plot was the number of quadrats a species occurred over the total number of quadrats per plot. Annual overall frequency for a species was calculated by combining frequency data from all monitoring plots within the planting zone where it occurred.

Seedling Survival and Growth

Percent survival of tree seedlings sampled at marsh sites and understory sites was calculated, as was the change in mean seedling height for each species over the period of record of each site. At each site, growth was examined over the planting zone's hydrologic gradient using linear regression and correlation. For each species, two tailed T-tests were used to compare mean seedling height between underplanting sites.

Soil Particle Size

Each soil sample was analyzed for percent composition (% clay, % silt, % sand). Samples were processed using the hydrometer method through WATERS Agricultural Laboratories in Camilla, GA.

RESULTS

RESEARCH SCHEDULE

The research schedule is summarized in Figure 40. The project was organized into thirteen broad tasks. The following tasks were completed:

- Selection of sites for descriptive transects, detailed hydrologic evaluation, and field trials
- Initial characterization of abiotic and biotic properties of CSA wetland areas using descriptive transects crossing more than 12,000 m^2 of 17 CSAs.
- Monitoring of a number of hydrologic parameters on 8 CSAs.
- Development of detailed water budget models and hydrologic temporal models of 7 CSAs
- Spatial rendering of water budget models with maps of inundation periods 6 CSAs
- Evaluation of ecohydrologic relationships between CSA vegetation and soils
- Evaluation of planted trees and ecosystem development in 37 plots on CSAs planted by Betty Rushton in the mid 1980s
- Field trials with planting of more than 11,700 trees and herbaceous plants on four CSAs
- Evaluation of the success of field trials
- Synthesis of all relevant data and information gleaned from the research into guidelines for wetlands on CSAs
- Organization and leadership of a workshop with representatives from the WOC project and FIPR to present guidelines for wetlands on clay and discuss best practices for wetland design including CSA construction, hydrologic modeling, and locating, planting, and monitoring wetlands
- Compilation of annual project reports
- Compilation of all project work into the final report

CHARACTERIZATION OF NATURALLY OCCURRING WETLANDS ON CSAs

Plant Community Evaluation

Transects where vegetation data were collected, totaling 45, with an average length of approximately 60 m, were established on 17 sites for a total length of 2,748 m. Species occurrences in terms of stem counts and total basal area are summarized for trees and shrubs in Tables 14 and 15. Table 16 lists species occurrences at the ground level in terms of frequency and mean cover class, with the latter as a range from 1 to 5 with 5 referring to over 75% coverage. Total basal area of trees and shrubs for 10 m intervals along each transect were calculated and are shown on part (b) in even-numbered figures of Appendix A.

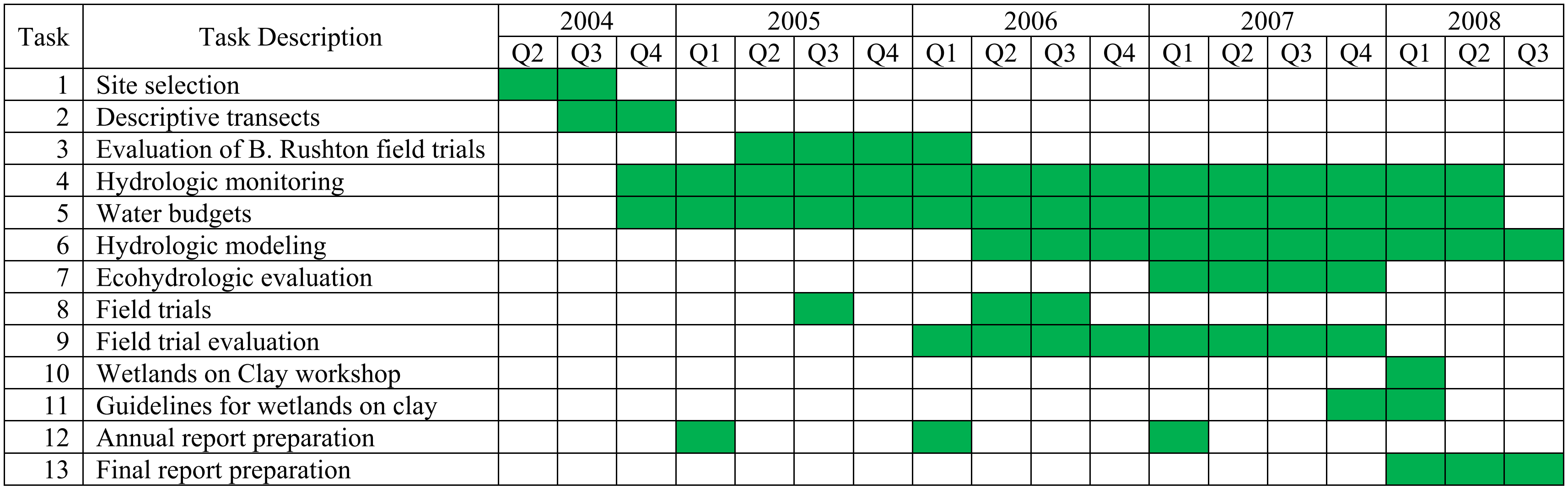


Figure 40. Wetlands on Clay Research Schedule.

Although marsh systems (no presence of a canopy layer) are present on parts of some CSAs, forested systems occur more frequently on CSAs, being present on 15 of 17 sites surveyed. *Salix caroliniana* accounted for nearly three times as many stems and twice as much basal area as all other canopy species combined (Table 14). It was also dominant in the shrub layer on the portions of transects further into the wetland. More *Ludwigia peruviana* (primrose willow) stems were present in the shrub layer than any other, but *Myrica cerifera* (wax myrtle) had thicker stems and occupied greater total basal area in this strata. But the canopy and shrub layer of naturally occurring CSA wetlands had relatively low species richness and evenness. The highest diversity occurred in the lowest layer, where the most prominent species were small floating aquatic plants, *Lemna minor*, *Spirodela polyrhiza*, and *Pistia stratiotes*, seedlings of the aforementioned woody plants, and *Typha spp*. *Imperata cylindrica* frequently occurred at the ground level within ecotone areas and often in high density, out competing other species in the niche it inhabited.

Table 14. Total Stems and Basal Area of Canopy Layer Species.

Species	Total Stems	Total Basal Area (m^2)
Salix caroliniana	1733	10.1
Taxodium distichum	270	3.6
Sapium sebiferum	174	0.7
Nyssa aquatica	53	0.5
Acer rubrum	87	0.3
Fraxinus caroliniana	152	0.3
Quercus laurifolia	17	0.1
Carya aquatica	7	0.1

Table 15. Total Stems and Basal Area of Shrub Layer Species.

Species	Total Stems	Total Basal Area (m^2)
Myrica cerifera	191	2.6
Salix caroliniana	698	1.8
Ludwigia peruviana	1104	0.8
Baccharis halimifolia	351	0.3
Sambucus canadensis	219	0.2
Schinus terebinthifolius	8	0.04
Cephalanthus occidentalis	24	0.01

Table 16. Dominant Species in Ground Layer.

Species	Frequency	Mean Cover Class
Lemna minor/Spirodela polyrhiza[1]	136	3.1
Ludwigia peruviana	67	1.2
Pistia stratiotes	42	2.4
Mikania scandens	39	1.3
Salix caroliniana	35	1.3
Typha spp.[2]	33	1.8
Imperata cylindrica	31	2.9
Eupatorium capillifolium	30	1.1
Polygonum hydropiperoides	28	1.4
Hydrocotyle umbellata	28	1.3
Sambucus canadensis	23	1.0
Cyperus odoratus	22	1.6
Pluchea odorata	19	1.3
Phyla nodiflora	16	2.4
Scirpus californicus	16	1.6
Alternanthera philoxeroides	16	1.1
Salvinia minima	15	2.9
Ptilimnium capillaceum	13	1.6
Ambrosia artemisiifolia	13	1.2
Baccharis halimifolia	12	1.2
Thelypteris hispidula	11	1.5
Limnobium spongia	10	1.4

[1]Occurring together nearly 100% of the time.
[2]Two closely related species of cattail with overlapping ranges, *latifolia* and *domenegensis*, are both included.

In comparison with natural wetlands in Florida and a selection of wetlands all over the Northern hemisphere, CSA wetlands are relatively species poor in shrub and tree species (Table 17). In some cases, only 1 species, usually *Salix caroliniana*, was present in a forested CSA wetland, although this is the case in some other natural forested systems.

In terms of average basal area of woody stems, CSAs exhibit less developed structure than other natural wetlands (Table 18). CSAs only had approximately one third of the average basal area of reference systems, although on some sites there was a high basal area density along sections of some transects. Stand basal area of willow averages across sites appears lower than willow in other parts of Florida or the Southern U.S. (Table 19) when it occurs as the dominant species in the canopy layer. However, in comparison with natural systems, it should be noted that CSAs are young systems and trees are still growing.

Table 17. Comparison of Average CSA Woody Species Richness with Reference Wetlands.

Wetland Type – Location (# of Sites)	No. of Woody Species		
	Mean	Min	Max
CSAs – Florida (14)[1]	2.9	1	6
Forested – Florida (23)[2]	10.3	1	26
Riverine Wetlands (30)[3]	8.3	1	23
Basin Wetlands (17)[3]	6.0	1	14

[1]Includes only non-planted species with stems > 2.5 in DBH.
[2]Brown and Tighe (1991). Min/max estimated as 2 times ± standard deviation.
[3]Lugo and others (1988).

Table 18. Comparison of Average CSA Stand Basal Area with Reference Wetlands.

Wetland Type – Location (# of Sites)	Basal Area (m^2/ha)		
	Mean	Min	Max
CSAs – Florida (11)[1]	13.1	6.2	23.5
Forested – Florida (23)[2]	38.4	5.5	107.7
Riverine – Global (32)[3]	37.8	12.0	92.3
Basin – Global (15)[3]	39.9	9.5	70.8

[1]Only sites where transects with canopies included.
[2]Brown and Tighe (1991). Min/max estimated as 2 times ± standard deviation.
[3]Lugo and others (1988). Figures from a literature review of studies of natural wetlands around the world.

Table 19. Comparison of Willow Growth in CSA and Reference Wetlands by Basal Area.

Species	Location	Age	Basal Area (m^2/ha)		Reference
			Avg	Max	
Salix caroliniana	CSAs	10-40	12	26	This study
Salix caroliniana	FL	50+	22	25	Lee and others (2005)
Salix nigra	SE US	50+	30	--	Pitcher and McKnight (1990)

Species generally occurred in restricted ranges along the ecotone-wetland elevation gradient. Minimum and maximum water levels, including maximums as indicated by SHWL indicators, were determined at 15 sites for individual species (Tables 20-33). Minimum water levels represent the lowest water level measured upon visits during the 2004 and 2005 growing seasons.

Table 20. Species and Water Levels: CFI R-6.

Species	Minimum Water Level (m)	Maximum Observed Water Level (m)	Maximum Water Level by Indicator (m)
Taxodium dis.	0.30	0.54	1.09
Typha spp.	0.00	0.38	0.93
Scirpus cal.	0.00	0.38	0.93
Fraxinus car.	0.19	0.37	0.67
Salix car.	0.09	0.29	0.58
Ludwigia per.	0.00	0.29	0.58

Table 21. Species and Water Levels: CFI R-8.

Species	Minimum Water Level (m)	Maximum Observed Water Level (m)	Maximum Water Level by Indicator (m)
Scirpus cal.	0.04	0.36	N/A

Table 22. Species and Water Levels: CFI SP-1.

Species	Minimum Water Level (m)	Maximum Observed Water Level (m)	Maximum Water Level by Indicator (m)
Salix car.	0.00	0.95	1.41
Acer rub.	0.00	0.68	1.41
Taxodium dis.	0.00	0.95	1.36
Fraxinus car.	0.00	0.95	1.36
Typha spp.	0.00	0.00	0.73
Ludwigia per.	0.00	0.35	0.54
Cyperus odo.	0.00	0.00	0.41
Myrica cer.	0.00	0.00	0.13

Table 23. Species and Water Levels: CFI SP-5.

Species	Minimum Water Level (m)	Maximum Observed Water Level (m)	Maximum Water Level by Indicator (m)
Scirpus cal.	0.00	0.85	N/A
Salix car.	0.04	0.59	0.73
Typha spp.	0.11	0.57	0.72
Taxodium dis.	0.11	0.21	0.48
Ludwigia per.	0.00	0.20	0.47
Imperata cyl.	0.00	0.00	0.30

Table 24. Species and Water Levels: Mosaic F2B.

Species	Minimum Water Level (m)	Maximum Observed Water Level (m)	Maximum Water Level by Indicator (m)
Salix car.	0.00	0.23	N/A
Typha spp.	0.00	0.23	N/A
Ludwigia per.	0.00	0.05	N/A
Baccharis hal.	0.00	0.00	N/A
Cyperus odo.	0.00	0.00	N/A
Imperata cyl.	0.00	0.00	N/A

Table 25. Species and Water Levels: Mosaic FG3.

Species	Minimum Water Level (m)	Maximum Observed Water Level (m)	Maximum Water Level by Indicator (m)
Taxodium dis.	0.72	0.89	1.08
Salix car.	0.05	0.73	1.02
Myrica cer.	0.00	0.09	0.29
Cyperus odo.	0.00	0.08	N/A
Ludwigia per.	0.00	0.00	N/A

Table 26. Species and Water Levels: Mosaic FGH1A.

Species	Minimum Water Level (m)	Maximum Observed Water Level (m)	Maximum Water Level by Indicator (m)
Ludwigia per.	0.00	0.39	N/A
Polygonum hyp.	0.00	0.39	N/A
Salix car.	0.00	0.00	N/A
Baccharis hal.	0.00	0.00	N/A

Table 27. Species and Water Levels: Mosaic H1.

Species	Minimum Water Level (m)	Maximum Observed Water Level (m)	Maximum Water Level by Indicator (m)
Salix car.	0.00	0.63	0.66
Polygonum hyp.	0.20	0.63	0.63
Imperata cyl.	0.00	0.60	0.63
Ludwigia per.	0.00	0.40	0.43
Cyperus odo.	0.20	0.36	0.42
Myrica cer.	0.00	0.00	0.00
Sambucus can.	0.00	0.00	0.00

Table 28. Species and Water Levels: Mosaic K5.

Species	Minimum Water Level (m)	Maximum Observed Water Level (m)	Maximum Water Level by Indicator (m)
Salix car.	0.00	1.30	N/A
Typha spp.	0.30	1.11	N/A
Ludwigia per.	0.00	0.87	N/A
Acer rub.	0.00	0.00	0.24
Myrica cer.	0.00	0.00	0.24
Imperata cyl.	0.00	0.00	0.44

Table 29. Species and Water Levels: PCS SA 01.

Species	Minimum Water Level (m)	Maximum Observed Water Level (m)	Maximum Water Level by Indicator (m)
Salix car.	0.00	1.36	N/A
Myrica cer.	0.00	0.47	N/A
Sambucus can.	0.00	0.33	N/A

Table 30. Species and Water Levels: PCS SA 10.

Species	Minimum Water Level (m)	Maximum Observed Water Level (m)	Maximum Water Level by Indicator (m)
Salix car.	0.00	1.74	N/A
Baccharis hal.	0.00	0.77	N/A

Table 31. Species and Water Levels: PCS SA 3A.

Species	Minimum Water Level (m)	Maximum Observed Water Level (m)	Maximum Water Level by Indicator (m)
Salix car.	0.00	1.53	N/A
Sambucus can.	0.00	0.94	N/A
Myrica cer.	0.00	0.00	N/A

Table 32. Species and Water Levels: PCS SA 04.

Species	Minimum Water Level (m)	Maximum Observed Water Level (m)	Maximum Water Level by Indicator (m)
Salix car.	0.00	0.38	0.42
Typha spp.	0.00	0.31	0.43
Polygonum hyp.	0.00	0.30	0.42
Baccharis hal.	0.00	0.00	N/A

Table 33. Species and Water Levels: Tenoroc-3.

Species	Minimum Water Level (m)	Maximum Observed Water Level (m)	Maximum Water Level by Indicator (m)
Salix car.	0.17	0.85	1.22
Taxodium dis.	0.09	0.70	1.21
Acer rub.	0.00	0.70	1.21
Myrica cer.	0.00	0.10	0.51

Although some species are sole occupants of one vertical stratum of some CSA wetlands, it is more broadly applicable to view CSAs wetlands in terms of plant communities when determining how soils and water levels affect their distribution. Tables 34-36 summarize the major vegetation communities identified at the tree, shrub, and ground layers on the vegetation monitoring transects in terms of the cumulative longitudinal coverage of the communities on all transects, cumulative occurrences of the communities, and the dominant plant species within each community. Plant communities along each transect and are shown on part (b) in the odd-numbered figures of Appendix A. Dominant species (or community type if one species does not display clear dominance) are listed by genus (or community type when applicable). For zones less than 5 m in length, labels for species are not visible. The dominance of these communities on CSAs is depicted in pie charts in Figure 41. For about half of the wetland areas surveyed, there was no presence of a community in the canopy layer. The same was apparent for the shrub layer in areas surveyed. Willow communities dominated the canopy layer, as was shown at the species level (Table 14), but primrose willow was the most common community in the shrub layer. Species mixes occurred at all strata but were most common at the ground level, in part due to greater species richness and less space occupied by a species. Some areas were dominated by cattails, cogongrass or a floating aquatic community.

Table 34. Dominant CSA Wetland Communities in the Canopy Layer.

Name	Longitudinal Coverage (m)	Occurrences	Most Frequent Species
Willow	742	47	*Salix caroliniana*
Mixed wetland forested	297	12	*Salix caroliniana, Taxodium distichum*
Cypress	110	9	*Taxodium distichum*
Mixed upland forested	34	3	*Myrica cerifera, Quercus laurifolia, Carya aquatica, Salix caroliniana*
Ash	24	1	*Fraxinus caroliniana*
Chinese tallow	20	2	*Sapium sebiferum*

Table 35. Dominant CSA Wetland Communities in the Shrub Layer.

Name	Longitudinal Coverage (m)	Occurrences	Most Frequent Species
Primrose willow	389	31	*Ludwigia peruviana*
Willow	266	22	*Salix caroliniana*
Mixed wetland shrubs	214	16	*Salix caroliniana, Ludwigia peruviana*
Saltbush	147	6	*Baccharis halimifolia*
Wax myrtle	69	6	*Myrica cerifera*
Elderberry	49	3	*Sambucus canadensis*
Mixed upland shrubs	15	1	*Myrica cerifera, Sambucus canadensis, Baccharis halimifolia*

Table 36. Dominant CSA Wetland Communities in the Ground Layer.

Name	Longitudinal Coverage (m)	Occurrences	Most Frequent Species
Mixed wetland herbaceous	778	42	*Lemna minor, Ludwigia peruviana, Phyla nodiflora, Hydrocotyle umbellata, Salix caroliniana, Polygonum hydropiperoides, Typha spp.*
Duckweed	387	27	*Lemna minor, Spirodela polyrrhiza*
Mixed upland herbaceous	232	20	*Eupatorium capillifolium, Ludwigia peruviana, Ambrosia artemisiifolia, Lemna minor, Paspalum notatum, Pluchea odorata*
Cattail	192	17	*Typha spp.*
Cogongrass	177	16	*Imperata cylindrica*
Water lettuce	125	6	*Pistia stratiotes*
Bulrush	73	4	*Scirpus californicus*
Smartweed	35	2	*Polygonum punctatum*
Sedge	19	3	*Cyperus odoratus*

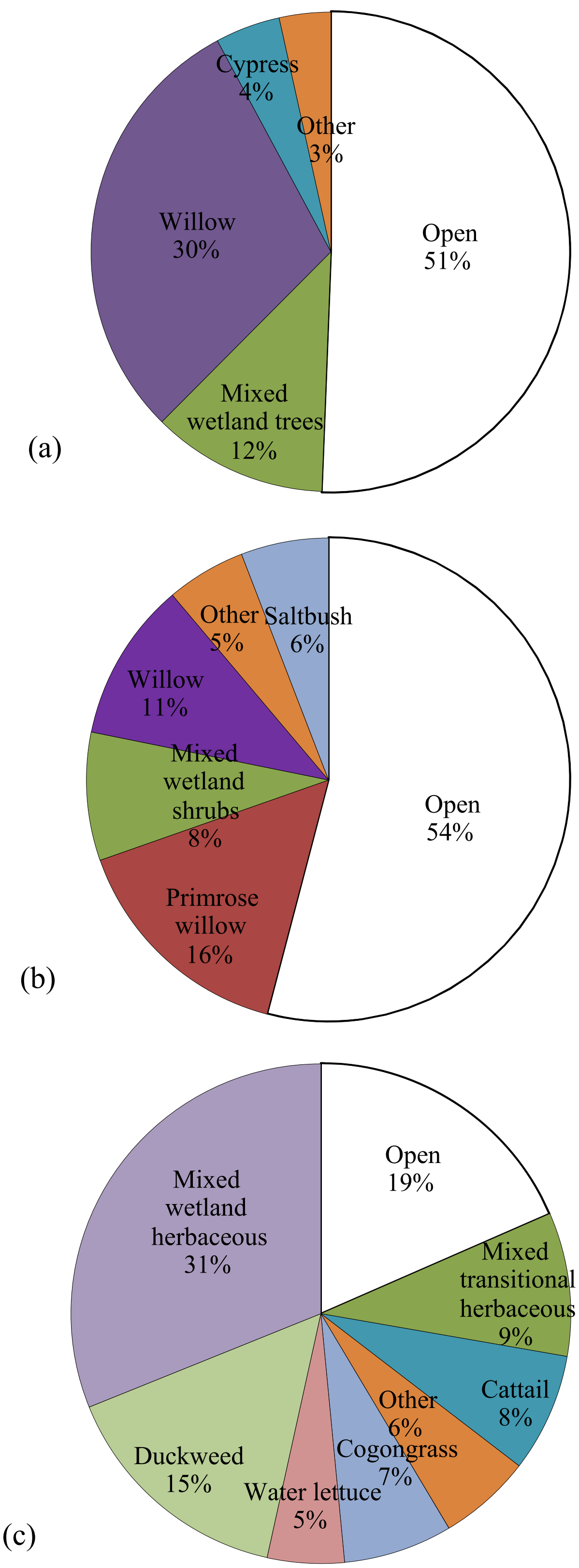


Figure 41. Vegetation Community Dominance in the Three Vertical Strata: (a) Canopy, (b) Shrub, and (c) Ground, on Sampled CSAs.

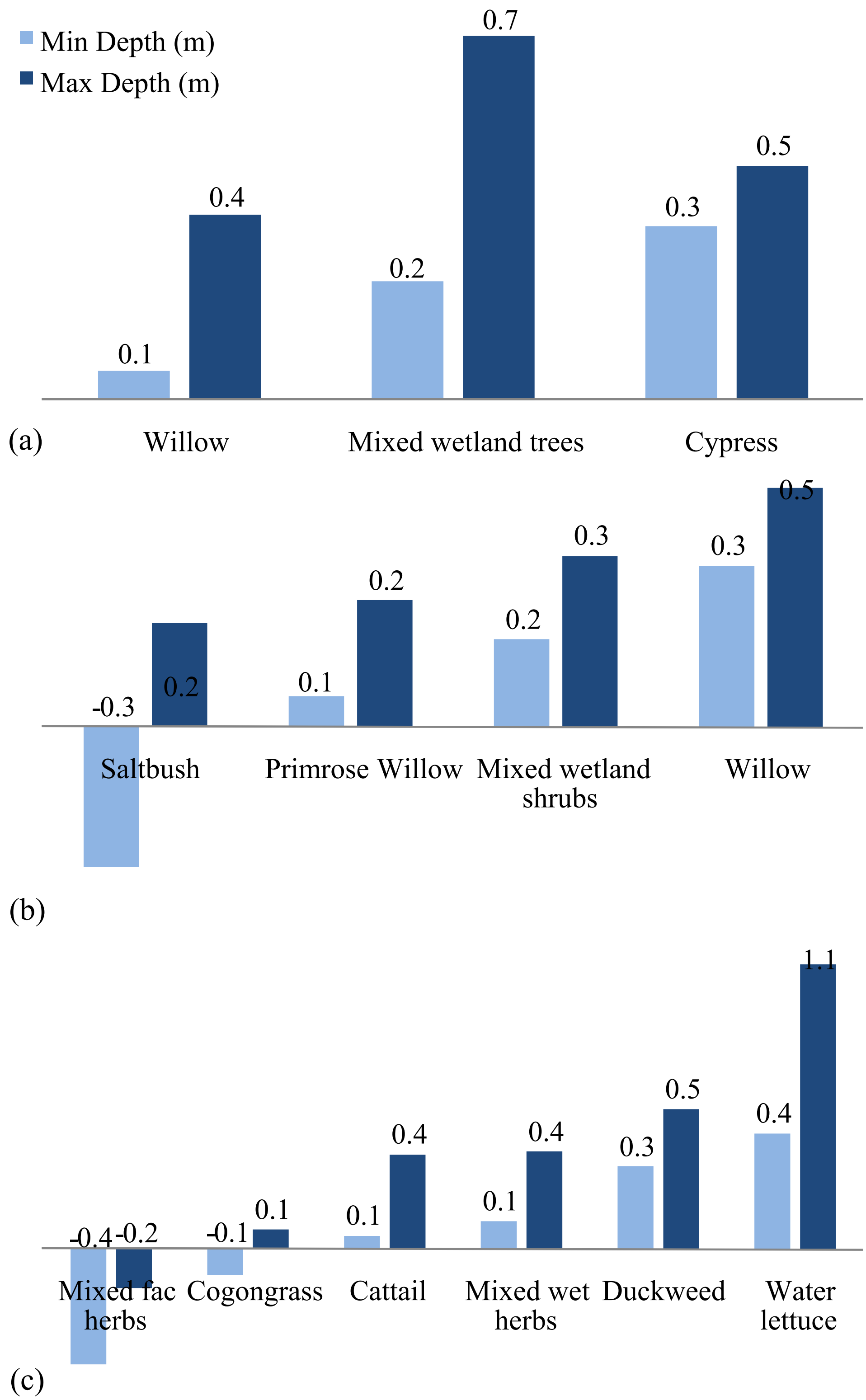


Figure 42. Average Minimum and Maximum Water Levels in Dominant Vegetation Communities in the (a) Canopy, (b) Shrub, and (c) Ground Layers.

Water levels had a strong influence over the wetland communities monitored. Community distributions with water levels are depicted by strata in Figure 42. Canopy and woody communities on CSAs typically occurred in areas with standing water, which supported mixed communities up to a depth 0.7 m on average. Shrub communities extended further into the ecotone, with *Baccharis halimifolia* (saltbush) often occupying the transitional zone, and primrose willow a zone of more moderate inundation. At the ground level, facultative herbs or cogongrass occupied the ecotone, *Typha spp.* or mixed wetland communities appeared in the zones with moderate inundation, and floating aquatics occupied the deepest areas.

Soils

Soil chemical characteristics of CSA wetlands are summarized in Table 37. Average available phosphorus and total nitrogen in soils sampled along transects was 606 and 2280 ppm, respectively. Percent organic matter, TKN, available P, and pH for soil samples taken along each transect are shown in part(a) of the even-numbered figures of Appendix A.

Table 37. Soil Chemical Characteristics Summary.

	pH	OM (%)	P (ppm)	TN (ppm)
Avg	7.3	11.6	607	2280
SD	0.6	6.1	559	1429
Min	5.8	1.0	129	307
Max	8.1	26.3	3430	7090

Table 38 compares organic matter accumulation and total phosphorus and nitrogen with findings from a study of minimally impaired wetlands in two broadly defined ecological regions in the southeastern U.S. Percent organic matter is not as high in CSAs likely due to the age of sites. Available phosphorus in CSAs (as a proxy but slight underestimate of TP) is more than double the amount in wetlands of the Southern Coastal plain, while nitrogen is less. As a results CSAs have an N:P ratio of about 4, which is much less than in the reference systems, especially the Southern Coastal Plain wetlands.

Table 38. Comparison of CSA Soil Characteristics with Reference Wetlands (Greco 2004)*.

Wetland Type (# Samples)	OM (%)	TP (ppm)	TN (ppm)	N:P
	Avg ± SD			
CSAs (56)	11.6 ± 6.1	607 ± 559	2280 ± 1429	4
Southern Coastal Plain (26)	42.1 ± 33.6	218 ± 152	5522 ± 2386	25
Southeastern Forested Plain (55)	23.2 ± 22.8	367 ± 283	2917 ± 1067	8

*Reference wetlands included wetlands in the U.S. EPA 'Southern Coastal Plain' ecoregion, which overlaps with all CSAs studied. The Southern Coastal Plain extends from central to north FL, and the Southeastern Forested Plain from northwestern FL into other southeastern states.

Correlations of total nitrogen (TN) and available phosphorus (P) with organic matter (OM) are presented in Figures 43 and 44. There was a strong increasing trend between TN and OM in CSA soils, and a close correlation. On the other hand, with increasing OM, no change in P was evident.

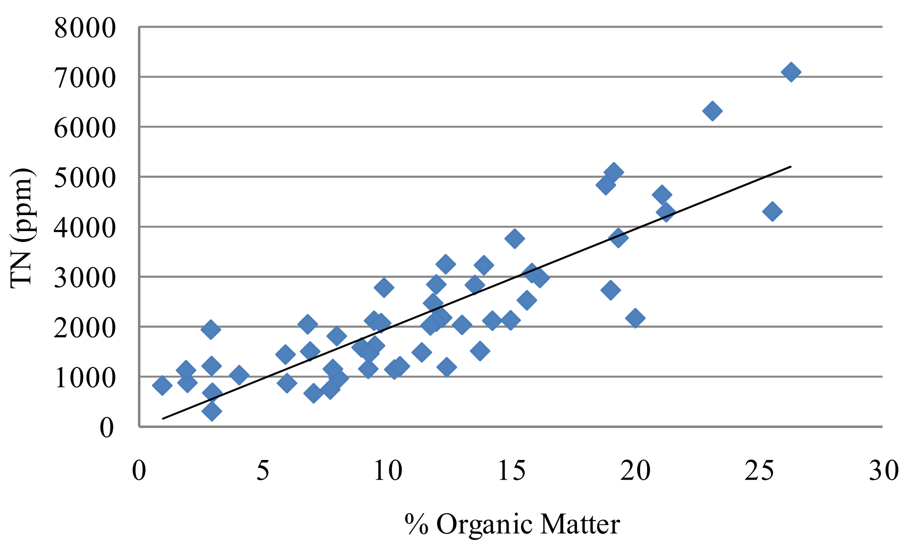


Figure 43. Total Nitrogen and Percent Organic Matter.

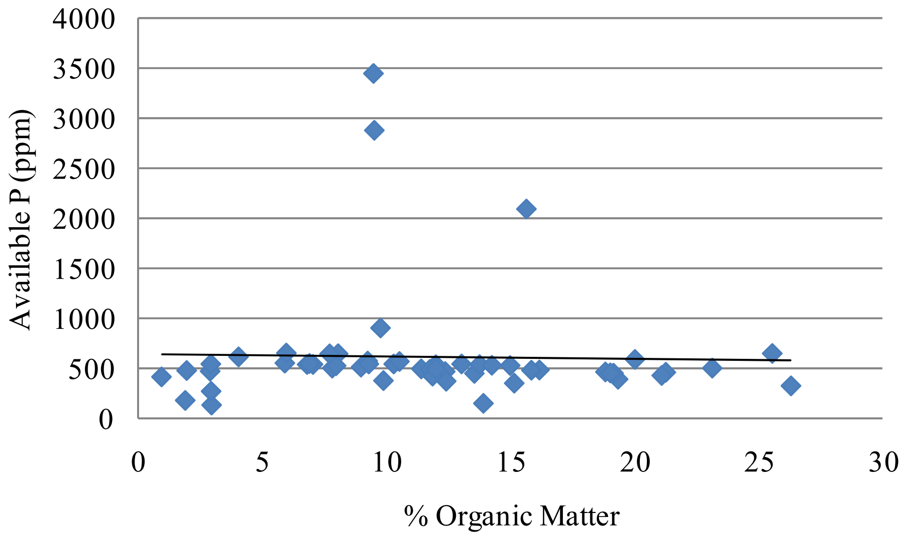
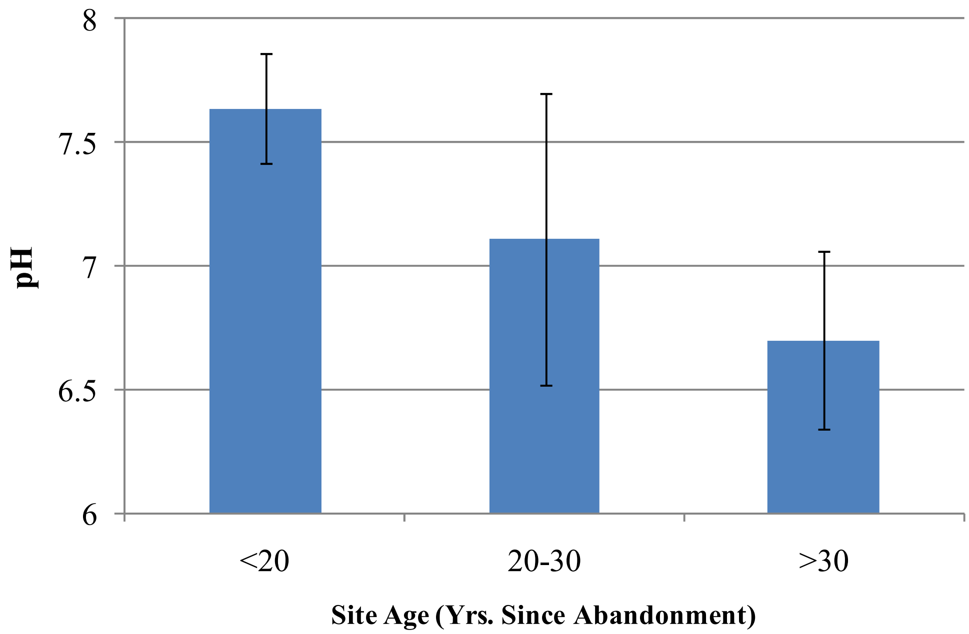


Figure 44. Available Phosphorus and Percent Organic Matter.

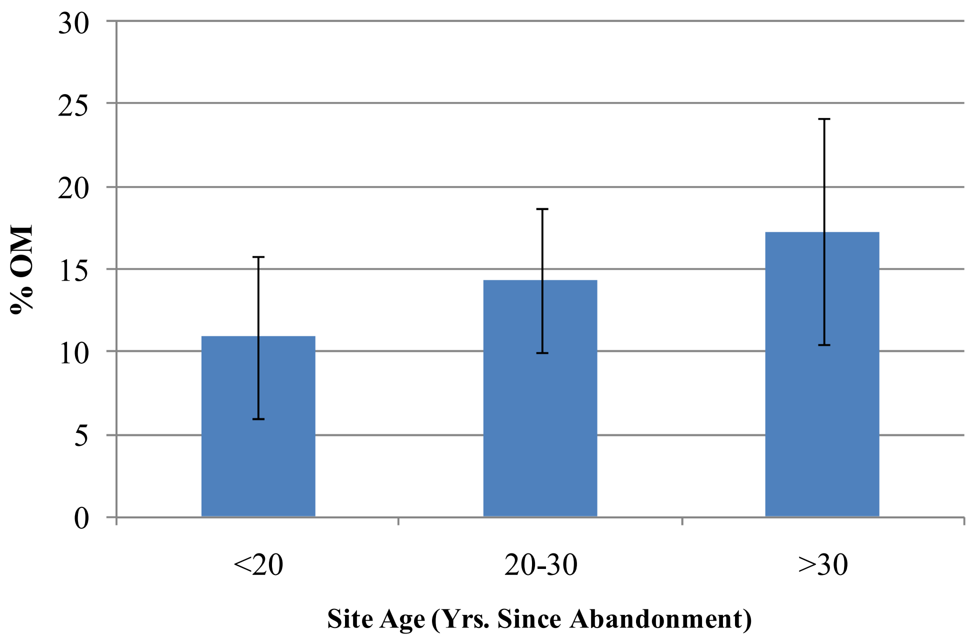
As sites increased in age, general trends toward neutral soils and increasing soil organic matter contents were present (Figures 45 and 46). However, there was considerable variation of organic matter within sites of similar age, as indicated by wide standard error range in Figure 46.

Figures 47-49 illustrate how typical CSA wetland vegetation communities coincide with soil chemical characteristics. The strongest signature of these relationships was the presence of cattail communities in areas of high available P. Mixed wetland shrub communities were associated with the highest percentages of soil organic matter, and cogongrass communities were associated with the highest TN measurements.



Note: Sites are grouped by age classes.

Figure 45. pH in CSA Soils by Age.



Note: Sites are grouped by age classes.

Figure 46. Percent Organic Matter in CSA Soils by Age.

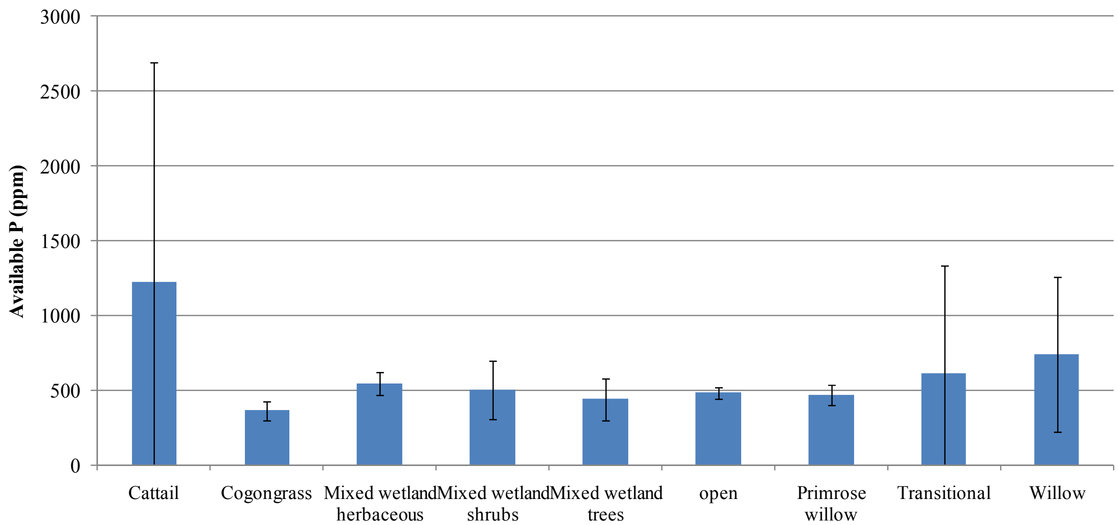


Figure 47. Available P in Soils with Presence of Distinct Wetland Communities.

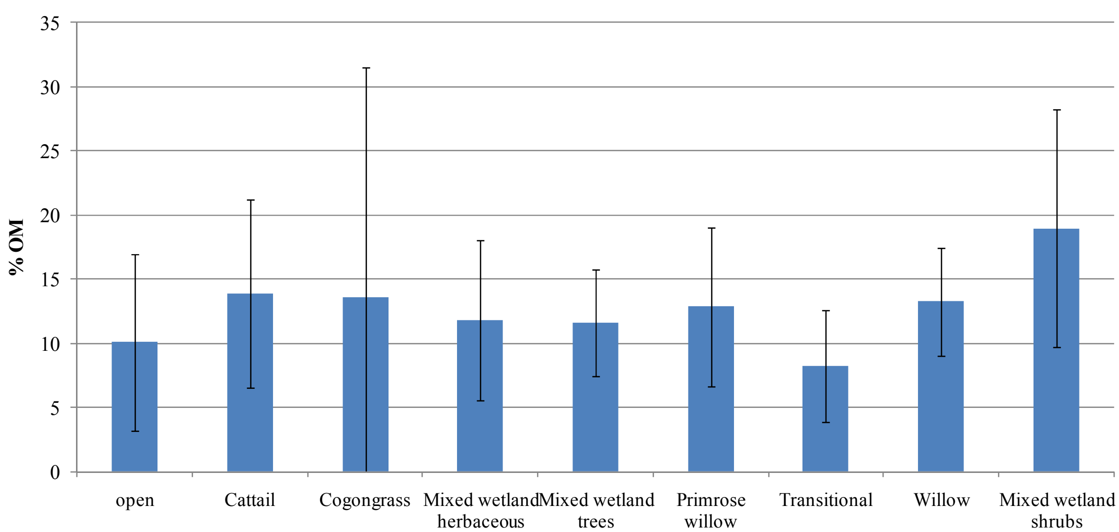


Figure 48. Percent Organic Matter in Soils with Presence of Distinct Wetland Communities.

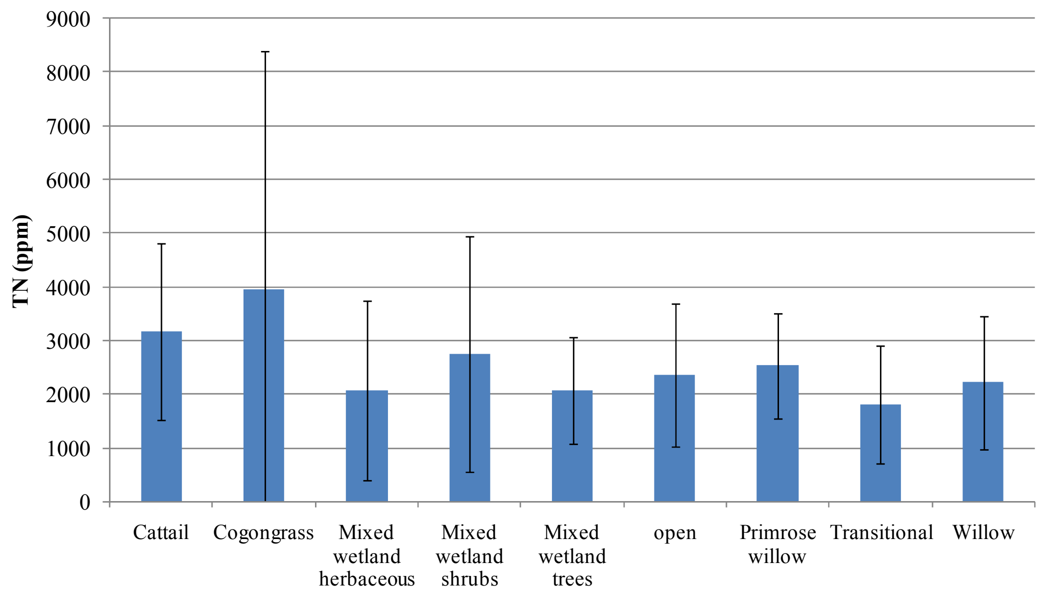
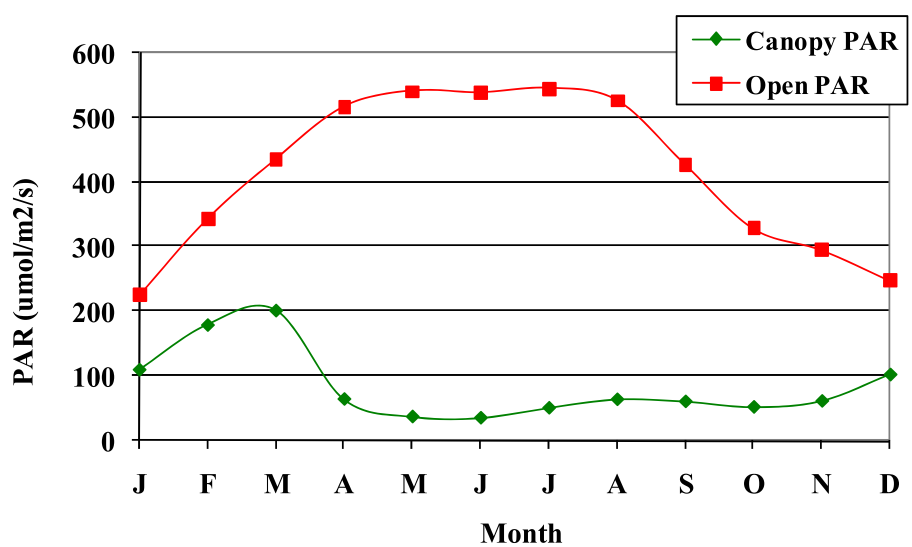


Figure 49. Total Nitrogen in Soils with Presence of Distinct Wetland Communities.

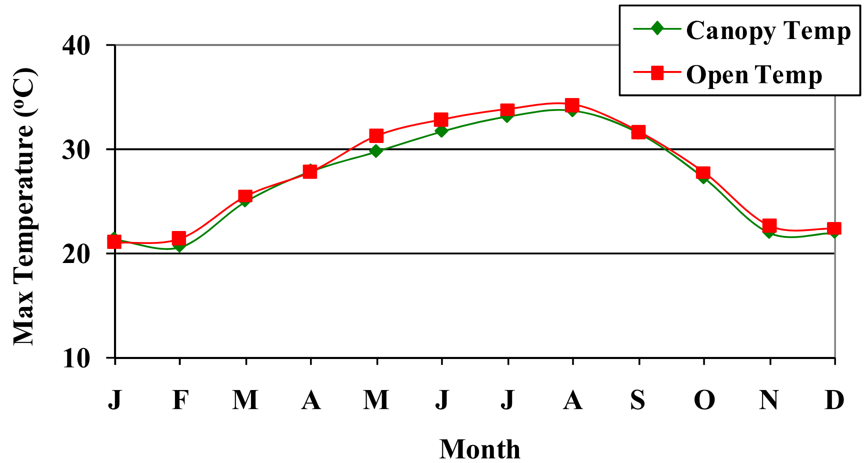
Microclimate

Light as measured by incoming photosynthetically active radiation (PAR) showed a reduction of light under the canopy which became exaggerated during the summer months when leaves were present (Figure 50). Maximum temperatures in the canopy from May to Oct were on average 0.6 ˚C lower than in the open area (Figure 51). Less difference in maximum temperature was detected in the cooler half of the year.



Note: Data are averaged by month over a two-year period.

Figure 50. PAR Comparison Under and Outside of the Canopy at SA 10.



Note: Data are averaged by month over a two-year period.

Figure 51. Temperature Comparison Under and Outside of the Canopy at SA 10.

Elevation Profiles

Relative elevation profiles along transects are illustrated in Figure 52. The pattern of elevation change reflects the transition from the edge of a wetland, or ecotone area, into the wetland area. The variation between elevation profiles of CSA wetlands was evident. Two general patterns were present: (1) Relatively flat areas with only minor (<20 cm) fluctuations, especially after the initial transition into the wetland, and (2) deeper depressions with a sharp gradient into the wetland from the ecotone.

Elevation change and depth along transects is summarized in Table 39. The slope in the transitional areas was sharper than the slope in the wetlands, however, there was considerable variation in the ecotone area depending on the wetland feature. The average depth along the transects varied from 0.1-2. It should be noted that wetlands on CSAs may extend to greater depths, but greater depths impeded monitoring and characterization.

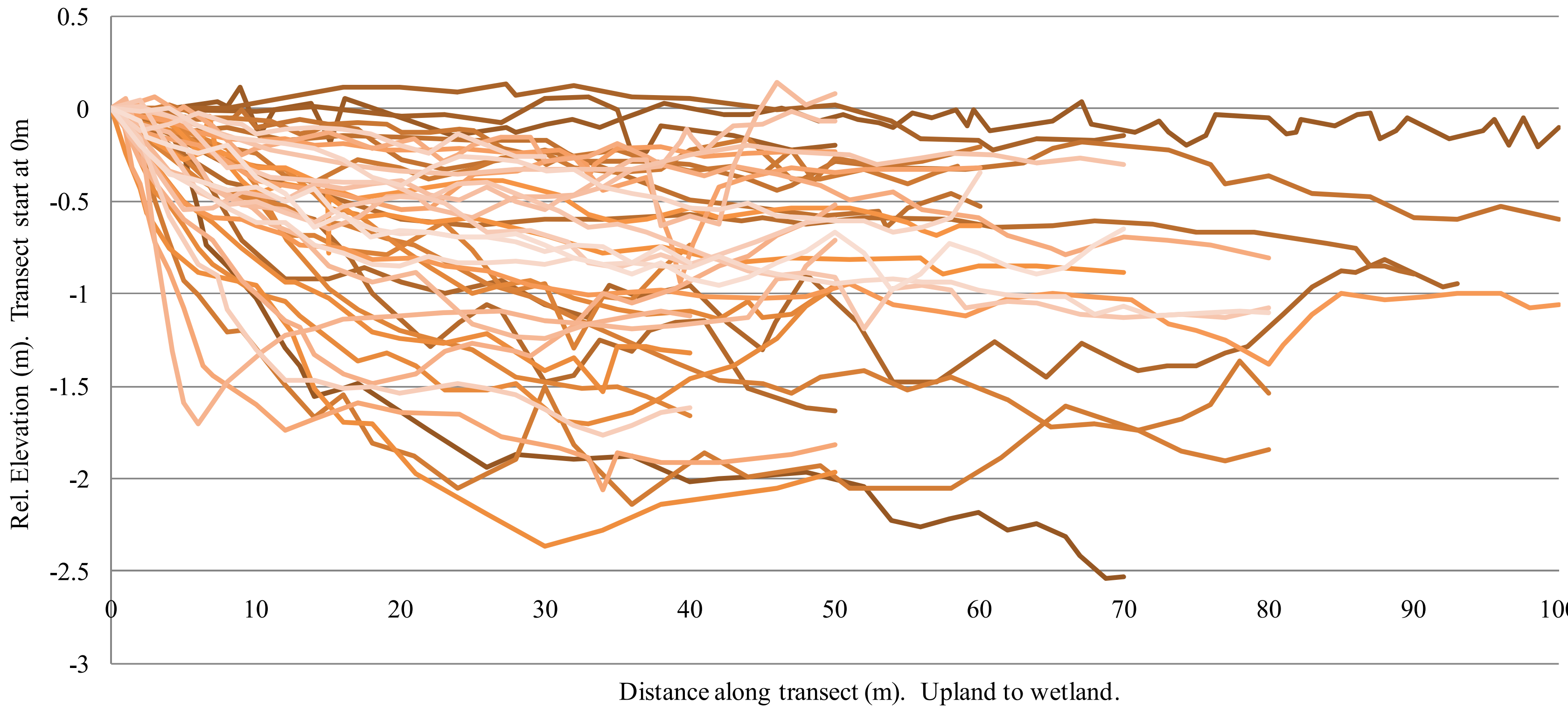
Table 39. Elevation Summary of Transects Through Ecotone and Wetland Areas on CSAs.

	Elev Change in Ecotone (%) n = 31	Elev Change in Wetland (%) n = 42	Wetland Depth (m) n = 42
Avg	7.6	3.5	0.8
SD	6.0	1.6	0.4
Min	1.4	1.1	0.1
Max	24.0	8.6	2.1

The elevation profiles in CSA wetlands were compared with those of natural wetlands from Florida as a reference (Table 40). CSAs wetlands varied in depth like natural Florida wetlands of many types, although the average depth places them in the range of wetlands of moderate depth. However, the wetland profiles reflected that the elevation profiles on CSAs are highly variable and feature dependent, with some features presenting very shallow depths and others very deep (Figure 52).

Table 40. Comparison of Depth in CSA and Reference Wetlands (Brown and Tighe 1991).

	CSA Wetland n = 42	Lake Fringe n = 2	Marsh n = 9	Cypress Dome n = 6	Bayhead n = 3	Hardwood Swamp n = 10
Avg	0.8	1.4	0.8	0.7	0.7	0.4
SD	0.4	0.2	0.5	0.3	0.3	0.7
Min	0.1	1.2	0.1	0.3	0.5	0.3
Max	2.1	1.5	1.7	1.2	1.0	2.3



Note: Data for 42 transects on 17 CSAs are presented, varying in length from 40 to 100 meters.

Figure 52. Transect Elevation Profiles.

Vegetation Mapping of Aerial Photographs

Land cover maps were developed for PCS SA 01 and PCS SA 03 for 1996 and 2003 and of PCS SA 04 for 2003 (Figures 53-57). Percent of these sites that were upland, wetland, and transition (*Baccharis halimifolia.*) were calculated based on the land cover maps (Table 41). In comparing the 1996 and 2003 photographs for PCS SA 01 and PCS SA 3A, there were increases of wetland cover of 9.8 and 3.1%, respectively. The land cover maps for 2003 were used in conjunction with observed water ranges for vegetation types (Tables 29, 31, and 32) to develop water depth maps (see Figures 58-60).

Table 41. Land Cover Percentages.

Site, Year	Wetland (%)	Upland (%)	Transition (%)
PCS SA 01, 1996	26.6	73.4	0
PCS SA 01, 2003	29.2	70.8	0
PCS SA 3A, 1996	55.7	44.3	0
PCS SA 3A, 2003	57.4	42.6	0
PCS SA 04, 2003	59.5	17.1	23.4

The premise behind this objective was to use on-site monitoring of vegetation communities to ground-truth vegetation mapping of recent aerial photographs and to relate inundation to dominant communities. Employing these established vegetation-hydrology linkages and vegetation signatures, vegetation would be mapped in a time series of aerial photographs and used to identify past hydrologic regimes. As a result of the large range of inundation depths experienced by dominant vegetation, especially *Salix caroliniana* which is the most conspicuous community observable on the photographs, it was difficult to relate hydrologic regime to vegetation maps. Furthermore, the quality of the older aerial photographs, where only general canopy features were evident, created substantial difficulties in addressing this objective. Some analyses have been performed using this procedure, as presented above, but because of the problems encountered and likely inaccuracies of the analyses, continued effort was not dedicated to meeting this objective.

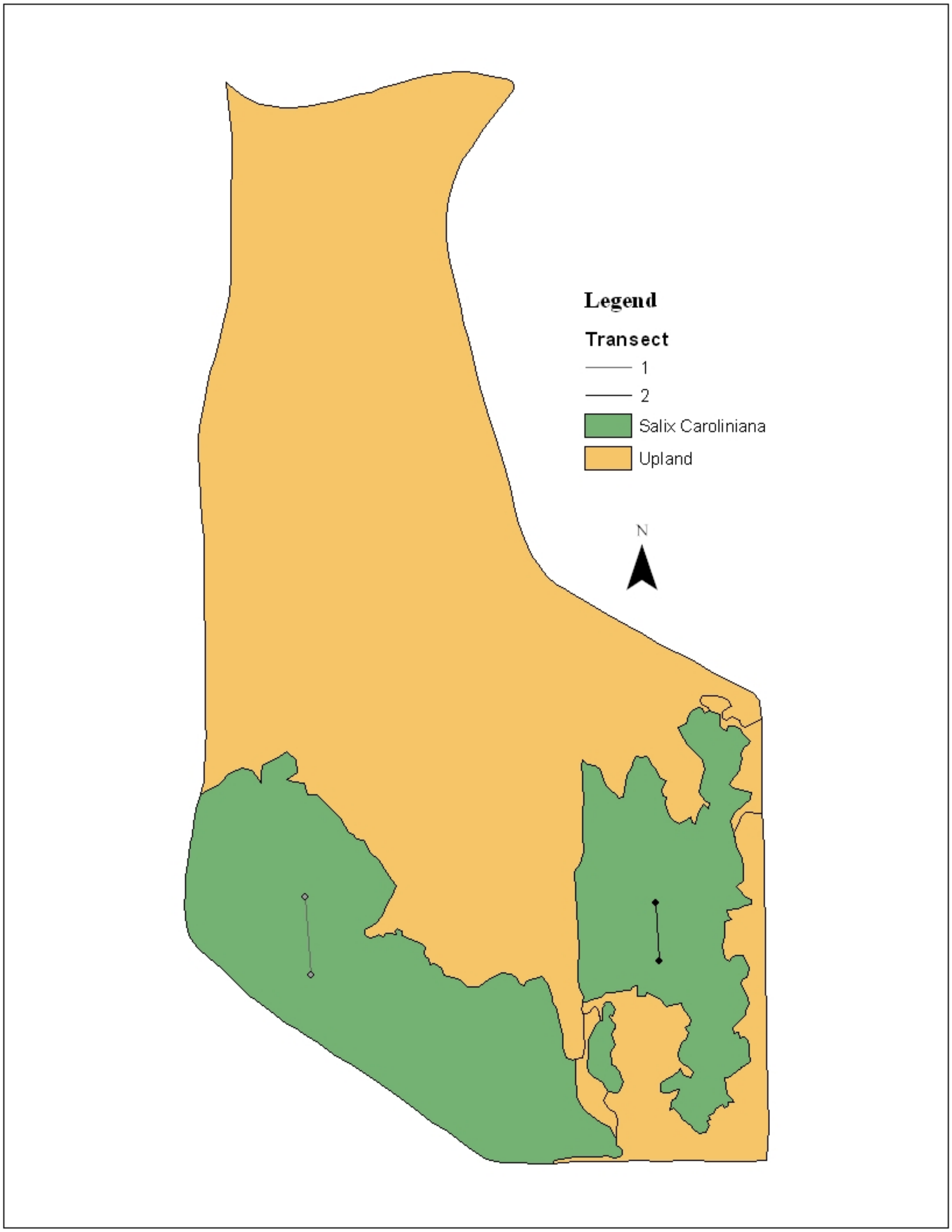


Figure 53. PCS SA 01 2003 Land Cover Map.

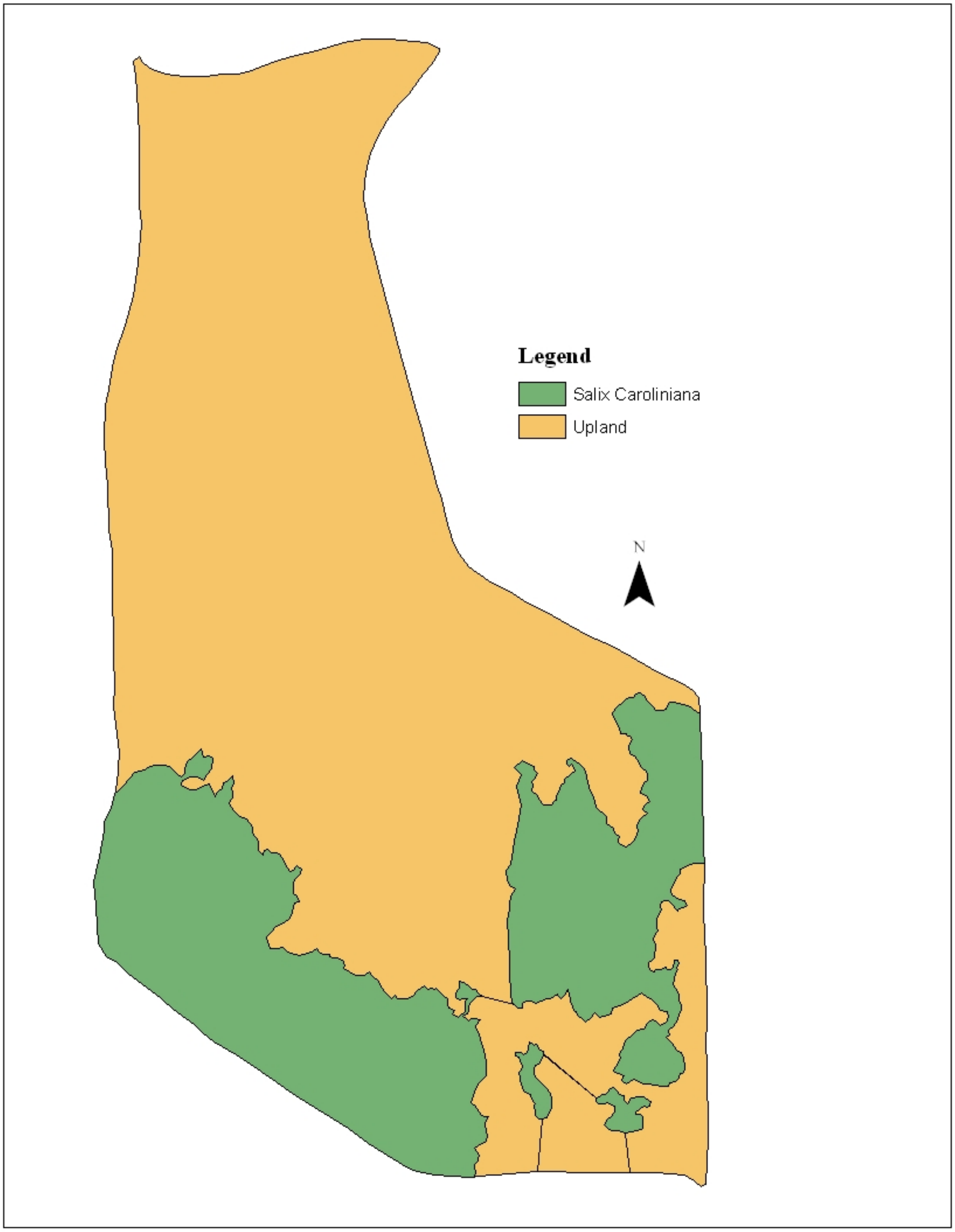


Figure 54. PCS SA 01 1996 Land Cover Map.

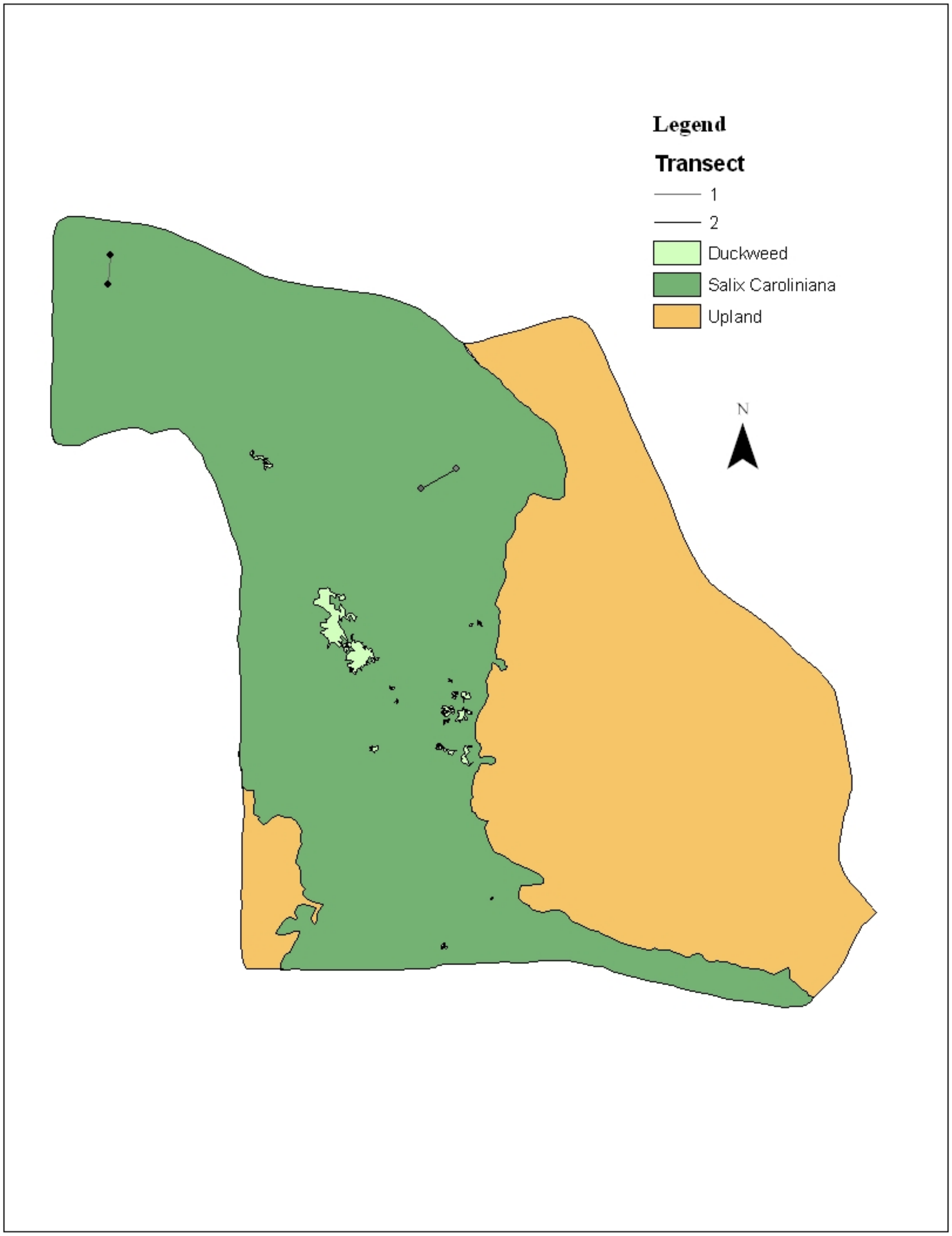


Figure 55. PCS SA 3A 2003 Land Cover Map.

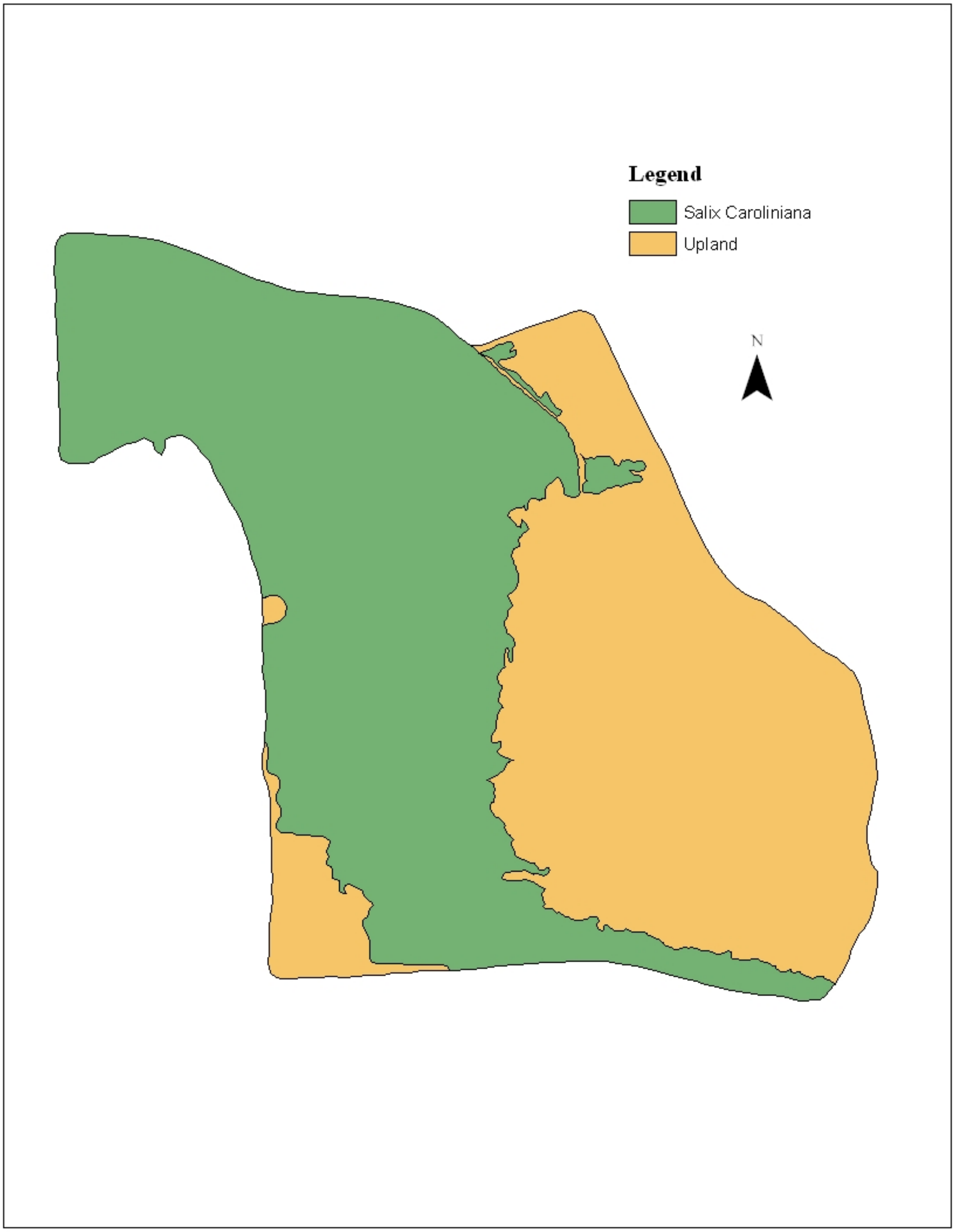


Figure 56. PCS SA 3A 1996 Land Cover Map.

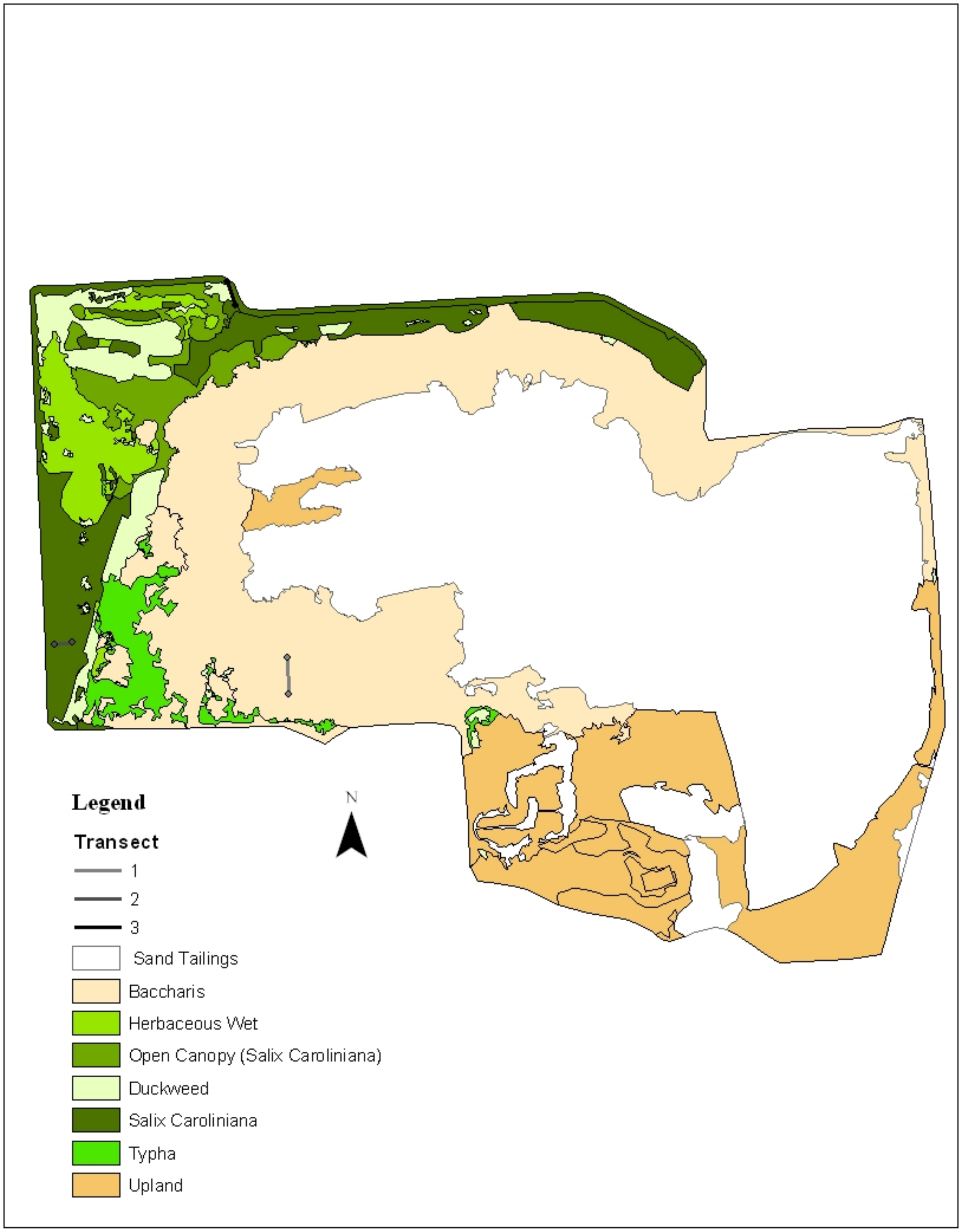


Figure 57. PCS SA 04 2003 Land Cover Map.

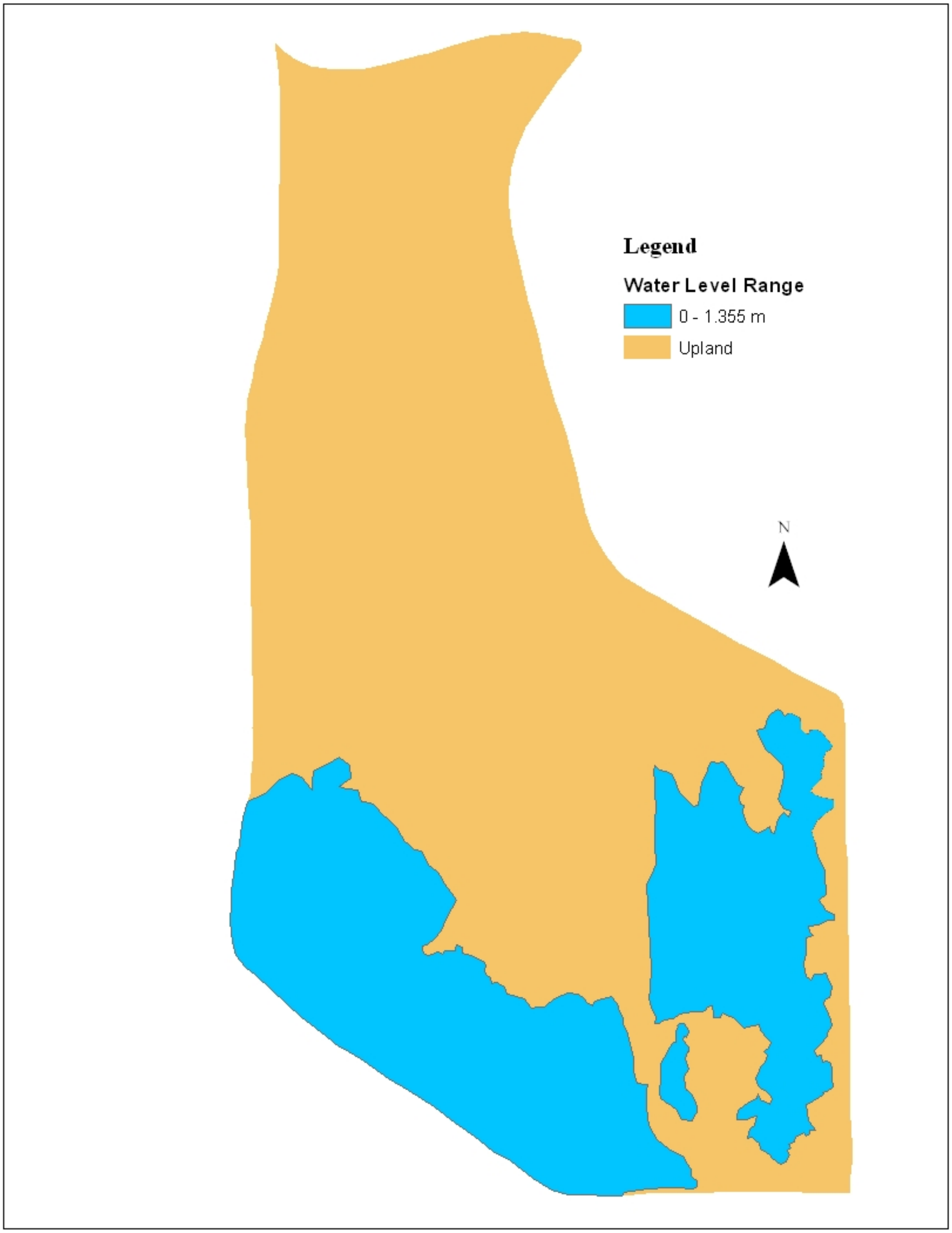


Figure 58. PCS SA 01 2003 Water Level Ranges.

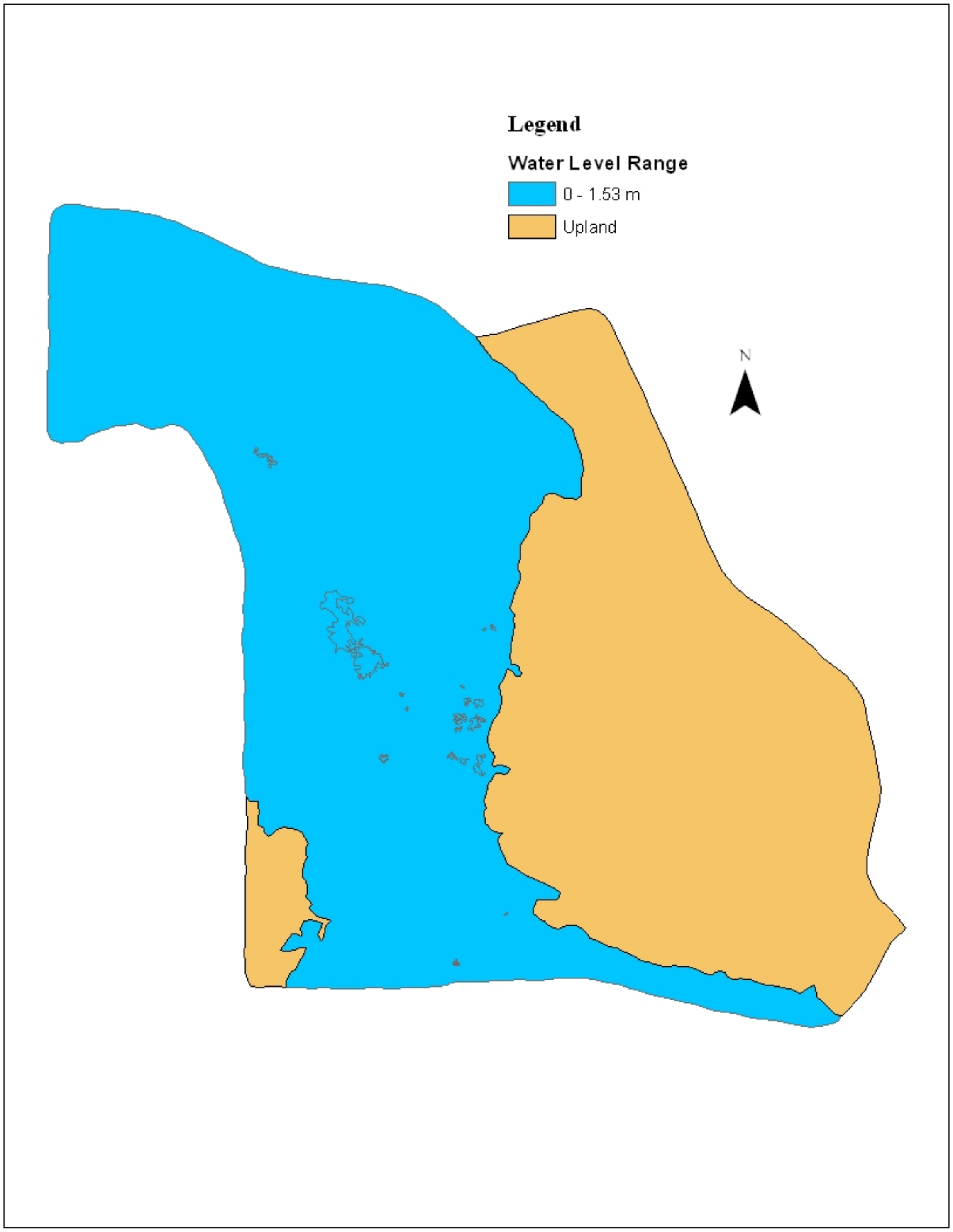


Figure 59. PCS SA 3A 2003 Water Level Ranges.

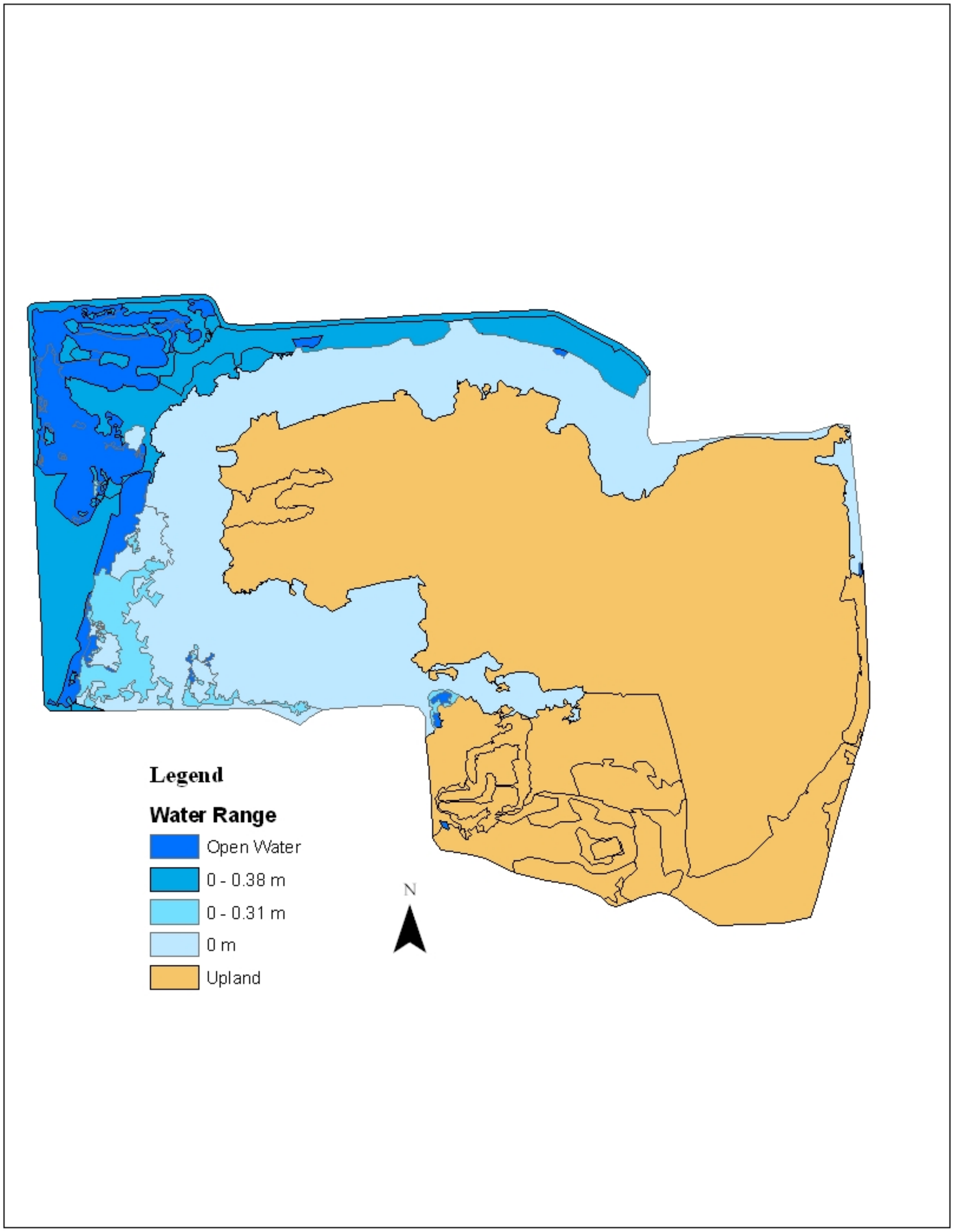


Figure 60. PCS SA 04 2003 Water Level Ranges.

HYDROLOGIC ANALYSIS AND MODELING

Hydroperiod Evaluation

Eight hydrology sites were established for monitoring precipitation and water levels within dominant surface water features (Figures 1-8). Three sites had multiple surface-water features instrumented with wells including Mosaic H1, PCS SA 10, and Mosaic K5 which had three, two, and two features instrumented, respectively. Surface water levels and precipitation for the sites are shown in Figures 61-76. The surface water levels of most sites experienced decline starting in spring 2006 through the remainder of the record, with some sites having significant dry periods due to the drought years of 2006 and 2007. PCS SA 01 experienced the reverse, however, with increasing water levels as a result of a beaver dam causing its outfall to become essentially inactive. Inserts were mortared in place in the outfall structure during May 2006 to completely block any outflow from PCS SA 01 so that an accurate water balance could be performed.

Three other sites experienced surface water outflow during the period of study, including PCS SA 10, Mosaic HP-10, and Mosaic K5. PCS SA 10 experienced outflow through a weir when Surface Water-1 reached 1.47 m and during August 2005 through April 2006 (Figure 63). Mosaic HP-10 experienced intermittent outflow throughout its record through a ditch network that exited the site via a buried pipe. It was determined that outflow from HP-10 occurred when surface water levels were at 0.45 m or above (Figure 75). There is a gap in the data for HP-10 from 2/13/08 to 3/12/08 due to equipment failure. Surface water outflow occurred from Mosaic K5 through a vegetated swale when surface water feature-1 (SW-1) reached 0.5 m, which happened frequently throughout the period of study (Figure 73).

Hydrologic regime in terms of water depths and frequency of flooding and drying significantly varied from site to site (Figure 77). The two PCS sites remained flooded throughout the study period and had more buffered regimes with less response to rain events, slower rates of decline, and greater water depths than the other six sites (Figures 61, 63, and 77). Mosaic K5 and HP-10 also experienced permanent flooding, with the exception of K5 during a brief period in June 2006, but with shallower water depths due to the elevation position of their outflow systems compared to that of PCS SA 10's (Figures 73, 75, and 77). Mosaic H1 Surface Water-1 (SW-1) and Williams Co. experienced a flashier regime with steeper declines, greater response to rain events, and significant dry periods with depths below ground reaching over 0.5 m and 1.5 m, respectively. These two systems have wetland slopes that are fairly flat versus the steep and deep systems of the two PCS sites, which partially accounts for the different hydrologic regimes (Figure 52). Tenoroc-4 and CFI SP-1 experienced the most extensive dry periods that occurred for the majority of the record for Tenoroc-4 and consistently since November 2006 at CFI SP-1. Despite wells being installed 1.3 m below ground at SP-1 and 1.54 m below ground at Tenoroc-4, water levels still dropped below the wells.

As such, it can only be concluded that water depths fell to at least or greater than the well depths for these two systems.

Figure 61. Surface Water Levels for PCS SA 01.

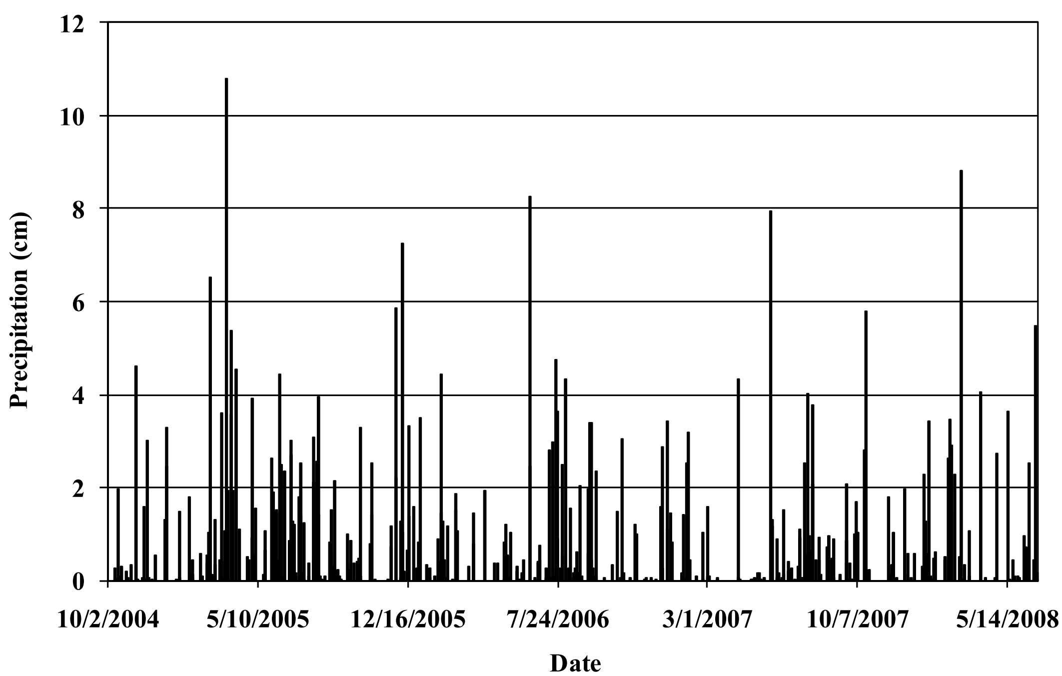


Figure 62. Precipitation for PCS SA 01.

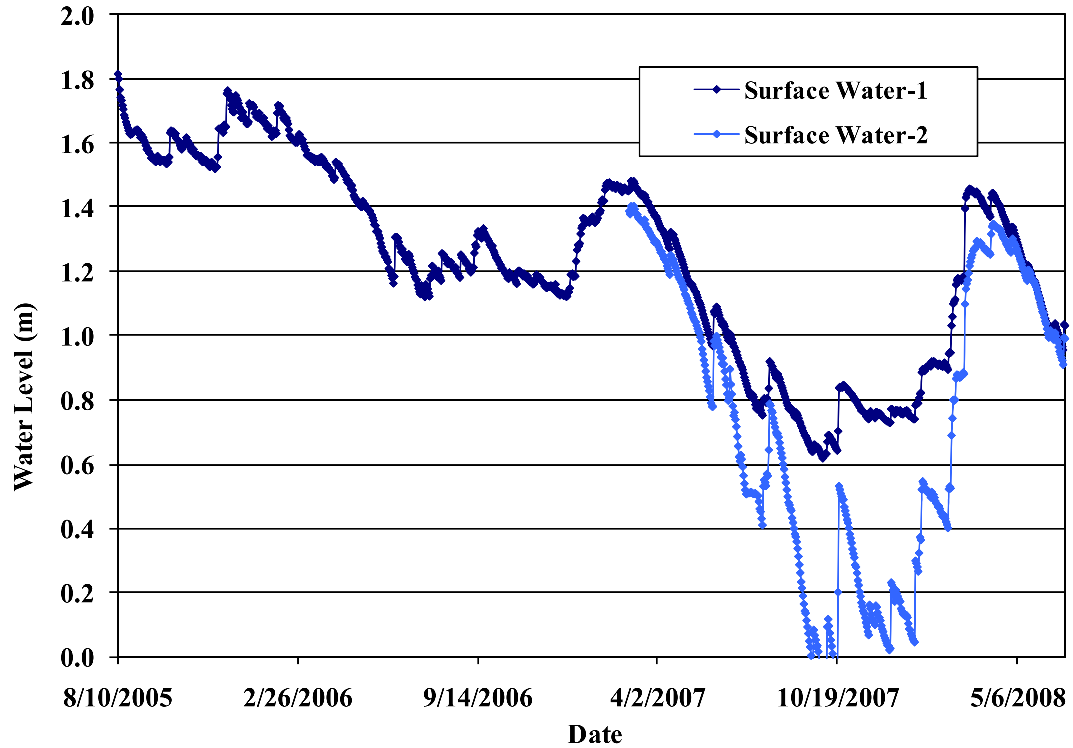


Figure 63. Surface Water Levels for PCS SA 10.

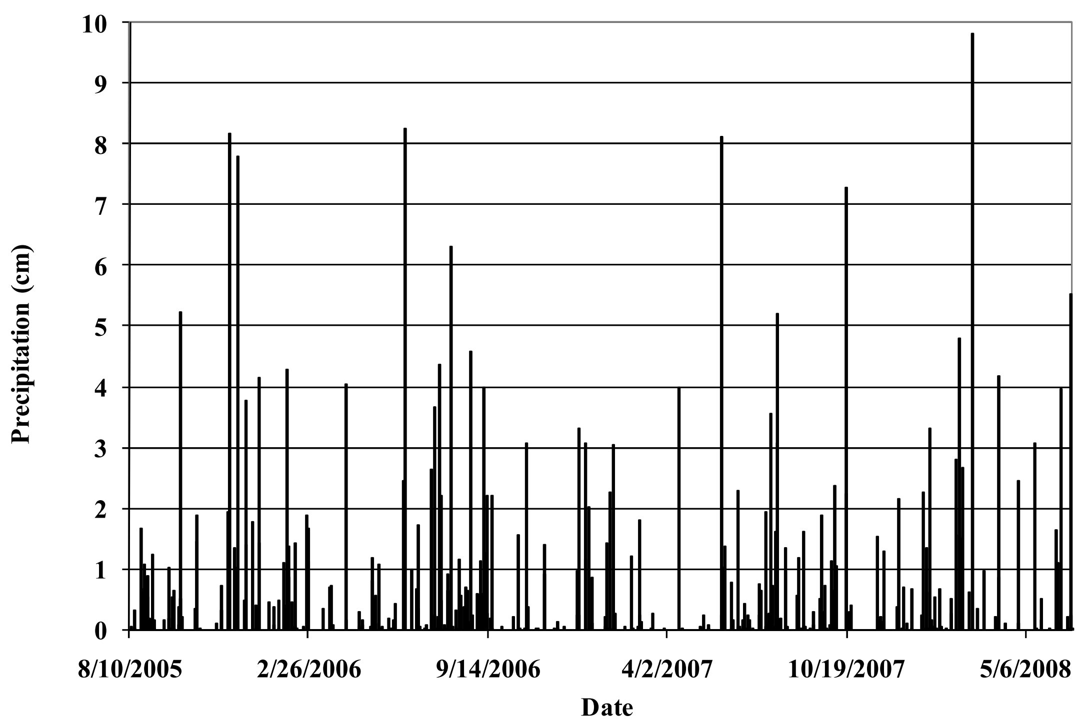


Figure 64. Precipitation for PCS SA 10.

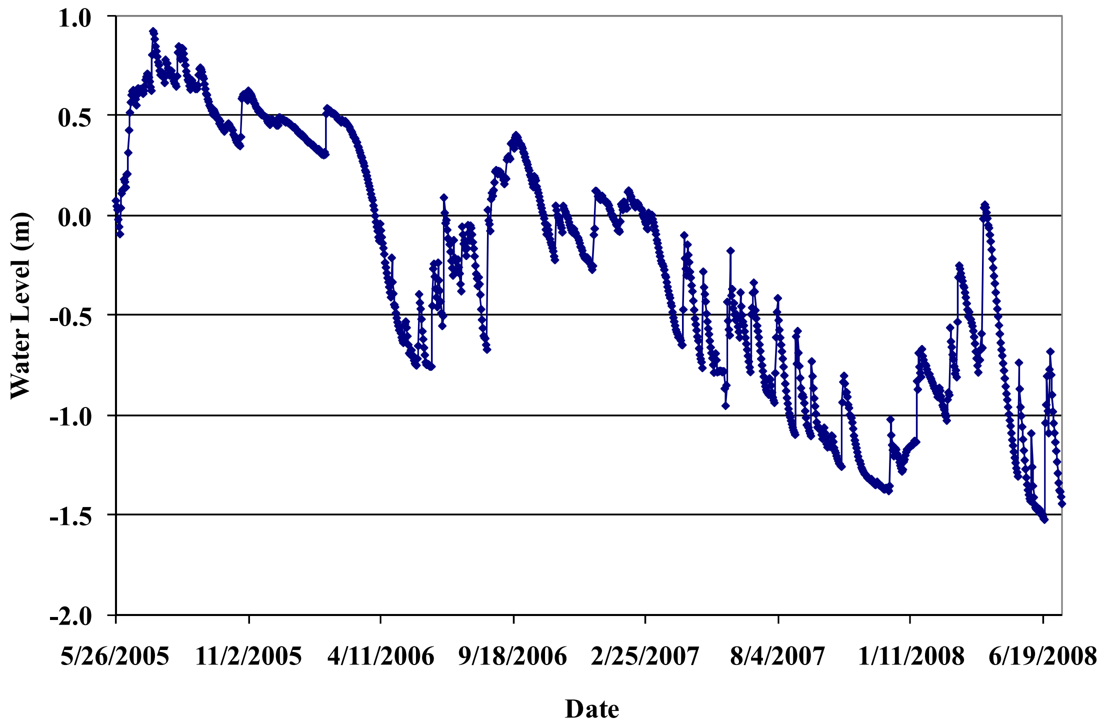


Figure 65. Surface Water Levels for Williams Co.

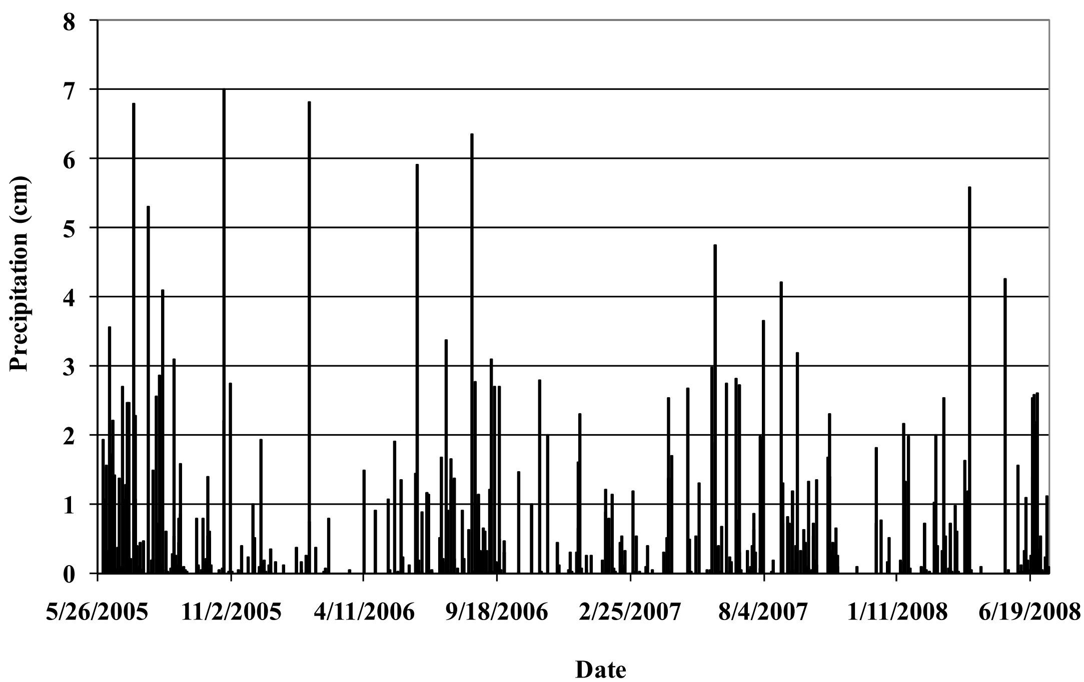


Figure 66. Precipitation for Williams Co.

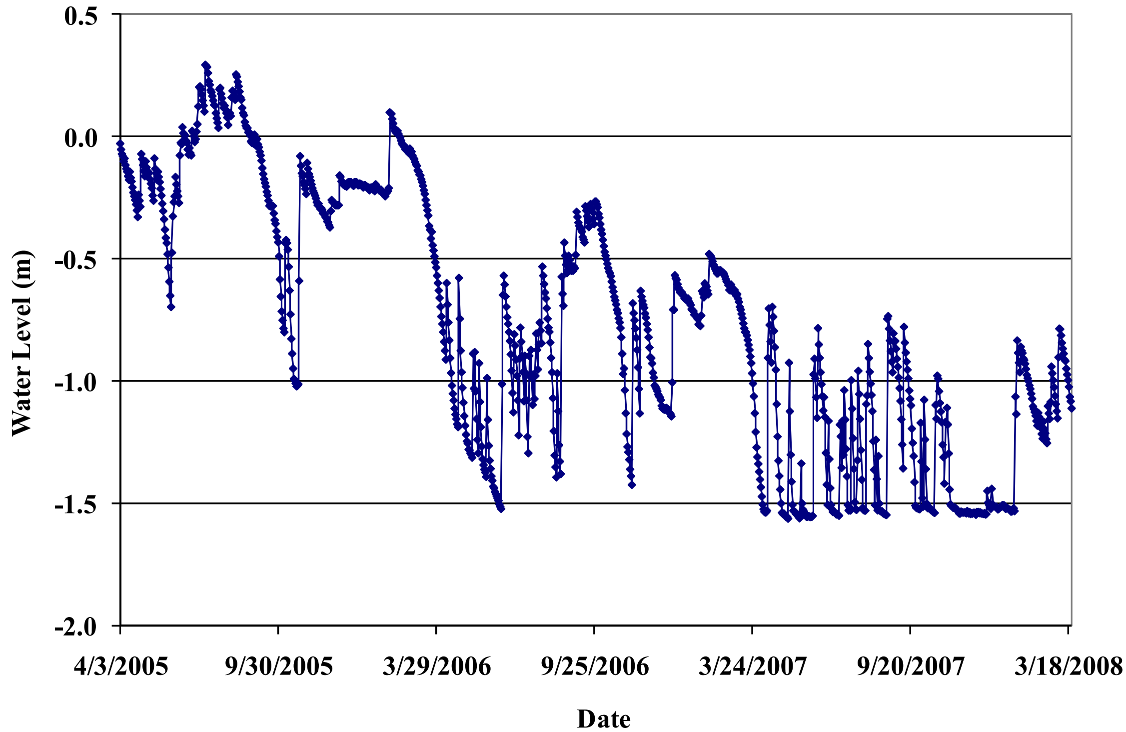


Figure 67. Surface Water Levels for Tenoroc-4.

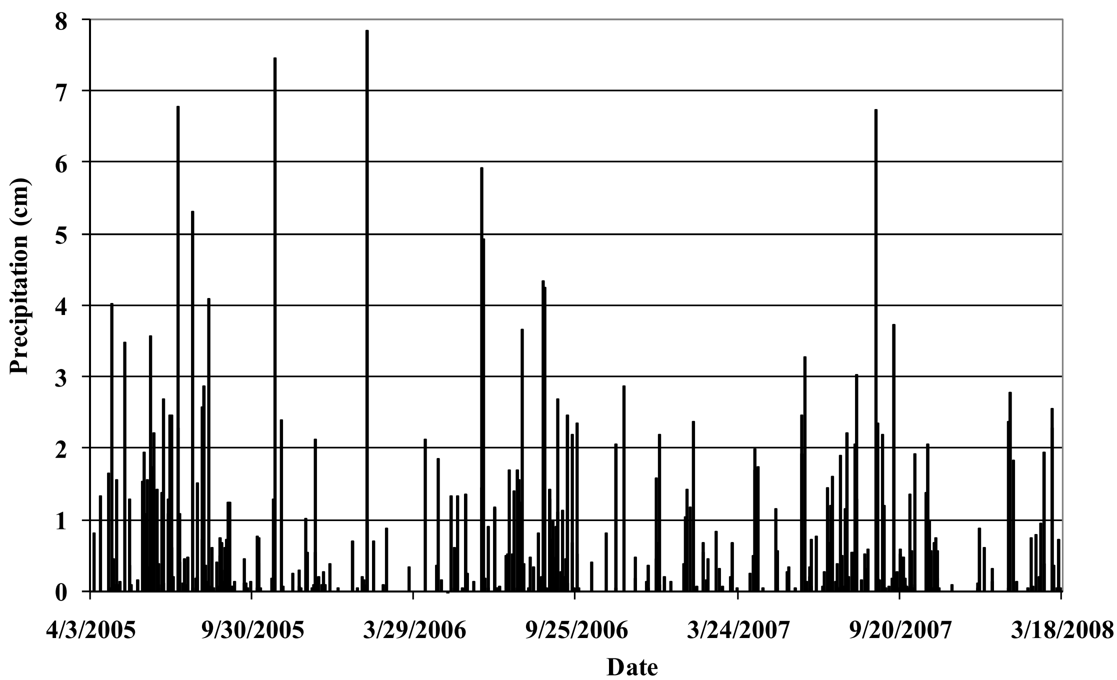


Figure 68. Precipitation for Tenoroc-4.

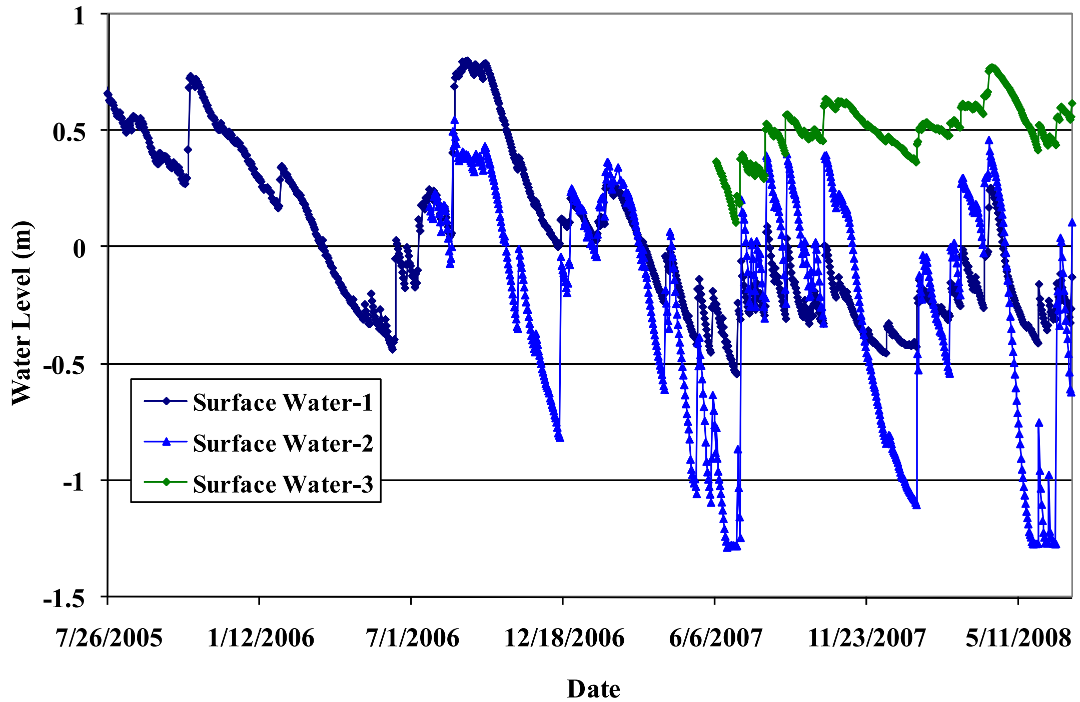


Figure 69. Surface Water Levels for Mosaic H1.

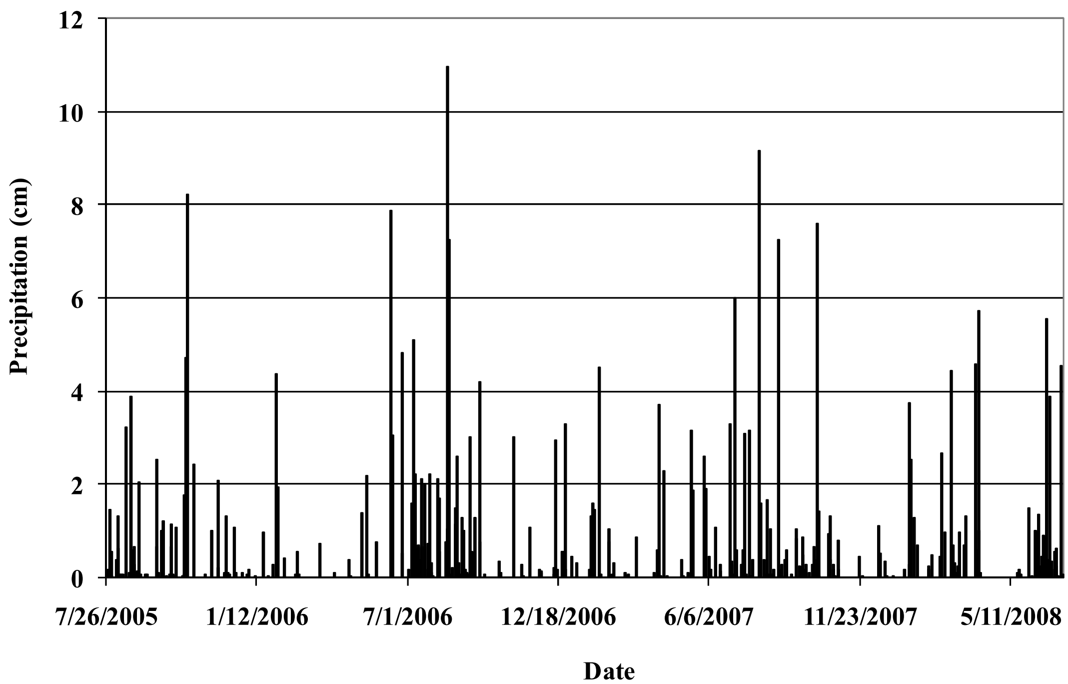


Figure 70. Precipitation for Mosaic H1.

Figure 71. Surface Water and Groundwater Levels for CFI SP-1.

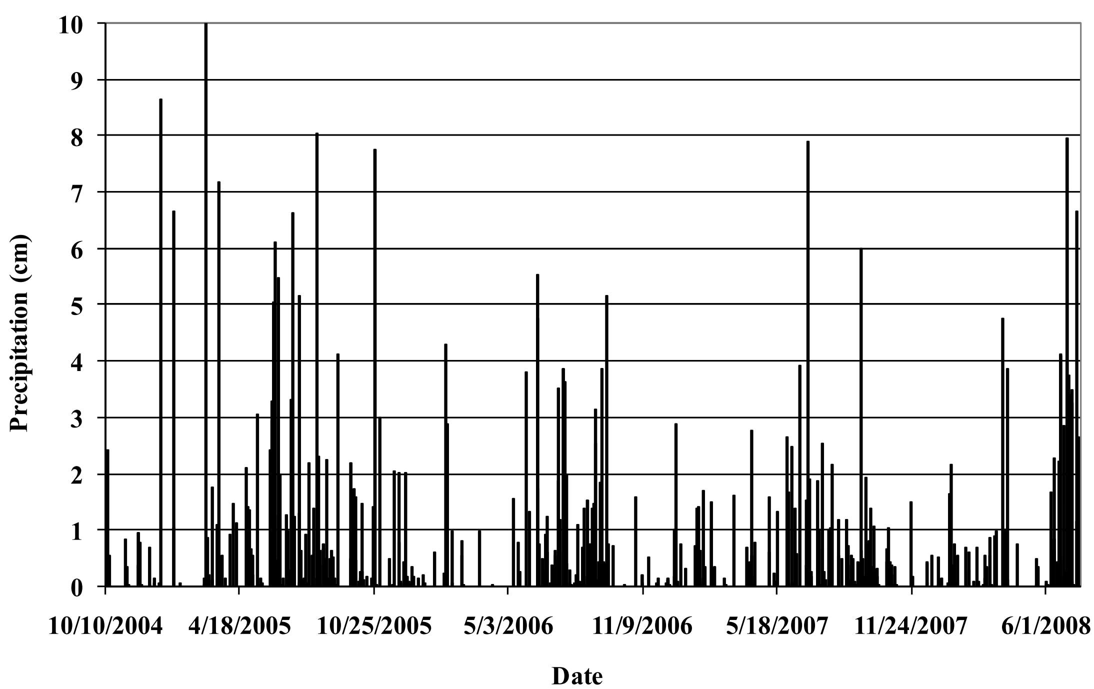


Figure 72. Precipitation for CFI SP-1.

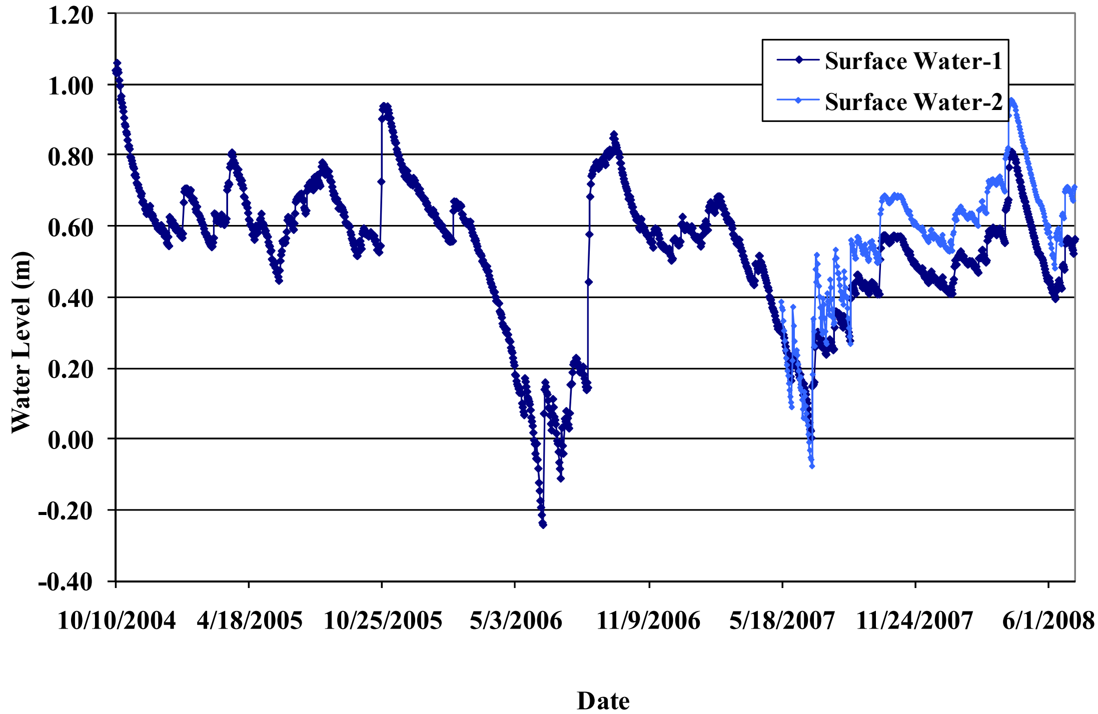


Figure 73. Surface Water Levels for Mosaic K5.

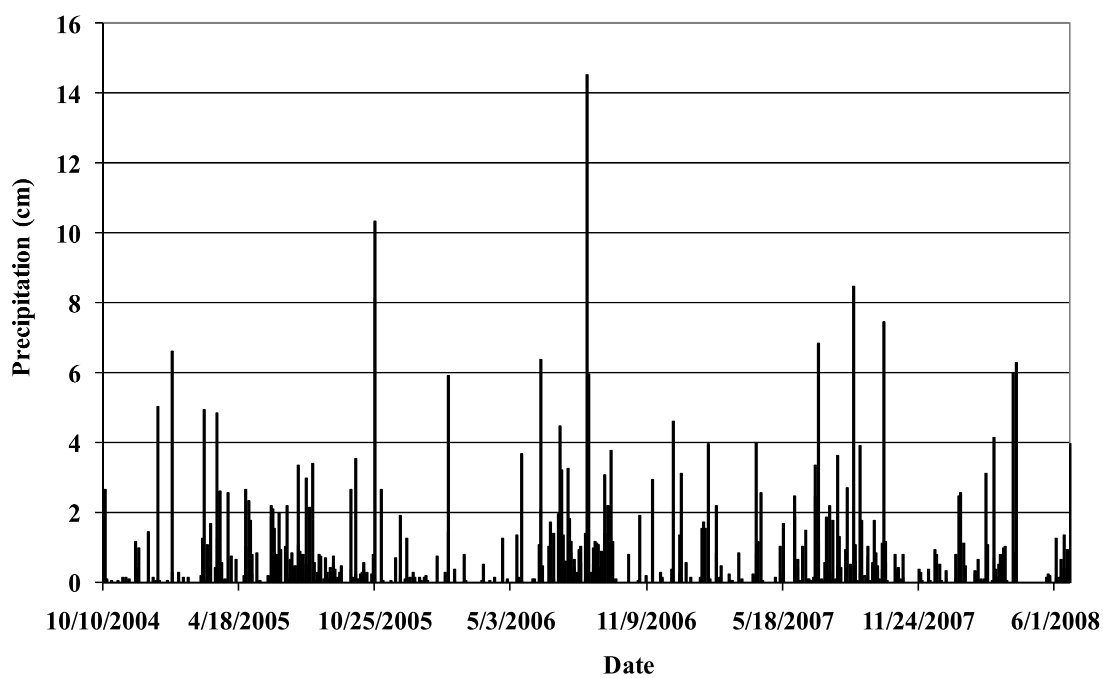


Figure 74. Precipitation for Mosaic K5.

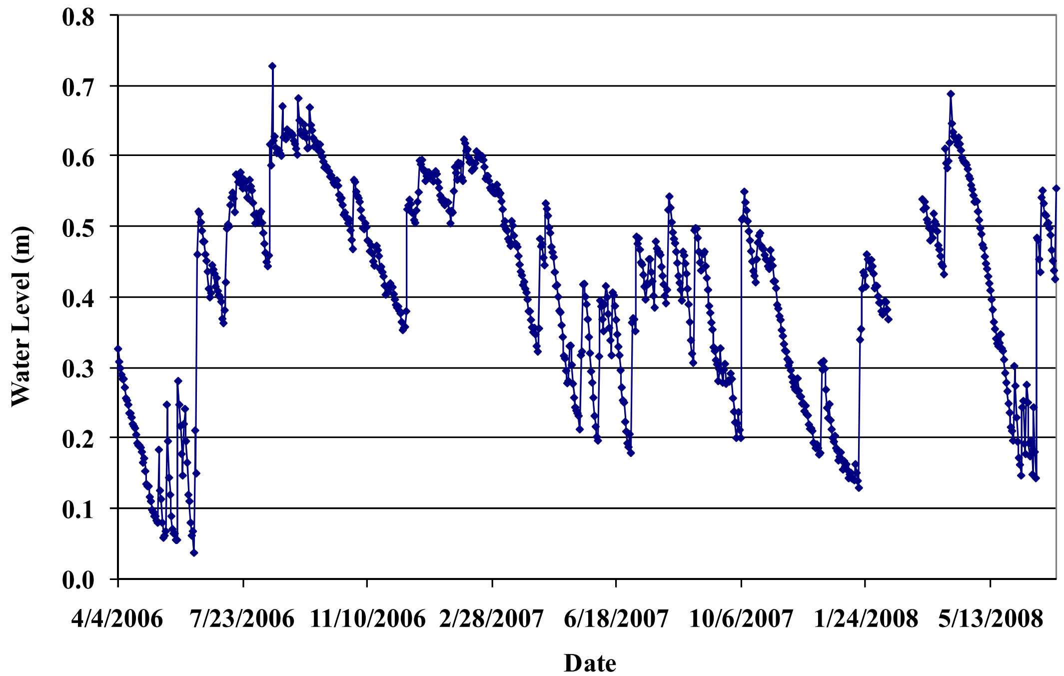


Figure 75. Surface Water Levels for Mosaic HP-10.

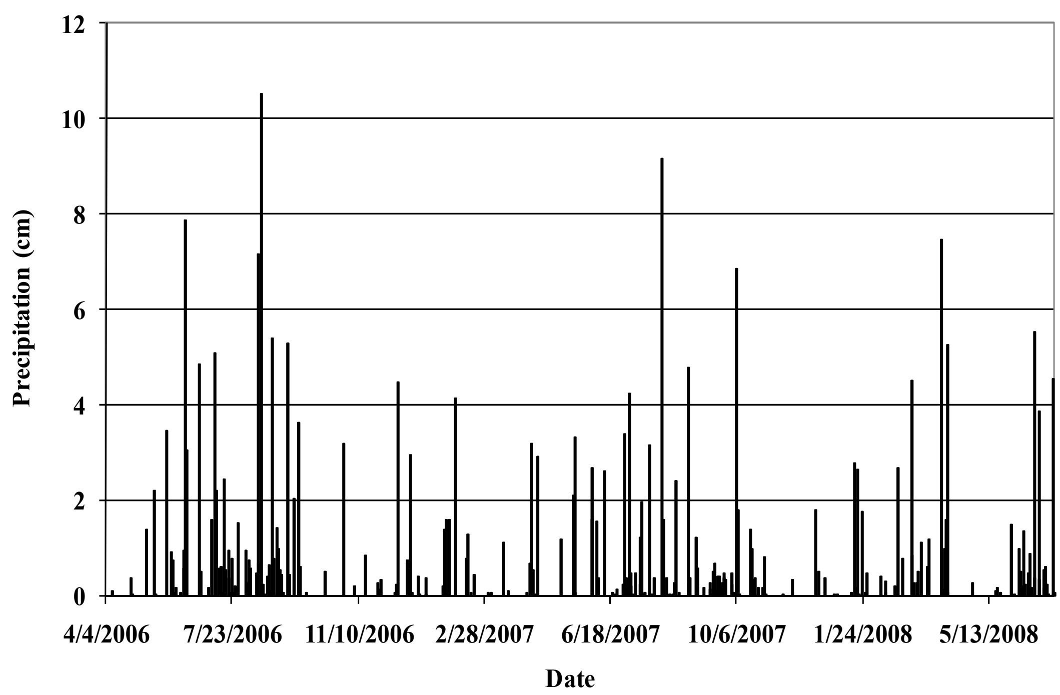


Figure 76. Precipitation for Mosaic HP-10.

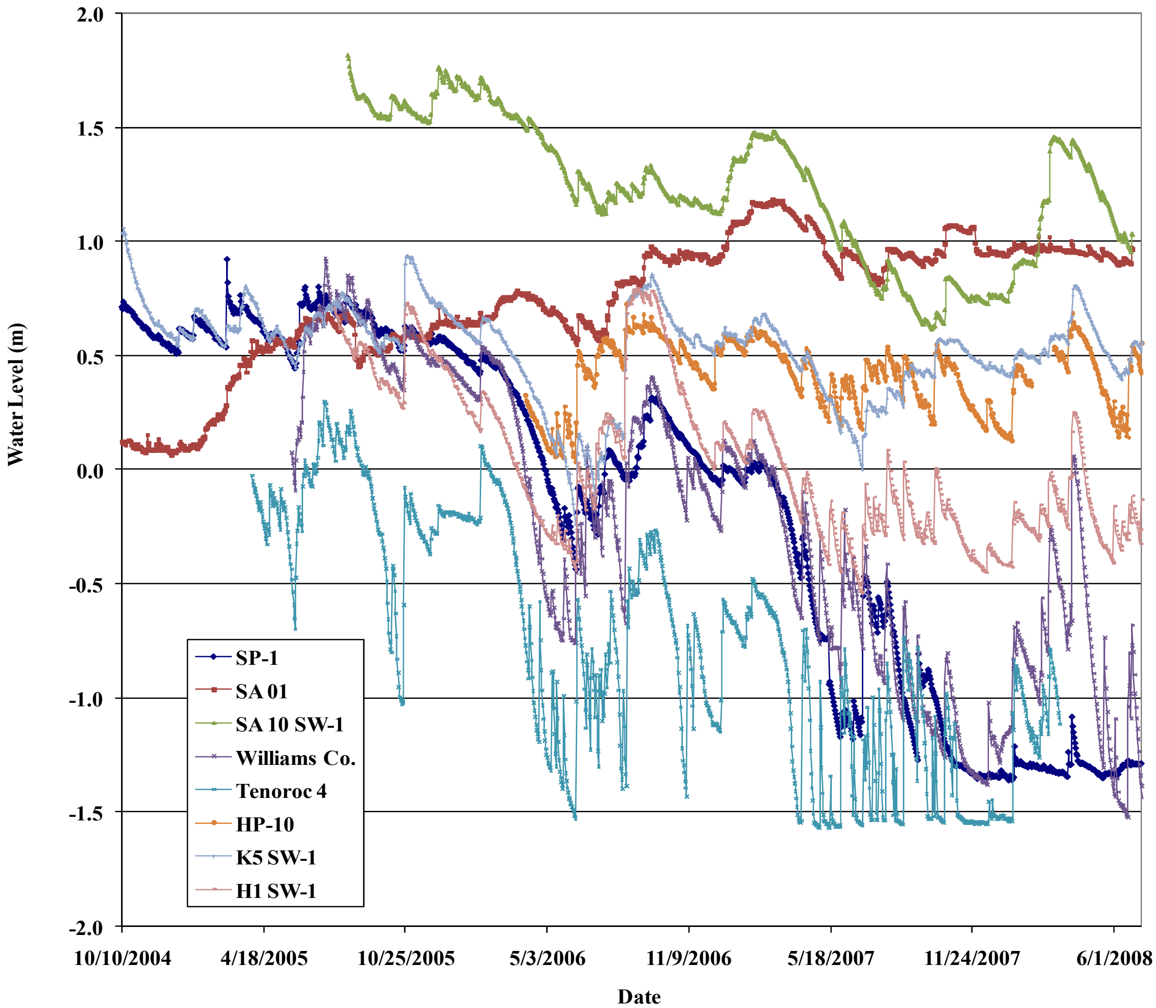


Figure 77. Surface Water Levels for All Eight Hydrology Sites.

Figure 78 compares the hydroperiods that occurred at Mosaic H1, PCS SA 10, and Williams Co. during 2006 to average hydroperiods measured at central Florida marsh and *Taxodium spp.* wetland systems (Bardi and others 2005). While the Mosaic H1 and Williams Co. systems experienced similar trends in terms of seasonal flooding and drying as the marsh and *Taxodium spp.* systems, their increases following rain events and declines were more pronounced, illustrating their flashy hydrologic regime. The buffered and deep system at PCS SA 10 had water depths much greater than typically occur in Florida wetland systems. Water levels from July 2005 to July 2007 for the same CSAs along with PCS SA 01 were compared to collected surface water level data from the Green Swamp system in central Florida (data provided by Southwest Florida Water Management District) (Figure 79). While it appears that some CSAs have more buffered and deeper systems compared to natural wetland systems, other CSA features experience similar regimes as natural wetland systems but which tend to be more fluctuating.

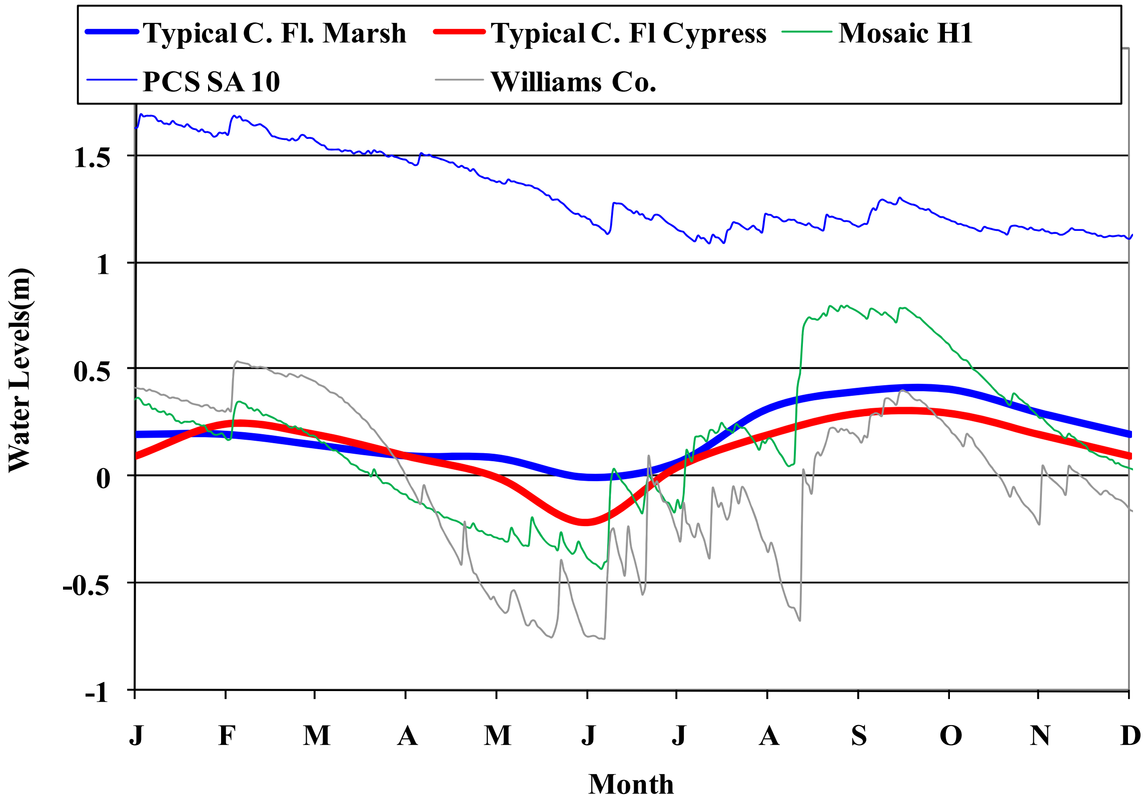


Figure 78. Comparison of Hydroperiods from PCS SA 10, Mosaic H1, and Williams Co. with Average Hydroperiods for a Typical Central Florida Marsh and Cypress Swamp.

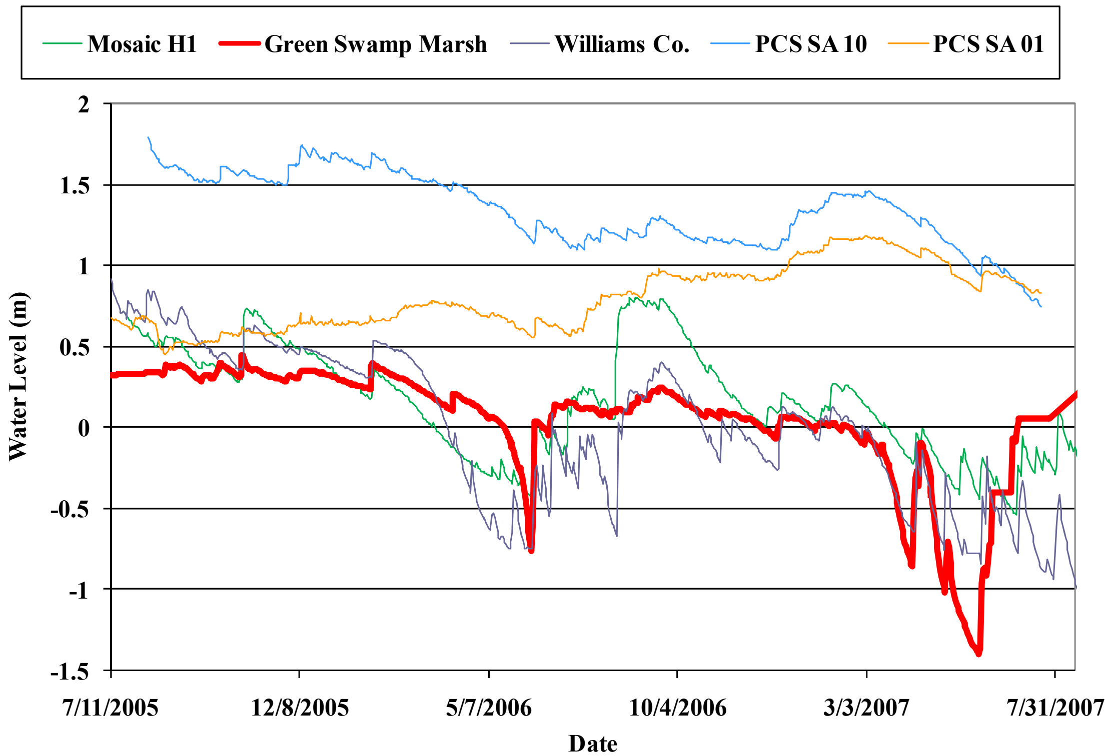


Figure 79. Comparison of Surface Water Levels from PCS SA 10, Mosaic H1,Williams Co., PCS SA 01, and Green Swamp Marsh System.

As previously mentioned, three sites had multiple surface-water features instrumented with wells (Figures 63, 69, and 73). The two surface water features instrumented at PCS SA 10 had similar rates of decline, responses to rain events, and depths when Surface Water-1 (SW-1) was over 1.0 m in depth (Figure 63). At lower levels, however, the 2nd feature (SW-2) had steeper drawdowns and increases and shallower water levels. The water level elevations at Surface Water-2 were determined in reference to the ground surface at the SW-1 well, demonstrating that indeed the two systems became connected when SW-1 had depths of or above 1.2 m (Figure 80). Both features at Mosaic K5 also experienced similar fluctuations at specific depths, around 0.4 m at SW-1, and different surface water signatures at lower water levels (Figure 73). Evaluating the water level elevation at SW-2 relative to SW-1 revealed that the two systems became connected when Surface Water-1 was at 0.39 m (Figure 81). The relationships between the two surface-water features at both PCS SA 10 and Mosaic K5 demonstrate that systems exist on CSAs that can be disconnected with different hydrologic regimes but which merge at higher water levels.

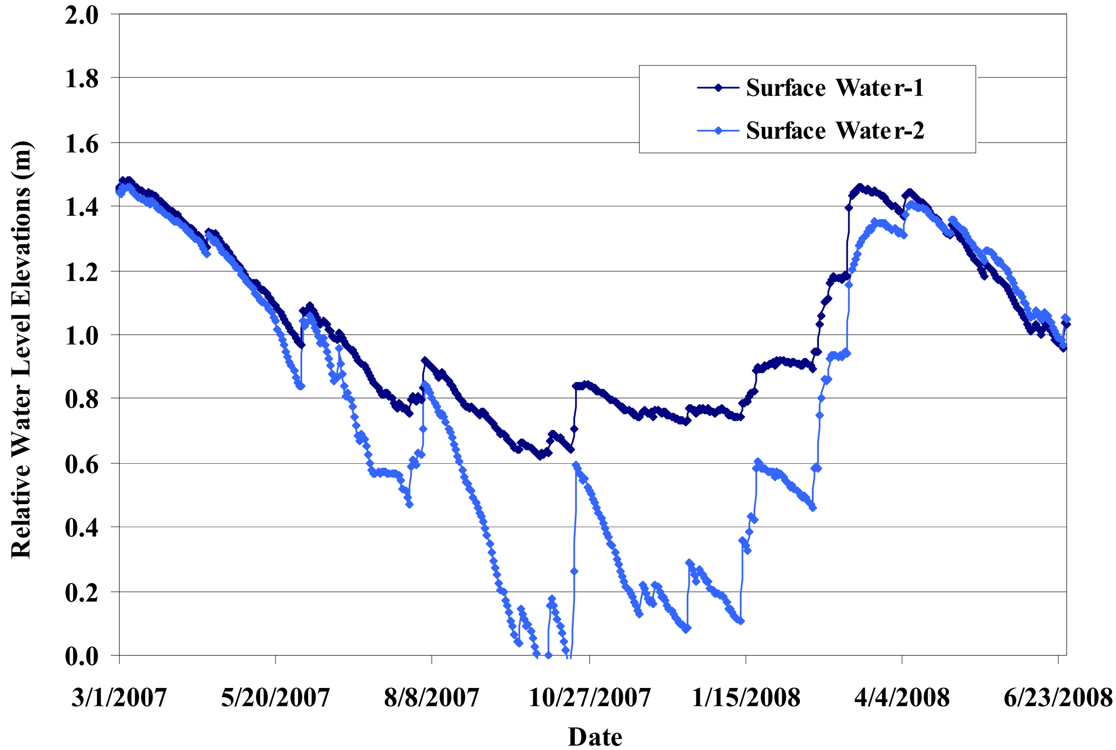


Figure 80. Surface Water Elevations Relative to SW-1 for PCS SA 10.

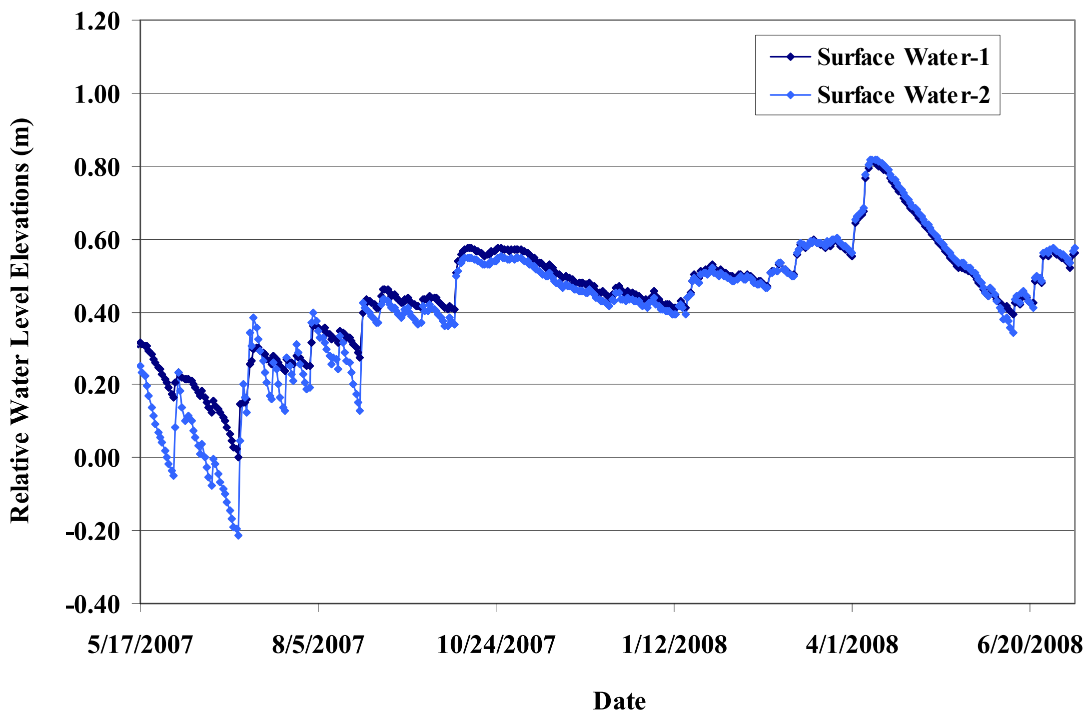


Figure 81. Surface Water Elevations Relative to SW-1 for Mosaic K5.

Three isolated surface water features were instrumented at Mosaic H1 and which had considerably different hydrologic regimes (Figure 69). Surface Water-3 (SW-3) experienced permanent flooding, a buffered hydroperiod, and greater water depths compared to the other two systems which had flashier regimes and extensive dry periods (Figure 69). The differing signatures illustrate that these three systems never became connected during the period of study and this is more evident when expressing SW-2 and SW-3 water level elevations relative to SW-1 (Figure 82). From the relative elevations, it appears that SW-2 and SW-3 are separate perched systems suggesting that the level pool assumption should not be applied across an entire CSA. Furthermore, these results reveal that not only does hydrologic regime vary among CSAs but also among individual surface water features within a single CSA.

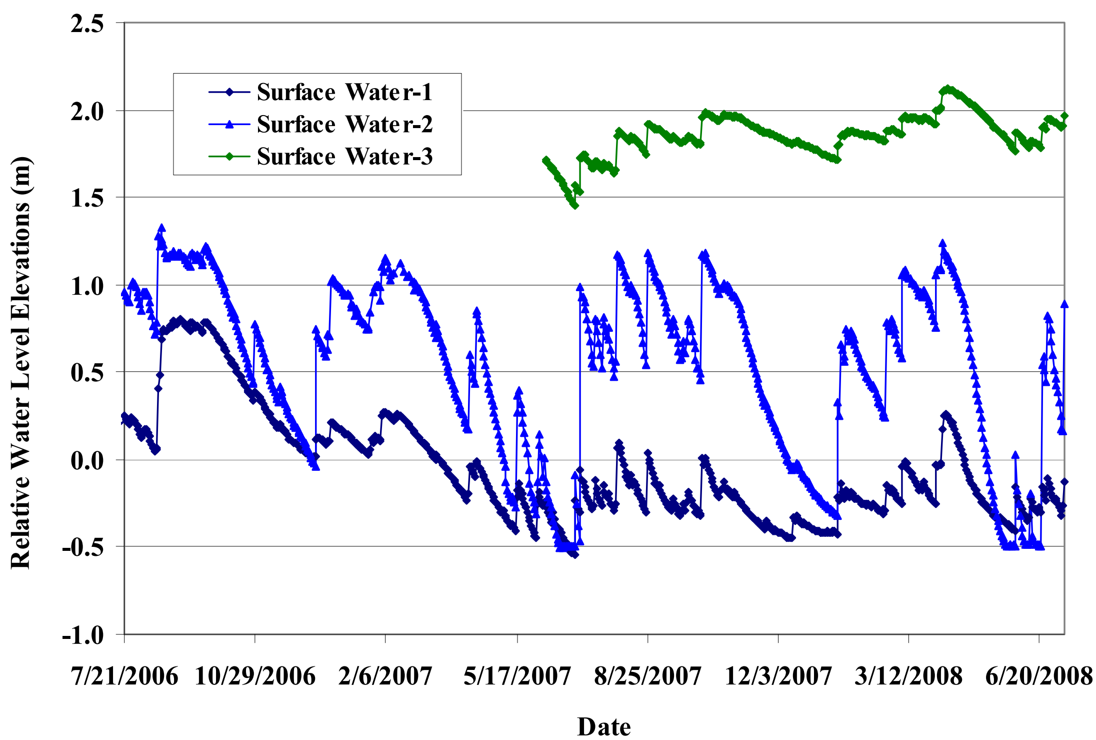


Figure 82. Surface Water Elevations Relative to Surface Water-1 for Mosaic H1.

Spatial Modeling of CSA Hydrology

The CSA spatial hydrology model results are split into four sections: watershed delineation maps, maximum and average water depth maps, calibration graphs and model coefficients, and hydropattern maps.

Watershed Delineation Maps

Mosaic H1. For the CSA H1 there were two monitored surface water features that were modeled, referred to as Surface Water one (SW-1) and Surface Water two (SW-2). For each surface-water feature the watershed scale was used that best matched the observed boundaries of the feature. For SW-1 the optimal watershed size was generated using a z-limit of 25 cm, whereas for SW-2 it was at a z-limit of 40 cm (Figures 83 and 84**)**. Regardless of the watershed delineation scale, H1 was a CSA that contained many small watersheds, as compared to some CSAs that were dominated by a single large watershed. Out of the 88 watersheds delineated using the 25-cm z-limit, only 18 were larger than 1 ha, whereas 10 watersheds were greater than 1 ha out of 28 watersheds total for the 40-cm z-limit.

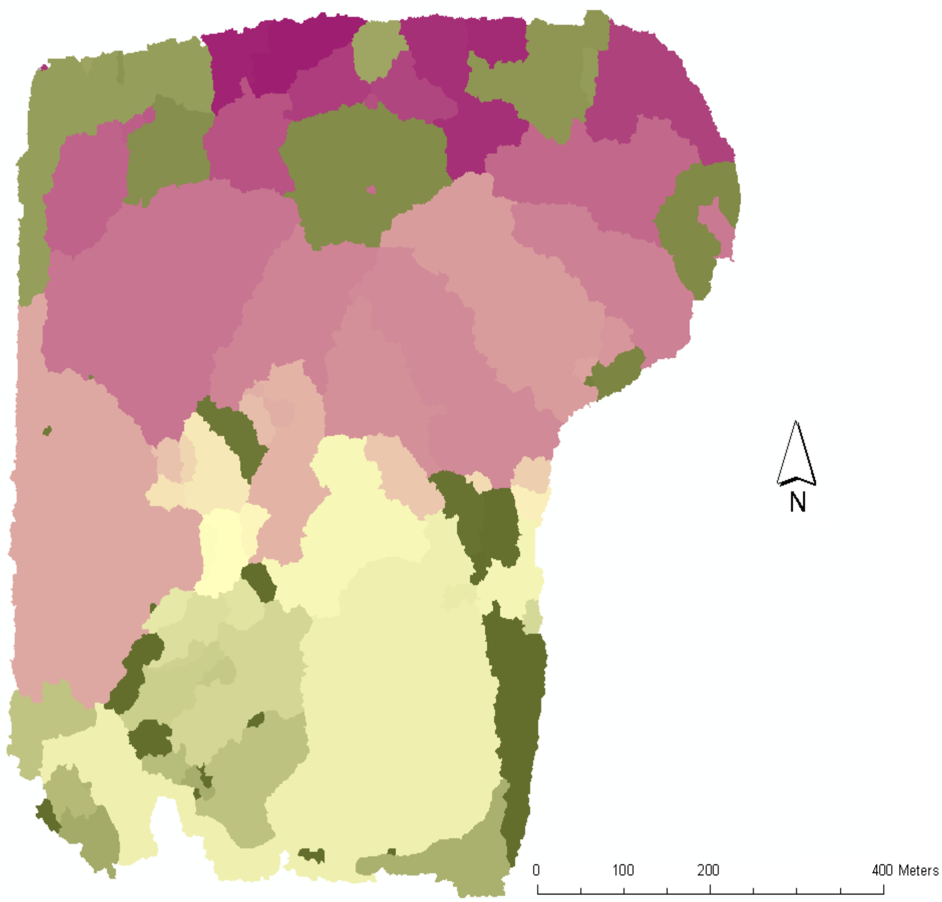


Figure 83. 88 Watersheds Delineated in H1 (for SW-1) with a 25-cm Z-limit.

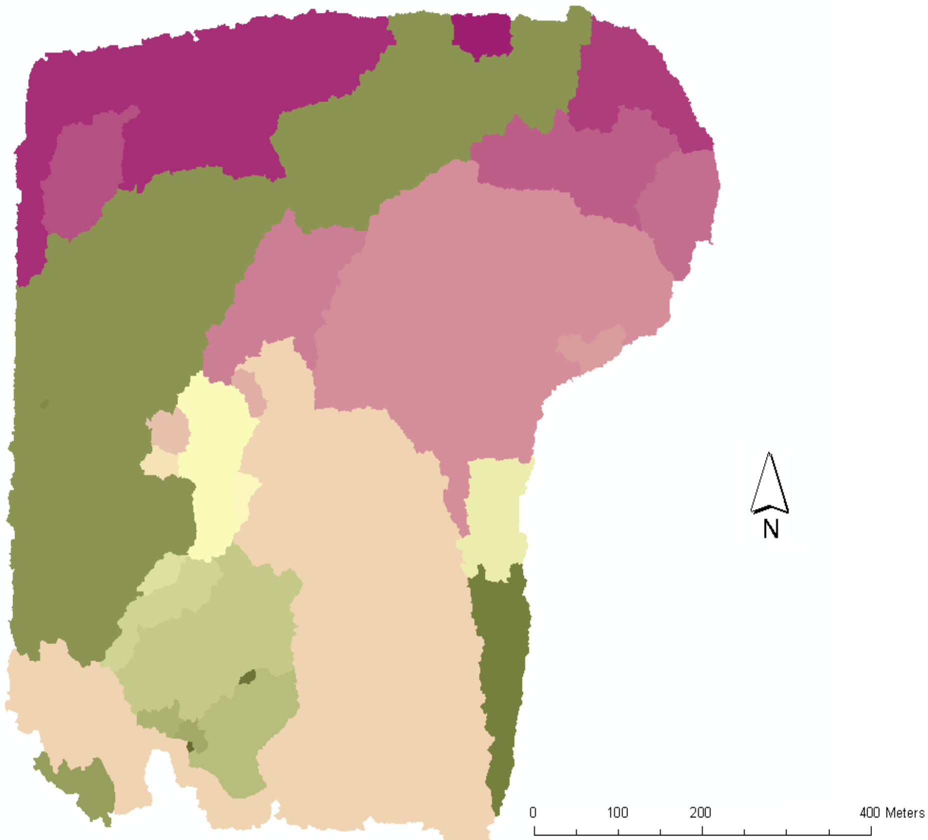


Figure 84. 28 Watersheds Delineated in H1 (for SW-2) with a 40-cm Z-limit.

PCS SA 01. SA 01 was separated into 33 watersheds using a z-limit of 50 cm, including 12 watersheds that had an area greater than 1 ha (Figure 85). For this site the two large watersheds at the southern end were known to contain surface water features, and the southwestern watershed was monitored.

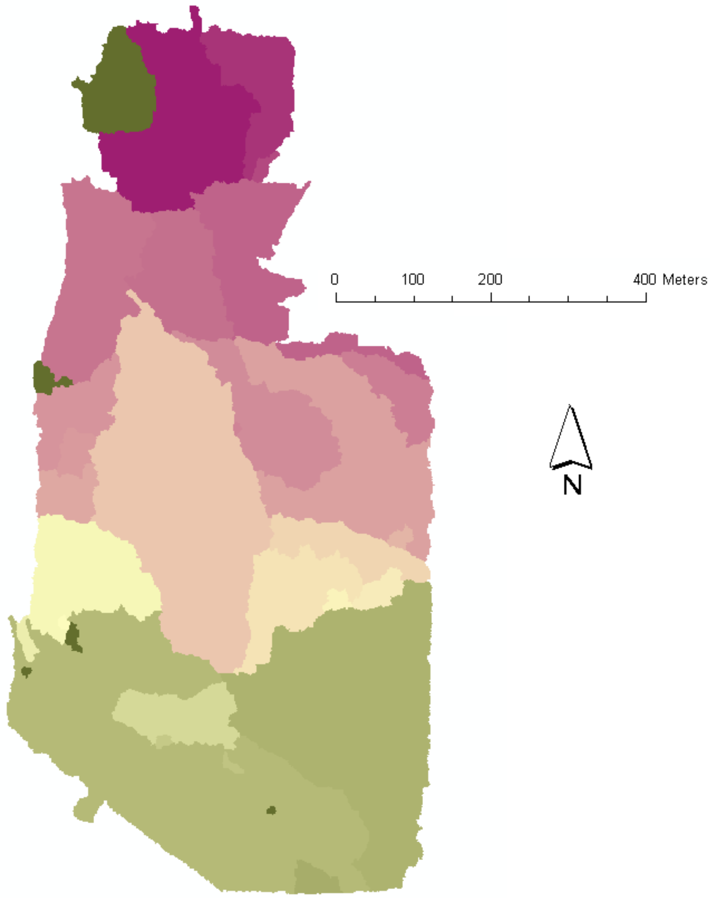


Figure 85. 33 Watersheds Delineated in SA 01 with a 50-cm Z-limit.

PCS SA 10. SA 10 was separated into 101 watersheds using a z-limit of 40 cm, but only nine watersheds had an area greater than 1 ha (Figure 86). The majority of this site was delineated as one large watershed, which was known to contain surface water throughout most of the eastern portion.

Figure 86. 101 Watersheds Delineated in SA 10 with a 40-cm Z-limit.

Mosaic K5. K5 was separated into six watersheds using a z-limit of 60 cm, and two of those watersheds had an area greater than 1 ha (Figure 87). The vast majority of this site was delineated as one large watershed, which contained surface water throughout most of the southern half. The other five watersheds were parts of a ditch that surrounded the CSA on the east and south sides.

Figure 87. Six Watersheds Delineated in K5 with a 60-cm Z-limit.

CF Industries SP-1. SP-1 was separated into five watersheds using a z-limit of 60 cm, and all five of those watersheds had an area greater than 1 ha (Figure 88). The site contained two larger watersheds, both of which contained surface water over the modeled period and were connected in the middle by a ditch.

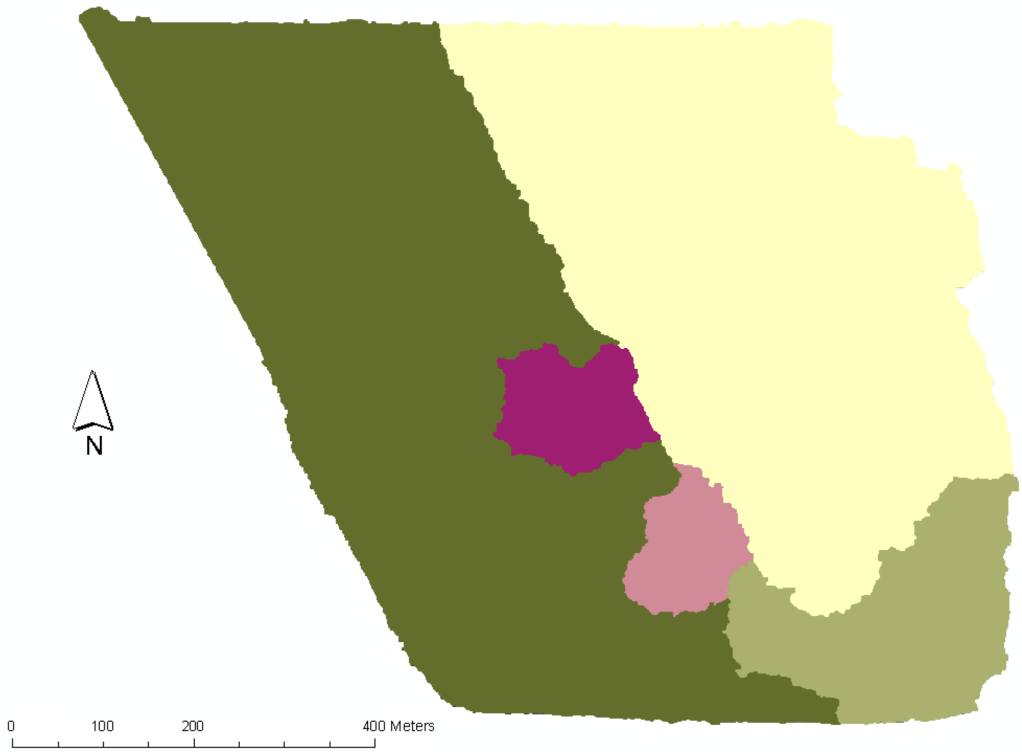


Figure 88. Five Watersheds Delineated in SP-1 with a 60-cm Z-limit.

Williams Co. The Williams Co. CSA was separated into 199 watersheds using a z-limit of 40 cm, and 46 of those watersheds had an area greater than 1 ha (Figure 89). There were many linear features throughout this CSA due to mining techniques, some of which contained water. The larger watersheds on the north side were known to hold water periodically.

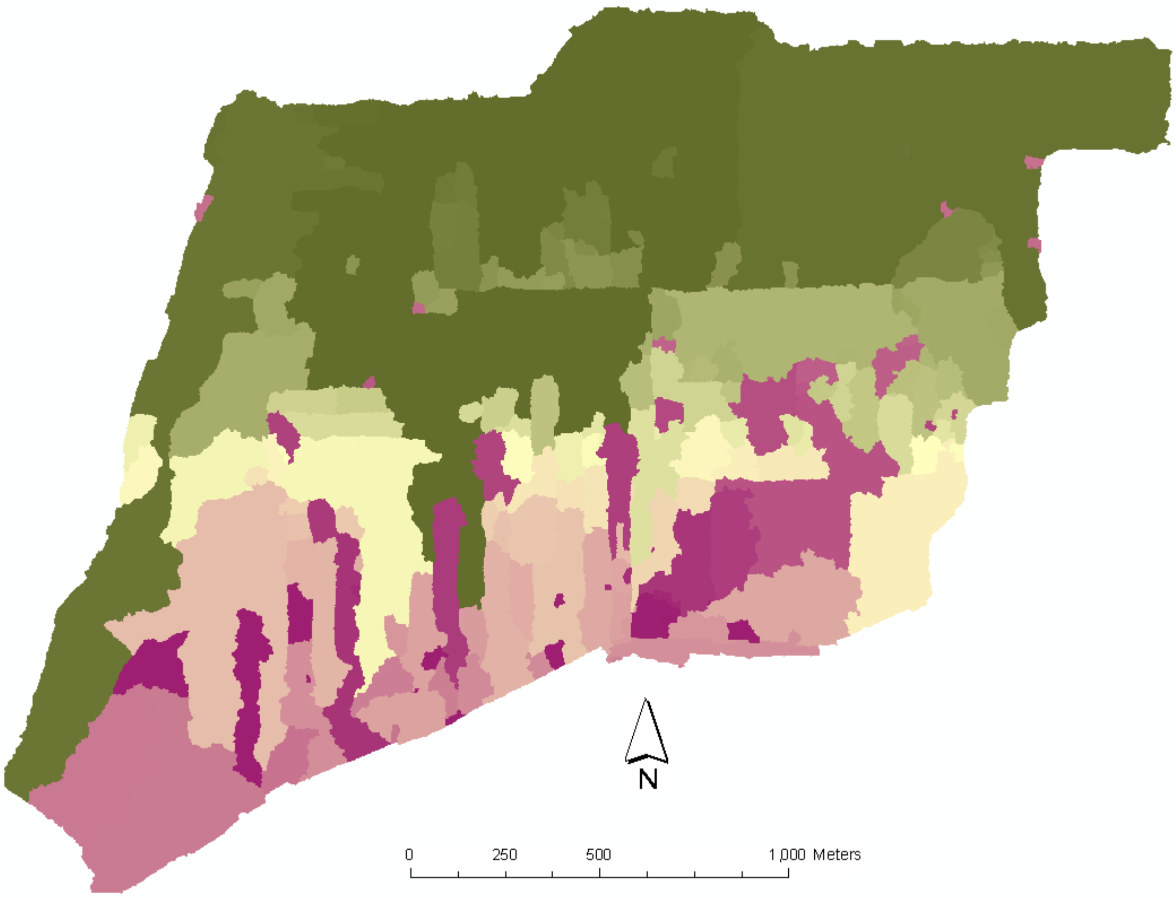


Figure 89. 199 Watersheds Delineated in Williams Co. with a 40-cm Z-limit.

Water Depth Maps

Mosaic H1. H1 SW-1 and SW-2 were the only watersheds that were able to be accurately modeled for the entire monitoring period because they were the only monitored watersheds that were dry when the LiDAR was flown, and therefore their topography (or bathymetry) was known. For the watershed SW-1, maximum and average water depth maps were created (Figures 90 and 91). The maximum water depth map shows that much of the watershed was inundated with a depth of 80 cm at the surface water well on 9/2/06. This map also shows possible connections to adjacent watersheds at multiple locations as shown by the presence of water at the watershed boundary. The average water depth map shows what the watershed looked like when there was a minimal amount of surface water, since the average measured water depth was 4 cm at the surface water well. H1 SW-2 had distinct water features within its watershed which combined at higher water levels such as were modeled in the maximum water depth map that was based on 55 cm at the surface water well (Figure 92). The average water depth map was not able to be created because the average water depth was below ground.

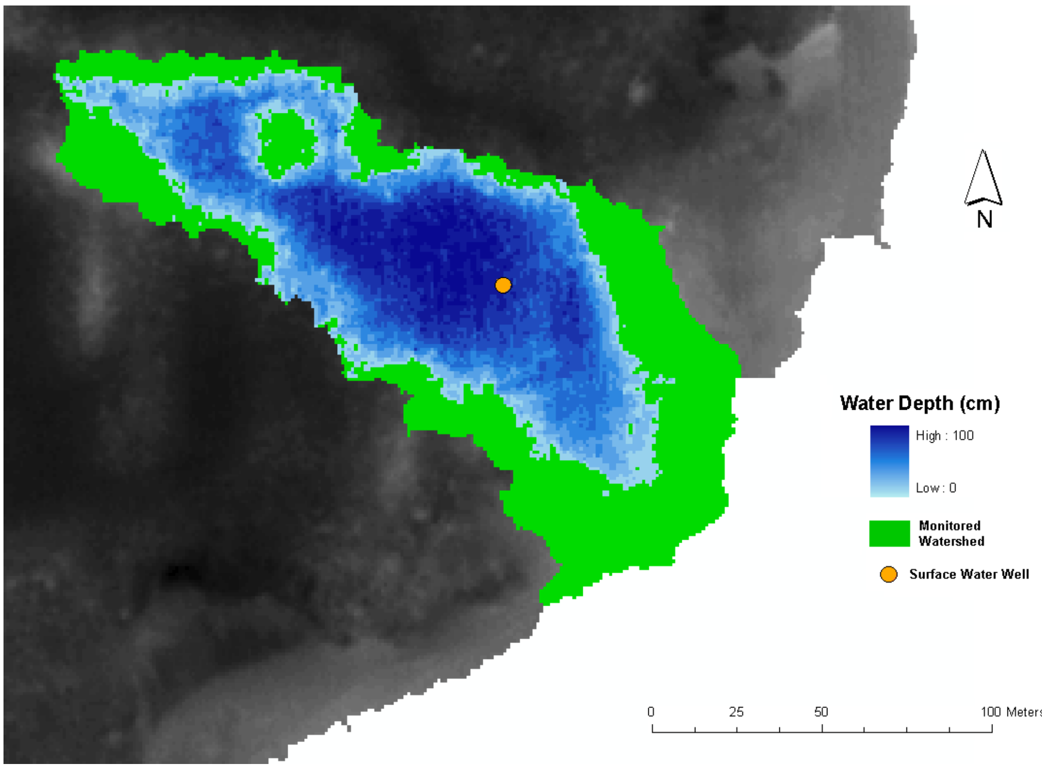


Figure 90. Maximum Water Depth Map for H1 SW-1 on 9/2/06.

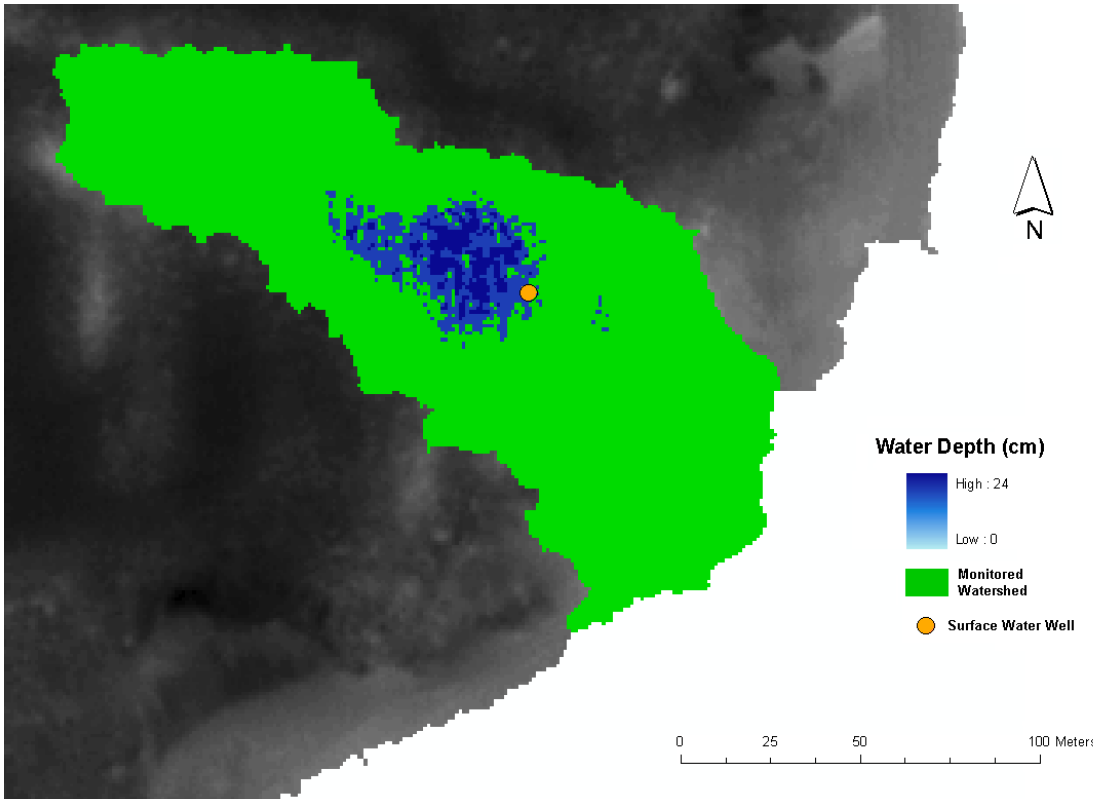


Figure 91. Average Water Depth Map for H1 SW-1 from 7/26/05 to 7/9/08.

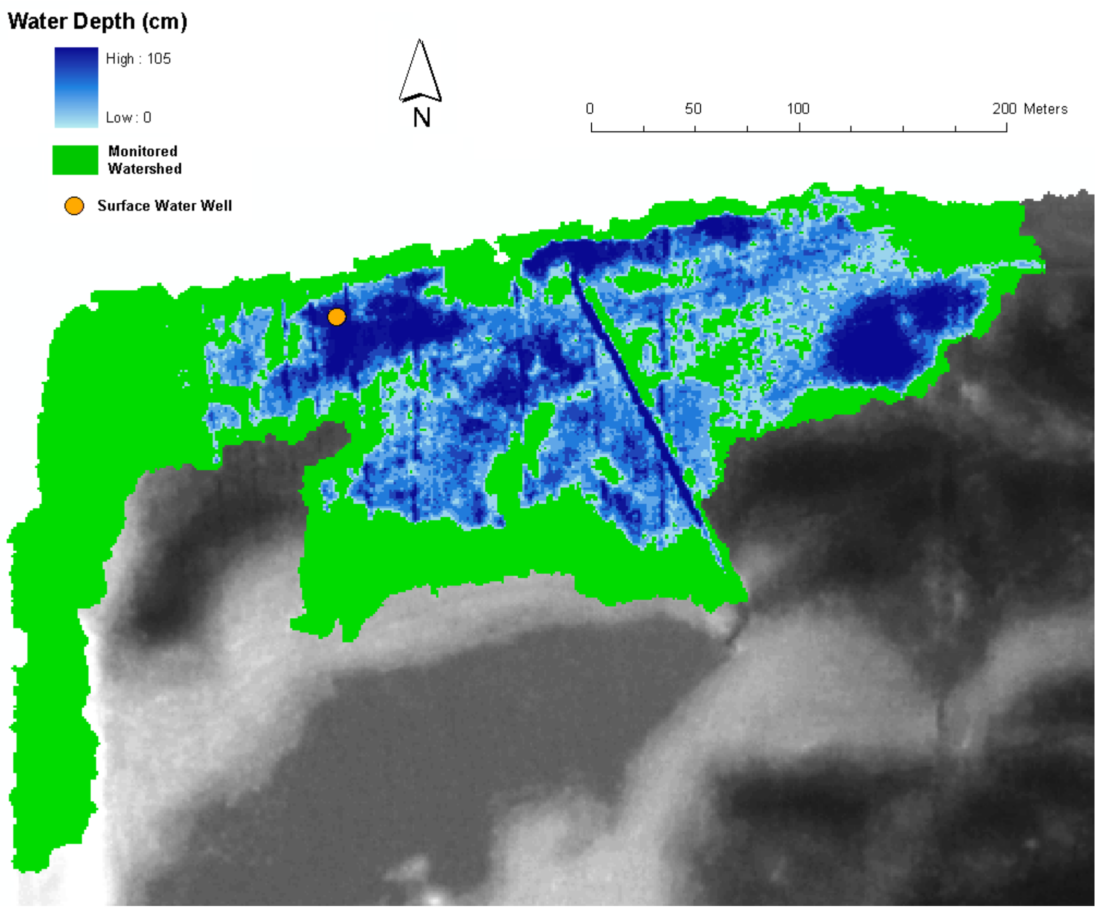


Figure 92. Maximum Water Depth Map for H1 SW-2 on 8/18/06.

PCS SA 01. For SA 01, the southwestern watershed was monitored for surface water levels, and maximum and average water depth maps were created for that water feature (Figures 93 and 94). The average water depth map had a patchy appearance due to the average water level measured (76 cm) being about equal to the 76 cm water level when the LiDAR was flown. Therefore, what should mostly be a continuous water surface was not captured due to errors in the DEM, possibly as a result of dense canopy in that area. Also the legend of each map lists the greatest depths at 112 cm and 71 cm for the maximum and average depth maps, respectively, but it should be noted that these values are relative to the water surface when LiDAR was flown, which was 76 cm at the surface water well. Therefore in these maps, the areas that contained standing water when LiDAR was flown represent water depths above this water elevation, and the actual water depth in these areas may be greater.

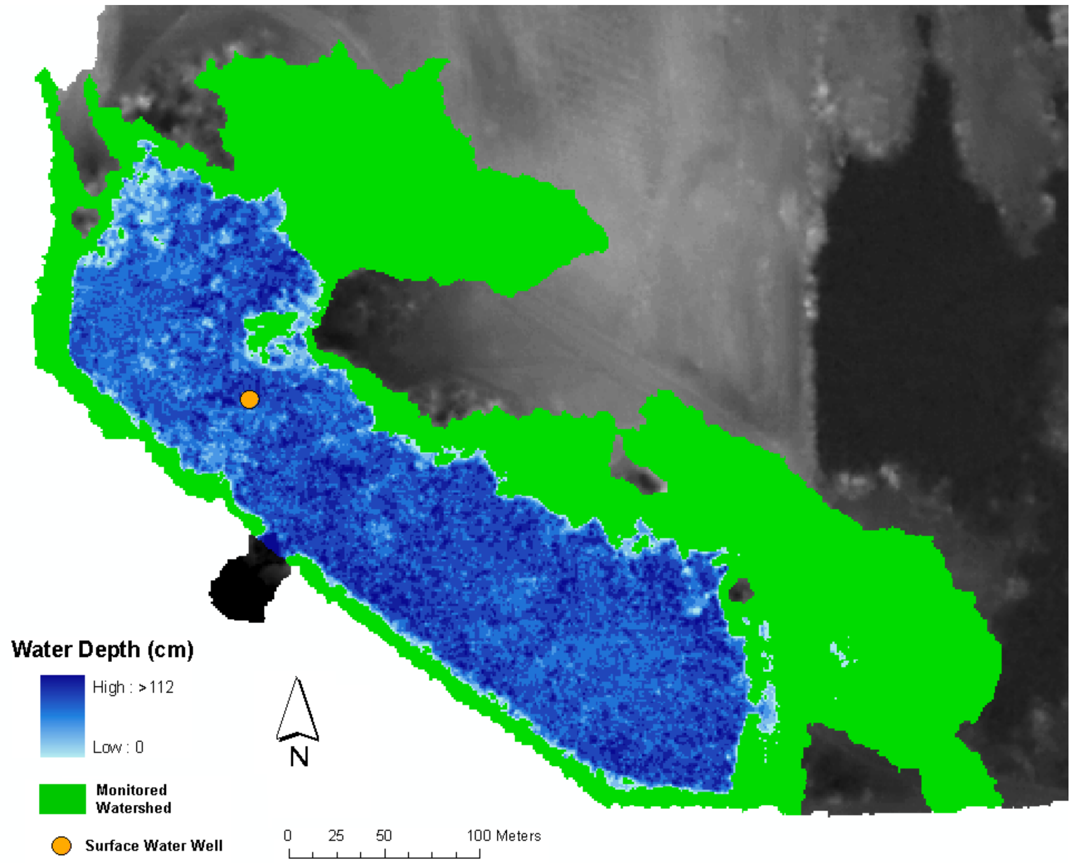


Figure 93. Maximum Water Depth Map for SA 01 on 3/2/07.

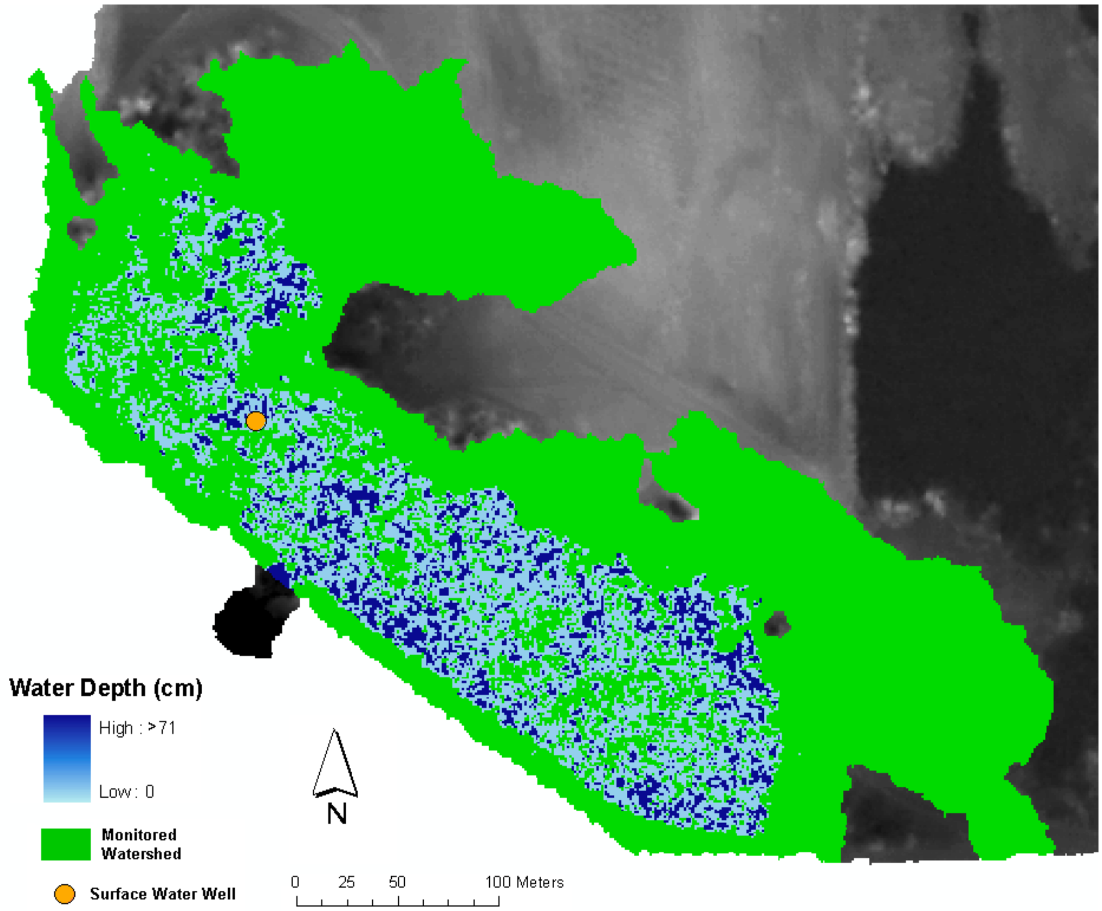


Figure 94. Average Water Depth Map for SA 01 from 10/2/04 to 6/27/08.

PCS SA 10. The large eastern watershed was monitored in SA 10, for which a maximum water depth map was created (Figure 95). The average water depth map was not able to be generated because the average water level (123 cm) was below the 151 cm water level when LiDAR was flown. The maximum water depth map shows that there was a significant portion of the watershed that was much deeper than the rest of the watershed, and that it is periodically hydraulically disconnected from the rest of the surface water in the watershed. The actual water depths throughout much of the monitored watershed were unknown since it contained standing water when LiDAR was flown. Therefore the water depths listed in the legend were relative to the water surface when LiDAR was flown, which was 151 cm at the surface water well. Based on what was known, it can be concluded that some of this watershed contained water in excess of 182 cm in depth, making it the consistently deepest watershed monitored.

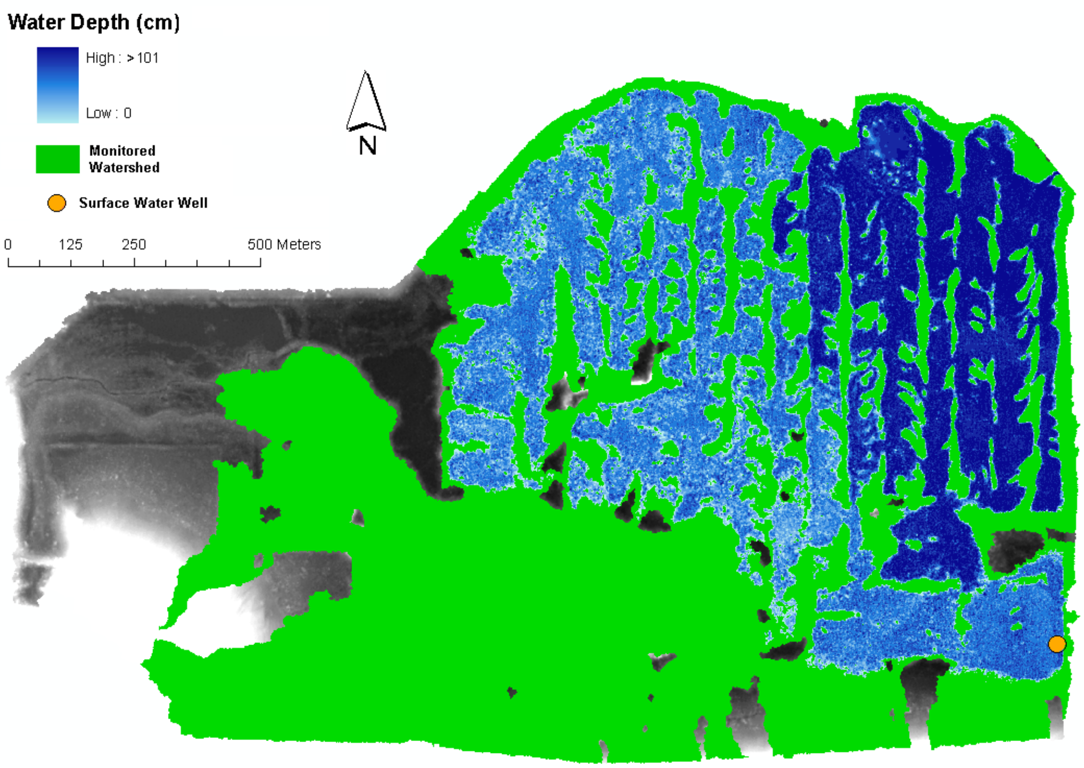


Figure 95. Maximum Water Depth Map for SA 10 on 8/10/05.

Mosaic K5. The monitored watershed for K5 included most of the CSA, for which maximum and average water depth maps were created (Figures 96 and 97). The maximum water depth map shows standing water throughout most of the watershed except the northern portion. The water depth varied with shallow areas in the north and west portions of the watershed and deeper areas in the south and east portions. The average water depth map shows that much of the shallow area in the maximum depth map dried out at lower water levels. Linear features due to mining techniques were apparent throughout the center of the watershed, particularly on the shallower west side. The connection between the main water feature and the outfall ditch can be seen in the southwest corner of the watershed. The greatest water depth modeled was 355 cm but this was due to inclusion of the outfall area in the watershed (as delineated by the GIS) which was at a much lower elevation than the rest of the CSA, and therefore does not reflect depths throughout most of the site. This was also the case for the average water depth map. The greatest depth not including the outfall area was 115 cm for the maximum water depth map, and 62 cm for the average water depth map. These depths did not take into account the water level when the LiDAR was flown, which was 41 cm at the surface water well, so some areas had depths greater than those just mentioned.

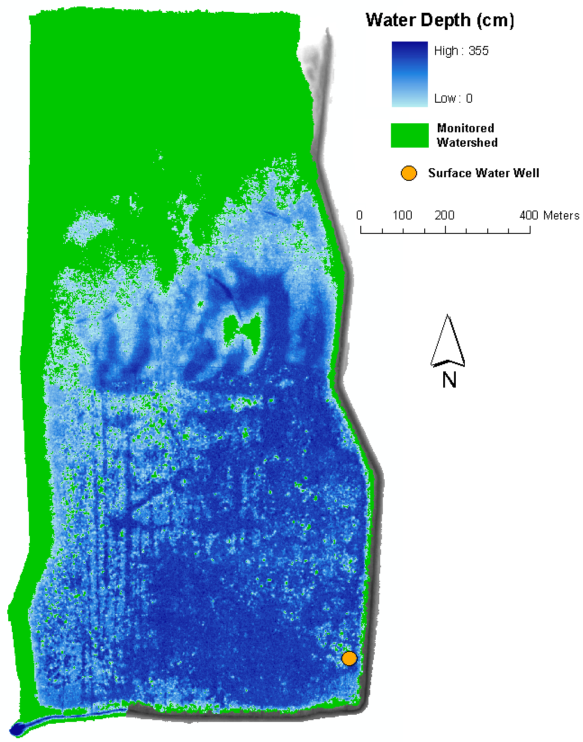


Figure 96. Maximum Water Depth Map for K5 on 10/12/04.

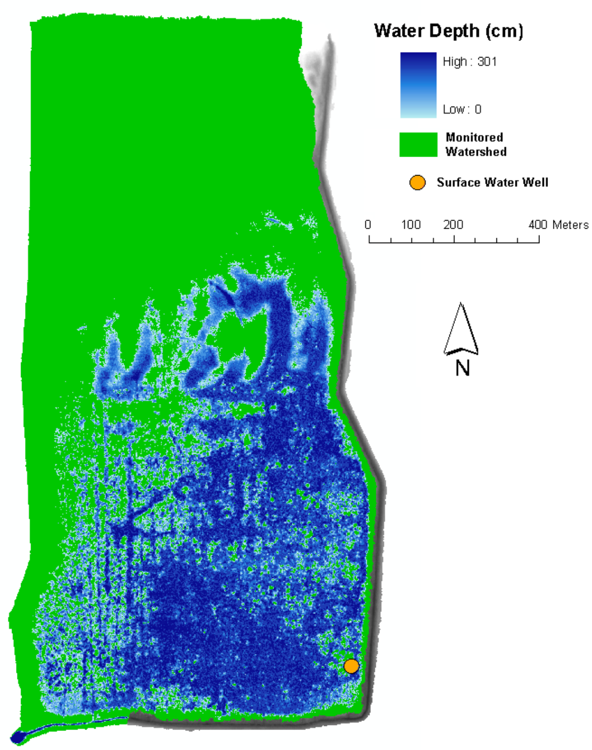


Figure 97. Average Water Depth Map for K5 from 10/10/04 to 7/10/08.

CF Industries SP-1. The eastern lobe of SP-1 was monitored for surface water levels, for which the maximum water depth map was created (Figure 98). The average water depth map was not able to be generated because the average water level during the monitoring period was below ground. The water depths at the maximum water level measured did not take into account the water level when the LiDAR was flown, which was 25 cm at the surface-water well.

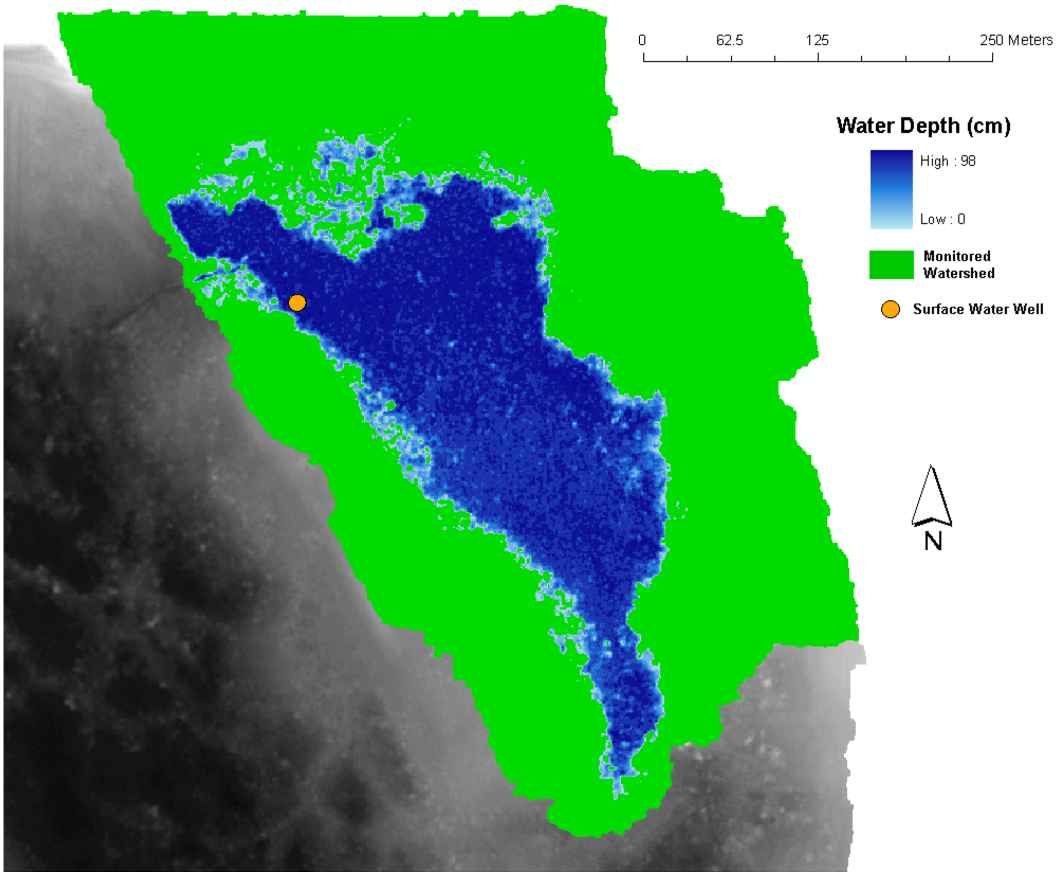


Figure 98. Maximum Water Depth Map for SP-1 on 2/28/05.

Williams Co. The northwestern corner of Williams Co. contained a surface-water feature that was monitored, for which the maximum water depth map was created (Figure 99). The irregularly elongated shape of this watershed was due to a ditch that was created to provide water for cattle, that intermittently drains into the water feature. The map shows possible connections to adjacent watersheds at multiple locations as shown by the presence of water at the watershed boundary in the northeastern section. The average water depth map was not able to be generated because the average water level during the monitoring period was below ground. These water depths at the maximum water level measured did not take into account the water level when the LiDAR was flown, which was 25 cm at the surface water well.

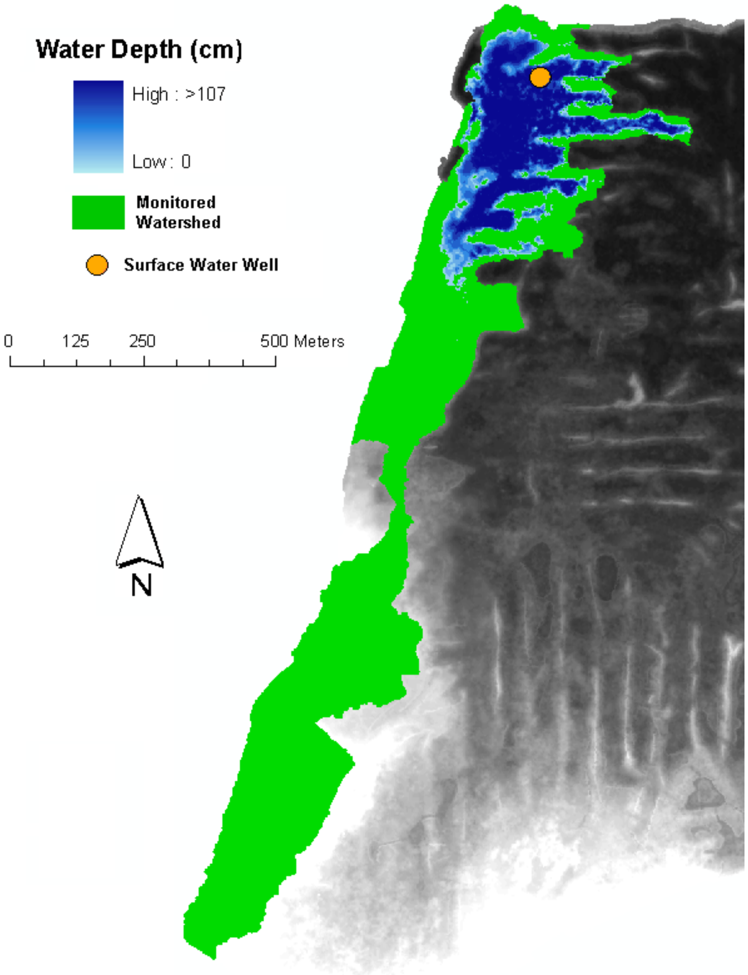


Figure 99. Maximum Water Depth Map for Williams Co. on 7/10/05.

Calibration

Mosaic H1. H1 SW-1 was simulated from 7/26/05 to 7/9/08 except for periods when the water level receded below ground (Figure 100). This model matched the measured surface water levels better than any other watershed modeled (R^2 = 0.95). The SW-1 model included three different simulation periods because water levels receded below ground before rising again.

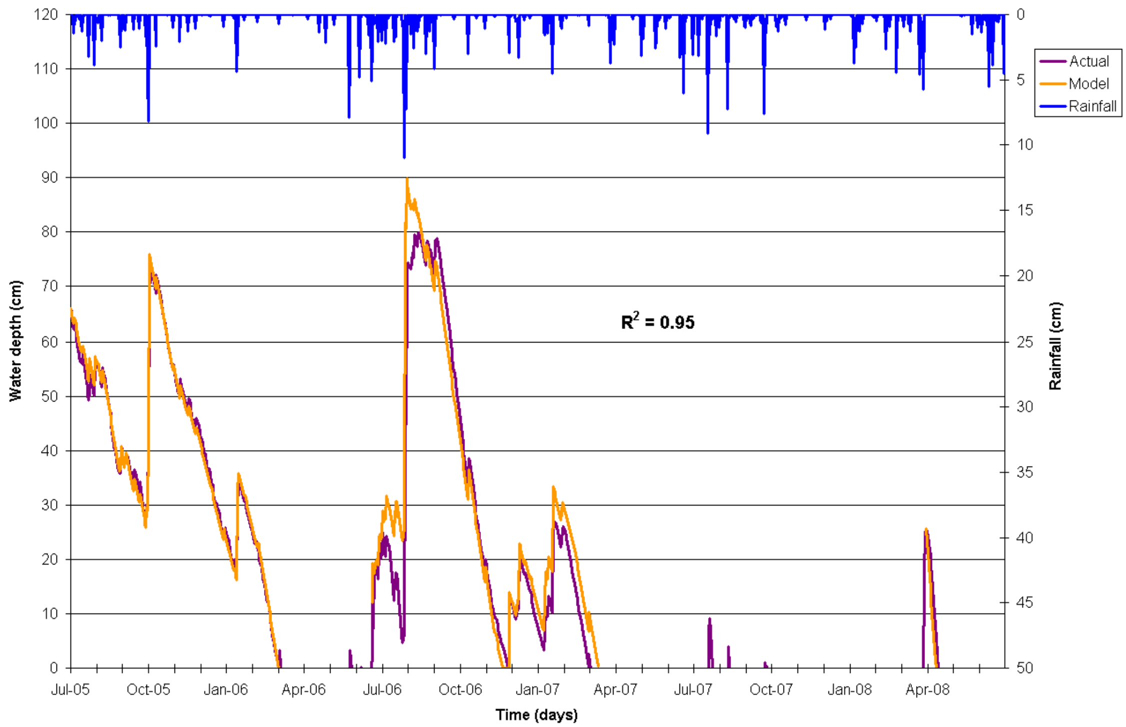


Figure 100. Comparison of Measured and Modeled Water Levels for H1 SW-1.

H1 SW-2 was simulated from 7/21/06 to 7/9/08 except for periods when the water level receded below ground (Figure 101). This model had the worst fit of the seven watersheds modeled (R^2 = 0.59). This was probably due to the fact that there was only surface water periodically, since this watershed rapidly dried out.

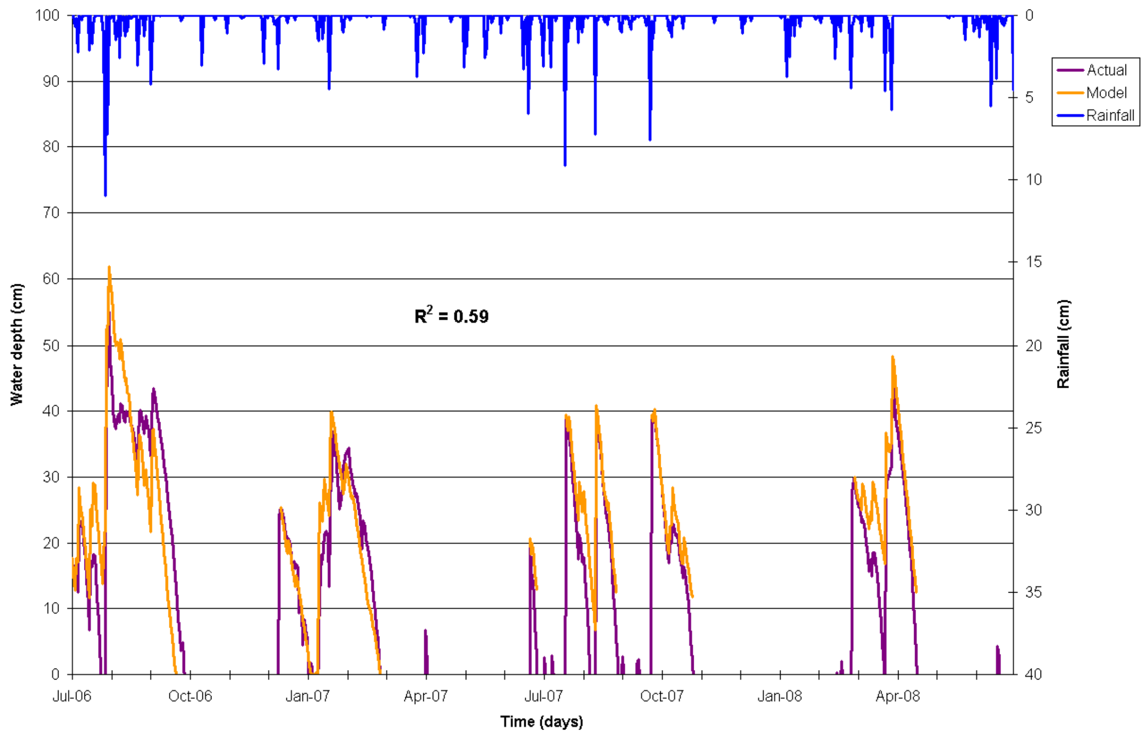


Figure 101. Comparison of Measured and Modeled Water Levels for H1 SW-2.

PCS SA 01. SA 01 had one of the longer simulation periods with appropriate surface water levels from 8/5/06 to 6/26/08 (692 days). The fit of the model yielded a coefficient of determination equal to 0.83, which was reasonable for this simplistic model (Figure 102). In the spring of 2007, the model began to exceed the measured water levels until they receded in the fall of 2007. At this point the model was adjusted to include a lower outfall height due to the removal of weir slats. In the spring of 2008, the model declined faster than the measured results, which can be explained by the need to use daily loss rates in the spring months that were a compromise between the springs of 2007 and 2008 in order to achieve a best fit between the modeled and measured water levels.

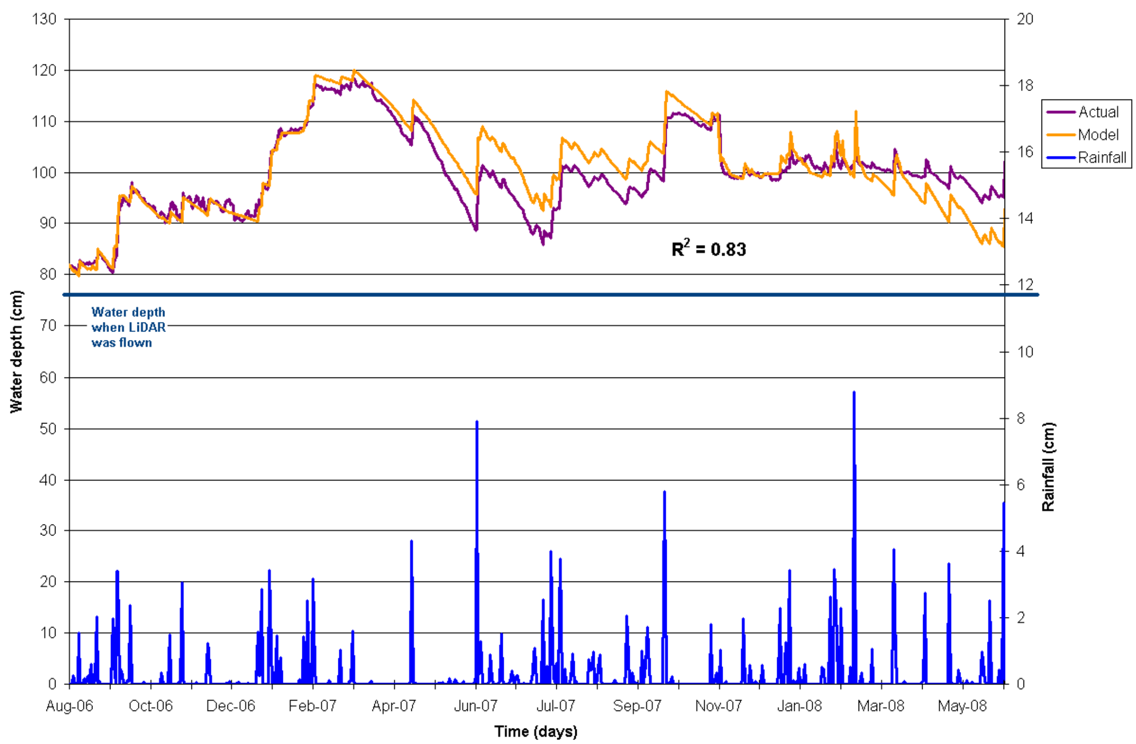


Figure 102. Comparison of Measured and Modeled Water Levels for SA 01.

PCS SA 10. SA 10 had one of the shortest simulation periods, from 8/9/05 to 4/4/06 (239 days), due to water levels being lower than the water level when LiDAR was flown for 23% of the monitoring period (Figure 103). The calibration graph shows a decent fit (R^2 = 0.70) between modeled and measured water level data. The main deviation occurred during January 2006; this may have been due to a change in outfall characteristics (such as clogging by debris) causing less water to flow out through the weir than modeled.

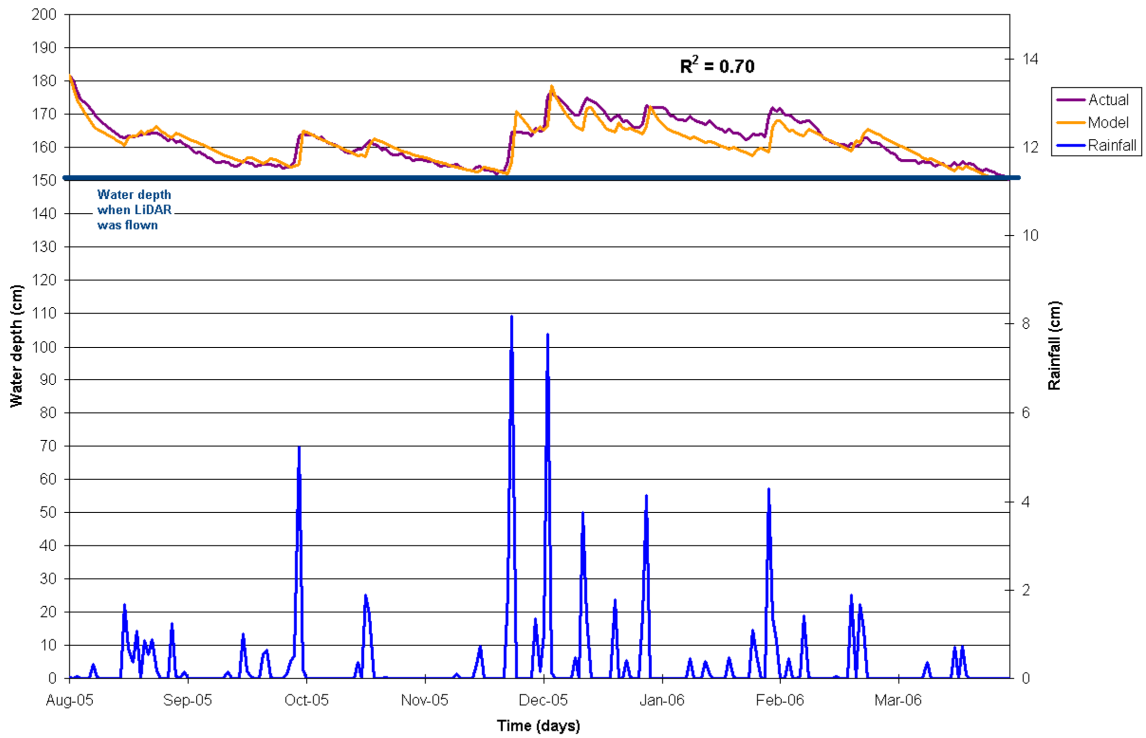


Figure 103. Comparison of Measured and Modeled Water Levels for SA 10.

Mosaic K5. The simulation period for K5 ran from 10/10/04 to 5/22/08, including 1021 days that were able to be modeled, which made this site the longest simulated (Figure 104). Overall there were three separate simulations because the water level receded below the LiDAR water level twice before increasing to applicable levels again. The spatial model worked well for this site in terms of fit between modeled and measured water levels ($R^2 = 0.82$).

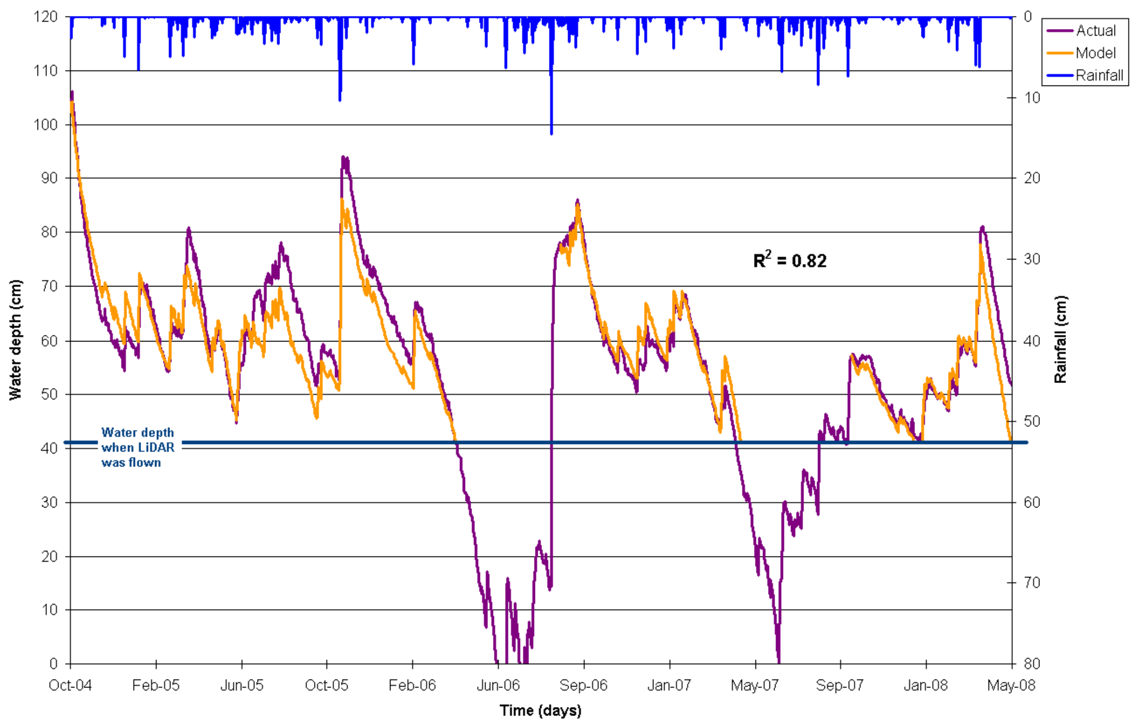


Figure 104. Comparison of Measured and Modeled Water Levels for K5.

CF Industries SP-1. SP-1 contained water levels above the LiDAR water level of 25 cm from 10/10/04 to 4/7/06 (544 days) until the site started to dry out (Figure 105). This simulation model generated the second worst fit (R^2 = 0.69) between measured and modeled water levels out of the seven watersheds modeled, but this was mainly due to the deviation of the modeled water level towards the end of the simulation period since the rest of the data matched well.

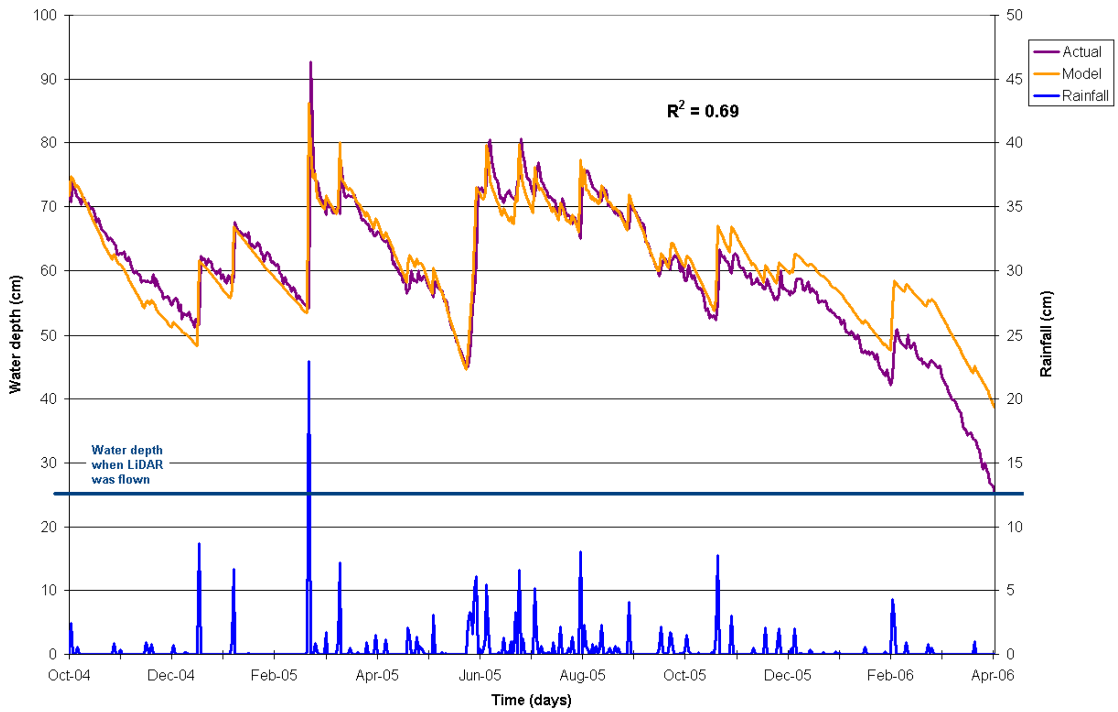


Figure 105. Comparison of Measured and Modeled Water Levels for SP-1.

Williams Co. Williams Co. had the next shortest simulation period of 287 days, from 6/16/05 to 3/29/06, due to the water feature drying (Figure 106). The model generated one of the best fits compared to measured water level data (R^2 = 0.88).

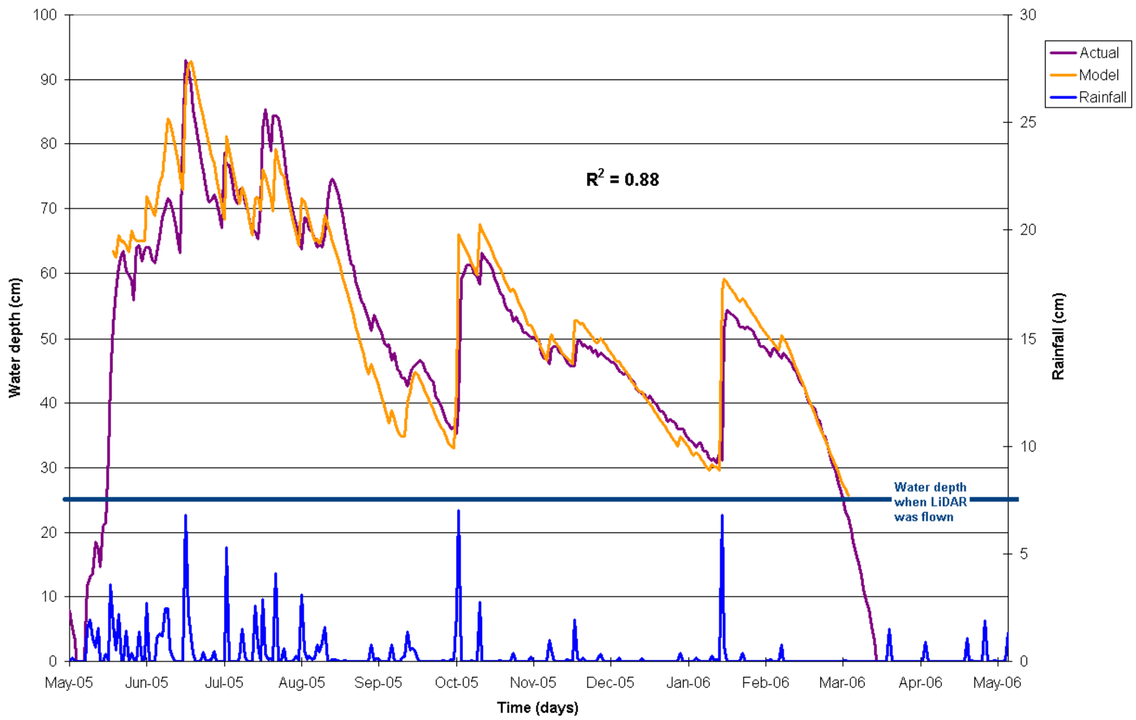


Figure 106. Comparison of Measured and Modeled Water Levels for Williams Co.

Model Coefficients. A summary of the model coefficients used in the water budget calculations are given in Table 42. H1 SW-1 included three different simulation periods, with different daily loss rates. The higher daily loss rates were necessary for the 2006-2007 period and the April 2008 period as compared to the 2005-2006 period, a fact that was attributed to the observation of large cracks that developed in the clay ground surface during the dry periods between simulation periods.

Another special adjustment for H1 SW-1 that generated a better fit was the runoff coefficient increase for large storm events. The coefficient was raised from 0.50 to 0.80 for rain events larger than 4 cm, and 2.60 for the one rain event larger than 10 cm. This adjustment seemed reasonable since this watershed appeared to be periodically connected to adjacent watersheds, and probably received inflow from these watersheds during these large rain events. The 10.9 cm storm event fit the measured data best when a runoff coefficient of 2.60 was used, which is fundamentally impossible, thus indicating that water was coming from outside the watershed.

Site	H1 – SW1	H1 – SW1	H1 – SW1	H1 – SW2	SA 1	SA 10	K5	SP-1	Williams Co.
Simulation Period - Start Date	7/26/2005	7/9/2006	7/9/2007	7/21/2006	8/5/2006	8/9/2005	10/10/2004	10/10/2004	6/16/2005
Simulation Period - End Date	7/8/2006	7/8/2007	7/9/2008	7/9/2008	6/26/2008	4/4/2006	5/22/2008	4/7/2006	3/29/2006
Simulation Period (days)	241	261	13	282	692	239	1021	544	287
LiDAR Water Depth (cm)	N/A	--	--	N/A	76	151	41	25	25
Max Water Depth (cm)	80	--	--	55	118	182	106	93	93
Avg Water Depth (cm)	4	--	--	< 0	76	123	53	< 0	< 0
Calibration									
Correlation Coefficient (R^2)	0.95	--	--	0.59	0.83	0.70	0.82	0.69	0.88
Runoff Coefficient	0.50	--	--	0.90	0.40	0.50	0.90	0.20	0.50
Daily Loss Rates (cm/day)									
Jan	0.70	0.70	N/A	1.40	0.00	0.50	0.30	0.35	0.60
Feb	0.70	0.70	N/A	1.40	0.10	0.60	0.35	0.30	0.60
Mar	1.00	1.00	N/A	1.40	0.25	0.60	0.50	0.50	1.00
Apr	N/A	N/A	2.20	2.00	0.35	0.60	0.70	0.65	N/A
May	N/A	N/A	N/A	N/A	0.45	N/A	0.80	0.85	N/A
Jun	N/A	N/A	N/A	N/A	0.50	N/A	0.75	0.80	1.00
July	1.00	1.00	N/A	2.40	0.70	N/A	0.75	0.70	2.20
Aug	1.00	1.20	N/A	2.50	0.40	0.70	0.70	0.70	2.00
Sept	1.00	1.20	N/A	2.40	0.35	0.55	0.60	0.70	1.50
Oct	0.90	1.20	N/A	2.00	0.25	0.55	0.40	0.60	0.90
Nov	0.90	1.20	N/A	1.40	0.20	0.35	0.35	0.50	0.90
Dec	0.70	0.70	N/A	1.40	0.10	0.50	0.35	0.30	0.60
Outflow									
Height 1 (cm)/ Outflow (cm/day)	--	--	--	--	100/0.50	166/1.00	60/0.20	75/3.50	--
Height 2 (cm)/ Outflow (cm/day)	--	--	--	--	105/3.00	171/1.50	70/0.30	80/5.00	--
Height 3 (cm)/ Outflow (cm/day)	--	--	--	--	110/6.00	176/3.00	80/0.40	85/7.00	--
Height 4 (cm)/ Outflow (cm/day)	--	--	--	--	--	--	90/0.75	90/10.00	--
Height 5 (cm)/ Outflow (cm/day)	--	--	--	--	--	--	100/1.75	--	--
Special									
Runoff Coefficient For Rain Events >4 cm	0.80	--	--	--	--	--	--	--	--
Runoff Coefficient For Rain Events >8 cm	2.60	--	--	--	--	--	--	--	--

Table 42. Summary of Water Budget Components.

The runoff coefficients had a wide range from 0.20 to 0.90, which reflects differences in geomorphological features that dictate runoff amounts during rain events. The daily loss rates also varied significantly between sites, with values as high as 2.20 cm/day at Williams Co. during July and as low as 0 cm/day at SA 01 during January. Most sites had less than 1.00 cm/day as a maximum daily loss rate. SA 01 had the least daily loss throughout the year, ranging from 0.00 cm/day in the winter to 0.70 cm/day in the summer. H1 SW-1, SA 10, K5, and SP-1 all showed less variability throughout the year, with a maximum difference 0.55 cm/day between summer and winter values for SP-1. Outflow was included in the water budget for SA 01, SA 10, K5, and SP-1, where SP-1 had the highest maximum outflow rate (10 cm/day) and K5 the lowest (1.75); however, it should be noted that this outflow rate occurred on only one day for SP-1.

Hydropattern Maps

Mosaic H1. The hydropattern map shows no values of 100% inundation during the simulation period because the feature was dry for part of each year (Figures 107 and

108). H1 SW-1 and SW-2 were the only features whose hydropattern map represents the entire monitoring period, since the monitored features on other CSAs contained water when the LiDAR was flown. Both watersheds had a significant amount of area that periodically contained surface water; however some of this area was dry for greater than 80% of the time (Tables 43 and 44).

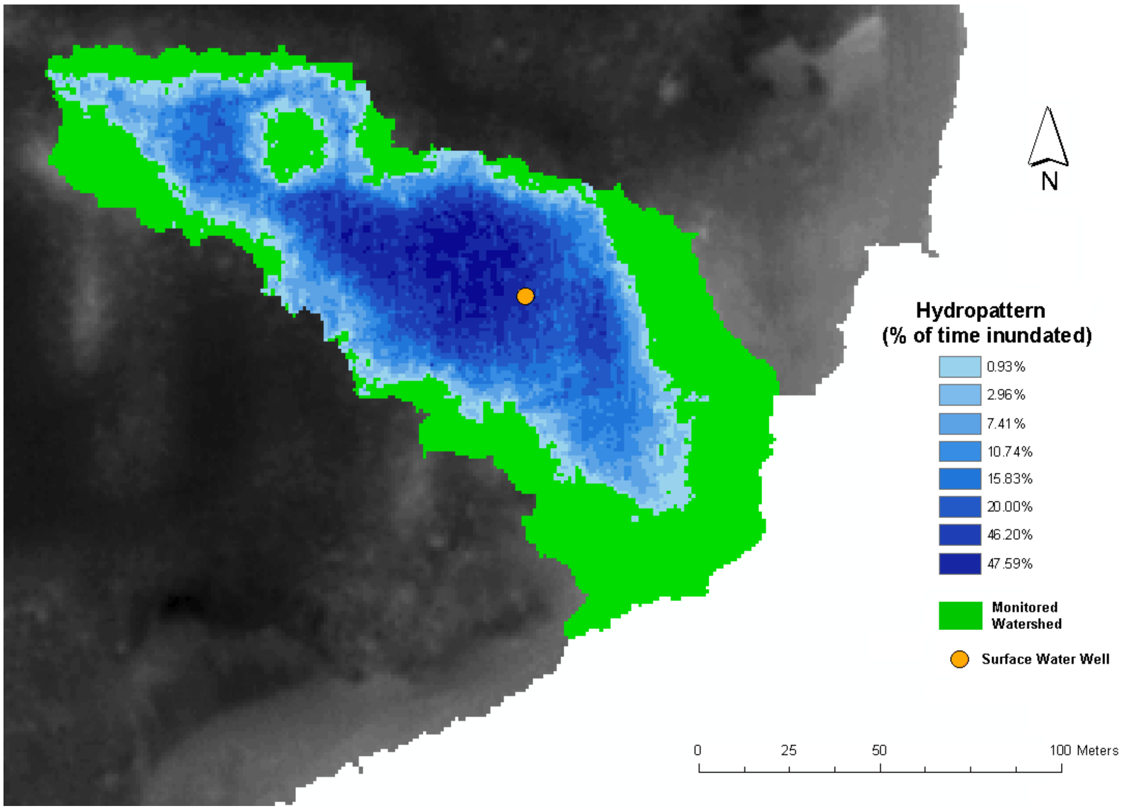


Figure 107. H1 SW-1 Hydropattern Map from 7/26/05 to 7/9/08.

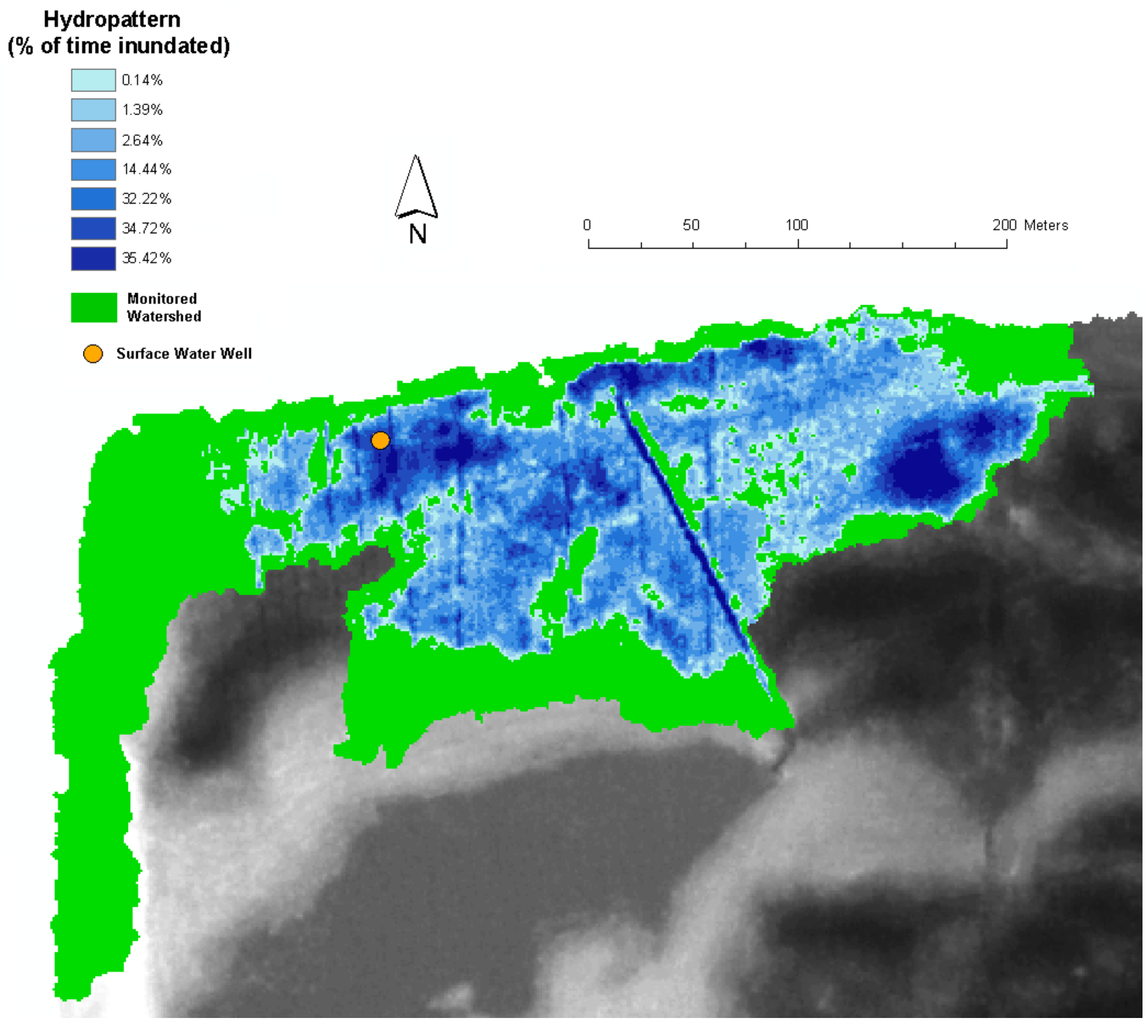


Figure 108. H1 SW-2 Hydropattern Map from 7/21/06 to 7/9/08.

Table 43. Hydropattern Area Inundated for H1 SW-1.

% of Time Inundated	0.93	2.96	7.41	10.74	15.83	20.00	46.20	47.59
Area (ha)	0.11	0.11	0.12	0.12	0.11	0.11	0.11	0.13
% of inundated area	11.90	12.12	12.86	13.00	12.13	12.02	11.94	14.05
% of watershed area	6.88	7.01	7.44	7.52	7.02	6.95	6.91	8.13

Table 44. Hydropattern Area Inundated for H1 SW-2.

% of Time Inundated	0.14	1.39	2.64	14.44	32.22	34.72	35.42
Area (ha)	0.28	0.40	0.62	0.60	0.34	0.23	0.23
% of inundated area	10.39	14.96	22.96	22.06	12.61	8.49	8.53
% of watershed area	5.40	7.77	11.93	11.46	6.55	4.41	4.43

PCS SA 01. The hydropattern map for SA 01 shows that a majority of the watershed contained standing water throughout the simulation period (Figure 109). About 1.58 ha experienced water level fluctuations (Table 45), most of which were located around the perimeter of the water feature, with a substantial area in the northwestern portion of the watershed. The simulation period did not include the 672 days (49% of monitoring period) for which the monitored water levels were lower than when the LiDAR was flown, which indicates that there was actually more area that fluctuated in the range appropriate for wetland plants.

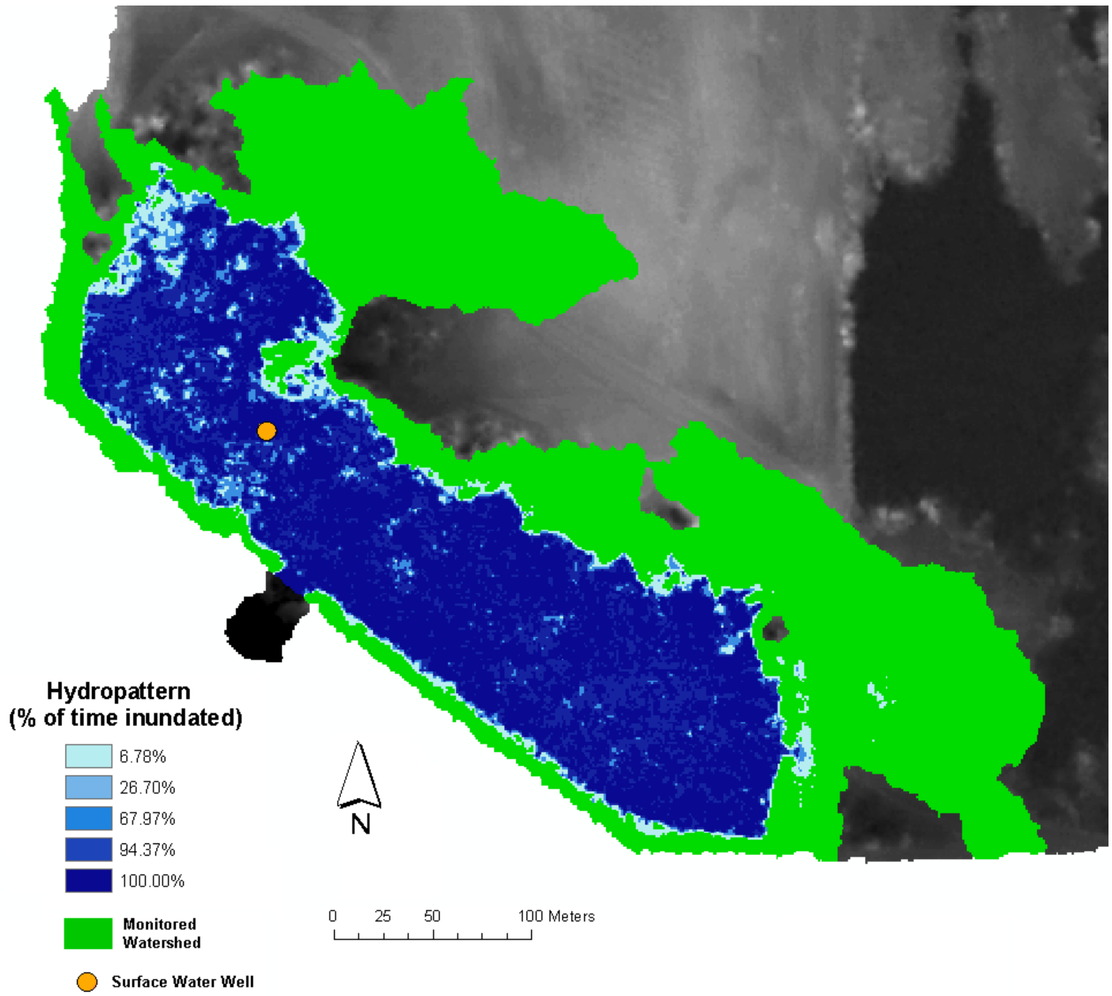


Figure 109. SA 01 Hydropattern Map from 8/5/06 to 6/26/08.

Table 45. Hydropattern Area Inundated for SA 01.

% of Time Inundated	6.78	26.70	67.97	94.37	100.00
Area (ha)	0.12	0.17	0.29	1.00	2.01
% of inundated area	3.30	4.76	7.96	27.94	56.03
% of watershed area	1.53	2.21	3.69	12.95	25.97

PCS SA 10. The SA 10 hydropattern map shows water accumulation in the north-central and eastern portions of the watershed (Figure 110). The north-central portion had much more spatial variability in the frequency of inundation, with a significant area that was barely inundated (10.61 ha) (Table 46). There were also 12.74 ha that were inundated about half of the simulation period. The simulation period did not include the 820 days (78% of the monitored period) for which the monitored water levels were lower than when the LiDAR was flown, which indicates that there was actually more area that fluctuated in the range appropriate for wetland plants.

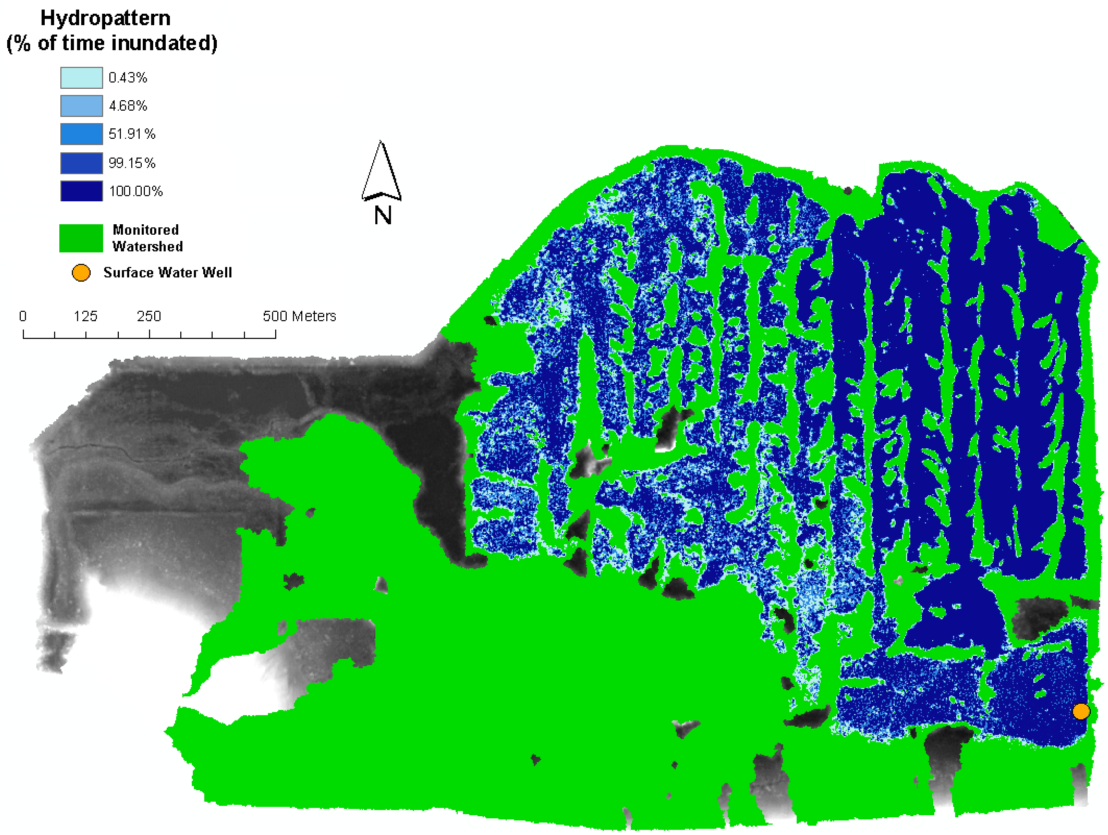


Figure 110. SA 10 Hydropattern Map from 8/9/05 to 4/4/06.

Table 46. Hydropattern Area Inundated for SA 10.

% of Time Inundated	0.43	4.68	51.91	99.15	100.00
Area (ha)	4.08	6.53	12.74	20.04	35.33
% of inundated area	5.18	8.30	16.19	25.46	44.88
% of watershed area	2.18	3.49	6.81	10.71	18.87

Mosaic K5. The hydropattern map for K5 shows many different periods of time inundated (Figure 111). Most of the site was inundated either infrequently (<13% of the time) or very frequently (>82% of the time) (Table 47). Only 6.51 ha, which represented

4.11% of the watershed area, was inundated between 13% and 82% of the time, and all of this area was inundated exactly 43% of the time. Where the map indicates 100% inundation over the simulation period, there was actually less than constant inundation in some of this area, since the entire monitoring period was not able to be simulated (LiDAR limitations).

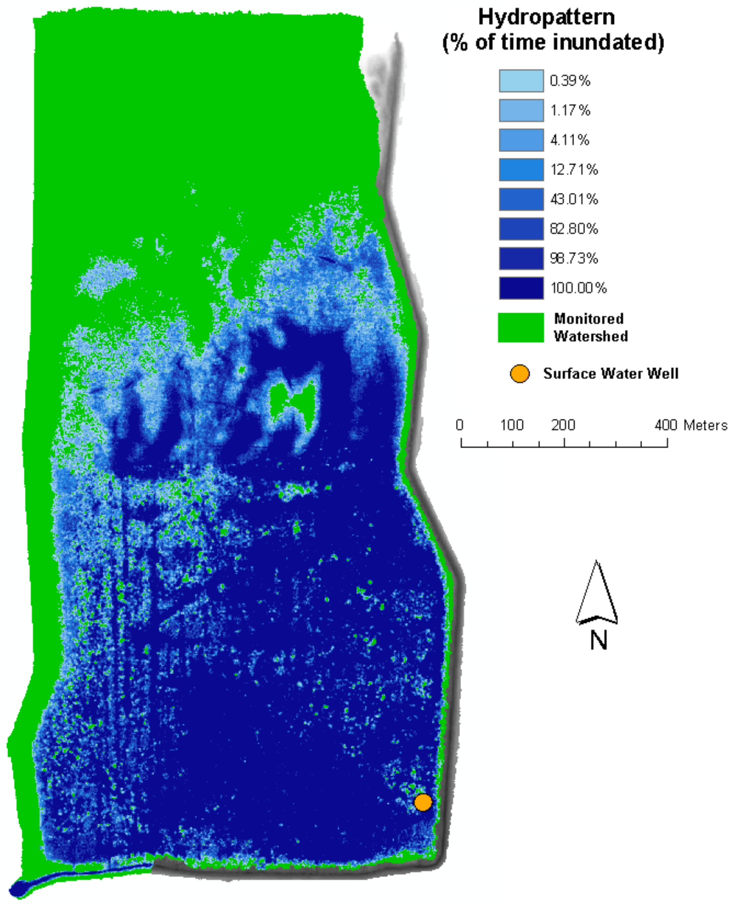


Figure 111. K5 Hydropattern Map from 10/10/04 to 5/22/08.

Table 47. Hydropattern Area Inundated for K5.

% of Time Inundated	0.39	1.17	4.11	12.71	43.01	82.80	98.73	100.00
Area (ha)	5.48	5.82	5.99	6.32	6.51	7.01	7.79	54.46
% of inundated area	5.52	5.86	6.03	6.36	6.55	7.05	7.84	54.80
% of watershed area	3.47	3.68	3.79	4.00	4.11	4.43	4.92	34.44

CF Industries SP-1. The SP-1 hydropattern map shows that much of the area that contained surface water during the simulation period remained inundated during that period (Figure 112). The area over which the water levels fluctuated was located mostly around the perimeter of the water feature (Table 48). Where the map indicates 100% inundation over the simulation period, there was actually less than constant inundation in some of this area, since the entire monitoring period was not able to be simulated (LiDAR limitations).

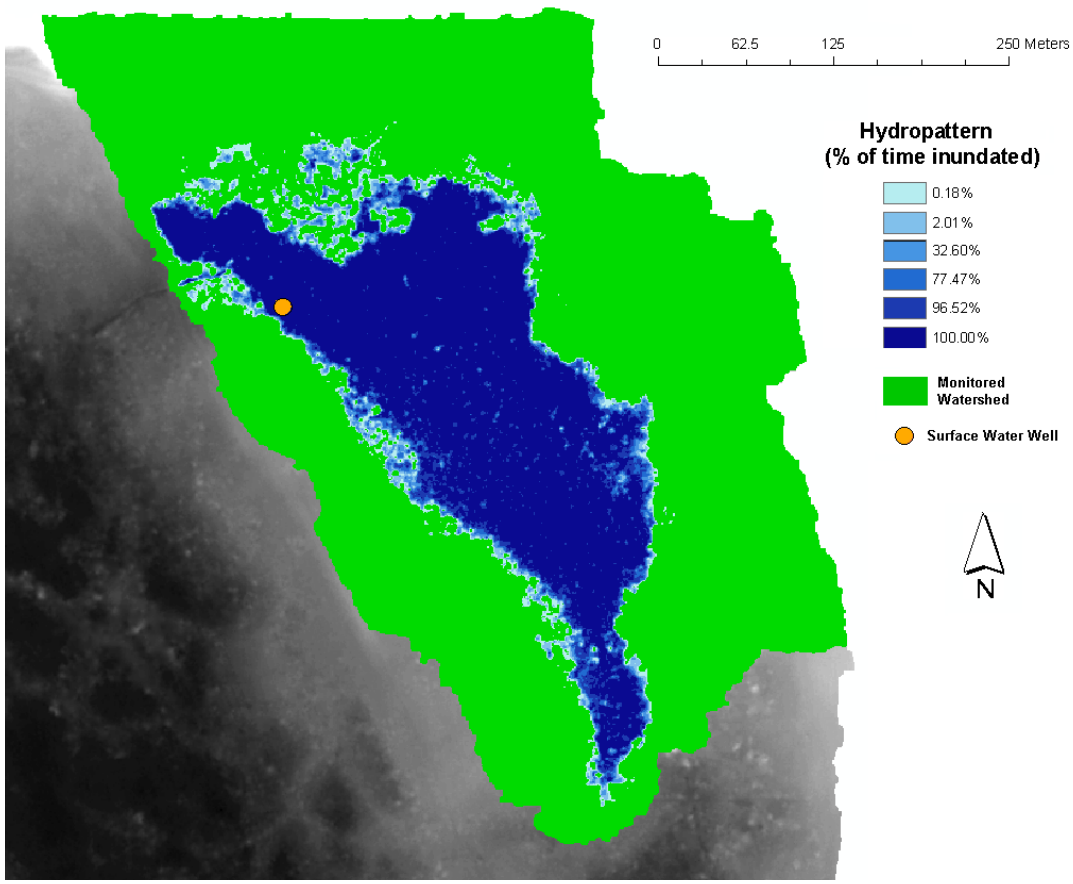


Figure 112. SP-1 Hydropattern Map from 10/10/04 to 4/7/06.

Table 48. Hydropattern Area Inundated for SP-1.

% of Time Inundated	0.18	2.01	32.60	77.47	96.52	100.00
Area (ha)	0.46	0.40	0.34	0.34	0.61	4.97
% of inundated area	6.41	5.66	4.80	4.78	8.55	69.80
% of watershed area	2.00	1.76	1.50	1.49	2.67	21.76

Williams Co. The Williams Co. hydropattern map shows that there were significant water level fluctuations in the surface water feature during the simulation period (Figure 113). The area over which the water levels fluctuated was located mostly around the perimeter of the water feature, but there were also areas within the interior of the feature that were inundated less than 70% of the time (Table 49). Where the map indicates 100% inundation over the simulation period, there was actually less than constant inundation in some of this area, since the entire monitoring period was not able to be simulated (LiDAR limitations).

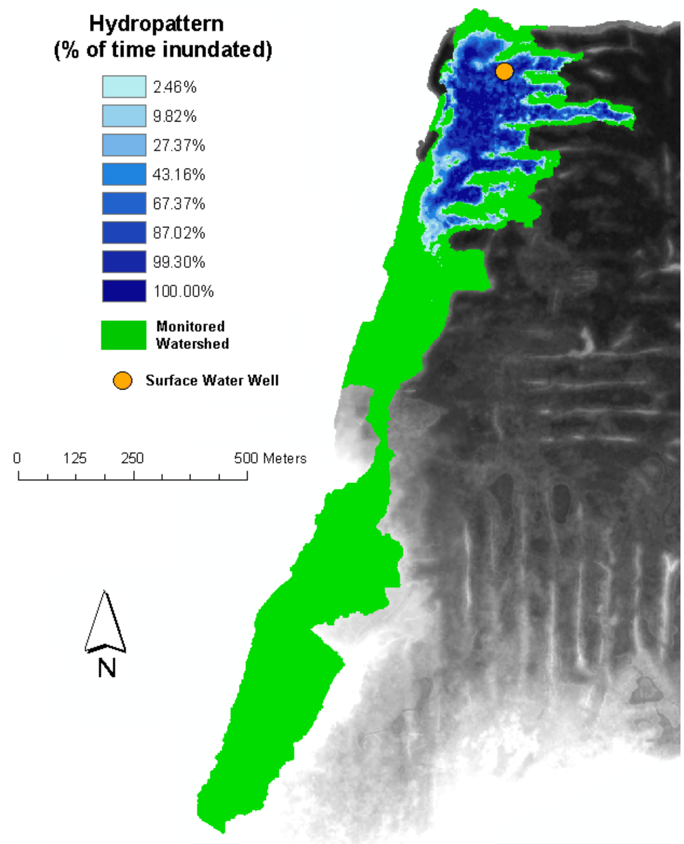


Figure 113. Williams Co. Hydropattern Map from 6/16/05 to 3/29/06.

Table 49. Hydropattern Area Inundated for Williams Co.

% of Time Inundated	2.46	9.82	27.37	43.16	67.37	87.02	99.30	100.00
Area (ha)	0.66	0.73	0.73	0.74	0.85	1.15	1.46	1.22
% of inundated area	8.74	9.71	9.64	9.87	11.29	15.19	19.41	16.15
% of watershed area	4.93	5.48	5.44	5.57	6.37	8.58	10.96	9.12

Water Budget

Evapotranspiration Estimation

Daily ET (mm/day) was estimated with seven empirical models using hourly climatic data from the three weather stations installed in canopy-free areas. Figure 114 shows the monthly averages of daily ET as estimated by the different methods using weather data from Williams Company during 2006. The daily ET estimates from 2/1/06 through 1/31/07 were summed to give total estimated ET (m) during that time period (Table 50). The Penman, Hargreaves, and Priestly Taylor methods resulted in the highest estimates. The Hamon method estimated similar rates during the summer months but much lower rates in the winter months.

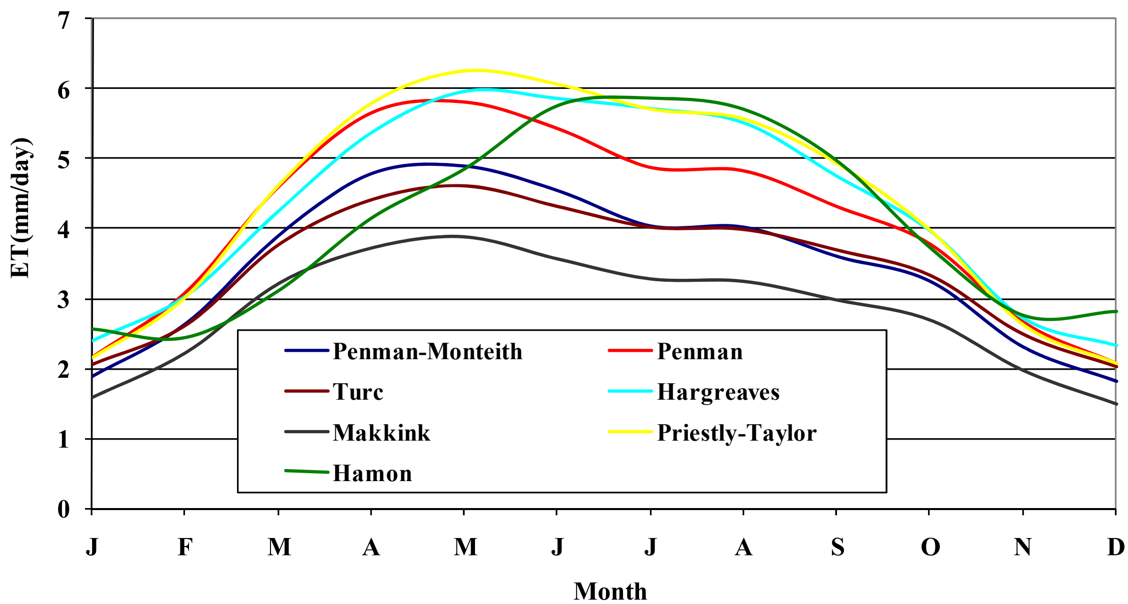


Figure 114. Monthly Averages of Daily ET Estimated with Different Methods.

Table 50. Total ET 2/1/06 through 1/31/07 at Williams Co.

Method	Total ET (m)
Penman-Monteith	1.28
Penman	1.55
Turc	1.26
Hargreaves	1.58
Makkink	1.03
Priestly-Taylor	1.60
Hamon	1.49

ET was estimated with the Penman method from January 2006 through June 2008 using on-site climatic data from the weather stations at Williams Co., PCS SA 10, and CFI SP-1. Monthly averages of these estimates are shown in Figure 115. The daily rates were totaled to calculate yearly ET in 2006 and 2007 (Table 51). The two central Florida sites, Williams Co. and CFI SP-1, had similar rates and seasonal trends with slightly lower yearly totals estimated at Williams Co., which is farther north. The northern site, PCS SA 10, had less total ET, which is to be expected with its milder climate. It had higher rates, however, during the later summer months possibly due to less convective thunderstorms occurring in north Florida compared to the central region.

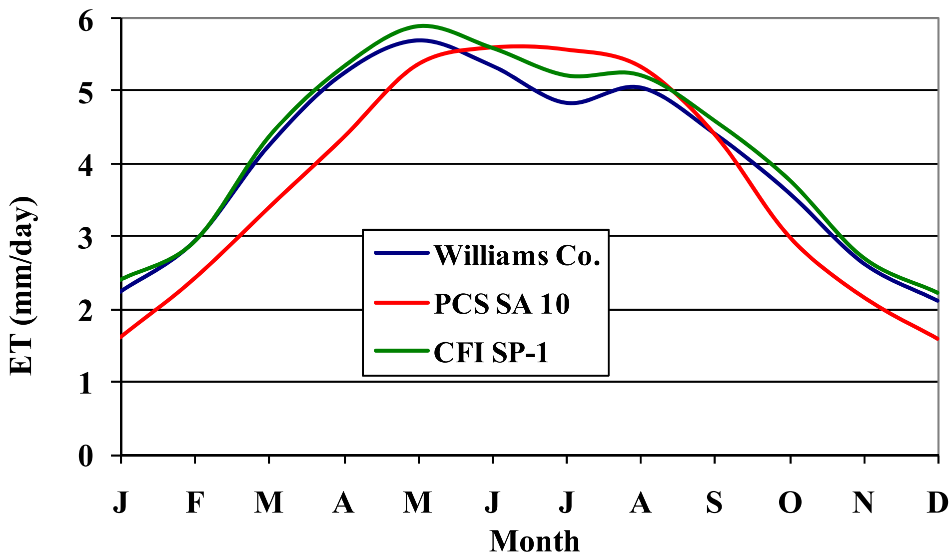


Figure 115. Monthly Averages of Daily ET with Penman Method for Williams Co., PCS SA 10, and CFI SP-1.

Table 51. Total ET (m) Estimated with Penman Method.

Site	2006	2007
Williams Co.	1.49	1.47
CFI SP-1	1.55	1.52
PCS SA 10	1.39	1.40

Runoff Analysis

Runoff analysis was performed for one surface water feature at each of the eight hydrology sites except for Tenoroc-4, which had limited periods of inundation. Additionally, another surface water feature at Mosaic H1, SW-3, was included in the analysis (Figure 69). Time periods when surface water outflow occurred at the sites with outfalls were excluded from the analysis. The relationships between event responses and magnitude of rain events were explored for all sites and linear regression resulted in the best fits for these relationships. Event responses versus rain events for PCS SA 01, PCS

SA 10, and Mosaic H1 SW-1 are shown in Figures 116-118. A 45° (1:1) line is also shown on the figures to highlight when the responses become greater than the rainfall amount. Event responses of PCS SA 01 and SA 10 followed the 1:1 line more closely than the responses at Mosaic H1 SW-1. The magnitude of event responses was greater at Mosaic H1 than at the PCS sites, with maximums of 0.35 m compared to 0.1 at PCS SA 01 and 0.12 at PCS SA 10, clearly showing the greater runoff experienced at Mosaic H1 SW-1.

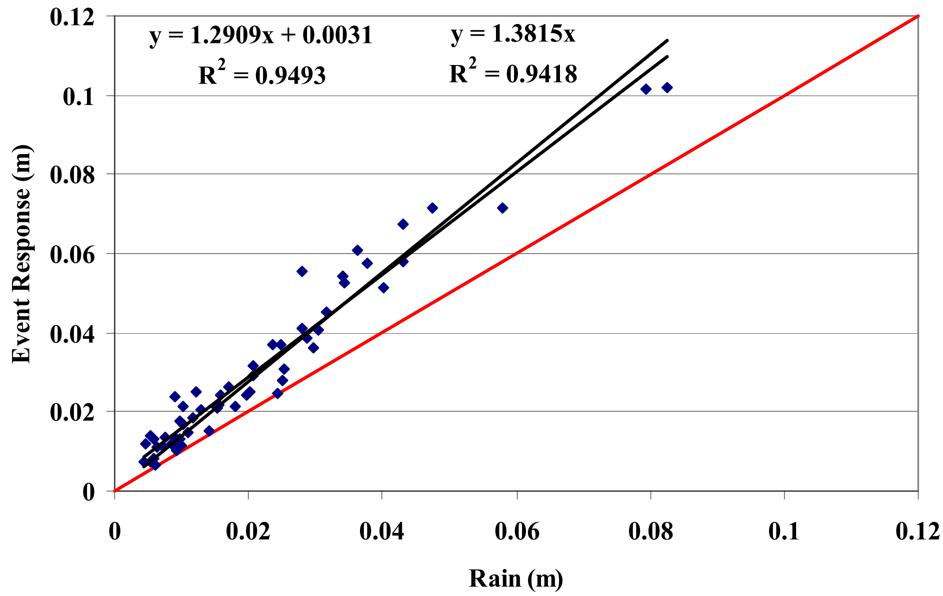


Figure 116. Event Response Versus Rain for PCS SA 01.

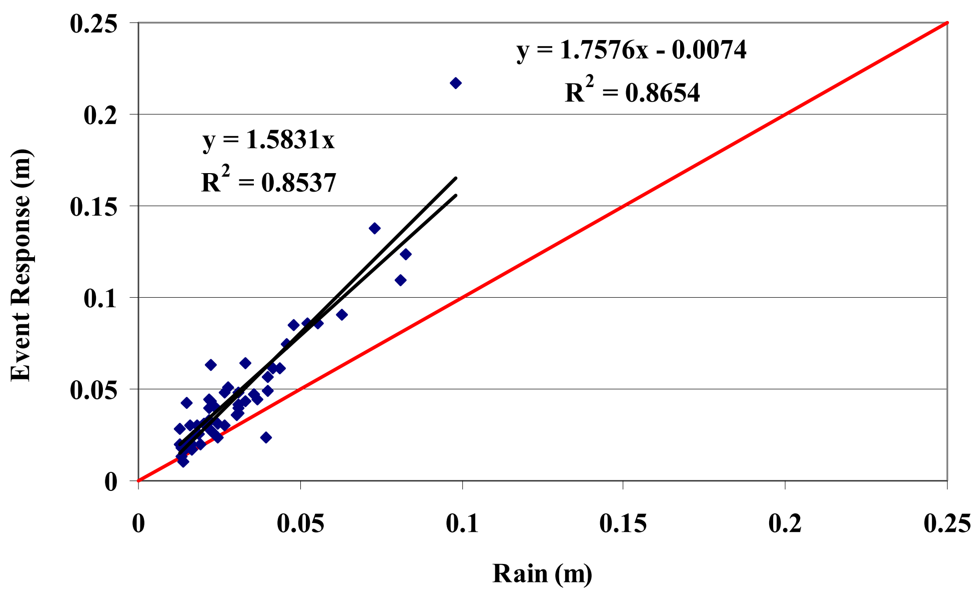


Figure 117. Event Response Versus Rain for PCS SA 10.

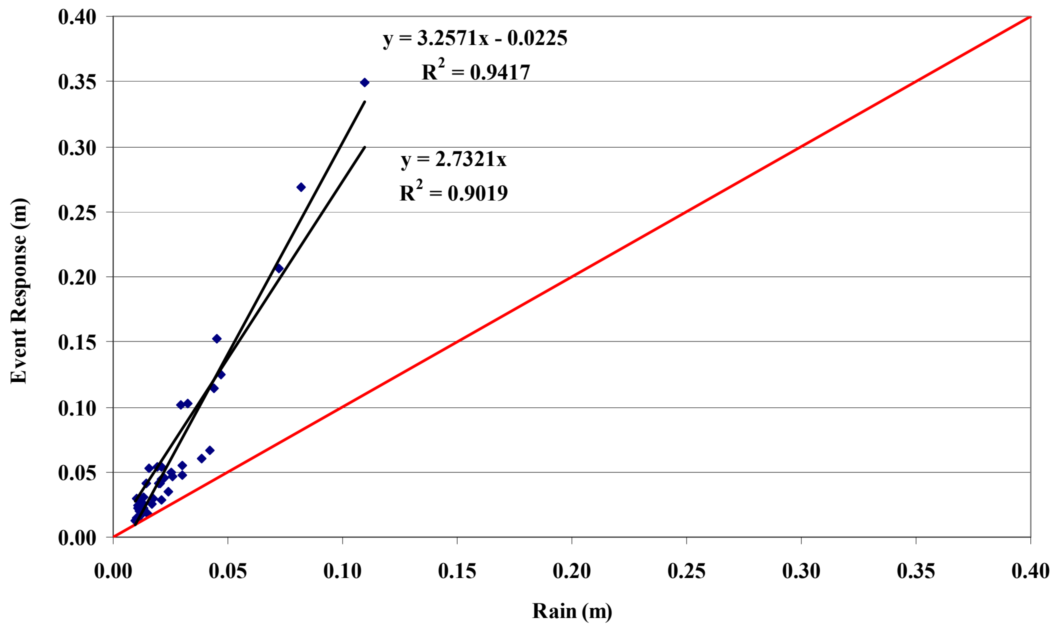


Figure 118. Event Response Versus Rain for Mosaic H1 SW-1.

Two regression lines of event response versus rain depth are shown in Figures 116-118. While the regression lines with a non-zero y-axis intercept produce a slightly better fit, setting the intercept to zero allows determination of runoff amounts. Event response represents the stage response plus ET predicted by Penman method during the day of the rain event, and subtracting the direct rain depth from the event response results in runoff depth. Therefore, subtracting one from the slope of the regression line for event response versus rain depth provides a quantification of runoff amounts in terms of percentages of rain amounts. These results demonstrate that PCS SA 01 and SA 10 typically experienced runoff amounts equaling 38% and 58% of direct rainfall, respectively, while runoff amounts equaling 173% of the direct rainfall occurred at Mosaic H1 SW-1. Furthermore, the strong linear relationships between event response and rain depth, which improve when the y-intercept is permitted to be non-zero, allow runoff to be predicted solely from rain data. The strong relationships also suggest that runoff was primarily influenced by event magnitude and less by antecedent conditions or upland-to-wetland ratios. This analysis did not include upland-to-wetland ratios and therefore did not account for contributing areas, which changed substantially with stage. The strong correlation between rain and runoff despite changing contributing area implies that runoff may be more controlled by a perimeter effect from the upland versus the actual size of the watershed of these sites, where microtopography and depressional storage may limit runoff.

Event responses plotted against rain depth for Mosaic H1 SW-3 and Mosaic K5 are shown in Figures 119 and 120. Strong linear relationships resulted for both sites, though the relationship for Mosaic K5 may be influenced by the maximum event response (Figure 119). Nevertheless, it appears that Mosaic K5 experienced more runoff than the two PCS sites and less than Mosaic H1 SW-1. SW-3 at Mosaic H1 experienced runoff depths that almost equaled direct rainfall (99.6%) compared to the 173% that occurred at the other surface water feature (SW-1) at Mosaic H1, demonstrating the different runoff characteristics at separate features within one CSA (Figures 118 and 120).

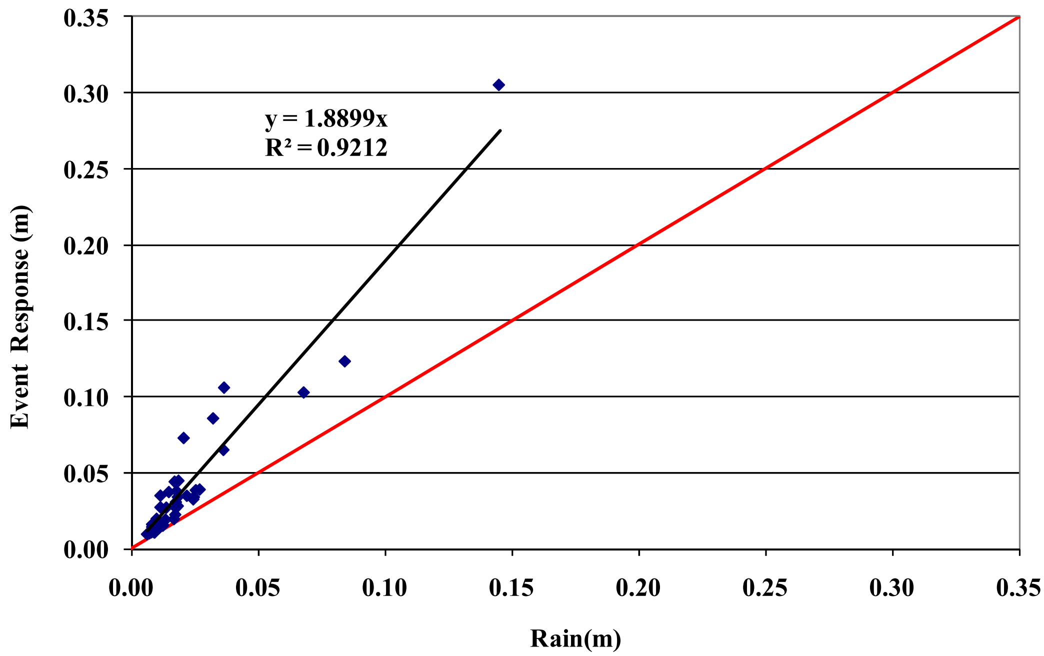


Figure 119. Event Response Versus Rain for Mosaic H1 K5.

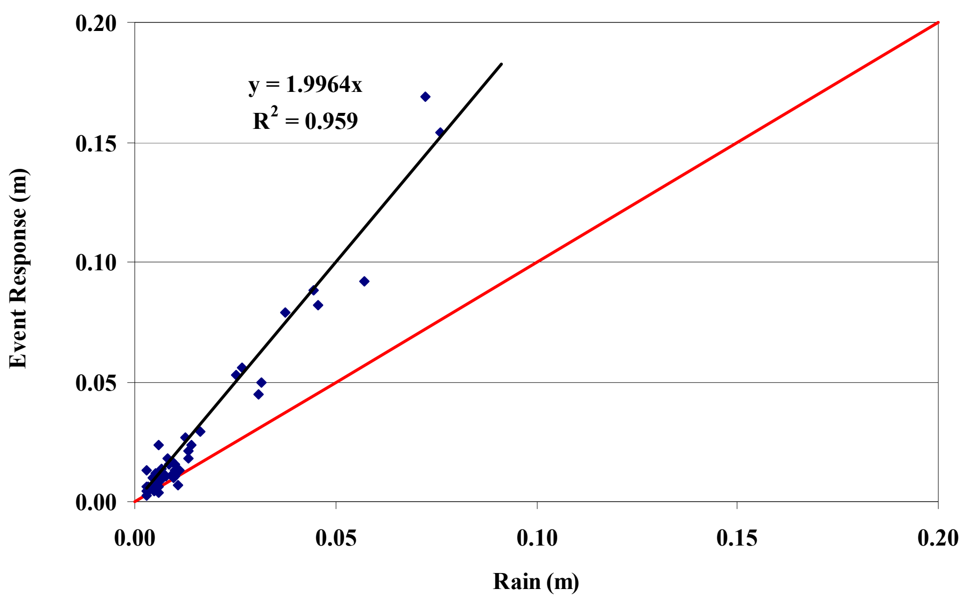


Figure 120. Event Response Versus Rain for Mosaic H1 SW-3.

The event responses at CFI followed the 1:1 line extremely closely and had a maximum of 0.09 m, which occurred during a rain event of 0.082 m (Figure 121). The regression line for responses experienced at CFI SP-1 had a slope near one, demonstrating the response to a rain event was primarily due to direct rainfall with small runoff depths equaling approximately 21% of rainfall. Again, the strong relationship allows prediction of response from rainfall data alone.

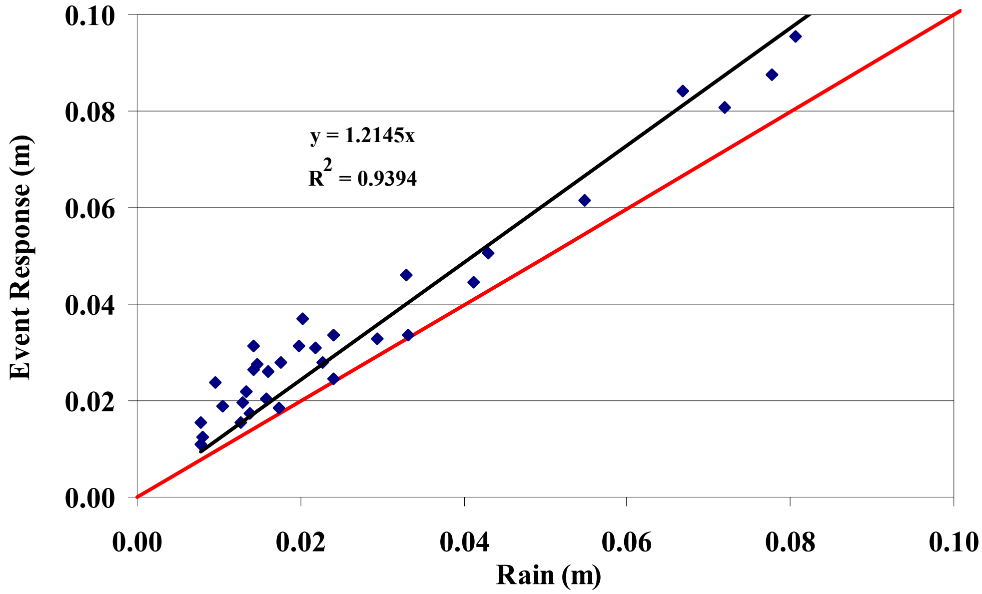


Figure 121. Event Response Versus Rain for CFI SP-1.

The event responses that occurred at Williams Co. and Mosaic HP-10 significantly departed from the 1:1 line and had maximums of 0.2 m and 0.35 m, respectively (Figures 122 and 123). Subtracting one from the slope of the regression lines reveals that runoff depths were approximately 1.5 and 3.8 times the rainfall at Williams Co. and Mosaic HP-10, respectively. The relationships between event responses and rain experienced at these sites, however, were not as strong as the previous sites presented, suggesting some influence from contributing areas and/or antecedent conditions on runoff events.

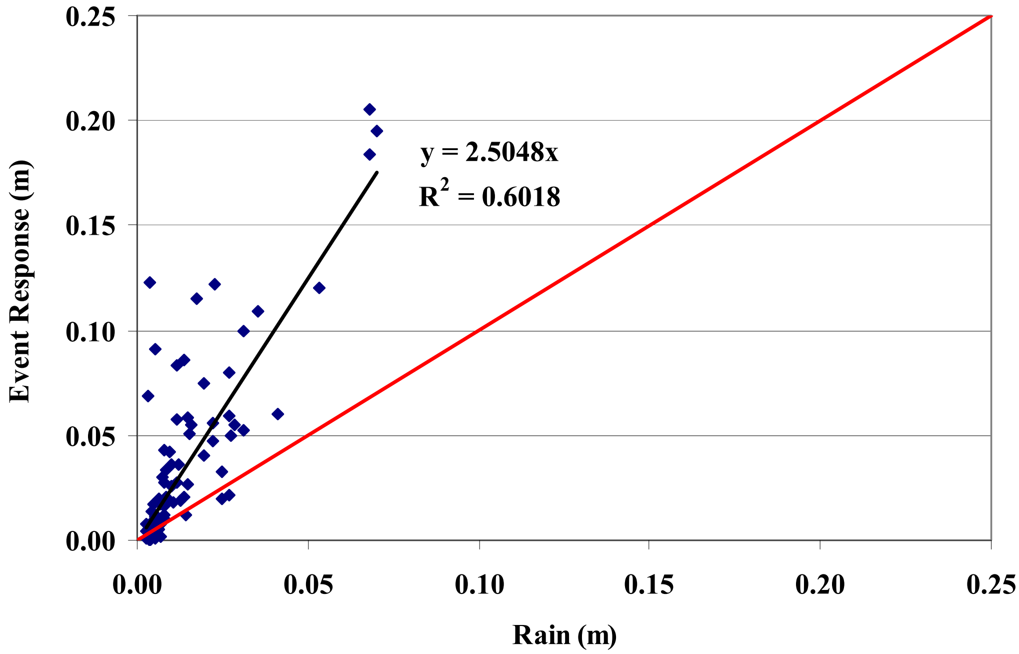


Figure 122. Event Response Versus Rain for Williams Co.

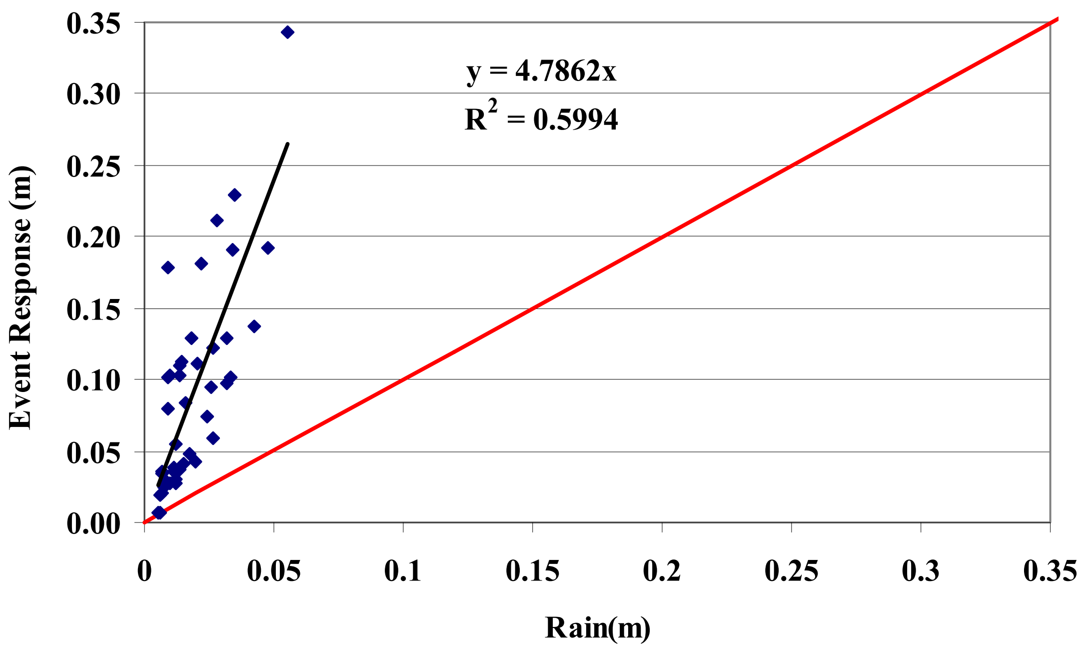


Figure 123. Event Response Versus Rain for Mosaic HP-10.

For at least six of the sites, the results from the event response analysis suggest that runoff may be adequately predicted with rainfall data independent of antecedent conditions or upland-to-wetland ratios. Even though this was the case, calculation and modeling of runoff coefficients were still performed. To observe the effect from scale of watershed delineation (See "Watershed Delineation," page 33), runoff coefficients were calculated for PCS SA 10 and Mosaic H1 SW-1 at three watershed scales. Upland-to-wetland ratios were modeled as a function of stage for each scale of watershed delineation and are shown in Figures 124 and 125. A power function performed better for PCS SA 10, whereas an exponential function resulted in the best fit for Mosaic H1 SW-1. The correlation coefficients indicated a good fit for the different scales of watersheds for both sites.

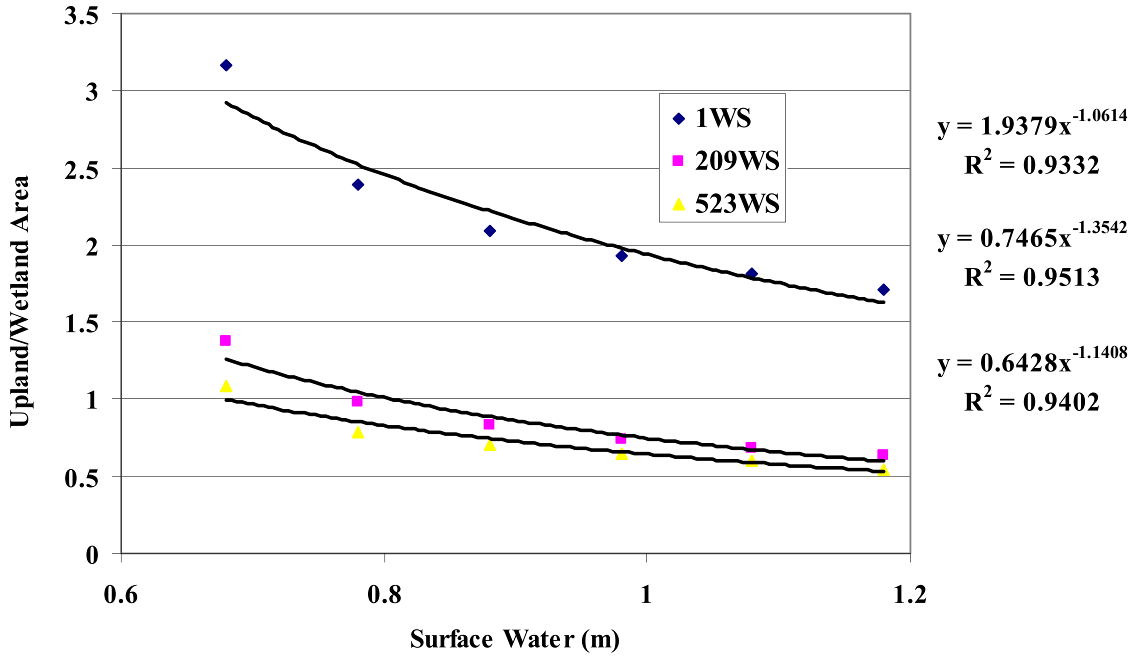


Figure 124. Upland/Wetland Area as a Function of Surface Water for Different Scales of Watersheds at PCS SA 10.

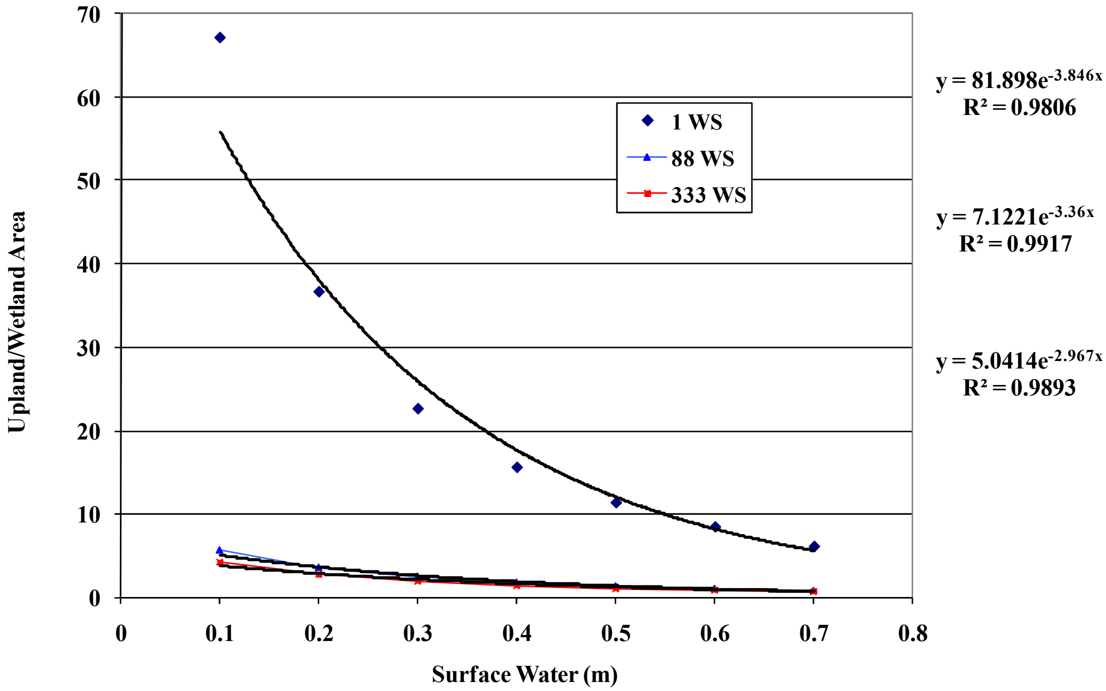


Figure 125. Upland/Wetland Area as a Function of Surface Water for Different Scales of Watersheds at Mosaic H1.

Using the upland-to-wetland area ratios, runoff depths (event response minus rain depth), surface water levels at midnight prior to the rain event, rainfall amount, and Equation 13, runoff coefficients were calculated. Runoff coefficients were calculated for each rain event and at the three scales of watershed analysis. Table 52 lists the average runoff coefficients using different scales of watershed analysis: large, medium and small. Large, medium, and small scales correspond to 1, 239, and 523 watersheds at PCS SA 10 and to 1, 88, and 333 watersheds at Mosaic H1 SW-1. Runoff coefficients increased with smaller scale analysis, as to be expected. When the entire CSA was treated as one watershed, the runoff coefficients were relatively small and not ones typically associated with clayey soils. As the number of watersheds was increased, Mosaic H1 SW-1's coefficients increased to values more typical of clayey soils. While the runoff coefficients for PCS SA 10 also increased, they remained much lower than the coefficients for Mosaic H1 SW-1, supporting similar results from the event response analysis (Figures 117 and 118).

Table 52. Average Runoff Coefficients for Different Scales of Watershed Analysis.

Scale of Watersheds	PCS SA-10	Mosaic H-1
Large	0.06	0.06
Medium	0.12	0.57
Small	0.20	0.66

Runoff coefficients for the remaining six features were calculated using the smallest scale of watershed analysis to determine upland-to-wetland ratios as functions of stage. Careful attention was placed when developing the ratios to account for the convergence of surface water features, as was observed in the hydroperiod analysis (Figures 80 and 81). As surface water features merge, so do their contributing areas—a fact that needs to be accounted for while calculating upland to wetland ratios—and this was done utilizing the multiple watershed approach (See "Watershed Delineation," page 33). Upland-to-wetland ratios were successfully related to stage for all eight features with squares of the correlation coefficients (R^2) equaling 0.91 or greater. The resulting average runoff coefficients (C) and upland-to-wetland ratios (Up/Wet) for all eight surface water features are shown in Table 53 along with the regression slopes of event response versus rain depths with y-intercept equal to zero. It should be noted that the runoff coefficients were dynamic and influenced by upland-to-wetland ratios and/or antecedent conditions and the numbers presented in Table 53 are the averages of the coefficients calculated. Likewise, the upland-to-wetland ratios presented here are averages of the ratios calculated, which varied substantially as they are a function of stage.

Table 53. Runoff Analysis Summary.

Site	Event Response Slope	Avg C	Avg Up/Wet
Mosaic H1 SW-1	2.73	0.66	2.4
Mosaic H1 SW-3	2.00	0.26	2.9
PCS SA 10	1.58	0.20	2.9
PCS SA 01	1.38	0.29	1.8
CFI SP-1	1.21	0.15	3.0
Mosaic HP-10	4.79	0.21	31.7
Mosaic K5	1.90	0.22	3.6
Williams Co.	2.50	0.32	5.4

Mosaic H1 SW-1 had the highest runoff coefficient, which was greater than the one of Mosaic HP-10 (Table 53). Mosaic HP-10, however, experienced the greatest runoff depths, with an event response slope of 4.79. The greater runoff depths despite a lower runoff coefficient results from Mosaic HP-10's much higher upland-to-wetland ratios. Mosaic HP-10 has a ditch network that connects all the surface water features within the site at certain stages, and therefore, the upland-to-wetland ratio at certain surface water depths was equivalent to the upland-to-wetland ratio of the entire site. The features at the other sites may only merge with nearby features but never connect through a ditch network; thus, their contributing area is strictly the upland area that surrounds them. Runoff coefficients must be coupled with contributing areas when using the coefficients to compare total runoff amounts. Williams Co. had the second highest runoff coefficient but it was comparable to that of PCS SA 01. The event response slope, however, for Williams Co. was much higher than that of PCS SA 01, which can be explained by a larger contributing area at Williams Co. PCS SA 01 also experienced less runoff than PCS SA 10 but had a higher runoff coefficient. Again, this can be explained by the contributing areas, which are much smaller at PCS SA 01. CFI SP-1 had both the lowest runoff coefficient and event response slope. Similar to the results in the event response analysis, Mosaic H1 SW-3 had a comparable contributing area to Mosaic H1 SW-1, but with a much lower runoff coefficient, demonstrating the different runoff response that may be experienced by separate features on an individual CSA.

The results from the event response analysis demonstrate that for most of the sites, runoff depth defined as event response minus rain depth can be predicted by estimating event response with rain data. The alternative is to predict runoff coefficients with rain and antecedent conditions and apply rain to the upland-to-wetland ratio with these coefficients to predict runoff depths. To determine if prediction of runoff could be improved with the inclusion of antecedent conditions and contributing area, multiple regression was employed to predict runoff coefficients. Using the calculated runoff coefficients and upland-to-wetland ratios, multiple regression with rainfall and the various antecedent condition indices were used to develop predictive models for runoff coefficients.

Using the antecedent condition indices for rainfall and ET history, multiple regression was performed for all sites. Mosaic H1 SW-1 and Williams Co. had models that resulted in correlation coefficients of 0.76 and 0.81, respectively. Predicted and actual runoff coefficients plotted for different rain events are shown in Figures 126 and 127. A linear trend versus rainfall is not evident for either site. The lack of this trend and the strong linear relationship between event response and rainfall amount at Mosaic H1 SW-1 with no inclusion of watershed area, reinforces that runoff at Mosaic H1 SW-1 may have been independent from watershed size and antecedent conditions. The resulting model for Williams Co., however, does predict runoff coefficients better than event response was predicted with rain depths, suggesting that antecedent conditions may have influenced runoff more at Williams Co. (Figures 127 and 122). Similarly, the multiple regression performed for Mosaic HP-10 resulting in an R^2 of 0.81 compared to an R^2 of 0.60 when predicting event response from rain depths (Figure 123). While event response was predicted at PCS SA 10 using rain depth with an R^2 of 0.85, multiple regression was able to better predict runoff coefficients ($R^2 = 0.94$).

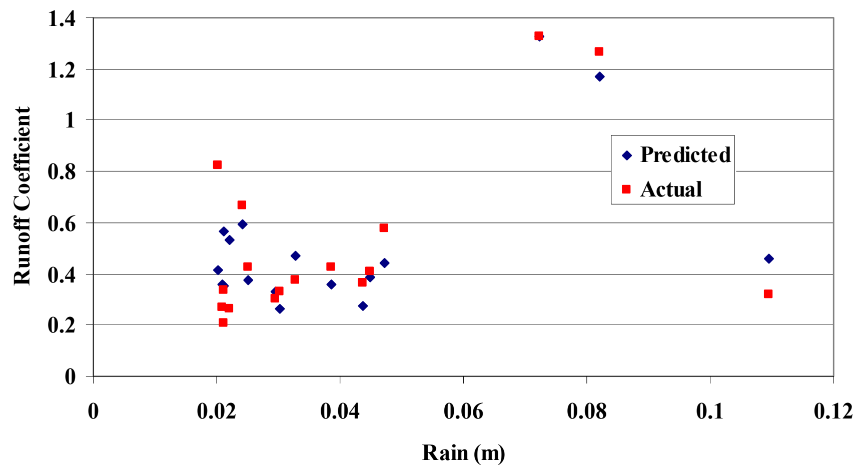


Figure 126. Actual and Predicted Runoff Coefficients Versus Rainfall for Mosaic H1 SW-1.

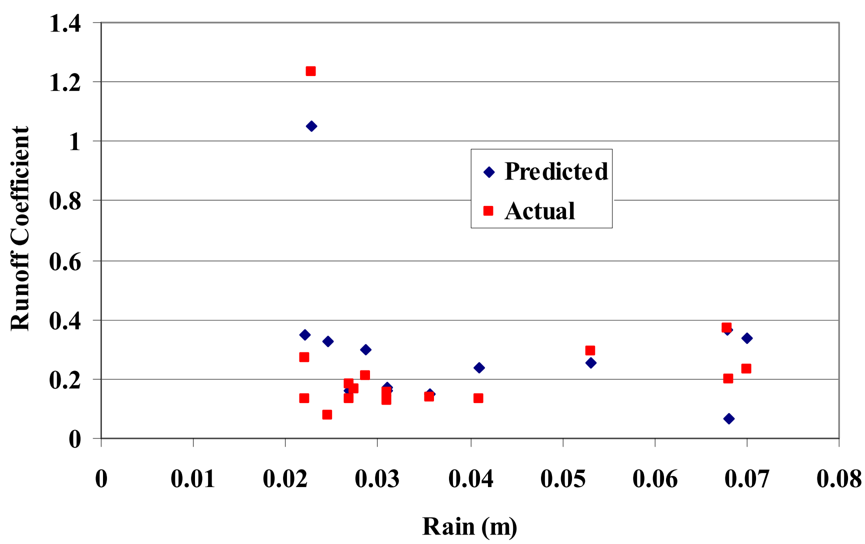


Figure 127. Actual and Predicted Runoff Coefficients Versus Rainfall for Williams Co.

The multiple regression models developed for the remaining four sites resulted in much lower R^2's compared to the other multiple regression models and to the R^2's that were achieved when predicting event response from rain depth (Table 54). The R^2's listed in Table 54 refer to the linear models that predict event response from rain but while allowing y-axis intercepts to be non-zero. So while it was mentioned above that both runoff coefficients and upland-to-wetland ratios are needed to evaluate runoff amounts, the dynamic nature of the coefficients may simply be a result of the varying contributing areas. The fact that most simple models for predicting runoff from rain performed better than the multiple regression models that included upland-to-wetland areas again suggests that runoff may have been primarily from a constant contributing area such as the edges of the feature that do not change significantly with stage.

Table 54. Runoff Analysis and Modeling Results.

Site	Event Response R^2	Multiple Regression R^2
Mosaic H1 SW-1	0.94	0.76
Mosaic H1 SW-3	0.97	0.46
PCS SA 10	0.87	0.94
PCS SA 01	0.95	0.30
CFI SP-1	0.96	0.23
Mosaic HP-10	0.60	0.82
Mosaic K5	0.92	0.37
Williams Co.	0.60	0.81

To further assess the models' ability to predict runoff amounts, positive actual event responses during periods of no outflow were summed and compared to predictions made by the various models. Since the event responses included rain events, the comparison was made between the sums of actual event responses and the sums of rain plus predicted runoff depths. Both the simple linear models and the multiple regression models for PCS SA 10, Mosaic H1 SW-1, and Williams Co. were evaluated (Table 55). In all three cases, the simple linear model resulted in higher total rainfall plus predicted runoff amounts, which were more comparable to the total of the recorded event responses. Such an analysis was not possible for Mosaic HP-10 as it experienced outflow quite frequently throughout its period of record.

Table 55. Actual and Predicted Rainfall plus Runoff Totals with Linear and Multiple Regression Models.

Site	Actual (m)	Simple Linear (m)	Multiple Regression (m)
PCS SA 10	1.02	1.01	0.91
Mosaic H1	2.87	2.82	2.63
Williams Co.	1.85	1.87	1.46

Only the linear models' abilities were evaluated for PCS SA 01, Mosaic K5, CFI SP-1, and Mosaic H1 SW-3 since it was found that the linear models performed for these sites far better than the multiple regression models (Table 56). All totals are within 5 cm except for Mosaic H1 SW-3, whose actual is over-predicted by 15 cm.

Table 56. Actual and Predicted Rainfall plus Runoff Totals with Linear Models.

Site	Rain+Predicted Runoff (m)	Actual (m)
PCS SA 10	2.13	2.10
Mosaic K5	2.73	2.70
CFI SP-1	2.28	2.32
Mosaic H1 SW-3	2.02	2.17

Surface Water Outflow

Mosaic K5 and Tenoroc-4 both have channels that allow outflow and which were instrumented in May 2006 with upstream and downstream wells, separated by at least 50 m. The water levels in the upstream and downstream wells, where the downstream levels are water level elevations relative to levels at the upstream well, are shown in Figures 128 and 129. The channel at Tenoroc-4 experienced high water levels but low flow, as indicated by the similar elevations experienced in the upstream and downstream wells. Comparing the channel surface water levels with the levels in Figure 67, it seems that the channel acted as a water feature rather than a channel draining feature. This channel connects to another larger channel before exiting over a weir. The elevation of the weir and the backflow force caused from the larger channel may have limited the flow from the Tenoroc-4 channel. The channel at Mosaic K-5 experienced lower levels and more frequent drying. Furthermore, there were positive head differences between the water elevations at the upstream and downstream wells, indicating outflow occurred from the site.

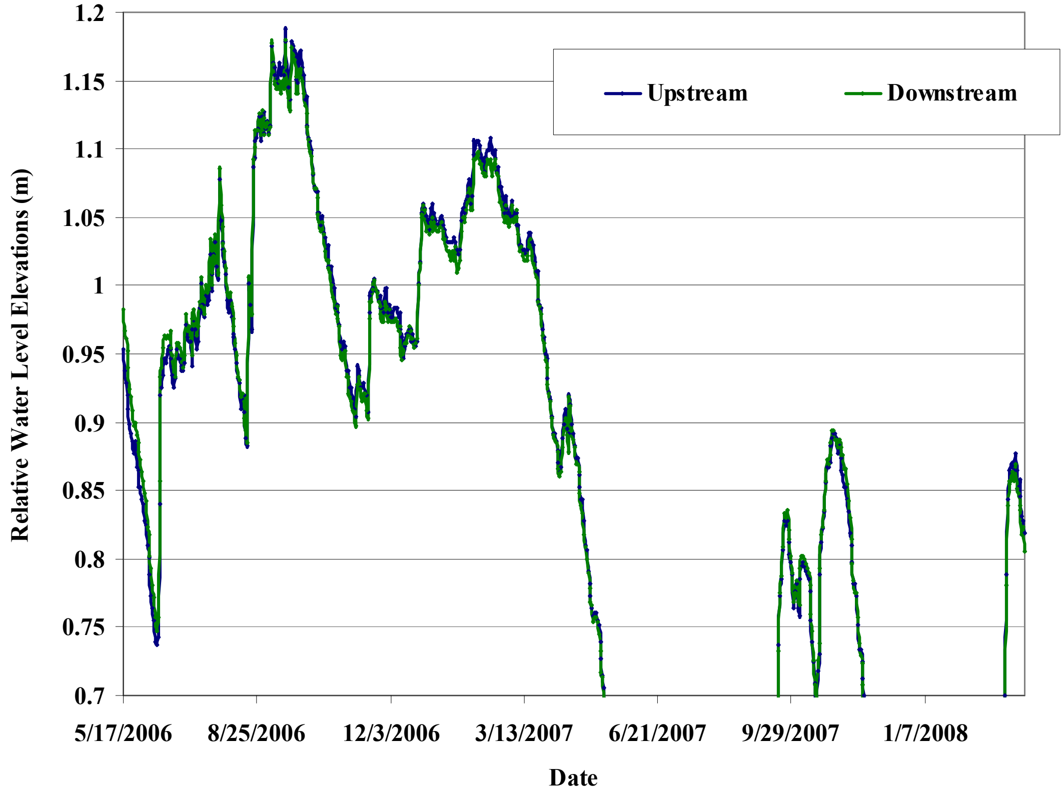


Figure 128. Relative Water Elevations from Upstream and Downstream Channel Wells at Tenoroc-4.

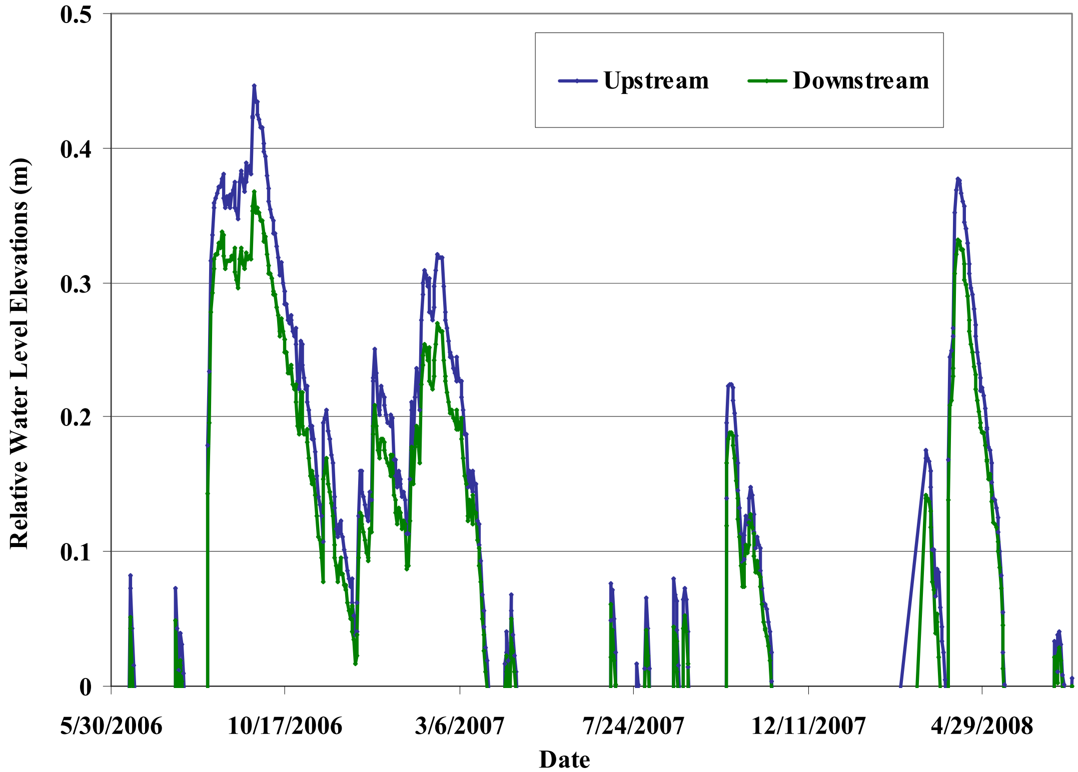


Figure 129. Relative Water Elevations from Upstream and Downstream Channel Wells at Mosaic K5.

A flow measurement was performed on 9/26/2006 at the upstream well using an ADV velocity meter and the velocity-area flow calculation technique, where velocities were measured at 6/10 depth and at 50 cm increments across the width of the channel. The resulting flow was 0.078 m^3/sec. The cross sectional profile of the channel at the upstream well was obtained as relative to the ground surface of the well, which was the lowest point in the profile (Figure 130). The profile was treated as a triangle to determine cross-sectional area and wetted perimeter with surface water levels at the upstream well at the time of velocity measurements. The measured flow and these parameters were used along with Equation 15, solving for the Manning's roughness coefficient which equaled 0.256 sec/$m^{1/3}$. Other measurement attempts of velocity were unsuccessful due to dry conditions in the channel during field visits. The resulting coefficient with the one measurement is on the high end for channels, which typically range from 0.1 to 0.15 for channels with high plant growth (Dingman 2002). However, a coefficient of 0.25 is often used for dense, vegetated swales. The high coefficient calculated from the one velocity measurement seems appropriate, as the channel was dominated by dense *Typha spp.*, including significant detritus.

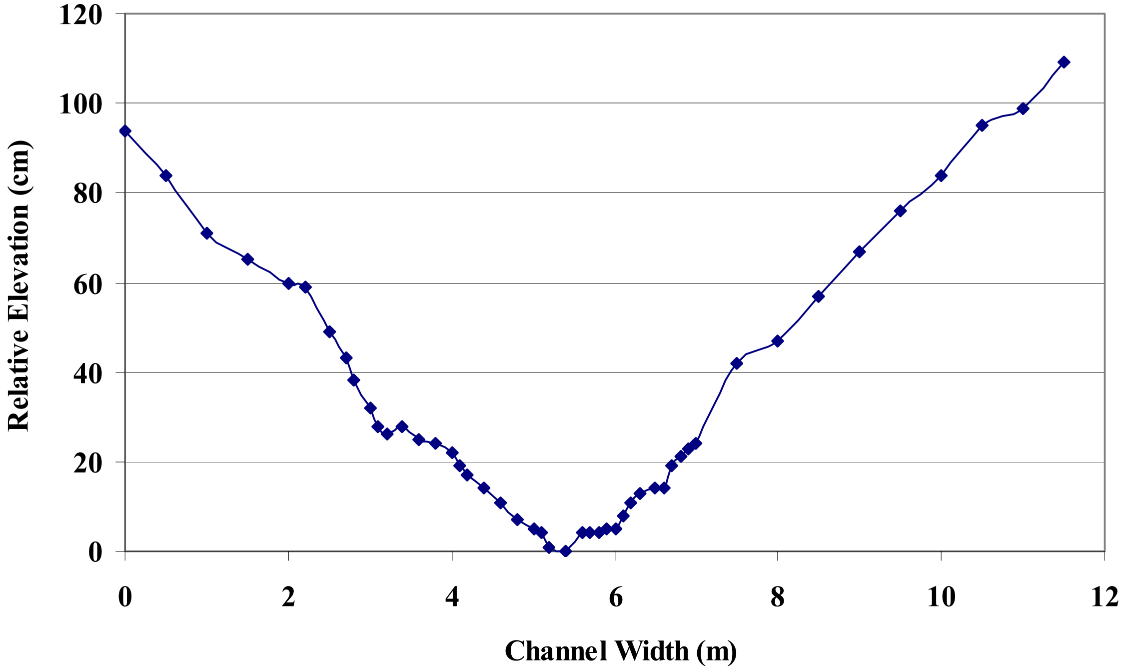


Figure 130. Channel Cross-Sectional Profile at the Upstream Well at Mosaic K5.

Using the continuous well data to determine cross sectional area and wetted perimeter and the daily average head difference along with Equation 15 and a Manning's coefficient of 0.256 sec/$m^{1/3}$, average daily flow (m^3) was calculated (Figure 131). The channel was visited on 8/17/06 when flow was occurring and it was determined that there were at least three connection points to SW-1 which facilitated flow from SW-1 to the channel. The channel does, however extend the whole length of the east side of the CSA and it could not be determined if flow from other isolated features would be possible. Assuming that the majority of channel flow was from SW-1 and that direct rainfall to the ditch was negligible, the outflow from SW-1 was calculated in terms of depth/day. The upland-to-wetland ratios as function of stage were used along with the flow data to determine daily outflow (cm) from SW-1 (Figure 132). During the 1.68 years that were analyzed for channel flow, 64.77 cm of outflow from SW-1 was estimated, resulting in 5.3 E^5 m^3 of total flow.

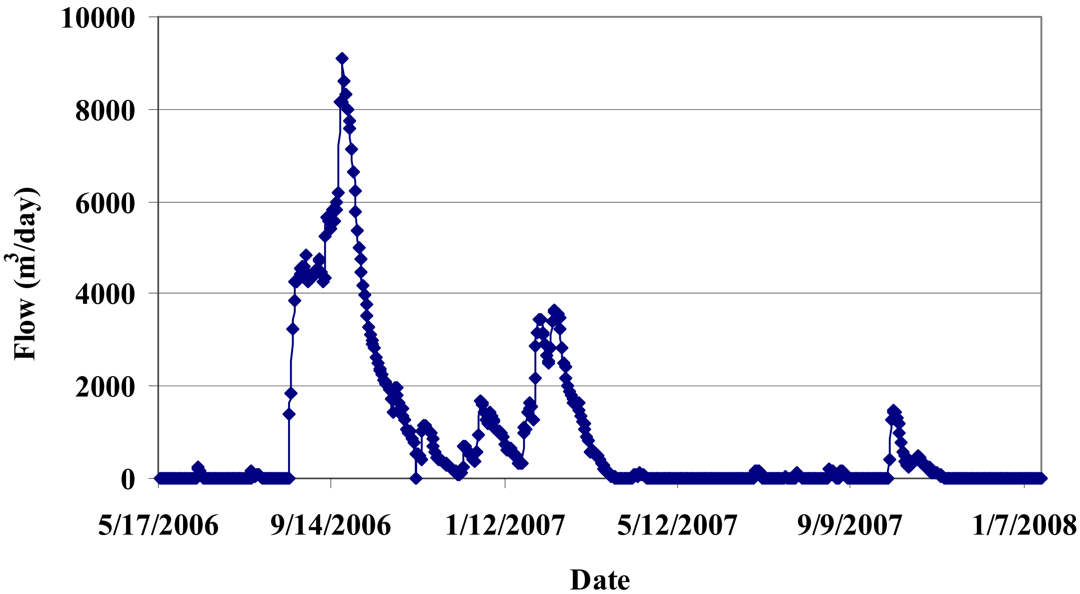


Figure 131. Channel Flow at Mosaic K5 (m^3/day).

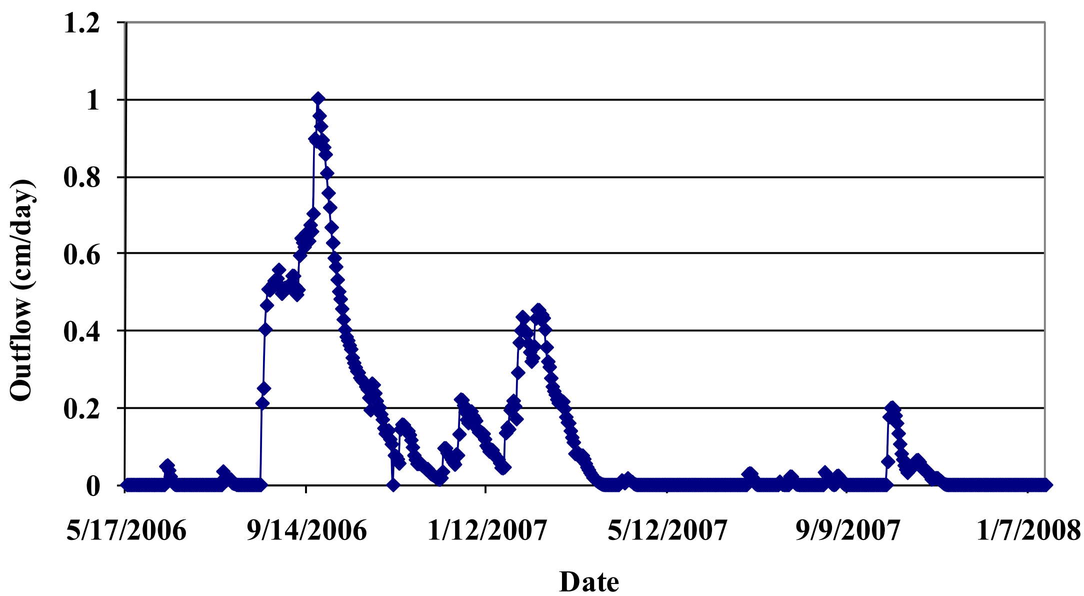


Figure 132. Channel Flow at Mosaic K5 (cm/day).

Two weirs exist at PCS SA 10 that created outflow when SW-1 had water levels of 1.47 m or above, which only occurred during the first part of the site's record, 8/5/05 to 4/27/06 (Figure 63). During this time, one of the weirs experienced short-circuiting of flow through cracks in the structure, thereby restricting any calibration of the weirs. The structure was repaired during a drier time in May 2006 to allow calibration after subsequent increases in water levels. Water levels never increased enough, however, to create outflow and therefore calibration was not performed. As discussed earlier, PCS

SA 01 had an active outfall which was affected by a beaver dam that was witnessed in early spring 2005. Installation of inserts in the structure during May of 2006 successfully blocked outflow and allowed more accurate runoff modeling and water balances.

Williams Co. has an outfall via a pipe through its dike which was visited on numerous field visits and when the site had the highest surface water levels recorded during the study period. At these times, surface water levels never reached the pipe invert and were always at least one meter below the invert. The pipe flow and ditch network at Mosaic HP-10 was never instrumented to provide data for flow analysis. It was determined with LiDAR elevations and site visits during pipe flow events that flow occurred at the site when surface water levels were above 0.45 m, which happened quite frequently (Figure 75). CFI SP-1 has a small channel which connects its two lobes. The instrumented feature is on the east lobe that drains to the west lobe which has an outflow off the entire CSA. Flow was only witnessed through the small connection channel during summer 2004 and it was determined that this occurred when surface water levels were approximately 0.80 m or above. Since the feature was instrumentated in fall 2004, water levels reached this level only once during a three day period in April 2005. Mosaic H1 had no active outflow paths from the entire site during the period of study.

Water Balances

Water balances were calculated for a surface water feature at all hydrology sites except for Tenoroc-4 due to its limited periods of inundation. Additionally, another surface water feature at Mosaic H1, SW-3, was evaluated. Analysis was performed for periods of time where constant flooding occurred. Since the outfalls at PCS SA 10 were active only in the beginning of the record, the balances were calculated for the period since the structure became inactive. Similarly, analysis for PCS SA 01 was done only for the period following the installation of inserts in the weir. Two different time periods, 239 and 238 days, for Mosaic H1 were evaluated where surface water data was positive. Four different periods of records at Mosaic K5 were analyzed including time periods when no channel outflow occurred and one longer period, 5/18/06 to 1/18/08, which had intermittent channel outflow and that encompassed some of the shorter times evaluated.

Water balances were calculated using Equation 17 and the assumption of zero infiltration along with rainfall, the various predictive runoff models, estimated daily ET with the Penman method, and daily surface water data. The channel well data and flow equation were included for the analysis of Mosaic K5. As a result of some of the sites having intermittent outflow during the period of analysis, actual stage responses could not be used to determine runoff contributions to the balances, as stage responses may also have been influenced by the outflow in addition to runoff and rain. Therefore, the predictive runoff models that were developed for times when there was no outflow were used to determine total runoff depths for the water balances. The predictive linear models with non-zero y intercepts were used for all sites except for Mosaic HP-10, where the multiple regression model was used since it performed better. While the multiple regression model also performed better for Williams Co., the results in Table 55 suggested the linear model may be adequate. Water balances, along with the residual totals and as depth per day for all evaluated features, are presented in Tables 57 and 58. Mosaic K5's water balances are shown separately in Table 58 with the inclusion of the channel flow. Since Mosaic HP-10 had frequent outflow that was not instrumented, its resulting residual includes outflow. Total runoff amounts were calculated as percentages of incoming rain for each feature (Table 59).

Table 57. Water Balance Results.

Site	Time Period	Rain (m)	Runoff (m)	ET (m)	Δ Storage (m)	Residual (m)	Residual/Day (mm)
Mosaic H1 SW-3	7/7/07 – 7/9/08	1.13	1.04	1.63	0.22	0.32	0.88
Mosaic HP-10	4/4/06 – 2/12/08	2.00	2.80	2.46	-0.09	2.43	3.98
PCS SA 01	6/1/06 – 11/27/07	1.48	0.65	2.33	0.45	-0.65	-1.19
CFI SP-1	10/10/04 – 4/10/06	1.94	0.38	2.23	-0.51	0.60	1.11
Mosaic H-1	7/26/05 – 3/22/06	0.57	0.69	0.87	-0.64	1.03	4.32
Mosaic H-1	7/12/06 – 3/6/07	0.79	0.98	0.83	-0.08	1.02	4.29
Williams Co.	6/14/05 – 4/4/06	0.87	1.00	1.04	-0.61	1.44	4.87
PCS SA 10	4/26/06 – 6/27/08	2.25	1.30	3.06	-0.43	0.92	1.16

Table 58. Water Balance Results for Mosaic K5.

Time Period	Rain (m)	Runoff (m)	ET (m)	Channel Outflow (m)	Δ Storage (m)	Residual (m)	Residual/Day (mm)
5/18/06 – 1/18/08	2.06	1.54	2.54	0.65	0.28	0.13	0.21
3/13/06 – 8/15/06	0.62	0.49	0.85	0.00	-0.08	0.34	2.17
3/27/07 – 10/4/07	0.76	0.57	1.03	0.00	-0.09	0.39	2.02
11/10/07 – 3/5/08	0.17	0.10	0.29	0.00	-0.05	0.03	0.29

Table 59. Runoff Depth as Percentages of Rain Depth.

Site	Time Period	Runoff/Rain (%)
Mosaic H1 SW-3	7/7/07 – 7/9/08	92.12
Mosaic HP-10	4/4/06 – 2/12/08	139.68
PCS SA 01	6/1/06 – 11/27/07	44.14
CFI SP-1	10/10/04 – 4/10/06	19.54
Mosaic H-1	7/26/05 – 3/22/06	119.79
Mosaic H-1	7/12/06 – 3/6/07	123.35
Williams Co.	6/14/05 – 4/4/06	115.19
PCS SA 10	4/26/06 – 6/27/08	57.68
Mosaic K5	5/18/06 – 1/18/08	74.90

Similar to the runoff analysis results, CFI SP-1 experienced the lowest runoff amounts, followed by the two PCS sites, then Mosaic K5 and H1 SW-3 (Tables 57-59). The highest runoff contributions to the water balances occurred at Mosaic HP-10, Mosaic H1 SW-1, and Williams Co. (Tables 57 and 59). The runoff amounts as percentages of rain are similar to the results of the slope of event response analysis (Tables 59 and 53). The runoff amounts calculated in the water balance of Mosaic HP-10, however, are much lower than what is predicted with the event response slope minus one due to the multiple regression model that was used in the water balance, which predicts less runoff (Tables 59 and 53). The runoff amounts in the water balances of Mosaic H1 SW-1 and Williams Co. are also less than what the regression slope analysis obtained, but to a lesser extent (Tables 59 and 53). The linear models were used in the water balances of these sites had y intercepts that were negative, which cause the predicted values to be less than what would be predicted using the regression lines with y intercepts equaling zero (Figures 118 and 122). The negative y intercepts represent thresholds of rain depths required to produce runoff events. The relative comparison of the runoff experienced by the eight different features is the same as found in the runoff analysis demonstrating the different runoff characteristics among and within CSAs.

The water balances of the different time periods for Mosaic K5 suggest that the channel outflow contribution was over-predicted with the channel well data and flow equation (Table 58). The two periods without channel flow and that include summer months resulted in daily residuals of over 2 mm, while the period of fall 2007 and winter 2008 had a residual of 0.29 mm/day. The long period which included times of channel flow and had it estimated with well data and flow equation had a daily residual of 0.21 mm. This residual is similar but smaller than the 0.29 mm/day experienced in the winter without channel flow. Since the time period with channel flow included 2 summers and 1.5 winters, its residual should not only be higher than what was experienced in the winter but approximately slightly higher than the average of the summer residual and winter residuals from the time periods without channel outflow. The fact that the residual is much lower than this, may suggest that the channel flow may be somewhat overestimated, causing a decrease in the residual. While the channel flow amounts may be fairly near what actually was outflow from the entire CSA, it includes direct rainfall to the channel itself and possibly flow from other features within the CSA that are isolated from the feature evaluated here.

The differences among the daily residuals from the water balances of all sites are apparent (Table 57). Excluding Mosaic HP-10 and Mosaic K5 with channel flow, Williams Co and Mosaic H1 SW-1 have the highest residuals, which are over 4 mm/day. The residuals for PCS SA 10, Mosaic H1 SW-3, and CFI SP-1 are comparable and near 1 mm/day, demonstrating the differences both among sites and within one site (Mosaic H1 SW-1 and SW-3). The analysis of PCS SA 01 resulted in the only negative residual, representing an inflow that may not have been accounted for in the balance. The positive residuals found in the balances of all other sites suggest underestimation of an outflow, be it ET and/or infiltration.

Dike Seepage

The water balances suggested that some underestimation of outflow occurred at most sites. Since surface water outflow did not occur during the times analyzed, with the exception of at Mosaic HP-10 and during one of the four analyzed periods of Mosaic K5, infiltration and/or underestimation of ET are reasonable explanations for the residuals.

Infiltration was defined as both the loss of surface water vertically to the local groundwater system and laterally across the dike as seepage (Figure 11). Three sites that had surface water in contact with the dikes were instrumented with a well on the dike and one on the downward slope of the dike (Figures 1, 2, and 4**)**. Discrete water levels within the wells were recorded during site visits. The water levels within the dike wells and surface water data from the wells in the adjacent surface water feature were used to develop groundwater profiles. Elevations of the ground surfaces of the wells were used to determine the profiles as relative elevation differences, with zero representing the elevation at the surface water well. Groundwater profiles recorded at the dike wells on different dates are shown for PCS SA 10, Mosaic H1, and PCS SA 01 in Figures 133, 134, and 135. The profiles are shown relative to land elevation profiles, and the water level elevation at zero distance represents the surface water levels in contact with the dikes. PCS SA 01 had an additional well installed at the toe of its dikes (Figure 135).

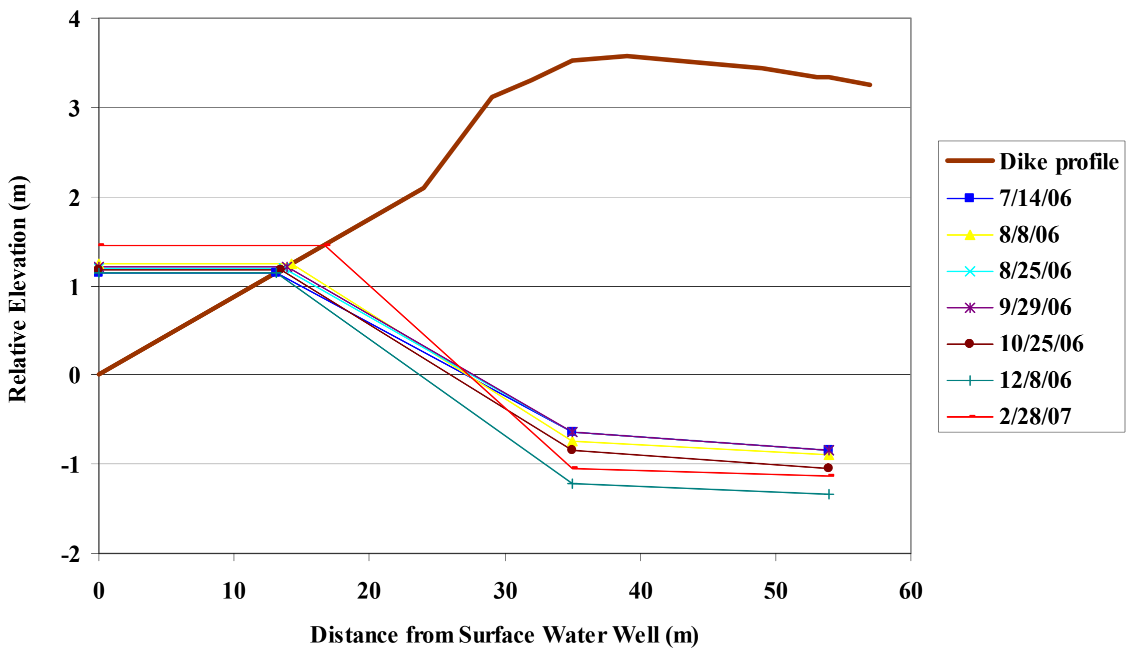


Figure 133. Groundwater Profiles Across the East Dike of PCS SA 10.

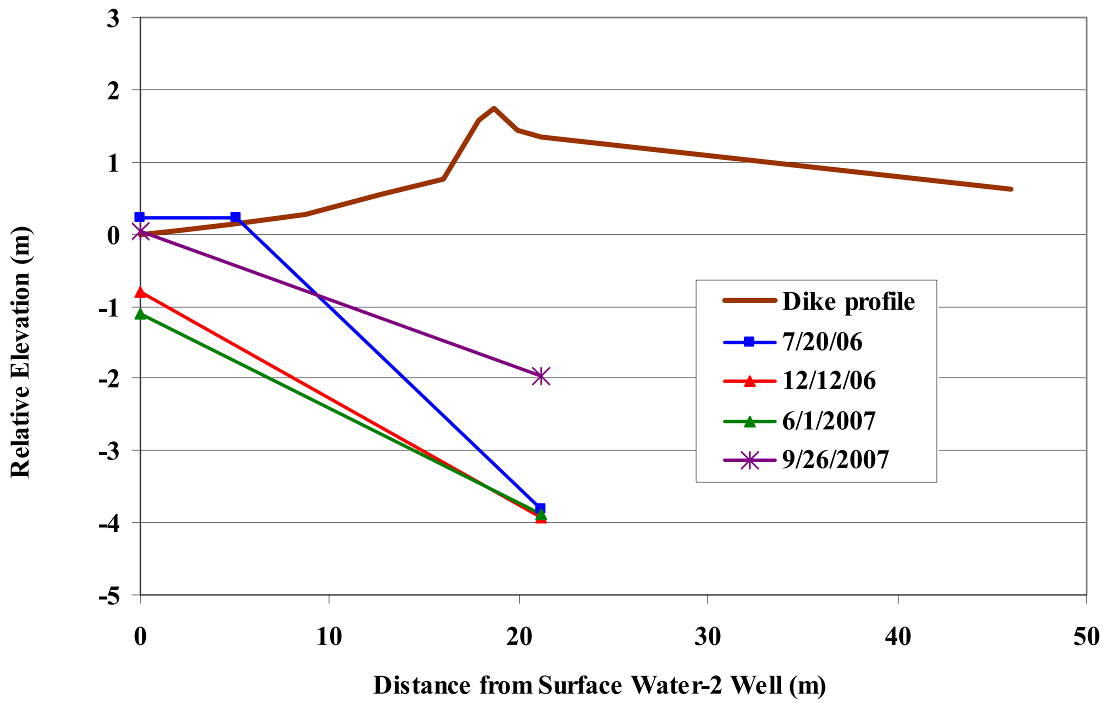


Figure 134. Groundwater Profiles Across the West Dike of Mosaic H1.

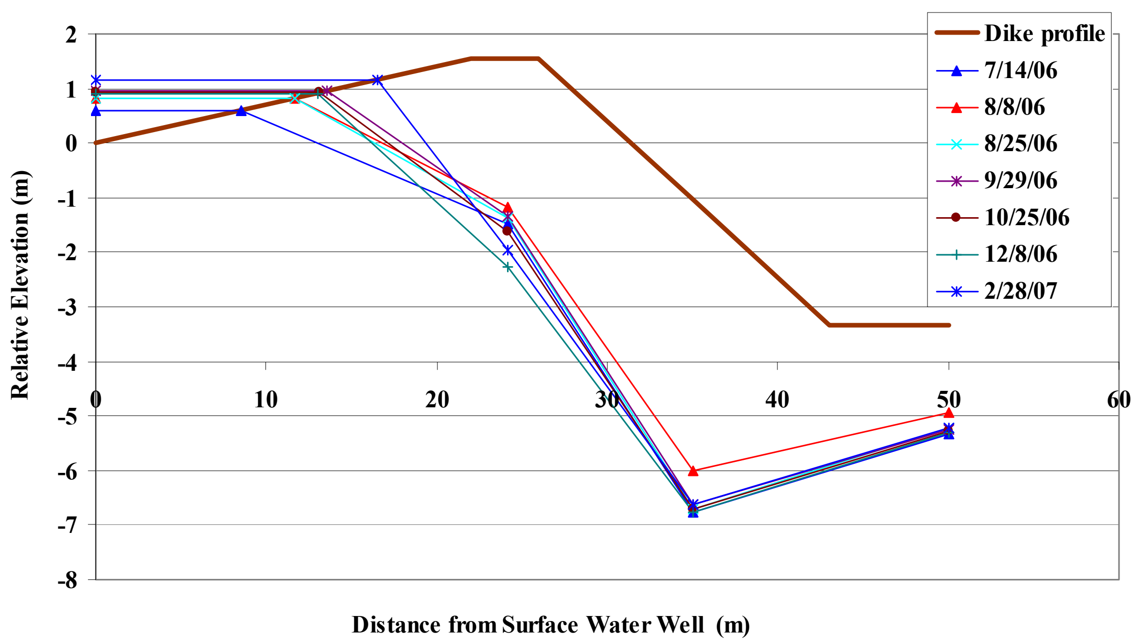


Figure 135. Groundwater Profiles Across the South Dike of PCS SA 01.

Measurements taken at the down slope well of Mosaic H1 are not shown (Figure 134). This well was installed 46 m away from the surface water well with a depth of only 4.2 m due to equipment failure and was dry during all site visits. The SW-2 well at

Mosaic H1 only experienced inundated conditions on two of the four measurement periods shown, and with minimal depths. Furthermore, only four measurement dates for Mosaic H1 are shown as a result of ponded conditions occurring on the dike due to rain the day before other site visits. The profiles across the dikes of the three sites indicate there were gradients to provide lateral dike seepage out of the CSAs.

Slug tests performed in the wells installed on top of the dikes and the Horslev method (Equation 18) were used to calculate average lateral saturated hydraulic conductivities, which are listed along with standard deviations in Table 60. Slug tests were done on three different occasions at each site. The average conductivities for the dike wells were used along with the groundwater profiles and Equation 19 to calculate dike seepage for each site. The steepest hydraulic gradient from the recorded profiles and the associated surface water levels were used to find maximum dike seepage from the record. Surface water levels used for PCS SA 10, Mosaic H1, and PCS SA 01 were 1.45, 0.22, and 1.16 m, respectively. Calculated dike seepages (cm/yr) for each site are shown in Table 61. The depth of water resting on the dike is represented by D, and seepage was calculated for D equal to the actual surface water in contact with the dike at the time of the steepest hydraulic gradient and to 1, 5, and 10 m to evaluate the sensitivity of the calculation to this parameter.

Table 60. Average Lateral Hydraulic Conductivities at Dike Wells.

Site	K_{sat} (cm/sec)	Std Dev
PCS SA 10	2.66E-05	1.60E-05
PCS SA 01	3.83E-06	1.48E-06
Mosaic H1	1.54E-06	5.49E-07

Table 61. Calculated Dike Seepage (cm/yr).

D (m)	PCS SA 10	PCS SA 01	Mosaic H1
Surface water	0.69	0.77	0.05
1	0.48	1.99	0.23
5	2.39	3.32	1.13
10	4.79	6.63	2.25

Seepage rates calculated with the actual surface water levels were low for all three sites, with a maximum of 0.77 cm/yr at PCS SA 01, due to its steeper hydraulic gradients (Table 61 and Figure 135). With 10 m of surface water in contact with the dike at PCS SA 01, seepage was estimated to be only 6.63 cm/year. Despite the fairly steep hydraulic gradients, seepage was calculated to be insignificant to the overall water budget as a result of the low measured hydraulic conductivities (Table 60). It should be noted, however, that the conductivities represent point conductivities and other portions of the dikes could have higher conductivities where more significant seepage may occur. Observation of hydrophytic vegetation along the toes of the dikes was done prior to the installation of the wells in an attempt to locate these zones to perform the dike studies. No significant differences in vegetation along the toes were found, however.

Local Groundwater Analysis

Groundwater wells were installed with pressure transducers in all eight hydrology sites (Figures 1-8). Up to three wells were installed per site, and the names of the wells referred to here are in reference to when they were installed versus their relative position to the adjacent surface water features. Groundwater-1 wells were installed along with the original surface water wells and immediately adjacent, within 10 m, to the water feature. Groundwater-2 wells were installed during the summer of 2006 and in upland contributing areas of the water features. The Groundwater-2 wells were installed in the topographic highest point and at least 100 m away from the feature. The pressure transducers used in the Groundwater-1 wells were removed and placed in the Groundwater-2 wells; therefore, Groundwater-1 records cease at the beginning of the Groundwater-2 records. Three sites, Williams Co., Mosaic K5, and Mosaic SP-1, had yet another well, Groundwater-3, later installed at an elevation between the Groundwater-2 and Groundwater-1 wells. Therefore, Groundwater-3 wells were lower and closer to the water feature than the Groundwater-2 wells but higher and farther away than the Groundwater-1 wells. Mosaic HP-10 had only one groundwater well installed, which never recorded water despite being 7.24 m below ground.

The groundwater levels recorded in the Groundwater-2 wells at the seven sites are shown in Figure 136. These levels represent water table depths at a topographic high area within the interior area of each CSA. Depths below ground ranged from 0.35 m to 3.7 m with the shallowest observed at PCS SA 10 and the deepest recorded at Tenoroc-4. Most sites experienced water tables ranging from 1 to 2 m below ground. Tenoroc-4, Williams Co. Mosaic H1, and CFI SP-1 experienced the deepest water tables, and these sites also had the most extensive dry times, which are associated with their deepest water table depths (Figures 136 and 77). Likewise, the sites with the most constant flooding, PCS SA 10, PCS SA 01, and Mosaic K5, experienced the shallowest water tables. Of these sites, PCS SA 01 had the deepest groundwater levels though it experienced deeper inundation than Mosaic K5, which typically had groundwater levels 0.5 m higher than those of PCS SA 01. This observation may be explained by the sand tailings at PCS SA 01 in which the Groundwater-2 well was installed.

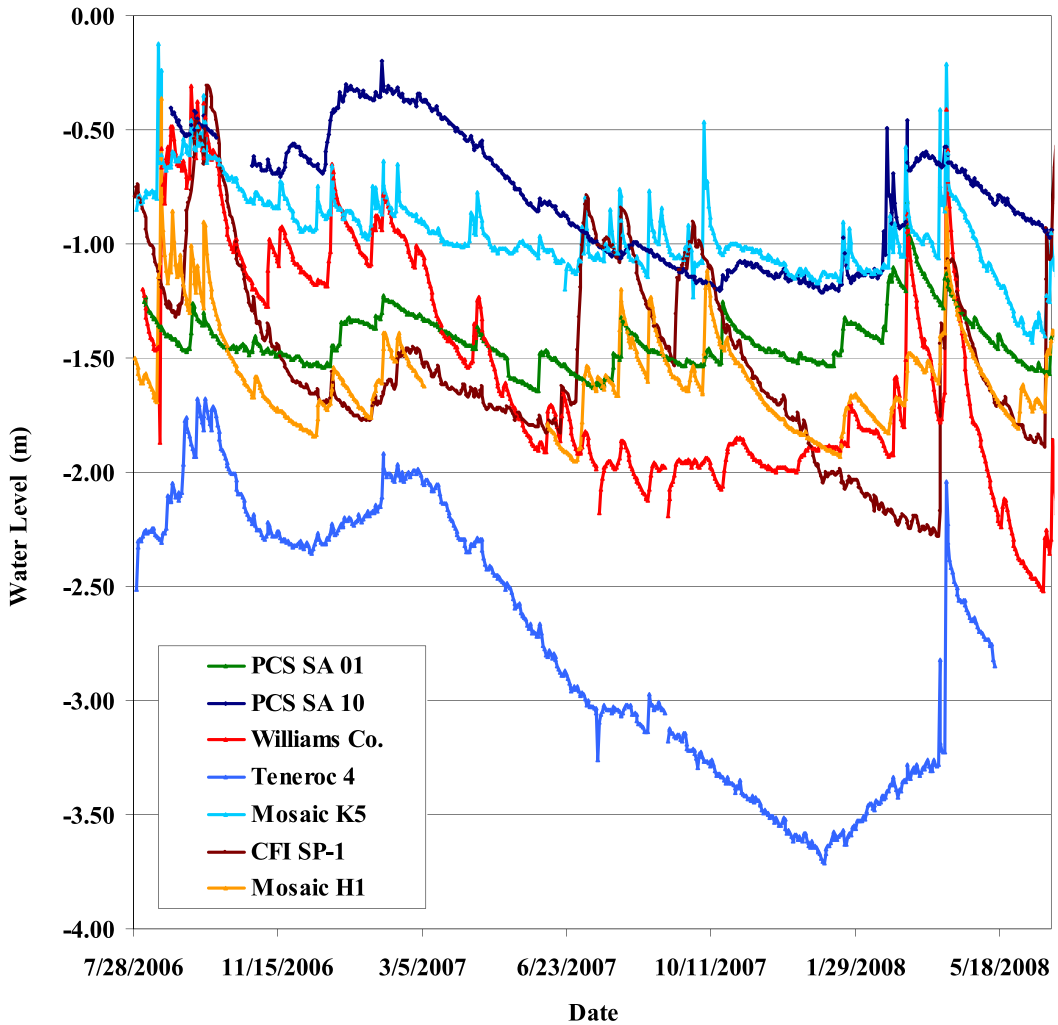


Figure 136. Groundwater Levels at Topographical High Interior Areas.

Slug tests were performed in the Groundwater-2 wells and on three different occasions. The tests and the Horslev Method (Equation 18) were used to calculate saturated lateral hydraulic conductivities of the local groundwater system that the wells penetrated, and the averages are shown in Table 62 along with standard deviations, depth of installation, and the soil type encountered during installation. Also listed in Table 62 are the elevation differences between the ground surfaces of the surface water wells and Groundwater-2 wells. The elevation differences are in reference to the SW-1 wells on all sites except for Mosaic H1. The Groundwater-2 well at Mosaic H1 was installed in an upland area that surrounds the SW-3 feature; therefore, the elevation difference between the SW-3 well and Groundwater-2 well is shown. It was not possible to perform a slug test in the well at Mosaic HP-10, as the well never recorded water. As previously mentioned, the Groundwater-2 well at PCS SA 01 was installed in pure sand tailings, and slug test attempts were unsuccessful since the increase in water levels after the slug was removed was almost instantaneous. The pure clay of the uplands instrumented at Mosaic K5, Williams Co., and Tenoroc-4 resulted in the lowest conductivities while much higher conductivities were measured at Mosaic H1, PCS SA 10, and CFI SP-1, which had sandier conditions. The extremely quick recovery after the slug test and the pure sand at PCS SA 01 suggest that its lateral hydraulic conductivity is higher than the ones measured at the other sites.

Table 62. Average Lateral Hydraulic Conductivities at Groundwater-2 Wells.

Site	K_{sat} (cm/sec)	Std Dev	Elev. Difference (m)	Well Depth (m)	Soil Type
Williams Co.	1.42E-06	2.06E-07	1.64	4.34	Unconsolidated clay
Mosaic H1	1.34E-04	6.94E-05	2.45	3.66	Sandy clay to pure clay
Tenoroc-4	4.55E-06	1.09E-06	1.20	5.23	Unconsolidated clay
Mosaic K5	3.01E-06	1.26E-06	1.44	3.95	Unconsolidated clay
CFI SP-1	1.20E-05	1.69E-06	1.82	4.07	Sandy clay
PCS SA 10	3.77E-04	1.47E-04	3.06	3.16	Sandy clay
PCS SA 01	--	--	2.70	3.68	Sand
Mosaic HP-10	--	--	2.27	7.24	Sand to sandy clay

Utilizing both surveying and LiDAR data, water levels recorded in all groundwater wells were determined relative to the ground surface of the surface water wells. Groundwater-1 and Groundwater-2 levels are shown relative to the elevation of surface water levels at PCS SA 01 in Figure 137. The proximity of the Groundwater-1 well at PCS SA 01 to the instrumented surface water feature resulted in near equal elevations and fluctuations. Though the Groundwater-1 level elevations were slightly less than the surface water level elevations, they were always within 8 cm, a magnitude that could be affected by small errors in surveying. Such a close matching of elevations and signatures demonstrates fairly good connection of the groundwater system immediately adjacent to the water feature. The Groundwater-2 levels, however, were higher relative to the surface water levels and with greater response to rain events and subsequent decline. This surrounding water table was on average 0.30 m higher than the surface water elevation, which suggests the potential for groundwater flow into the water feature that would have been controlled by the lateral hydraulic conductivity. While a conductivity was not measured at PCS SA 01 Groundwater-2 well, it is estimated that it was higher than the conductivities measured at all other sites.

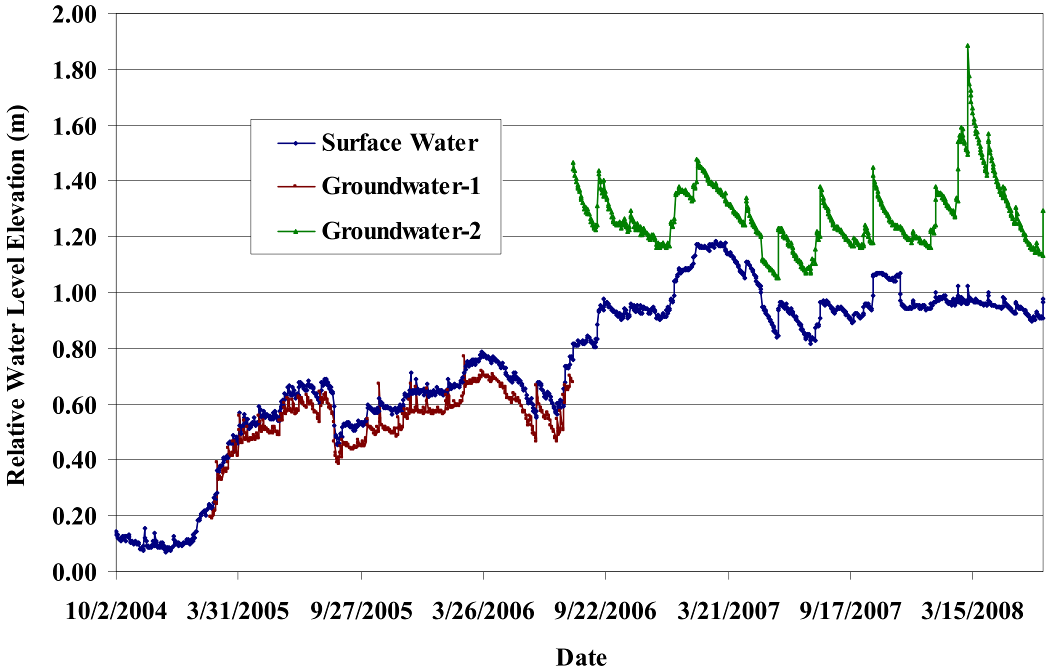


Figure 137. Groundwater Levels Relative to Surface Water at PCS SA 01.

In contrast to what was observed at PCS SA 01, the Groundwater-1 level elevations at PCS SA 10 were lower than the surface water levels and with a different signature (Figure 138). While the proximity of this well to the water feature is comparable to that of PCS SA 01, the well at PCS SA 10 was installed just inside of the dike and on its down slope. The lower elevations recorded in the Groundwater-1 wells suggest a hydraulic gradient that could provide dike seepage, similar to the results found in the dike seepage analysis (Figure 133). The Groundwater-2 well was installed on the other side of the feature and in the interior of the CSA. The water elevations at this well were an average of 1.2 m above the surface water elevations. These results reveal that a hydraulic gradient existed from the interior potentially providing groundwater inflow to the water feature, but that there was also a gradient to allow lateral outflow through the dikes. Each of those flows would have been largely controlled by the conductivities of the respective paths. The measured conductivity at the Groundwater-2 well was an order of magnitude higher than the one measured at the dike well (Tables 62 and 60).

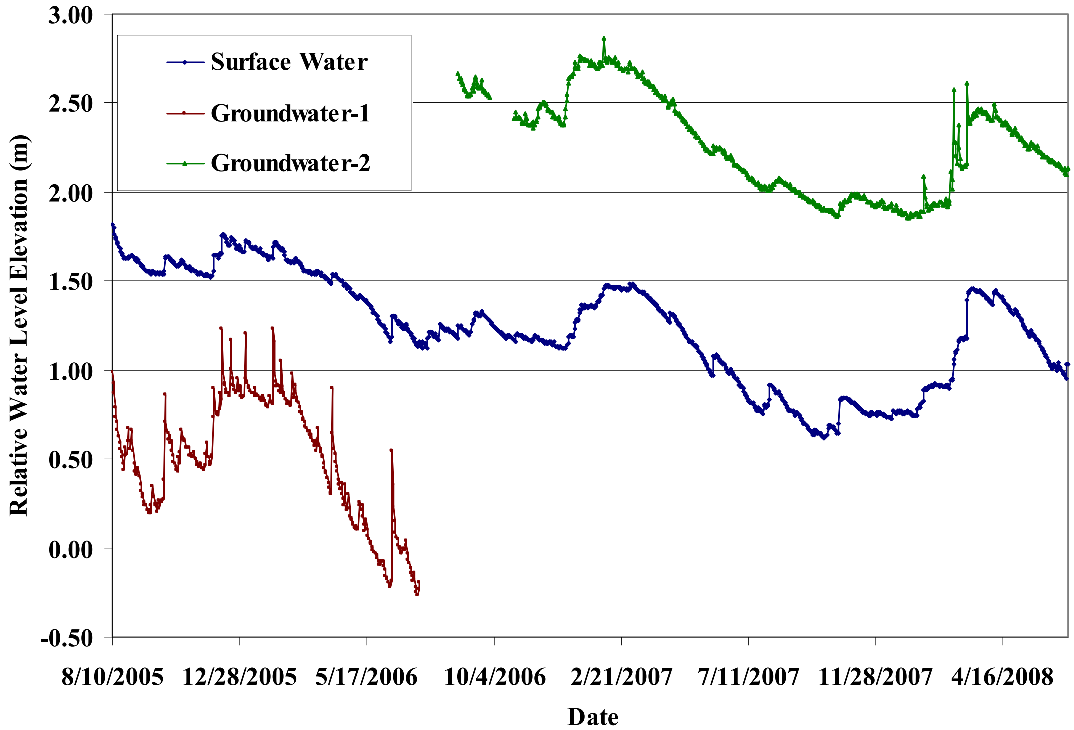


Figure 138. Groundwater Levels Relative to Surface Water at PCS SA 10.

The elevation of the Groundwater-1 levels at Tenoroc-4 were higher but with similar signatures compared to the elevation of the levels recorded at the surface water well, which were often below ground and thus also groundwater levels (Figure 139). Throughout the entire record of Groundwater-2, there was no inundation at the surface water system, and often the surface water well was dry which occurred around 1.5 m below ground. That said, it is hard to conclude much from the relative elevation differences between Groundwater-2 and surface water levels. It appears, though, that the groundwater system at Groundwater-2 may have been lower than the system at the surface water well, but by a maximum of only 0.20 m. The elevation data used to calculate the differences at this site were strictly from LiDAR data and a difference of only 20 cm is close to the accuracy of that technology. Furthermore, nothing can be concluded for what levels would have been at Groundwater-2 during the first part of the surface water data record when much higher water levels were experienced and evidence of potential groundwater flow toward the system was observed.

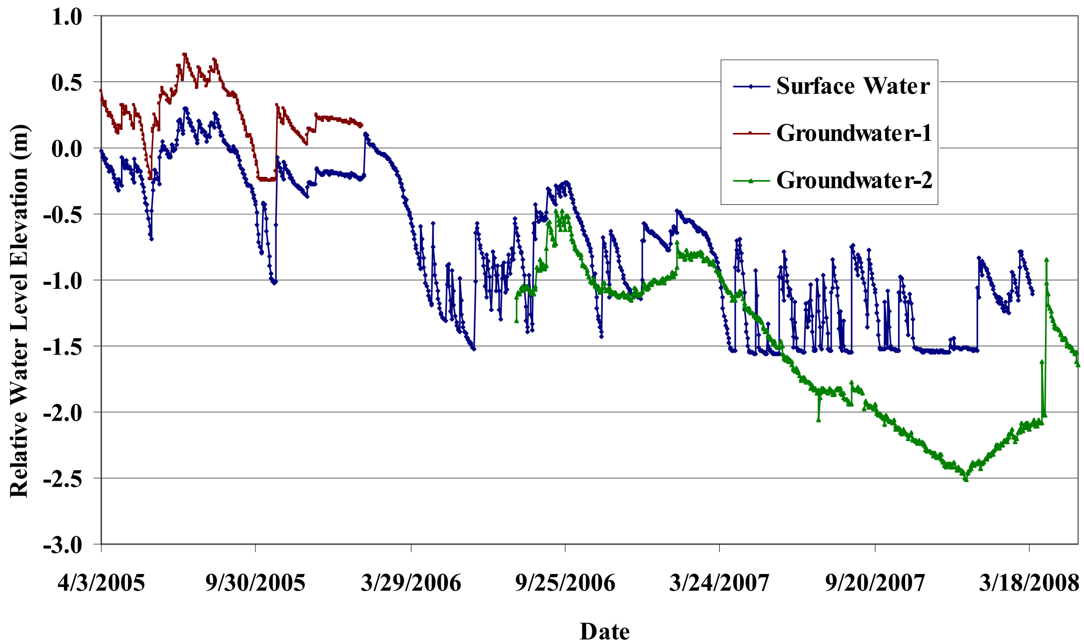


Figure 139. Groundwater Levels Relative to Surface Water at Tenoroc-4.

Data from all three surface water wells and both groundwater wells are shown for Mosaic H1 in Figure 140, with all water elevations relative to SW-1. Similar to Tenoroc-4 and PCS SA 01, the groundwater levels (Groundwater-1) just adjacent to the surface water feature were similar in elevation and signature to the surface water system. The elevations of Groundwater-2 and SW-3 suggest that the hydraulic gradient created the potential for groundwater flow from Groundwater-2 to SW-3. SW-3 elevations were above the water elevations of the other two surface water systems, indicating the potential direction of continued groundwater flow. Again, this flow would have been strongly regulated by the hydraulic conductivities of the system. While the conductivity measured at the Groundwater-2 well was one of the highest measured (Table 62), it should be noted that the sandier upland where the well was installed is not characteristic of the whole site and strictly surrounds the Surface Water-3 system. The majority of the site is pure clay, which may have conductivities similar to the ones measured at Williams Co., Tenoroc-4, and Mosaic K5 (Table 62) and that may limit groundwater flow from feature to feature. This limitation is supported by the fact that all three surface water systems have very different water elevations and signatures. The higher conductive soil and elevated water table that surrounded SW-3 suggests higher potential for groundwater flow into the SW-3 feature (Figure 141).

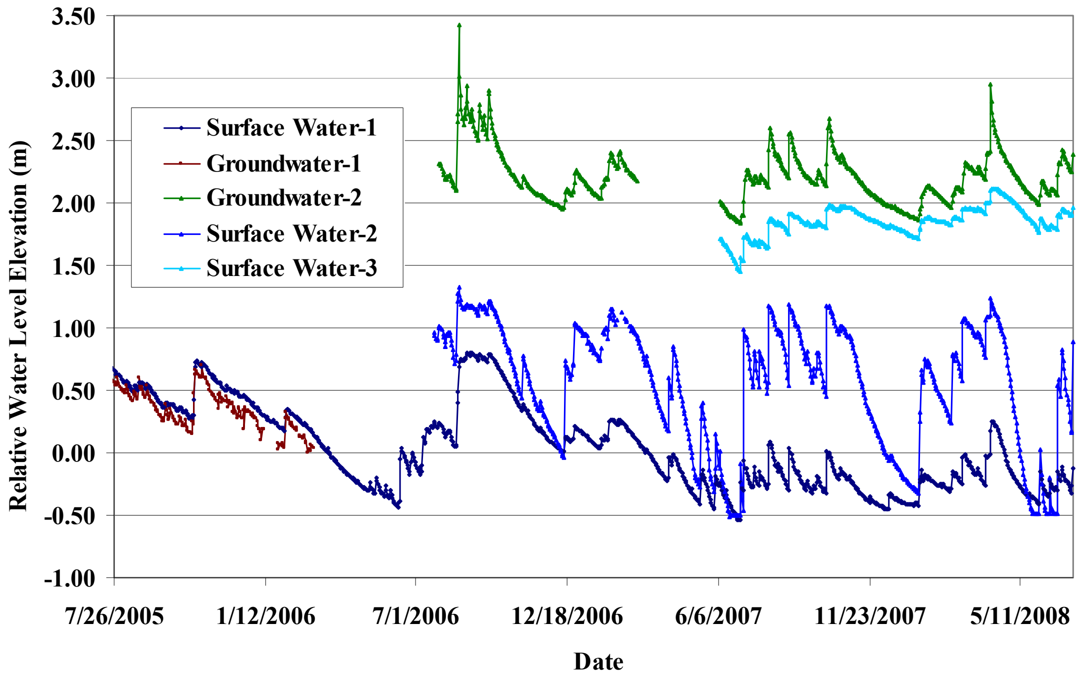


Figure 140. Groundwater Levels Relative to Surface Water-1 at Mosaic H1.

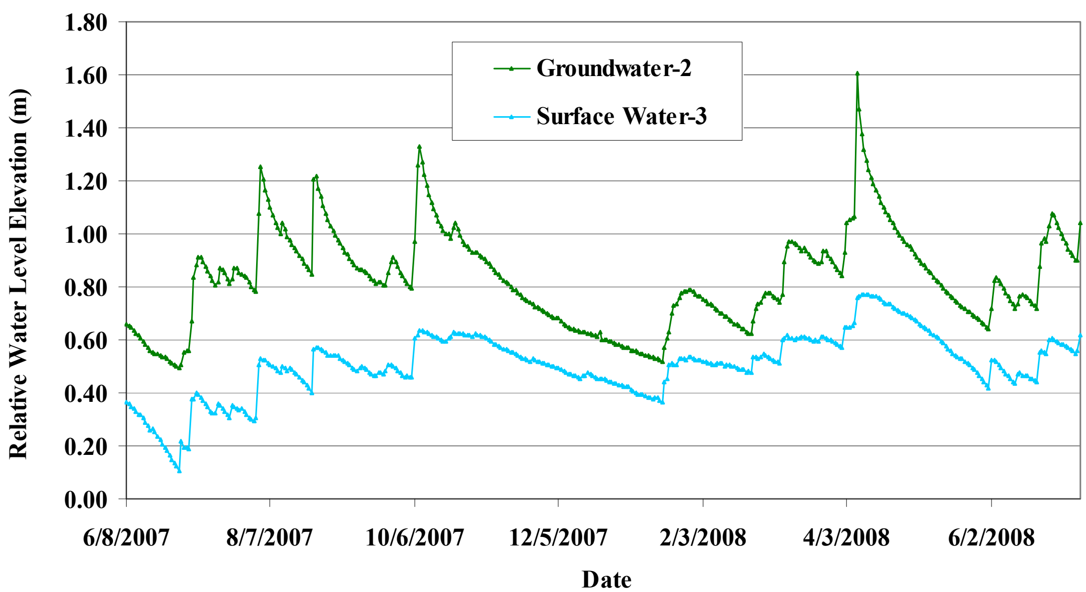


Figure 141. Groundwater Levels Relative to Surface Water-3 at Mosaic H1.

Three groundwater wells were installed at Williams Co. and CFI SP-1. Groundwater-2 well was installed in the most upland area surrounding the surface water feature, and Groundwater-3 in an intermediate zone between Groundwater-2 and the Groundwater-1 wells. As with the other sites, Groundwater-1 wells were just adjacent to the water features. Similar to other sites, both Williams Co. and CFI SP-1 had Groundwater-1 levels with elevations and signatures almost equal to those recorded in the surface water wells (Figures 142 and 143). Caution should be placed when evaluating Figure 143, as both the SW-1 and Groundwater-3 wells became dry after 10/18/07 and values thereafter should be disregarded. Similar to the Groundwater-1 levels, the Groundwater-3 wells had elevations almost equivalent to the levels measured at the surface water wells. It should be noted that no inundation occurred at the surface water wells at either site since installation of Groundwater-3 wells. As such, the similar elevations recorded at the surface water and Groundwater-3 wells demonstrate that a connected local groundwater system had occurred between the wells and in the lower areas of the two sites. The Groundwater-2 elevations of both sites were above the elevations recorded in the surface water and Groundwater-3 wells, suggesting groundwater flow towards these lower areas. The measured hydraulic conductivity at CFI SP-1 is an order of magnitude higher than the one measured at Williams Co., where groundwater flow may have been more limited. The average difference between the Groundwater-2 and the water elevations in the lower areas was 0.97 m at CFI SP-1 compared to 0.72 m at Williams Co.

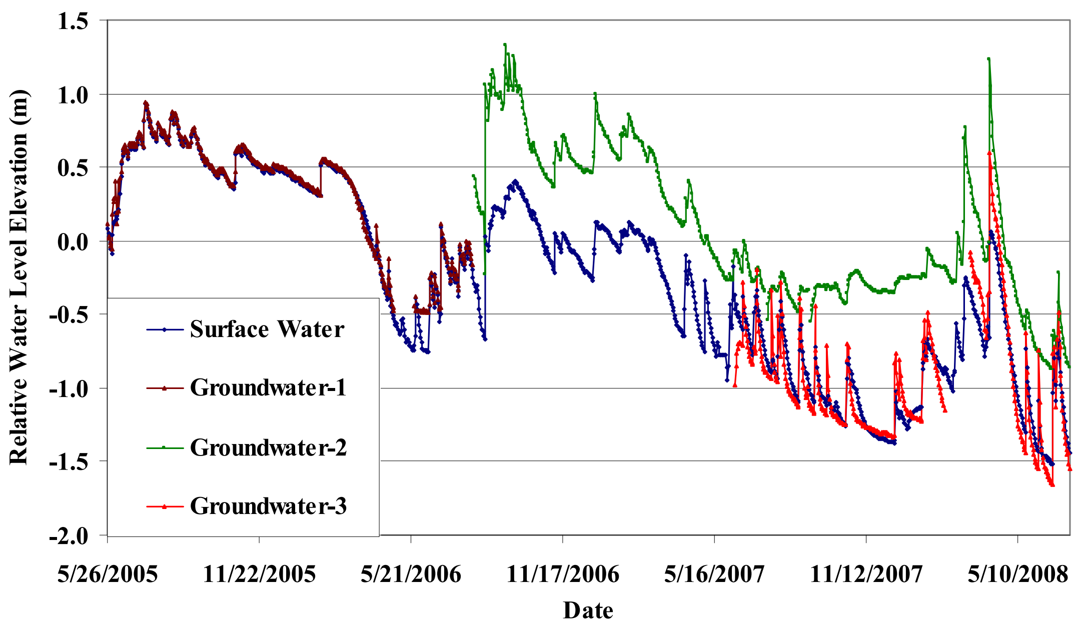


Figure 142. Groundwater Levels Relative to Surface Water at Williams Co.

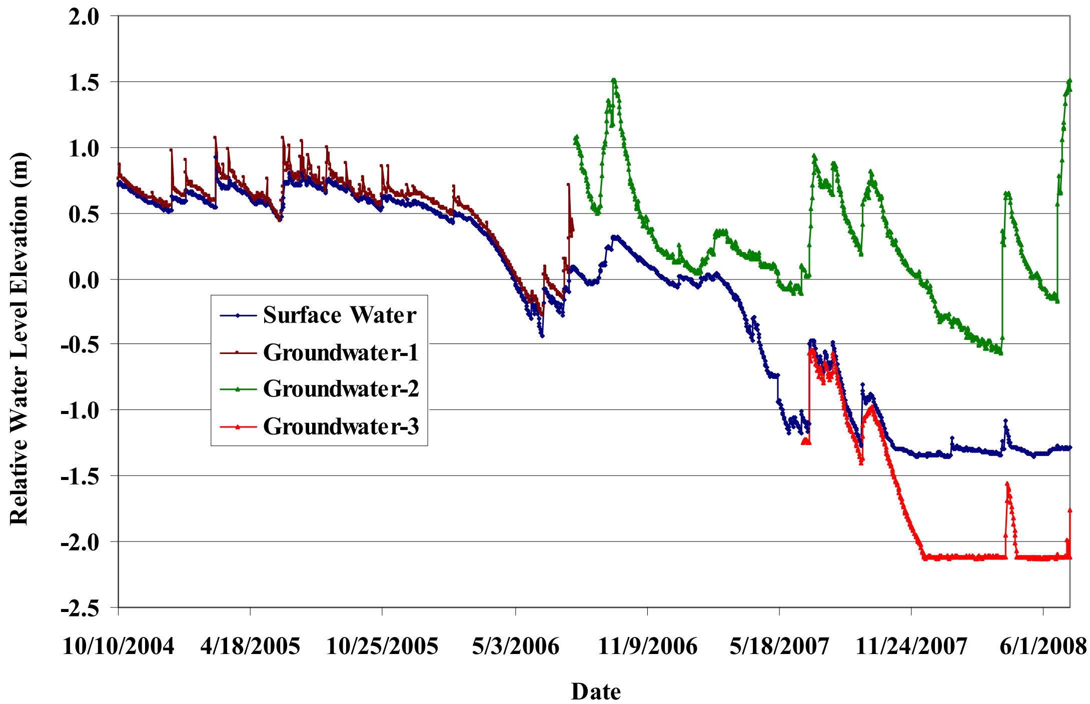


Figure 143. Groundwater Levels Relative to Surface Water at CFI SP-1.

The Groundwater-1 well at Mosaic K5 was installed just inside of the dike that is adjacent to Surface Water-1, whereas the Groundwater-2 was installed on the other side of the feature and more in the interior of the site. This was also the case at PCS SA 10, and similar to its results, the Groundwater-1 water elevations at Mosaic K5 are lower than the surface water elevations, again indicating the potential for groundwater outflow through the dike (Figures 138 and 144). Unlike the other sites, the Groundwater-2 elevations at Mosaic K5 were at times equal, above, or below the surface water elevations (Figure 144). While the elevations were equal for extended period since installation, the onset of summer 2007 resulted in more drying of the surface water feature compared to the Groundwater-2 system. At this time, the elevations indicated the potential for groundwater inflow into the water feature. With increases of surface water levels through the later part of the summer, however, the Groundwater-2 elevations became lower than the surface water, implying that a reversal of potential groundwater flow had occurred. The Groundwater-2 elevations were also above the water elevations recorded in the Groundwater-3 well through most of summer 2007. However, it appears that the Groundwater-2 and Groundwater-3 systems became connected near the time when the Groundwater-2 elevations fell below the surface water elevations, suggesting flow from the water feature towards the now-connected groundwater system may have occurred. Such a transient potential groundwater role was not observed at the other sites, but again this role is limited by the hydraulic conductivity of the groundwater systems.

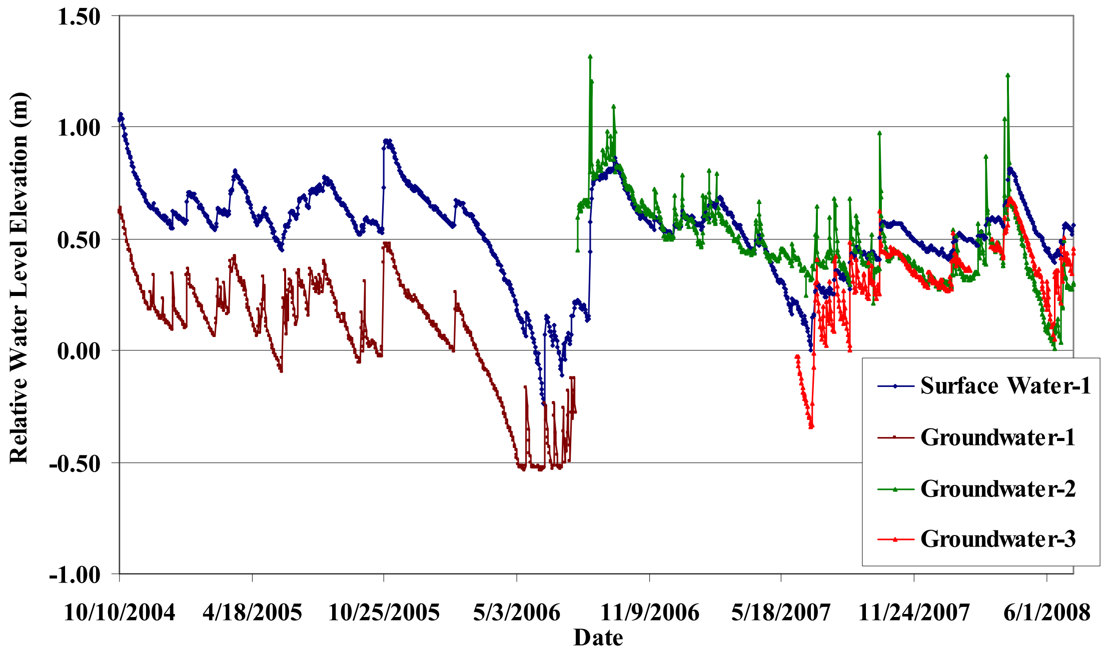


Figure 144. Groundwater Levels Relative to Surface Water-1 at Mosaic K5.

As previously mentioned, the one groundwater well installed at Mosaic HP-10 never recorded water despite the bottom of the well being 4.94 m below the bottom of the surface water well. Such a deep surrounding water table compared to the constantly flooded surface water system suggests that the water features on HP-10 may have been fairly disconnected from and perched above any local groundwater of the system. This site is atypical compared to the other sites and other CSAs, as it had sand tailings deposited on the entire site that were later mined.

Drawdown Analysis

The resulting residuals from the water balance analysis suggested an underestimation of ET and/or infiltration. The dike seepage analysis gave evidence, for at least the sites evaluated, that lateral dike seepage may not have been a significant component of the possible infiltration. An analysis of daily declines of instrumented surface water features was performed in an effort to separate groundwater and ET flows. Average daily losses using the less accurate pressure transducers and calculated with linear regression of surface water levels during time periods with no rain or surface outflow are shown in Tables 63-69. The losses calculated during particular months were averaged, along with their associated correlation coefficients (R^2), and are presented as monthly average daily decline. Also included in the tables are average monthly ET rates estimated with the Penman method. The estimated ET rates were subtracted from the loss slopes to calculate the differences, Δ (mm/day), which are the residual values that represent any error in ET estimation plus potential infiltration losses. The residual values for each site were averaged and are presented at the bottom of the tables. In general, the correlation coefficients suggested good fits by the linear regressions. Tenoroc-4 had the largest residual value, suggesting significant infiltration occurred (Table 65). It should be noted, however, that though the analyses were performed during time periods when the sites were inundated, Tenoroc-4 constantly had low surface water levels used in analysis. At such levels, the surface water declines can be strongly affected and increased by the associated decline in the surrounding soils and their much lower specific yield (Hill and Neary 2007). Caution should be used when interpreting the results from Tenoroc-4. Mosaic K5, Williams Co., and Mosaic H1 also had large residual values which were calculated when there was sufficient surface water (Tables 66, 69, and 133). These results suggest infiltration occurred at these sites. Similar to the water balances, PCS SA 10, PCS SA 01, and CFI SP-1 have the lowest daily decline and thus residuals.

Table 63. Drawdown Analysis Summary for PCS SA 10 SW-1.

Month	Slope (mm/day)	R^2	ET (mm/day)	Δ (mm/day)
May	-7.48	0.91	4.88	-2.60
June	-10.03	0.88	5.51	-4.52
July	-10.67	0.92	6.24	-4.43
August	-7.03	0.74	5.92	-1.11
Sept	-7.01	0.84	5.04	-1.97
			Average =	-2.92

Table 64. Drawdown Analysis Summary for PCS SA 01.

Month	Slope (mm/day)	R^2	ET (mm/day)	Δ (mm/day)
May	-9.14	0.85	4.88	-4.26
June	-6.22	0.78	5.51	-0.71
July	-6.60	0.80	6.24	-0.36
			Average =	-1.78

Table 65. Drawdown Analysis Summary for Tenoroc-4.

Month	Slope (mm/day)	R^2	ET (mm/day)	Δ (mm/day)
June	-17.53	0.92	5.51	-12.01
July	-22.27	0.97	6.24	-16.03
August	-21.72	0.97	5.92	-15.80
Feb	-11.30	0.87	2.67	-8.63
			Average =	-13.12

Table 66. Drawdown Analysis Summary for Mosaic K 5 SW-1.

Month	Slope (mm/day)	R^2	ET (mm/day)	Δ (mm/day)
May	-12.62	0.91	4.88	-7.73
June	-15.24	0.96	5.51	-9.73
August	-11.18	0.84	5.92	-5.26
			Average =	-7.57

Table 67. Drawdown Analysis Summary for CFI SP-1.

Month	Slope (mm/day)	R^2	ET (mm/day)	Δ (mm/day)
Nov	-3.47	0.63	3.06	-0.41
Dec	-2.37	0.52	2.15	-0.22
Feb	-3.81	0.88	2.67	-1.14
March	-5.59	0.98	3.08	-2.51
April	-7.70	0.93	3.73	-3.98
May	-8.13	0.97	4.88	-3.25
July	-8.13	0.85	6.24	-1.89
Aug	-7.37	0.92	5.92	-1.45
Sept	-6.60	0.79	5.04	-1.57
Oct	-6.10	0.93	3.93	-2.17
			Average =	-1.86

Table 68. Drawdown Analysis Summary for Mosaic H1 SW-1.

Month	Slope (mm/day)	R^2	ET (mm/day)	Δ (mm/day)
Aug	-17.72	0.90	5.92	-11.80
Sept	-10.92	0.96	5.04	-5.88
Oct	-9.31	0.91	3.93	-5.39
Nov	-9.40	0.99	3.06	-6.34
Dec	-7.49	0.89	2.15	-5.34
Feb	-6.86	0.96	2.67	-4.19
March	-8.64	0.92	3.08	-5.55
			Average =	-6.36

Table 69. Drawdown Analysis Summary for Williams Co.

Month	Slope (mm/day)	R^2	ET (mm/day)	Δ (mm/day)
July	-23.37	0.93	6.24	-17.13
Aug	-25.15	0.95	5.92	-19.23
Sept	-14.48	0.90	5.04	-9.44
Oct	-9.40	0.94	3.93	-5.47
Nov	-9.40	0.86	3.06	-6.34
Dec	-4.32	0.90	2.15	-2.17
Jan	-4.57	0.83	2.46	-2.11
Feb	-5.33	0.83	2.67	-2.66
March	-13.21	0.99	3.08	-10.13
			Average =	-8.30

The monthly variability in the residual values for most sites also suggests that summer ET may have also been underestimated and contributed to the residuals. The daily declines from all sites were averaged to obtain average daily declines during the growing season and the non-growing season. These average rates are listed in Table 70 along with associated averages for Penman-estimated ET and ET rates measured in lysimeter studies of a Florida *Typha spp.* marsh and *Cladium jamaicense* marsh (Mao and others 2002). The differences between the average daily decline and various ET estimates were 6.15 to 7.94 mm during the growing season compared to differences of 3.9 to 5.2 mm during the non-growing season. Additionally, at some sites summer residual values were substantially higher than ones calculated in the wintertime, indicating that ET may have been underestimated during the summer months (Tables 68 and 69). ET during the winter months also could have been underestimated, just not to the same degree as in the summer. These results suggest that infiltration could be occurring at some sites and at different rates. If it is assumed that infiltration is fairly constant throughout the year, then the seasonal variability in the residuals also suggests that not all of the residual decline can be explained with infiltration and that a portion is due to underestimation of ET.

Table 70. Average Daily Decline and ET Estimates (mm/day).

	Growing Season	Non-Growing Season
Average of 8 sites	12.08	7.27
Penman ET	4.84	3.37
Fl *Typha* Marsh ET[1]	4.14	2.17
Fl *Cladium* Marsh ET[1]	5.93	2.06

[1]Mao and others (2002).

To more accurately and quantitatively separate groundwater and ET flows, highly accurate pressure transducers were deployed in surface water features at six sites during late 2006. The new transducers replaced the less accurate ones in the original surface water wells at Mosaic H1, PCS SA 01, PCS SA 10, and Williams Co., while a new well, SW-2, was installed with the transducer at Mosaic K5. Mosaic H1 had two sites instrumented with the transducers, SW-1 and SW-3. The accuracy of the transducers allowed the diurnal signatures of the water level declines during periods with no rain or surface water outflow to be evaluated. It was assumed that ET is negligible during the night hours, and only surface water levels above 15 cm were evaluated to avoid effects from soil-specific yields. Equation 20 was used to calculate daily ET and infiltration. Periods up to 15 consecutive days when no surface water outflow or rain events had occurred were analyzed for all six sites. Surface water levels at Mosaic K5 SW-2 during June 2008 are shown in Figure 145 along with a closer view of fluctuations in a two-day period. The diurnal signature is evident with declines during the day and almost flat-lined at night. ET rates of 1.62 cm/day and infiltration rates of -0.04 cm/day were calculated using the White method. The minimal negative infiltration, or groundwater inflow to the water feature (exfiltration), indicates that the average daily decline of 1.56 is primarily due to ET.

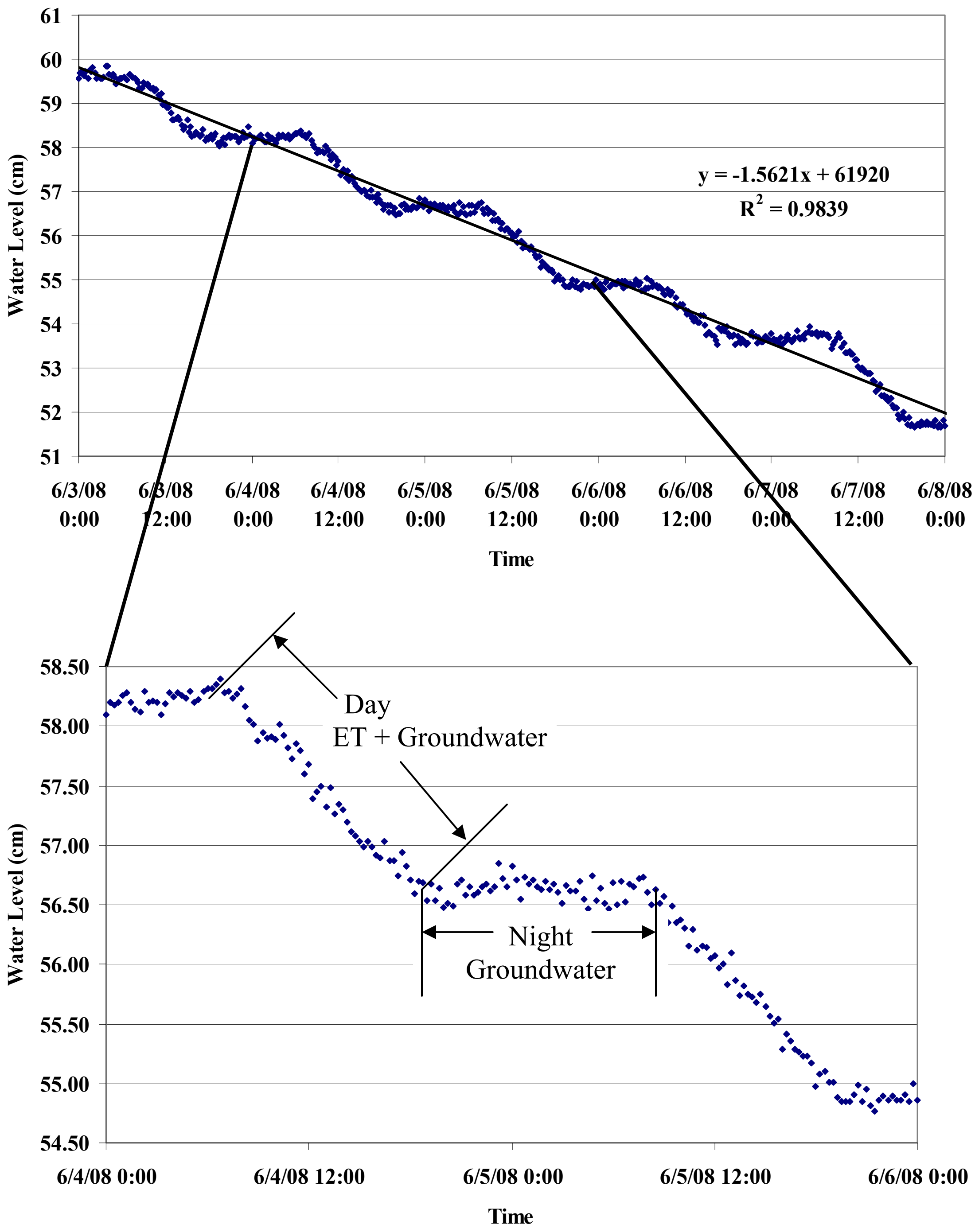


Figure 145. Surface Water Level Decline and Example of White Method at Mosaic K5 Surface Water-2.

Examples of the diurnal curves typically observed at PCS SA 10 and SA 01 are shown in Figures 146 and 147. The associated calculated rates for ET during this period were 1.61 and 1.77 cm/day for PCS SA 10 and SA 01, respectively. While the ET rates were similar at each site, the infiltration rates were much different, which cause different average daily declines as estimated with the linear regression slope. The infiltration rate calculated for PCS SA 10 was -0.54 cm/day compared to -1.35 cm/day at PCS SA 01. Both indicate exfiltration of groundwater flow into the surface water features causing less daily declines than observed during a similar period at Mosaic K5 SW-2 (Figures 146, 147, and 145). It should be noted, however, that at higher levels when SW-2 at Mosaic K5 became connected to SW-1, higher exfiltration rates were observed, suggesting higher exfiltration occurred at SW-1 compared to SW-2 when the two were not connected. The exfiltration rates observed with nighttime surface water levels at PCS SA 10, PCS SA 10, and Mosaic K5 were not constant rates. The surface water levels increased at a fast rate immediately following sundown but this rate decreased with time and to almost zero at PCS SA 10 (Figure 146). While much more pronounced at PCS SA 10, this phenomenon was still apparent at PCS SA 01 (Figures 146 and 147). The decreasing rate of groundwater inflow at night demonstrates transient groundwater flows that are highest when the gradient is the maximum just after sundown. The White method assumes the groundwater flow rate that is experienced during the day is equivalent to the rate observed at night. The method used here somewhat deviated from the original White method since it took the conservative estimate of nighttime exfiltration/infiltration rates by calculating it with surface water levels from 10 pm to 6 am. Therefore, this calculation did not include the faster rates observed from sundown to 10 pm.

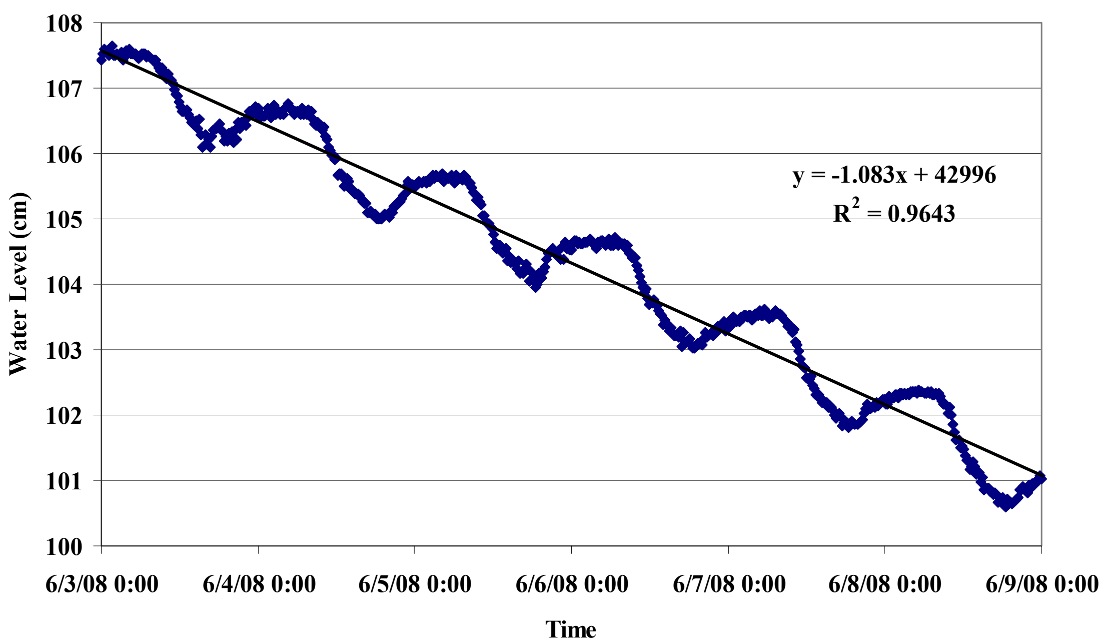


Figure 146. Surface Water Level Decline at PCS SA 10.

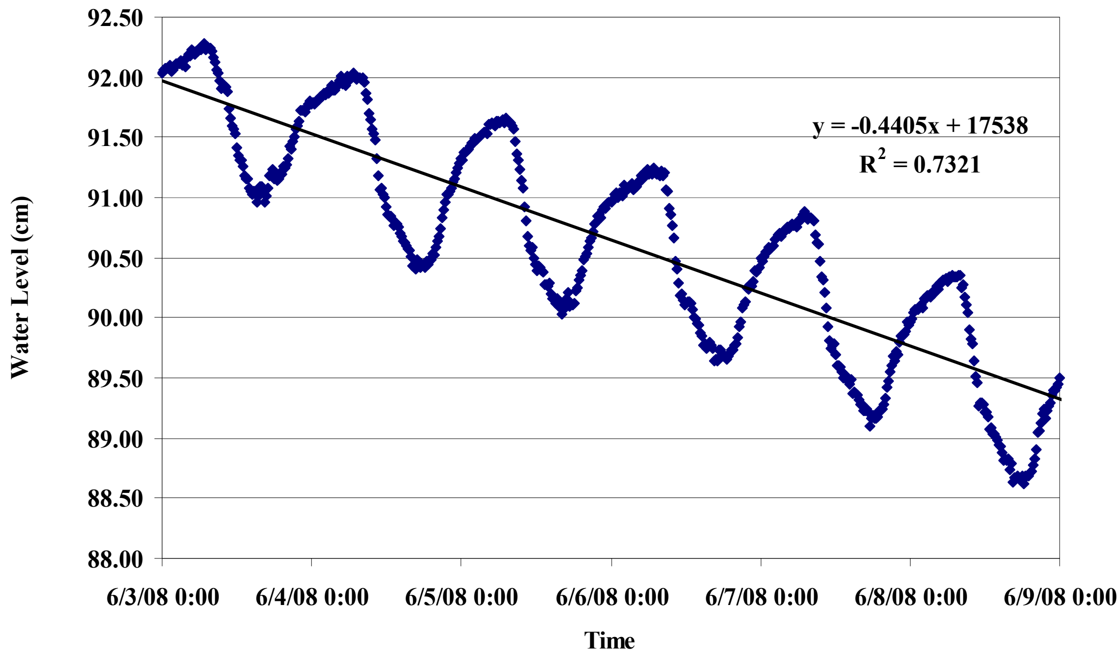


Figure 147. Surface Water Level Decline at PCS SA 01.

Surface water fluctuations observed at Mosaic H1 SW-3 that resulted in an calculated ET rate of 0.6 cm/day and an infiltration rate of -0.8 cm/day are shown in Figure 148. The transient groundwater rates at night were even more obvious at this site, and often surface water level diurnal signatures switched and were more similar to those seen in Figure 145.

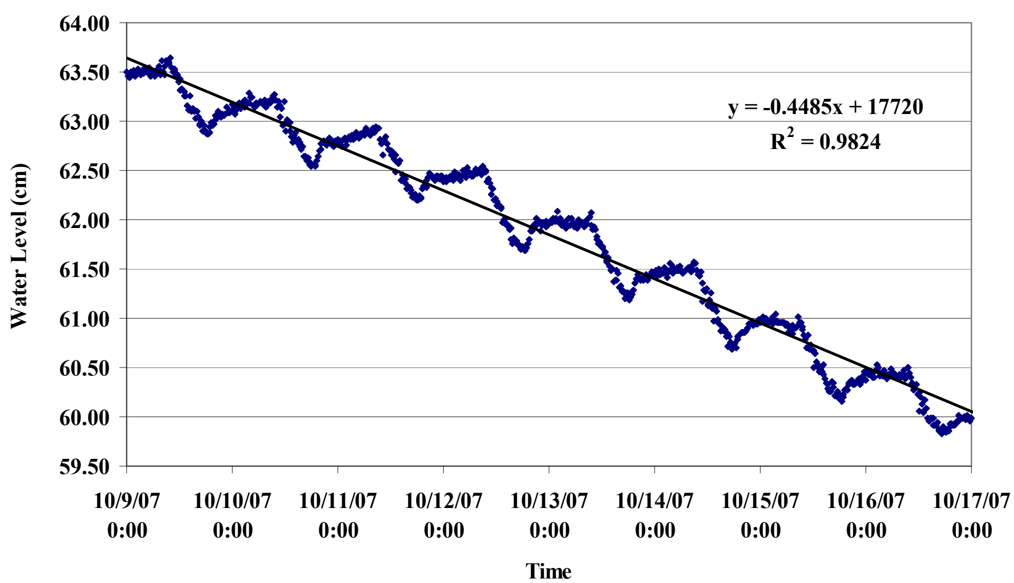


Figure 148. Surface Water Level Decline at Mosaic H1 SW-3.

There were limited periods with surface water levels over 15 cm at Mosaic H1 Surface Water-1. Winter surface water level decline is shown in Figure 149, where the immediate rise in levels after sundown is again apparent but is then followed by declines caused by infiltration events. ET was calculated to be 0.39 cm/day and average infiltration was 0.49 cm/day. Positive infiltration was also calculated for all other inundated periods at Mosaic H1 SW-1. The instrumented features at both CFI SP-1 and Williams Co. never were sufficiently inundated since installation of the new transducers; therefore, surface water declines were not evaluated.

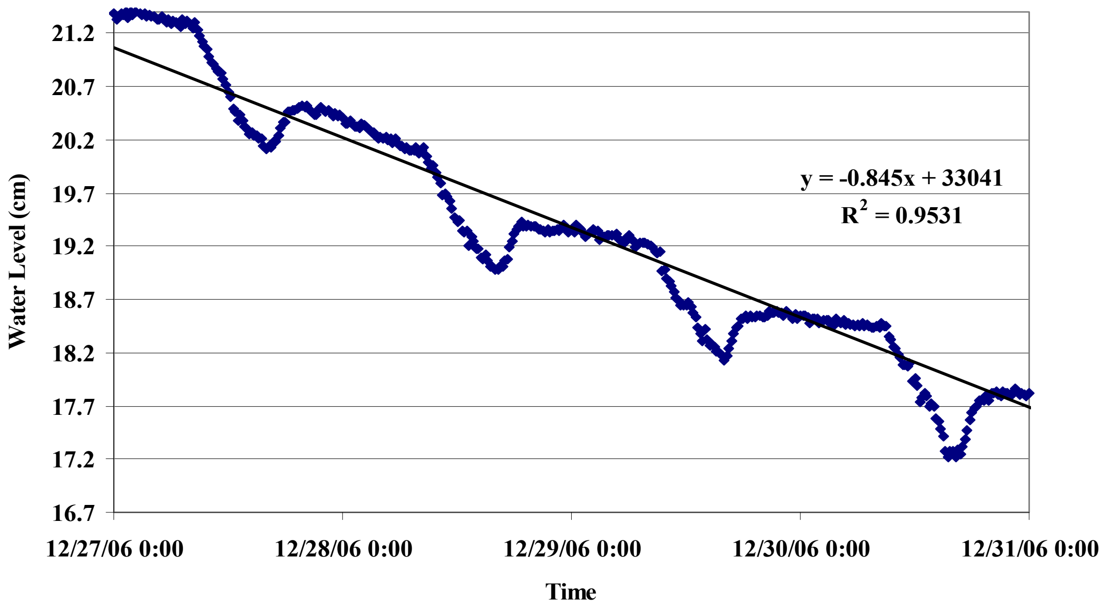


Figure 149. Surface Water Level Decline at Mosaic H1 SW-1.

Rates calculated with the White method and total daily decline calculated with the slope of linear regression lines were averaged to obtain monthly averages of the various daily rates. The rates are listed for Mosaic K5 SW-2 along with the number of days sampled in each month and average daily Penman-predicted ET in Table 71. The monthly averages did at times include months from 2007 and 2008. Calculated ET rates with the White method increased from winter to summer, as was to be expected, and were much higher than what was estimated with the Penman method. The infiltration rates were generally more negative in the winter compared to the summer when at times rates became positive, indicating groundwater outflow from the water feature. The average infiltration rate for all periods analyzed was -0.82 cm/day. Similar to the ET rates, the daily declines increased in the summer and were also typically higher than Penman-predicted ET.

Table 71. Calculated Rates, Daily Decline, and Penman ET (cm/day) at Mosaic K5.

Month	Sampled Days	ET	Infiltration	Daily Decline	Penman ET
January	10	1.42	-1.07	0.40	0.23
February	15	1.26	-0.91	0.29	0.29
March	12	1.51	-1.00	0.49	0.43
April	20	1.07	-0.56	0.60	0.52
May	23	1.56	-0.70	0.85	0.57
June	12	1.74	0.10	1.84	0.53
July	13	2.72	0.29	3.06	0.48
August	8	2.62	-0.75	1.84	0.50
September	9	1.62	-0.70	0.87	0.44
October	9	0.82	-0.68	0.22	0.36
November	20	1.05	-0.68	0.41	0.26
December	12	1.34	-0.98	0.26	0.21

Of particular interest is that the daily declines significantly increased from winter to summer and much more than the calculated ET rates increased. The calculated rates were generally over 1.0 cm/day, even in the winter months when typical ET rates were 0.2 to 0.4 cm/day (Table 71). Such differences cause suspicion of the rates calculated with the White method. While the most conservative estimate of nighttime infiltration rates was calculated by not including the immediate response after sundown, the large negative, or exfiltration, rates that still resulted may cause overestimation in the ET calculation. Since the exfiltration rate is applied during the day, the ET rate includes the observed daytime decline plus the estimated daytime groundwater inflow. For the method to be accurate, the assumption of groundwater flow rates in the day being equal to what is observed at night must hold true. The largest calculated rates of groundwater inflow were observed during the winter when the calculated ET rates were substantially above typical values. This said, caution should be used when evaluating the calculated rates. One explanation is that groundwater inflows are much higher during the nighttime after a gradient was established by loss of surface water via ET and that these inflow rates should not have been applied during the day. The declining exfiltration rates, at times to zero, that were observed at night give support to this possible explanation (Figures 138-141). Assuming that the groundwater inflows observed at night do not occur during the day and that the change in surface water levels during the daytime are solely from ET would result in more conservative ET estimates.

Considering the total daily decline as daily ET is the most conservative estimate of ET using surface water fluctuations when increases are observed at night. This assumes a transient groundwater flow where the gradient established during the day causes groundwater inflow at night, but the onset of ET causes a groundwater trough to establish around the water feature that would result in groundwater outflow during the day. If it is assumed that the inflow and outflow rates are mirrored, then net daily groundwater inflow is zero. Again, this is an offered explanation for the most conservative estimates, which are still higher than Penman-predicted ET but more reasonable values (Table 71). However, this estimate should not be used during periods when positive infiltration was calculated, such as in the case of the June and July calculations, when infiltration rates increased the daily decline above ET rates. The daily declines in the winter months are only slightly larger than the Penman-predicted estimates, while the spring and summer months, excluding June and July, have declines that average 1.8 times what the Penman method estimates.

The days sampled and the monthly averaged daily rates for PCS SA 10 and PCS SA 01 are shown in Tables 72 and 73. Average infiltration rates were always negative at both sites, with an average of -0.64 and -1.2 cm/day at SA 10 and SA 01, respectively. Similar to Mosaic K5, the calculated ET rates were near or above one cm/day even in the winter months. Also similar to Mosaic K5, the most conservative estimates of ET, daily declines, at PCS SA 10 were greater than the Penman-estimated rates (Tables 72 and 71). Daily declines at PCS SA 10 were on average 1.5 times greater than Penman ET. The daily declines at PCS SA 01, however, were typically less than Penman-predicted ET and on average 25% less. The similar calculated ET rates of the two PCS sites but the smaller daily decline at SA 01 can possibly be explained by the larger average exfiltration rates of SA 01. The diurnal signatures of PCS SA 01 had less transient groundwater inflows observed at night, which never approached zero as was often experienced at the other sites.

Table 72. Calculated Rates, Daily Decline, and Penman ET (cm/day) at PCS SA 10.

Month	Sampled Days	ET	Infiltration	Daily Decline	Penman ET
January	9	0.92	-0.68	0.32	0.16
February	4	1.24	-0.96	0.24	0.24
March	13	1.69	-1.28	0.40	0.34
April	17	1.44	-0.85	0.66	0.44
May	31	1.45	-0.63	0.91	0.54
June	13	1.44	-0.43	1.03	0.56
July	0	--	--	--	0.56
August	14	1.02	-0.11	0.89	0.53
September	21	0.92	-0.22	0.70	0.44
October	10	1.00	-0.48	0.54	0.30
November	21	0.99	-0.61	0.41	0.22
December	12	1.01	-0.78	0.25	0.16

Table 73. Calculated Rates, Daily Decline, and Penman ET (cm/day) at PCS SA 01.

Month	Sampled Days	ET	Infiltration	Daily Decline	Penman ET
January	10	1.10	-1.00	0.17	0.16
February	11	1.22	-0.96	0.27	0.24
March	17	1.77	-1.53	0.25	0.34
April	16	1.72	-1.50	0.31	0.44
May	16	1.70	-1.41	0.29	0.54
June	30	1.65	-1.22	0.46	0.56
July	13	1.43	-0.78	0.69	0.56
August	12	1.23	-0.88	0.35	0.53
September	18	1.11	-0.78	0.35	0.44
October	10	1.13	-0.89	0.27	0.30
November	21	1.00	-0.91	0.14	0.22
December	10	1.30	-1.23	0.06	0.16

The days sampled and the monthly averaged daily rates for Mosaic H1 SW-3 and SW-1 are shown in Tables 74 and 75. As with the other sites, the average daily infiltration calculated for SW-3 were negative, with two exceptions, suggesting exfiltration (Table 74). The calculated exfiltration rates at Mosaic H1 SW-3, however, were less than those calculated for the other sites, causing the daily decline values to be near the calculated ET rates (Table 74). The daily declines at SW-3 were on average 1.5 times greater than the Penman-estimated rates. While only a few periods at Mosaic H1 SW-1 were appropriate for analysis, the average infiltration rates were always positive, with a maximum of 0.77 cm/day groundwater outflow from the water feature. These results reveal differences in the hydrologic regime of separate water features within one CSA.

Table 74. Calculated Rates, Daily Decline, and Penman ET (cm/day) at Mosaic H1 Surface Water-3.

Month	Sampled Days	ET	Infiltration	Daily Decline	Penman ET
January	10	0.49	-0.07	0.39	0.23
February	13	0.48	-0.15	0.32	0.29
March	12	0.76	-0.32	0.46	0.43
April	12	0.50	-0.23	0.35	0.52
May	20	1.05	-0.20	0.85	0.57
June	25	1.06	-0.06	1.04	0.53
July	15	1.01	-0.05	1.03	0.48
August	10	0.84	0.01	0.87	0.50
September	12	0.68	0.05	0.74	0.44
October	9	0.60	-0.18	0.45	0.36
November	22	0.60	-0.15	0.46	0.26
December	12	0.61	-0.22	0.40	0.21

Table 75. Calculated Rates, Daily Decline, and Penman ET (cm/day) at Mosaic H1 Surface Water-1.

Month	Sampled Days	ET	Infiltration	Daily Decline	Penman ET
January	17	0.54	0.19	0.73	0.23
February	11	0.71	0.23	0.96	0.29
March	--	--	--	--	0.43
April	6	1.08	0.77	1.84	0.52
May	--	--	--	--	0.57
June	--	--	--	--	0.53
July	--	--	--	--	0.48
August	--	--	--	--	0.50
September	--	--	--	--	0.44
October	--	--	--	--	0.36
November	--	--	--	--	0.26
December	4	0.39	0.45	0.85	0.21

As previously mentioned, no sufficient inundation occurred at CFI SP-1 and Williams Co. since the time of installation of the new transducers. Without assuming a specific yield, calculation of ET and infiltration rates with below-ground water fluctuations is not possible and any such calculations would be strongly affected by the assumed specific yield. A more qualitative analysis was performed, however, to simply identify if exfiltration or infiltration was occurring at these sites. Water table fluctuations for Williams Co. are shown in Figure 150, where the decline at night, and thus infiltration, is apparent. Of the 19 periods analyzed for this site, 11 had a signature similar to the one shown in Figure 150, suggesting that infiltration occurred at this site frequently. Furthermore, though any calculated rates without an accurate specific yield would be erroneous, a relative comparison of the calculated infiltration rate to the calculated ET rate gives an idea of the magnitude of infiltration. The rates for 145 days were calculated with a specific yield of one and on average the infiltration was 25.2% of calculated ET. While in previous discussions, a positive infiltration was defined as loss of surface water to the local groundwater, infiltration here refers to loss of local groundwater to adjacent regions, vertically and/or laterally. The direction of this groundwater movement suggests the potential direction for surface water loss at Williams Co. There were limited periods appropriate for analysis of CFI SP-1 as water levels were often below the well depth at CFI SP-1 since the installation of the newer transducers. Five periods totaling 28 days were evaluated for CFI SP-1 and negative infiltration was observed during all periods, with signatures similar to those of the PCS SA 10 and Mosaic H1 SW-3 (Figures 146 and 148). It should be noted, however, that the fact that no infiltration was observed when water levels were below ground does not fully imply that infiltration would not have occurred with inundated conditions.

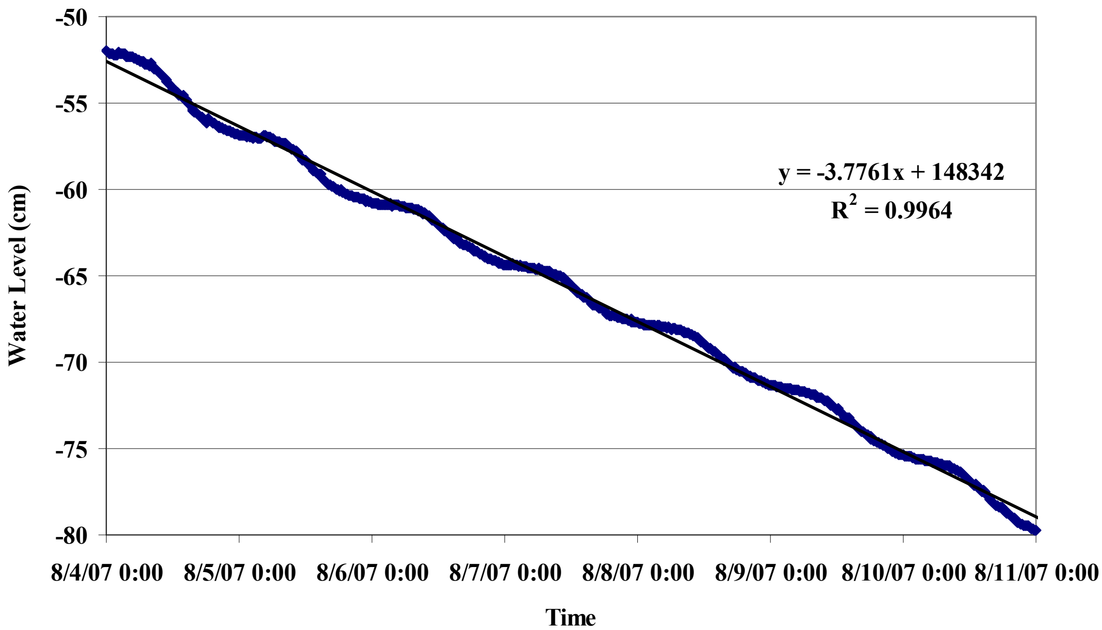


Figure 150. Water Table Decline at Williams Co.

Both the drawdown analysis with the less accurate transducers and the diurnal analysis with the more accurate transducers revealed differences among sites that were similar to the differences observed in the water balance analysis. The higher residuals at Williams Co. and Mosaic H1 may be explained by possible infiltration, while most sites' residuals may be more due to underestimation of ET, as no evidence of infiltration was found for other sites. Furthermore, the negative residual calculated for PCS SA 01 could have resulted from constant groundwater inflow, which was suggested by this analysis, since the daily declines were actually less than Penman-predicted ET.

Temporal Hydrologic Modeling

Temporal models were developed for seven instrumented surface water features using the simple linear runoff models, rainfall data, and ET estimated with the Penman method along with actual surface water data for calibration. Only positive surface water data were used to create the models and, therefore, modeling was not performed for the surface water feature at Tenoroc-4. Furthermore, Mosaic HP-10 was excluded from modeling as a result of the limited knowledge of the surface water outflow regime. The linear models with the y-axis intercept not set to zero were employed in these models due to their success in predicting event responses. Since these runoff models require only rainfall data, upland-to-wetland ratios and runoff coefficients were not included in the temporal models. While the diurnal curve analysis suggested some sites may have experienced groundwater inflow, zero groundwater flows were initially applied to the models. Only in the situation where a model underestimated stage, were groundwater inflows (exfiltration) included. In the cases where the initial models overestimated stage, the use of both increased ET rates using seasonal coefficients and infiltration rates were explored.

The model results for PCS SA 10 with no inclusion of an infiltration loss and for time periods after surface outflow had ceased are shown in Figure 151. The responses following rain events were similar between the actual and predicted surface water levels, with the exception of a large rain event in March 2008. The losses, however, were underestimated by the model and stage was overestimated. An infiltration term equal to the daily residual value, 1.16 mm/day, calculated in the water balance analysis, was applied (Figure 152). Though the final predicted stage matched the actual, declines during the summer months are underestimated while winter declines are overestimated, suggesting seasonal variability in the residuals possibly from underestimation of ET. Penman-estimated ET rates were multiplied by seasonal coefficients which were manipulated to maximize the fit of the model. Excluding infiltration and multiplying the estimated ET rates by 1.5 during the growing season and by 1.1 in the non-growing season resulted in the best model (Figure 153). The model resulted in a non-linear correlation coefficient equal to 0.87, which increased to 0.94 when excluding the 10 cm rain event in March 2008 that was underestimated by the model. A non-growing season coefficient near 1.0 suggests that the residual value calculated in the water balances is due to underestimation of ET, especially in the growing season, and that infiltration may be negligible at this site. A coefficient of 1.5 is fairly high to use with the Penman model, but is in the range of some results from other studies of wetland ET (Drexler and others 2004).

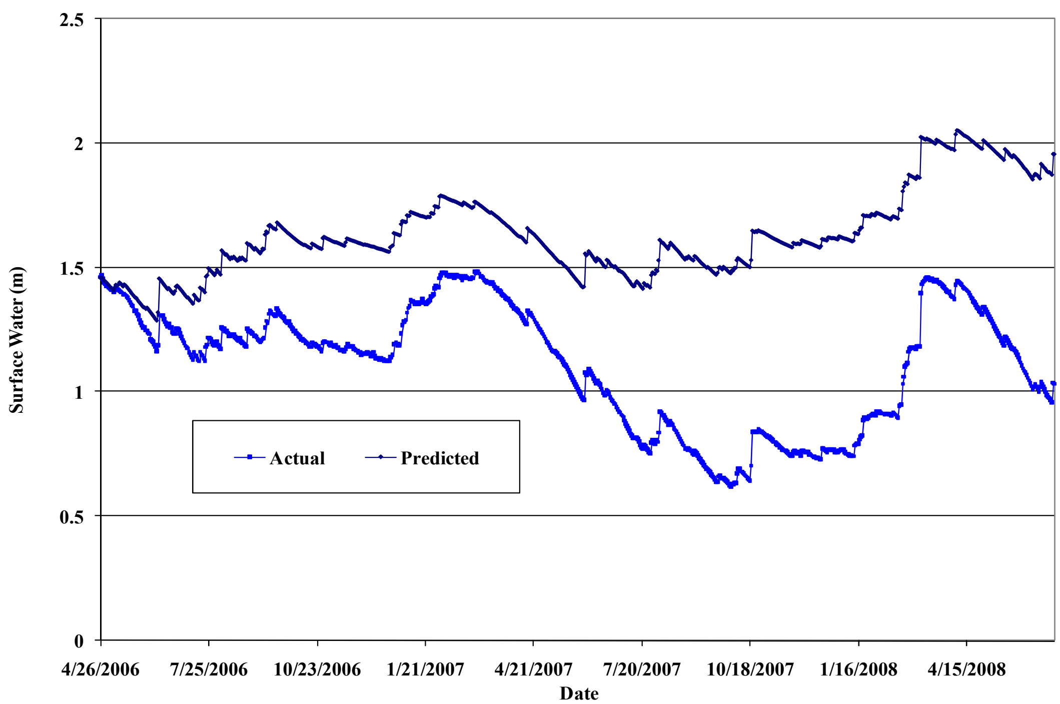


Figure 151. Temporal Model Results with No Infiltration Term for PCS SA 10.

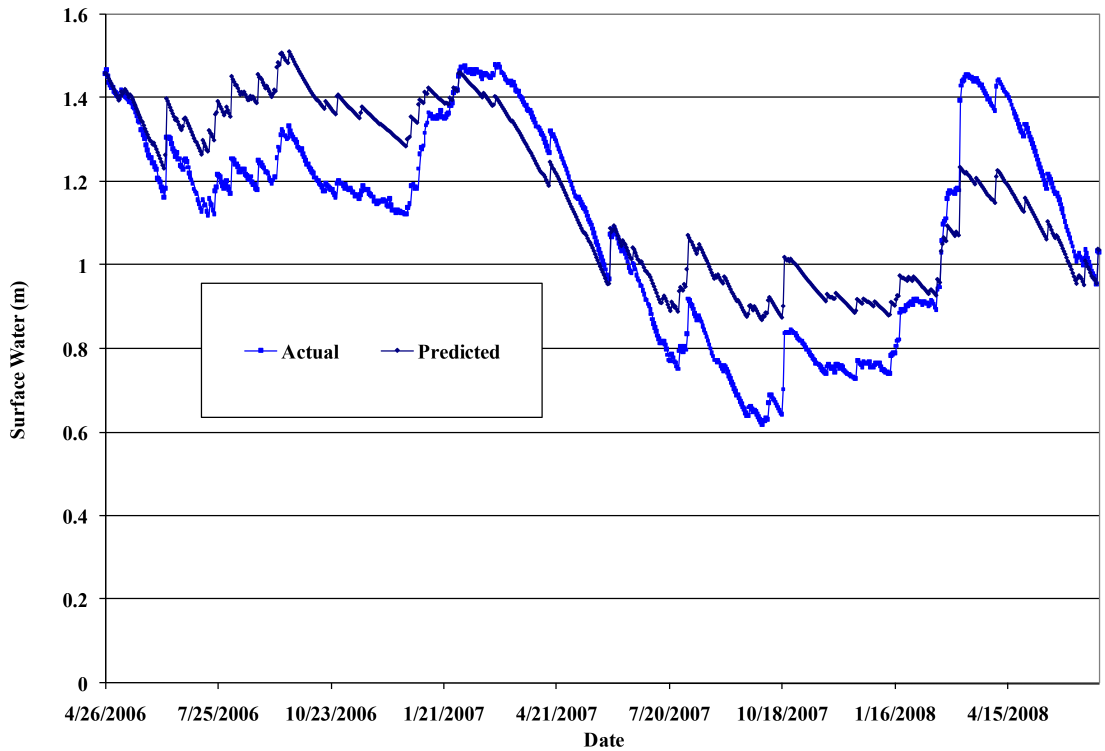


Figure 152. Temporal Model with I = 1.16 mm/day for PCS SA 10.

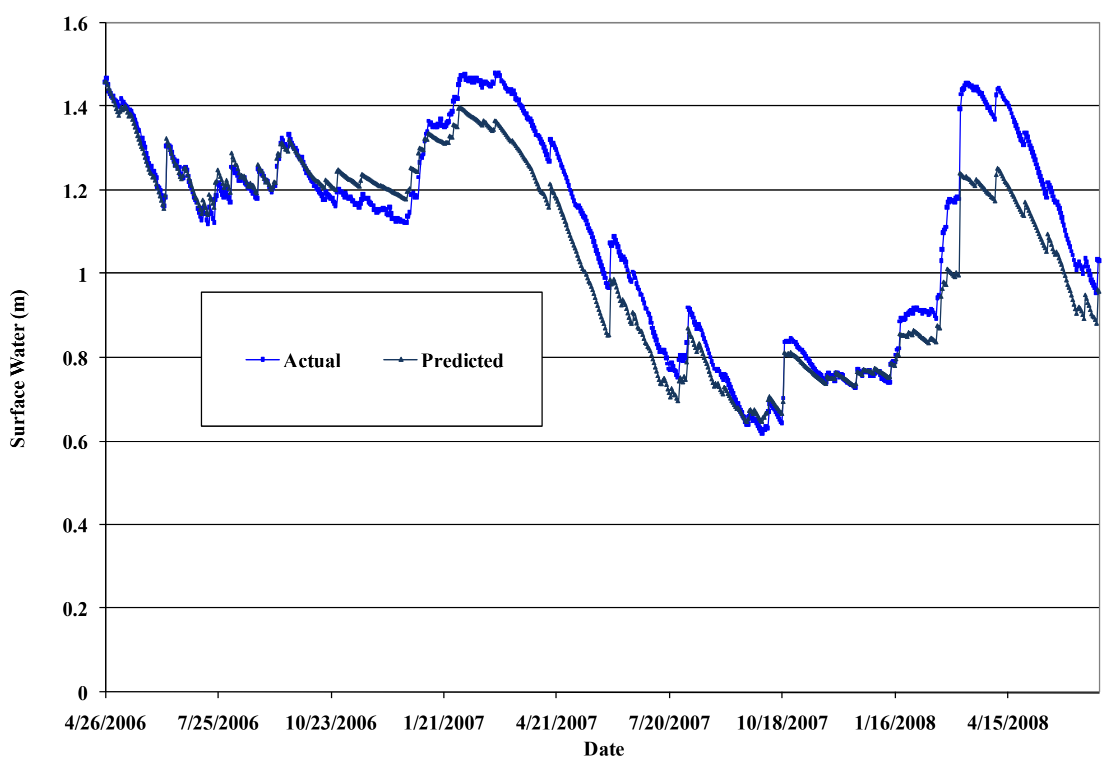


Figure 153. Temporal Model for PCS SA 10 with Manipulated ET and Zero Infiltration.

The model results with no inclusion of an infiltration loss are shown for Williams Co. in Figure 154. The responses following rain events were similar between the actual and predicted surface water levels, but the losses were substantially underestimated by the model. An infiltration term equal to the daily residual value, 4.87 mm/day, calculated in the water balance analysis, was included (Figure 155). The ending predicted and actual stages match, but as a result of underestimating summer decline and overestimating winter decline. The seasonal variation in declines for this site was greater than it was for PCS SA 10 (Tables 69 and 63). A seasonal coefficient for the non-growing season of 1.0 and an infiltration term was calculated to allow the predicted non-growing season decline to match the actual decline. An infiltration of 1.5 mm/day resulted, which was used to calculate a seasonal coefficient for the growing season to match predicted growing season decline with actual. A growing season coefficient of 2.5 resulted, which is well out of the range for reported values of crop coefficients. The non-growing season coefficient was decreased to 0.6 to again solve for infiltration and a non-growing season coefficient. A non-growing season coefficient of 0.6 was used, which is within the range of a *Typha spp.* dominated marsh during the winter (Drexler and others 2004). An infiltration of 2.54 mm/day and a growing season coefficient of 2.1 were found to produce the best-fitting model (Figure 156). The model fit the data well, with a correlation coefficient of 0.95. These results suggest that infiltration is significant at this site and that ET had a more pronounced seasonal variation than what was predicted or experienced at PCS SA 10.

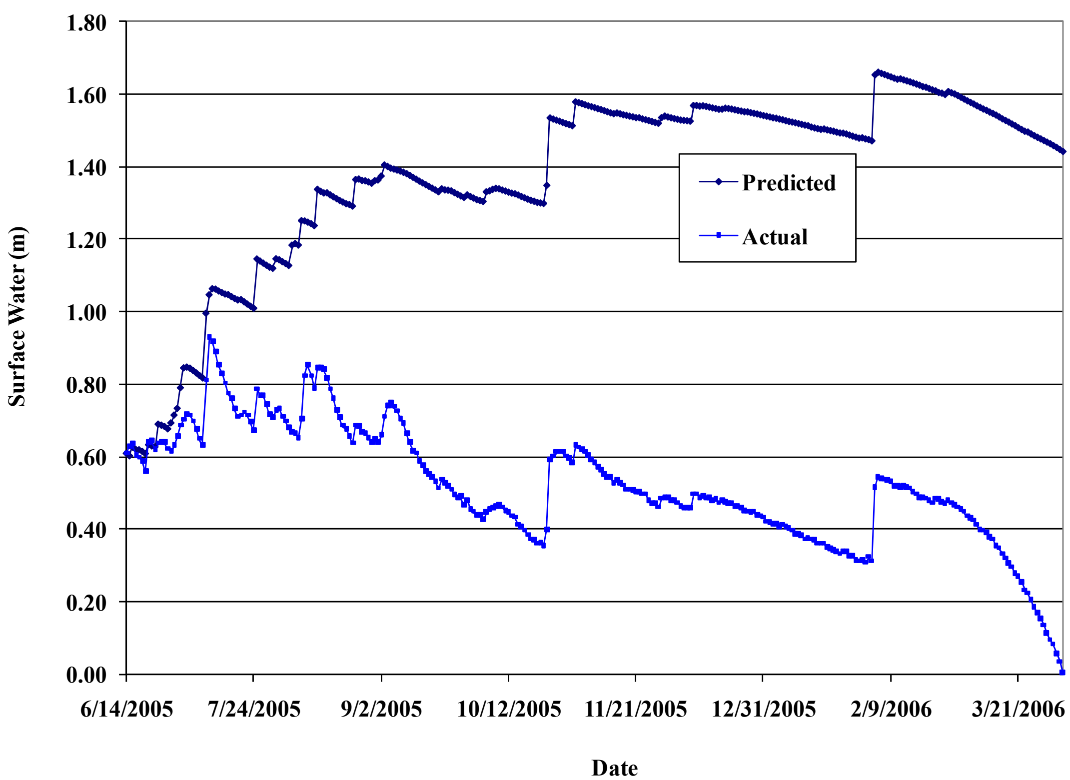


Figure 154. Temporal Model Results with Zero Infiltration for Williams Co.

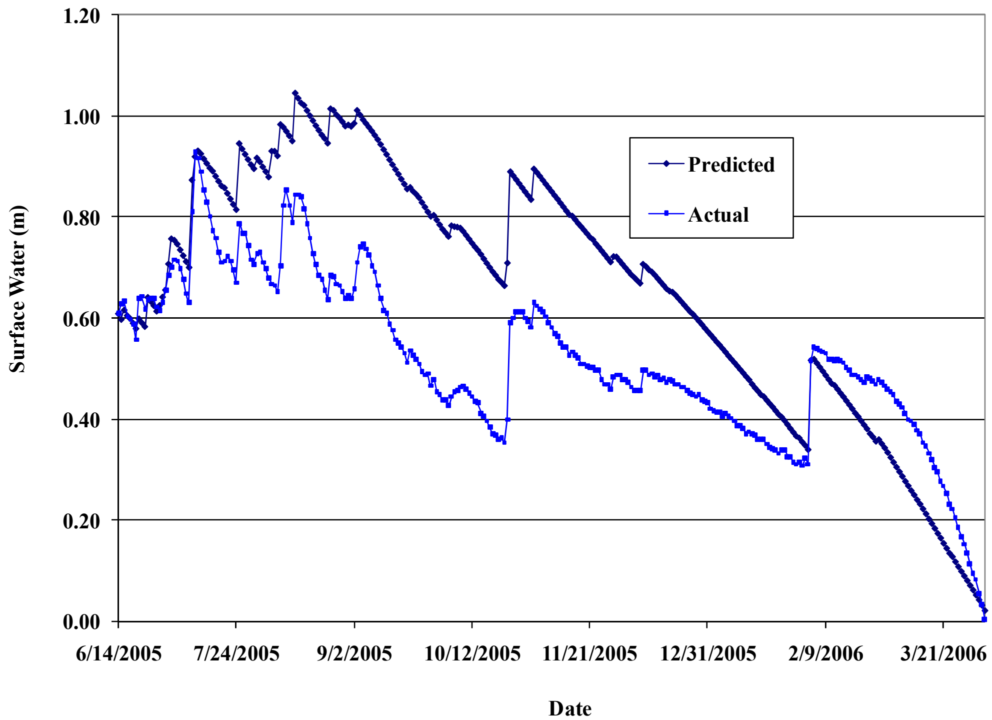


Figure 155. Temporal Model with I = 4.87 mm/day for Williams Co.

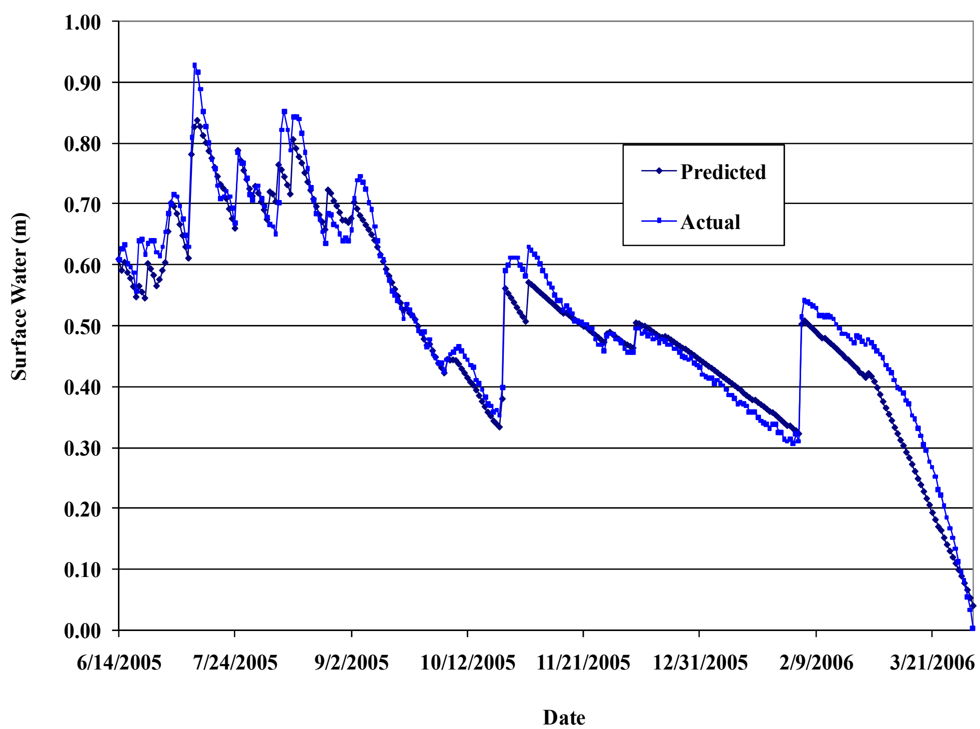


Figure 156. Temporal Model for Williams Co. with Manipulated ET and I = 2.54 mm/day.

Similar to the modeling results of PCS SA 10 and Williams Co., solely applying Penman-predicted ET and the linear runoff models while excluding infiltration overestimated stage at Mosaic H1 Surface Water-3 and CFI SP-1. Including the water balance residual resulted in end-stage matching but by underestimating growing season and overestimating non-growing season declines. The best models for both features were developed by excluding infiltration and applying seasonal coefficients. A non-growing coefficient equal to 1.0 and a growing season equal to 1.4 were applied to the Mosaic H1 Surface Water-3 model, which resulted in a correlation coefficient of 0.93 (Figure 157). Growing and non-growing season coefficients were 1.3 and 1.1, respectively, for the CFI SP-1 model, which had a correlation coefficient equal to 0.94 (Figure 158).

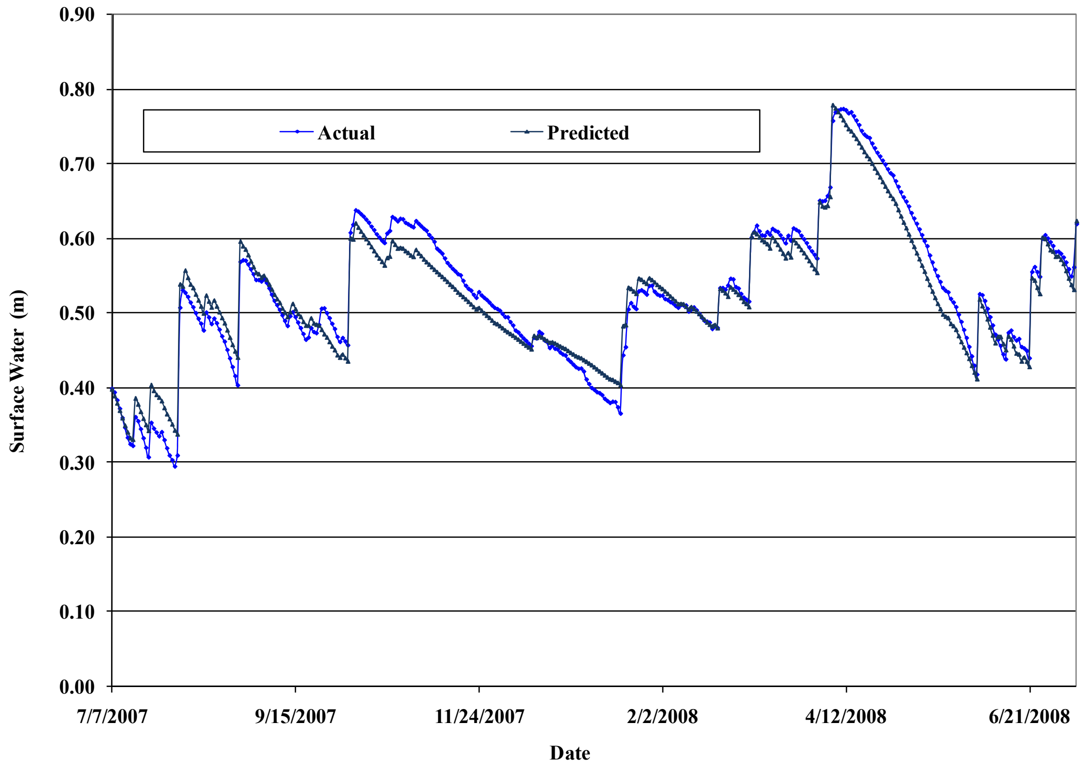


Figure 157. Temporal Model for Mosaic H1 Surface Water-3.

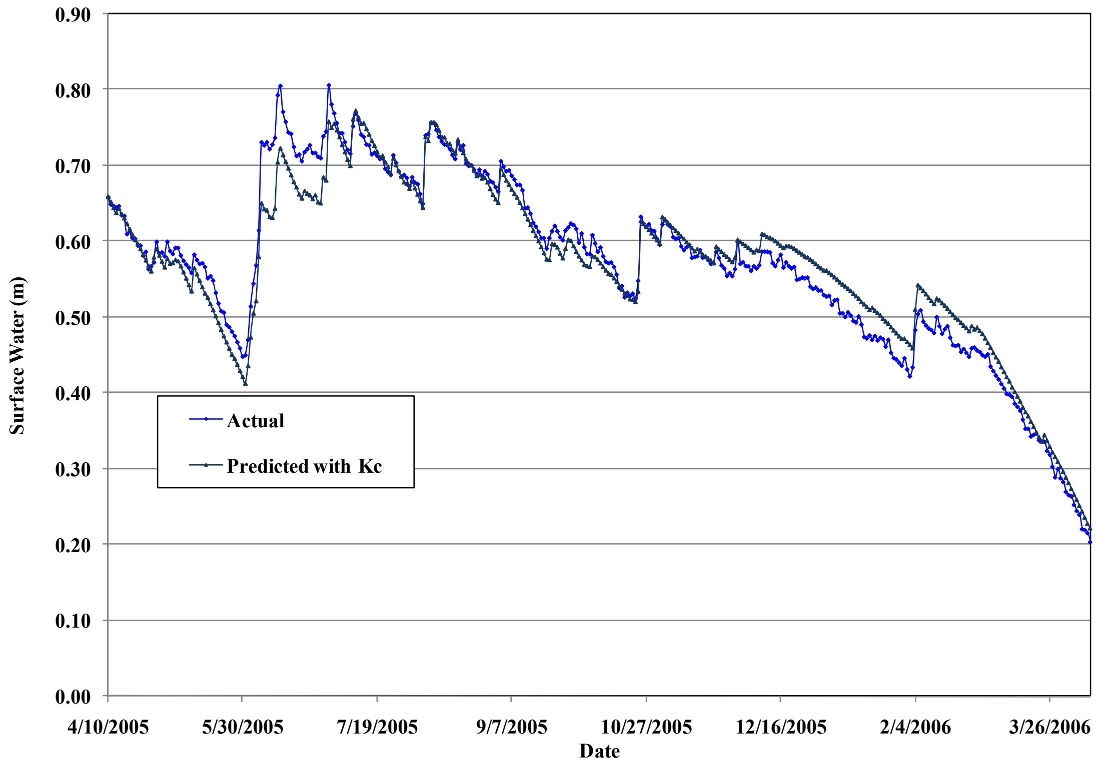


Figure 158. Temporal Model for CFI SP-1.

Three different time periods with no channel flow were used in model development of Mosaic K5 SW-1. Again, similar to the other modeling results, models initially overestimated stage. One non-growing season was modeled that required no seasonal coefficient or infiltration term to accurately predict stage with a correlation coefficient of 0.99 (Figure 159). Two different growing season time periods were modeled that had residual values slightly over 2 mm/day. Since the non-growing season model required no infiltration, only seasonal coefficients were included in the growing season models. A seasonal coefficient of 1.4 resulted in the best fits for both periods, with correlation coefficients of 0.93 and 0.99 (Figures 160 and 161, respectively). Both growing season models underestimated declines and responses to rain events at stages below 10 cm, likely due to the specific yield of surrounding soils.

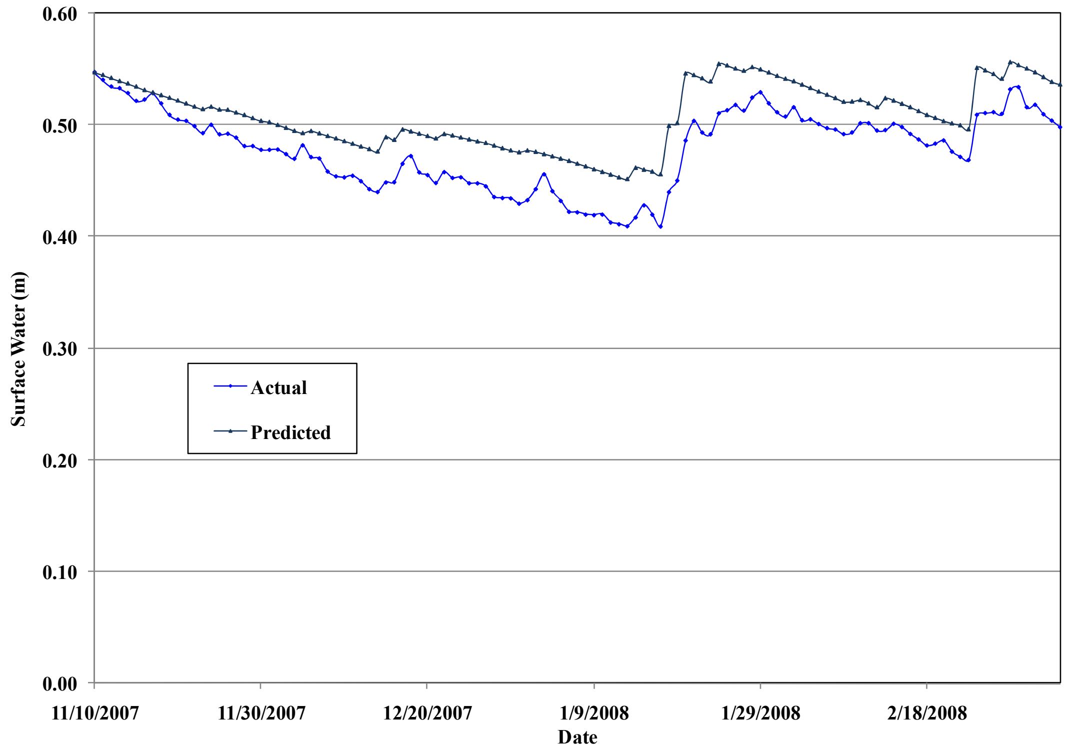


Figure 159. Temporal Model for Mosaic K5 11/10/07 to 3/5/08.

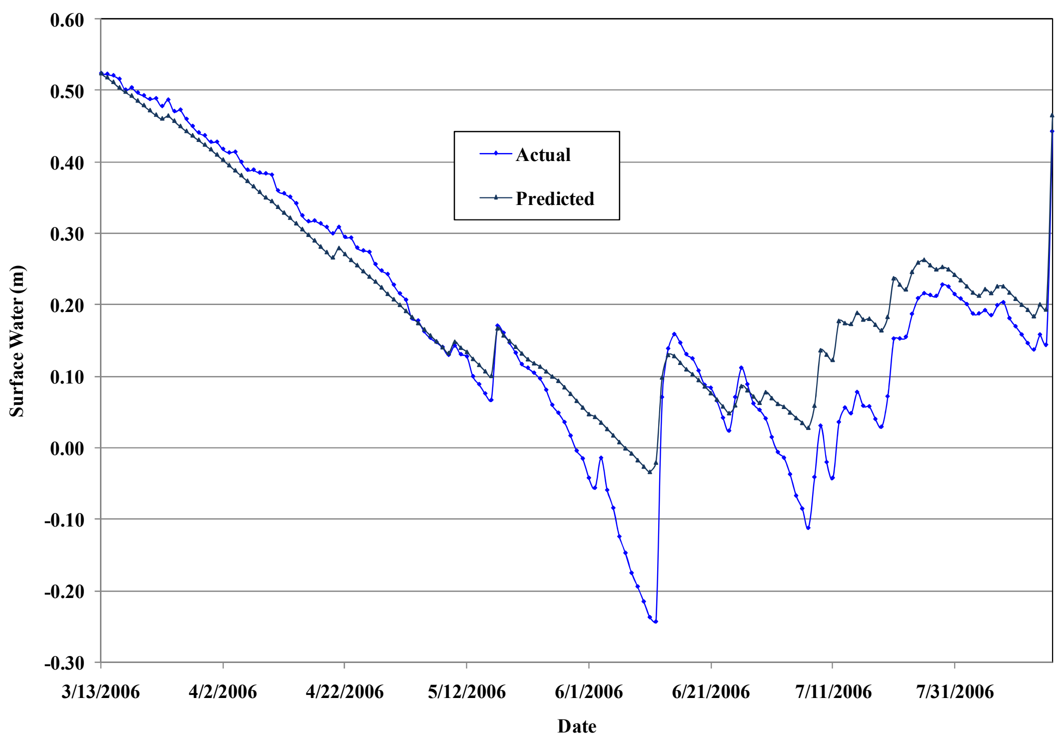


Figure 160. Temporal Model for Mosaic K5 3/13/06 to 8/15/06.

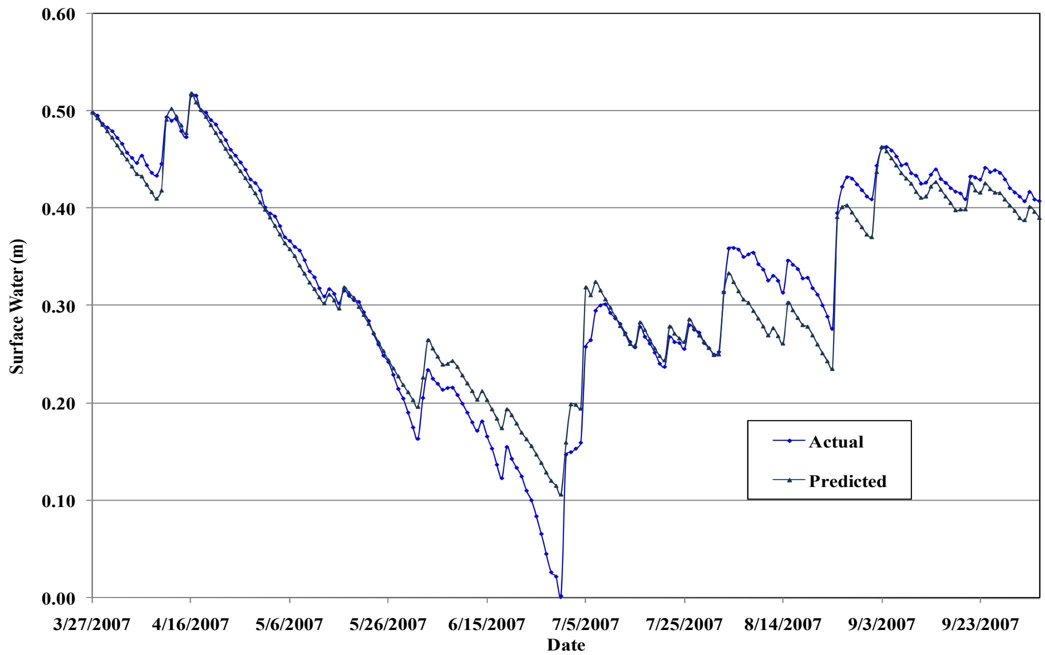


Figure 161. Temporal Model for Mosaic K5 3/27/07 to 10/4/07.

A model for Mosaic K5 to include channel flow was developed by applying the seasonal coefficients that were used in the three models without channel flow and including daily outflow (m/day) that was calculated in the surface water outflow analysis. The time period with the channel well data available, May 2006 to January 2008, was used. Predicted and actual stages are shown in Figure 162, which resulted in a correlation coefficient of 0.91. Again, declines and responses to rain events were underestimated when stage was near zero. While the predicted response to the large rain event late in August 2006 is agreement with the actual response, the subsequent decline is overestimated, causing the actual stage to be above the predicted for the most of the remaining period. A possible explanation to this underestimation is similar to the one given in the water balance analysis. At such high stage, the channel flow may not solely originate from the modeled feature but could include flow from other isolated features, as well as from direct rainfall from such a large event. Other declines are better predicted, however, including during times of channel flow, demonstrating utility of the seasonal coefficients and the channel flow estimations.

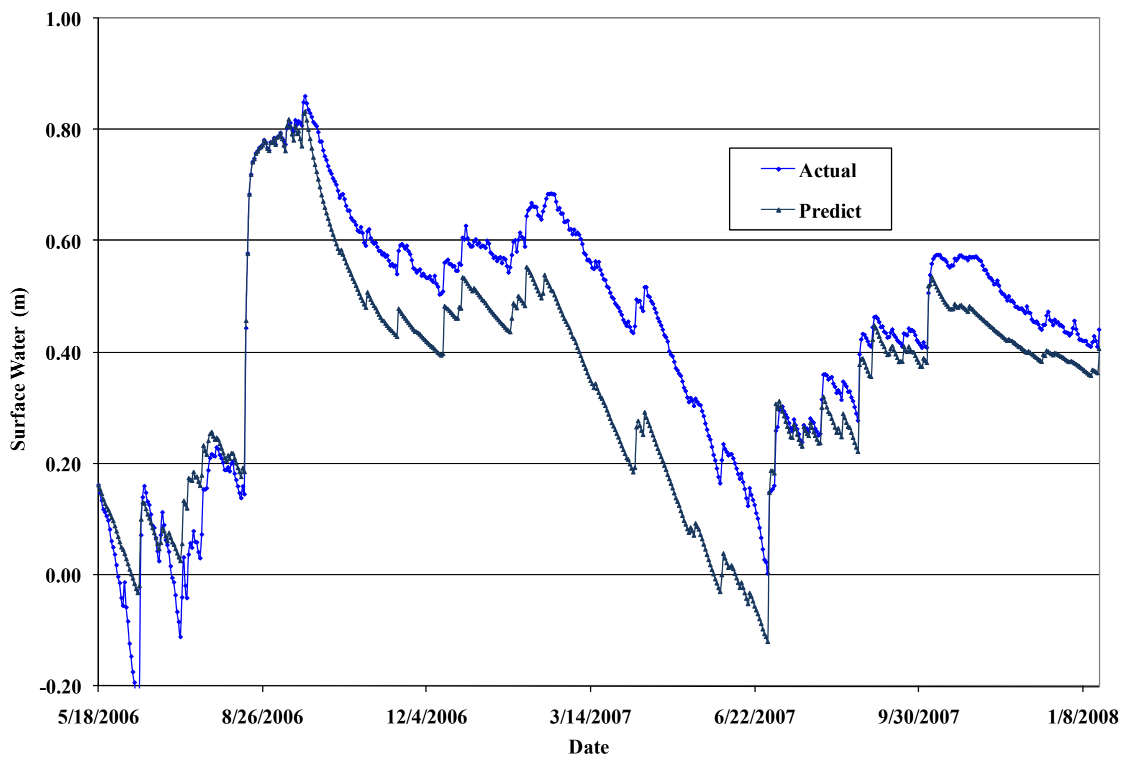


Figure 162. Temporal Model for Mosaic K5 5/18/06 to 1/18/08.

The model results with no inclusion of an infiltration loss are shown for Mosaic H1 Surface Water-1 in Figure 163. The model was applied to two separate time periods and not during times where surface water levels were below zero. As with the other sites, the model substantially underestimates losses and overpredicts stage. An infiltration term equal to the residual value of 4.30 mm/day was included in the model, which resulted in marked improvement (Figure 164). Some declines, however, were still underestimated, especially during the period after the feature became dry. Seasonal coefficients were manipulated with infiltration equal to 3.00 mm/day, to best fit the times before the dry period (Figure 165). The lower infiltration rate applied required a coefficient of 1.7 for the growing season and 1.2 for the non-growing season to produce the best fit (Figure 165). While the adjusted model sufficiently predicted the declines that were experienced prior to the dry period, it still underestimated the declines after the dry period. Increasing the infiltration rate to 4.00 mm/day and applying the same seasonal coefficients caused somewhat better prediction of decline during the second period but resulted in the first period's decline being overestimated (Figure 166). While the predicted stage better matched the end stage of the second period, it still underestimated decline and the end matching was a result of the model underestimating response to rainfall in February 2007.

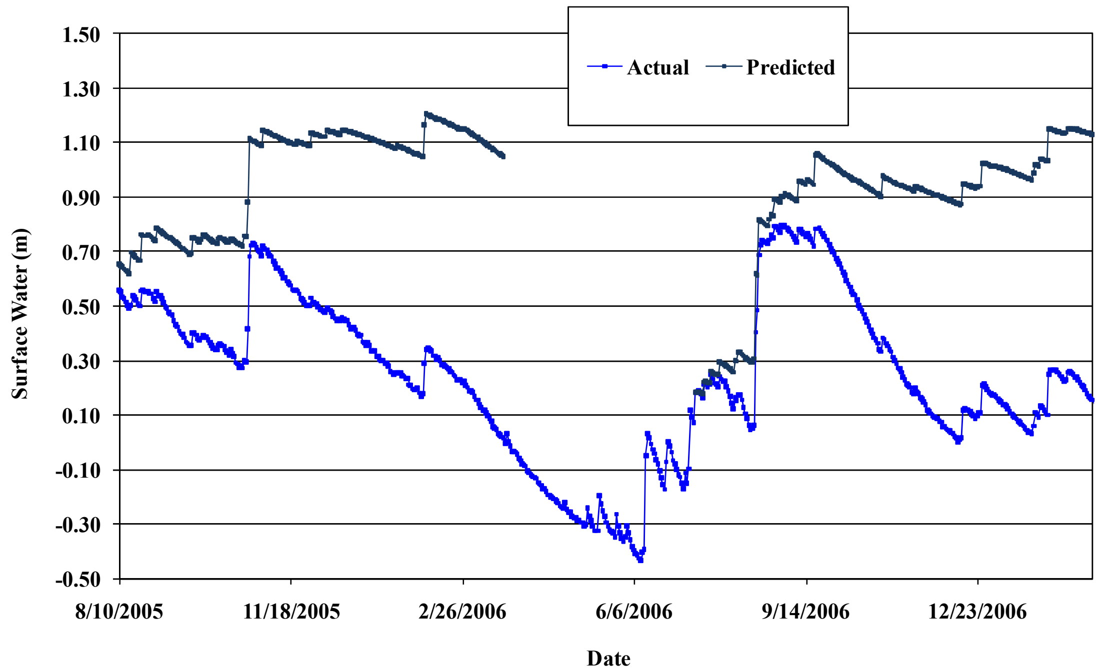


Figure 163. Temporal Model Results with Zero Infiltration for Mosaic H1.

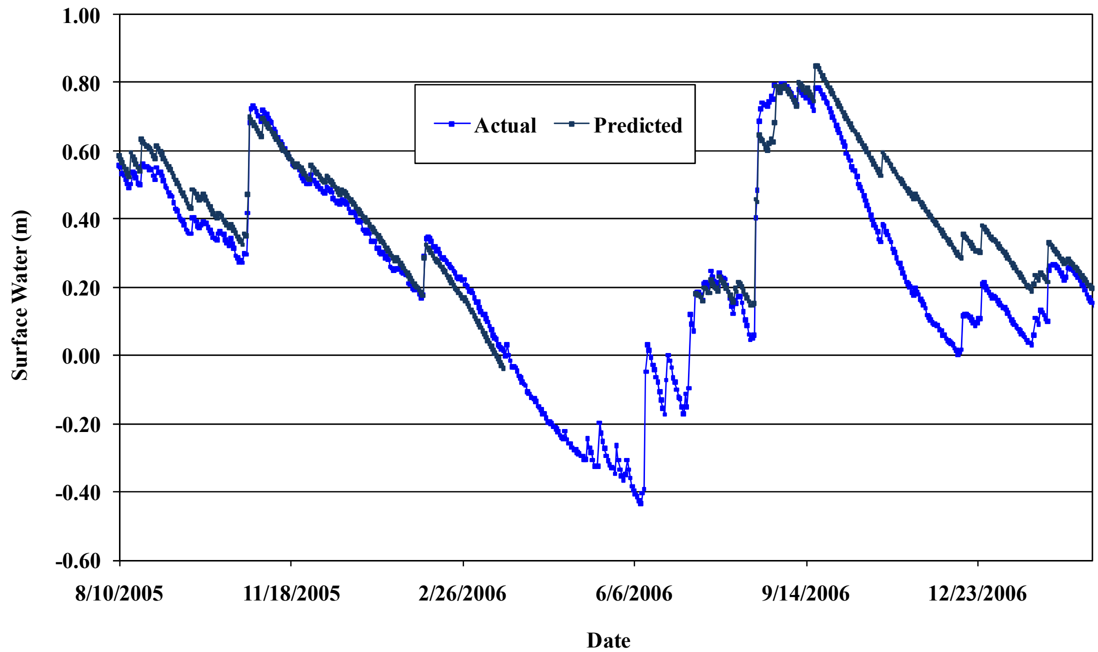


Figure 164. Temporal Model with I = 4.30 mm/day for Mosaic H1.

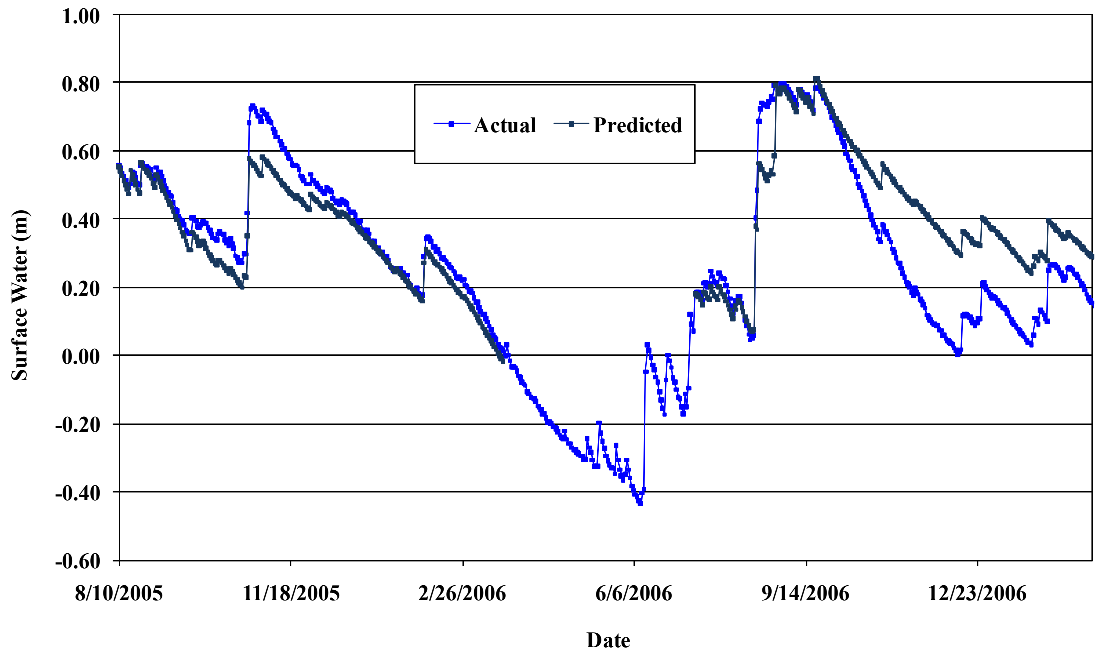


Figure 165. Temporal Model for Mosaic H1 with Manipulated ET and I = 3.00 mm/day.

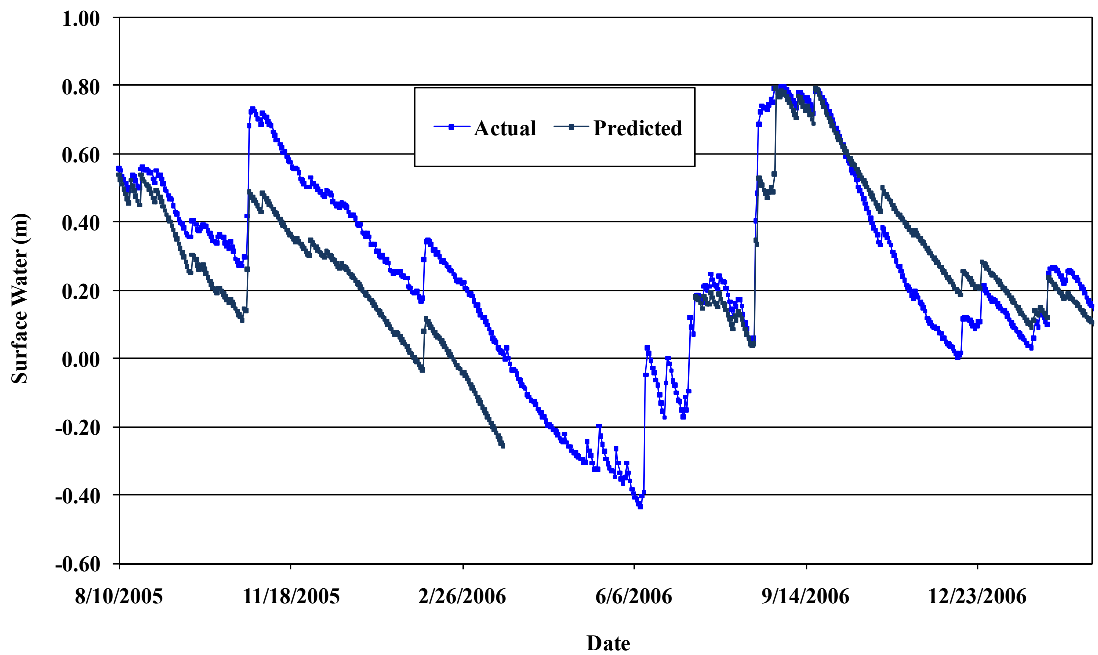


Figure 166. Temporal Model for Mosaic H1 with Manipulated ET and I = 4.00 mm/day.

An analysis of the surface water data collected at Mosaic H1 SW-1 suggested that the rate of losses was different during the two periods, which occurred at similar times in the year and at similar water depths. Different infiltration terms for the two periods were manipulated but with the same seasonal coefficients for ET to produce the best overall fit. Infiltration was adjusted to 2.79 mm/day for the period prior to the dry times and to 5.08 mm/day for the period after inundation occurred again (Figure 167). Though the predicted stage was less than actual stage at the end of the period, the slopes of the declines are better predicted, with an overall correlation coefficient of 0.83. Again, underestimating stage responses was responsible for the lower predicted stages rather than the rates of loss. These results suggest that not only is ET higher than what the Penman method predicted, but that also significant infiltration occurred that seemed to increase after a dry period. The latter observation could have possibly been a result of cracks developing in the feature with drying of the clay soils.

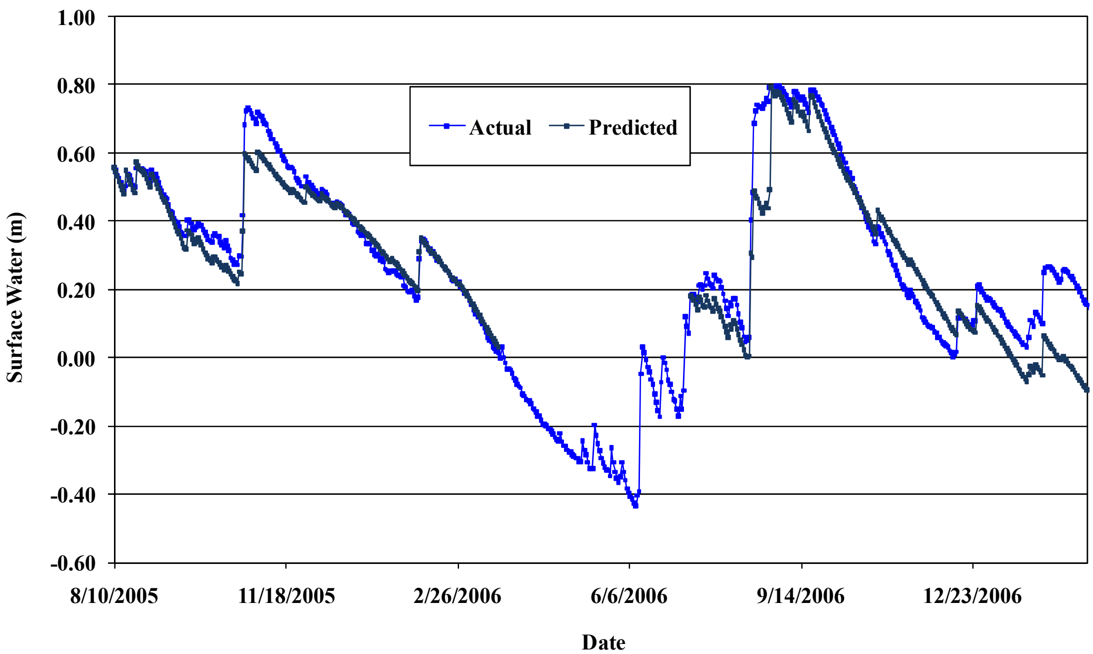


Figure 167. Temporal Model for Mosaic H1 with Manipulated ET, I_1 = 2.79 mm/day, and I_2 = 5.08 mm/day.

A negative residual resulted from the water balance analysis of PCS SA 01, suggesting groundwater inflow may have occurred. A model was developed using the linear models for runoff and Penman-predicted ET while excluding exfiltration (Figure 168). Unlike the initial models for the other sites, the model overestimated declines during the entire period and underpredicted stage. An exfiltration term of 1.2 mm/day, equal to the negative residual from the water balance, was included. This caused considerable improvement, giving a correlation coefficient of 0.92 (Figure 168). If it is assumed that this site could have also experienced elevated ET rates, then the exfiltration term would increase since seasonal coefficients were not applied to the Penman estimates in this model.

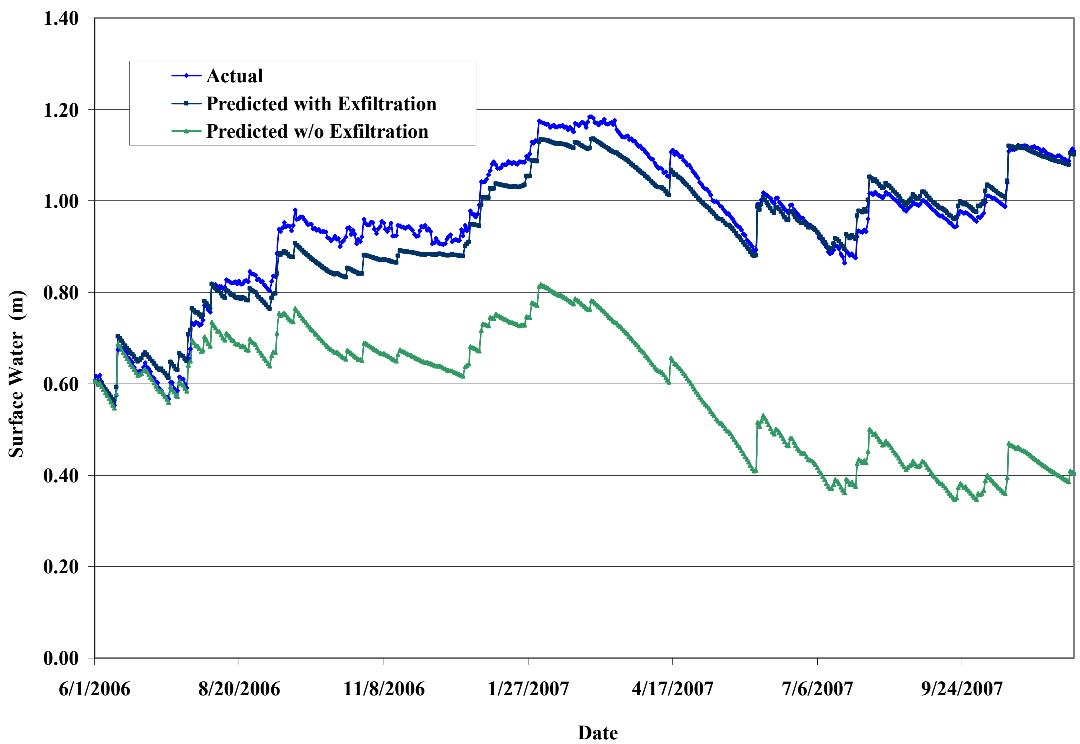


Figure 168. Temporal Models for PCS SA 01.

The seasonal coefficients and infiltration rates included in the models of the seven surface water features are listed in Table 76 along with the non-linear correlation coefficients. The infiltration rates of Williams Co. and Mosaic H1, the negative infiltration rate of PCS SA 01, and the need for seasonal coefficients support the results from the water balance and drawdown analyses.

Table 76. Modeling Results Summary.

Site	Seasonal ET Coefficients		Infiltration (mm/day)	R^2
	Growing	Non-Growing		
PCS SA 10	1.5	1.0	0	0.87
Williams Co.	2.1	0.6	2.54	0.95
Mosaic H1 SW-3	1.4	1.0	0	0.93
CFI SP-1	1.3	1.1	0	0.94
Mosaic K5	1.4	1.0	0	0.91
Mosaic H1 SW-1	1.7	1.2	2.79; 5.08	0.83
PCS SA 01	1.0	1.0	-1.20	0.92

Ecohydrology

Moisture Release Curves

Moisture release curves were developed in the laboratory using soils sampled at Mosaic K5, CFI SP-1, and Williams Co. to relate volumetric water content (VWC) to water potential in terms of negative cm of water pressure. Soil was sampled at the three different ecohydrology stations (upland, transitional, and saturated) of each site to construct a curve per station. The resulting moisture release curves for the three different stations at Mosaic K5 had good agreement and allowed a composite curve using all data to be evaluated (Figure 169). The curve demonstrated that saturation for soils at Mosaic K5 occur at a VWC near 0.60 m^3/m^3. The inflection point of the curve with decreasing VWC was near -200 cm H_2O, which demonstrates an approximate capillary fringe of 2 m. Assuming a permanent wilting point (PWP) of -1.5E^4 cm H_2O (Dingman 2002), soils at Mosaic K5 may experience PWP pressures at a VWC near 0.42 m^3/m^3.

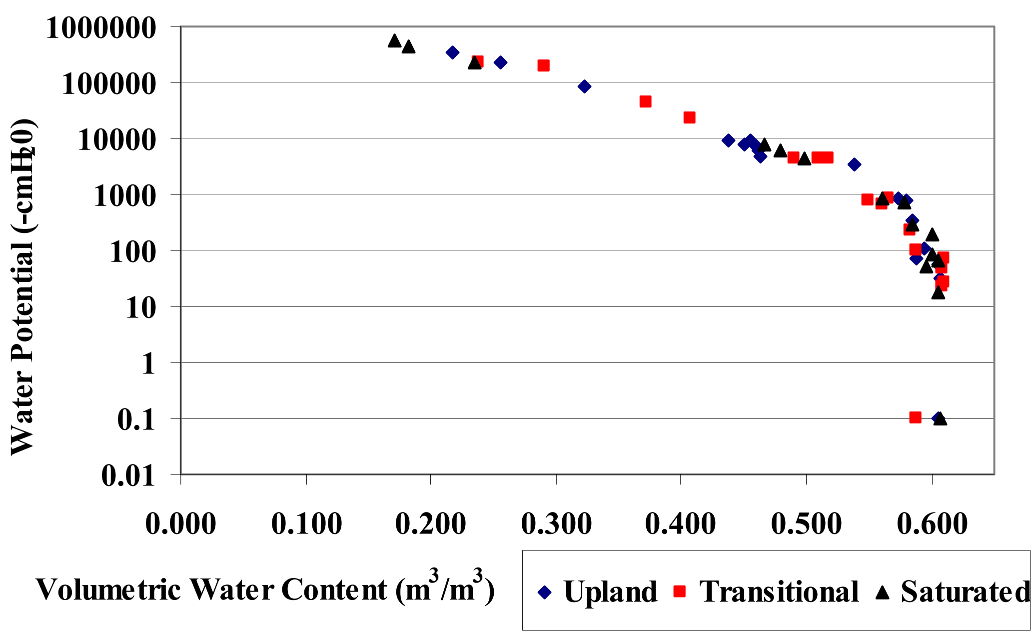


Figure 169. Moisture Release Curves for Mosaic K5.

Similar to the curves developed for Mosaic K5, the three curves for Williams Co. were in good agreement and allowed a composite curve to analyzed (Figure 170). The curve illustrates that saturation occurs when VWC is approximately 0.60 m^3/m^3. The inflection point occurred near -100 cm H_2O, corresponding to a capillary fringe approximately 1 m for the soils at Williams Co. PWP may be induced at VWCs near 0.42 m^3/m^3.

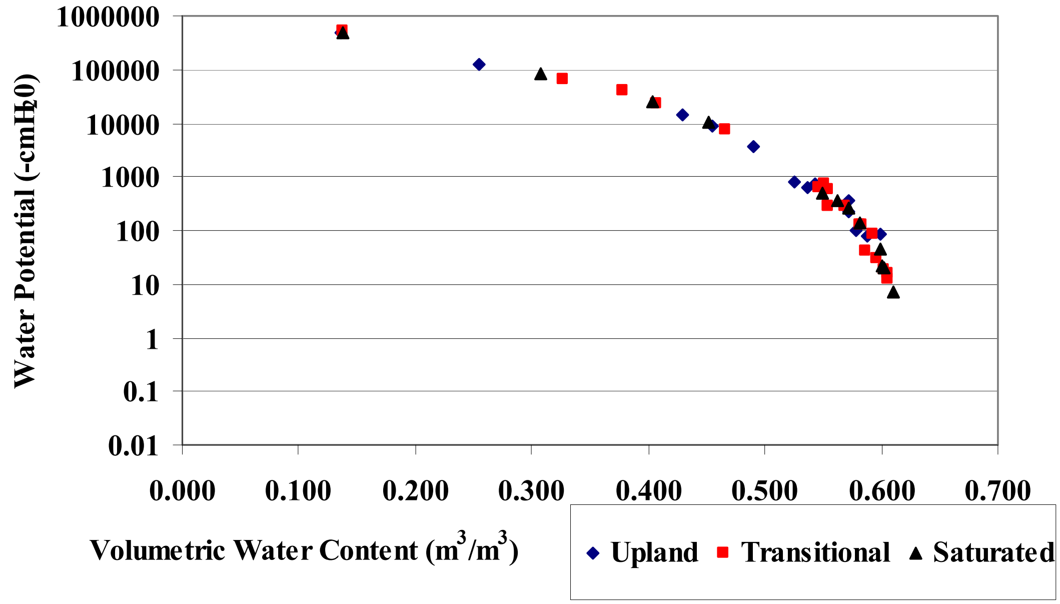


Figure 170. Moisture Release Curves for Williams Co.

The moisture release curves for CFI SP-1 were much different comparing the transitional and saturated zones to the upland zones. The upland soils were dominated by sand with very little clay as a result of this site receiving sand-clay mix, causing spatial soil heterogeneity. The curve for the transitional zone was developed using soil sampled 40 cm below the ground surface, which was more dominated by clay. The top 30 cm in this zone was similar to the sandy soil sampled in the upland. The entire soil profile at the saturated zone was similar to the sampled soil from the transitional zone. The curves developed for the transitional and saturated zones were comparable, allowing a composite curve to be evaluated (Figure 171). Soils at the saturated zone and 30 cm below ground at the transitional zone experience saturation at VWCs approximately 0.57 m^3/m^3. The curves suggest a capillary fringe of these soils near 1 m and that PWP may be induced at VWCs near 0.37 m^3/m^3.

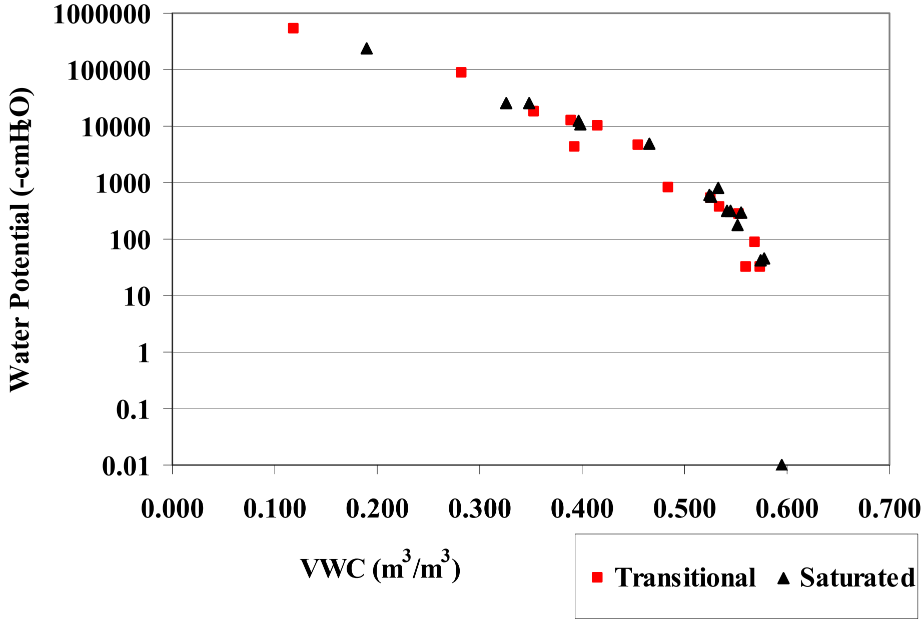


Figure 171. Moisture Release Curves for Transitional and Saturated at CFI SP-1.

The sand-dominated upland at CFI SP-1 resulted in a dissimilar curve compared to the curves developed for the other stations and to the composite curves for Mosaic K5 and Williams Co. (Figures 169-172). Saturation for the sandy soils occurs near 0.33 m^3/m^3 (Figure 172). The inflection point occurred at -20 cm H_2O, suggesting a much lower capillary fringe compared to the clayey soils. PWP pressures may be induced when the soils reach VWCs approximately 0.1 m^3/m^3, and field capacity occurs near 0.12 m^3/m^3.

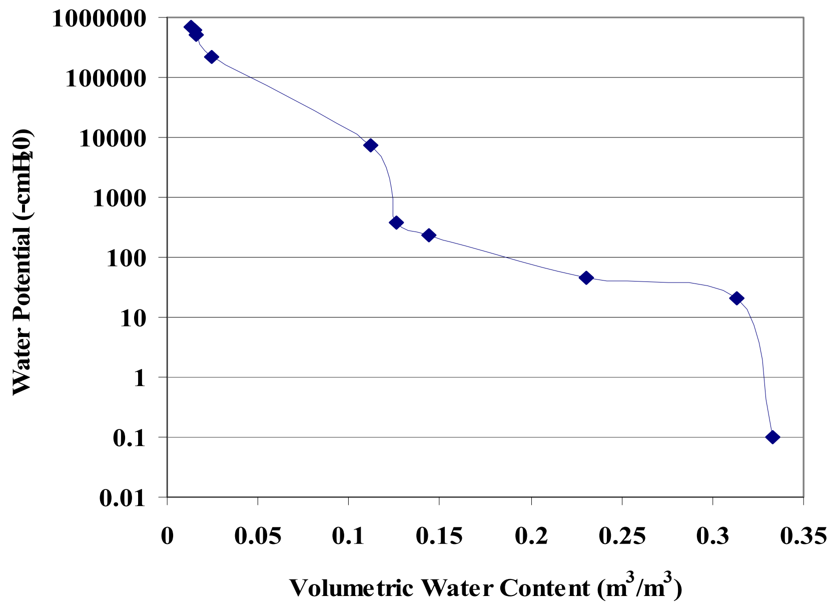


Figure 172. Moisture Release Curves for Upland at CFI SP-1.

Soil Moisture Analysis

Logging soil moisture probes were installed at various depths at the three ecohydrology stations of Mosaic K5, Williams Co., and CFI SP-1. Groundwater levels at Mosaic K5 recorded in the Groundwater-2 well from 5/15/07 to 9/27/07 are shown in Figure 173 and the corresponding soil moisture levels are shown in Figure 174. The soil moisture probes were installed within 1 m of the Groundwater-2 well, with an elevation change of less than 5 cm. The logger for the soil moisture probes experienced equipment failure on 9/27/07 and the 25-cm probe stopped recording on 9/14/07. Responses to increases in groundwater levels were much more pronounced at the 10-cm probe and to a lesser extent in the deeper probes. The 70- and 100-cm probes recorded levels with minimal variation from 0.62 and 0.59 m^3/m^3, respectively. The 25-cm probe recorded soil moisture levels consistently over 0.5 m^3/m^3, as did the 40-cm probe, with the exception during the end of the record. The 10-cm probe frequently recorded soil moisture levels below 0.40 m^3/m^3, with a minimum of 0.27 m^3/m^3 that corresponded to the deepest recorded water table.

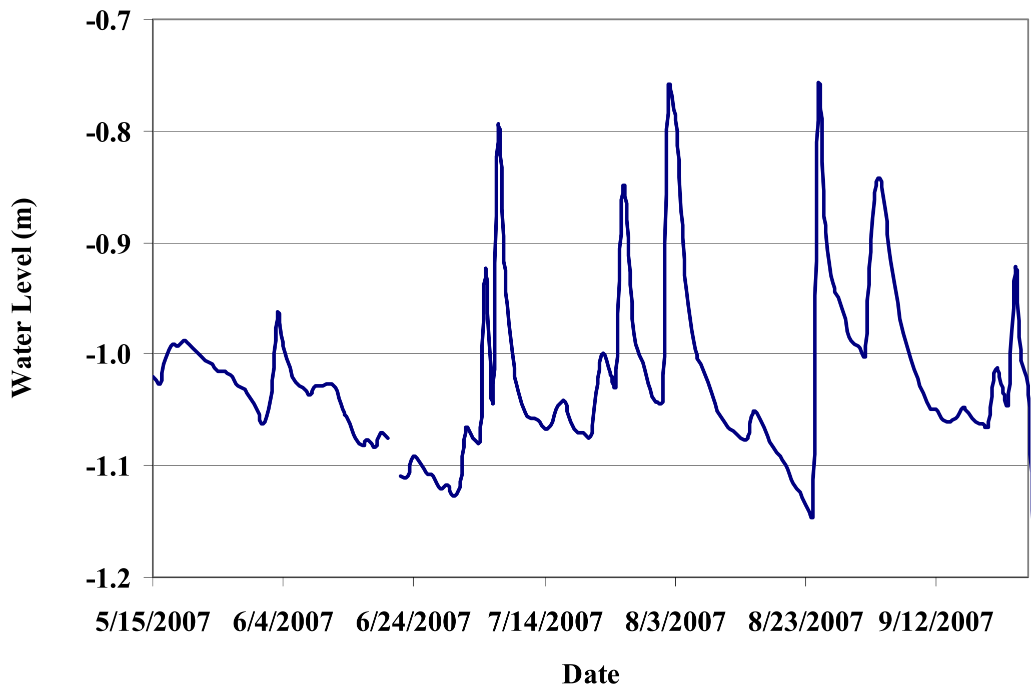


Figure 173. Groundwater-2 Levels at Mosaic K5.

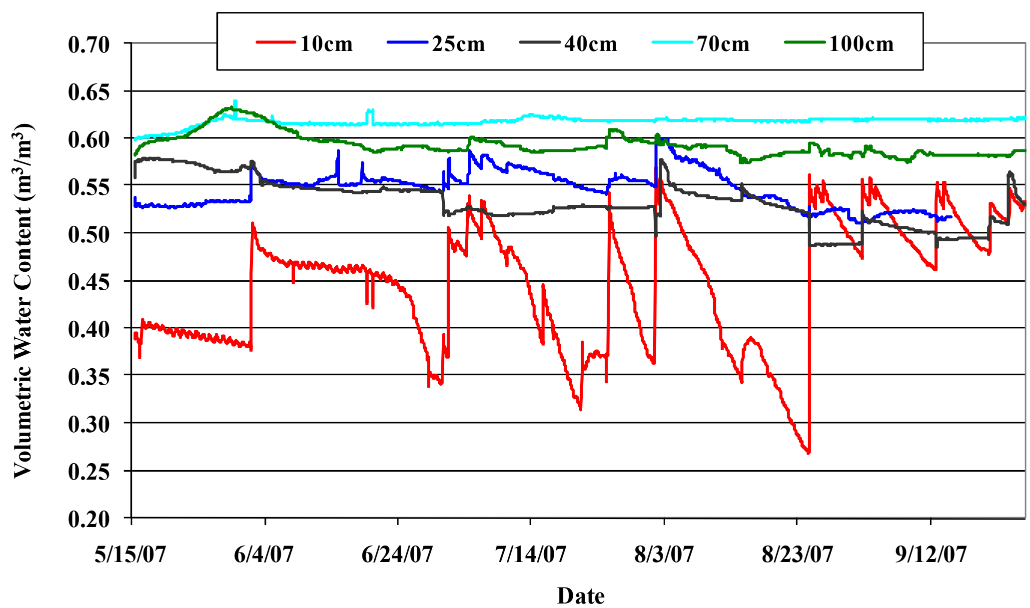


Figure 174. Upland Zone Soil Moisture Levels at Mosaic K5.

Groundwater levels recorded at Groundwater-3 at Mosaic K5, which was installed immediately adjacent to the soil moisture probes at the transitional station, are shown in Figure 175. The corresponding soil moisture levels recorded at the transitional zone are shown in Figure 176. Both the 25- and 70-cm probes experienced failure and stopped recording on 9/7/07. The 100-cm probe only worked briefly before it stopped recording on 6/25/07. The probes at 10, 25, and 40 cm all responded to the decreasing water table during June 2007 and recorded soil moisture levels below 0.3 m^3/m^3. The 70- and 100-cm probes did not experience lower levels during this decline period. Though the water table at this station was consistently closer to the ground surface compared to the water table at the upland station, lower soil moisture levels were recorded (Figures 173-176). While all probes except for the 10-cm probe recorded near or above 0.5 m^3/m^3 at the upland station, the 10-, 25-, and 40-cm probes' zone often fell below this VWC at the transitional station and experienced many more fluctuations (Figures 174 and 176). Similar to the groundwater levels, the 10- and 40-cm probes recorded more constant and elevated levels at the beginning of October and end of the growing season (Figures 175 and 176).

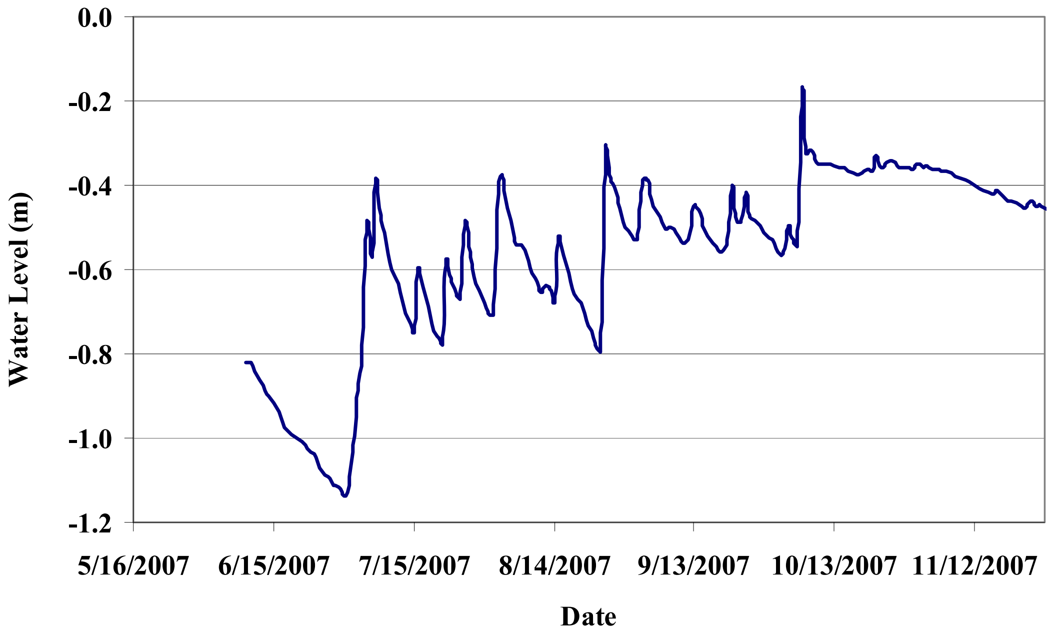


Figure 175. Groundwater-3 Levels at Mosaic K5.

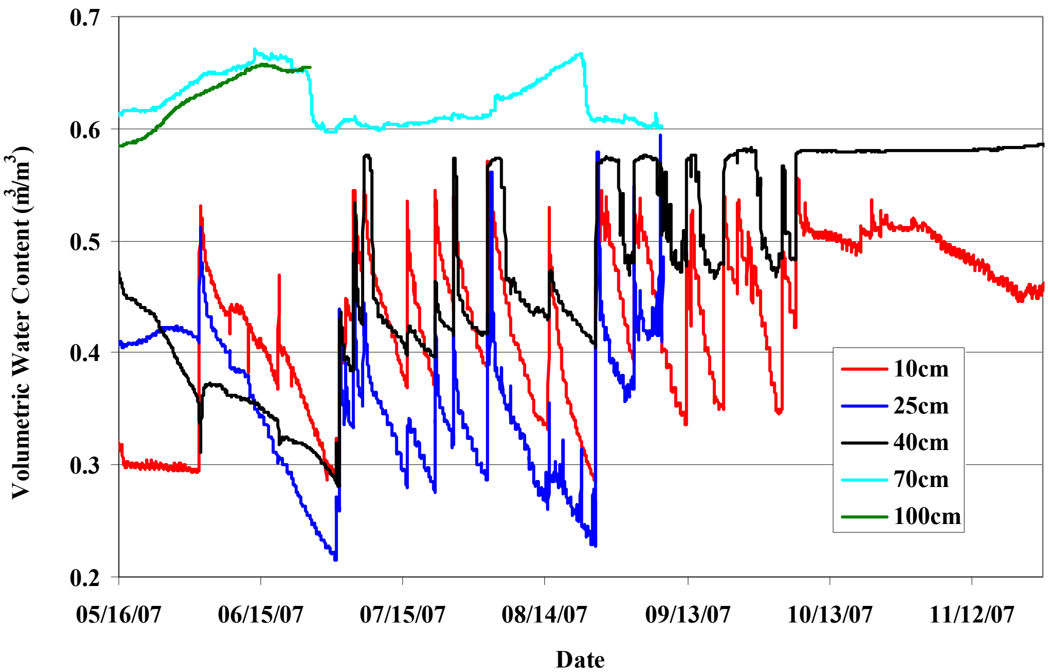


Figure 176. Transitional Zone Soil Moisture Levels at Mosaic K5.

Soil moisture levels recorded at the saturated zone of Mosaic K5 from 5/16/07 to 11/14/07 are shown in Figure 177. A well was not installed near this station but the ground surface at this location was 20 cm lower than the well at the transition zone. The 20- and 50-cm probes experienced equipment failure during the record. All probes recorded values over 0.52 m^3/m^3 and with the exception of two time periods, always over 0.55 m^3/m^3, including the shallower probes. The 20-cm difference in elevation between the two stations apparently caused significantly different soil moisture regimes, possibly a result of differences in water table depths (Figures 176 and 177). Similar to the transitional zone, the two working probes recorded more constant levels at the onset of the non-growing season.

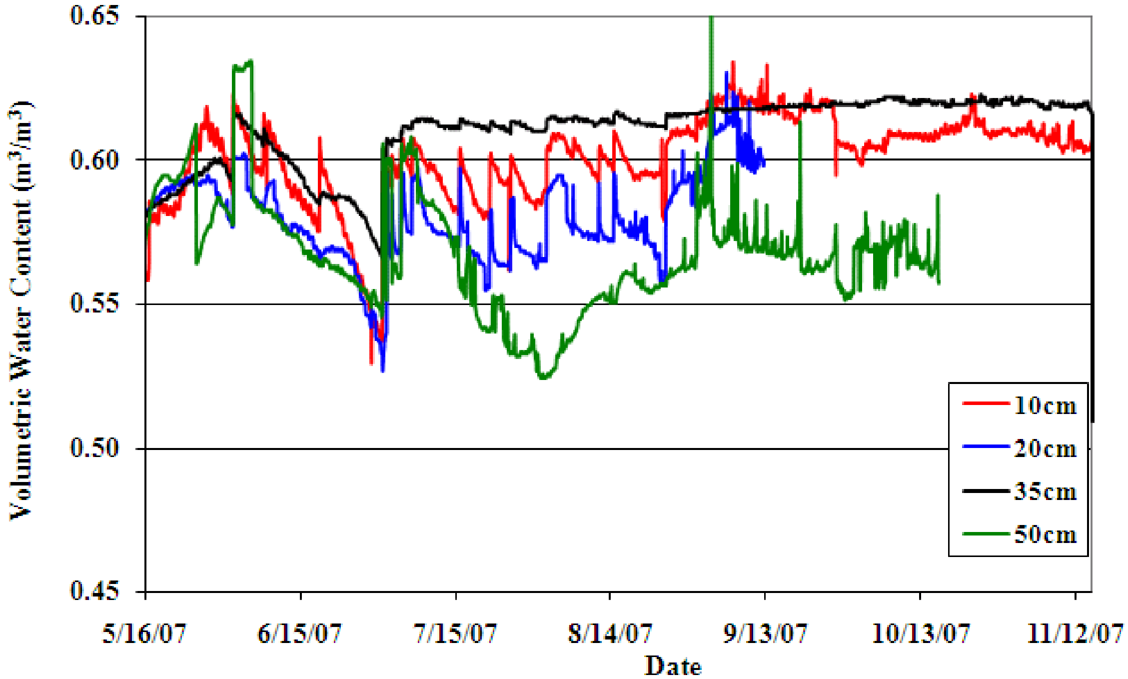


Figure 177. Saturated Zone Soil Moisture Levels at Mosaic K5.

The soil moisture probes at the upland station of Williams Co. were installed immediately adjacent to the Groundwater-2 well. Groundwater levels from 6/9/2007 to 9/25/07 and the associated soil moisture levels at the upland station are shown in Figures 178 and 179. The upland station experienced deeper water table depths than were observed at the upland station of Mosaic K5 (Figures 178 and 173). The water table was approximately and consistently 2 m below ground at Williams Co. and slight declines corresponded with decreasing soil moisture levels at the 10- and 25-cm probes (Figures 178 and 179). The 40- and 70-cm probes constantly recorded VWCs between 0.55 and 0.60 m^3/m^3. Of particular interest are the recordings from the 100-cm probe, which recorded levels similar to the shallower 40- and 70-cm probes, except during the month of July. The levels recorded at the 100-cm probe during July actually were less than those recorded at the shallowest probes. Other than this period, the 100-cm probe always recorded values higher than the 10- and 25-cm probes and ones near saturation, even at times when the 10- and 25-cm probes recorded the minimum values of the entire record. This observation suggests that a possible crack had intermittently developed around the 100-cm probe during July.

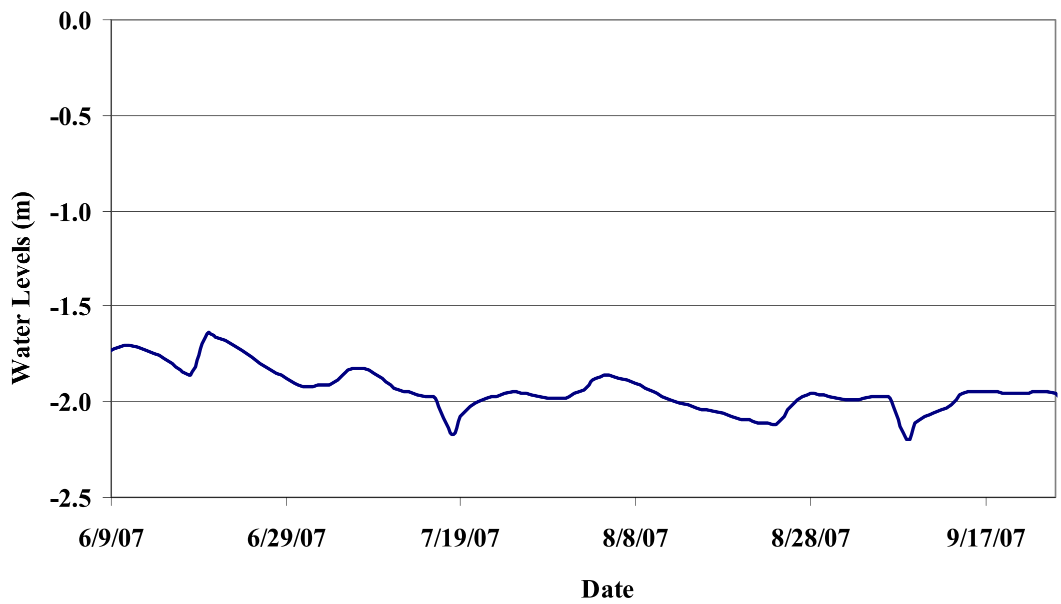


Figure 178. Groundwater-2 Levels at Williams Co.

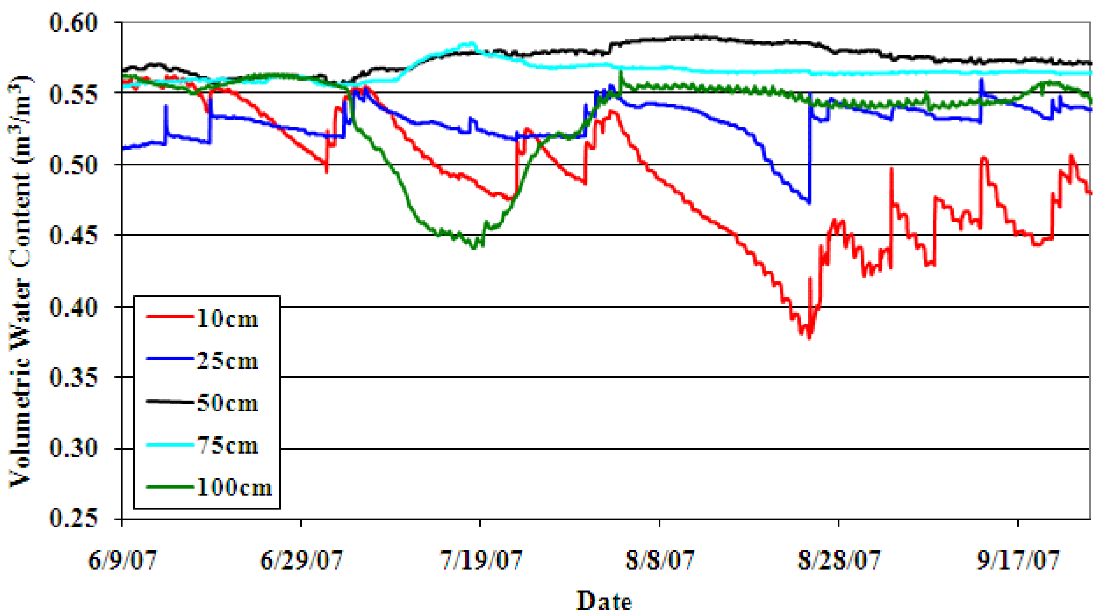


Figure 179. Upland Zone Soil Moisture Levels at Williams Co.

Soil moisture levels and the groundwater levels recorded at the adjacent well, Groundwater-3, for the transition station at Williams Co. are shown in Figures 180 and 181. The Groundwater-3 well was installed after the installation of the soil moisture probes, and the immediate increase in groundwater levels was somewhat due to equilibration of the well. The 100-cm probe, data not shown, only recorded for 9 days after installation before equipment failure and recorded values similar to those recorded at the 40-cm probe. The 40-cm probe stopped recording on 8/13/07. The groundwater levels at the transitional station also reached 2 m below, similar to the levels at the upland station, but were much more variable and often shallower (Figures 180 and 178). The soil moisture levels recorded at the 10-cm probe fluctuated between 0.40 and 0.50 m^3/m^3, and corresponded to changes in water table depth (Figure 181). The 25- and 40-cm probes were less affected by changes in groundwater levels, consistently recording over 0.55 m^3/m^3. The deepest working probe, however, recorded net drying over the entire period, with values approaching 0.45 m^3/m^3. The differences between the recordings of the 70-cm probe and the shallower probes (25 and 40 cm) may be a result of a developed crack or localized soil moisture regimes. The latter could be caused by heterogeneity in soil with depth.

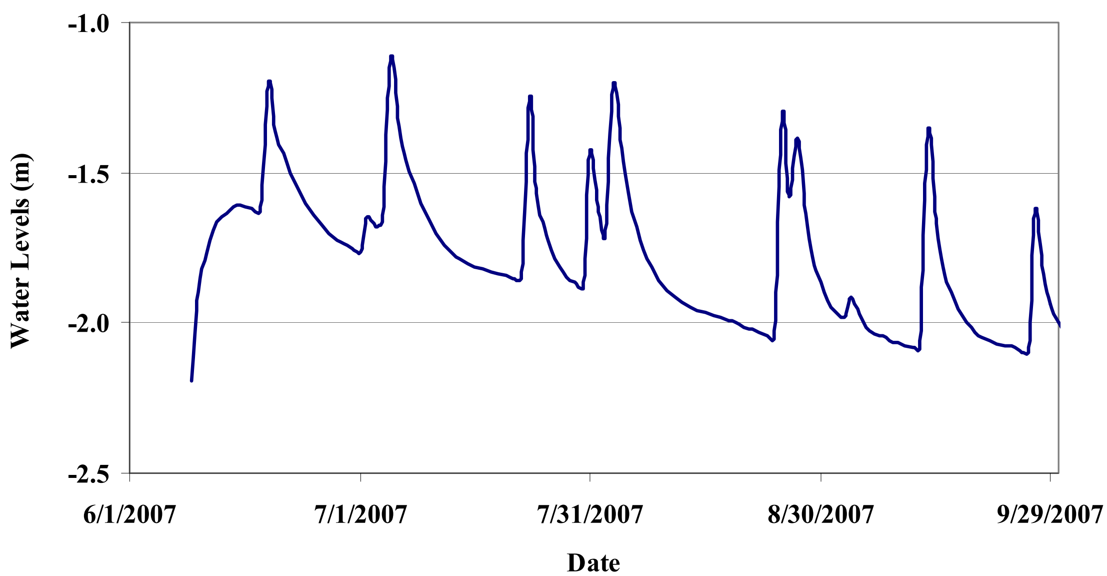


Figure 180. Groundwater-3 Levels at Williams Co.

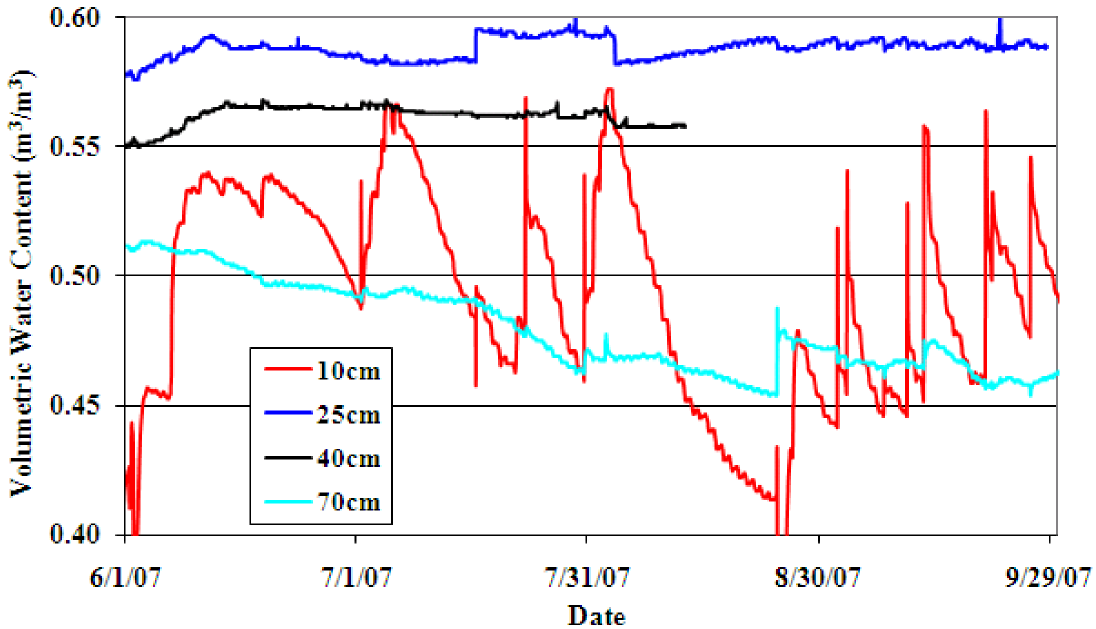


Figure 181. Transitional Zone Soil Moisture Levels at Williams Co.

Soil moisture probes were installed adjacent to the surface water well at Williams Co. This region, referred to here as the saturated zone, is one of the lowest elevation areas on the site but had not experienced inundation since fall 2006 (Figure 65). The water levels during the soil moisture data collection period ranged between 0.5 and 1.35 m below the ground surface. The fact this area had been historically inundated and soil moisture levels were collected during an extended dry period allowed an evaluation of soil moisture regimes during such a scenario. The 50-cm probe experienced failure soon after installation and was repaired on 8/4/07 when the 35-cm probe, which stopped logging on 7/9/07, was also repaired. The 10-, 20-, and 50-cm probes recorded values above 0.55 m^3/m^3, except during a drier period in late July (Figures 182 and 183). Higher values, however, were recorded at these probes during the end of the summer when lower groundwater levels were observed. The 35-cm probe also recorded near 0.60 m^3/m^3 before it stopped logging, but began recording much lower values shortly after repair. Values recorded by the 35-cm probe were slightly above 0.55 m^3/m^3 for 10 days after installation, but substantially decreased with the water table decline (Figures 182 and 183). The other probes, both shallower and deeper, did not record decreasing soil moisture levels during this time. Furthermore, the 35 cm probe recorded immediate increases up to 0.5 m^3/m^3 after rain events, with subsequent declines back to values below 0.3 m^3/m^3. The extreme drying observed at this probe and the fact that it recorded such drastic responses to, and drying following, rain events strongly suggests a crack had developed around the probe.

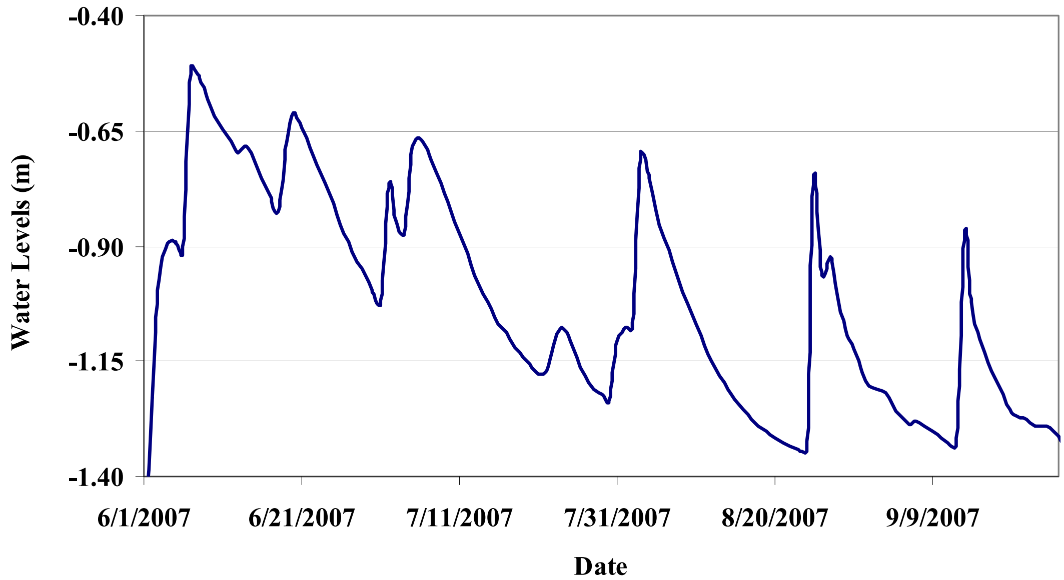


Figure 182. Groundwater Levels at Saturated Zone of Williams Co.

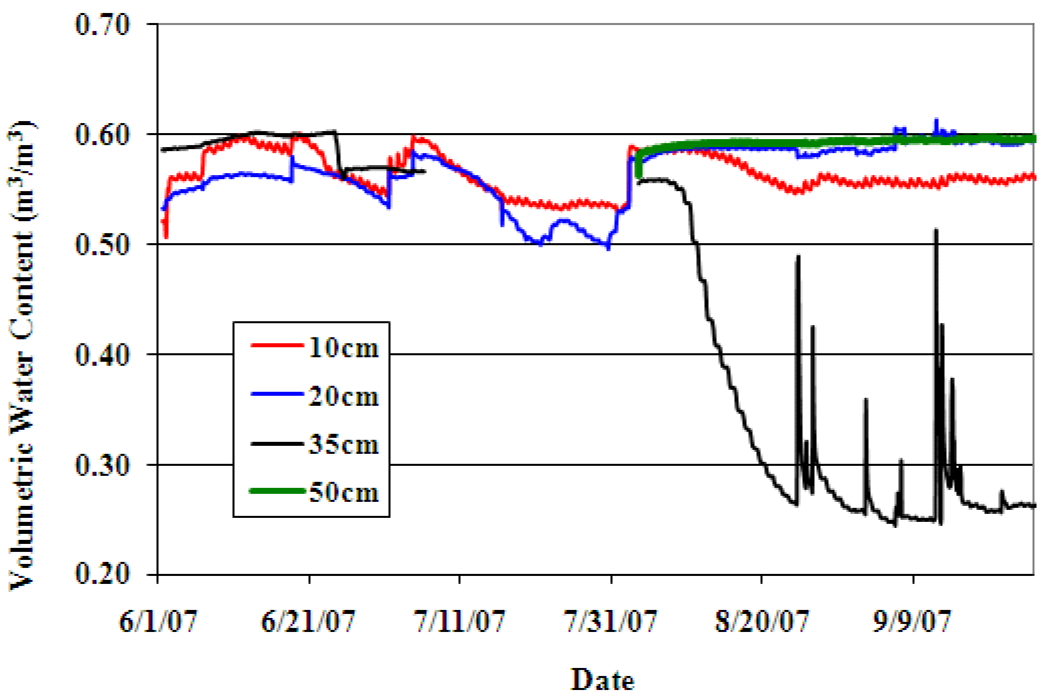


Figure 183. Saturated Zone Soil Moisture Levels at Williams Co.

The groundwater and soil moisture levels recorded at the upland station of CFI SP-1 are shown in Figures 184 and 185. The 10-cm probe stopped recording in early October 2007. All probes recorded fluctuating soil moisture levels that ranged from 0.08 to 0.2 m^3/m^3 and that corresponded to increases and decreases of groundwater levels (Figure 185). The sandier soils at this station resulted in the deeper probes recording more dynamic regimes and all probes recording much lower VWCs compared to the other upland systems evaluated (Figures 174, 179, and 185).

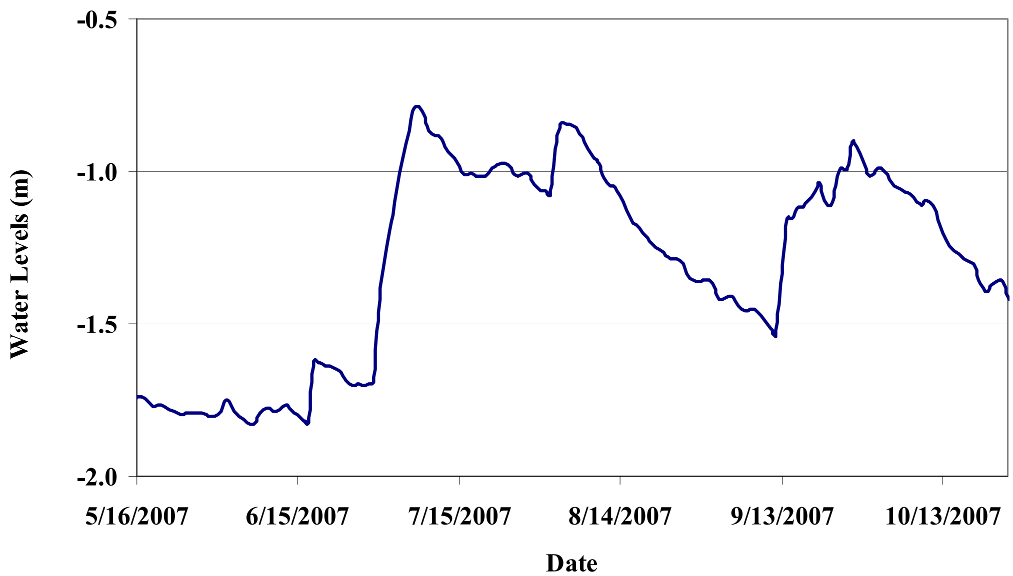


Figure 184. Groundwater-2 Levels at CFI SP-1.

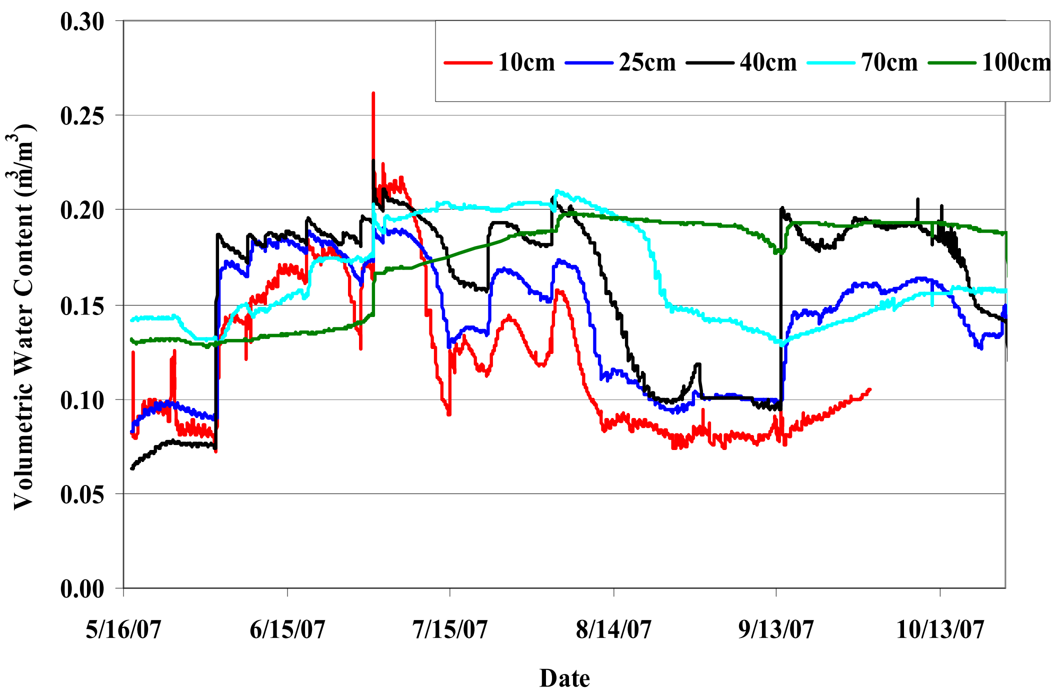


Figure 185. Upland Zone Soil Moisture Levels at CFI SP-1.

Groundwater and soil moisture levels recorded at the transitional station of CFI SP-1 are shown in Figures 186 and 187. Groundwater-3 was installed immediately adjacent to the probes and at a later date. The immediate increases in water levels after installation were somewhat due to well equilibration. The 10- and 25-cm probes recorded values consistently under 0.30 m^3/m^3 and that were more variable compared to the 40- and 100-cm probes which recorded values between 0.45 and 0.60 m^3/m^3. The top 30 cm of soil at this region was similar to the sandy soils at the upland station, whereas below 30 cm was typically more clay-dominated. The different soil types resulted in markedly different soil moisture regimes with depth. Interestingly, the 70-cm probe's recordings were below recordings from both the shallower 40-cm probe and the deeper 100-cm probe and were more fluctuating, similar to the 10- and 25-cm probes. Soil heterogeneity with depth and some interlacing sand near the 70-cm probe may have caused the lower and more fluctuating VWCs.

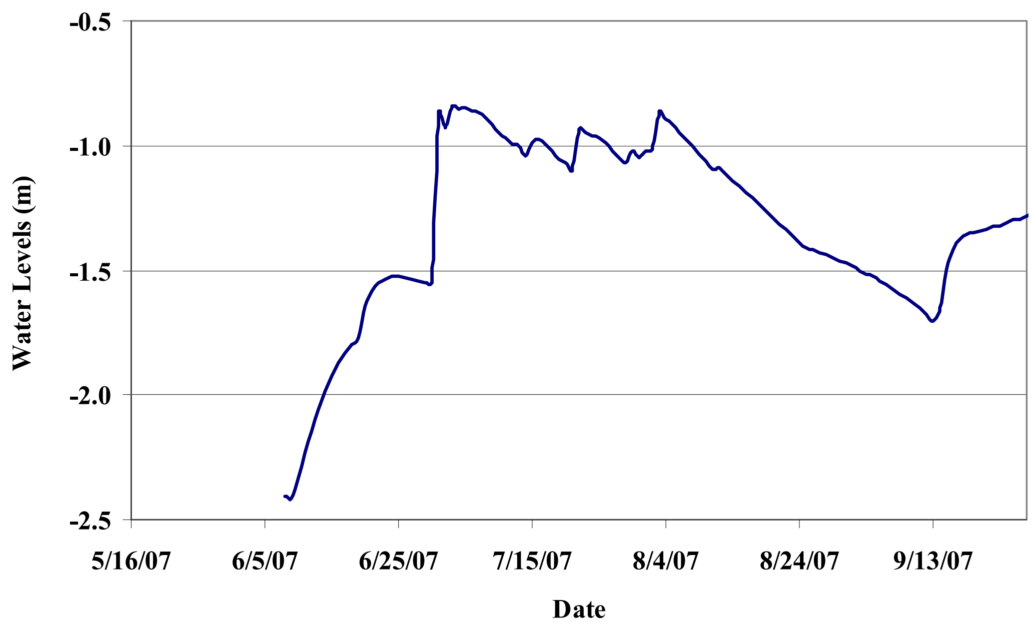


Figure 186. Groundwater-3 Levels at CFI SP-1.

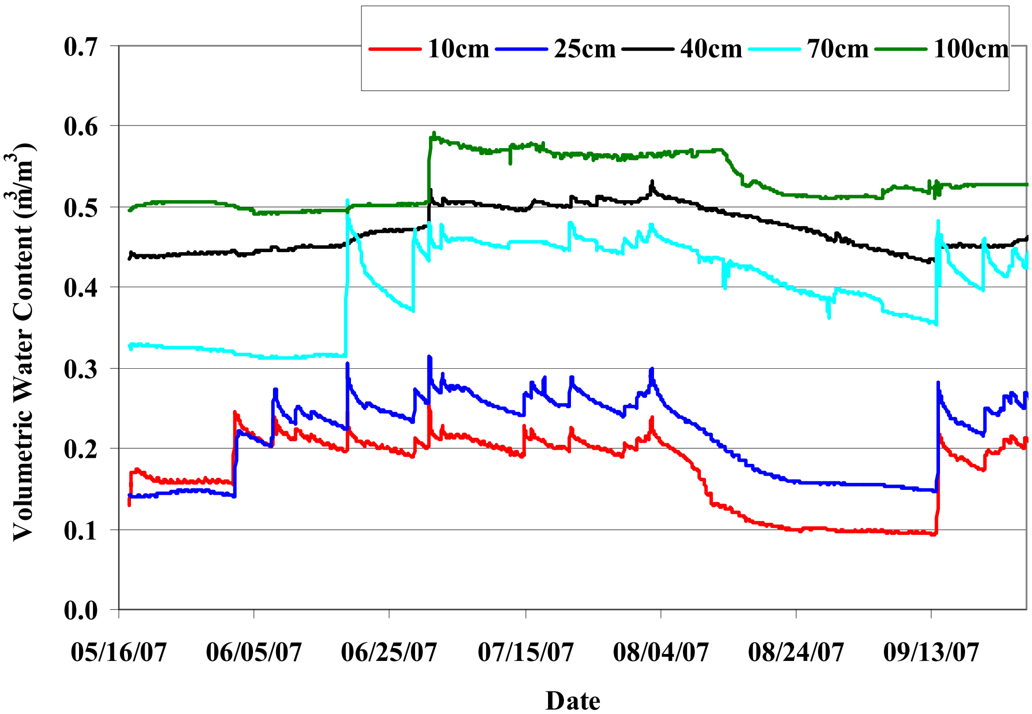


Figure 187. Transitional Zone Soil Moisture Levels at CFI SP-1.

Similar to Williams Co., soil moisture probes were installed adjacent to the surface water well at CFI SP-1, referred to here as the saturated station. The surface water feature at CFI SP-1 had not experienced inundation since fall 2006 (Figure 71). The groundwater levels recorded at the surface water well and the soil moisture levels are shown in Figures 188 and 189. The 35-cm probe stopped recording in early August. All probes recorded values over 0.45 m^3/m^3, with the deepest probe consistently recording near 0.6 m^3/m^3. The 60-cm change in groundwater levels did not cause much variation in the soil moisture levels (Figures 188 and 189). The shallowest probe actually recorded values higher than those recorded at the 20- and 35-cm probes and often near 0.55 m^3/m^3, again suggesting possible soil heterogeneity with depth.

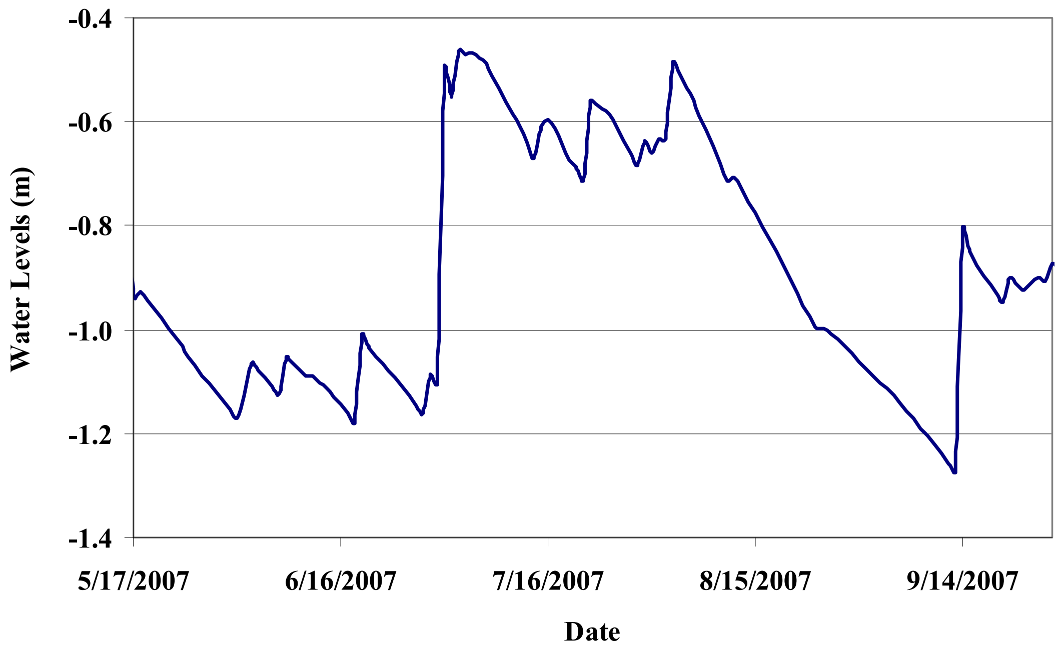


Figure 188. Groundwater Levels at Saturated Zone of CFI SP-1.

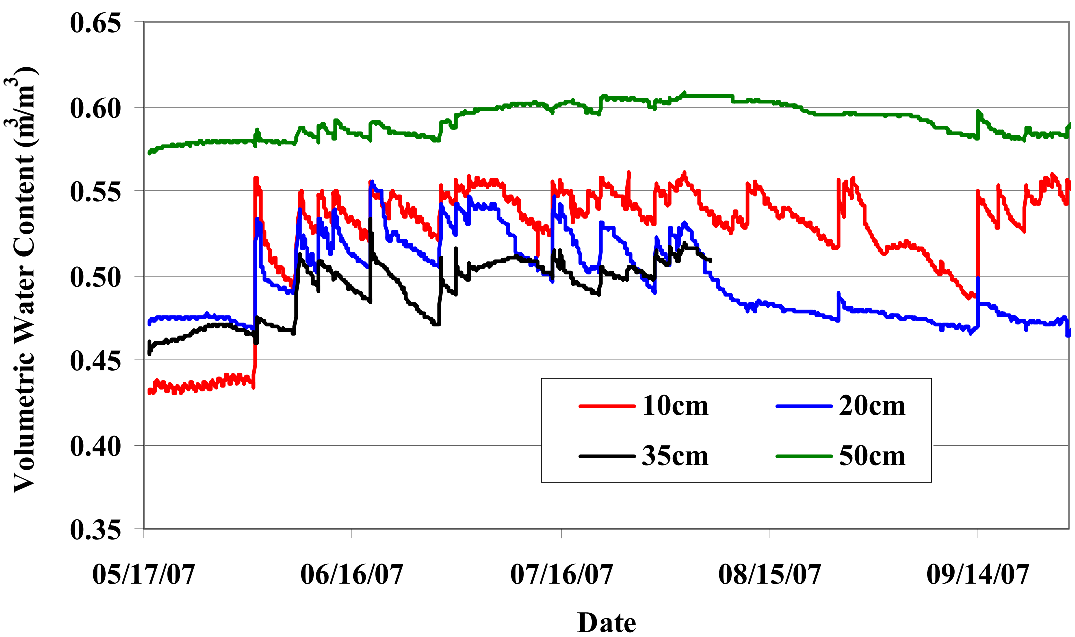


Figure 189. Saturated Zone Soil Moisture Levels at CFI SP-1.

Root Biomass Analysis

Salix caroliniana roots were sampled at three trees surrounding the soil moisture stations of Mosaic K5 and CFI SP-1 using the coring method and up to 1.5 m below the ground surface. Additionally, three trees in inundated regions at Mosaic K5 were sampled. Averages of dry root biomass per depth at each station of Mosaic K5 are shown in Figures 190-193. Roots were only successfully sampled up to 1.05 m below ground at the inundated zone as a result of the water pressure limiting sufficient removal of the samples when sampling the deeper cores. The deepest peak of biomass allocation occurred at the upland zone and a depth of 105 cm (Figure 190). The upland zone also had the highest total root biomass. There was a slight trend of shallower biomass allocation with wetter conditions, but with biomass equal to or greater than 3 kg/m^3 at a depth of 90 cm even in the wettest locations (Figures 190-193).

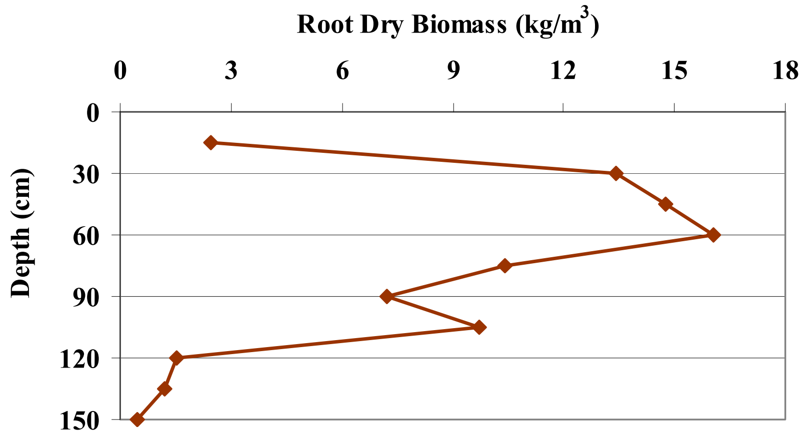


Figure 190. Upland Zone Root Biomass at Mosaic K5.

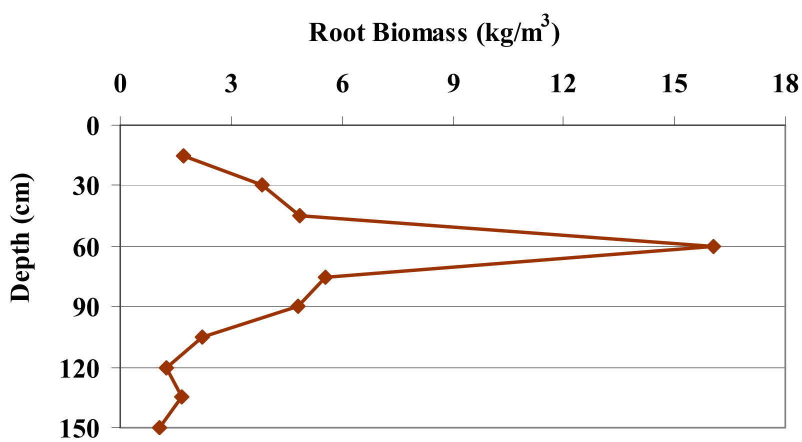


Figure 191. Transitional Zone Root Biomass at Mosaic K5.

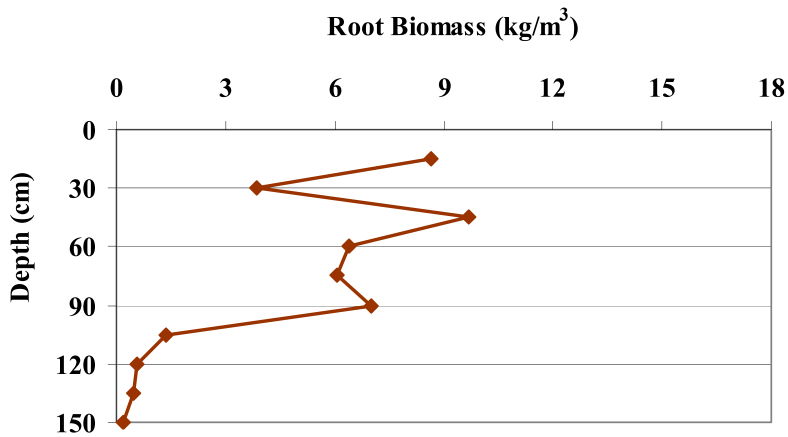


Figure 192. Saturated Zone Root Biomass at Mosaic K5.

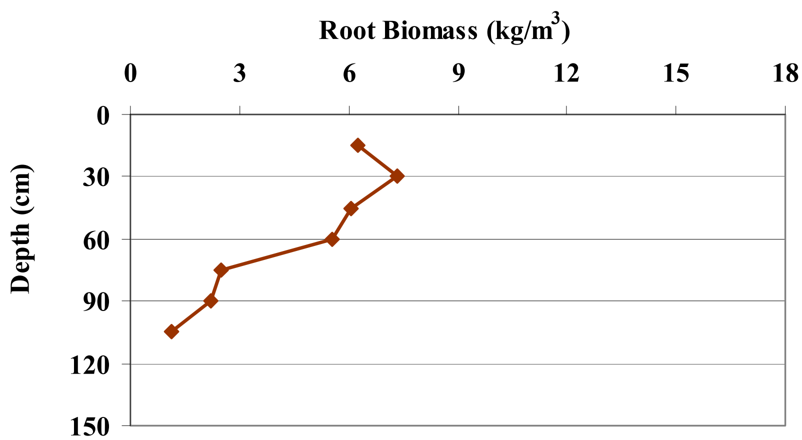


Figure 193. Inundated Zone Root Biomass at Mosaic K5.

Averages of dry root biomass per depth for the upland, transitional, and saturated stations at CFI SP-1 are shown in Figures 194-196. Root biomass allocation appeared to be different at CFI SP-1 from Mosaic K5, with peaks being more restricted to shallower depths (Figures 194-196). Furthermore, the total allocation was lower at CFI SP-1 compared to the upland, transitional, and saturated stations at Mosaic K5 (Figures 190-192 and 194-196). It should be noted that the trees sampled at the transitional and saturated zones of CFI SP-1 had adventitious rooting, whereas the trees sampled at these zones of Mosaic K5 did not. Only the inundated trees at Mosaic K5 had adventitious rooting. The adventitious rooting on the sampled trees at CFI SP-1 demonstrates that historical flooding had occurred at both the saturated and transitional zones. Flooded conditions in these zones could have resulted in root biomass allocation being similar to that observed at the inundated conditions of Mosaic K5 (Figures 193, 195, and 196). The upland trees sampled at CFI SP-1 had shallower and less total biomass than the upland trees at Mosaic K5 (Figures 194 and 190). Anecdotally speaking, it seemed that the sampled roots of the upland trees at Mosaic K5 tended to vertically spread, whereas the roots at the upland of CFI SP-1 had a more lateral spread.

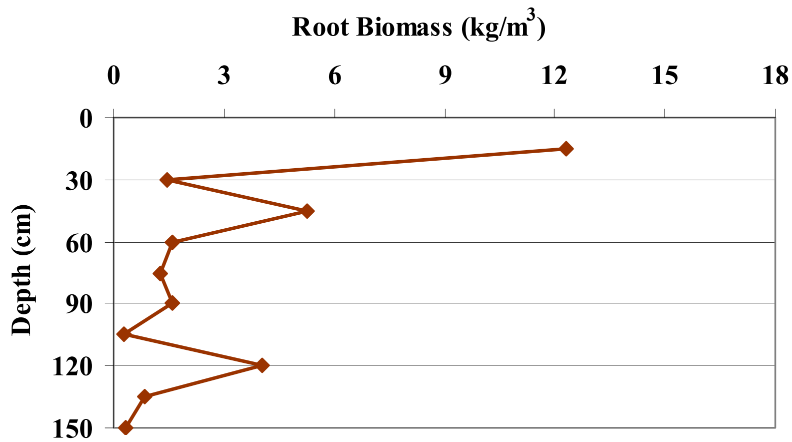


Figure 194. Upland Zone Root Biomass at CFI SP-1.

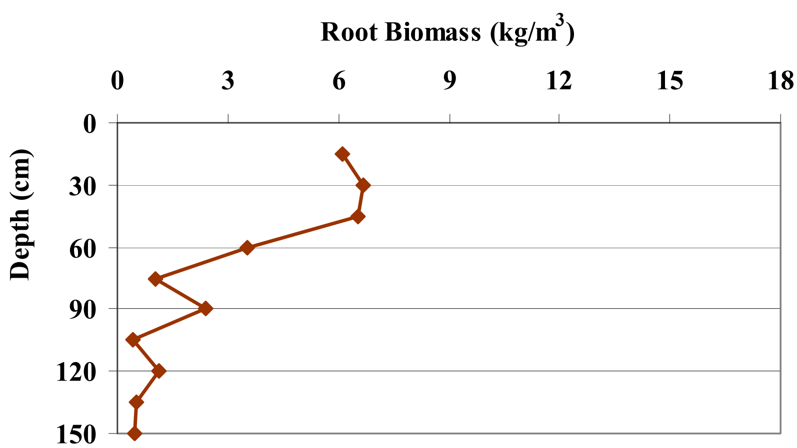


Figure 195. Transitional Zone Root Biomass at CFI SP-1.

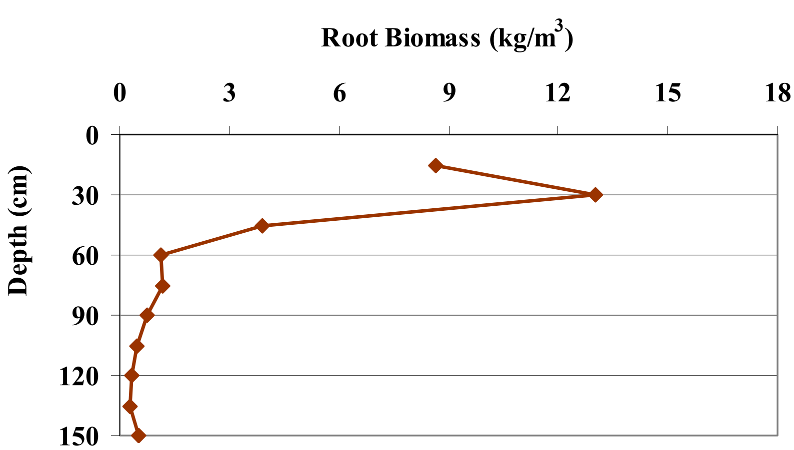


Figure 196. Saturated Zone Root Biomass at CFI SP-1.

Transpiration Analysis

Transpiration data was collected over 14 days at each ecohydrology station of Mosaic K5 and CFI SP-1. Four stems of separate *Salix caroliniana* trees were instrumented at each station and hourly flow rates were collected for each stem. An example of the raw data collected is shown in Figure 197. The hourly flow rates were converted to daily flow rates for each stem and were divided by the stem cross-sectional area to obtain daily stem flow rates as g H_2O/cm^2/day. The daily stem flow rates were indexed with the associated daily Penman-estimated ET, and the flow rates for the four stems were averaged to obtain average stem flow/Penman ET for each station. Indexing with the daily Penman-predicted ET allowed comparison among stations and sites while excluding climatic effects. Average daily stem flow rates at the four stations of Mosaic K5 are shown in Figure 198 along with standard deviations. The upland stems had the highest average flow rates, followed by the transitional stems. The flow rates for the wetter regions, saturated and inundated zones, were comparable and lower than the upland and transitional zones. These results suggest that stress was not induced in the most upland and driest regions.

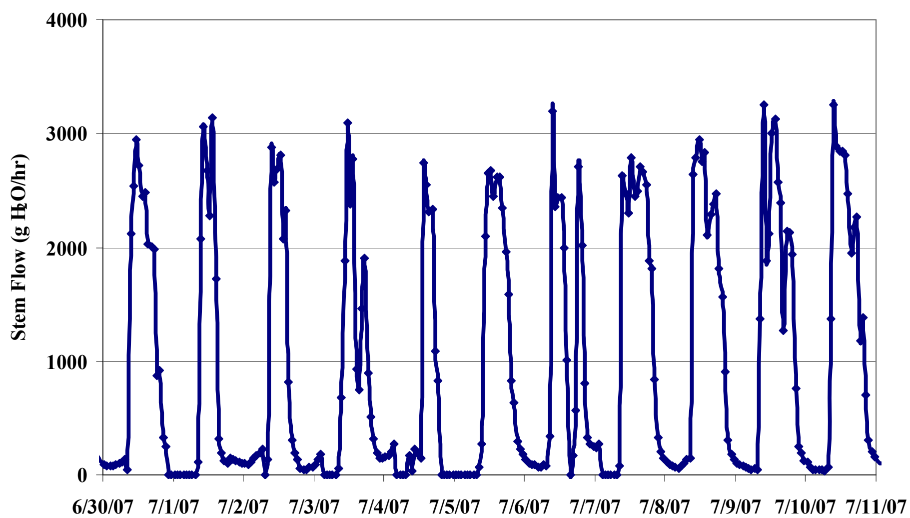


Figure 197. Transpiration Data for Stem at Upland Zone at Mosaic K5.

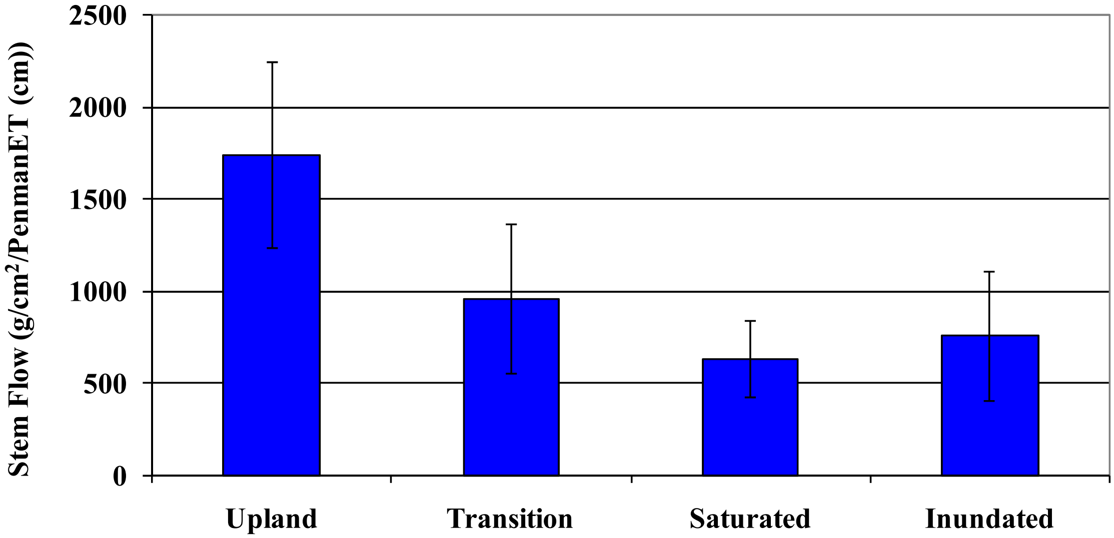


Figure 198. Stem Flow at Mosaic K5.

Basal area from a 30 × 6 m^2 belted transect that surrounded each station, Equation [22], and the daily stem flow rates of each stem were used to scale up the flow data to obtain daily stand level transpiration (cm/day). A daily stand level transpiration was calculated for each daily stem flow rate and each was indexed with Penman-predicted evapotranspiration. The daily stand transpiration rates indexed with Penman rates were averaged to calculate average daily stand transpiration/Penman ET (cm/cm) for each station (Figure 199). Scaling up to the stand level at Mosaic K5 resulted in an opposite trend compared to the stem flow rates (Figures 199 and 198). Stand level transpiration in the inundated conditions was on average 1.8 times greater than the total ET rate as estimated with the Penman method. The average of the indexed rates at both the transition and saturated zone were over one, also demonstrating stand transpiration rates greater than Penman-predicted ET. While stem flow rates were lower in the wetter regions, scaling up to stand level reveals that stress was not induced in flooded conditions and transpiration rates were higher than typical ET rates (Figure 199).

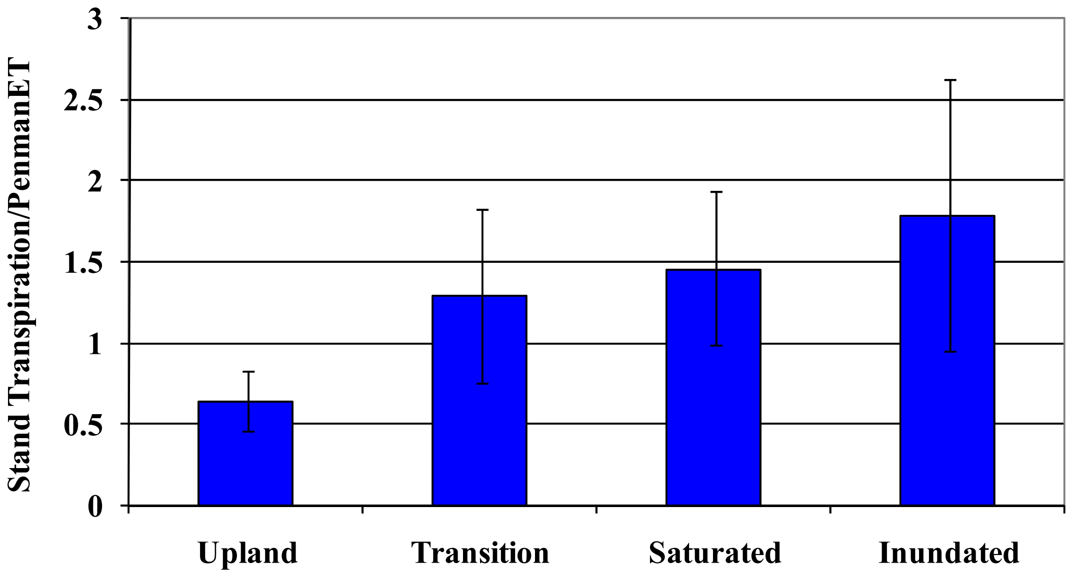


Figure 199. Stand Level Transpiration at Mosaic K5.

Total basal areas measured at transects surrounding each station and average leaf area per stem area of the instrumented stems are shown for Mosaic K5 in Figure 200. Tent caterpillars caused significant damage to the canopies of the instrumented stems at the transitional area, and thus no leaf area data is shown. Total basal area increased with wetter conditions, causing higher stand transpiration rates in those areas (Figure 200 and 199). With more basal area, however, there were more trees and less space for canopies to spread, causing less total leaf area per stem area in the wetter regions compared to the upland zone (Figure 200). With more leaf area per stem area, higher stem flow rates were possible in the upland zone (Figure 198).

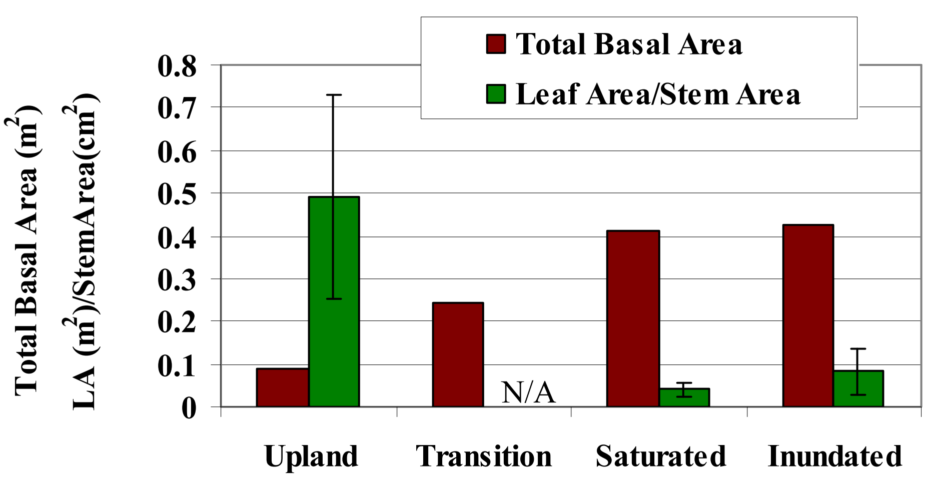


Figure 200. Total Basal Area on Transects and Leaf Area per Stem Area of Instrumented Stems at Mosaic K5.

Similar to the results at Mosaic K5, stem flow rates were higher in the drier zones at CFI SP-1, while stand transpiration was higher in the wetter zones (Figures 201 and 202). Collection of data in an inundated zone was not possible at CFI SP-1 since the entire site was dry during summer 2007. Average stand transpiration was 1.22 times greater than Penman-estimated ET at the saturated zone and 1.07 times greater at the transition zone (Figure 202). The upland zone had stand transpiration rates much lower, and on average 70% lower, than total ET predicted by the Penman method. A similar comparison to the stand transpiration and stem flow rates as performed with the rates at Mosaic K5, suggests little stress in either the driest or wettest regions of CFI SP-1 studied (Figures 201 and 202).

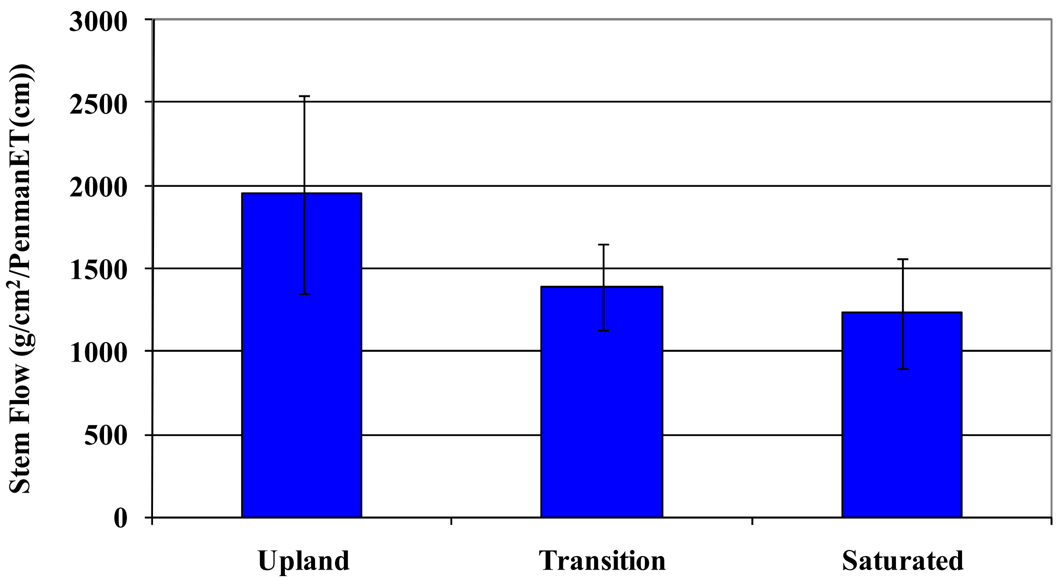


Figure 201. Stem Flow at CFI SP-1.

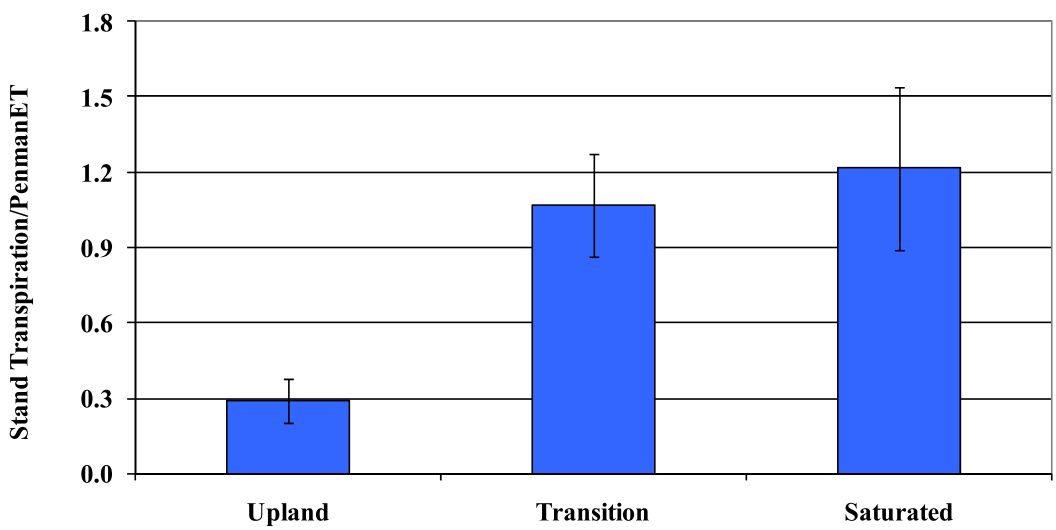


Figure 202. Stand Level Transpiration at CFI SP-1.

Total basal area increased with wetter conditions at CFI SP-1, again explaining the higher stand transpiration rates in the wetter regions (Figures 203 and 202). The low basal area at the upland zone resulted in much lower stand transpiration rates (Figures 203 and 202). There was higher leaf area per stem area in the upland regions than the wetter regions, but leaf area was much more variable among the instrumented stems at the upland region (Figure 203). While the stem flow rates were higher at the transitional zone compared to the saturated zone, average leaf area per stem area of instrumented stems was lower (Figure 203).

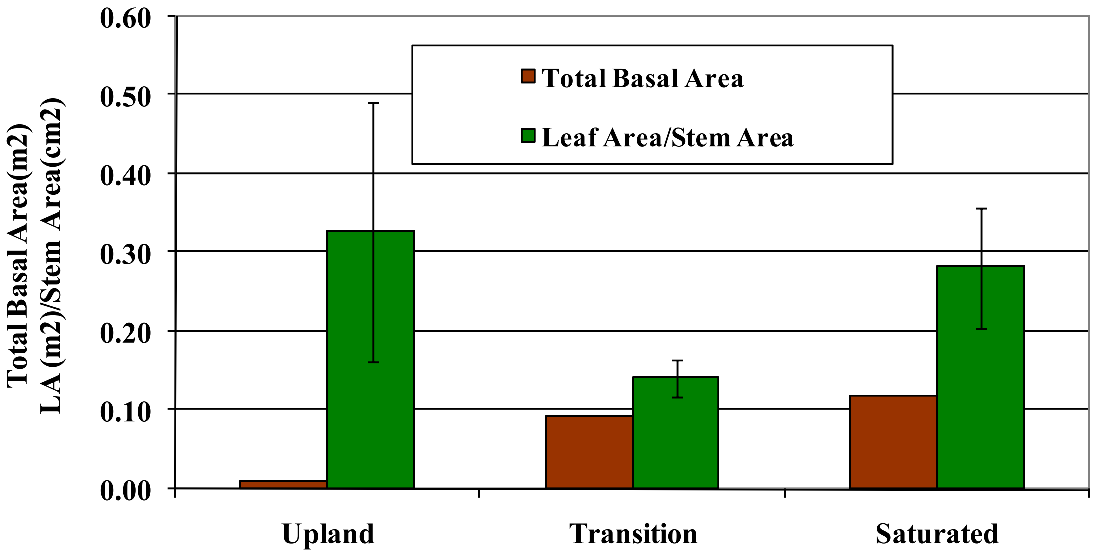


Figure 203. Total Basal Area on Transects and Leaf Area per Stem Area of Instrumented Stems at CFI SP-1.

Mosaic K5 experienced higher stand transpiration rates in the upland, transitional, and saturated zones compared to the rates measured at these zones of CFI SP-1 (Figure 204). The results from the transitional and saturated zones of both sites and the inundated zone of Mosaic K5 demonstrated that transpiration from *Salix caroliniana* alone was over typical total ET, as estimated with the Penman method. It should be noted, however, that *Salix caroliniana* was by far the dominant species in the transitional, saturated, and inundated conditions and likely contributed the most to total ET of the stands.

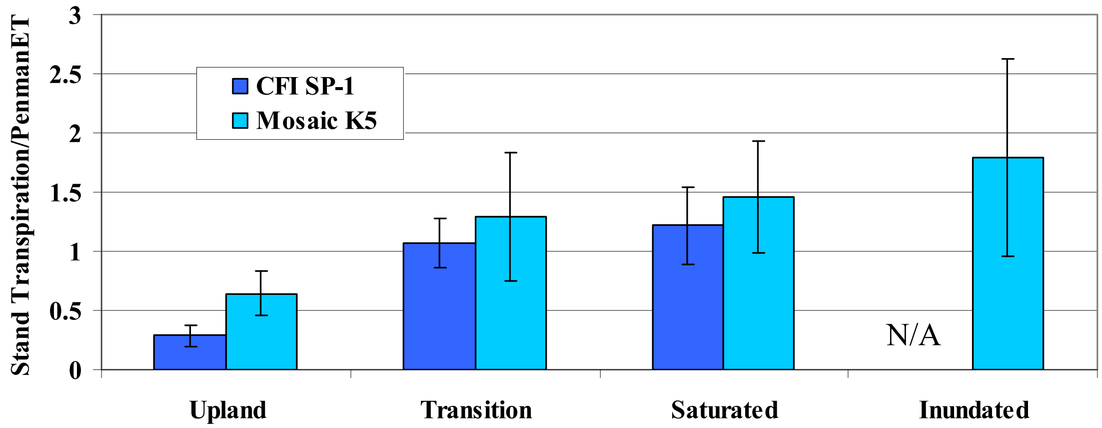


Figure 204. Stand Transpiration at CFI SP-1 and Mosaic K5.

EVALUATION OF B. RUSHTON FIELD TRIALS

Tree Populations in Relation to Environmental Factors

Results pertaining to tree populations are described below, first for cypress-gum plots (see Table 4 and Figure 14 for descriptions), and second for hydric swamp plots (see Table 5 and Figure 15 for descriptions).

Tree Survival in Cypress-Gum Plots

Table 77 summarizes the planted tree survival percentages after 1, 3, and 20 years. Aggregating all three species, trees at the CFI site had the highest survival after 20 years (78%), and trees at TEN had the lowest (22%). Mortality of *Taxodium distichum* was less than 2%/year between years 3 and 20 at all but the TEN site. Survival of *Fraxinus pennsylvanica* after 20 years was the poorest at HOM (8%), but highest of the three species at CFI (70%) and TEN (25%), two of the four sites with cypress-gum plots. *Nyssa aquatica* had the poorest survival in the initial year at the sites, but the survival rate between years 3 and 20 was the best of the three species at OHW and TEN, and better than *Fraxinus pennsylvanica* at CFI and HOM. Aggregating all sites, *Taxodium distichum* survived best after 20 years (34%), though *Fraxinus pennsylvanica* had the best survival at the end of the three years (70%). Compared with the survival rate during the first year, all species had an improved annual survival rate between years 3 and 20.

Table 77. Tree Survival from Initial Planting in 25 Sampled Cypress-Gum Plots.

Species	No. Planted	% Survival		
		1 Yr	3 Yrs	20 Yrs
Fraxinus pennsylvanica	651	72	70	29
Nyssa aquatica	837	44	34	18
Taxodium distichum	837	66	55	34

Tree Survival in Hydric Swamp Plots

Table 78 summarizes tree survival in hydric swamp plots after 1 and 19 years. *Acer rubrum*, *Fraxinus caroliniana, Taxodium distichum,* and *Ulmus americana* were the only species present in sampled plots after 19 years. No individuals of *Gordonia lasianthus*, *Nyssa sylvatica*, *Persea palustris*, *Quercus laurifolia*, or *Sabal palmetto* were found surviving in any of the plots after 19 years. Only one individual of *Ulmus americana* survived 19 years. Of the three other surviving species, total survival after the first year for each was greater than 85%. Survival of *Acer rubrum* after 19 years was 20% or less at all sites, with no surviving individuals found at TEN. *Fraxinus caroliniana* had the best survival in hydric swamp plots. At both OHW and TEN, all individuals survived after 19 years, a few having resprouted after the original stem died during the initial year. About half of *Taxodium distichum* trees that were surviving after one year survived 19 years, except at OHW where 20-year survival was only 12%, due to high mortality in two plots.

Table 78. Tree Survival from Initial Planting in 12 Sampled Hydric Swamp Plots.

Species	No. Planted	% Survival	
		1 Yr	19 Yrs
Acer rubrum	126	94	6
Fraxinus caroliniana	72	99	82
Taxodium distichum	216	89	31

Monthly Water Level Sampling vs. Continuous Recording

On the two sites (CFI and TEN) where continuous data-logging water level recorders were stationed, the average of hourly recorded water levels from these recorders was close to the average of monthly sampled water levels taken on field visits. At CFI, the average of the monthly sampled water levels was 0.66 m, and the average of the hourly sampled water levels was 0.65 m. At TEN, the average of the monthly sampled water levels was -0.18 m, and the average of the hourly sampled water levels was -0.15 m. The close proximity of the monthly and hourly sampled water levels (within 3 cm) indicates that the monthly sampled water levels provided an accurate average water level for the time period on these two sites.

Hydrology in Cypress-Gum Plots

Figure 205 shows the percentage of a plot that was inundated at the time of monthly water level sampling. Variation of the inundated area occurs within and between sites, with some obvious trends apparent. Plots at CFI demonstrated a range of inundation, varying from R1 which was almost totally inundated on all dates, to plot R6, which was at most 15% inundated. Thus all trees at R1 stood in standing water much of the season, whereas water level was below ground for most trees in R6. Nearly all eight plots at HOM were inundated upon every visit. At OHW, plots R2A and B, adjacent plots on a pond fringe, were more than 50% inundated in 4 of 5 months sampled, whereas about one-third of R1A, which crosses a drainage channel, was consistently covered in water. At TEN all plots were dry in May but for most of the season more than 50% of R2A and R2B were inundated. R5A, R5B, R6A, R6B, R7A, and R7B are on a pond fringe, and all plots were mostly inundated when sampled in July and August, but on visits earlier and later in the season were wet only in the deepest ends, if at all.

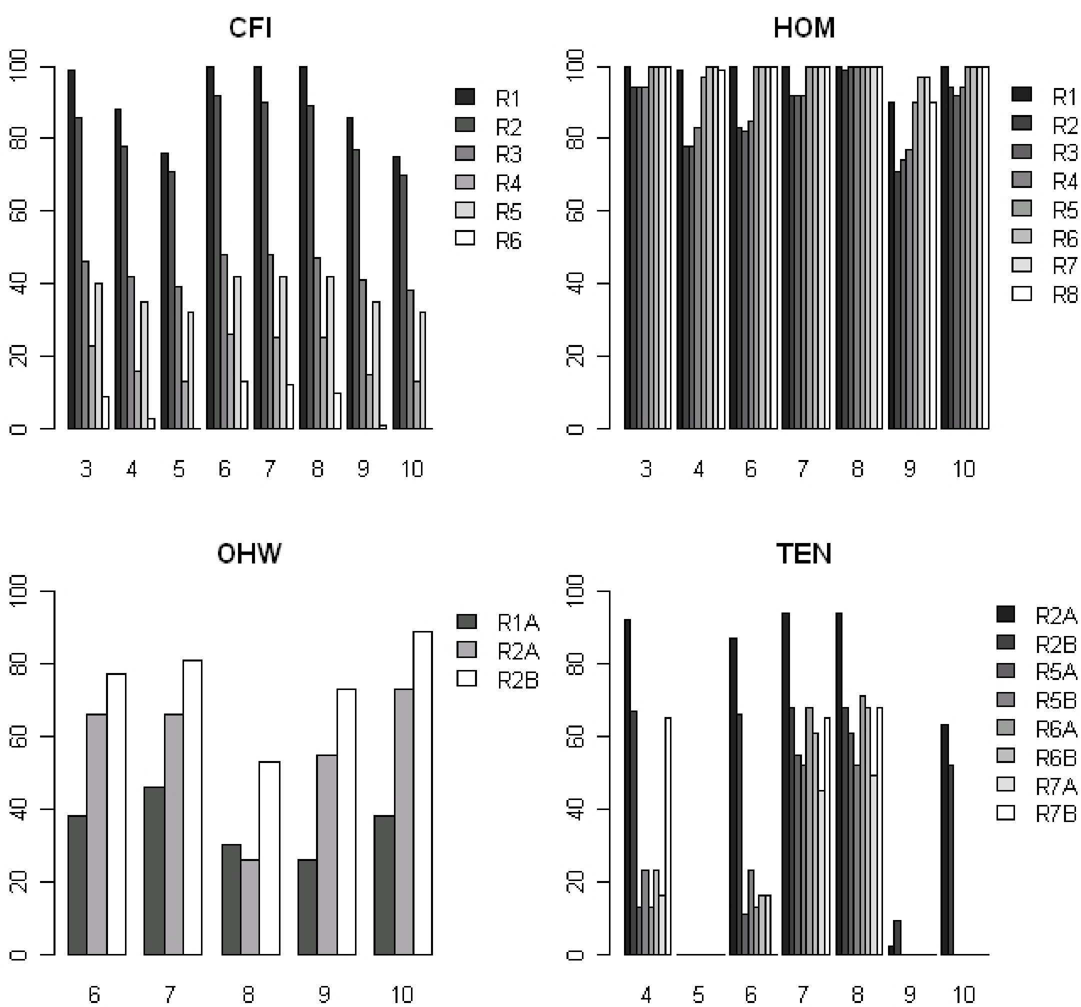


Figure 205. Percentage of Plot Inundated at Time of Monthly Sampling for All Sampled Months in 2005 on Cypress-Gum Plots.

Hydrology in Hydric Swamp Plots

Occupying less of a gradient than cypress-gum plots, hydric swamp plots exhibit a more uniform response to water level than cypress-gum plots (see Figure 206). Many plots were inundated through the season, including all plots at PRP and H1 and H4 at OHW, whereas others, such as OHW H2, were dry at every sampling. Sites at TEN all were completely dry when sampled during May, and only H2 and H3 had a small area inundated at the September sampling, but during other months H2 and H3 were completely inundated. TEN and OHW both had two rather wet and two dry sites, whereas at PRP, all sites were wet.

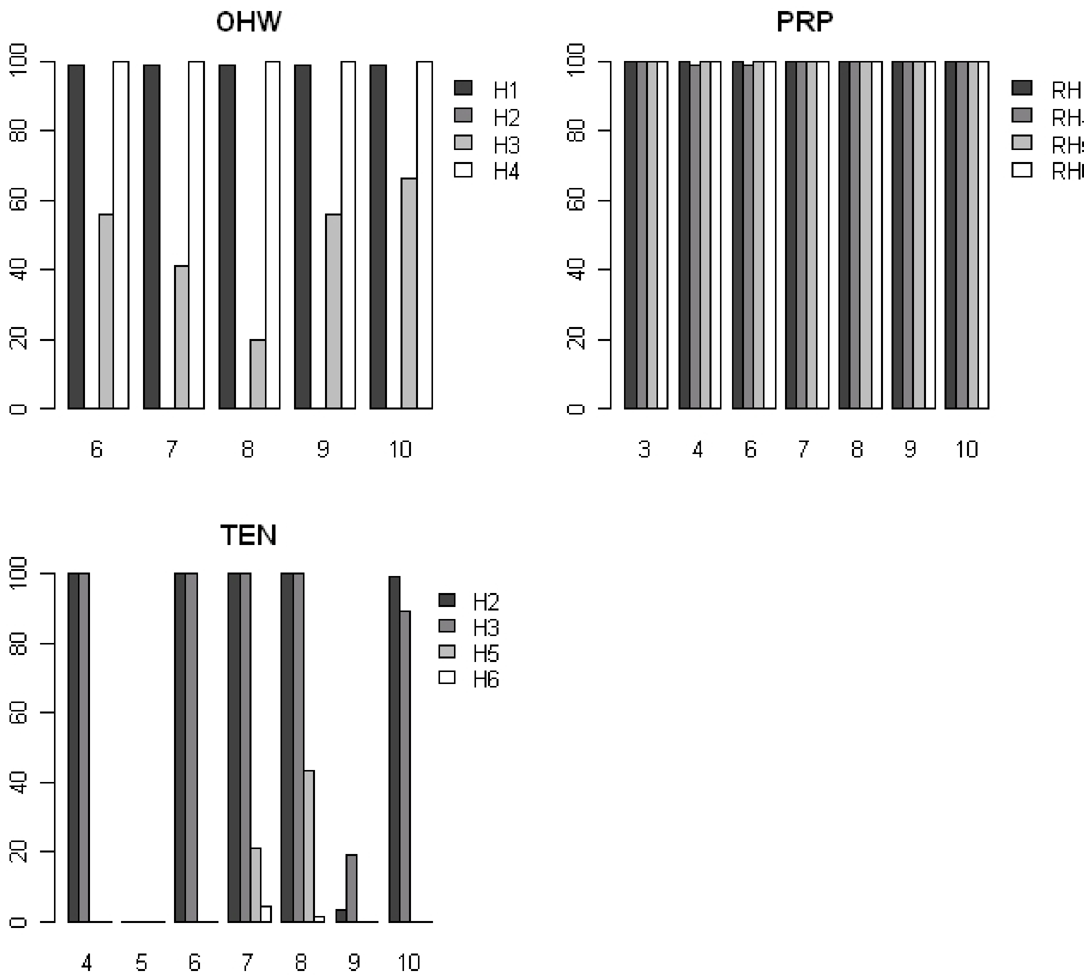
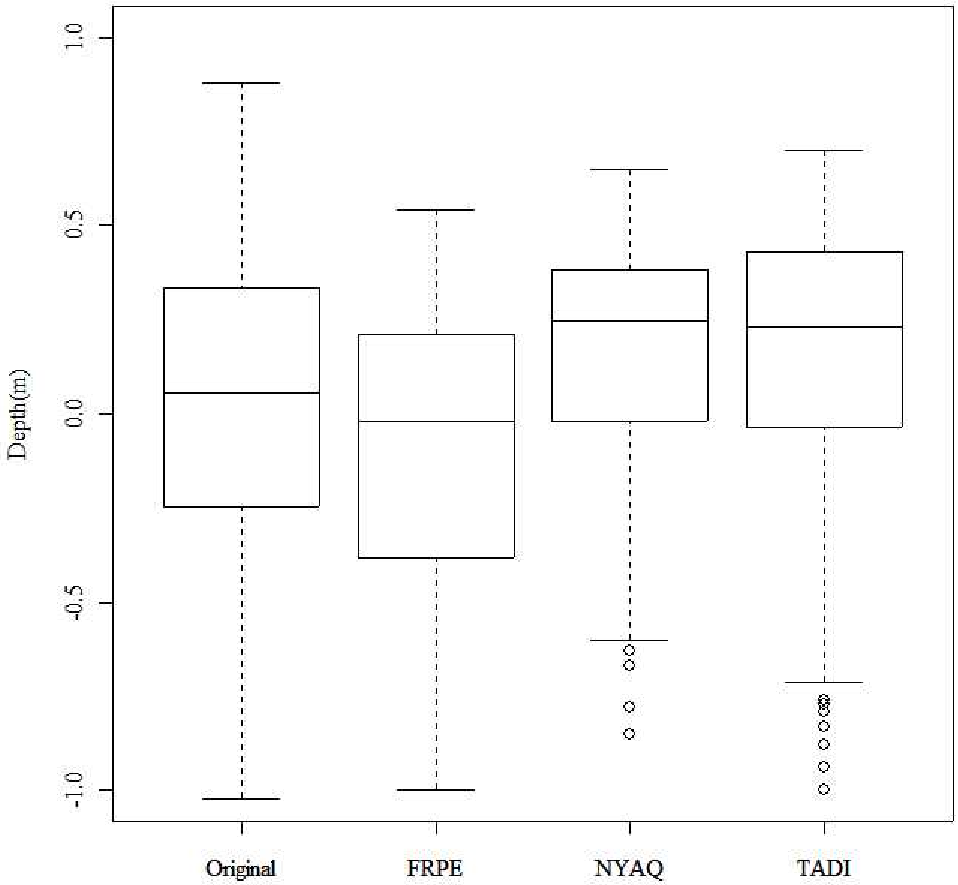


Figure 206. Percentage of Plot Inundated at Time of Monthly Sampling for All Sampled Months in 1995 on Hydric Swamp Plots.

Tree Survival and Hydrology in Cypress-Gum Plots

A box plot (Figure 207) showing the average depth of water for the planted trees by species is shown in comparison with a box (first from left) showing the average water depth for the plots. As all species were initially planted along the entire water level gradient in a plot, this box represents the distribution of water depths at all original planting locations. A comparison of this first box of all planting locations with plots of surviving individuals of each species shows where trees survived along the water level gradient. The range of surviving *Fraxinus pennsylvanica* extends from a water depth of 0.5 to -1.0 m, excluding the deeper portion of the original range. The population of surviving *Nyssa aquatica* and *Taxodium distichum* withstood more inundation than the population of surviving *Fraxinus pennsylvanica*. Only a few outliers of the two populations occur where the average water level was below -0.6 m. *Taxodium distichum*, which had the highest survival, occurs along a broader continuum of water depths than *Nyssa aquatica*. No individuals of any of the three species survived in the deepest part of the originally planted range.



Note: The distributions are presented as box plots that break the data into four quartiles. The middle box represents the 25-75th percentiles, which includes the median value represented by the middle line. The upper and lower hashes represent the 0 and 100 percentiles. The circles beyond the lower hash are outliers.

Figure 207. Distribution of Average Water Depth Inside Original Plot Boundaries of Cypress-Gum Transects, and at the Locations of Surviving Trees for Each of the Species Planted (*Fraxinus pennsylvanica, Nyssa aquatica,* and *Taxodium distichum*).

In Figures 208-210, tree survival after 1 and 20 years is compared by species for all cypress-gum plots within the same site. For instance, in the bottom chart in Figure 48, all the *Taxodium distichum* surviving after years 1 and 20 on the six adjacent transects of CFI are classified by 2005 average water depth either at the tree base, or the former location of the tree for those that died between years 1 and 20. At CFI, all three species appear to be tolerant of the range in which they survived after year 1. In other words, water depth did not preclude 20-year survival on this site. However, water level may have had an effect on likelihood of survival. More *Nyssa aquatica* trees died than lived in the shallower water depths. Once established, *Taxodium distichum* and *Fraxinus* *pennsylvanica* appear to be capable of tolerating the entire water level range. At OHW (Figure 209), *Fraxinus pennsylvanica* appears to have a much more limited water tolerance range, as only trees with an average water depth of 0.2-0.3 m survived. *Fraxinus pennsylvanica* did not tolerate the wetter locations of this transect, though it appears to have tolerated the same average depth at CFI. Only a few *Nyssa aquatica* survived and they appear to have tolerated depths between 0.2 and 0.4 m, as *Taxodium distichum* appears to have tolerated those depths as well as 0.0-0.2 m. At TEN (Figure 210), *Fraxinus pennsylvanica* tolerated the drier locations where it established, but not in locations with average water levels above the ground surface (0.0 m). *Taxodium distichum* survived where water levels were higher than -0.3 meters. *Nyssa aquatica* survival was poor across the range.

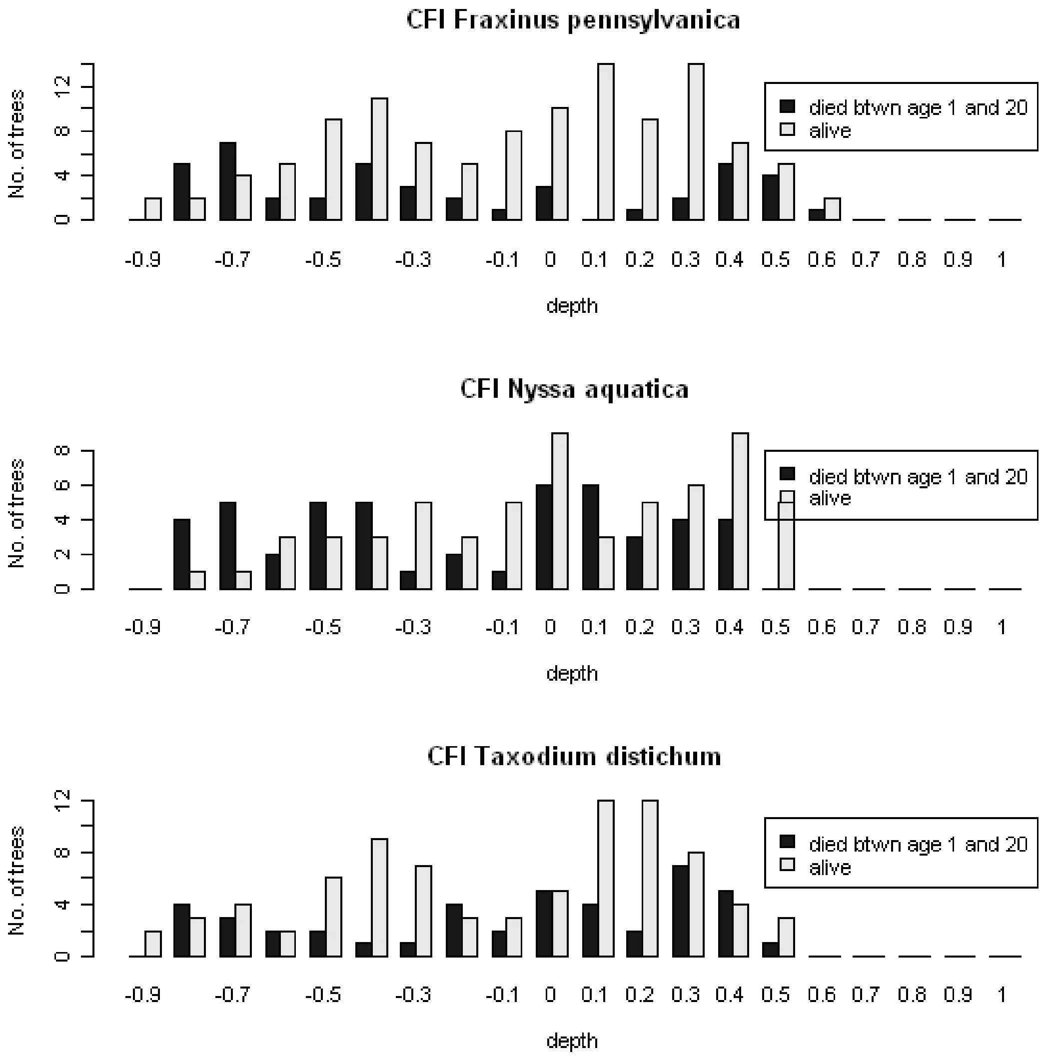
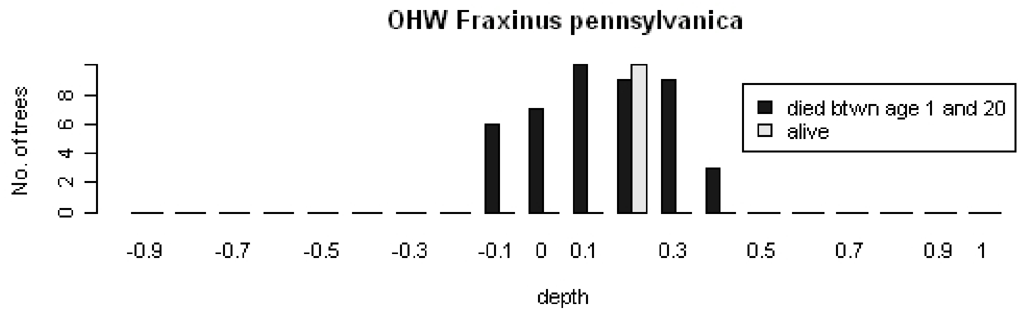


Figure 208. Number of Planted Trees that Died Between Years 1 and 20 and Trees Alive in 2005, in 0.1 m Depth Classes on CFI (Sand-Clay) on Plots R1-R6.



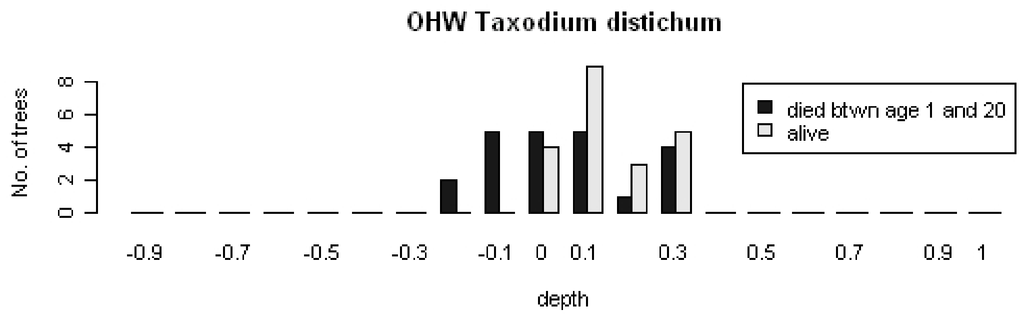


Figure 209. Number of Planted Trees that Died Between Years 1 and 20 and Trees Still Alive, in 0.1 m Depth Classes on OHW (Clay) Plots R2A and R2B.

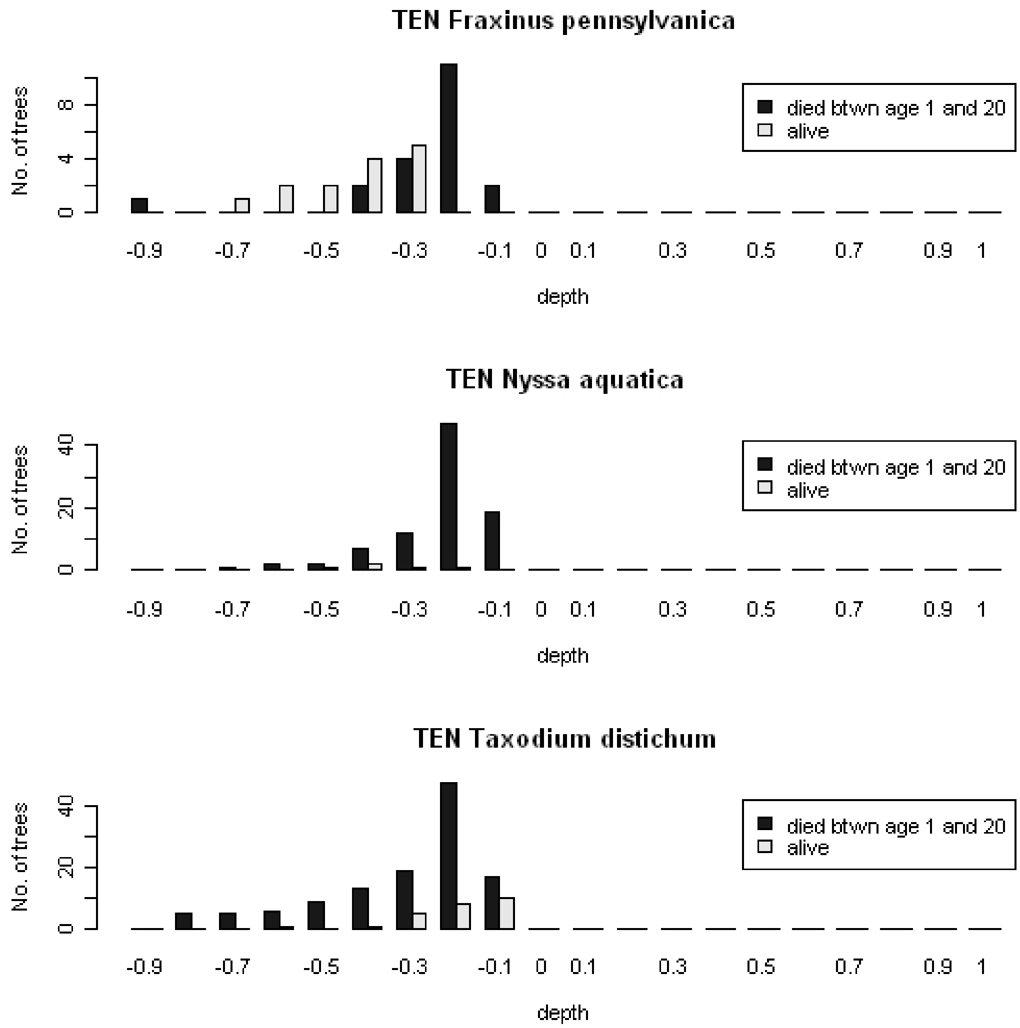
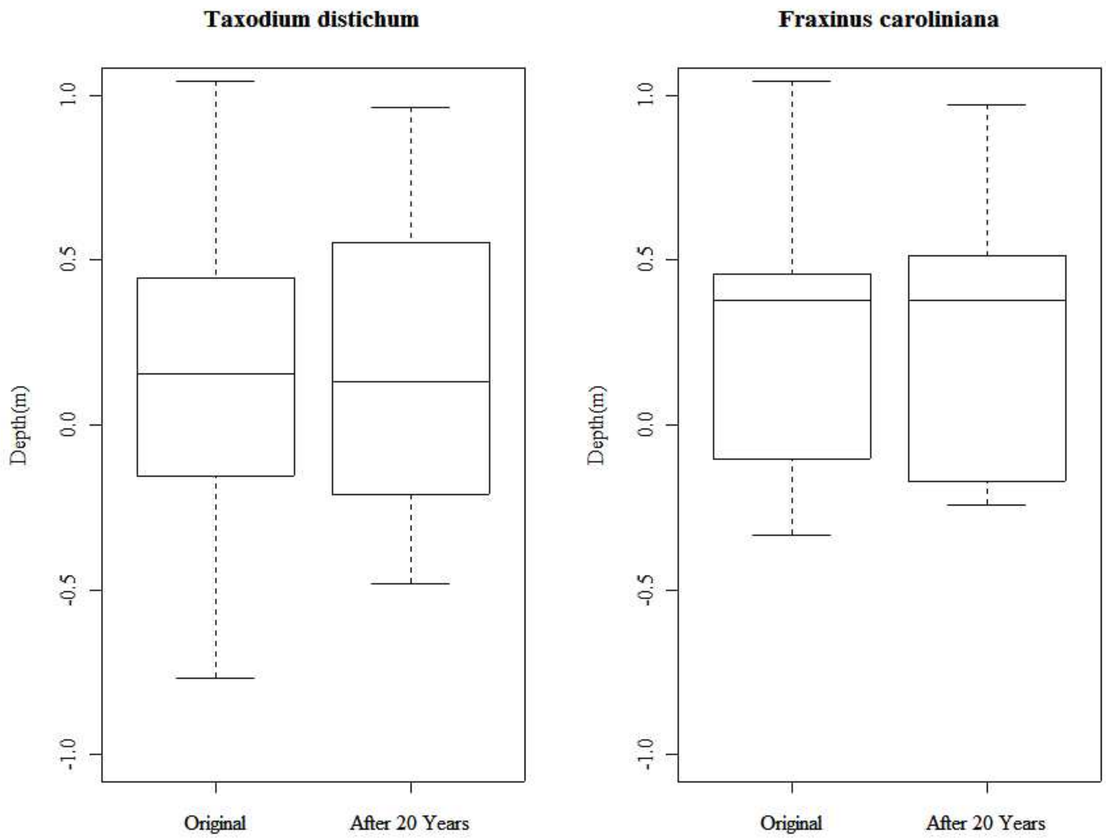


Figure 210. Number of Planted Trees that Died Between Years 1 and 20 and Trees Still Alive, in 0.1 m Depth Classes on TEN (Clay) Plots R5A, R5B, R6A, R6B, R7A, and R7B.

Tree Survival and Hydrology in Hydric Swamp Plots

Figure 211 shows distributions of *Taxodium distichum* in hydric swamp plots where average depths at the surviving trees ranged from -0.5 to 0.9 meters. This species was not found in drier locations from -0.75 to -0.5 m and not in the wettest locations where average depth was >0.9 m. The range of original planting locations of *Fraxinus caroliniana* was similar to that for *Taxodium distichum* but not drier than -0.3 m because it was not planted in the drier plots. Surviving individuals were not found where average depths were < -0.2 or >0.9 m.

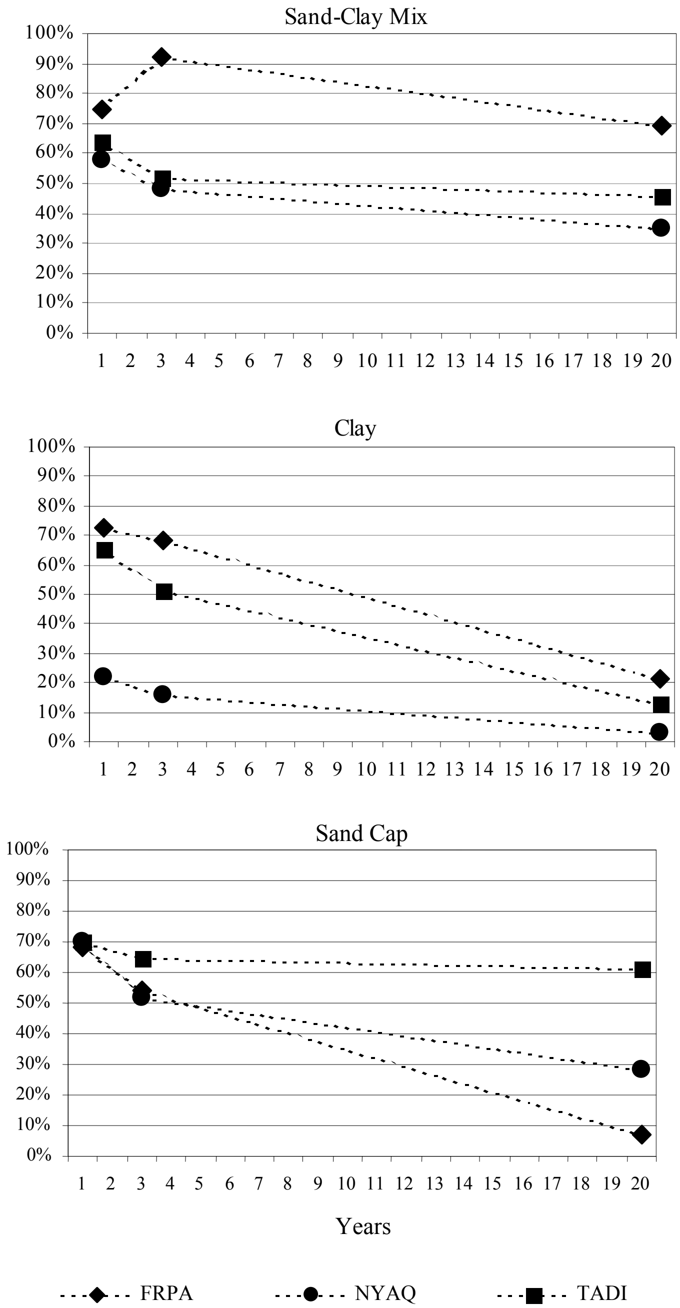


Note: See Figure 47 for explanation of box plot construction.

Figure 211. Distribution of Average Water Depth inside Plots Boundaries (Original) and at Locations of Surviving Trees.

Tree Survival and Soil Type

Figure 212 summarizes tree survival on the sand-clay (CFI), sand-capped (HOM), and 3 clay sites (OHW, PRP, TEN). Trees growing on sites with clay soils had the lowest survival after 20 years. CFI, the sand-clay site, had the best overall survival. Though *Nyssa aquatica* survived poorly on the clay sites after the first year, the survival rate between years 3 and 20 on clay was better than on the sand-cap site (HOM) and similar to that on the sand-clay site (CFI). *Taxodium distichum* average survival rate between years 3 and 20 was poorest on the clay sites at about 97% yr^{-1}, and high on both the sand-cap and sand-clay sites, at >99% $year^{-1}$. The population of *Fraxinus pennsylvanica* declined about 50% on the clay and sand-cap sites between years 3 and 20. Due in part to resprouting, almost as many *Fraxinus pennsylvanica* trees were alive at CFI after 20 years as there were after 1 year, where a very high percentage (70%) survived.



Note: The middle graph presents average survival at the 3 clay sites; the other soil types are represented by 1 site each.

Figure 212. Percentage of Planted Trees Surviving in Cypress-Gum Plots by Soil Type after Approximately 1 (Rushton 1988), 3 (Rushton and Paulic 2001), and 20 Years.

Tree Growth Comparison between Sand-Clay and Clay Sites

The population of trees planted on the sand-capped site was exposed to higher water levels than tree populations on clay and sand-clay sites. Comparison of the effects of soil medium on tree growth could not include sand-capped sites because there was no control for the effect of water level on tree growth. But assuming that similar hydrologic regimes can be inferred from similar average water depth at the tree base during the 2005 season, tree populations in clay and sand-clay may be compared to examine the effects of soil medium on tree growth. In 2005, all surviving trees on clay occurred within the range of water depths to which trees growing in sand-clay were exposed (see Figure 213).

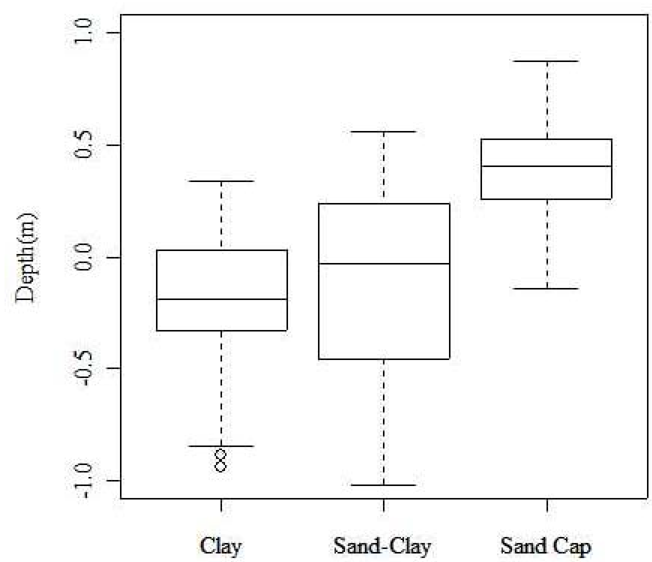


Figure 213. Distribution of Average Water Depth in Cypress-Gum Plots Grouped by Soil Type.

The basal areas of all trees surviving in clay were compared, by species, to basal areas of trees surviving on sand-clay in the same water level range (-0.75 to 0.25 m). Trees with a basal area < 7.8 cm^2 were assumed to be re-sprouts, and they were eliminated from the growth comparison. The basal area of the remaining trees was then log-transformed for normality. The results of T-tests to determine if a significant difference existed between the growth of trees on clay and sand-clay are presented in Table 79.

Table 79. Comparison of Trees Growing in Different Soil Media by Species, Among Those Occurring in Similar Water Depth in 2005.

Species	No. of Trees		Mean of Log (Basal Area)		P-Value
	Clay	Sand-Clay	Clay	Sand-Clay	
Fraxinus pennsylvanica	45	89	4.24	4.44	0.20
Nyssa aquatica	13	37	4.61	3.96	0.02*
Taxodium distichum	82	67	5.23	5.44	0.63

*Significantly different at the 95% confidence level.

Taxodium distichum trees from both cypress-gum plots and hydric swamp plots were considered in the analysis. Growth of *Fraxinus pennsylvanica* and *Taxodium distichum* on clay and sand-clay was not statistically different. Growth of *Nyssa aquatica* was better (at a 95% confidence level) on clay, however there were only 13 *Nyssa aquatica* trees surviving on clay, a very small percentage of those originally planted.

A two-way ANOVA was conducted (Tables 80 and 81) for species of trees planted on the sand-clay mix versus clay in order to simultaneously compare the effect of water level and soil type on tree DBH and determine if there were any interactions between the two variables.

Table 80. Results of a Two-Way ANOVA Comparing the Effect of Two Soil Types (Clay and Sand-Clay) and Two Water Levels (Shallow and Deep) on *Fraxinus pennsylvanica* DBH.

Variable	P-Value
Soil Type	0.39
Water Level	0.52
Interaction	0.16

Table 81. Results of a Two-Way ANOVA Comparing the Effect of Two Soil Types (Clay and Sand-Clay) and Two Water Levels (Shallow and Deep) on *Taxodium distichum* DBH.

Variable	P-Value
Soil Type	0.40
Water Level	0.02*
Interaction	0.01*

*Significantly different at the 95% confidence level.

Trees on the sand-cap site (HOM) were eliminated from consideration because of higher water levels. Prior to the test, the basal area of all trees was natural log-transformed for normality. For comparison of water levels, trees were split into 'shallow' and 'deep' classes, depending on whether the depth of water at the tree was higher or lower than the median water level for trees growing in both soil types. For *Fraxinus pennsylvanica*, trees with an 2005 average water table level of less than -0.25 m were grouped as 'shallow' and those with a water level greater than -0.25 m were grouped as 'deep.' *Fraxinus pennsylvanica* did not show a significant difference in DBH in different soil types or water levels and the test gave no evidence of an interaction effect. *Taxodium distichum* trees were split into 'shallow' and 'deep' classes using the average water level of 0.0 m. This test showed a significant effect for water level and for the interaction of water level and soil type. Trees in deep water had an average basal area of 5.4 cm^2, 0.4 cm^2 greater than trees in shallow water, but the variance in basal area was also much higher for deep trees (1.53 to 1.19). Though planted on both soils, survival of

Nyssa aquatica in clay was too low to allow for a comparison of the effects of soil type and water level on growth for this species.

Initial Tree Growth and 20-Year Tree Survival

Records of tree height on cypress gum plots after one year were paired with tree survival records within the same plot to determine if trees that grew faster during the first year were more likely to survive 20 years. Tree height records after one year were available for 6 plots on CFI, 2 plots on OHW, and 6 plots on TEN. Of the trees with a height record, 296 were surviving in 2005 and 408 were dead. A T-test was performed to determine if the heights of the trees after one year were different for these two groups, after the height was square root-transformed to satisfy the condition of similar between-group variance. The outcome, a p-value of 2.2e-16, indicated with a very high level of confidence that, among all the trees planted, those taller after one year were more likely to survive in the long-term.

Among the six plots on TEN, the average height of planted trees after one year was 35 cm, in comparison with 95 cm at CFI. Twenty-year survival of the TEN trees was 17%, versus 54% at CFI. Among these plots there is a strong correspondence between tree height after 1-year and 20-year survival.

Site Disturbance and Tree Survival

On a number of sites, disturbance factors directly caused mortality or damage to the planted trees within the initial year of establishment or in years since. Where records of these disturbances exist, they are presented in Table 82.

Table 82. Site Disturbance Record.

Site	Plot(s)	Disturbance		
		Fire	Heavy Grazing	Mechanical
CFI	1,3,4,5,6	-	-	-
	2	-	-	+
HOM	1,2,3,4	-	-	-
	5,6,7,8	-	+	-
	1A, 2A, 2B	-	-	-
OHW	H1, H4	-	-	-
	H2, H3	+	-	-
PRP	H1, H2, H3, H4	+	-	-
	5A, 5B	-	+	+
	6A, 6B, 7A, 7B	-	+	-
TEN	H5	-	+	-
	H2, H3, H6	-	-	-

+ Record of incidence, - No record of incidence.

Fire, heavy grazing, and mechanical disturbance (tractors, etc.) are three recorded disturbances known to have influenced a number of plots. A fire occurred in two hydric swamp plots (as well as in a number of cypress-gum plots not monitored in this study) that lie within a gully between two spoil piles on OHW within the first few years after planting (Rushton 2005). This or perhaps multiple events likely caused heavy mortality in these plots. Multiple fires burned into all four of the hydric swamp plots in PRP, where dead trunks of trees blackened from burning still stand as evidence. On HOM, four transects were subjected to grazing by cattle during their initial years of the experiment (Rushton 1988). In one basin of TEN, heavy herbivory negatively affected tree growth and survival during the first year (Rushton 1988). Segments of a few transects were damaged by earth-moving equipment, including the first 8 meters of CFI R2 and the first few meters of both TEN 5A and 5B. Numerous other disturbances may have occurred without leaving any direct or anecdotal evidence, including prolonged flood events, drought or heavy winds.

Recruited Trees

In a few cases, seedlings and mature trees of the same species as planted trees ('recruited trees') were found in abundance inside seedling sample plots, whereas in some plots no recruited trees were found. These trees are potentially offspring of the trees planted by Rushton, but in numerous cases, the trees planted by Rushton were not the only plausible source of recruited trees. Tree populations in plots are presented in Table 83, where they are ranked by the ratio of the number of surviving planted trees to the number of recruited trees (reproductive ratio).

Populations are defined in this table as all trees of a given species within the seedling sampling area of a plot. Only populations with at least one surviving tree and one planted tree are listed; 30 populations met this criterion. Where another plausible source for the recruited trees exists, this source is mentioned in the table. In nine populations, the number of recruited trees was greater than or equal to the number of planted trees. In two of these populations, the number of recruited trees was approximately 100 times greater than the number of planted trees. But in both of these two populations, there are clear seed sources other than the planted trees. Additional plantings of *Taxodium distichum* adjacent to or within sampling areas since 1985 occurred at CFI and HOM, but locations of those plantings were not available and thus trees not planted by Rushton could have either been planted later or are offspring of trees from another planting.

Table 83. Plots with Potential Offspring of Planted Trees Ordered by Reproductive Ratio.

Site	Plot	Species	# Planted Trees	# Recruited Trees	Reproductive Ratio (Planted/Recruited)	Possible Alternate Source for Non-Planted?	Alternate Source	Rank
TEN	H3	TADI	1	133	0.01	Y	Other plots	1
OHW	H2	ACRU	3	223	0.01	Y	Floodplain	2
CFI	R4	TADI	4	9	0.44	Y	Other planting	3
OHW	H1	FRCA	4	8	0.50	N		4
PRP	RH6	TADI	2	4	0.50	N		5
CFI	R5	TADI	10	16	0.63	Y	Other planting	6
OHW	H1	TADI	2	3	0.67	N		7
CFI	R2	TADI	12	13	0.92	Y	Other planting	8
OHW	R2B	FRPE	1	1	1.00	N		9
OHW	H4	FRCA	16	15	1.07	N		10
OHW	R1A	TADI	4	3	1.33	N		11
CFI	R3	NYAQ	8	6	1.33	N		12
CFI	R1	TADI	15	11	1.36	Y	Other planting	13
TEN	R6A	TADI	2	1	2.00	N		14
CFI	R3	TADI	24	9	2.67	Y	Other planting	15
HOM	R1	TADI	25	8	3.13	Y	Other planting	16
TEN	H6	TADI	17	4	4.25	N		17
CFI	R4	FRPE	14	3	4.67	N		18
OHW	H4	TADI	5	1	5.00	N		19
CFI	R5	FRPE	15	3	5.00	N		20
PRP	RH1	TADI	10	2	5.00	N		21
OHW	R2B	TADI	8	1	8.00	N		22
TEN	H5	TADI	9	1	9.00	N		23
TEN	H6	FRCA	19	2	9.50	N		24
CFI	R6	TADI	20	2	10.00	Y	Other planting	25
PRP	RH5	TADI	12	1	12.00	N		26
CFI	R3	FRPE	25	2	12.50	N		27
CFI	R1	NYAQ	15	1	15.00	N		28
HOM	R3	TADI	15	1	15.00	Y	Other planting	29
PRP	RH5	FRCA	20	1	20.00	N		30

New Seedling Survival

The *Taxodium distichum* seedling (0-100 cm in height) population at TEN H3 was the largest of any plot sampled in June, with 128 individuals. By November, the population had been reduced to 52 individuals. As location of the seedlings was noted only to the nearest meter and seedlings were not tagged, it was not possible to track individual seedling growth with certainty. However, size class distributions of the seedling populations during both periods reveal in which segments of the population that mortality occurred (Figure 214). A comparative look at the two distributions reveals a close match between trees in classes > 20 cm, but a discrepancy between the number of trees in the first two classes in the two distributions, which likely represent seedlings that germinated in the spring of 2005. In June there were a total of 87 trees in the first two classes, whereas there were only 10 in November. The size of Class 3 in November indicated that only a few of these trees likely grew into a larger size class during this period. The water level record reveals that the water was between -0.5 and 0.0 m in May at the locations where the 87 individuals of less than 20 cm stood in June. Of those seedlings, 72 were completely inundated in water during the June and July sampling.

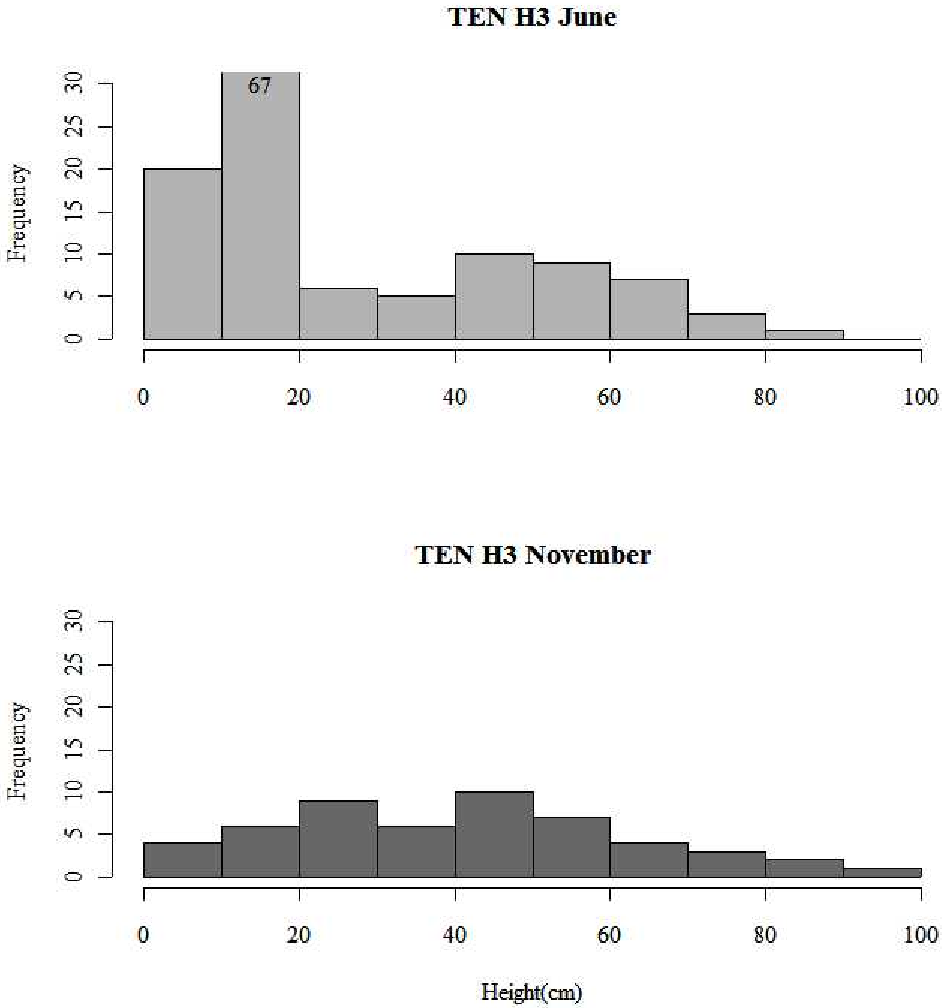
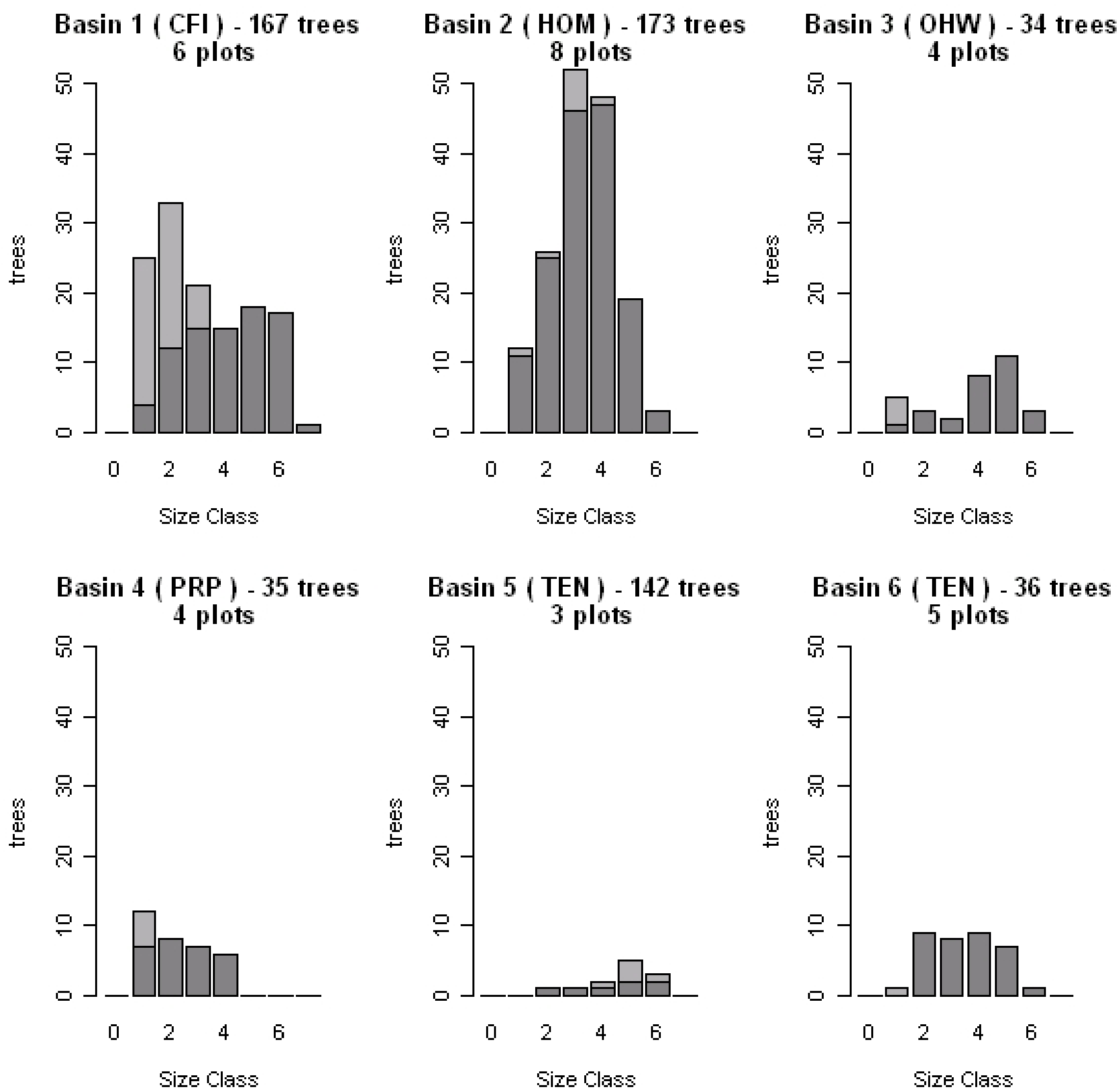


Figure 214. Size Class Distributions of *Taxodium distichum* Seedlings at Ten H3 Counted in June and November, 2005.

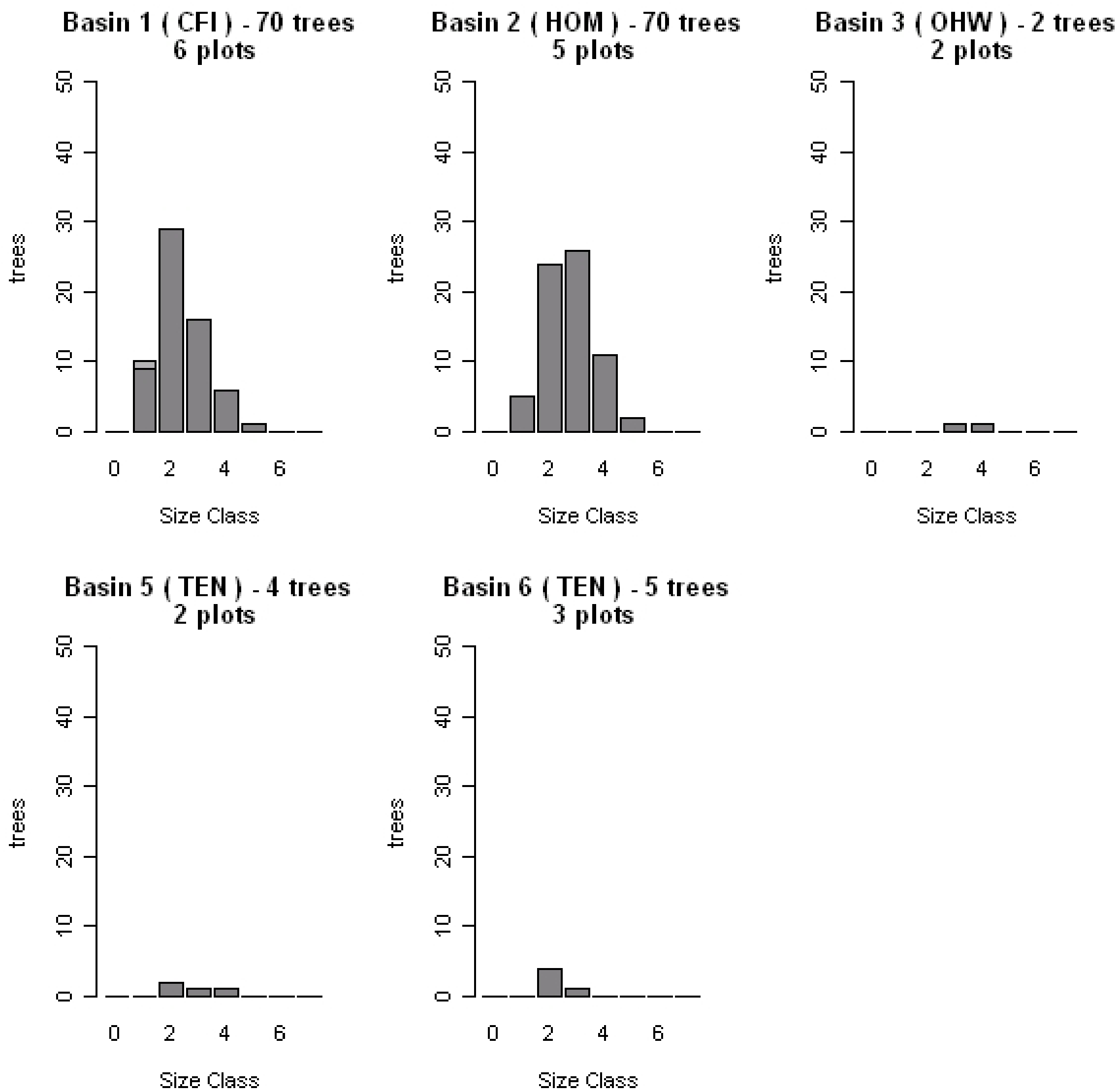
Tree Population Size Class Distributions

Figures 215-218 show size class distributions of *Taxodium distichum*, *Nyssa aquatica*, *Fraxinus pennsylvanica*, and *Fraxinus caroliniana*. The composition of each size class is split into planted and recruited trees. Populations of *Taxodium distichum* are shown in six basins in Figure 55. Trees at CFI are the most evenly distributed across size classes. Recruited trees at CFI appear in the first four size classes. At HOM there is a more normal-shaped distribution, with obvious omissions in the seedling class (Class 0). At OHW, PRP, and TEN there are fewer trees, in part because some of the plots were hydric swamp plots, where fewer trees of a species were planted, and in part because of lower survival. The first basin at TEN had an exceptionally high number of seedlings (see Table 31, row 1). Four trees in Classes 4, 5, and 6 in this basin appear as 'recruits' but are actually trees planted by Rushton in a plot not included in this study that overlapped with the recruited tree sampling area.



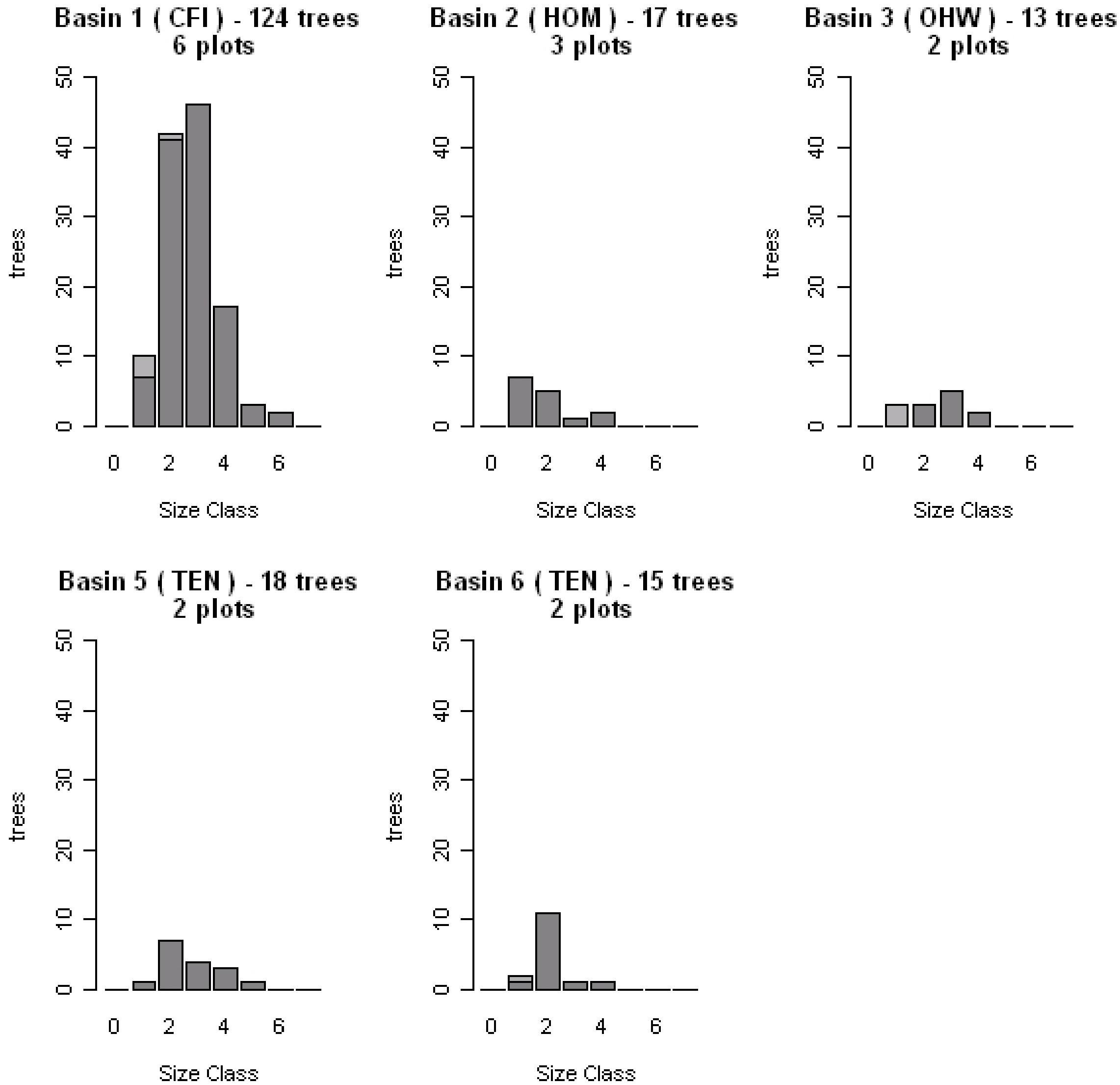
Note: Light sections represent recruited trees; dark sections planted trees. The size classes represent the following dbh ranges: 0: no dbh; 1: 0-5cm; 2: 5-10cm; 3: 10-15cm; 4: 15-20cm; 5: 20-30cm; 6: 30-40cm; 7: >40cm.

Figure 215. Size Class Distributions of *Taxodium distichum* in Six Basins on Five CSAs.



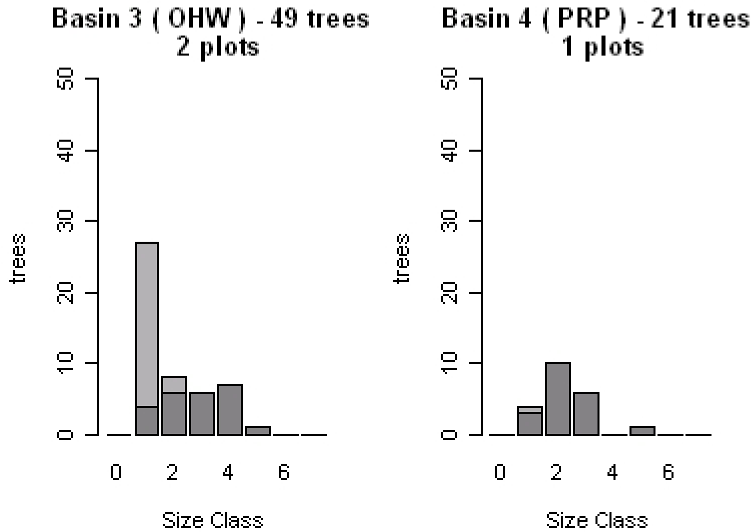
Note: Light sections represent recruited trees; dark sections planted trees. The size classes represent the following dbh ranges: 0: no dbh; 1: 0-5cm; 2: 5-10cm; 3: 10-15cm; 4: 15-20cm; 5: 20-30cm; 6: 30-40cm; 7: >40cm.

Figure 216. Size Class Distributions of *Nyssa aquatica* in Five Basins on Four CSAs.



Note: Light sections represent non-planted trees; dark sections planted trees. The size classes represent the following dbh ranges: 0: no dbh; 1: 0-5cm; 2: 5-10cm; 3: 10-15cm; 4: 15-20cm; 5: 20-30cm; 6: 30-40cm; 7: >40cm.

Figure 217. Size Class Distributions of *Fraxinus pennsylvanica* in Five Basins on Four CSAs.



Note: The size classes represent the following dbh ranges: 0: no dbh; 1: 0-5cm; 2: 5-10cm; 3: 10-15cm; 4: 15-20cm; 5: 20-30cm; 6: 30-40cm; 7: >40cm.

Figure 218. Size Class Distribution of *Fraxinus caroliniana* in Two Basins on Two CSAs.

Nyssa aquatica populations were too small in basins at OHW and TEN such that trees were only distributed between 2-3 middle range size classes (Figure 216). CFI has a small number of seedlings but the approximately the same number relative to other size classes in comparison with its *Taxodium distichum* population.

The CFI basin had six times as many surviving *Fraxinus pennsylvanica* as the other basins and a normal shaped population distribution (Figure 217), but the distributions of the populations are similar in the other basins, albeit they were lacking in smaller trees. Only a small number of *Fraxinus caroliniana* were planted in two basins and in both cases there are more individuals than originally planted (Figure 218).

Matrix Population Model

The model for *Taxodium distichum* at CFI used the records of 266 trees to construct the transition matrix (Figure 219). The lambda (λ) of this transition matrix was 1.005; the model predicts that at steady state the population will increase but at a slow pace. The population projection for the next 50 years shows at first a slowing decline from 150 to a low of about 120 trees after 20 years, but then growing again to 130 at the end of 50 years (Figure 220). The model for the *Taxodium distichum* population on the OHW basin used records of 106 trees for construction of the transition matrix (Figure 221), with no trees presently in the largest size class. The λ of this transition matrix was 0.991, indicating a slow long-term population decline. The model predicted that after 50 years, the tree population would fall from 36 to 16 trees in the basin (Figure 222). Though the λ values represent potential opposite long-term trajectories (the former ‘growing’ and the latter ‘declining’) for the two populations, the model does not predict drastic population change for either basin within the next 50 years.

CLASS	0	1	2	3	4	5	6	7
0[	0.699	0.012	0.013	0.015	0.023	0.034	0.051	0.076]
1[	0.173	0.751	0.	0.	0.	0.	0.	0.]
2[	0.	0.212	0.804	0.	0.	0.	0.	0.]
3[	0.	0.	0.139	0.731	0.	0.	0.	0.]
4[	0.	0.	0.	0.223	0.725	0.	0.	0.]
5[	0.	0.	0.	0.	0.244	0.902	0.	0.]
6[	0.	0.	0.	0.	0.	0.078	0.946	0.]
7[	0.	0.	0.	0.	0.	0.	0.04	0.991]

Figure 219. Transition Matrix for CFI SP-1 *Taxodium distichum* Population Model.

Relative to the mature tree population size, the larger number of new seedlings at CFI compared to OHW resulted in slightly higher fecundity values, or the probability of creating a successful offspring. These values are depicted in the first row of the transition matrices.

The stasis values, or the probability of remaining in the same size class over the year, are presented along the diagonal. These values are similar for the two sites. Predicted growth values (the value below the diagonal) were also similar at both sites. Because no trees were present in the largest size class at OHW, there was no probability of advancement into the largest size class at OHW, which does not represent a realistic scenario.

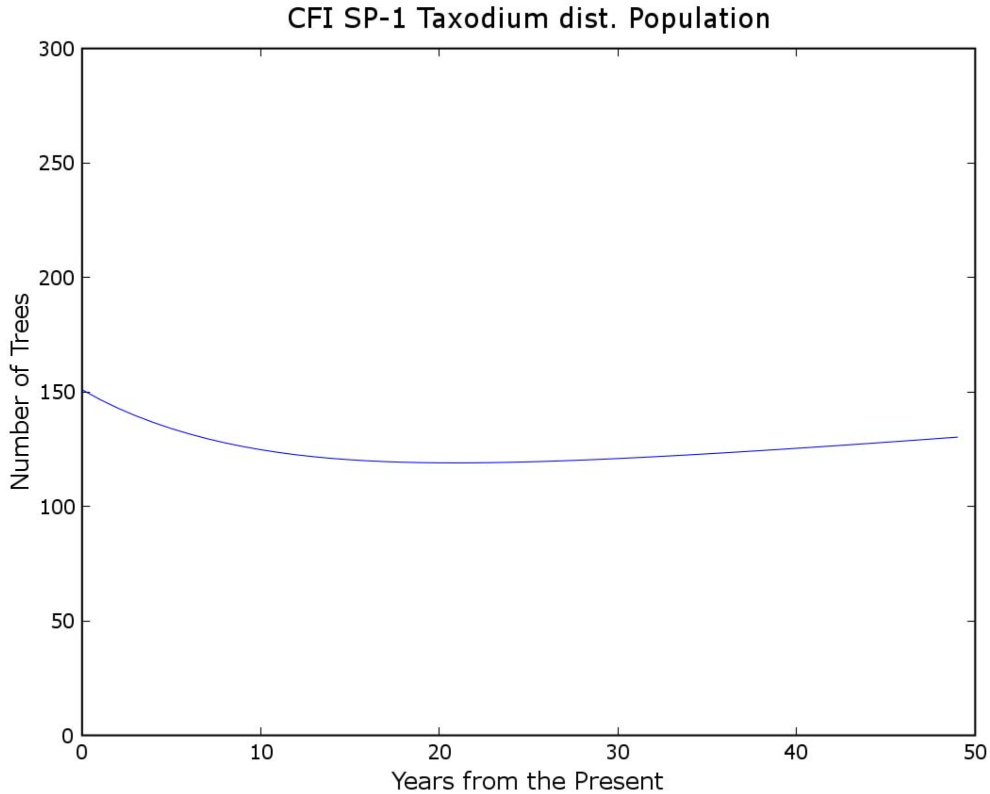


Figure 220. Model-Predicted Population Change of CFI SP-1 *Taxodium distichum.*

CLASS	0	1	2	3	4	5	6	7
0[	0.67	0.005	0.006	0.004	0.006	0.01	0.015	0.022]
1[	0.247	0.775	0.	0.	0.	0.	0.	0.]
2[	0.	0.194	0.799	0.	0.	0.	0.	0.]
3[	0.	0.	0.109	0.732	0.	0.	0.	0.]
4[	0.	0.	0.	0.188	0.78	0.	0.	0.]
5[	0.	0.	0.	0.	0.189	0.938	0.	0.]
6[	0.	0.	0.	0.	0.	0.042	0.986	0.]
7[	0.	0.	0.	0.	0.	0.	0.	0.991]

Figure 221. Transition Matrix for OH Wright *Taxodium distichum* Population Model.

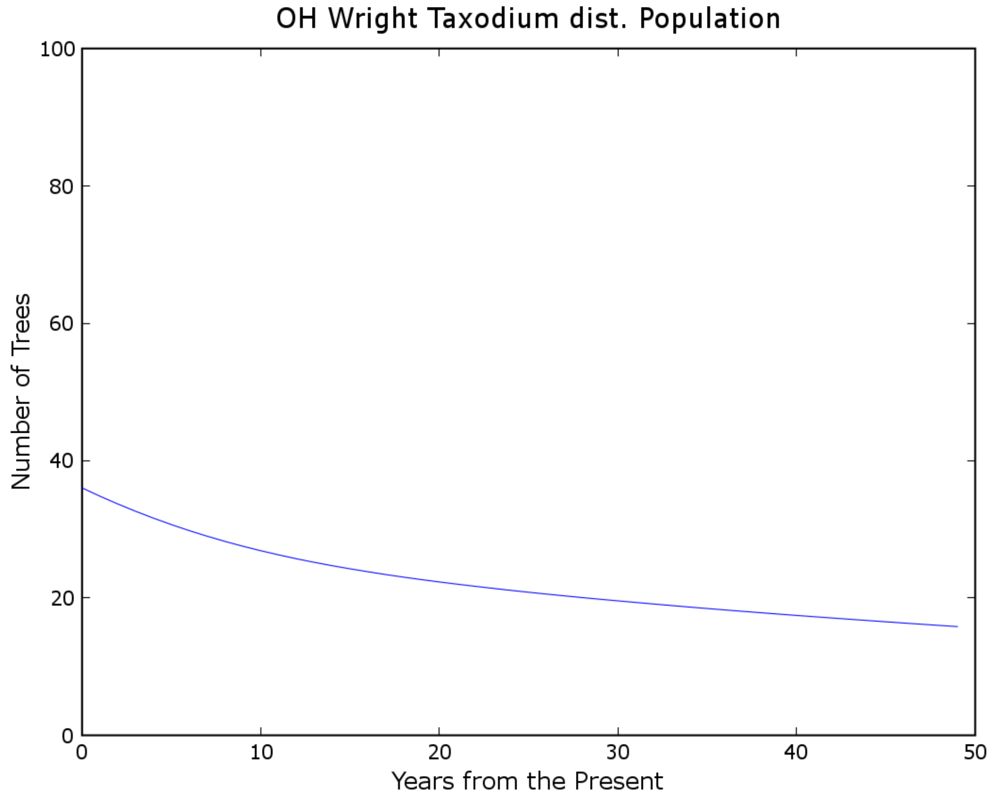
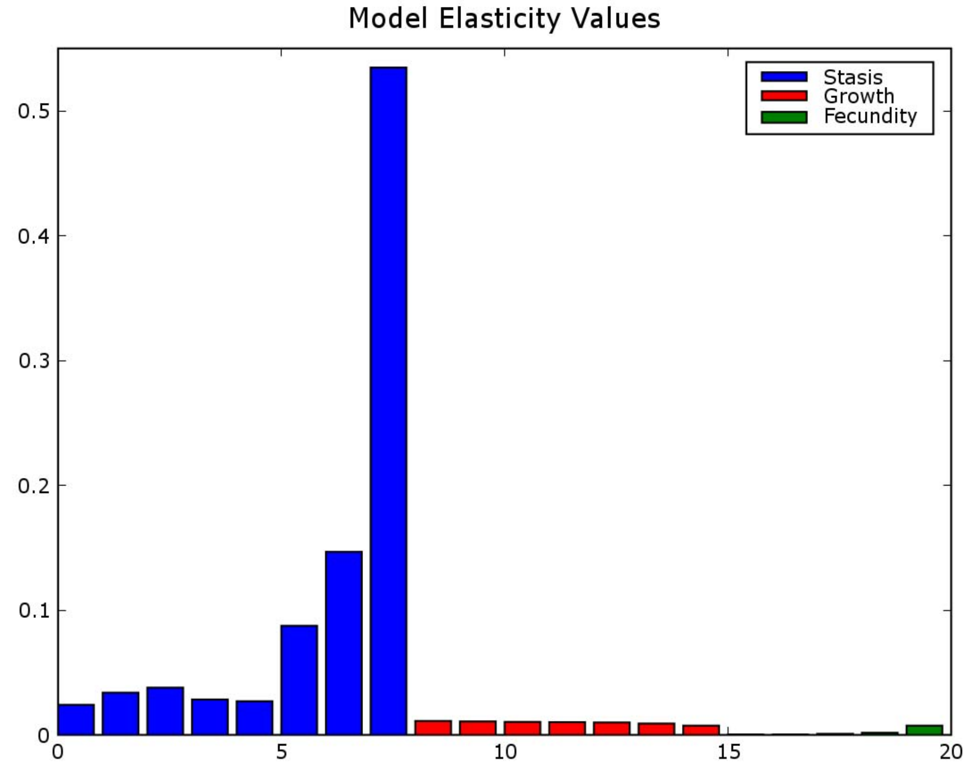


Figure 222. Model-Predicted Population Change of OH Wright *Taxodium distichum*.



Note: For each parameter type (stasis, growth, fecundity) the first bar from the left represents size class 0 with the bars to the right corresponding to size class 1,2,3… to 7.

Figure 223. Model Elasticity Values Showing Sensitivity of Different Parameters.

Figure 223 shows the results of the elasticity analysis of the CFI model. The elasticity analysis was nearly identical for the OHW model. This analysis shows the chief importance of the stasis values for the largest three size classes. Though there are different growth rates for the two populations, the stasis values for the last size class were 0.99 for both models, suggesting that 99 of 100 trees in the largest size class are likely to survive a given year. This value was, according to the sensitivity analysis, nearly five times as important as any other value in the transition matrix.

Ecosystem Development in Rushton and Reference Plots

Comparisons between pairs of a Rushton plot(s) and a reference plot were made based on the canopy cover, plot vegetation including trees, shrubs, and understory vegetation, and percent soil organic matter. Samples from Rushton plots were only considered when basal area density of Rushton trees was > 10 m^2/ha in the sample area.

Selection of Plots for Comparison

Table 84 presents all the Rushton plots and subplots ordered by basal area of Rushton trees. The plots/subplots considered in the comparative analysis with reference plots are those listed above the dotted line. A distinction was drawn at a basal area of 10 m^2/ha below which survival in plots was so poor as to potentially nullify the effect of planted species on the surrounding environment. This distinction was drawn based on an arbitrary but clear break in the basal area in plots/subplots between the plot with a basal area of approximately 13 m^2/ha and the next lowest with a basal area of approximately 8 m^2/ha. Five hydric swamp plots and one complete cypress-gum plot along with portions of five others were thus removed from consideration in the following comparative analysis.

In addition to the Rushton plots removed from consideration, one subplot of the reference plot at CFI SP-1 was removed from consideration upon the realization that this segment had been subjected to repeated disturbance from mowing and would not be representative of reference conditions.

Topographic Comparison of Rushton and Reference Plots

Table 85 shows a comparison of topography and water levels in Rushton plots and their corresponding reference plots, which are the highlighted items appearing at the bottom of the groups of Rushton plots. In most cases all reference plot variables including average change in elevation, average water depth, and minimum and maximum water depth fell within three standard errors of the mean of the variable for the corresponding Rushton plots.

Table 84. Rushton Plots/Subplots Ranked by Planted Tree Basal Area (m^2/ha).

Site	Plot	Subplot	Type	Rushton Tree Basal Area (m^2/hec)
CFI	R1	1	CG	226
HOM	R2	2	CG	158
CFI	R3	3	CG	154
HOM	R1	2	CG	142
CFI	R3	2	CG	140
TEN	H6	NA	HS	128
CFI	R1	3	CG	108
CFI	R2	3	CG	107
CFI	R2	1	CG	105
CFI	R6	2	CG	96
CFI	R6	1	CG	96
HOM	R1	3	CG	92
OHW	R2A	1	CG	90
HOM	R4	2	CG	88
CFI	R2	2	CG	87
CFI	R6	3	CG	87
CFI	R3	1	CG	86
CFI	R5	1	CG	86
HOM	R4	1	CG	82
CFI	R1	2	CG	82
CFI	R5	2	CG	80
CFI	R4	3	CG	79
HOM	R6	1	CG	77
HOM	R3	2	CG	76
OHW	R2B	1	CG	75
CFI	R5	3	CG	65
HOM	R1	1	CG	63
HOM	R5	1	CG	63
HOM	R7	3	CG	61
HOM	R7	1	CG	58
OHW	R2A	2	CG	57
HOM	R6	2	CG	55
HOM	R2	3	CG	50
HOM	R7	2	CG	48
TEN	H5	NA	HS	42
CFI	R4	2	CG	41
HOM	R6	3	CG	36
OHW	R2B	2	CG	35
HOM	R3	3	CG	35
TEN	R2B	1	CG	32
HOM	R5	3	CG	31
HOM	R5	2	CG	26
OHW	H4	NA	HS	24
OHW	H1	NA	HS	24
HOM	R4	3	CG	22
TEN	H2	NA	HS	21
PRP	H5	NA	HS	19
HOM	R2	1	CG	18
TEN	R2B	2	CG	17
TEN	R2B	3	CG	16
HOM	R3	1	CG	15
OHW	R1A	1	CG	14
PRP	H1	NA	HS	14
TEN	R2A	1	CG	13

Table 85. Topography and Water Level[a] Comparison of Rushton and Reference Plots.

Pair	Site	Plot	Plot Type	Avg Δ Elev (m)	Avg Depth (m)	Min Depth (m)	Max Depth (m)	% Inundation
1	CFI	R1	CG	0.04	0.36	0.07	0.63	100
1	CFI	R2	CG	0.03	0.32	-0.07	0.58	90
1	CFI	R3	CG	0.03	-0.05	-0.43	0.34	48
1	CFI	R4	CG	0.05	-0.30	-0.51	0.30	40
1	CFI	R5	CG	0.04	-0.19	-0.77	0.39	42
1	CFI	R6	CG	0.03	-0.36	-0.93	0.11	13
1	CFI	5	CG-Ref	0.05	0.00	-0.49	0.68[b]	55
2	HOM	R1	CG	0.04	0.39	0.13	0.71	100
2	HOM	R2	CG	0.04	0.30	-0.11	0.61	94
2	HOM	R3	CG	0.03	0.32	-0.09	0.51	94
2	HOM	R4	CG	0.03	0.33	-0.07	0.51	94
2	HOM	R5	CG	0.06	0.52	0.09	0.91	100
2	HOM	R6	CG	0.05	0.55	0.15	0.76	100
2	HOM	R7	CG	0.05	0.51	0.16	0.74	100
2	HOM	T1	CG-Ref	0.04	0.42	-0.10	0.63	94
3	OHW	H1	HS	0.01	0.45	-0.01	0.55	99
3	OHW	H4	HS	0.04	0.37	0.19	0.51	100
3	OHW	H1R	HS-Ref	0.02	0.39	0.02	0.50	100
4	OHW	R1A	CG	0.05	0.07	-0.06	0.21	81
4	OHW	T1	CG-Ref	0.03	0.11	0.02	0.25	100
5	OHW	R2A	CG	0.03	0.11	0.02	0.21	100
5	OHW	R2B	CG	0.04	0.20	-0.14	0.35	90
5	OHW	T2	CG-Ref	0.03	0.37[b]	0.10	0.50[b]	100
6	PRP	H1	HS	0.02	0.61	0.55	0.71	100
6	PRP	H5	HS	0.02	0.73	0.37	1.12	100
6	PRP	H1R	HS-Ref	0.04[b]	0.53	0.18	0.85	100
7	TEN	H2	HS	0.01	0.23	0.17	0.27	100
7	TEN	H2R	HS-Ref	0.02	0.25	0.2	0.3	100
8	TEN	H5	HS	0.02	-0.06	-0.21	0.09	14
8	TEN	H5R	HS-Ref	0.01	-0.12	-0.20	-0.07	0
9	TEN	H6	HS	0.01	-0.04	-0.17	0.02	4
9	TEN	H6R	HS-Ref	0.02	-0.03	-0.09	0.15	13
10	TEN	R2A	CG	0.05	0.08	-0.10	0.18	81
10	TEN	R2B	CG	0.03	0.07	-0.41	0.29	68
10	TEN	T1	CG-Ref	0.048	-0.01[c]	-0.59	0.25	62

[a]Water depth data from July for all plots.
[b]More than 3 standard errors from the mean of Rushton plots.
[c]Less than 3 standard errors from the mean of Rushton plots.

Plot Basal Area in Rushton and Reference Plots

Table 86 provides data on plot basal area from Rushton and reference plots. For all but TEN R2A and R2B, the plot basal area (m^2/hec) in reference plots was less than in Rushton plots. The mean plot basal area in Rushton plots was up to 12 times greater than in corresponding reference plots. Typically the difference in plot basal area between

Rushton and reference plots grew as planted species made up a larger portion of the plot basal area in a Rushton plot.

Table 86. Plot-Scale Basal Area Comparison in Rushton and Corresponding Reference Plots.

Pair	Site	Plots		Percent of BA from Rushton Trees		Mean BA (m^2/hec)[a]		Standard Deviation BA (m^2/hec)	
		Rushton	Ref	Rushton	Ref	Rushton	Ref	Rushton	Ref
1	CFI	R1 R2 R3 R4 R5 R6	T5	93	NA	**107**	25	26	NA
2	HOM	R1 R2 R3 R4 R5 R6 R7	T1	97	NA	**64**	6	23	NA
3	OHW	H1 H4	H1R	82	NA	**29**	7	5	NA
4	OHW	R1A	T1	30	NA	**48**	16	NA	NA
5	OHW	R2A R2B	T2	71	NA	**90**	7	13	NA
6	PRP	H1 H5	H1R	62	NA	**27**	11	4	NA
7	TEN	H2	H2R	47	NA	**45**	17	NA	NA
8	TEN	H5	HR	79	NA	**53**	20	NA	NA
9	TEN	H6	H6R	98	NA	**131**	11	NA	NA
10	TEN	R2A R2B	T1	51	NA	33	**47**	13	7

[a]Bolded numbers indicate a difference of more than 1 standard deviation.

In Table 87, the basal area comparison is examined within subplots of cypress-gum plots and their corresponding reference plots. The comparison revealed additional heterogeneity of plot basal area at CFI and HOM at this subplot scale. Using the subplot basal area data enables a comparison of a Rushton and a reference plot with a T-test. Plot basal area was natural log-transformed for normality and a T-test was performed to determine if a significant difference existed between the Rushton and reference plot basal area. Rushton plots had significantly more basal area at the 90% confidence level in three plots. In the one pairing where mean reference basal area was greater, the difference was not significant.

Table 87. Subplot-Scale Basal Area Comparison in Rushton and Corresponding Reference Plots.

Pair	Site	Plots		Samples		Mean BA (m^2/hec)		Standard Deviation BA (m^2/hec)		P-Value from T-Test
		Rushton	Ref	Rushton	Ref	Rushton	Ref	Rushton	Ref	
1	CFI	R1 R2 R3 R4 R5 R6	T5	17	2	107	25	39	7	0.06
2	HOM	R1 R2 R3 R4 R5 R6 R7	T1	21	3	64	6	37	3	0.007
5	OHW	R2A R2B	T2	4	2	90	7	12	7	0.19
10	TEN	R2A R2B	T1	4	2	33	47	10	7	0.24

Percent Canopy Cover

Table 88 compares percent canopy cover determined from canopy photos in Rushton and reference plots. In 7 of 10 pairs, Rushton plots had more canopy cover than corresponding reference plots. In the remaining 3 pairs, reference plots' canopy cover was within 1% of that of the Rushton plots. Except at HOM, there was not a difference between the canopy cover in Rushton and reference plots of more than 10%. Figure 224 demonstrates the trend in canopy cover as plot basal area increases at HOM, which is typical of other sites. As subplot basal area increases, the canopy cover increases steeply and then levels out between 80 and 90%.

Table 88. Percent Canopy Cover Comparison in Rushton and Corresponding Reference Plots.

Pair	Site	Plots		Mean		SD
		Rushton	Ref	Rushton	Ref	Rushton
1	CFI	R1 R2 R3 R4 R5 R6	T5	0.88	0.85	0.03
2	HOM	R1 R2 R3 R4 R5 R6 R7	T1	**0.82**	0.30	0.03
3	OHW	H1 H4	H1R	**0.86**	0.76	0.02
4	OHW	R1A	T1	**0.91**	0.89	NA
5	OHW	R2A R2B	T2	0.90	0.90	0.01
6	PRP	H1 H5	H1R	**0.79**	0.68	0.08
7	TEN	H2	H2R	0.89	0.89	NA
8	TEN	H5	HR	0.89	0.89	NA
9	TEN	H6	H6R	**0.90**	0.88	NA
10	TEN	R2A R2B	T1	0.87	0.88	0.02

[a]Bolded numbers indicate a difference of more than 1 standard deviation.

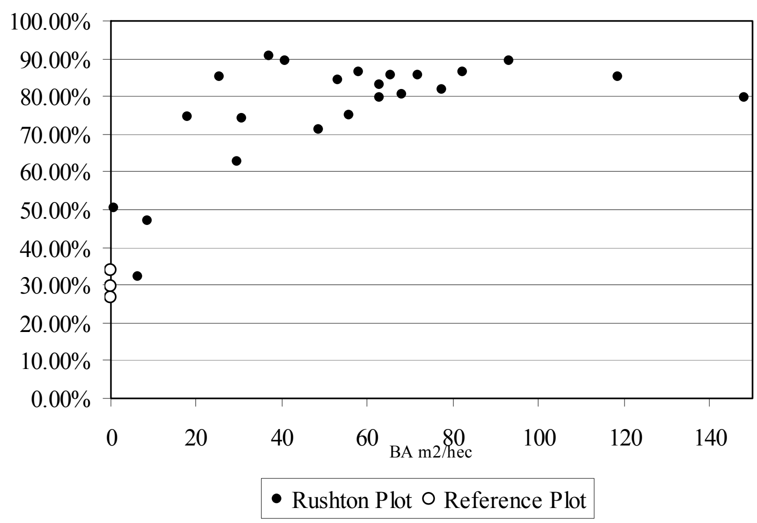


Figure 224. Subplot Basal Area and Percent Canopy Cover at HOM.

Soil Organic Matter

Table 89 provides a comparison of the percent soil organic matter found in samples of the top 10 cm of the soil in Rushton and reference plots. At CFI, HOM, and PP, soil organic matter was greater in Rushton plots, but in most pairings at the older sites of OHW and TEN, percent soil organic matter was higher in reference plots. In all cases the differences between the Rushton and reference plots as indicated by T-tests were significant at the 90% confidence level. At HOM there was a very wide range of organic matter within the Rushton plots that was not present at the other sites.

Table 89. Percent Soil Organic Matter Comparison in Rushton and Corresponding Reference Plots.

Pair	Site	Plots		Samples		Mean % OM[a]		Standard Deviation % OM		P-Value from T-Test
		Rushton	Ref	Rushton	Ref	Rushton	Ref	Rushton	Ref	
1	CFI	R1 R2 R3 R4 R5 R6	T5	153	18	9.06	7.47	3.39	1.96	0.01
2	HOM	R1 R2 R3 R4 R5 R6 R7	T1	57	27	8.78	4.7	11.24	3.06	0.01
3	OHW	H1 H4	H1R	54	27	10.19	9.2	1.65	1.2	3.00E-03
4	OHW	R1A	T1	9	18	10.31	13.4	3.59	5.11	0.08
5	OHW	R2A R2B	T2	36	18	10.8	13.14	3.82	3.92	0.04
6	PRP	H1 H5	H1R	36	27	10.21	7.7	3	1.54	6.00E-05
7	TEN	H2	H2R	27	27	9.26	11.58	1.25	2.49	1.00E-04
8	TEN	H5	H5R	27	27	8.06	**14.06**	1.69	2.94	1.60E-11
9	TEN	H6	H6R	27	27	11.48	**14.84**	3.2	2.68	1.00E-04
10	TEN	R2A R2B	T1	36	18	8.1	10.28	3.1	4.29	0.07

[a]Bolded numbers indicate a difference of more than 1 standard deviation.

Table 90 compares Rushton and reference plot percent organic matter by site. The variation between reference plots on different sites is greater than the variation between Rushton plots on different sites. The average % OM in Rushton sites varies between 8.5 and 10.5%.

Table 90. Percent Soil Organic Matter Summarized by Site and Plot Type.

Site	Mean % OM	
	Rushton	Ref
CFI	9.06	7.47
HOM	8.78	4.70
OHW	10.42	11.52
PRP	10.21	7.70
TEN	9.16	12.91

Understory Vegetation

Table 91 presents a comparison of the understory coverage in Rushton and reference plots. Inconsistent differences occur between the Rushton and reference plots. Among the Rushton plots, the highest cover occurs at CFI, where ferns were planted underneath the drier portions of the plots. Understory coverage at OHW is consistent at around 30% for Rushton plots, lower than at other sites.

Table 91. Average Percent Understory Cover Comparison Between Rushton and Reference Plots.

Pair	Site	Plots		Samples		Average Cover %[a]		
						Mean		SD
		Rushton	Ref	Rushton	Ref	Rushton	Ref	Rushton
1	CFI	R1 R2 R3 R4 R5 R6	T5	34	6	0.96	0.84	0.18
2	HOM	R1 R2 R3 R4 R5 R6 R7	T1	63	9	0.94	0.86	0.17
3	OHW	H1 H4	H1R	18	9	0.32	0.22	0.20
4	OHW	R1A	T1	3	6	0.35	0.25	NA
5	OHW	R2A R2B	T2	12	5	0.32	**0.86**	0.12
6	PRP	H1 H5	H1R	12	9	0.83	**1.20**	0.20
7	TEN	H2	H2R	9	9	0.59	0.62	NA
8	TEN	H5	H5R	9	9	**0.68**	0.58	NA
9	TEN	H6	H6R	9	8	**0.38**	0.15	NA
10	TEN	R2A R2B	T1	12	6	0.47	0.37	0.21

[a]Bolded numbers indicate a difference of more than 1 standard deviation.

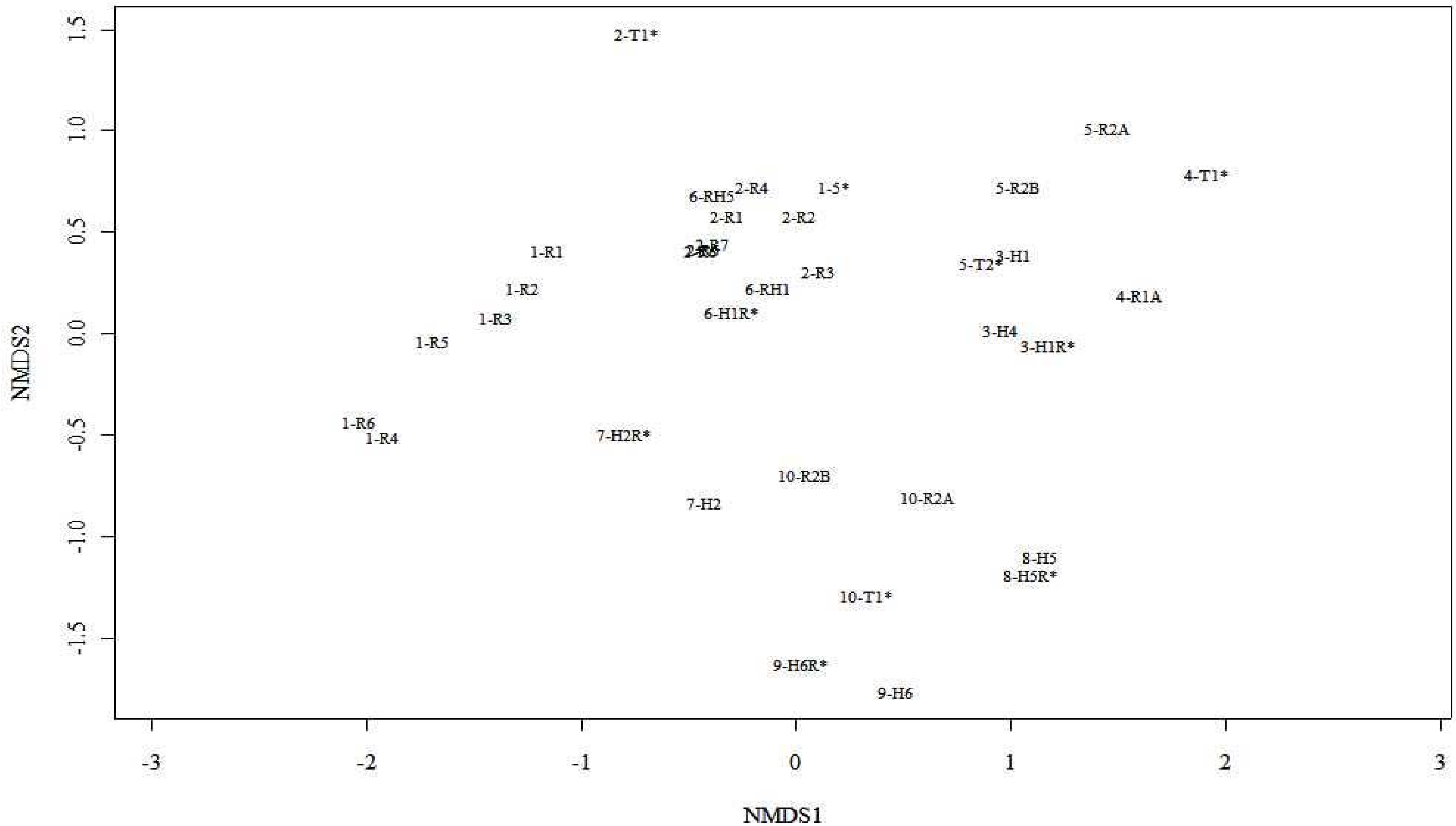
Table 92 summarizes species richness and evenness among pairs of Rushton and references plots. No consistent signal of a difference in richness and evenness is apparent between Rushton and reference plots. The average number of species occurring in Rushton plots is never more than 13, whereas reference plots at CFI and TEN have as many as 21 and 20 species. Species evenness, a measure of the evenness of the distribution of species in a plot, follows a similar trend to species richness when comparing within Rushton and reference pairs. The range of both richness and evenness is greater in the reference than in the Rushton plots.

Table 92. Species Richness and Evenness Comparison in Rushton and Reference Plots.

Pair	Site	Plots		Species Richness[a]			Species Evenness[a]		
				Mean		SD	Mean		SD
		Rushton	Ref	Rushton	Ref	Rushton	Rushton	Ref	Rushton
1	CFI	R1 R2 R3 R4 R5 R6	T5	12	**21**	3.9	0.59	0.70	0.15
2	HOM	R1 R2 R3 R4 R5 R6 R7	T1	12	12	3.4	0.51	**0.86**	0.12
3	OHW	H1 H4	H1R	7	**9**	0.7	0.71	**0.86**	0.06
4	OHW	R1A	T1	9	8	NA	0.81	0.85	NA
5	OHW	R2A R2B	T2	**12**	7	2.1	0.84	0.45	0.00
6	PRP	H1 H5	H1R	5	**7**	0.7	**0.70**	0.53	0.02
7	TEN	H2	H2R	4	3	NA	**0.51**	0.27	NA
8	TEN	H5	H5R	13	**20**	NA	0.80	0.77	NA
9	TEN	H6	H6R	**10**	5	NA	0.78	0.77	NA
10	TEN	R2A R2B	T1	9	**12**	2.1	0.72	0.77	0.05

[a]Bolded numbers indicate a difference of more than 1 standard deviation.

The ordination of species assemblages based on the average cover of species can be a useful means of visualizing the similarity of assemblages in different plots. Figure 225 presents the result of an Nonmetric Multidimensional Scaling (NMDS) of the most prevalent species in the plots. The diagram shows a clear separation of sites and pairs. CFI reference plots are clustered on the left side, with the drier plots R-6 and R-4 close together and R1, the wettest site, on the other end. The CFI reference sites are closer to the HOM Rushton plot. All the HOM Rushton plots (names starting with ‘2’) are clustered among themselves and the 3 PRP sites (names starting with ‘6’). The HOM reference site is isolated from the other groups. Plots in both the OHW (names starting with ‘3’,’4’, and ‘5’) and TEN sites are clustered within their respective sites. Overall there is a much greater difference in species assemblages between sites than within sites or within Rushton-reference pairs.



Note: Plot names are condensed to 'pair-plot name' with a '*' added to the reference plots. The greater the distance between the plots, the less similar their species assemblages.

Figure 225. NMDS Plot of Understory Species Assemblages.

Relationship among Measures of Ecosystem Development

Table 93 contains correlations among selected ecosystem development variables by site. Rushton and reference plots are combined in this analysis by the site they originate from. Differences in the relationship strength and the direction of the relationships between these variables occur between different sites.

Table 93. Correlation Matrices for Ecosystem Development Variables by Site.

CFI									
	Rush_BA	Depth	Range	Tot BA	Canop Cov	U Cover	U Richness	U Evenness	Soil OM
Rush_BA	1.00	0.35	-0.72	1.00	-0.03	0.55	-0.91	-0.68	-0.08
Depth	0.35	1.00	-0.65	0.37	-0.83	0.82	-0.23	-0.87	-0.70
Range	-0.72	-0.65	1.00	-0.75	0.42	-0.45	0.65	0.72	0.16
Tot_BA	1.00	0.37	-0.75	1.00	-0.06	0.54	-0.92	-0.68	-0.07
Canop_Cov	-0.03	-0.83	0.42	-0.06	1.00	-0.47	0.00	0.54	0.46
U_Cover	0.55	0.82	-0.45	0.54	-0.47	1.00	-0.36	-0.93	-0.80
U_Richness	-0.91	-0.23	0.65	-0.92	0.00	-0.36	1.00	0.47	-0.16
U_Evenness	-0.68	-0.87	0.72	-0.68	0.54	-0.93	0.47	1.00	0.73
Soil_OM	-0.08	-0.70	0.16	-0.07	0.46	-0.80	-0.16	0.73	1.00
HOM									
	Rush_BA	Depth	Range	Tot BA	Canop Cov	U Cover	U Richness	U Evenness	Soil OM
Rush_BA	1.00	-0.23	-0.47	0.99	0.71	-0.33	-0.28	-0.95	0.27
Depth	-0.23	1.00	0.17	-0.30	-0.09	0.79	-0.60	-0.01	-0.86
Range	-0.47	0.17	1.00	-0.40	-0.34	0.09	-0.03	0.41	-0.24
Tot_BA	0.99	-0.30	-0.40	1.00	0.66	-0.43	-0.23	-0.91	0.30
Canop_Cov	0.71	-0.09	-0.34	0.66	1.00	0.09	0.12	-0.68	0.33
U_Cover	-0.33	0.79	0.09	-0.43	0.09	1.00	-0.19	0.15	-0.62
U_Richness	-0.28	-0.60	-0.03	-0.23	0.12	-0.19	1.00	0.48	0.58
U_Evenness	-0.95	-0.01	0.41	-0.91	-0.68	0.15	0.48	1.00	-0.14
Soil_OM	0.27	-0.86	-0.24	0.30	0.33	-0.62	0.58	-0.14	1.00
OWH									
	Rush_BA	Depth	Range	Tot BA	Canop Cov	U Cover	U Richness	U Evenness	Soil OM
Rush_BA	1.00	-0.33	-0.16	0.95	0.34	-0.12	0.71	0.32	-0.19
Depth	-0.33	1.00	0.75	-0.57	-0.46	0.21	-0.65	-0.47	-0.31
Range	-0.16	0.75	1.00	-0.32	-0.48	0.11	-0.48	-0.13	-0.53
Tot_BA	0.95	-0.57	-0.32	1.00	0.42	-0.17	0.80	0.40	-0.16
Canop_Cov	0.34	-0.46	-0.48	0.42	1.00	0.29	0.03	-0.33	0.49
U_Cover	-0.12	0.21	0.11	-0.17	0.29	1.00	-0.19	-0.72	0.49
U_Richness	0.71	-0.65	-0.48	0.80	0.03	-0.19	1.00	0.52	-0.04
U_Evenness	0.32	-0.47	-0.13	0.40	-0.33	-0.72	0.52	1.00	-0.41
Soil_OM	-0.19	-0.31	-0.53	-0.16	0.49	0.49	-0.04	-0.41	1.00
PRP									
	Rush_BA	Depth	Range	Tot BA	Canop Cov	U Cover	U Richness	U Evenness	Soil OM
Rush_BA	1.00	0.93	-0.14	0.86	0.89	-0.94	-1.00	0.93	0.78
Depth	0.93	1.00	0.24	0.61	1.00	-1.00	-0.95	0.73	0.49
Range	-0.14	0.24	1.00	-0.62	0.33	-0.22	0.06	-0.49	-0.73
Tot_BA	0.86	0.61	-0.62	1.00	0.53	-0.63	-0.82	0.99	0.99
Canop_Cov	0.89	1.00	0.33	0.53	1.00	-0.99	-0.92	0.66	0.40
U_Cover	-0.94	-1.00	-0.22	-0.63	-0.99	1.00	0.96	-0.75	-0.51
U_Richness	-1.00	-0.95	0.06	-0.82	-0.92	0.96	1.00	-0.90	-0.73
U_Evenness	0.93	0.73	-0.49	0.99	0.66	-0.75	-0.90	1.00	0.95
Soil_OM	0.78	0.49	-0.73	0.99	0.40	-0.51	-0.73	0.95	1.00
TEN									
	Rush_BA	Depth	Range	Tot BA	Canop Cov	U Cover	U Richness	U Evenness	Soil OM
Rush_BA	1.00	-0.22	-0.16	0.96	0.43	-0.06	0.05	0.27	-0.19
Depth	-0.22	1.00	-0.20	-0.22	-0.08	0.30	-0.79	-0.88	-0.27
Range	-0.16	-0.20	1.00	0.04	-0.13	-0.34	0.10	0.35	-0.47
Tot_BA	0.96	-0.22	0.04	1.00	0.47	-0.11	0.09	0.28	-0.29
Canop_Cov	0.43	-0.08	-0.13	0.47	1.00	-0.23	-0.01	-0.20	-0.29
U_Cover	-0.06	0.30	-0.34	-0.11	-0.23	1.00	0.22	-0.26	-0.26
U_Richness	0.05	-0.79	0.10	0.09	-0.01	0.22	1.00	0.64	0.18
U_Evenness	0.27	-0.88	0.35	0.28	-0.20	-0.26	0.64	1.00	0.10
Soil_OM	-0.19	-0.27	-0.47	-0.29	-0.29	-0.26	0.18	0.10	1.00

Note: Correlations between Rushton_BA and effect variables (last five) are highlighted in gray.

Two hydrologic variables, average depth and range of average depth are included in the correlations, along with the total Rushton tree basal area. The response variables included are total basal area, canopy cover, understory cover, understory richness, understory evenness, and percent soil organic matter, and their relationship to Rushton basal area is of primary interest, though their correlations among one another are also worth noting.

At all sites Rushton basal area is strongly positively correlated with total basal area, as was apparent in Table 86, which showed that Rushton trees made up the majority of total basal area in most Rushton plots. However, the correlation with canopy cover is less clear. At CFI correlation is nearly absent because all plots, including Rushton and reference, have very similar canopy coverages (see Table 88). The trend is more positive at the sites where reference plots have less canopy cover. The correlations between Rushton basal area and cover are mostly negative, except at CFI where understory planting occurred, though the relationship is weak at the older sites of OHW and TEN. Rushton basal area ranges from being strongly negatively correlated with understory richness at PRP to being strongly positively correlated at OHW. The correlations between understory evenness also range from strong negative to strong positive. OWH and TEN show the same direction of correlation for all response variables. HOM and PRP, the wettest sites, also show the same direction of correlation in all variables but species evenness.

WETLAND REVEGETATION FIELD TRIALS

Results are presented for marsh revegetation sites (H1, PPW-3), seedling underplanting sites (SA 10, TEN-1, H1u), and monitoring sites (PPW-1, PPW2).

Marsh Revegetation Sites

For each site, hydrologic and soil characteristics are summarized first. Frequency data for planted and volunteer vegetation is presented and compared over two growing seasons. Wetland tree seedling survival and growth are evaluated. The effect of wetland hydroperiod is examined as it pertains to the survival and growth of planted species and the composition and dynamics of volunteer species.

H1 Marsh

Hydrology. Hydrologic conditions after revegetation in October 2005 at the Hooker's Prairie 1 (H1) site were influenced by two extremes: the dramatic spikes in water levels occurring late in the 2005 and 2006 growing season, followed by extended periods of drought, most pronounced during the 2006-2007 growing season. Historic average annual precipitation (cm) for Bartow, Florida, as well as cumulative precipitation (cm) at the CSA for the monitoring periods of September 2005-August 2006 and September 2006-August 2007, is given in Figure 226. Totals for both years were below the historic average, with cumulative rainfall between 2006 and 2007 much lower than the preceding year due to drought conditions. Monthly totals for both years, as well as historic monthly values for Bartow, Florida, are given in Figure 227.

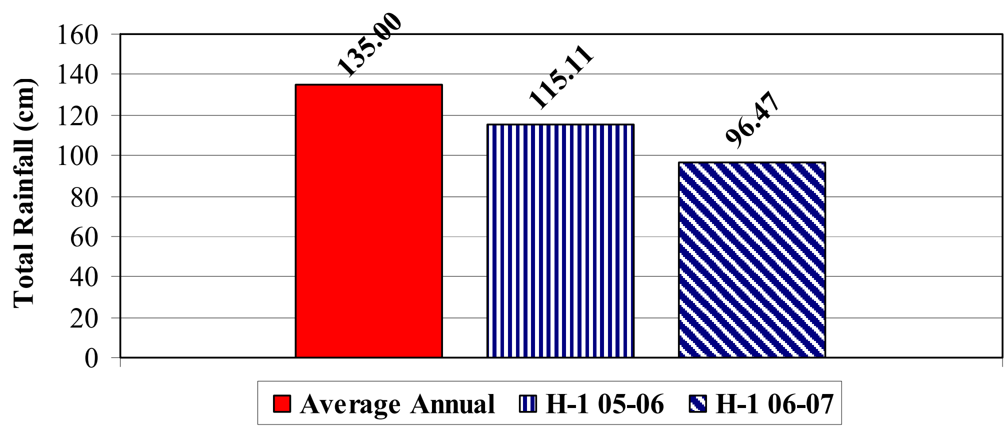


Figure 226. Precipitation Totals at the H1 Marsh.

Note: Historic annual average precipitation for Bartow, Florida was provided by the Southeast Regional Climate Center (SERCC 2008).

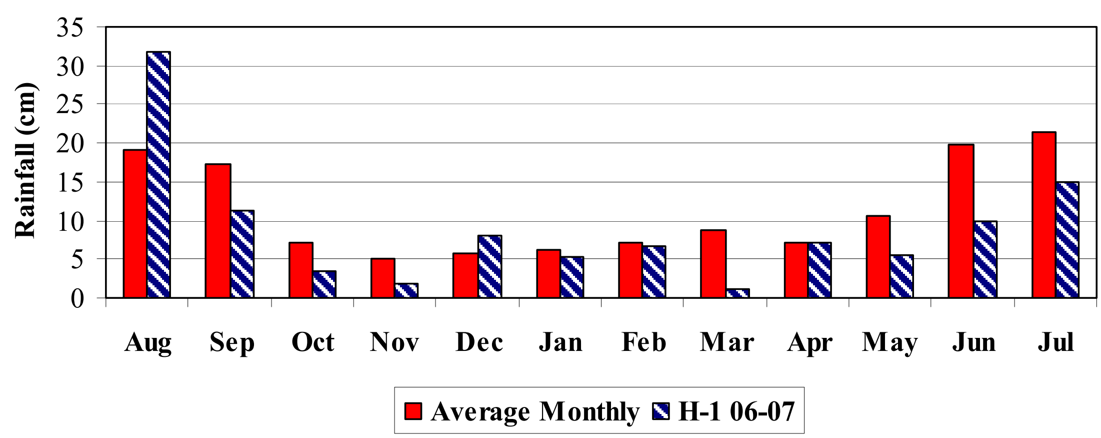


Figure 227. Monthly Precipitation at the H1 Marsh.

Note: Historic monthly average precipitation for Bartow, Florida was provided by the Southeast Regional Climate Center (SERCC 2008).

Figure 228 presents water levels (m) at the H1 marsh over the period of record. Surface water levels at the well were approximately 0.4 m at the time of planting, but rose to 0.73 m approximately two weeks after planting on 10/03/2005. The well remained mostly inundated through the end of March 2006. Below average monthly rainfall during the 2006 growing season kept surface water levels below the ground surface through 07/09/2006, and then they fluctuated between the ground surface and 0.25 m until above-average precipitation in August 2006 drove the water level at the well to 0.8 m. Below-average rainfall from September through November caused water levels at the wetland to drop, and like 2005, water levels remained low, fluctuating below the ground surface between March and July, 2007. Although precipitation was higher than average during August 2007, the wetland water budget was already at a deficit, and so inundation only occurred twice during the 2007 growing season and only to a maximum depth of 0.09 m.

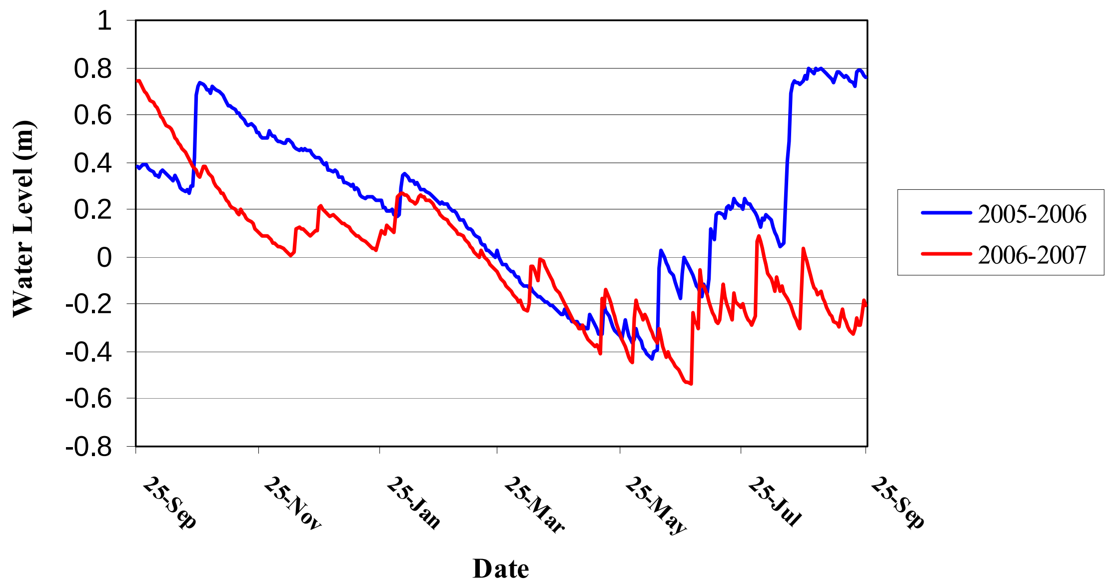


Figure 228. Water Levels at the H1 Marsh Surface Water Well.

Hydrologic conditions experienced over the period of record between 2005 and 2007 for monitoring plots and transects are presented in Tables 94 and 95. Values for various attributes of wetland hydroperiod were calculated from 10/3/2005 to 10/02/2006 ('05-'06) and then from 10/03/2006 through 09/27/07 ('06-'07). Water levels and flooding frequency at the marsh in the second year after planting were lower than the first year due to drought conditions over 2006 and 2007. Average water levels declined by 0.23 m, maximum flooding depth decreased by 0.14 m, and minimum water table depths were 0.10 m lower. As a result, the percentage of time the planting zones, and root zones of the herbaceous and woody species, were inundated was also lower. At plots and transects, percent inundation fell between 37% and 16%, with wetter plots experiencing the greatest declines in inundation (Table 95). At drier monitoring locations within the graminoid planting zone and tree transects, inundation declined to between 7% and 0.5% of the year. Inundation, specifically during the 2007 growing season, only occurred within the bulrush planting zone and the wettest areas in the western flag marsh planting zone.

Substrate. The majority of the H1 marsh revegetation site is classified as clay soil (Table 96). Sand tailings from the dike to the east of the revegetation site were present at Transect 1 (T1), FM1, SG1, and GR1. The transition zone from pure sand soils to clay is narrow, with GR2 classified as sandy clay, and all other plots to the west classified as clay. Two bulrush monitoring plots, BR1 and BR2, contained a high sand content, and were classified as sandy clay loam soils.

Table 97 presents percent organic matter at each monitoring plot and transect for the H1 marsh. Organic matter content is generally high across the entire site, indicating wet conditions have persisted at the site for at least the past several years, with several exceptions. The bulrush monitoring plots, BR1 and BR2, located in the deepest, and therefore wettest, part of the marsh had the highest percentage of organic matter per sample. As would be expected, drier sampling plots at each planting zone had lower organic matter content than wetter plots (Tables 94 and 95). All four sampling locations located on pure sand substrate (Table 96) (SG1, FM1, GR1, and T1) had much lower organic matter content relative to the rest of the site.

Table 94. Hydrologic Characteristics at the H1 Marsh.

Plot	Average Water Level ('05-'06) (cm)	Minimum Water Level ('05-'06) (cm)	Maximum Water Level ('06-'07) (cm)	Average Water Level ('06-'07) (cm)	Minimum Water Level ('06-'07) (cm)	Maximum Water Level ('06-'07) (cm)	Flooding Frequency ('05-'06)	Flooding Frequency ('06-'07)
FM1	-0.02	-0.68	0.55	-0.25	-0.79	0.42	3	2
FM2	-0.14	-0.80	0.44	-0.37	-0.90	0.30	4	1
FM3	-0.02	-0.68	0.56	-0.25	-0.78	0.42	4	2
FM4	0.13	-0.53	0.70	-0.11	-0.64	0.57	3	2
FM5	0.13	-0.53	0.70	-0.12	-0.64	0.57	3	3
FM6	0.02	-0.65	0.59	-0.23	-0.75	0.45	4	3
SP1	-0.29	-0.95	0.29	-0.52	-1.05	0.15	2	1
SP2	-0.20	-0.86	0.37	-0.44	-0.97	0.24	2	1
BR1	0.14	-0.52	0.72	-0.09	-0.62	0.58	3	3
BR2	0.17	-0.49	0.74	-0.07	-0.60	0.61	2	3
SG1	-0.11	-0.77	0.46	-0.34	-0.87	0.33	4	1
SG2	-0.10	-0.76	0.48	-0.33	-0.86	0.34	4	1
SG3	-0.01	-0.67	0.56	-0.24	-0.77	0.43	4	2
SG4	0.00	-0.66	0.57	0.53	-0.77	0.44	4	2
GR1	-0.27	-0.93	0.31	-0.50	-1.03	0.17	3	1
GR2	-0.16	-0.82	0.41	-0.40	-0.93	0.28	3	1
GR3	-0.18	-0.84	0.40	-0.41	0.29	0.26	2	1
GR4	-0.30	-0.96	0.28	-0.53	-1.06	0.14	3	1
GR5	-0.25	-0.91	0.32	-0.48	-1.01	0.19	2	1
T1NW	-0.24	-0.91	0.33	-0.49	-1.01	0.19	2	1
T1SW	-0.25	-0.92	0.32	-0.50	-1.02	0.18	2	1
T1NE	-0.42	-1.08	0.15	-0.67	-1.18	0.02	2	1
T1SE	-0.36	-1.03	0.21	-0.61	-1.13	0.07	2	1
T2NW	-0.38	-1.04	0.19	-0.63	-1.15	0.06	2	1
T2SW	-0.30	-0.96	0.27	-0.55	-1.07	0.14	2	1
T2NE	-0.21	-0.87	0.36	-0.46	-0.98	0.23	2	1
T2SE	-0.15	-0.81	0.42	-0.40	-0.92	0.29	3	1
T3NW	-0.34	-1.00	0.23	-0.59	-1.11	0.10	2	1
T3SW	-0.40	-1.07	0.17	-0.65	-1.17	0.03	2	1
T3NE	-0.41	-1.07	0.16	-0.66	-1.18	0.03	2	1
T3SE	-0.42	-1.08	0.15	-0.67	-1.19	0.02	2	1
T4NW	-0.27	-0.93	0.30	-0.52	-1.03	0.17	2	1
T4SW	-0.22	-0.88	0.35	-0.47	-0.99	0.22	2	1
T4NE	-0.40	-1.06	0.17	-0.65	-1.17	0.04	2	1
T4SE	-0.31	-0.98	0.26	-0.57	-1.08	0.12	2	1

Table 95. Percent (%) Inundation for Monitoring Plots and Transects at the H1 Marsh.

Plot	Percent Inundation ('05-'06)	Percent Inundation ('06-'07)	Growing Season % Inundation ('05-'06)	Growing Season % Inundation ('06-'07)	Root Zone % Inundation ('05-'06)	Root Zone % Inundation ('06-'07)	Growing Season Root Zone % Inundation ('05-'06)	Growing Season Root Zone % Inundation ('06-'07)
FM1	48.77	13.65	20.48	0.00	75.07	53.20	57.75	20.10
FM2	32.60	7.52	21.60	0.00	68.22	40.95	46.01	6.22
FM3	49.86	14.21	22.54	0.00	75.62	53.20	58.69	20.10
FM4	66.03	34.26	42.25	2.87	85.75	71.03	76.06	50.72
FM5	66.03	33.15	42.25	2.39	85.75	70.19	76.06	48.36
FM6	54.52	16.99	26.76	0.00	77.81	55.71	62.44	23.94
SP1	22.47	3.34	20.66	0.00	55.07	17.83	26.76	0.00
SP2	29.04	5.29	21.13	0.00	64.38	27.86	39.44	1.44
BR1	66.85	37.33	43.66	4.31	86.58	72.98	77.46	54.07
BR2	68.22	40.95	46.01	6.22	88.49	77.44	80.75	61.72
SG1	37.26	8.36	21.60	0.00	70.14	45.13	49.30	8.61
SG2	38.08	8.91	21.60	0.00	70.68	46.24	50.23	9.57
SG3	50.41	14.76	22.54	0.00	76.16	54.04	59.62	21.53
SG4	52.05	15.88	23.94	0.00	77.26	54.60	61.50	22.49
GR1	24.11	3.90	20.66	0.00	57.81	19.50	29.58	0.00
GR2	31.51	5.85	21.60	0.00	66.85	37.33	43.66	4.31
GR3	30.68	5.57	21.60	0.00	66.03	34.26	42.25	2.87
GR4	21.37	3.34	20.66	0.00	52.88	16.43	24.41	0.00
GR5	26.30	4.18	21.13	0.00	60.55	22.28	33.33	0.00
T1NW	26.30	4.18	21.13	0.00	73.15	50.14	54.46	14.83
T1SW	25.75	4.18	21.13	0.00	72.88	49.58	53.99	13.88
T1NE	16.99	0.56	20.66	0.00	63.29	25.07	37.56	0.48
T1SE	18.90	1.67	20.66	0.00	66.30	35.38	42.72	3.35
T2NW	18.36	1.39	20.66	0.00	65.48	31.75	41.31	2.39
T2SW	21.37	3.06	20.66	0.00	70.14	45.13	49.30	8.61
T2NE	28.77	5.01	21.13	0.00	75.34	53.20	58.22	20.10
T2SE	31.51	6.69	21.60	0.00	80.27	59.05	66.67	30.14
T3NW	19.45	2.23	20.66	0.00	67.40	39.00	44.60	5.26
T3SW	17.53	0.84	20.66	0.00	63.56	26.74	38.03	0.96
T3NE	17.53	0.84	20.66	0.00	63.56	25.91	38.03	0.48
T3SE	16.99	0.56	20.66	0.00	63.01	24.79	37.09	0.00
T4NW	24.11	3.90	20.66	0.00	72.33	48.75	53.05	12.44
T4SW	28.22	4.74	21.13	0.00	75.07	52.37	57.75	18.66
T4NE	17.81	1.11	20.66	0.00	64.11	27.02	38.97	0.96
T4SE	20.82	2.79	20.66	0.00	69.32	42.62	47.89	7.18

Table 96. Soil Texture Determinations at the H1 Marsh.

Sample Location	Soil Type	% Sand	% Clay	% Silt
BR1	Sandy Clay Loam	63.20	21.60	15.20
BR2	Sandy Clay Loam	55.20	29.60	15.20
SP1	Clay	25.60	66.80	7.60
SP2	Clay	15.20	74.80	10.00
FM1	Sand	100.00	0.00	0.00
FM2	Clay	21.20	46.80	32.00
FM3	Clay	35.60	56.80	7.60
FM4	Clay	35.20	53.60	11.20
GR1	Sand	100.00	0.00	0.00
GR2	Sandy Clay	48.00	41.60	10.40
GR3	Clay	27.20	66.40	6.40
GR4	Clay	19.60	74.80	5.60
GR5	Clay	24.00	69.60	6.40
SG1	Sand	100.00	0.00	0.00
SG2	Clay	25.60	68.80	5.60
SG3	Clay	25.60	68.80	5.60
SG4	Clay	25.60	68.80	5.60
T1	Sand	100.00	0.00	0.00
T2	Clay	24.00	69.60	6.40
T3	Clay	19.60	74.80	5.60
T4	Clay	25.60	66.80	7.60

Table 97. Percent (%) Organic Matter at the H1 Marsh.

Sampling Location	Percent Organic Matter (μ ± σ) (%)
BR1	39.74 ± 3.56
BR2	36.81 ± 5.43
SP1	15.94 ± 0.16
SP2	17.08 ± 3.90
FM1	1.82 ± 1.61
FM2	16.61 ± 1.67
FM3	28.36 ± 9.07
FM4	24.89 ± 14.29
SG1	1.19 ± 0.14
SG2	17.52 ± 0.96
SG3	18.67 ± 3.16
SG4	17.01 ± 1.34
GR1	1.50 ± 0.43
GR2	15.37 ± 1.87
GR3	22.25 ± 1.38
GR4	17.27 ± 1.61
GR5	18.77 ± 1.76
T1	0.56 ± 0.13
T2	13.98 ± 0.88
T3	12.55 ± 0.98
T4	15.43 ± 2.93

Bulrush Planting Zone. *Scirpus californicus* (giant bulrush) was planted in the deeper portion of the marsh, an area occupied by cattail before clearing in 2005. Figure 229 presents frequency data for *Scirpus californicus*. The initial overall frequency for *Scirpus californicus* was low due to the spike in water level following planting in 2005, which caused individual plants to dislodge from the clay soil and float to the water surface, thus decreasing the species' presence. While both monitoring plots, BR1 and BR2, experienced similar hydrologic conditions (Figure 230), bulrush frequency never rebounded at BR2 after the 2005 growing season. Overall frequency declined from 2005 to 2006, due to the absence of individuals at BR2, but rebounded to its initial value as the species continued to spread within BR1. From 2005 to 2007, *Scirpus californicus* increased each year in overall frequency at the BR1 monitoring plot, indicating survival and growth within portions of the planting area. By 2006, *Scirpus californicus* was also present within the spike rush and graminoid planting zones. By 2007, the spread of *Scirpus californicus* shifted mainly to the west and southwest of the bulrush planting zone into the spike rush and flag marsh planting zones, and was no longer present at drier areas within the graminoid planting zone (GR2, GR3).

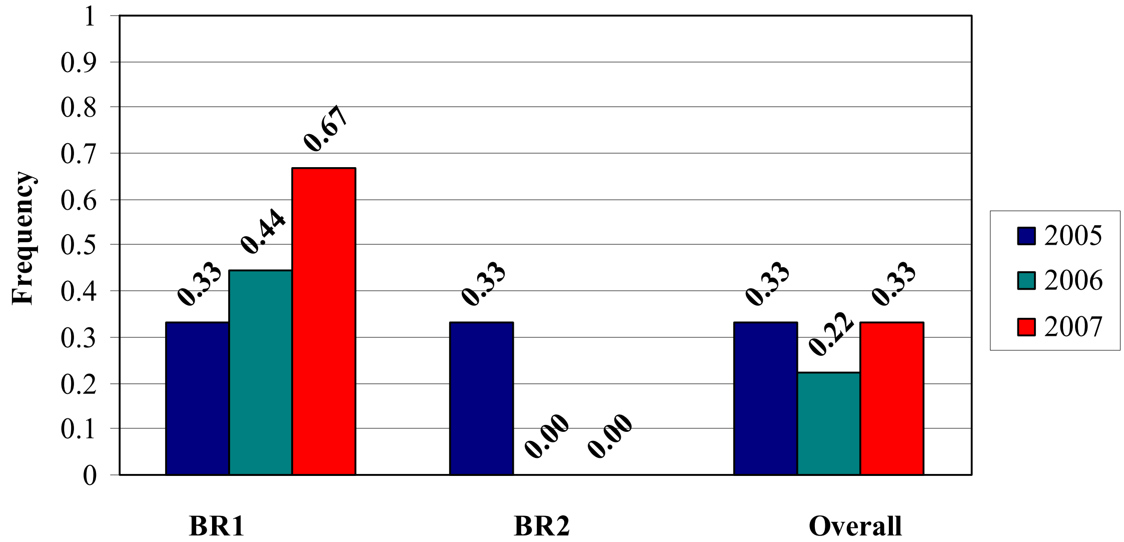


Figure 229. Bulrush Frequency at the H1 Marsh.

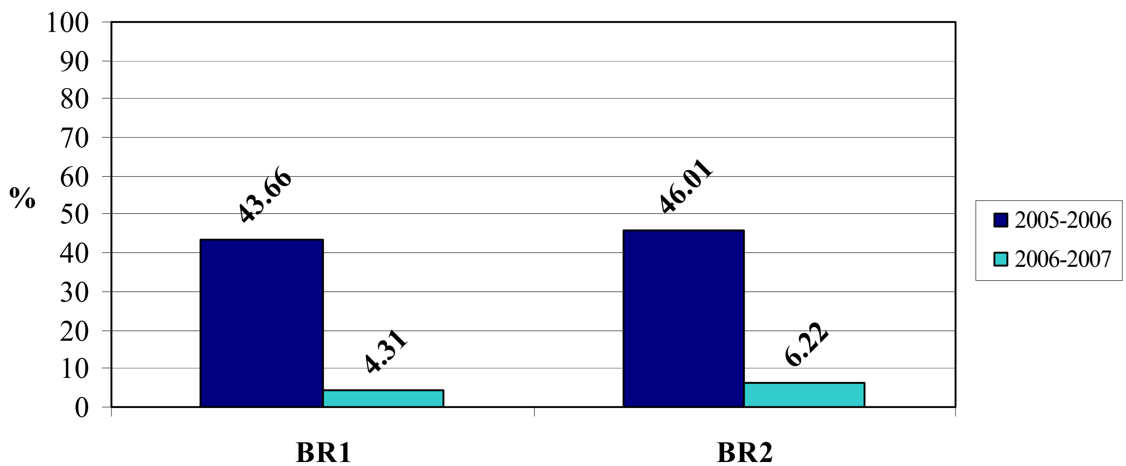


Figure 230. Bulrush Planting Zone Percent Inundation (%) at the H1 Marsh.

Detailed information on volunteer species frequency in all planting zones is presented in Appendix C. After planting occurred in 2005, sparse *Typha latifolia* was present. By 2006 its overall frequency within the planting area had increased to 0.83. *Eupatorium capillifolium* (dog fennel) had also recruited to the majority of the bulrush planting zone. By 2007, *Eupatorium capillifolium* was no longer present, but *Typha latifolia*, *Polygonum hydropiperoides*, and *Pluchea odorata* (sweet scent) had overall frequencies of 1.00, 1.00, and 0.056, respectively, although the majority of *Typha latifolia* appeared stressed.

Spike Rush Planting Zone. *Eleocharis cellulosa* (club-rush), was planted in two areas along the northern edge of the site. *Eleocharis cellulosa* increased in overall frequency between 2005 and 2007 (Figure 231). By 2007, *Eleocharis cellulosa* was pervasive in an area to the south of the spike rush planting zone that is similar in elevation and hydroperiod with the original planting area. *Eleocharis cellulosa* and *Scirpus californicus* were the only two planted species found present outside their initial planting areas within two years of planting.

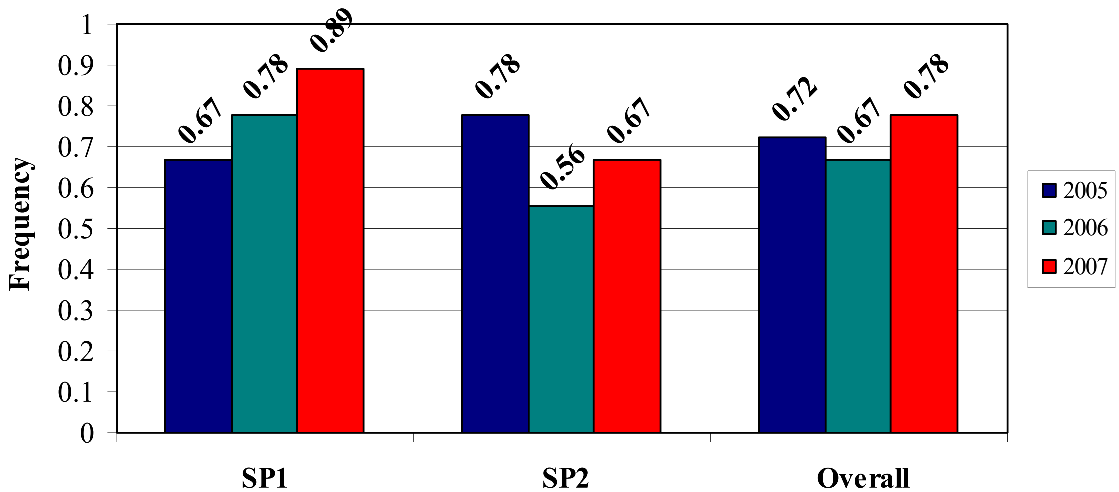


Figure 231. Spike Rush Frequency at the H1 Marsh.

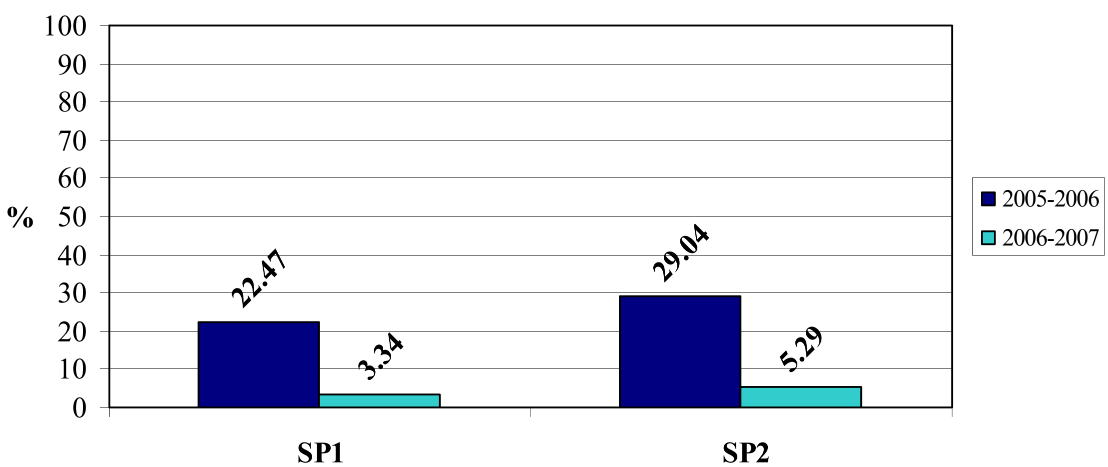


Figure 232. Spike Rush Planting Zone Percent Inundation (%) at the H1 Marsh.

In 2005, drier areas within the planting zone were occupied by *Ludwigia peruviana* and *Commelina diffusa*, while *Typha latifolia* and *Momordica charantia* (balsam pear) occupied wetter areas. *Polygonum hydropiperoides*, *Eupatorium capillifolium*, and *Baccharis halimifolia* (Eastern baccharis) recruited heavily to the planting area in 2006. By 2007, *Ludwigia peruviana*, *Eupatorium serotinum* (lateflowering thoroughwort), *Polygonum hydropiperoides*, *Mikania scandens*, *Baccharis* *halimifolia*, and *Eupatorium capillifolium* were present throughout the entire planting zone at high overall frequencies, with *Ludwigia peruviana* and *Eupatorium serotinum* forming a thick shrub canopy (1-2 m in height) throughout the planting zone.

Woody Shrub Invasion. Between 2006 and 2007, a thick shrub canopy established over the majority of the spike rush, flag marsh, saw-grass, graminoid, and tree planting zones, as shown in Figure 233. Photographs documenting the encroachment of woody shrubs over the period of record are given in Appendix C. The canopy transitioned in species dominance from *Pluchea odorata* and *Eupatorium serotinum* in wetter areas to *Ludwigia peruviana*, *Baccharis halimifolia*, and *Myrica cerifera* (wax myrtle) at the drier wetland edge. The canopy's height, cover, and stem density increased with distance from the center of the marsh, reaching 3-4 meters in height within the graminoid and wetland tree planting area.

Flag Marsh Planting Zone. Three flag marsh species, *Sagittaria lancifolia* (bulltongue arrowhead), *Thalia geniculata* (bent alligator-flag), and *Pontederia cordata* (pickerelweed), were planted on the eastern and western sides of the marsh. Figure 234 presents frequency data for the three species. Despite small declines between 2005 and 2006, *Sagittaria lancifolia* performed well at the FM1, FM2, and FM3 monitoring plots. Plot FM4, where survival was poor, experienced the greatest inundation in 2005 and 2006 and water depths to 0.63 m immediately after planting (Figure 235). Overall frequency increased to 0.69 over the period of record (0.93 with FM4 excluded). Overall frequency for *Pontederia cordata* increased slightly from 0.44 to 0.50 from 2005-2007. The highest frequencies of *Pontederia cordata* were found in 2006 and 2007 at FM1 and FM2, the driest plots (Figure 234). *Thalia geniculata* declined in overall frequency within the planting zones, with no individuals present within monitoring plots after two years. *Thalia geniculata* was only present in one monitoring plot in 2006, but was observed to be present, reproducing vegetatively, and flowering throughout the eastern flag marsh planting zone.

Additional monitoring plots, FM5 and FM6, were established in 2007 to capture flag marsh species' frequency west of the FM4 monitoring plot. *Sagittaria lancifolia* was abundant in the area west of FM4 in 2007, with frequencies of 0.66 and 0.33 at FM5 and FM6. *Pontederia cordata* was not present within the random plots, but was visually observed within the area. No individuals of *Thalia geniculata* were present in the monitoring plots nor observed within the area.

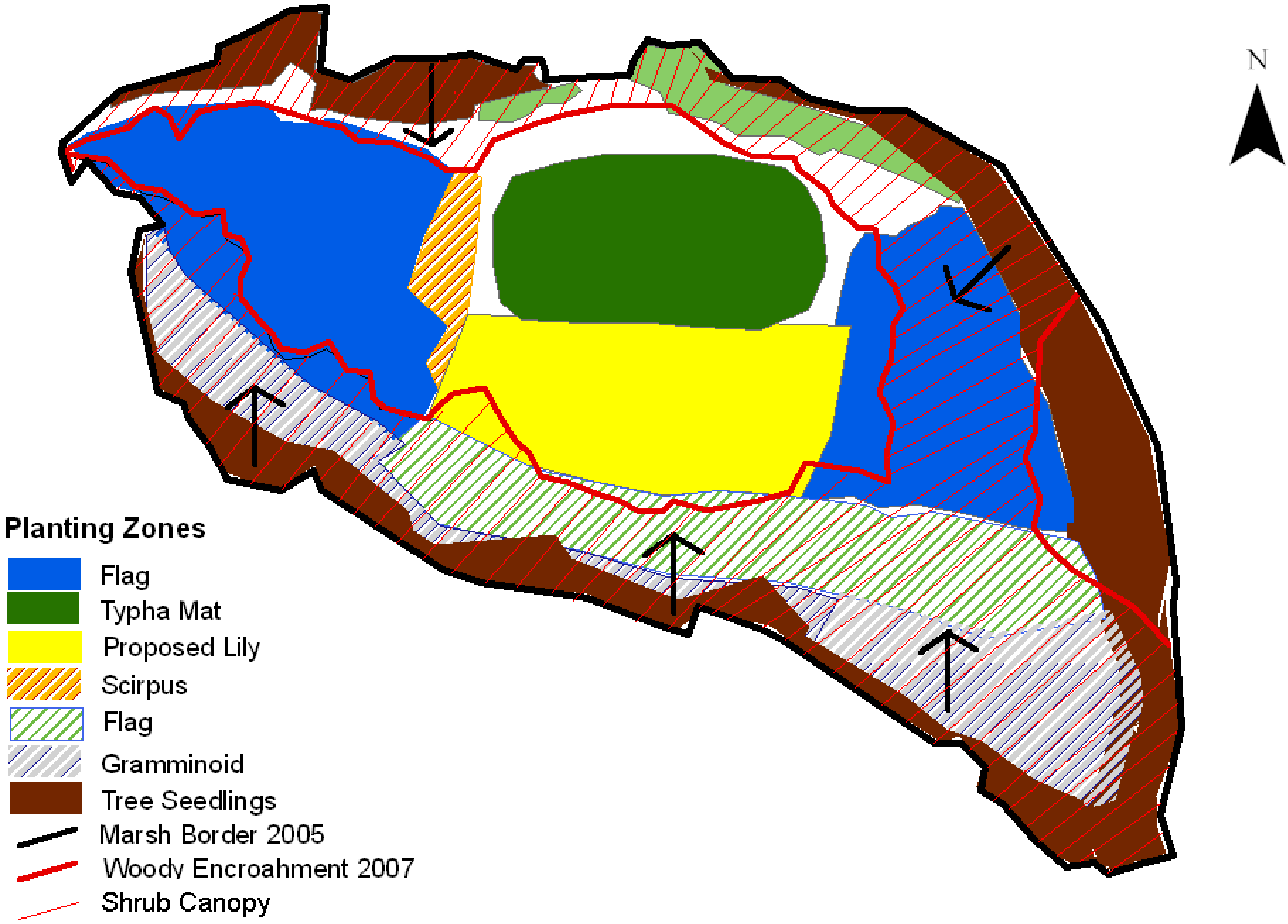


Figure 233. Extent of Woody Shrubs at the H1 Marsh.

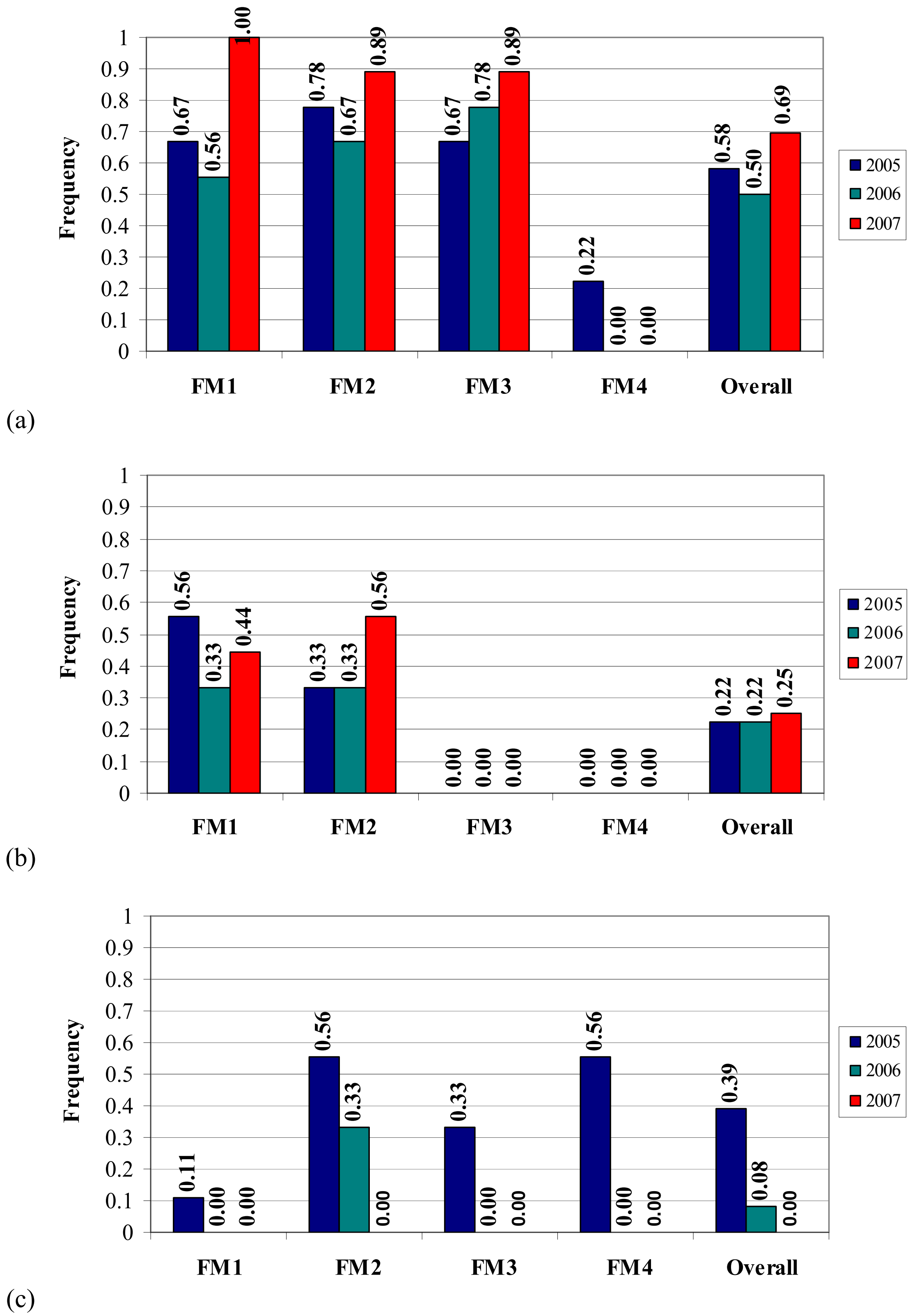
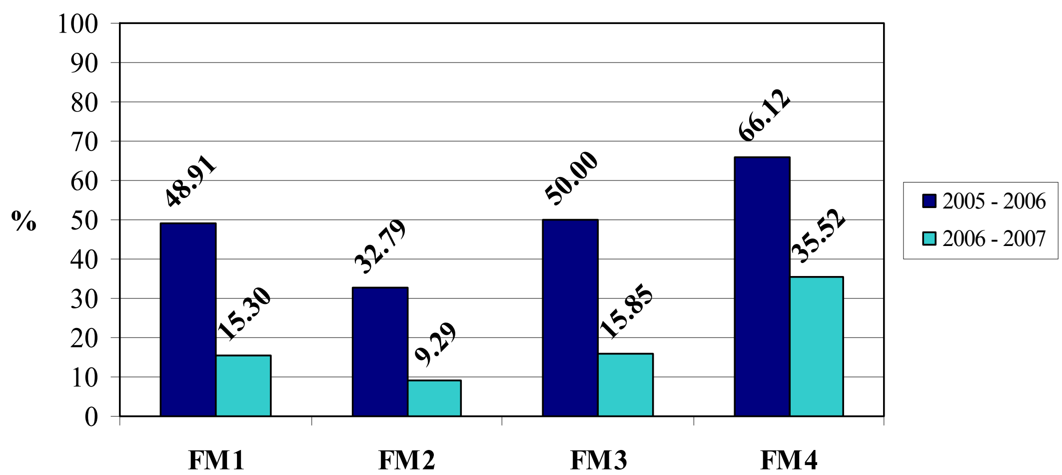


Figure 234. (a) *Sagittaria lancifolia*, (b) *Pontederia cordata*, and (c) *Thalia geniculata* Frequency at the H1 Marsh.



a)

(b)

Figure 235. (a) Percent Inundation (%) and (b) Water Levels at the Flag Marsh Planting Zone at the H1 Marsh.

Volunteer species were not present within planting areas in 2005, but *Eupatorium capillifolium* volunteered to high frequencies at FM1 and FM4 in 2006. By 2007, *Ludwigia peruviana*, *Eupatorium serotinum*, and *Pluchea odorata* had formed a thick shrub canopy over the eastern planting zone, while the western zone was dominated by a groundcover of *Polygonum hydropiperoides* and sparse *Pluchea odorata*.

Saw-grass Planting Zone. *Cladium jamaicense* (saw-grass) declined in frequency from 2005 to 2006, but rebounded slightly in 2007 (Figure 236). Although frequency declined at all monitoring plots between 2005 and 2006, the worst survival for the species occurred at the wettest plot, SG4 (Figure 237). In 2007, *Cladium jamaicense* maintained its 2006 frequency at SG2 and increased in frequency at SG1, with the size of surviving plants in these plots increasing from 2005 through 2007.

Volunteer species were not present within the planting zone in 2005, but by 2006, *Eupatorium capillifolium* and *Ludwigia peruviana* were present in all plots, with *Baccharis halimifolia* occurring only on drier areas and *Typha latifolia* only at wetter. By 2007, a thick shrub canopy of *Ludwigia peruviana* and *Eupatorium serotinum* had established over the entire planting zone (Figure 233).

Graminoid Planting Zone. Five graminoid species were planted along the southern edge of the H1 marsh. Most species were not successful, with the exceptions of *Spartina bakerii* (spartina grass) and *Juncus effusus* (soft rush), as seen in Figures 238-243. *Juncus effusus* was the only species of the five to increase in overall frequency between 2005 and 2007 (Figure 238). The species either maintained or declined in frequency at four of five monitoring plots between 2005 and 2006, but as conditions at the H1 marsh became drier between 2006 and 2007 (Tables 23 and 24), the species increased or maintained frequency at all five plots. *Spartina bakerii* experienced a similar trend in establishment and growth over the first growing season, with declines in frequency over the first growing season and an increase over the second (Figure 239). *Muhlenbergia capillaris* (hairawn muhly grass) was initially present in GR3, GR4, and GR5. Frequency declined to zero at all three plots between 2005 and 2006, and never reestablished within monitoring plots over the second growing season (Figure 240). *Panicum hemitomon* (maidencane) and *Bacopa caroliniana* (lemon bacopa) similarly occurred in three monitoring plots in 2005 and declined to zero overall frequencies in 2006, with no rebound in 2007 (Figures 241 and 242). *Peltandra virginica* (green arrow arum) maintained a slight frequency at only one plot over the entire period of record (Figure 243).

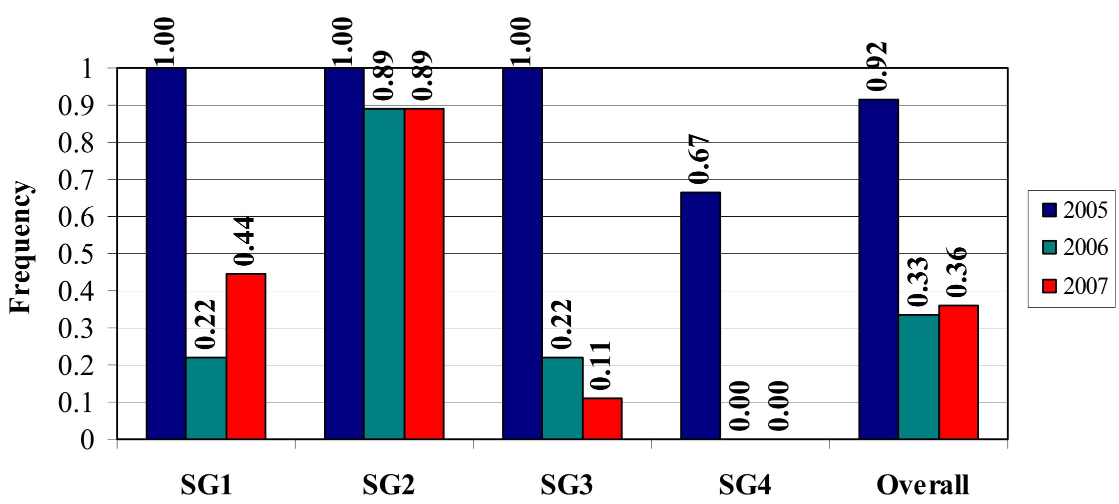


Figure 236. Saw-grass Frequency at the H1 Marsh.

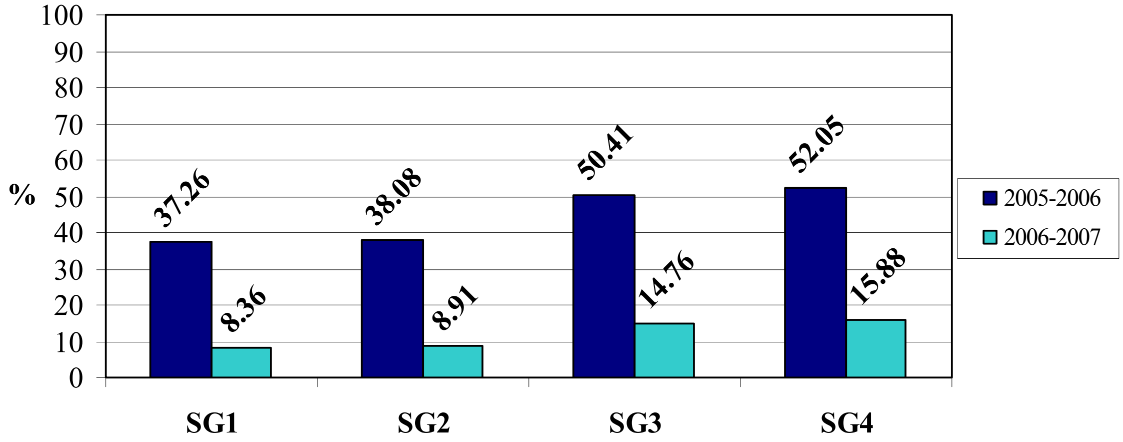


Figure 237. Saw-grass Planting Zone Percent Inundation (%) at the H1 Marsh.

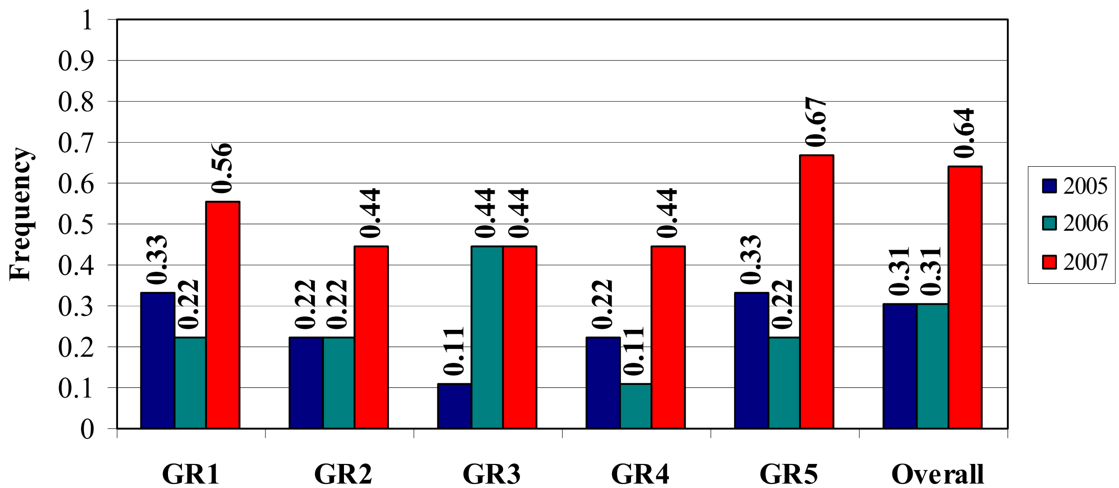


Figure 238. *Juncus effusus* Frequency at the H1 Marsh.

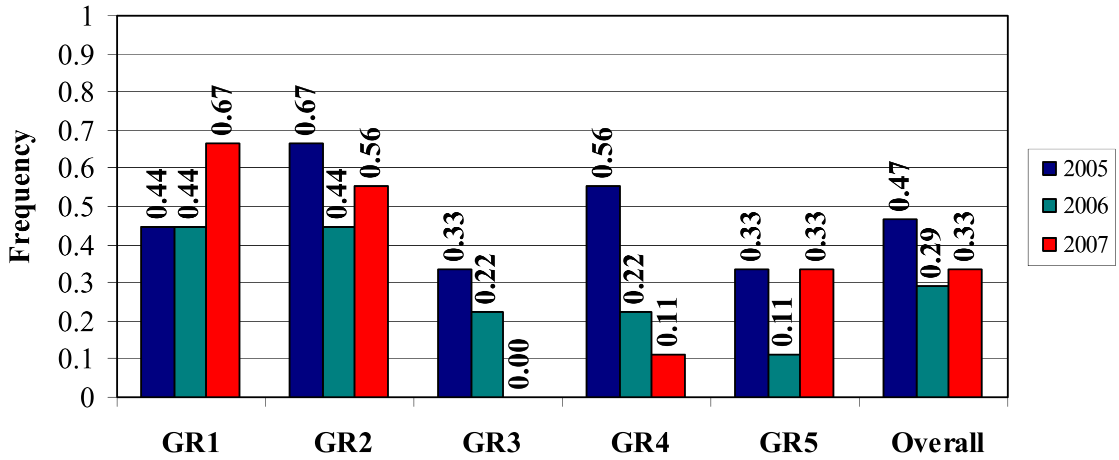


Figure 239. ***Spartina bakerii*** **Frequency at the H1 Marsh.**

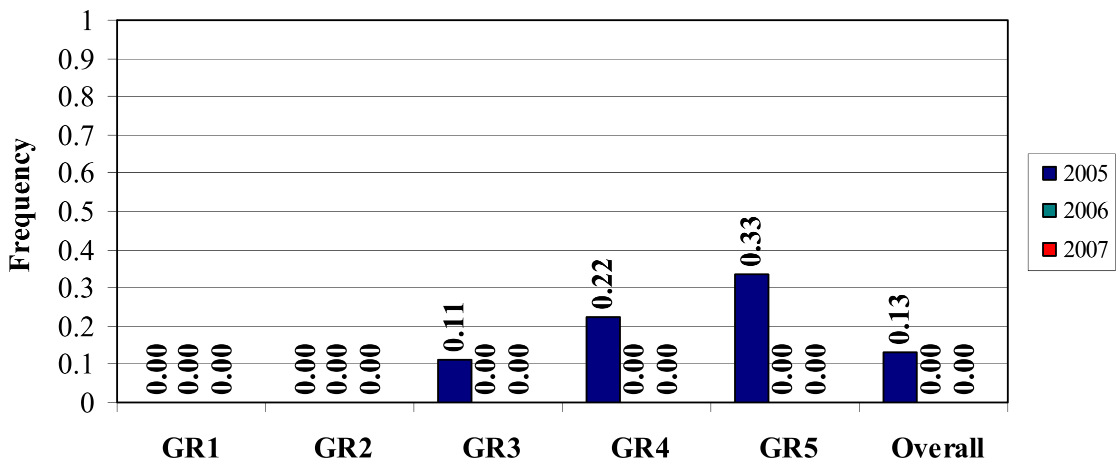


Figure 240. ***Muhlenbergia capillaris*** **Frequency at the H1 Marsh.**

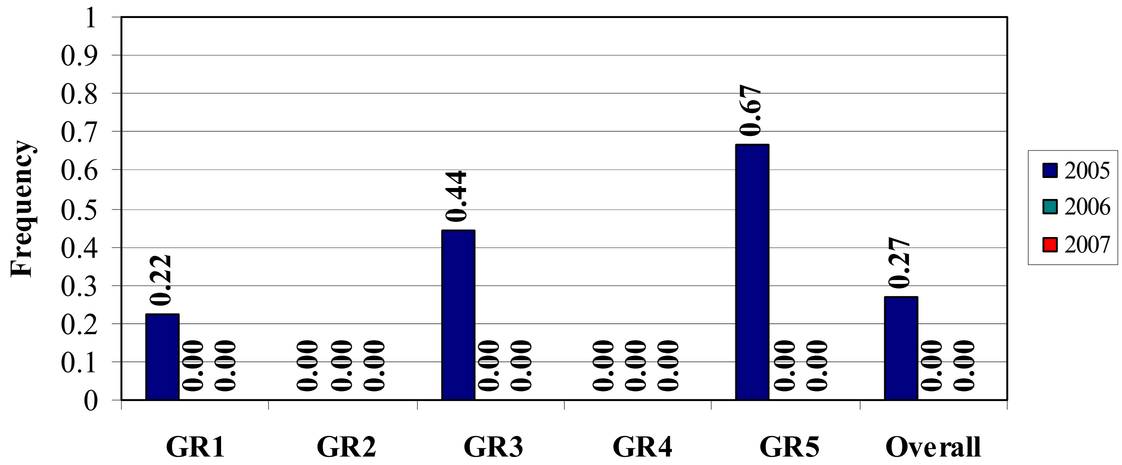


Figure 241. ***Panicum hemitomon*** **Frequency at the H1 Marsh.**

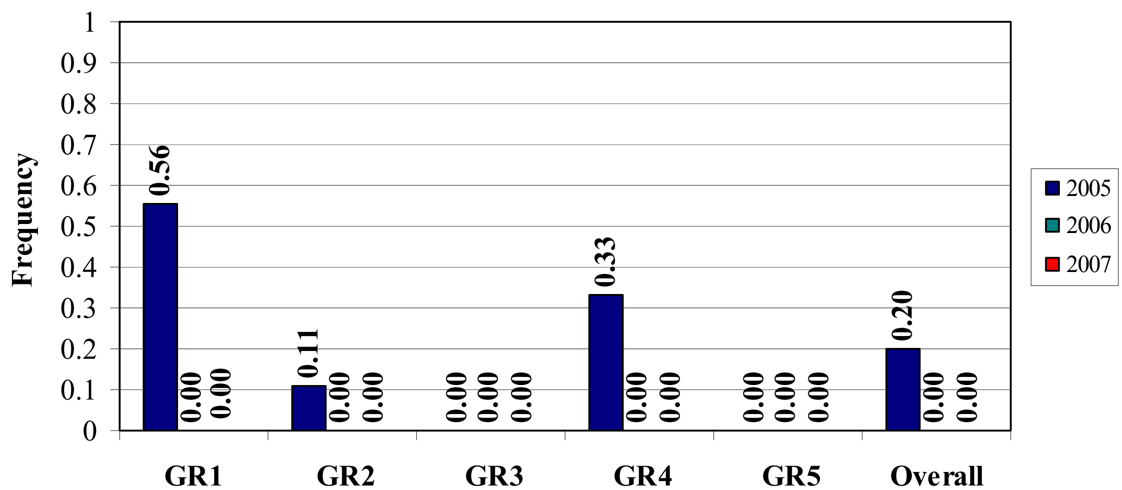


Figure 242. ***Bacopa caroliniana*** **Frequency at the H1 Marsh.**

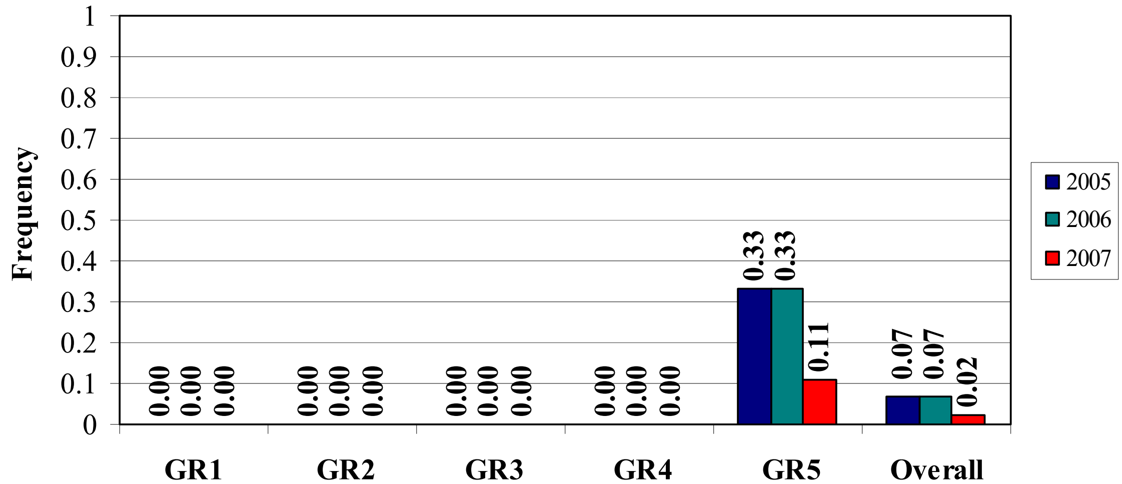


Figure 243. *Peltandra virginica* Frequency at the H1 Marsh.

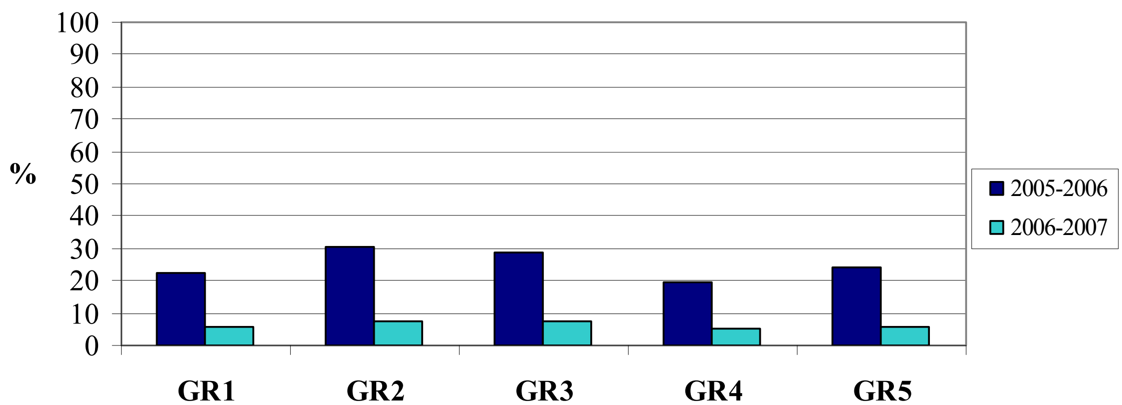


Figure 244. Graminoid Planting Zone Percent (%) Inundation at the H1 Marsh.

In 2005, several volunteer species were present at low overall frequencies, especially at the driest monitoring plots, GR1 and GR4 (Figure 244). *Eupatorium capillifolium*, *Polygonum hydropiperoides*, and *Baccharis halimifolia* were present throughout the planting zone at high frequencies in 2006. By 2007, a thick shrub canopy had established over the planting area, which transitioned in dominance from *Eupatorium serotinum* and *Pluchea odorata* to *Baccharis halimifolia* and *Ludwigia peruviana* along the site's hydrologic gradient, from wet to dry.

Lily Marsh. The site was supplemented with two species of floating leaf aquatics in May of 2006. *Nymphaea odorata* (fragrant water lily) and *Nuphar polysepala* (spatterdock) were planted in the deeper, central portion of the marsh. Stem length was appropriate for the planting area's water depth (personal communication, J. Allen, RSS field services) and floating leaves rested on the water's surface the day of planting. Two weeks after planting occurred, on 08/18/2006, water levels had risen by approximately 20 cm from the planting day water level. No individuals of either species were present when monitored (0% survival, frequency = 0) at two weeks and two and twelve months after planting.

Tree Seedling Survival. Nine species of wetland trees were planted around the periphery of the marsh in 2005. Overall wetland tree seedling survival, an aggregate of the nine planted species, was 81% in 2006 and 56% in 2007 (n = 140). Figure 245 presents seedling survival data. Seedling survival for species was relatively high after two growing seasons, with the exceptions of *Persea palustris* (swamp bay) and *Hypericum fasciculatum* (peelbark St. John's wort), which both had zero survival after the second growing season.

Taxodium distichum had the best survival in 2006 and 2007, maintaining 100% (n = 19) survival two years after planting. *Itea virginica* (Virginia willow) (85%, n = 21), *Cephalanthus occidentalis* (button bush) (92%, n = 13), *Fraxinus caroliniana* (pop ash) (85%, n = 20), and *Hypericum fasciculatum* (peelbark St. John's wort) (87%, n = 8) also had high survival in 2006. *Nyssa sylvatica var. biflora* (62%, n = 8) and *Persea palustris* (53%, n = 13) had the worst survival after the first growing season. Along with *Taxodium distichum*, *Cephalanthus occidentalis* was able to maintain a high percent survival through 2007 (84%, n = 13). The tree seedling periphery was supplemented in August, 2006, with a population of *Annona glabra* (pond apple) tree seedlings. *Annona glabra* had 85% (n = 27) survival after one year.

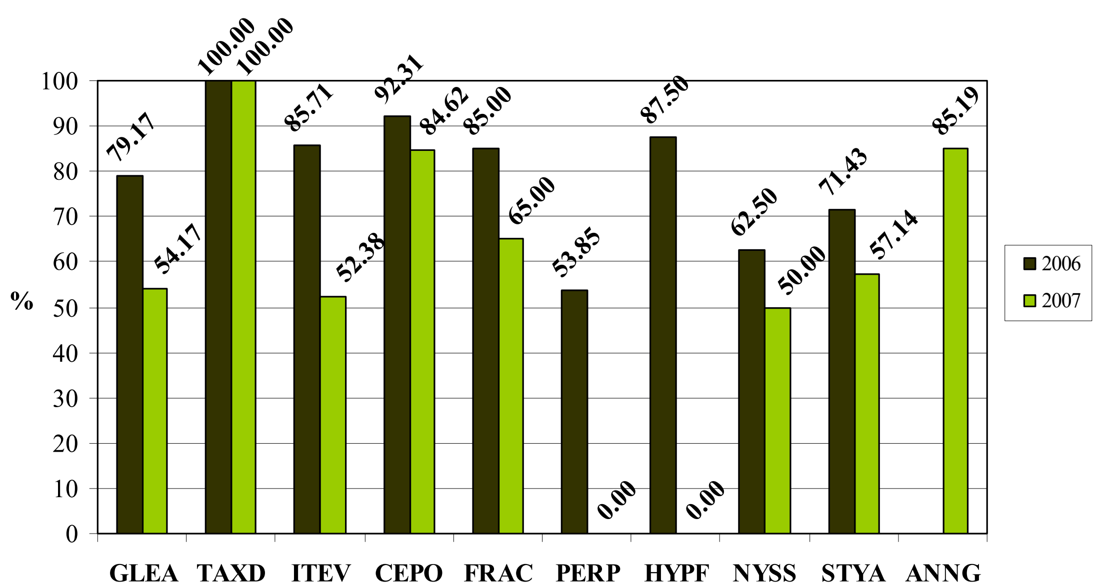


Figure 245. Percent Survival (%) for Seedlings (2005-2007) at the H1 Marsh.

Seedling Growth. Height data for seedlings is presented in Table 27. *Gleditsia aquatica* (45.7 cm, 52%) and *Taxodium distichum* (54.79 cm, 61%) had the largest mean increase and largest percent (%) change in mean height from over the period of record. *Annona glabra* had a larger increase in mean seedling height and percent change in height after one growing season than all other species after two growing seasons, with the exception of *Taxodium distichum*. *Cephalanthus occidentalis* increased in mean height 11.9% between 2005 and 2007, although the stand only increased by 0.57% between 2005 and 2006. *Itea virginica* and *Nyssa sylvatica var. biflora* both experienced better growth over the first growing season. *Fraxinus caroliniana* experienced low growth both years, with mean height increasing 3.9% from 2005-2007. *Styrax americana*, *Persea palustris*, and *Hypericum fasciculatum* all experienced minimal to no increase in mean height, with *Styrax americana* experiencing a negative change in mean height due to seedling dieback and the other two species declining in survival to zero over the period of record.

Seedling Survival, Hydrology, Substrate, and Volunteer Vegetation. Survival and change in height for each seedling were graphed against the difference in elevation (dElevation) between the seedling location and the surface water well to observe any trends in survival and growth as they related to hydrology. Graphs for seedling survival and change in height as they pertained to water levels are contained in Appendix C. Percent inundation experienced at tree transects ranged from 32% to 17% in the first year after planting and 7% to 0.6% in the second. *Taxodium distichum* was able to survive and grow along the entire gradient over which seedlings were planted, both in the first and second years after planting. *Hypericum fasciculatum* and *Persea palustris* also survived and grew along the planting zone's gradient, but from 2006 to 2007, experienced 100% mortality of planted seedlings. Growth was mixed for *Cephalanthus occidentalis*, but sampled seedlings were able to survive in 2006 and 2007, except for the driest areas along the gradient. *Itea virginica* seedlings seemed to survive and grow better at wetter sampling areas from 2005 to 2006, but experienced mortality and seedling dieback along the entire gradient from 2006 to 2007. Survival and growth was mixed along the planting zone's gradient from 2005 through 2007 for *Nyssa sylvatica var. biflora*, *Styrax americana*, *Fraxinus caroliniana*, and *Gleditsia aquatica.* However, most planted species had low sample sizes and species survival was certainly affected by other environmental variables that interact strongly with hydrologic conditions, such as substrate and volunteer vegetation.

Since the four belted transects experienced a similar range of wetland hydroperiod (Tables 94 and 95), seedlings were aggregated by transect in order to compare survival on differing substrates. T2, T3, and T4 were classified as clay with roughly equal parts sand and silt and had similar organic matter content, while T1 was 100% sand and had only 0.5 ± 0.1% organic matter (Tables 96 and 97). Table 99 lists overall survival by transect. In 2005, T1 had the highest survival (96%, n = 58) of the four belted transects used to monitor seedlings. T4 also had relatively high seedling survival after one year. T1, located on the most well drained substrate, unbuffered by clay, had the highest decline from 2006 to 2007, despite having the least woody shrub encroachment of the four transects.

Table 98. Height Data for Seedlings at the H1 Marsh.

Species	Average Height (05)	σ (05)	MIN (05)	MAX (05)	Average Height (06)	σ (06)	MIN (06)	MAX (06)	Average Height (07)	σ (07)	MIN (07)	MAX (07)
GLEA	87.38	16.23	32.00	110.00	117.58	37.88	65.00	215.00	133.08	72.85	74.00	300.00
TAXD	89.63	12.03	65.00	105.00	108.53	10.09	95.00	135.00	144.42	33.37	80.00	195.00
ITEV	67.76	16.37	25.00	95.00	76.11	18.19	40.00	110.00	71.00	36.92	19.00	118.00
CEPO	80.46	14.54	60.00	105.00	80.92	17.84	55.00	105.00	90.55	22.43	60.00	131.00
FRAC	69.15	9.76	55.00	85.00	70.76	20.15	35.00	105.00	71.85	20.91	43.00	101.00
PERP	42.62	7.10	30.00	55.00	41.43	8.52	30.00	50.00	0.00	0.00	0.00	0.00
HYPF	47.50	8.86	35.00	60.00	51.43	12.15	35.00	65.00	0.00	0.00	0.00	0.00
NYSS	65.75	12.42	45.00	85.00	77.75	6.34	70.00	85.00	81.75	27.40	46.00	111.00
STYA	69.36	11.83	52.00	90.00	70.00	6.12	60.00	75.00	59.13	18.73	31.00	84.00
ANNG	N/A	N/A	N/A	N/A	83.11	18.77	23.00	110.00	133.53	19.42	77.00	175.00

Species	Δ Average Height ('05-'06) (cm)	Δ Average Height ('06-'07) (cm)	Δ Average Height ('05-'07) (cm)	% Change ('05-'06)	% Change ('06-'07)	% Change ('05-'07)
GLEA	30.20	15.50	45.71	34.57	13.19	52.31
TAXD	18.89	35.89	54.79	21.08	33.07	61.13
ITEV	8.35	-5.11	3.24	12.32	-6.72	4.78
CEPO	0.46	9.63	10.08	0.57	11.90	12.53
FRAC	1.61	1.08	2.70	2.34	1.53	3.90
PERP	-1.19	-41.43	-42.62	-2.78	-100.00	-100.00
HYPF	3.93	-51.43	-47.50	8.27	-100.00	-100.00
NYSS	12.00	4.00	16.00	18.25	5.14	24.33
STYA	0.64	-10.88	-10.23	0.93	-15.54	-14.75
ANNG	N/A	50.42	N/A	N/A	60.66	N/A

Table 99. Percent Survival (%) by Transect at H1 Marsh.

Transect	Trees Present (2005)	Trees Present (2006)	Trees Present (2007)	% Survival ('05-'06)	% Survival ('06-'07)
T1	58	56	34	96.55	58.62
T2	21	14	14	66.67	66.67
T3	22	14	11	63.64	50.00
T4	39	30	20	76.92	51.28
Overall	140	114	79	81.43	56.43

In 2005, the tree planting zone was mostly free of volunteer vegetation. By 2006, *Eupatorium capillifolium* had recruited to the tree planting zone along with *Polygonum hydropiperoides*, *Baccharis halimifolia*, and *Ludwigia peruviana*, which was particularly dense at the periphery of the marsh. In 2007, *Baccharis halimifolia, Ludwigia peruviana,* and *Eupatorium serotinum* were the dominant volunteer species in the planting zone. The shrub canopy was taller than the height of nearly all tree seedlings, with the exception of seedlings at T1, where the canopy was not fully developed, and several individuals of *Gleditsia aquatica* at T4.

PPW-3

Hydrology. The PPW-3 marsh revegetation site experienced drought conditions lasting from planting in August 2006 through the end of the monitoring period in August 2007. Total precipitation in both 2006 and 2007 for Bartow, Florida, was far less than 135 cm, the historical average annual rainfall for the area. PPW-3 received a total of 99.77 cm of precipitation over the period of record (August 2006-September 2007). Monthly rainfall values, for all growing season months except April, were lower than historic average monthly rainfall totals for Bartow, Florida (Figure 246). Figure 247 presents water levels at the PPW-3 well. The wetland site was dry and water levels at the well remained at -0.31 m or lower during the first month after planting. Flooding at the surface water well occurred once for the period of record, for nine days, from 09/10/2006 through 09/19/2006, a little over one month after planting occurred in August. After September 2006, water levels fell to over -1.00 m below the ground surface. A precipitation event, which occurred between 02/02/2007 and 02/03/2007, caused water levels at the well to increase to a depth of -0.19 m.

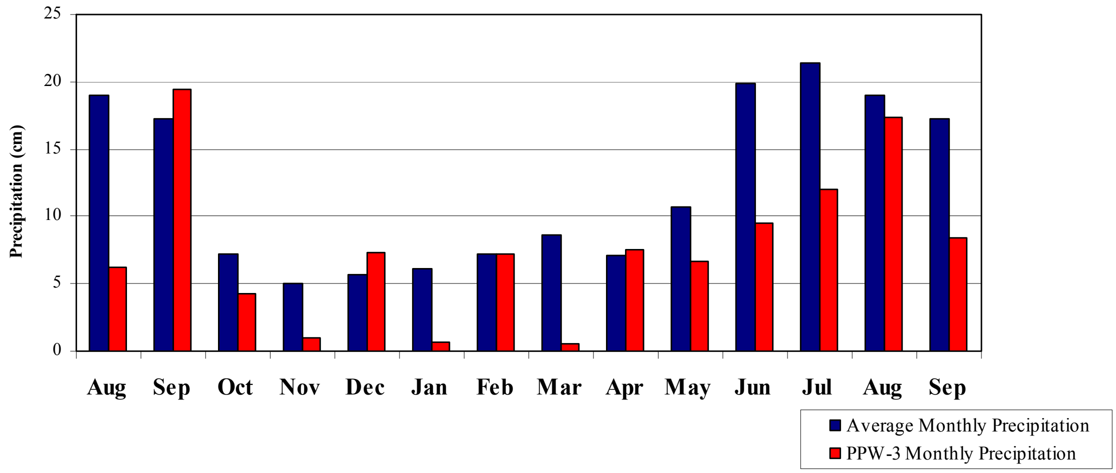


Figure 246. Monthly Precipitation for Polk County, Florida, and the PPW-3 Marsh (08/2006 through 09/2007).

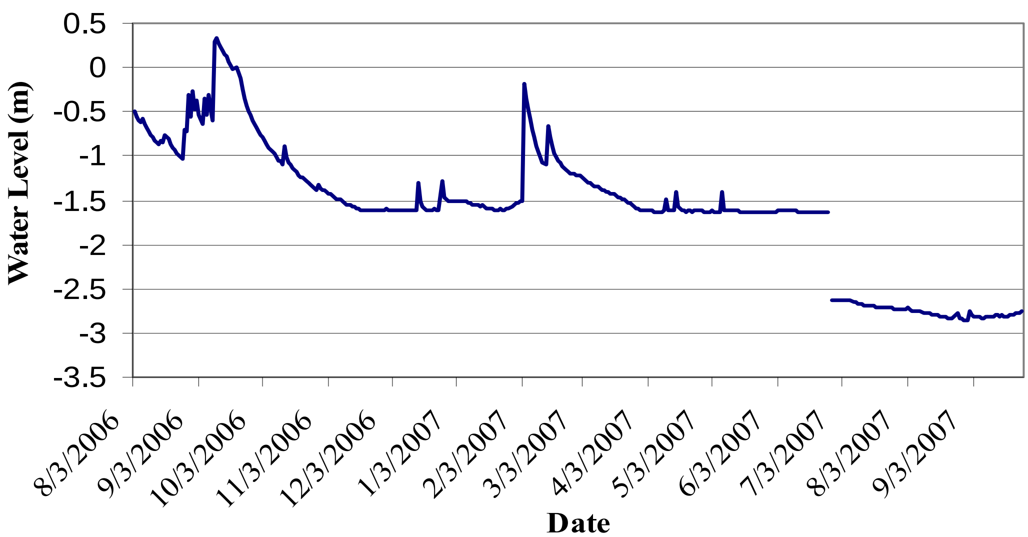


Figure 247. Water Levels at the PPW-3 Surface Water Well.

The well was established 1.62 m below the ground surface, with the assumption that water levels in the deepest portion of the wetland would not fall below this depth. Water levels appeared to remain static at approximately -1.62 m between 12/01/2006 and 02/02/2007 and also from 03/30/2007 through 06/26/2007 (Figure 247). After examining these readings, another well was constructed on 06/27/2007 to a depth of -3.09 m and water levels were found to be -2.6 m below the ground surface. For this reason, water levels recorded at -1.62 m can most likely be assumed to be below this value for the dates listed above. Additionally, water levels between 03/30/2007 and 06/27/2007 likely experienced a sloped decline from -1.59 m on 03/29/07 to -2.60 m on 06/27/2007. After 06/27/2007, water levels never declined to the secondary well depth, -3.09 m. Water levels remained very low between June and August of 2007.

Hydrologic characteristics for plots and transects at PPW-3 are presented in Table 100. This includes water level data during the period of record as well as the percent of time the soil surface and root zone (depths of 30.48 cm for herbaceous species and 50.00 cm for tree seedlings) were inundated. As might be expected with the drought conditions experienced during the period of record, annual mean water levels were very low, ranging between -1.32 m at Saw-grass plot 2 (SG2) and -2.19 m at the driest end of Transect 1 (T1). Percent time inundated was minimal, with three herbaceous monitoring plots never inundated and only six of the eight plots experiencing inundation of their root zones. Maximum inundation of the root zone was only 6% for SG2, the wettest plot. The revegetation was designed with the spike rush and flag marsh planting zones in the deepest portions of the marsh, however shifts in planting design on the planting day led the saw-grass planting zone to experience the highest average water levels and inundation. The spike rush planting zone only experienced inundation within the root zone for 0.8% (SP1) and 3% (SP2) of the time over the period of record, and plots within the flag marsh planting for 3% (FM1) and 4% (FM2) percent of the time. Inundation never occurred at either graminoid planting zone monitoring plot, and only one plot experienced minimal root zone inundation.

Table 100. Hydrologic Data for Monitoring Plots and Transects at PPW-3.

Plot	Mean Water Level (cm)	Minimum Water Level (cm)	Maximum Water Level (cm)	Percent Inundation	Root Zone Percent Inundation[ab]
Well	-141.45	-274.25	32.46	2.47	4.38
T1 start	-219.45	-352.25	-45.54	0.00	0.55
T1 end	-140.45	-273.25	33.46	2.74	7.67
T2 start	-205.45	-338.25	-31.54	0.00	1.64
T2 end	-171.45	-304.25	2.46	0.27	3.84
T3 start	-186.45	-319.25	-12.54	0.00	3.01
T3 end	-154.45	-287.25	19.46	1.64	5.75
SP1	-195.95	-328.75	-22.04	0.00	0.82
SP2	-161.45	-294.25	12.46	1.10	3.29
SG1	-145.95	-278.75	27.96	2.19	4.38
SG2	-132.45	-265.25	41.46	3.29	6.03
FM1	-159.45	-292.25	14.46	1.37	3.56
FM2	-144.45	-277.25	29.46	2.19	4.38
GR1	-208.95	-341.75	-35.04	0.00	1.92
GR2	-179.45	-312.25	-5.54	0.00	0.00

[a]The root zone depth for herbaceous species is assumed to be 30.48 cm (12.00in) below the ground surface.
[b]The root zone depth for tree seedlings is assumed to be 50.00 cm (19.68in) below the ground surface.

Each of the belted transects was inundated less than 3% of the time at their lowest elevation end (Table 100). Root zone inundation occurred along all portions of the belted transects, ranging between 0.55% and 7.67% of the time along T1, 1.64% and 3.84% at T2, and 3.01% and 5.71% at T3. No monitoring plots or portions of belted transects were inundated or experienced any root zone inundation between March 2007 and August 2007. Due to rain events, the majority of inundation and root zone inundation for plots and transects over the period of record occurred during September. Due to the inadequacy of the initial surface water well depth, annual average and minimum water levels are likely much lower than presented for plots and transects.

Substrate. Particle size distribution for soils at monitoring plots and transects are presented in Table 101. The majority of the site was classified as clay, with percent sand ranging between 18% and 24%. Flag marsh Monitoring Plot 1 (FM1) was classified as a silt, with 29% sand, 26% clay, and 45% silt. Only monitoring Transect 3 (T3) is classified as sand, with 91% sand, 6% clay, and 3% silt. Percent organic matter (%) for each plot and transect is listed in Table 102. Organic matter ranged between 6.28 ± 0.65% and 10.06 ± 1.02% across the entire site. Although percent organic matter in wetlands is variable, these are relatively low amounts for a wetland system with moderate clay content, indicating the ephemeral nature of this wetland system. Soils at this wetland can be classified as mineral soils, containing less than 12% organic matter and less than 60% clay (Cowardin and others 1979). Wetter monitoring plots at PPW-3 don't always exceed drier plots in organic matter content, and sampling locations at tree transects (drier locations) have more organic matter per sample than the wetter, herbaceous planting areas.

Table 101. Soil Texture Determinations at PPW-3.

Sample Location	Soil Type	% Sand	% Clay	% Silt
FM1	Loam	28.80	26.40	44.80
FM2	Clay	20.80	52.40	26.80
SG1	Clay	20.80	36.40	42.80
SG2	Clay	20.80	36.40	42.80
SP1	Clay	24.80	30.40	44.80
SP2	Clay	18.40	44.40	37.20
GR1	Clay	18.80	46.40	34.80
GR2	Clay	20.80	42.40	36.80
T1 start	Clay	18.40	50.40	31.20
T1 end	Clay	20.80	36.40	42.80
T2 midpoint	Clay	14.40	52.40	33.20
T3 midpoint	Sand	90.80	6.40	2.80

Table 102. Percent Organic Matter (%) at PPW-3.

Sampling Location	Percent Organic Matter (μ ± σ) (%)
FM1	6.28 ± 0.65
FM2	7.87 ± 0.54
SP1	6.85 ± 1.76
SP2	8.95 ± 0.63
SG1	6.39 ± 0.41
SG2	8.57 ± 0.84
GR1	9.17 ± 0.61
GR2	8.97 ± 0.84
T1 Start	10.06 ± 0.34
T1 20m	9.24 ± 0.85
T1 End	8.68 ± 0.37
T2 Start	9.38 ± 0.40
T2 End	10.06 ± 1.02
T3 Start	9.13 ± 0.52
T3 End	9.03 ± 0.67

Spike Rush Planting Zone. *Eleocharis cellulosa* (club-rush) was planted to the south of the flag marsh planting zone. Frequency data are given in Figure 248. *Eleocharis cellulosa* decreased in overall frequency between 2006 and 2007. Frequency at both monitoring plots declined to zero after one year. Neither monitoring plot experienced inundation within the first month of planting, and while SP2 experienced greater above-ground and root-zone inundation in September 2006, the survival of *Eleocharis cellulosa* was equally poor at the wetter (SP2) and drier (SP1) monitoring plots. All *Eleocharis cellulosa* individuals within the planting zone experienced dieback, when qualitatively observed on 10/12/2006.

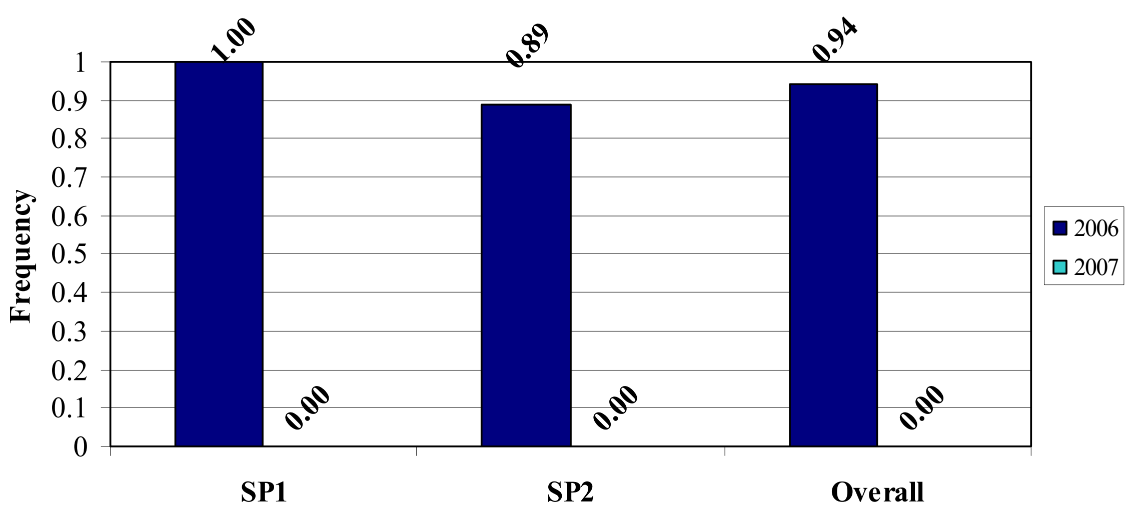


Figure 248. Spike Rush Frequency at PPW-3.

Detailed information on volunteer species frequency in all planting zones is presented in Appendix C. No volunteer species were present when the planting zone was first monitored in 2006. After the 2007 growing season, *Polygonum hydropiperoides*, *Eupatorium capillifolium* (dog fennel), *Boehmeria cylindrica* (smallspike false nettle), and *Ambrosia artemisiifolia* (common ragweed) were present at high overall frequencies throughout the spike rush planting zone.

Flag Marsh Planting Zone. Three flag marsh species, *Sagittaria lancifolia* (bulltongue arrowhead), *Thalia geniculata* (bent alligator-flag), and *Pontederia cordata* (pickerelweed) were planted north of the spike rush planting zone. *Sagittaria lancifolia* increased in overall frequency from 0.22 to 0.27 from 2006 to 2007 (Figure 249). FM1, the drier plot, maintained its initial frequency, and FM2, the wetter plot, increased in frequency over one year. *Thalia geniculata* (bent alligator-flag) survived well, maintaining its initial overall frequency of 0.16 (Figure 250). *Pontederia cordata* (pickerelweed) had the worst survival of the three flag marsh species, with a decline in overall frequency from 0.33 to 0.11 between 2006 and 2007 (Figure 251). The species maintained its low initial frequency in the wetter plot, and declined in frequency from 0.55 to 0.22 at FM1, the drier plot. Low initial frequencies for all three species are a result of planting density and to a lesser degree, some non-random planting. The species mixture was planted at an approximate density of one plant per 1.7 m^2 and monitored using 9 m^2 plots. Also, although workers were instructed to plant the flag marsh zone with an even distribution of the mixture, inevitably several areas within the planting zone were clumped with the same species.

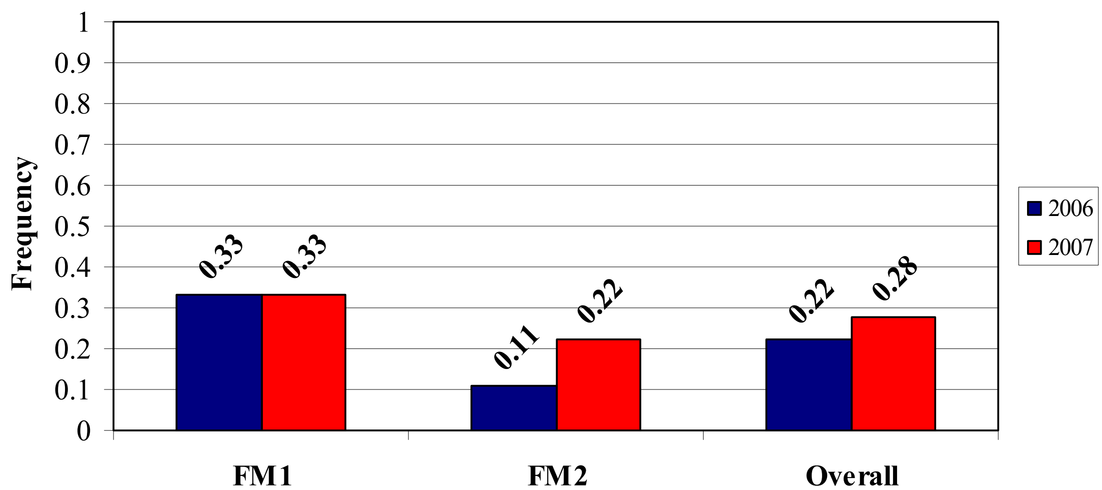


Figure 249. *Sagittaria lancifolia* Frequency at PPW-3.

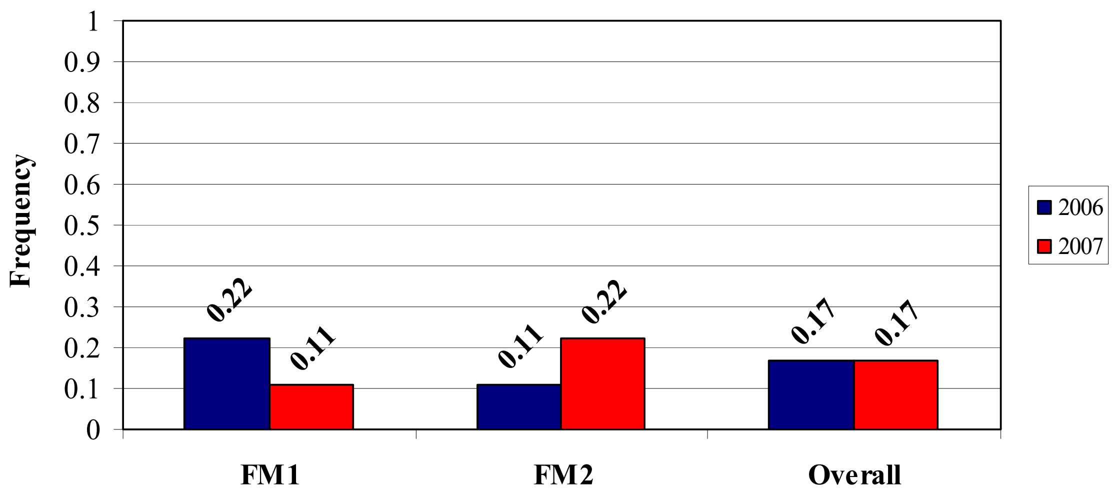


Figure 250. ***Thalia geniculata*** **Frequency at PPW-3.**

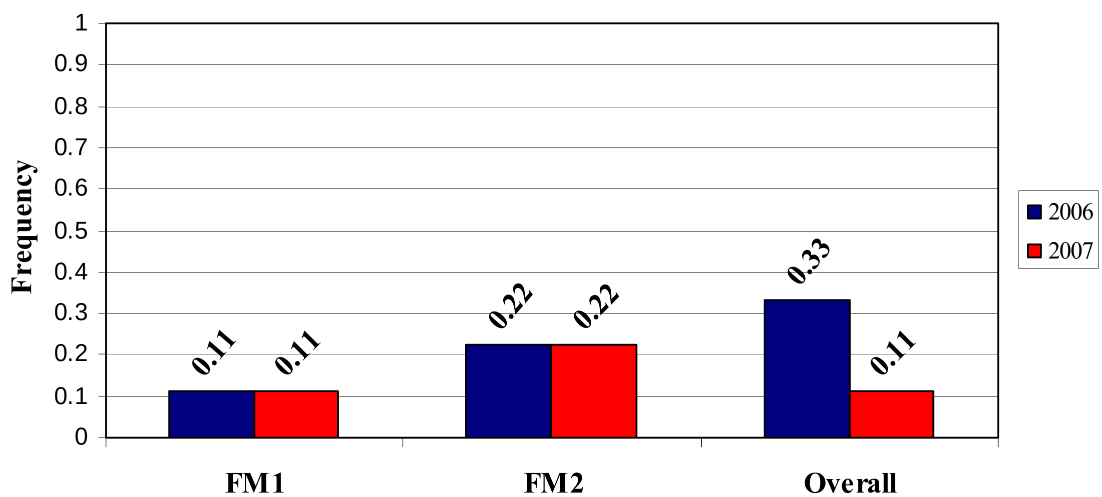


Figure 251. ***Pontederia cordata*** **Frequency at PPW-3.**

No volunteer species were present when the planting zone was first monitored in 2006, however several species were present when the planting zone was monitored in 2007. *Polygonum hydropiperoides, Eupatorium capillifolium, Boehmeria cylindrica*, and *Cyperus virens* (green flatsedge) volunteered to high overall frequencies within the flag marsh planting zone.

Saw-grass Planting Zone. *Cladium jamaicense* (saw-grass), planted north of the flag marsh planting zone, declined in overall frequency between 2006 and 2007, with both monitoring plots experiencing declines in frequency from 0.88 to 0.44 (Figure 252). Both the wetter (SG2) and drier (SG1) plots experienced equal declines in frequency. No volunteer species were present when the planting zone was first monitored in 2006. *Eupatorium capillifolium, Polygonum hydropiperoides*, and *Indigofera hirsuta* volunteered to high overall frequencies within the saw-grass planting zone in 2007.

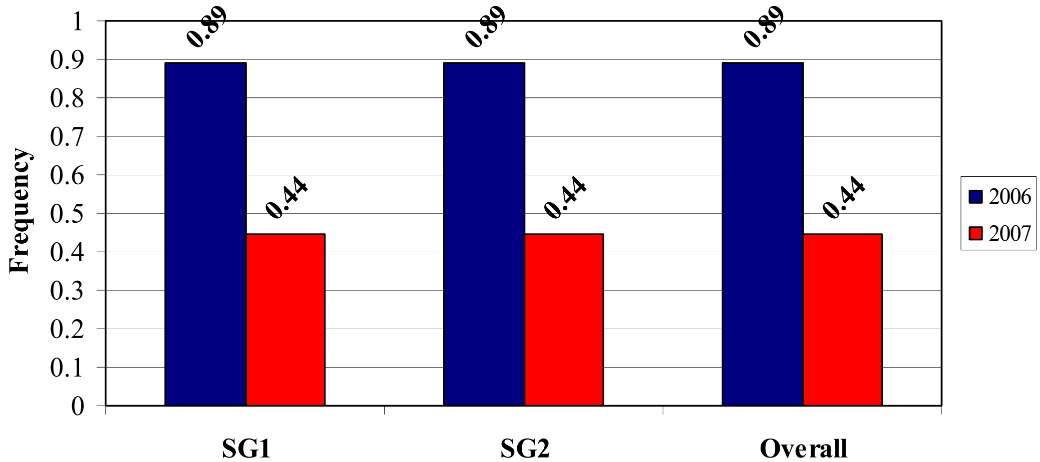


Figure 252. Saw-Grass Frequency at PPW-3.

Graminoid Planting Zone. *Spartina bakerii* (spartina grass), *Juncus effusus* (soft rush), *Muhlenbergia capillaris* (hairawn muhly), and *Bacopa caroliniana* (lemon bacopa) were planted within the graminoid planting zone. Initially, the four species were to be evenly distributed throughout the planting zone, but due to dry conditions on the day of planting and the small size of available *Bacopa caroliniana* individuals, planting for this species was concentrated along the wetter, eastern edge of the planting zone. All species experienced total declines in overall frequency, with no individuals present within either monitoring plot after one year (Figure 253). Only two *Muhlenbergia capillaris* individuals were observed within the entire planting area in 2007. Individuals of *Indigofera hirsuta* were present within GR1 and GR2 at low frequencies when the planting zone was first monitored in 2006. *Eupatorium capillifolium*, *Boehmeria cylindrica*, *Polygonum hydropiperoides*, *Lythrum alatum*, *Ambrosia artemisiifolia* (common ragweed), and *Cyperus virens* were present throughout the graminoid planting zone.

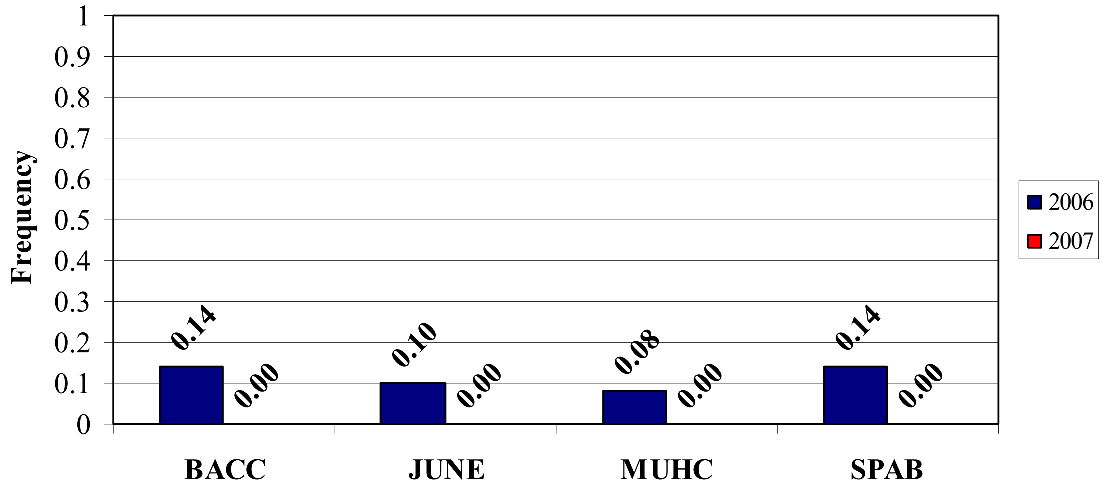


Figure 253. Graminoid Planting Zone Species Overall Frequency at PPW.

Tree Seedling Survival. Ten species of wetland trees were planted along the northern and eastern periphery of the PPW-3 marsh planting. Seedling survival is presented in Figure 254 and Table 32. Overall wetland tree seedling survival, an aggregate of the ten planted species, excluding *Taxodium distichum* and *Annona glabra* seedlings planted within the central portion of the marsh, after one year was 62%. *Cephalanthus occidentalis* (common button bush) (96%, n = 51) and *Persea palustris* (swamp bay) (95%, n = 40) had the best first-year survival of all planted species at the marsh periphery. *Gleditsia aquatica* (water locust) (90%, n = 54), *Fraxinus caroliniana* (pop ash) (79%, n = 49), *Taxodium distichum* (75%, n = 62), and *Itea virginica* (Virginia willow) (72%, n = 85) also survived well after one year. *Annona glabra* (6%, n = 61*),* *Hypericum fasciculatum* (peelbark St. John's wort) (25%, n = 43), and *Nyssa sylvatica var. biflora* (swamp tupelo) (27%, n = 54) had the lowest survival after one year. Percent survival after one year for *Taxodium distichum* and *Annona glabra*, within the central marsh planting, was 100% and 0%, respectively (Table 32). The remains of the *Annona glabra* seedlings were desiccated, broken, and sometimes uprooted.

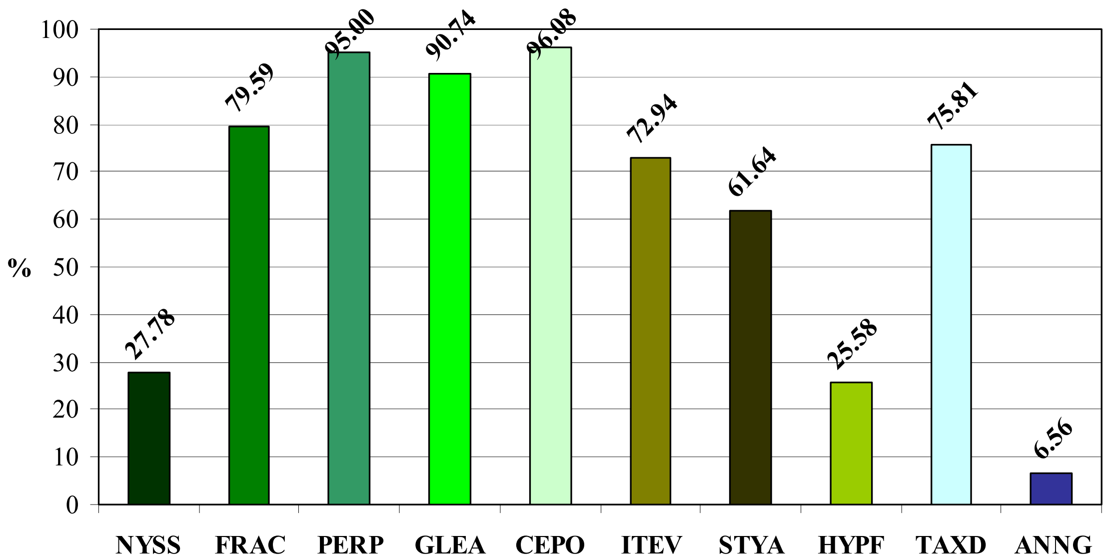


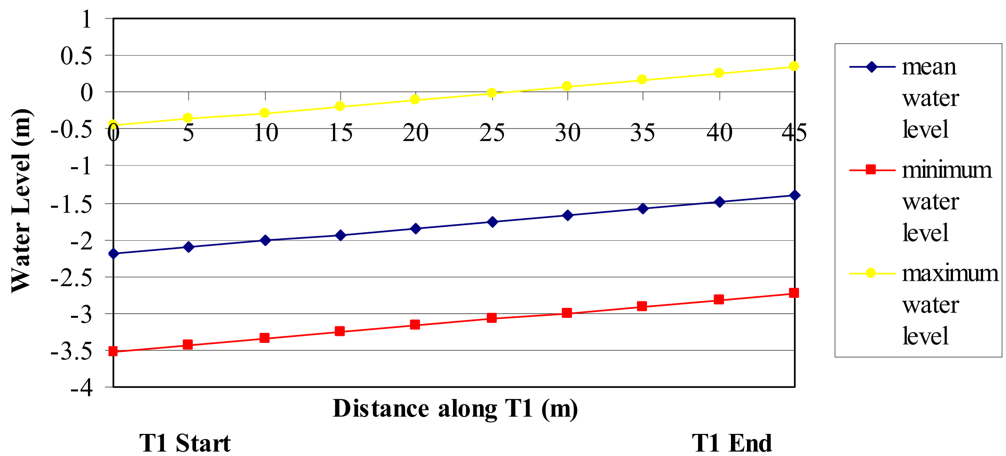
Figure 254. Percent Seedling Survival (%) at PPW-3.

Seedling Growth. Table 103 presents height data for seedlings at PPW-3. *Fraxinus caroliniana* had the largest increase in mean seedling height (cm) and *Itea virginica* had the highest % increase in mean height. *Nyssa sylvatica var. biflora* had the lowest positive increase in mean seedling height (cm) and % increase in mean height. *Taxodium distichum* and *Annona glabra* both experienced declines in mean seedling height (cm) between 2006 and 2007. Mean height for *Taxodium distichum* in the central marsh planting increased by 42%, from 88.3 cm ± 13.1 cm to 126.0 cm ± 17.8 cm ($\mu \pm \sigma$). Dieback of tree seedlings at PPW-3 may have been caused by one or more stressors, including root shock immediately after planting, drought stress, animal herbivory, or competition with dominant volunteer species within the tree seedling zone.

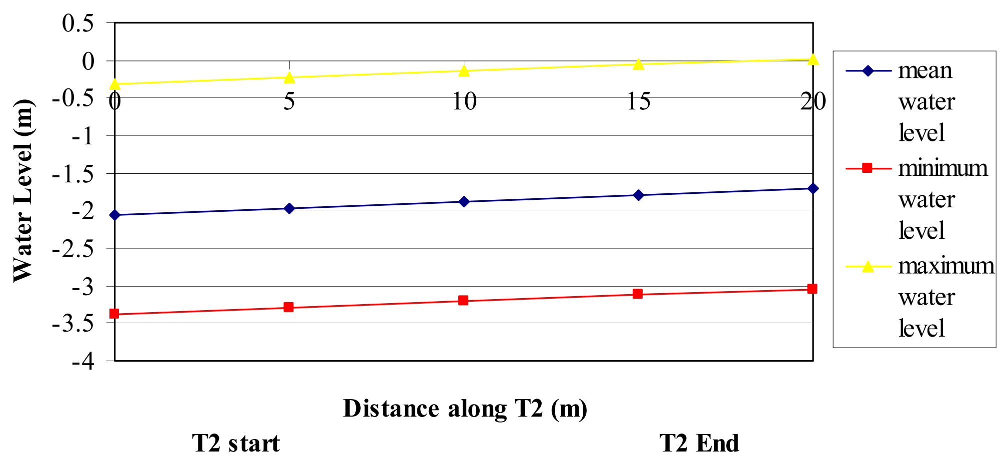
Table 103. Seedling Survival and Height at PPW-3.

Species	n	n Present in 2007	% Survival	% Decline in Survival	Average Height (06)	STDDEV Height (06)	STD ERROR 06	Min Height (06)	Max Height (06)	Average Height (07)	STDDEV Height (07)	STD ERROR (07)	Min Height (07)	Max Height (07)	Δ Mean Height (cm)	% Change Mean Height
NYSS	54	15	27.78	-72.22	53.09	8.20	1.12	40.00	85.00	53.93	8.70	2.25	35.00	67.00	0.84	1.58
FRAC	49	39	79.59	-20.41	81.60	19.58	2.80	30.00	115.00	125.51	41.43	6.63	55.00	229.00	43.91	53.81
PERP	40	38	95.00	-5.00	55.33	11.45	1.81	40.00	100.00	78.82	21.35	3.46	29.00	133.00	23.49	42.46
GLEA	54	49	90.74	-9.26	69.44	21.67	2.95	28.00	125.00	103.10	63.37	9.05	38.00	258.00	33.66	48.47
CEPO	51	49	96.08	-3.92	49.10	5.84	0.82	33.00	60.00	68.84	30.13	4.30	24.00	150.00	19.74	40.20
ITEV	85	62	72.94	-27.06	52.80	12.92	1.40	20.00	97.00	85.68	29.72	3.77	30.00	153.00	32.88	62.28
STYA	73	45	61.64	-38.36	42.21	9.09	1.06	25.00	65.00	47.36	23.34	3.48	21.00	170.00	5.15	12.20
HYPF	43	11	25.58	-74.42	44.33	11.76	1.79	22.00	75.00	54.91	10.77	3.25	35.00	71.00	10.58	23.88
TAXD	62	47	75.81	-24.19	85.39	11.56	1.4675	55.00	110.00	70.73	42.69	6.22693	20.00	156.00	-14.65	-17.16
ANNG	61	4	6.56	-93.44	87.34	13.02	1.67	50.00	125.00	80.50	39.97	0.00	25.00	138.00	-6.84	-7.84
OVERALL	572	359	62.76	-37.24												
TAXD Central	22	22	100.00	0.00	88.27	13.11	2.79	57.00	110.00	126.00	17.83	3.80	89.00	164.00	37.73	42.74
ANNG Central	21	0	0.00	-100.00	89.52	9.53	2.08	71.00	105.00	0.00	0.00	0.00	0.00	0.00	-89.52	-100.00

(a)



(b)



(c)

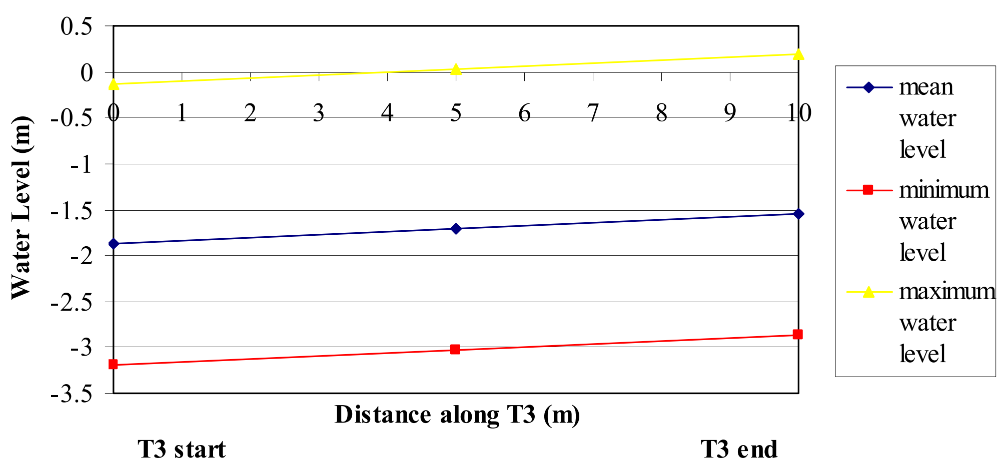


Figure 255. Water Level Data for (a) T1, (b) T2, and (c) T3 at PPW-3.

Seedling Survival, Hydrology, Substrate, and Volunteer Vegetation. *Taxodium distichum* and *Annona glabra* seedlings planted within the flag marsh and spike rush planting zones in the central portion of the marsh experienced hydroperiods similar to monitoring plots FM1, FM2, and SP2 (Table 100). Seedlings experienced average water levels ranging from -139.45 cm to -156.45 cm. Wetland hydroperiod within this planting area was not sufficiently wet enough to support *Annona glabra* seedlings. Survival of *Annona glabra* in the tree planting zone was slightly higher than individuals of this species planted in the central portion of the marsh; however, survival was very low at both sampling locations (Table 103). *Taxodium distichum*, however, thrived within this planting area and had higher survival than *Taxodium distichum* seedlings sampled in the tree planting zone.

Figure 255 presents water level data for the three belted transects used to monitor tree seedlings. Belted Transect 3 (T3) begins at a lower elevation than Transect 1 (T1) and Transect 2 (T2), and therefore has a higher average water level and most likely, greater soil moisture than at least the first 15 m of T1 and first 10 m of T2. In addition to increased water availability, volunteer vegetation was less dense at T3 and had a higher sand content than T1 or T2. Six planted species had higher % survival at T3 than T2 or T1 (Table 104). Four of seven species, present at all three transects, had higher mean growth at T3. Increase in mean height was low at all three transects for *Annona glabra*, due to low percent survival, and *Taxodium distichum* as a result of dieback and breakage of seedlings.

Table 104. Seedling Survival by Transect at PPW-3.

Species	T1	T2	T3
NYSS	37.50	7.69	22.22
ANNG	7.89	5.56	0.00
FRAC	83.33	54.55	86.67
PERP	95.00	93.33	100.00
TAXD	72.73	76.92	100.00
GLEA	92.68	91.67	N/A
CEPO	94.74	85.71	100.00
ITEV	77.78	58.82	100.00
STYA	63.33	55.56	85.71
HYPF	33.33	7.14	N/A

Note: Species with the highest survival are highlighted in red.

Seedling survival and growth were graphed against the average water level experienced by seedlings. Graphs are found in Appendix C, excluding those for *Persea palustris* and *Fraxinus caroliniana*. Both *Persea palustris* and *Fraxinus caroliniana* survived well along the entire gradient, and both had weak, positive correlations of change in height with increasing average water levels (0.34 and 0.31, respectively), although the coefficients of determination (r^2) were low (Figures 256 and 257). Survival and growth were low for *Annona glabra* and *Hypericum fasciculatum* along the entire hydrologic gradient within the tree planting zone, which ranged in average water level from -219.5 cm at the driest end to -146.5 cm at the wettest. *Nyssa sylvatica var. biflora* had slightly higher survival at the wetter end of the gradient, but percent survival and growth for the sample was very low. *Styrax americana* and *Taxodium distichum* survived along the entire gradient, but growth for both species was generally low. No correlation was found between average water levels and seedling growth for *Gleditsia aquatica*, *Cephalanthus occidentalis*, and *Itea virginica*.

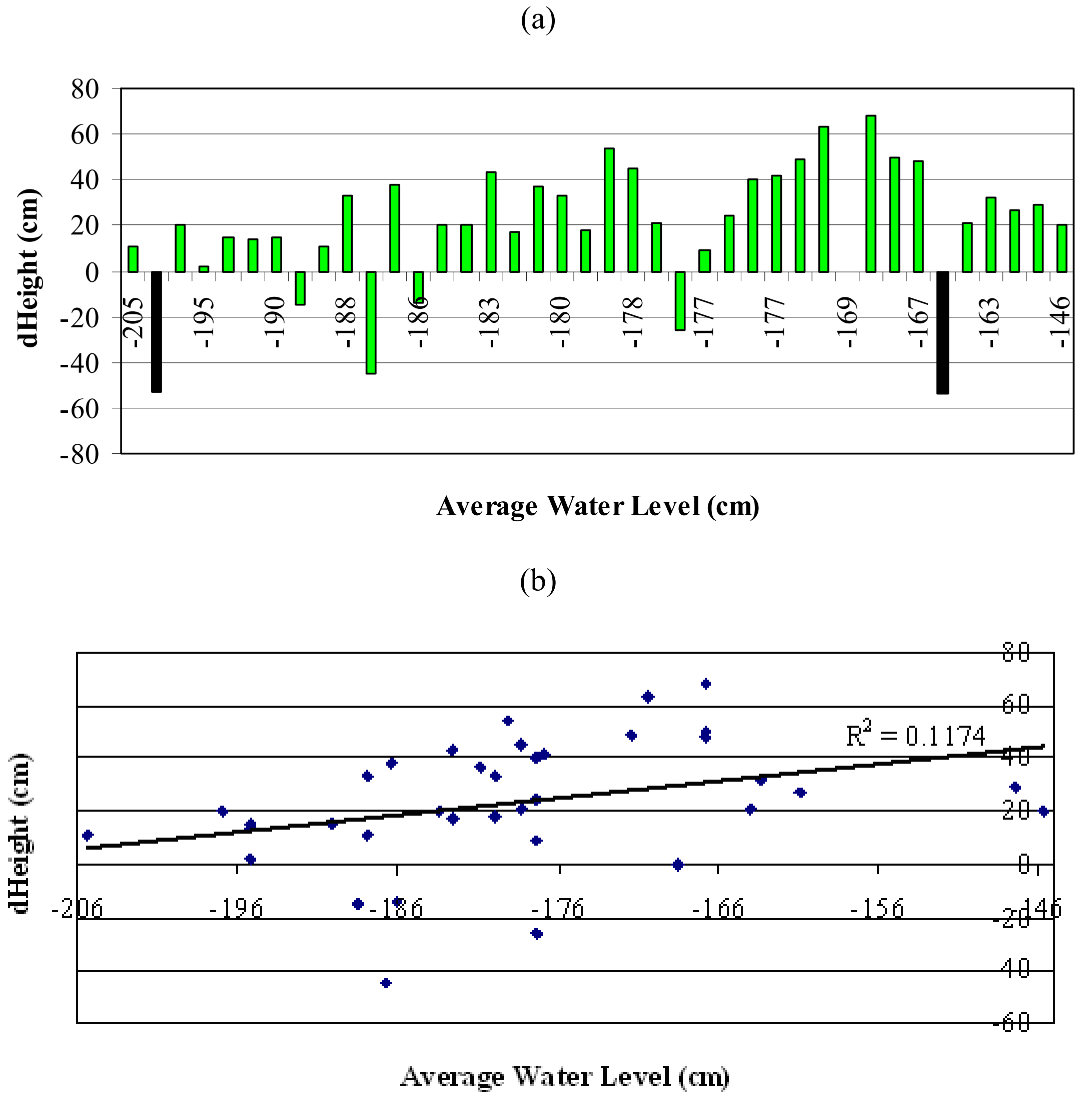
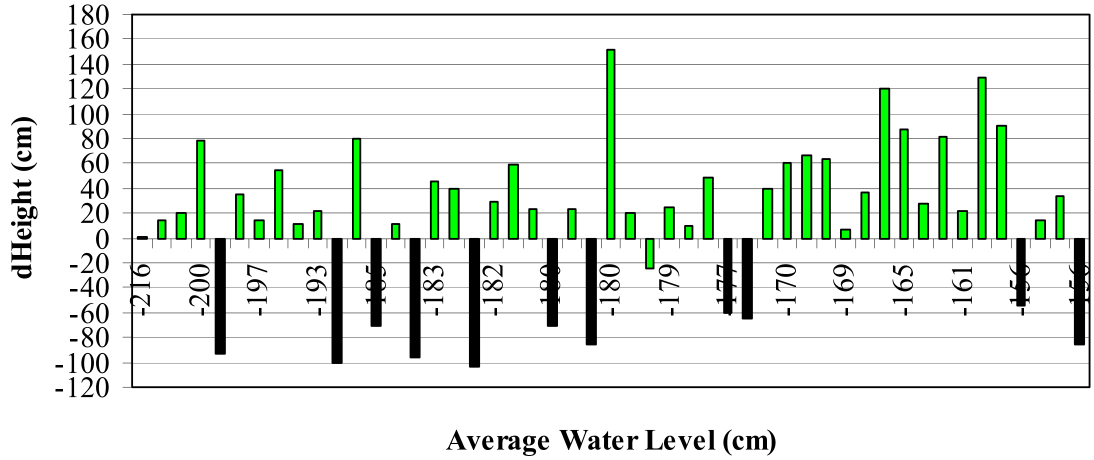


Figure 256. *Persea palustris* Survival and Growth at PPW-3.

(a)



(b)

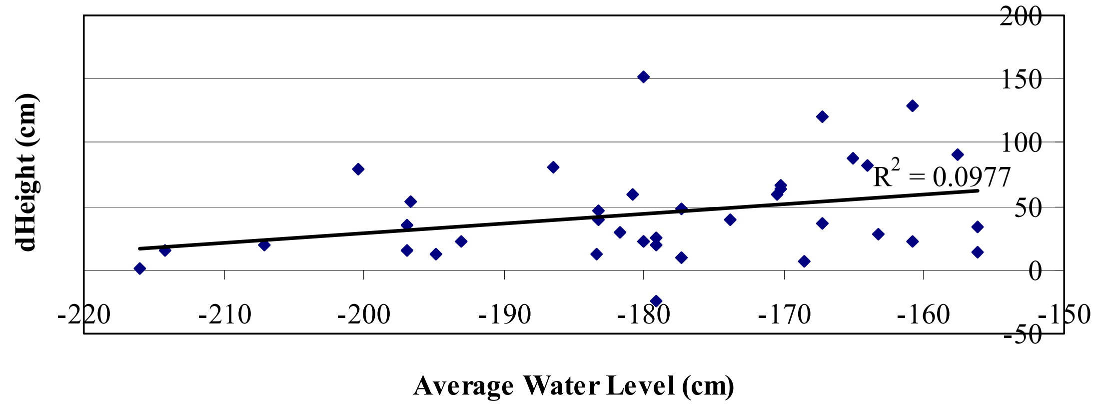


Figure 257. ***Fraxinus caroliniana*** **Survival and Growth at PPW-3.**

Indigofera hirsuta (hairy indigo) was observed growing throughout the tree planting zone during the 2006 growing season. This species is an erect, reseeding summer annual legume that can reach heights of 121 cm to 213 cm if not grazed, typically occurring in upland ecosystems (Chambliss and Ezenwa 2002). At PPW-3, the maximum height of *Indigofera hirsuta* ranged 130 cm at lower elevations to 200 cm at higher elevations along the eastern edge of the planting zone. The species was observed pinning some seedlings to the ground surface and causing breakage of others; however, many tree seedlings appeared to be thriving. At the end of the 2007 growing season, *Eupatorium capillifolium* (dog fennel) had volunteered across the entire tree planting zone and *Indigofera hirsuta* was no longer dominant. Since *Indigofera hirsuta* can easily survive upland conditions similar to those at PPW-3 over the period of record, competition with *Eupatorium capillifolium* may have eliminated the majority of the population over the 2007 growing season. Along T1, *Eupatorium capillifolium* ranged in height from 100 cm to approximately 400 cm, with stem density exceeding 50 stems per square meter. Height and density ranged between 100 cm and 200 cm at T2 and T3, with a variety of other species present, including *Baccharis halimifolia* (eastern baccharis), *Indigofera hirsuta*, *Sambucus canadensis* (elderberry), *Ambrosia artemisiifolia* (common ragweed), *Phytolacca americana* (pokeweed), *Polygonum hydropiperoides* (swamp smartweed), *Sambucus canadensis*, *Ludwigia peruviana* (Peruvian primrose willow), *Passiflora incarnata* (purple passion flower), and *Lythrum alatum* (winged loosestrife).

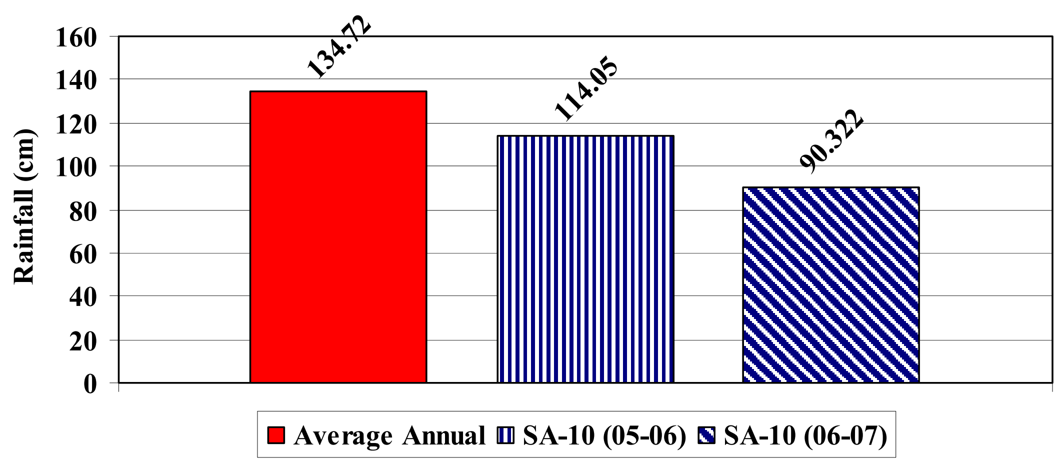
Seedling Underplanting Sites

Hydrologic conditions at the seedling underplanting sites are summarized and compared. Underplanting sites are further characterized through the analysis of substrate and existing canopy and understory vegetation. Seedling survival is explored at each site, along each site's hydrologic gradient, as well as between sites. Height data is examined for each site and differences in growth are evaluated between sites. The relationship between seedling growth and hydrology is explored.

Site Characteristics

Precipitation. Annual, as well as monthly, precipitation varied among the sites, although all three were affected by drought conditions lasting from planting in 2006 through monitoring in 2007. Monthly and yearly rainfall amounts at each site are presented in Figures 258-260. Total precipitation for the 2005-2006 (08/01/2005-07/31/06) and 2006-2007 (08/01/2006-07/31/2007) rain years is lower than the historic annual average for all three underplanting sites. At SA 10, both years are below 134.72 cm, although the 2006-2007 year is lower than the preceding year by approximately 24.00 cm. This same trend in annual rainfall can be seen at the H1u site. At TEN-1, total precipitation for 2005-2006 and 2006-2007 are roughly the same, with both totals approximately 38.00 cm below the historic annual average. Growing season months in 2007 for all three sites were almost all lower than historic average rainfall for those months.

(a)



(b)

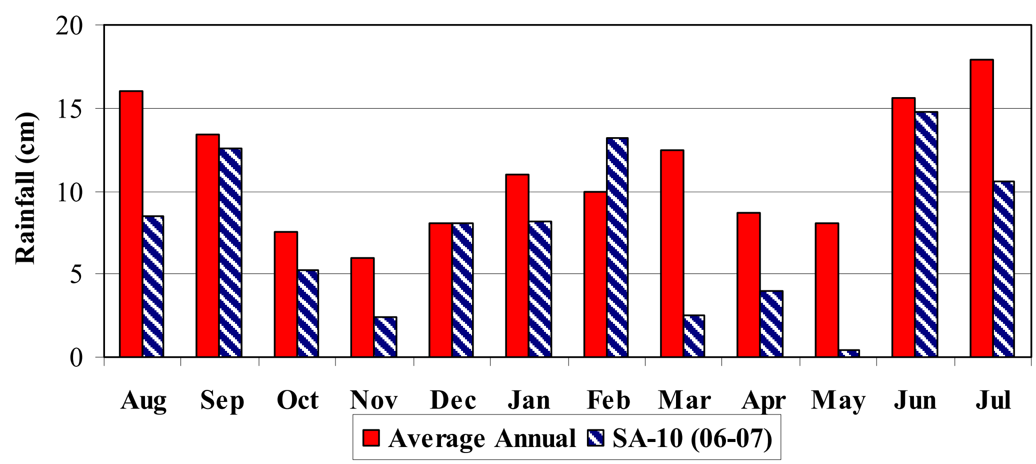


Figure 258. (a) Annual and (b) Monthly Precipitation Totals at SA 10.

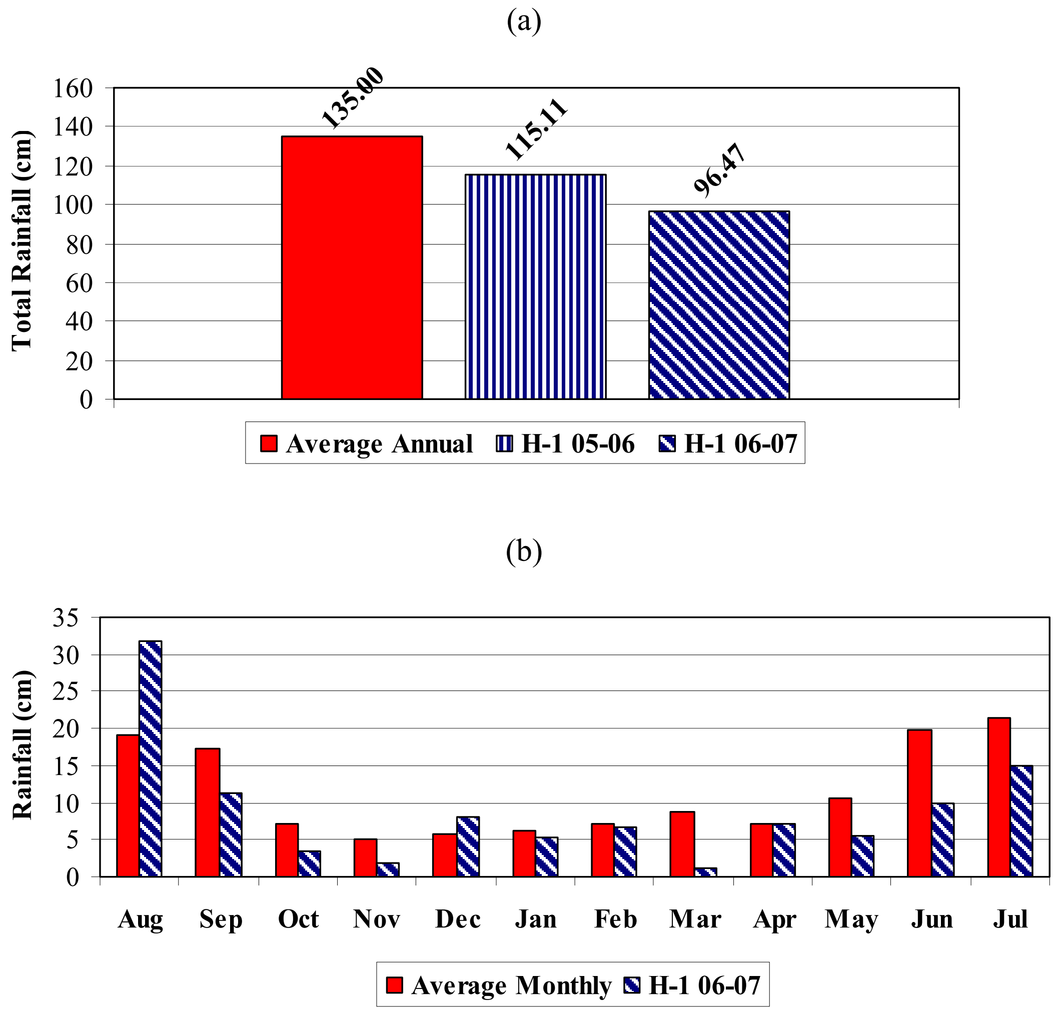
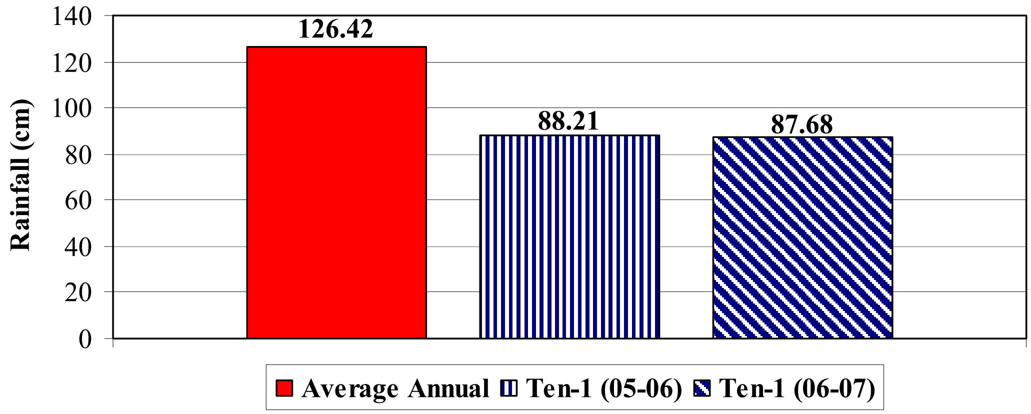


Figure 259. (a) Annual and (b) Monthly Precipitation Totals at H1u.

(a)



(b)

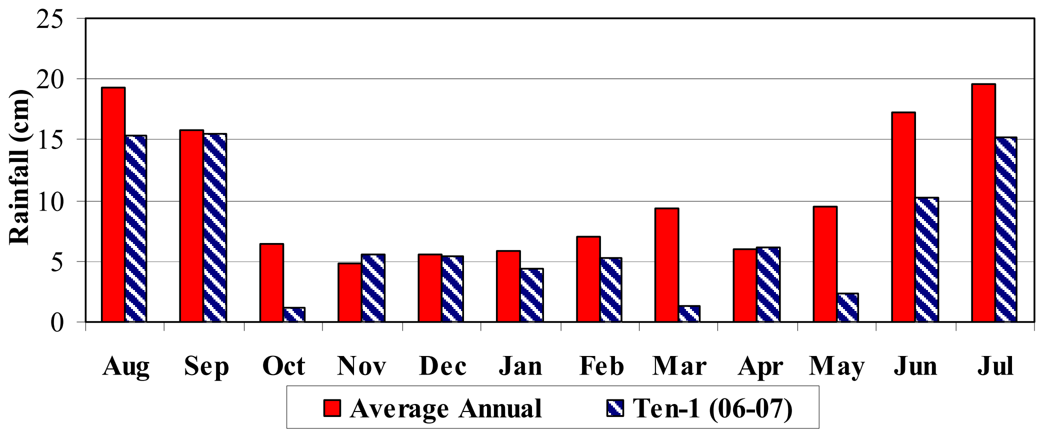


Figure 260. (a) Annual and (b) Monthly Precipitation Totals at TEN-1.

Wetland Hydrology. Hydrologic conditions differ between the three underplanting sites, Hooker's Prairie 1 (H1u), PCS SA 10 (SA 10), and Tenoroc-1 (TEN-1), due to each site's unique wetland water budget and topography. The underplanting at SA 10 took place at the low-slope, wetland edge of a permanently ponded wet feature with an average slope of 0.02. Figure 261 displays water level fluctuations at the surface water well between August of 2006 and July of 2007. Average monthly water levels at the surface water well located in the ponded feature are lower from 2006-2007 than the previous year's due to a lack of rainfall in the region.

Water levels along transects through the planting area are assumed to be directly connected with water levels at the surface water well. The height of water at a point along the transect is the surface water level minus the change in elevation between the well and corresponding transect point. Average water levels, along the three transects used to characterize wetland hydroperiod at SA 10, are presented in Figures 262, 263, and 264. Table 105 lists the planting zone and tree species that coincide with each transect. Average water levels at the driest portion of the site never exceeded -1.6 m in depth below the ground surface. Small hummocks in the ground surface account for the uneven slope of water levels along transects. Inundation over the period of record (July 2006 through July 2007) only occurred for the first 8 m of Transect 1 (T1), the first 3 m of Transect 2 (T2), and the first 0.5 m of Transect 3 (T3), Transect 5 (T5), and Transect 6 (T6). No aboveground inundation occurred at the site during the 2007 growing season; however, the substrate at SA 10 appeared moist at every site visit. The low slope gradient allowed for saturation and available moisture within the seedlings' root zone over a larger portion of the planting area, especially during the growing season when available moisture is critical. The root zone for tree seedlings was assumed to be the top 0.5 m of soil. When groundwater levels were within -0.5 m of the ground surface, it was assumed that the capillary fringe that exists in mainly clayey soils would cause the soil to be saturated to at least -0.25 m, the planting depth for seedlings, thus saturating the seedlings' root zone. This is most likely an underestimation of the capillary fringe depth, with published values estimating depth in clayey soils to be between 0.5 m and 1 m (Dingman 2002, Bell 2004).

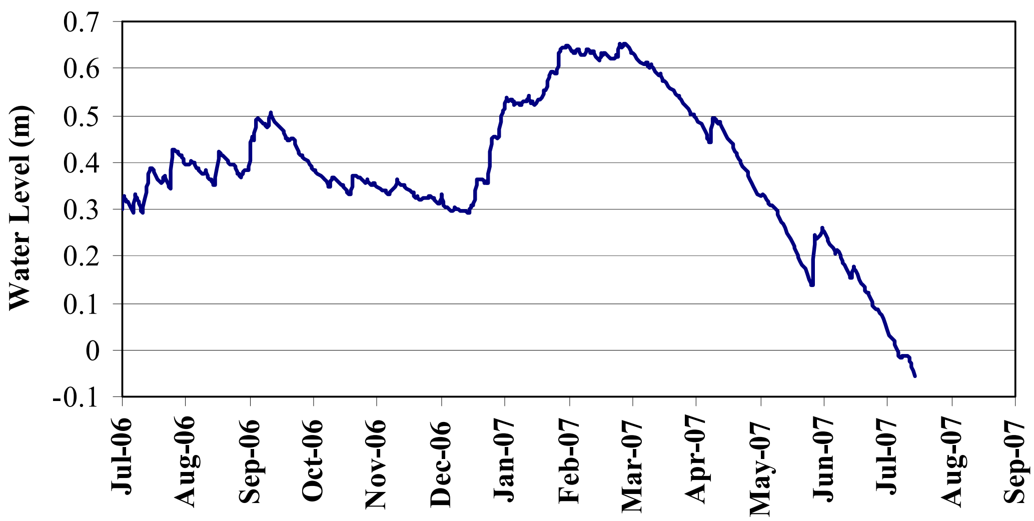


Figure 261. Water Levels at the SA 10 Surface Water Well.

Table 105. Transects, Planting Zones, and Tree Species at (a) SA 10, (b) H1u, and (c) TEN-1.

Transect	Zone	Species
1	1	ITEV
	1	TAXD
	1	LIQS
	1	TAXA
2	1	ITEV
	1	TAXD
	1	TRAP
3	1	FRAP
	1	CEPO
	2	ULMA
	2	CARA
4	2	BETN
	2	CARA
	2	QUEL
5	2	NYSS
	3	PLAO
	3	LIRT
	3	CELL
	3	MAGV
6	3	PLAO
	3	MAGV
	3	CORF
	3	QUEM
	3	ILEC
7	3	QUEM
	3	ILEC
	3	NYSA
	3	QUEN

(a)

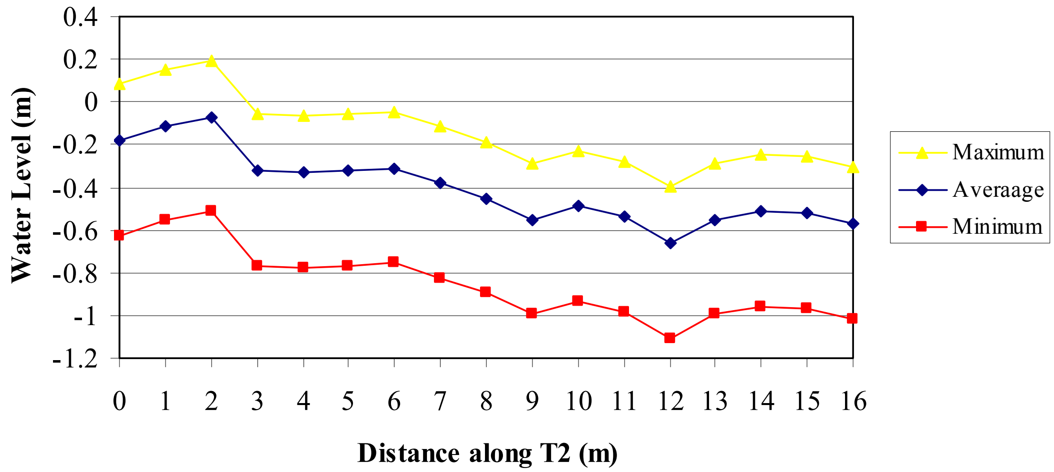
Transect	Zone	Species
T1	1	FRAC
	1	NYSS
	1	TAXD
T2	2	CELL
	2	ULMA
	2	ILEC
T3	3	QUEN
	3	LIQS
	3	SABM
	3	MAGV
	3	CARA

(b)

Transect	Zone	Species
T1	1	TAXD
	1	NYSS
	1	FRAC
	1	ITEV
	1	TAXA
T2	2	CELL
	2	ULMA
	2	QUEL
	2	CARA
	2	ILEC
	2	BETN
T3	3	QUEN
	3	SABP
	3	CORF
	3	QUEM
	3	LIRT
	3	LIQS
	3	MAGV

(c)

(a)



(b)

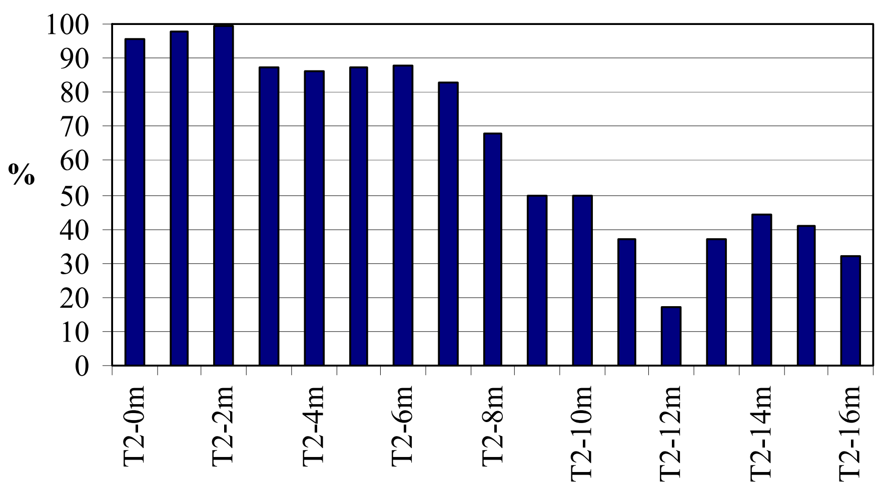
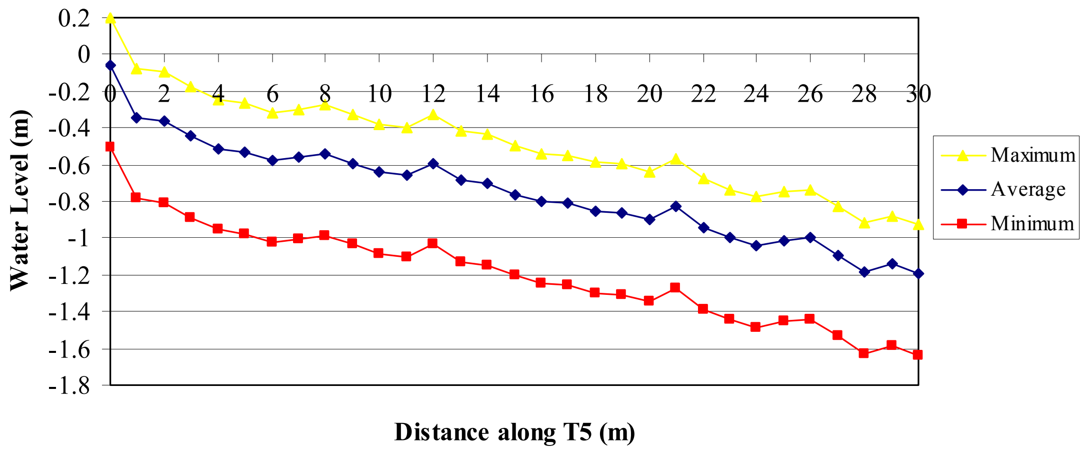


Figure 262. T2 (a) Average Water Levels and (b) Root Zone Inundation (%) at SA 10.

(a)



(b)

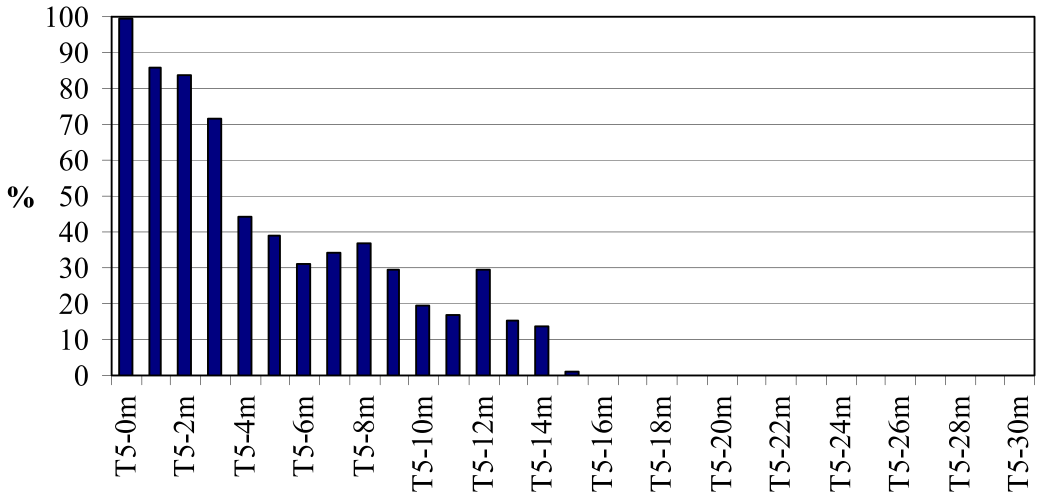
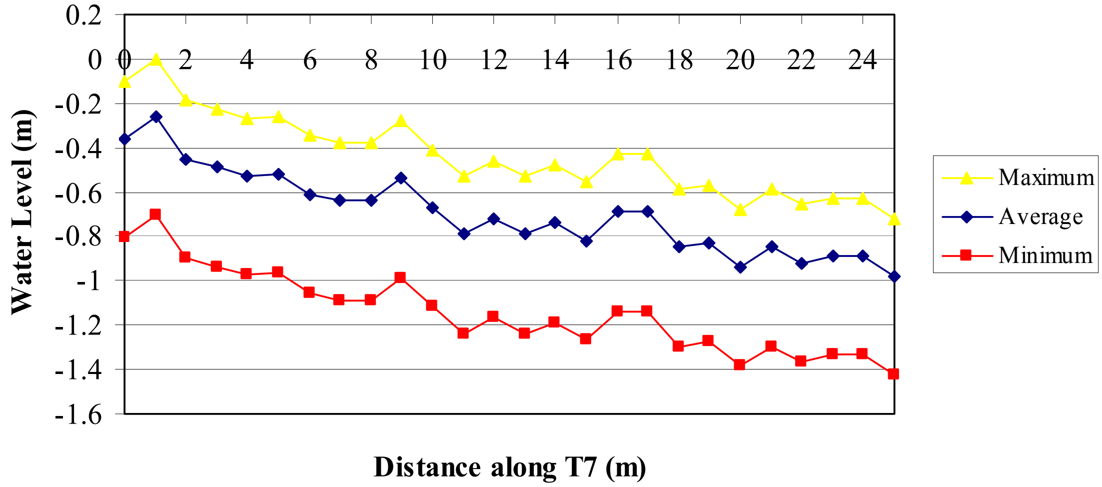


Figure 263. T5 (a) Average Water Levels and (b) Root Zone Inundation (%) at SA 10.

(a)



(b)

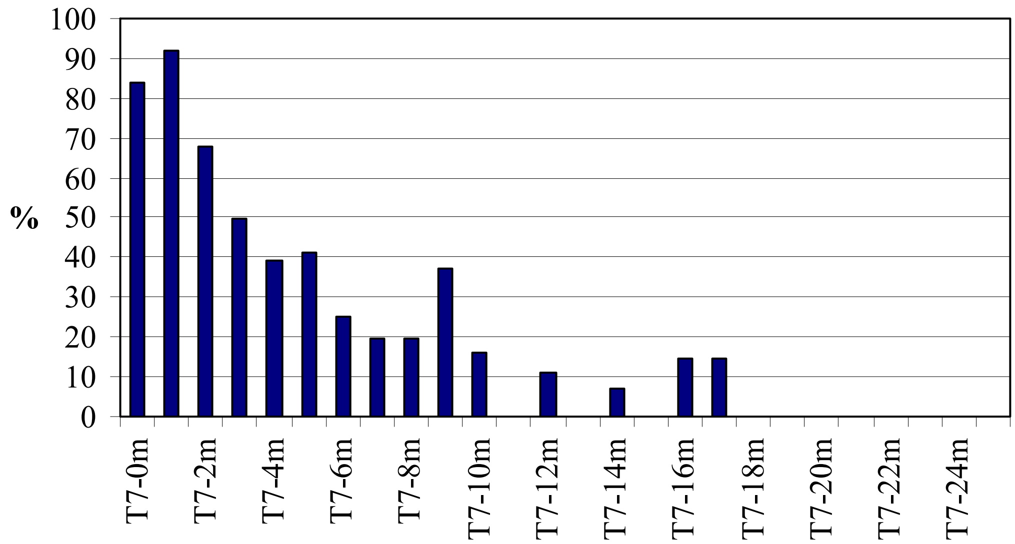


Figure 264. T7 (a) Average Water Levels and (b) Root Zone Inundation (%) at SA 10.

The underplanting at H1u experienced hydrologic conditions similar to SA 10, and its topography is characterized as well by a low slope of 0.02 and hummock features within the planting area. The H1 underplanting site is located within a swath of wetland area that runs along the northern dike of the CSA. Water backs up along the edge of the dike, flooding into the planting area. The surface water well is located north of the underplanting in a deeper wetland area that dries down at times throughout the year due to a lack of connectivity with other wetland areas on the CSA and climatic conditions. Water levels at the H1 surface water well are presented in Figure 265. Like SA 10, the surface water well was inundated during and after planting at the end of the 2006 growing season. The water level fluctuated during the 2007 growing season, with the well inundated once between March and July of 2007. Sharper declines and rebounds in water levels contrast with steadier increases and decreases seen at SA 10.

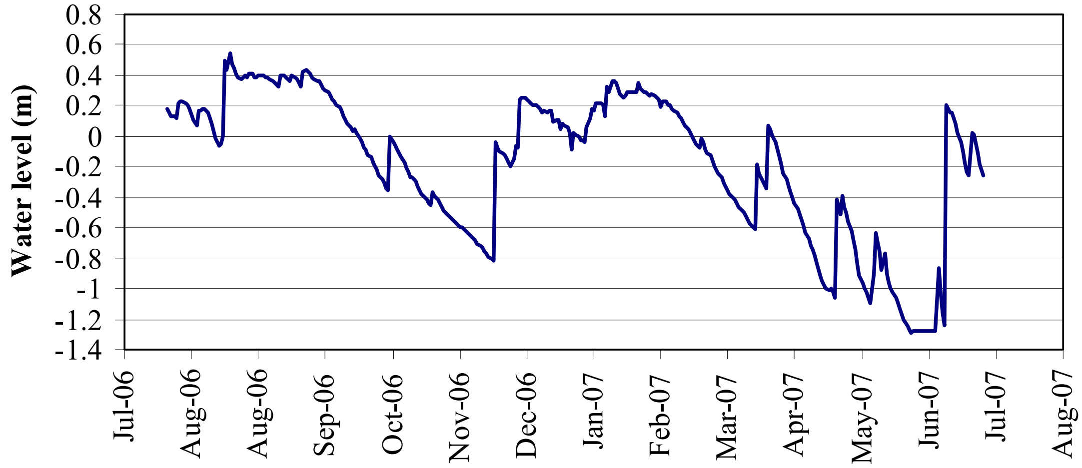
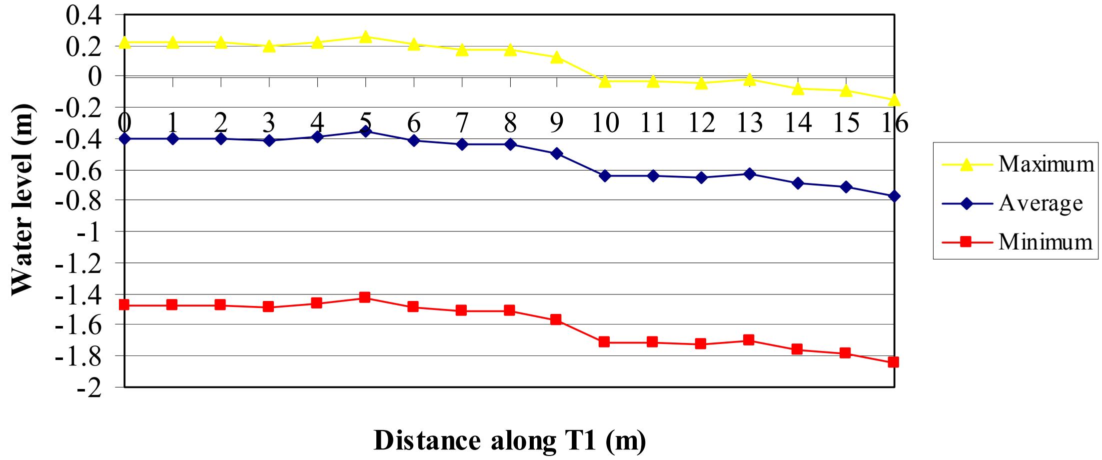


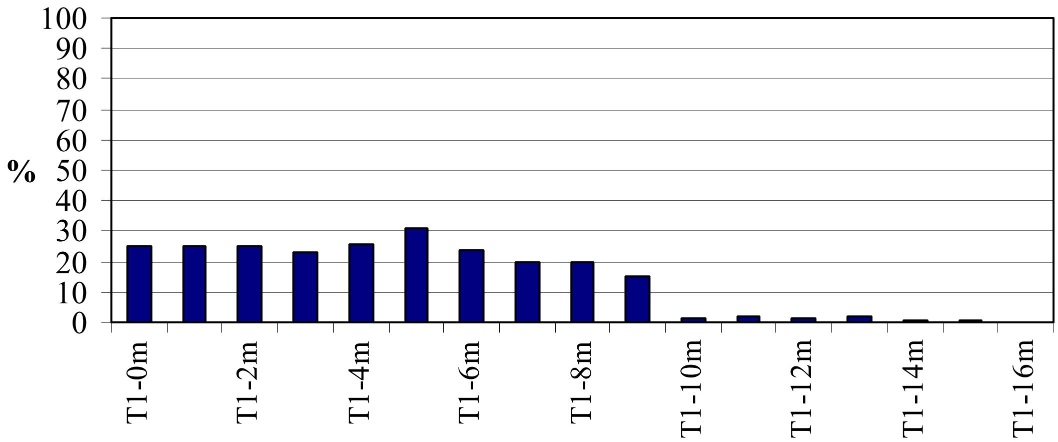
Figure 265. Water Levels at H1u Surface Water Well.

Average water levels as well as above-ground and root-zone inundation for the three transects used to characterize the three planting zones at H1u are presented in Figures 266, 267, and 268. T1, used to characterize Zone 1, experienced above-ground inundation along its entirety, with the exception of the driest portion of the planting zone. The low slope within Zone 1, a change in elevation of only 0.4 m, accounts for similar hydrologic conditions with the entire zone. Average water levels and root-zone inundation along T2, used to characterize Planting Zone 2, were similar to T1, but the zone experienced less above-ground inundation. Average water levels and root-zone inundation within Zones 1 and 2 at H1u are similar to those of Zones 1 and 2 at SA 10. Zone 3 at H1u experienced almost no above-ground inundation, with average water levels slightly lower than in Zones 1 and 2. Table 34 lists the planting zone and tree species that coincide with each transect at the H1u.

(a)



(b)



(c)

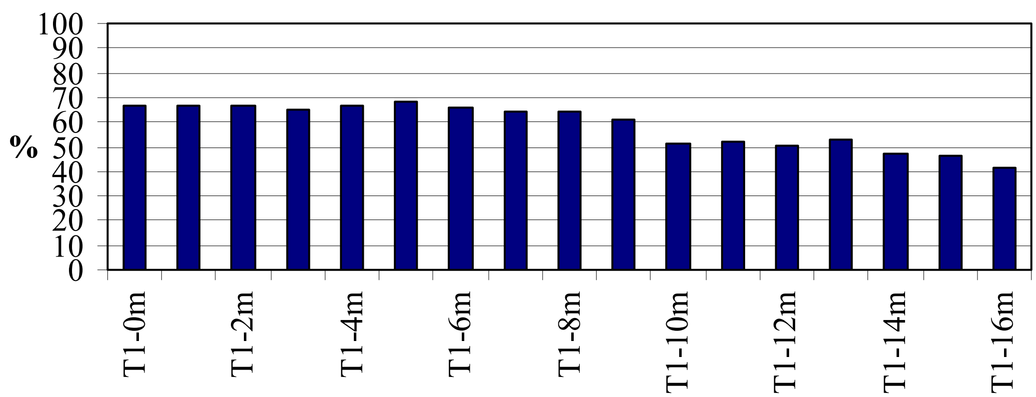


Figure 266. T1 (a) Average Water Levels, (b) Inundation (%), and (c) Root Zone Inundation (%) at H1u.

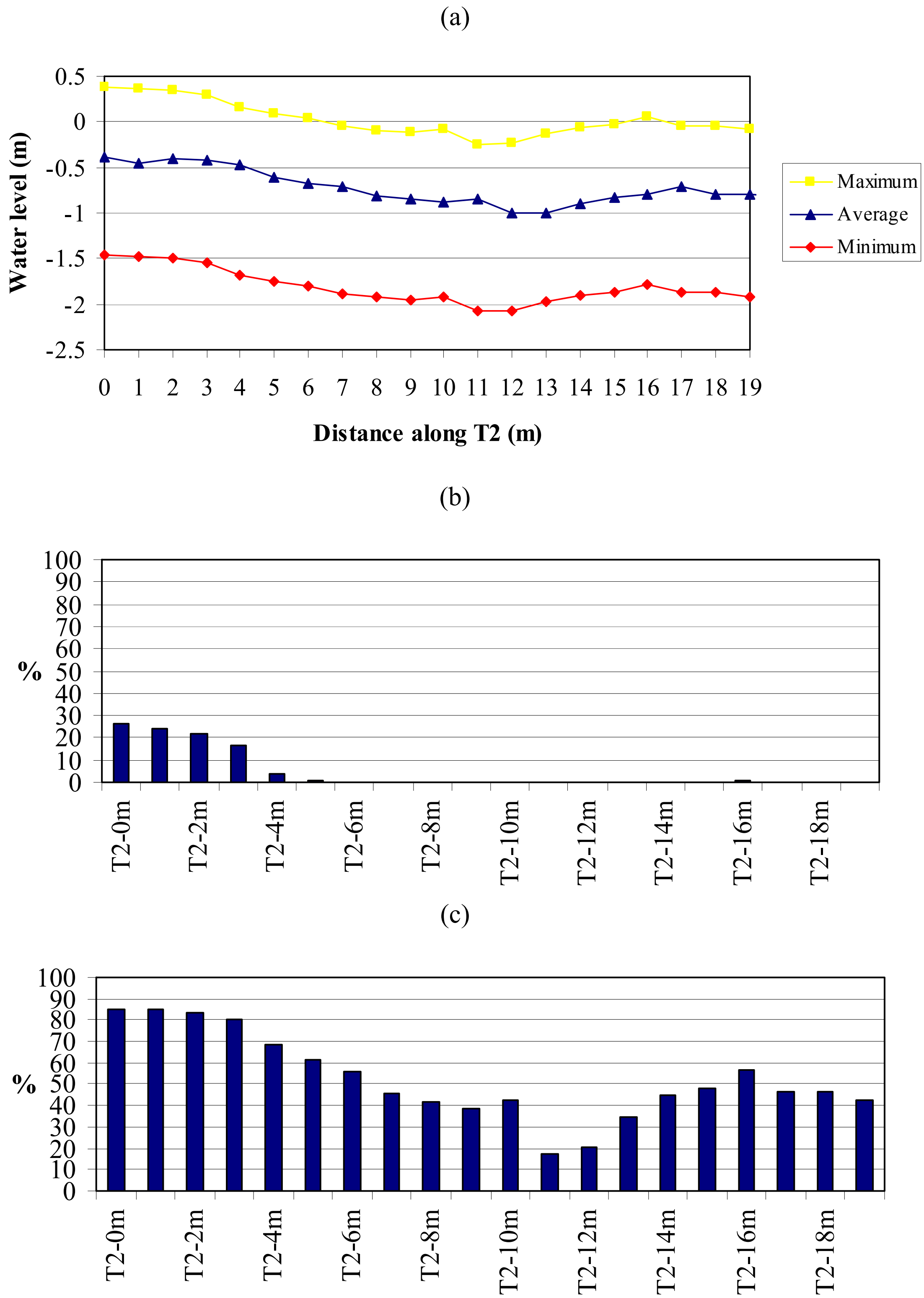
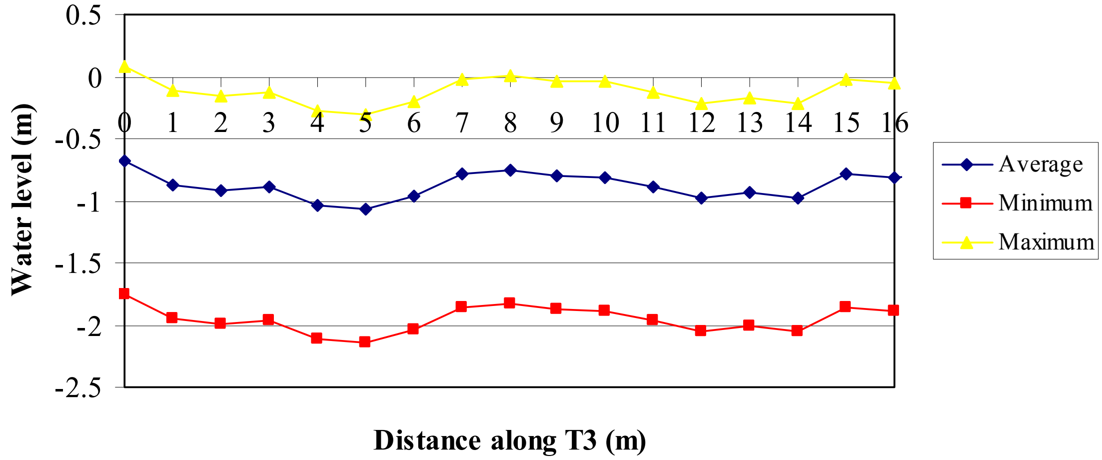
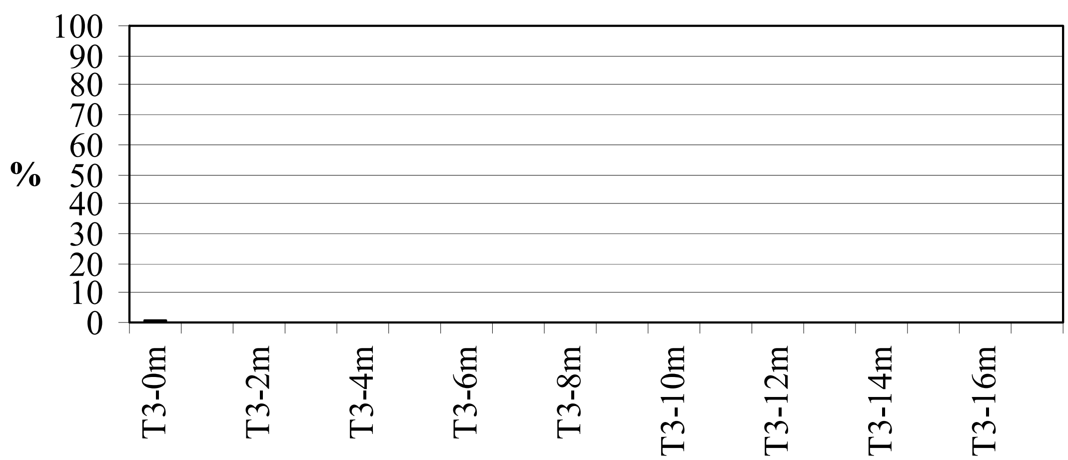


Figure 267. T2 (a) Average Water Levels, (b) Inundation (%), and (c) Root Zone Inundation (%) at H1u.

(a)



(b)



(c)

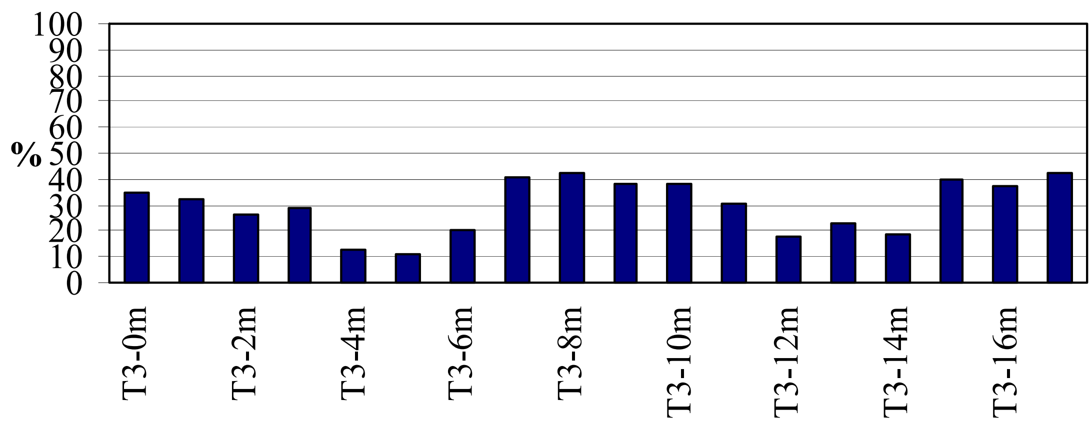


Figure 268. T3 (a) Average Water Levels, (b) Inundation (%), and (c) Root Zone Inundation (%) at H1u.

The underplanting at TEN-1 took place along a fairly steep gradient that stretches from a *Typha latifolia* marsh to the inside wall of the CSA dike. Figure 269 displays the water levels at the surface-water well located at the lowest elevation within the marsh. Like SA 10 and H1u, the surface-water feature was flooded at the end of the 2006 growing season, as well as between January and April of 2007, but not nearly to the depths seen at SA 10 or H1u. After a slight inundation occurred in April, water levels declined much faster and to much greater depths compared with SA 10 (Figure 261). Increases in water levels following rain events are also more pronounced, possibly due to infiltration and drainage from the CSA dike, which contains a higher proportion of sand than the clayey soils inside the CSA. Since species were planted along a steeper gradient and the adjacent water feature was so infrequently wetted, most average water levels along transects were lower at TEN-1 than those at SA 10 and H1u.

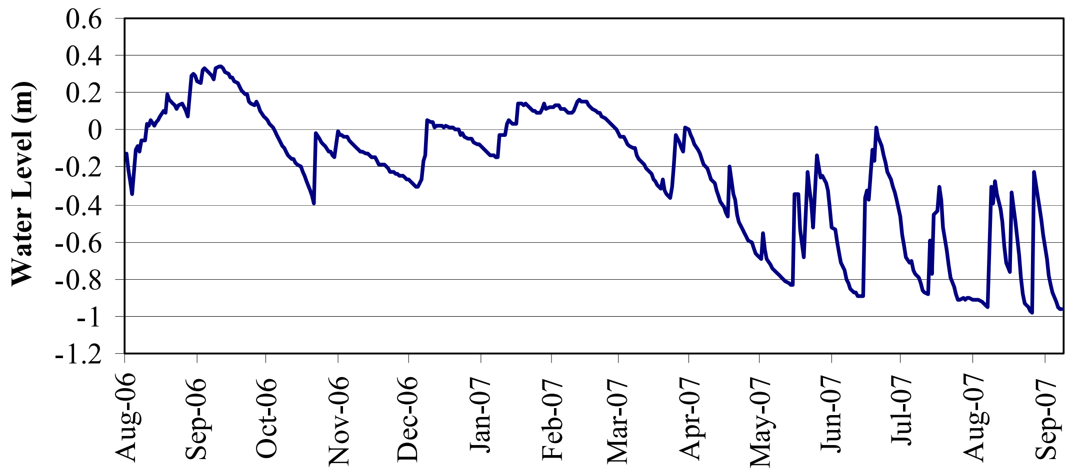
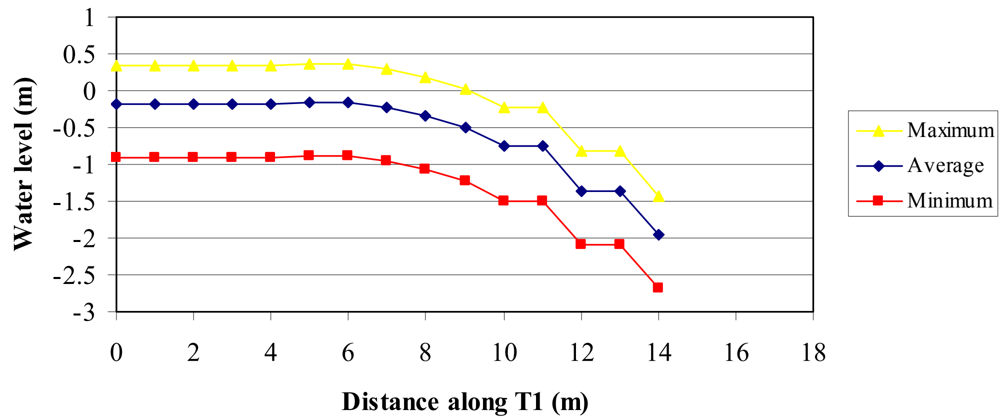


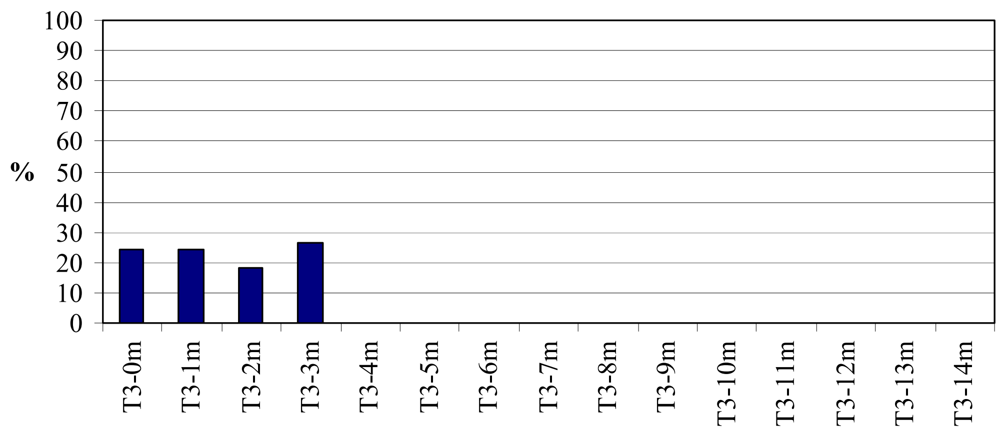
Figure 269. Water Levels at TEN-1 Surface Water Well.

Each of the three transects used to characterize the three planting zones at TEN-1 encountered the toe of slope associated with the dike wall (Figures 270, 271, and 272). This caused wetter conditions at the beginning of each transect, with a sharp transition to drier conditions as the planting zone encountered the dike. The first half of Transect 1 (T1), used to characterize Zone 1 at TEN-1, experienced average water levels within -0.5 m of the ground surface, some above-ground inundation, and root-zone inundation for the majority of the year and growing season. These conditions are reflective of those seen at Zone 1 of SA 10 and H1u. Inundation is limited after 8 m along T1. T2 and T3, which were used to characterize Zones 2 and 3 at TEN-1, encountered the toe of slope sooner than T1 and thus much drier conditions.

(a)



(b)



(c)

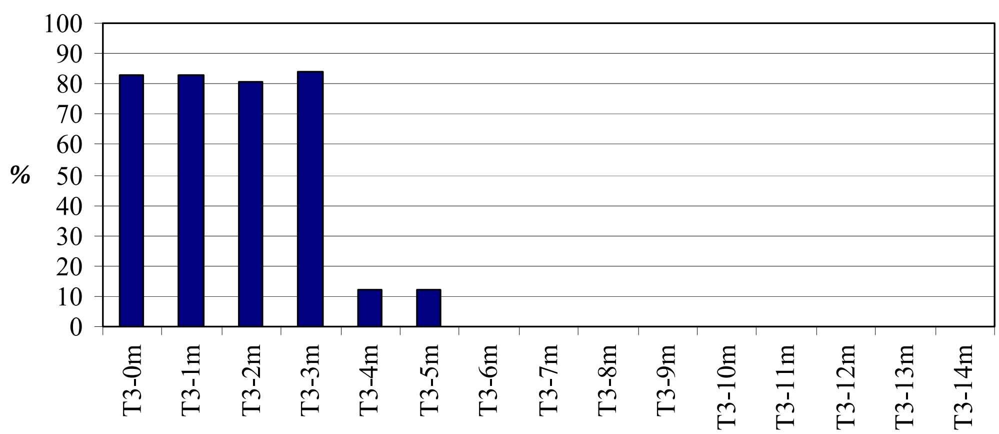
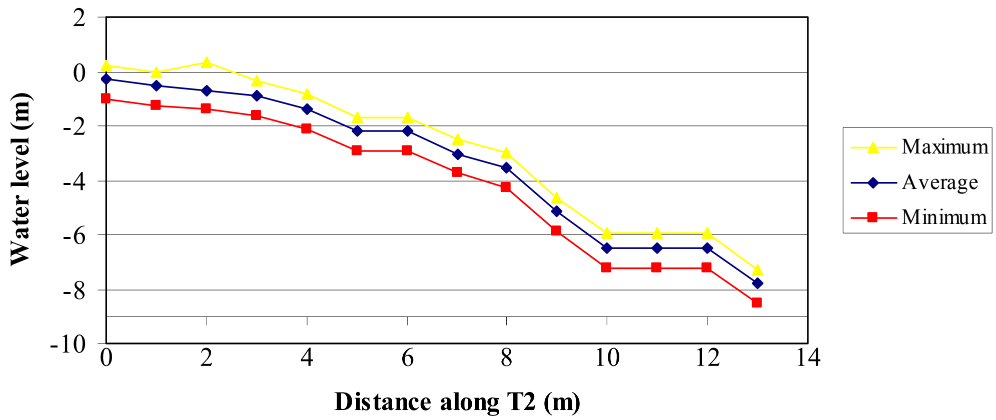
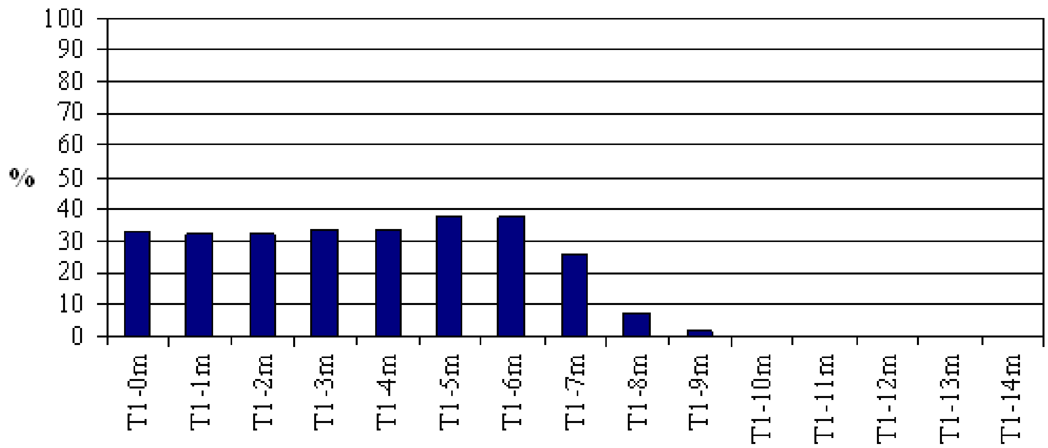


Figure 270. T1 (a) Average Water Levels, (b) Inundation (%), and (c) Root Zone Inundation (%) at TEN-1.

(a)



(b)



(c)

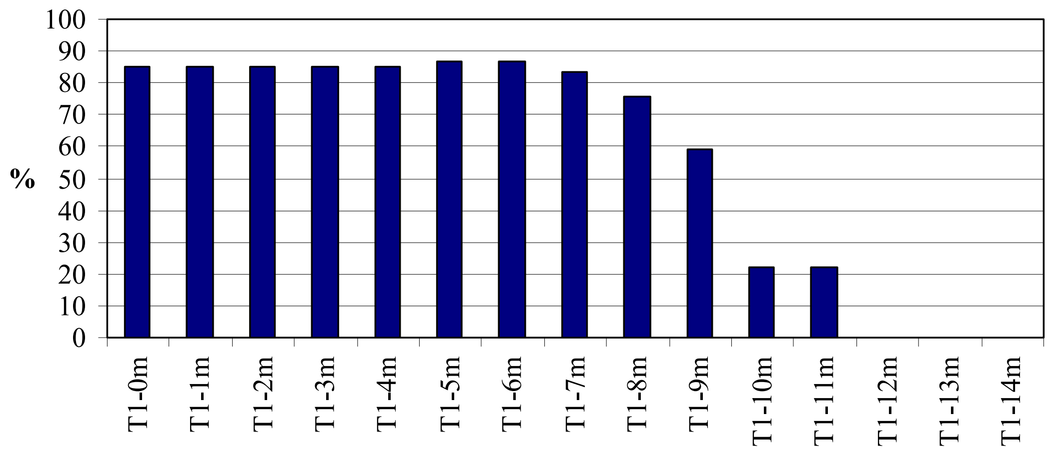
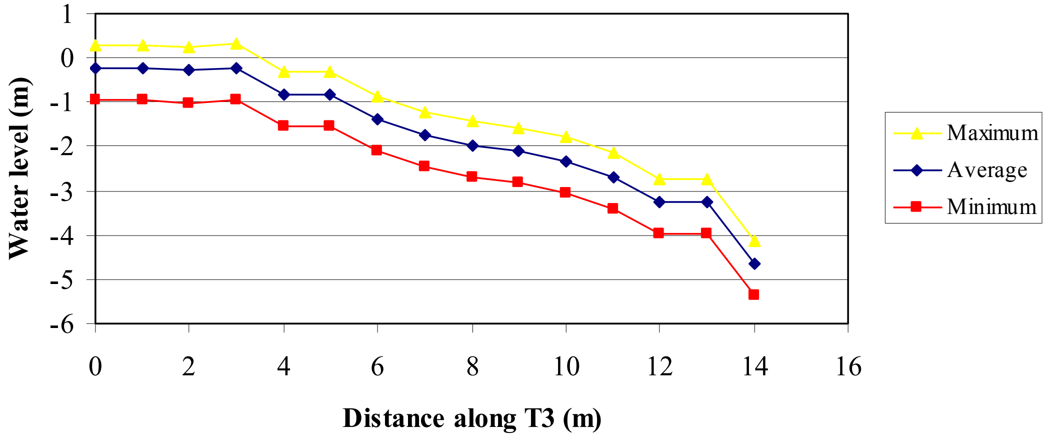
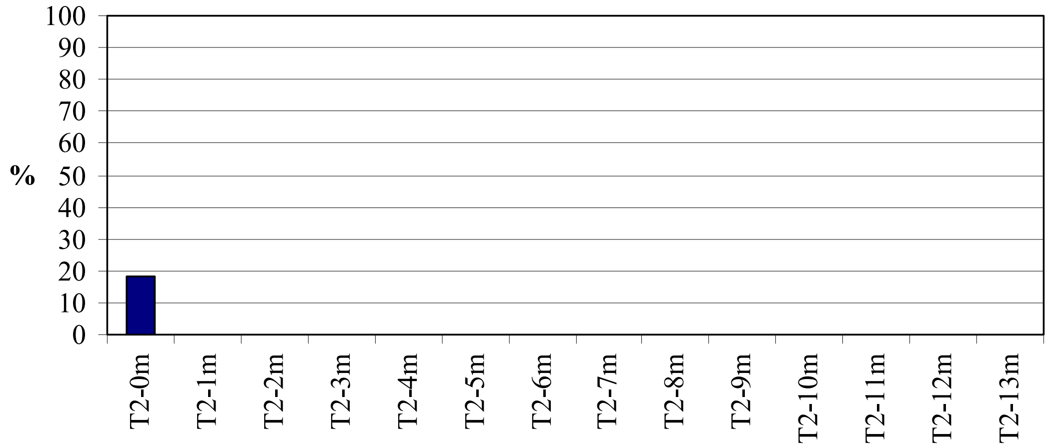


Figure 271. T2 (a) Average Water Levels, (b) Inundation (%), and (c) Root Zone Inundation (%) at TEN-1.

(a)



(b)



(c)

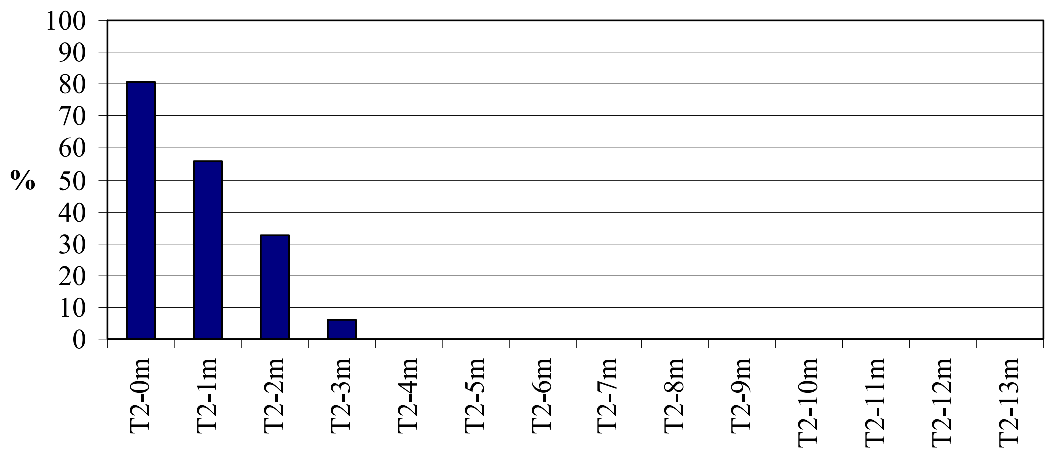


Figure 272. T3 (a) Average Water Levels, (b) Inundation (%), and (c) Root Zone Inundation (%) at TEN-1.

Substrate. The majority of the SA 10 underplanting site is classified as clay, with percent sand ranging between 16% and 24% (Table 106). Particle size distributions for samples taken at H1u are listed in Table 107. Like SA 10, the majority of this underplanting site was classified as clay, with percent (%) sand ranging between 17% and 32%. The underplanting at TEN-1 is located at the base of an overburden dike, where overburden material from the mining process meets the clay fill, thus sand makes up the majority of each sample across the planting zones (Table 108). Because of high sand content in the substrate, the soil drains more quickly after saturation and may have a narrower capillary fringe than mainly clay soils. Water levels after rainfall events may be more pronounced due to drainage from the adjacent sand dike that may continue well after the event has ceased. This can be seen in Figure 269 during the 2007 growing season.

Table 106. Particle Size Distribution at SA 10.

Sample Location	Soil Type	% Sand	% Clay	% Silt
Zone 1, 5m	Silty Loam	20.00	12.80	67.20
Zone 1, 15m	Clay	20.00	66.40	13.60
Zone 2, 5m	Clay	16.00	75.20	8.80
Zone 2, 15m	Clay	24.80	65.60	9.60
Zone 3, 5m	Clay	16.00	75.20	8.80
Zone 3, 15m	Clay	24.80	65.60	9.60

Table 107. Particle Size Distribution at H1u.

Sample Location	Soil Type	% Sand	% Clay	% Silt
Zone 1, 5m	Clay	17.60	56.00	26.40
Zone 1, 15m	Clay	18.40	72.00	9.60
Zone 2, 5m	Clay	17.60	76.80	5.60
Zone 2, 15m	Clay	21.60	72.80	5.60
Zone 3, 5m	Clay	32.80	60.80	6.40
Zone 3, 15m	Clay Loam	25.60	32.40	42.00

Table 108. Particle Size Distribution at TEN-1.

Sample Location	Soil Type	% Sand	% Clay	% Silt
Zone 1, 5m	Sandy Clay	56.80	36.40	6.80
Zone 1, 10m	Sandy Loam	72.80	20.40	6.80
Zone 2, 5m	Sandy Clay Loam	64.80	26.40	8.80
Zone 2, 10m	Sandy Loam	76.40	18.40	5.20
Zone 3, 5m	Sandy Loam	80.40	12.40	7.20
Zone 3, 10m	Sandy Loam	74.40	10.80	14.80

Percent organic matter (% OM) at SA 10, H1u, and TEN-1 are listed in Tables 109, 110, and 111. Percent organic matter at SA 10 ranges between 10% and 20%, with the wetter sampling locations containing a larger percentage of organic matter per sample. At H1u, values for organic matter range between 10% and almost 12% (11.75 ± 0.58). Although wetter areas of the planting zones have a higher percentage of organic matter, the range in organic matter content is narrow, reflecting less variation in water levels across the site. As would be expected with drier site conditions and a higher sand content, values for percent organic matter at TEN-1 are the lower than both SA 10 and H1u.

Table 109. Percent (%) Organic Matter at SA 10.

Sampling Location	Organic Matter ($\mu \pm \sigma$) (%)
Zone 1, 5m	20.64 ± 2.13
Zone 1, 15m	11.30 ± 0.55
Zone 2, 5m	13.02 ± 0.78
Zone 2, 15m	11.29 ± 0.40
Zone 3, 5m	10.48 ± 1.20
Zone 3, 15m	10.01 ± 0.59

Table 110. Percent (%) Organic Matter at H1u.

Sampling Location	Organic Matter ($\mu \pm \sigma$) (%)
Zone 1, 5m	11.75 ± 0.58
Zone 1, 15m	11.50 ± 0.64
Zone 2, 5m	11.86 ± 0.46
Zone 2, 15m	11.29 ± 0.48
Zone 3, 5m	10.96 ± 0.31
Zone 3, 15m	10.46 ± 0.30

Table 111. Percent (%) Organic Matter at TEN-1.

Sampling Location	Organic Matter ($\mu \pm \sigma$) (%)
Zone 1, 5m	6.94 ± 0.99
Zone 1, 10m	3.96 ± 1.03
Zone 2, 5m	3.10 ± 0.15
Zone 2, 10m	3.63 ± 0.48
Zone 3, 5m	5.42 ± 1.21
Zone 3, 10m	4.03 ± 0.33

Canopy Cover. Species composition of the existing forest canopies and associated canopy cover are unique to each underplanting site. The canopy at the SA 10 underplanting site is composed of *Salix caroliniana* (Carolina willow), *Acer rubrum* (red maple), and *Taxodium distichum* (bald cypress) and ranges between 5 m and 7 m in height. *Salix caroliniana* is the dominant tree species that grows throughout the ponded wet feature north of the planting area. The canopy above the planting zones consisted of a mixture of all three species, with *Taxodium distichum* being the least dominant. Canopy cover measurements are presented in Table 112. Percent canopy cover at SA 10 ranges between 69% and 80%.

Table 112. Canopy Cover (%) at SA 10 and TEN-1.

Site	Planting Zone	Station	Percent Cover
TEN-1	1	5m	73.62
	1	15m	78.74
	2	5m	58.88
	2	15m	76.44
	3	5m	59.96
	3	15m	64.51
SA-10	1	5m	77.75
	1	15m	80.54
	2	5m	68.99
	2	15m	73.14
	3	5m	77.70
	3	15m	72.50

A canopy consisting of *Salix caroliniana* was also present at H1u. However, previous herbiciding and burning of the CSA in 2005 had left the majority of the canopy without foliage and many trees appeared dead, although standing, when the site was planted in 2006. When the site was monitored in 2007, the majority of *Salix caroliniana* individuals that were standing in 2006 had fallen over within the underplanting site. Planting Zone 3 appeared to be the least affected by tree fall, and several surviving *Salix caroliniana* trees were still standing.

At TEN-1, the canopy over the wettest parts of Zones 1 through 3 consisted primarily of *Salix caroliniana* and *Sapium sebiferum* (Chinese tallow). As the planting zones transition from wet to dry, the canopy shifts in dominance from *Salix caroliniana* and *Sapium seriferum* to *Sapium sebiferum* and *Quercus nigra*, with *Baccharis halimifolia* (eastern baccharis) and *Schinus terebinthifolius* (Brazilian pepper) interspersed. When the site was monitored in 2007, it was observed that *Schinus terebinthifolius* was present not only in the drier portions of the planting zones but also underneath the *Salix caroliniana* canopy in the wetter portions. This species as well as *Sapium seriferum* are both invasive nonindigenous species, which grow aggressively in both terrestrial and more aquatic environments in Florida and can outcompete native species (Burks 1996). At TEN-1, percent canopy cover ranges between 59% and 79%, with lower coverage in the wetter areas (Table 112).

Understory Vegetation. Understory vegetation varies between the three sites, but several species were found at more than one site including *Eupatorium capillifolium* (SA 10 and H1u), *Eupatorium serotinum* (late-flowering thoroughwort) (H1u and TEN-1), *Pluchea odorata* (sweetscent) (H1 and TEN-1), and *Lygodium japonicum* (Japanese climbing fern) (SA 10 and TEN-1). Understory vegetation at SA 10 was dominated by the fern *Thelypteris hispidula var. versicolor* (hairy maiden fern). The vines *Ampelopsis arborea* (peppervine), *Momordica charantia* (balsampear), and the non-native *Lygodium japonicum* (Japanese climbing fern) were found growing within the underplanting site on seedlings in 2007. At H1u, in 2006, the wettest parts of Planting Zones 1, 2, and 3 were dominated by *Polygonum hydropiperoides* (swamp smartweed) and *Pluchea odorata* (sweetscent). Those species transitioned, first to *Eupatorium capillifolium* (dog fennel) and then to *Imperata cylindrica* (cogon grass), an aggressive, non-native invasive grass species, along the site's hydrologic gradient, from wet to dry. Understory vegetation at TEN-1 in 2006 consisted mainly of *Pluchea odorata*, *Polygonum hydropiperoides*, *Ludwigia peruviana*, and the two vines *Campsis radicans* (trumpet creeper) and *Parthenocissus quinquefolia* (Virginia creeper) in the wetter areas of the underplanting site. The understory at drier areas, on the side of the CSA dike, consisted of *Rubus argutus* (sawtooth blackberry), *Campsis radicans,* and *Parthenocissus quinquefolia*.

Survival

SA 10. Twenty species of wetland trees were planted as one-gallon seedlings at SA 10 on 07/18/2006. Figure 273 displays percent survival after one year for species planted in Zone 1, the wettest planting zone, at SA 10. Survival was high for all species within this zone. *Taxodium distichum* (n = 30) and *Liquidambar styraciflua* (sweet gum) (n = 25) had the best survival, and *Cephalanthus occidentalis* (n = 23) had the worst. Figure 274 displays percent survival after one year for species planted in Zone 2. All five planted species had high survival. *Nyssa sylvatica var. biflora* (swamp tupelo) (n = 25), *Ulmus americana* (elm) (n = 24), and *Quercus lyrata* (overcup oak) (n = 22) had the best survival. At Zone 3, the driest planting zone, all nine species survived well, with four of the nine planted species having 100% survival after one year (Figure 279); *Platanus occidentalis* (sycamore) (n = 24), *Liriodendron tulipifera* (tulip poplar) (n = 23), *Celtis laevigata* (hackberry) (n = 25), and *Quercus nigra* (n = 25). *Ilex cassine* (dahoon holly) (86%, n = 23) and *Nyssa aquatica* (88%, n = 27) had the lowest percent survival after one year. Overall survival for the entire site, using an aggregate of the twenty planted species, was 94% (n = 493). Nine of the 20 planted species had 100% survival after one year. Due to only slight variations in average water levels along transects and lack of above-ground inundation, species survival was compared with its distance along each monitoring transect (wet to dry). This assumes that distance along a transect compares survival along a gradient from wetter to drier conditions. No clear trend in survival was found along the hydrologic gradient at the site; however, the two seedlings of *Taxodium ascendens* which experienced the wettest conditions (>76%) did not survive, and *Cephalanthus occidentalis* had less mortality in the drier portion of the planting zone.

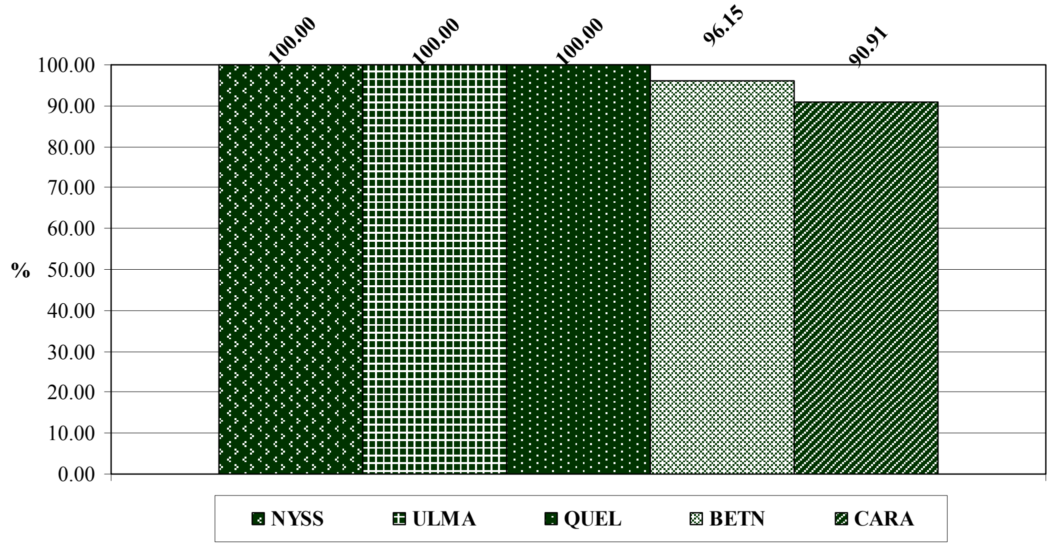


Figure 273. Percent Survival (%) for Zone 1 at SA 10.

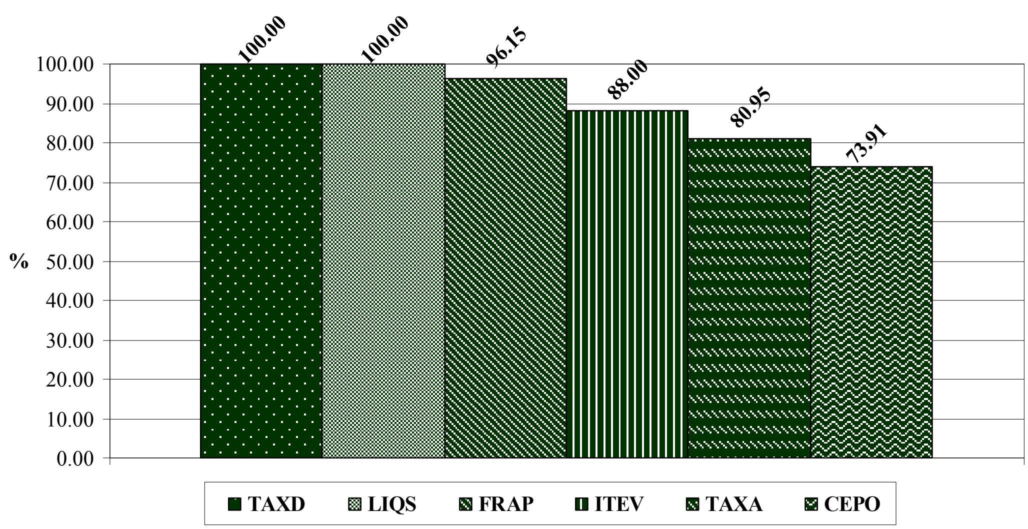


Figure 274. Percent Survival (%) for Zone 2 at SA 10.

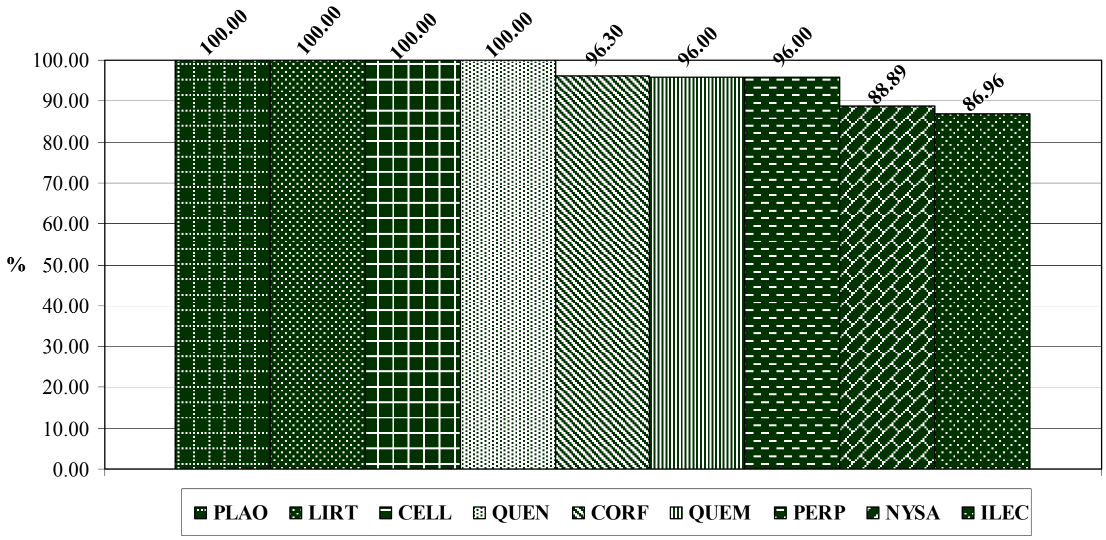


Figure 275. Percent Survival (%) for Zone 3 at SA 10.

H1u. Eleven species of wetland trees were planted as one-gallon seedlings at H1u on 07/11/2006. In contrast to SA 10, H1u is located in central Florida and so the more southerly occurring ash species, *Fraxinus caroliniana* (pop ash), was used instead of *Fraxinus pennsylvanica* (green ash). Also, the palm species *Sabal palmetto* (cabbage palm) was chosen, as this species is common to the edge of freshwater wetlands in Florida (Alexander 1995); however, the nursery delivered another variety of palm, *Sabal minor* (dwarf palmetto). This species is commonly found on well drained soils and is tolerant of drought conditions, but has a FACW status in Florida (Gilman 1999).

Figure 276 presents seedling survival at H1u. At Zone 1, the wettest area, *Fraxinus caroliniana* had the best survival with 81% (n = 27) of seedlings present after one year. *Nyssa sylvatica var. biflora* had the worst survival with only 36% (n = 25) of seedlings present. Species within this zone were subject to breakage and mortality due to the toppling of the *Salix caroliniana* stand. At Zone 2, *Celtis laevigata* (84%, n = 25) and *Ulmus americana* (64%, n = 25) had the best survival, while *Ilex cassine* had the worst (40%, n = 25). The majority of Zone 2 was also subject to damage due to the toppling of *Salix caroliniana*, as well as recruitment of *Imperata cylindrica* within the dry, canopy-free areas at the southern end of Zone 2. The majority of seedling decline for *Celtis laevigata* occurred in the drier portion of Zone 2, while mortality for the other two species occurred throughout the zone. At Zone 3, three of the five species had the highest percent survival of all planted species at H1u; *Quercus nigra* (96%, n = 25), *Liquidambar styraciflua* (90%, n = 20), and *Sabal minor* (84%, n = 26) (Figures 4-29). *Carya aquatica* had the worst survival at Zone 3 with only 50% (n = 22) of seedlings present after one year. Mortality for *Carya aquatica* and *Magnolia virginiana*, the species with the lowest percent survival in Zone 3, occurred throughout the planting zone and no trends in mortality along the hydrologic gradient were found. This zone had the least amount of damage from toppling *Salix caroliniana* individuals.

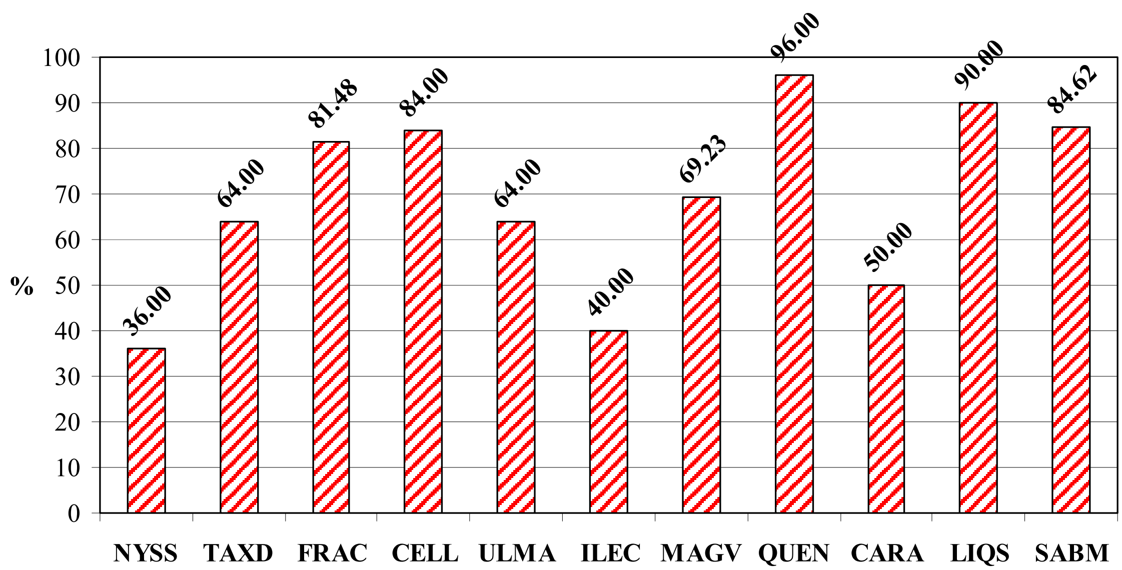


Figure 276. Percent Survival (%) for Seedlings at H1u.

TEN-1. Eighteen species of wetland trees were planted at TEN-1 on 08/06/2006. The site was planted with the same species as SA 10 with the exceptions of *Platanus occidentalis* (sycamore) and *Nyssa aquatica* (water tupelo). When TEN-1 was first monitored on 08/18/2006, the site appeared extremely dry and the foliage of several species planted in Zones 2 and 3 had already died back.

Figure 277 displays percent survival for species planted within Zone 1 at TEN-1. *Taxodium distichum* (100%, n = 24) and *Fraxinus caroliniana* (91%, n = 24) had the best survival at Zone 1, while *Nyssa sylvatica var. biflora* (37%, n = 24) had the worst, with seedling mortality occurring throughout the planting area. Percent survival for the six species planted within Zone 2 at TEN-1 is presented in Figure 278. *Ulmus americana* (92%, n = 25) had the best survival, and *Ilex cassine* (47%, n = 23) and *Quercus lyrata* (52%, n = 25) had the worst. *Quercus lyrata* had poor survival within the entire planting zone, while *Ilex cassine*, *Carya aquatica*, and *Betula nigra* had the majority of seedling mortality within Zone 2, 8 m or greater along T2 (Figure 271). After this point within the planting zone, average water levels fall below -3.52 m in depth, and also no inundation within the root zone occurs during the period of record. *Sabal palmetto* had the best survival at Zone 3 (100%, n = 26) (Figure 279). *Quercus nigra* (88%, n = 27), *Cornus foemina* (swamp dogwood) (84%, n = 26), and *Magnolia virginiana* (sweet bay) (82%, n = 23) also survived well after one year. *Liquidambar styraciflua* had the lowest percent survival (48%, n = 25), and this species as well as *Liriodendron tulipifera* (57%, n = 26), experienced the highest mortality after 5 m along T3, where average water level drops to below -1 m and the root zone is never inundated (Figure 272). Several species were able to exist from the wettest to the driest conditions along the dike's gradient: *Ulmus americana*, *Celtis laevigata*, *Sabal palmetto*, *Quercus nigra*, *Cornus foemina*, and *Magnolia virginiana*.

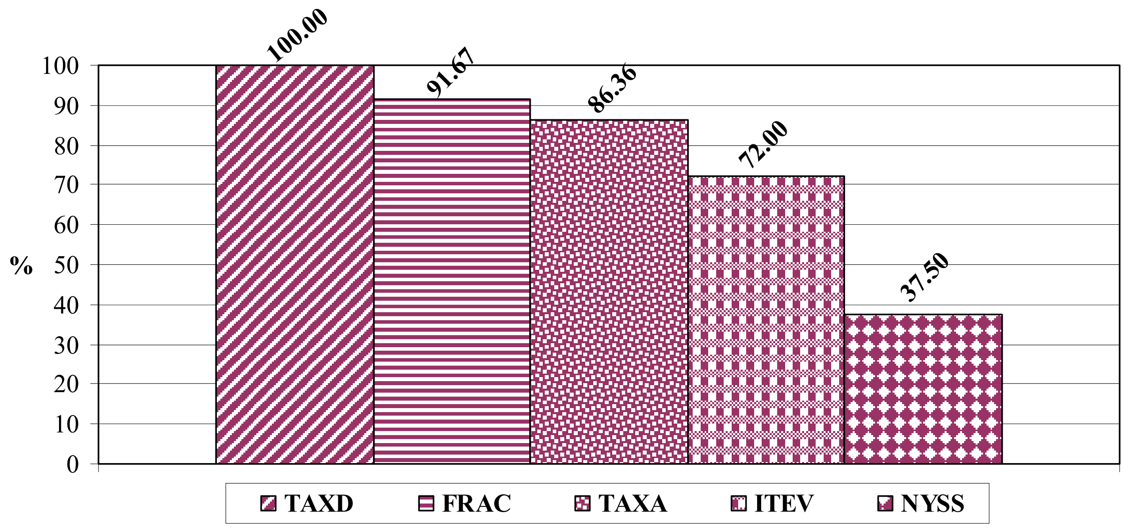


Figure 277. Percent (%) Survival for Zone 1 at TEN-1.

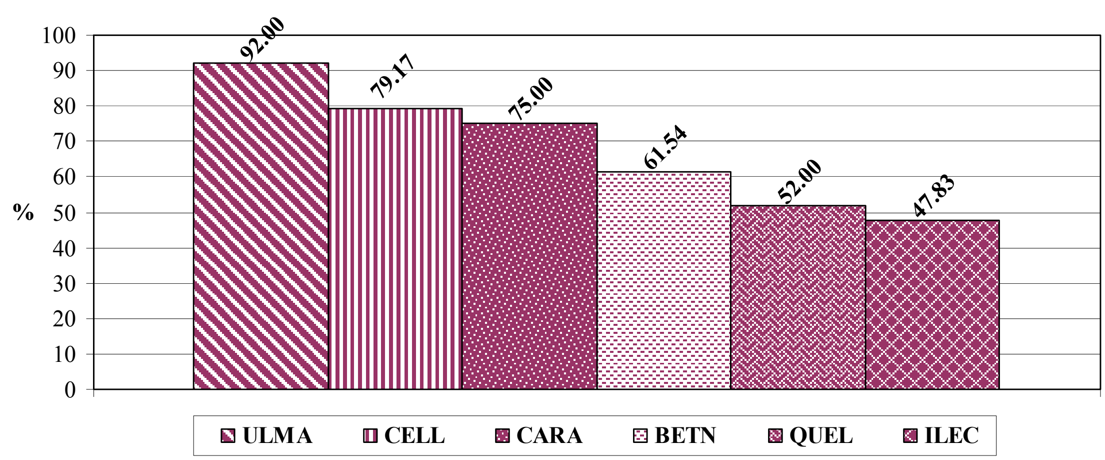


Figure 278. Percent Survival (%) for Zone 2 at TEN-1.

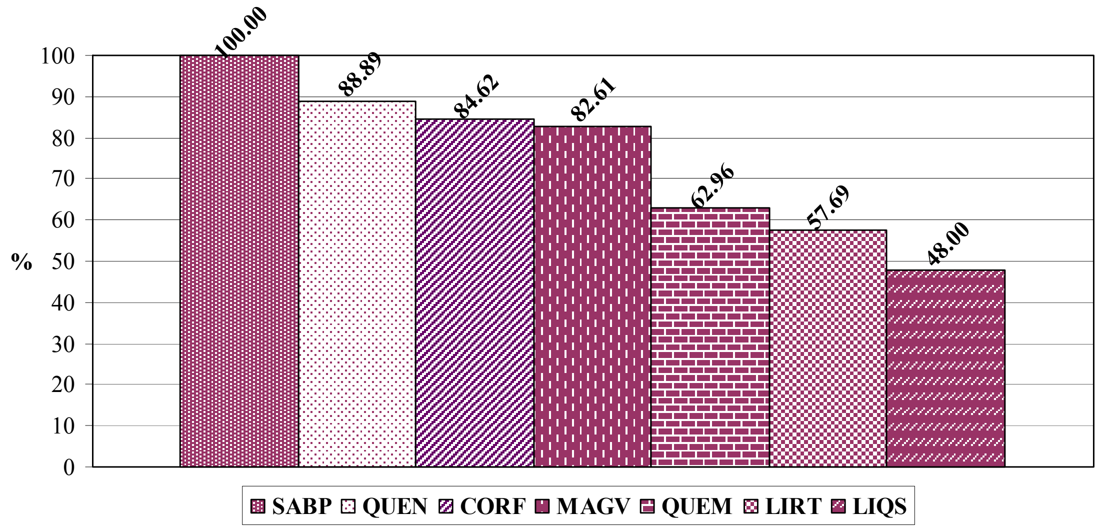


Figure 279. Percent Survival (%) for Zone 3 at TEN-1.

Overall. SA 10 (94%, n = 493) had the highest overall percent survival for seedlings among the three underplanting sites, and H1u (69%, n = 271) had the worst. Seventy-three percent (n = 477) of planted seedlings survived at TEN-1. Overall survival for all seedlings from the three underplanting sites (n = 1210) was 81%. Table 113 compares survival of each species by site.

Table 113. Seedling Survival at Underplanting Sites.

Species	SA-10	TEN-1	H-1
NYSS	100	37.5	36
TAXD	100	100	64
FRAC	n/a	91.6	81.5
FRAP	96.2	n/a	n/a
CELL	100	79.2	84
ULMA	100	92	64
ILEC	87	47.8	40
MAGV	96	82.6	69.2
QUEN	100	88.9	96
CARA	90.9	75	50
LIQS	100	48	90
SABP	n/a	100	n/a
SABM	n/a	n/a	84.6
ITEV	88	72	n/a
TAXA	80.9	86.4	n/a
BETN	96.1	61.5	n/a
QUEL	100	52	n/a
LIRT	100	57.7	n/a
CORF	96.3	84.6	n/a
QUEM	96	63	n/a
CEPO	73.9	n/a	n/a
NYSA	88.9	n/a	n/a
PLAO	100	n/a	n/a
Max	100	100	96
Min	73.9	37.5	36

Eight species of wetland trees were planted within a Planting Zone 1, the wettest planting area at each site. *Fraxinus caroliniana*, *Fraxinus pennsylvanica*, *Taxodium distichum*, *Taxodium ascendenes*, *Itea virginica*, and *Cephalanthus occidentalis* all survived well, despite differences in Planting Zone 1 hydrology between sites. *Liquidambar styraciflua*, included within Zone 1 of SA 10 due to spacing constraints, survived well at both the wetter and drier locations (Zone 3 at H1u and TEN-1). Survival was highest for *Nyssa sylvatica var. biflora* at SA 10 (100%, n = 25) where average water levels ranged between -0.06 m and -1 m, with root inundation within most of its planting zone (Figure 263).

Eight species of wetland trees were also planted within a Planting Zone 2. *Ulmus americana* survived well where average water levels ranged from -0.2 m to -7.0 m and root zone inundation ranged from 0 to 93% of the period of record (Figures 263, 271). *Betula nigra*, *Quercus lyrata, Carya aquatica, Celtis laevigata, and Ilex cassine* all survived best at SA 10 where conditions were the wettest and the canopy and subcanopy were stable. *Quercus nigra* and *Magnolia virginiana* survived well at Zone 3 at all three sites, across a large range in hydrologic conditions. *Quercus michauxii*, *Cornus foemina*, *Liriodendron tulipifera* were all planted in Zone 3 at SA 10 and TEN-1. While *Cornus foemina* survived well at both sites, *Quercus michauxii* and *Liriodendron tulipifera* did not survive well at portions of the TEN-1 Planting Zone 3, where average water levels ranged between -1.39 m and -4.67 m (Figure 272).

Growth

SA 10. Figure 280 presents height data for tree seedlings planted at SA 10. Change in mean seedling height after one year ranged between 42.34 cm and -9.18 cm for the twenty planted species. *Platanus occidentalis* had the highest increase in mean seedling height (42.34 cm) as well as the highest percent change in mean seedling height (29%) of the 20 planted species. *Itea virginica* had the lowest positive increase in average height (4.06 cm), and *Quercus lyrata* had the lowest positive percent change in height (4%). *Cephalanthus occidentalis* and *Betula nigra* had declines in mean seedling height and negative percent change in height after one year.

H1u. Figure 281 presents height data for the planted tree species at H1u. Change in mean seedling height for the remaining ten species ranged from 6.92 cm to -24.86 cm. Six of the ten species experienced decline in mean seedling height. *Ilex cassine* had the largest increase in mean seedling height (6.92 cm) and the highest percent change (18.46%) in height at H1u; however, this species had one of the lowest percent survivals (40%, n = 25) at the site. *Taxodium distichum* and *Ulmus americana* had the largest decrease in mean height over the period of record. Both species appear to have been negatively affected by the toppling *Salix caroliniana* canopy, but were able to survive at 64% after one year.

TEN-1. Height data for the underplanting at TEN-1 is presented in Figure 282. *Ulmus americana* had the greatest increase in mean seedling height and percent increase in height at TEN-1. However, this species varied in growth across the planting area. Minimum height only increased from 23 cm to 24 cm, but maximum height increased by 135 cm and the σ increased to 49.65 cm in 2007. *Quercus lyrata* declined in mean height by -8.11 cm, and had the worst growth at TEN-1. This species also had low survival (52%) and was observed wilting shortly after planting occurred.

Overall. Table 114 compares percent change in mean seedling height for each species at each site. Eight species, planted at all three sites, had the highest percent change in mean height at SA 10. H1u had the worst growth for all but two site comparisons. Two-tailed T-tests, with F-tests for equal variance, were used to test for significant differences in mean seedling growth (or decline) for species between sites.

Results are presented in Table 115. Mean growth was significantly lower ($p < 0.025$) at H1u for *Nyssa sylvatica*, *Taxodium distichum*, *Fraxinus caroliniana* (only H1u vs. TEN-1), *Celtis laevigata*, *Carya aquatica* (only H1u vs. SA 10) and *Ulmus americana*. At TEN-1, mean growth for *Ulmus americana* and *Quercus nigra* was significantly higher ($p < 0.025$) than for those two species at either SA 10 or H1u.

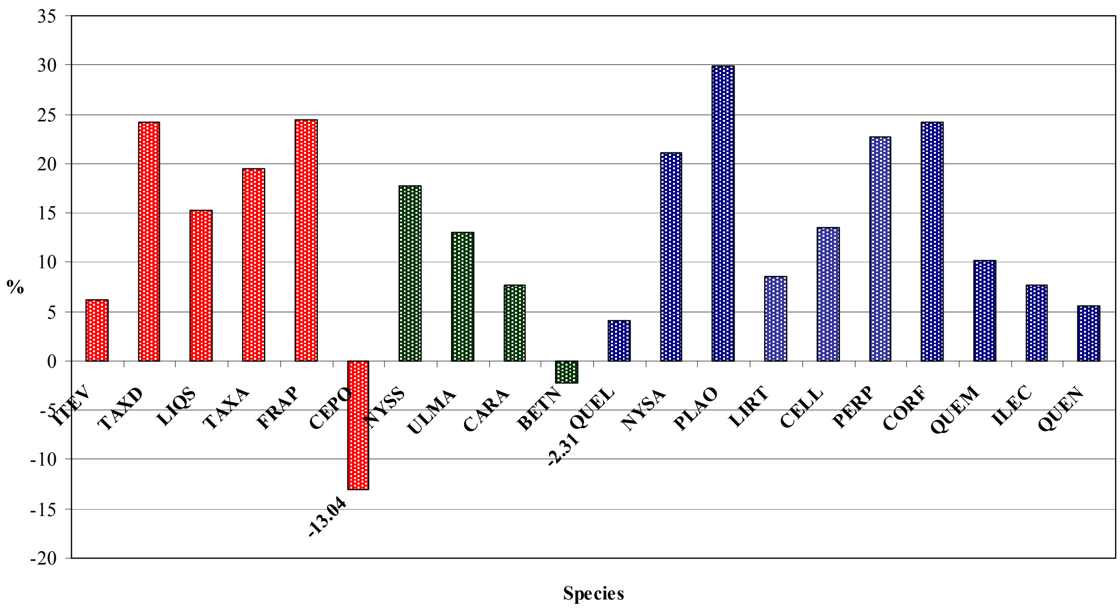


Figure 280. Percent Change (%) in Seedling Height at SA 10.

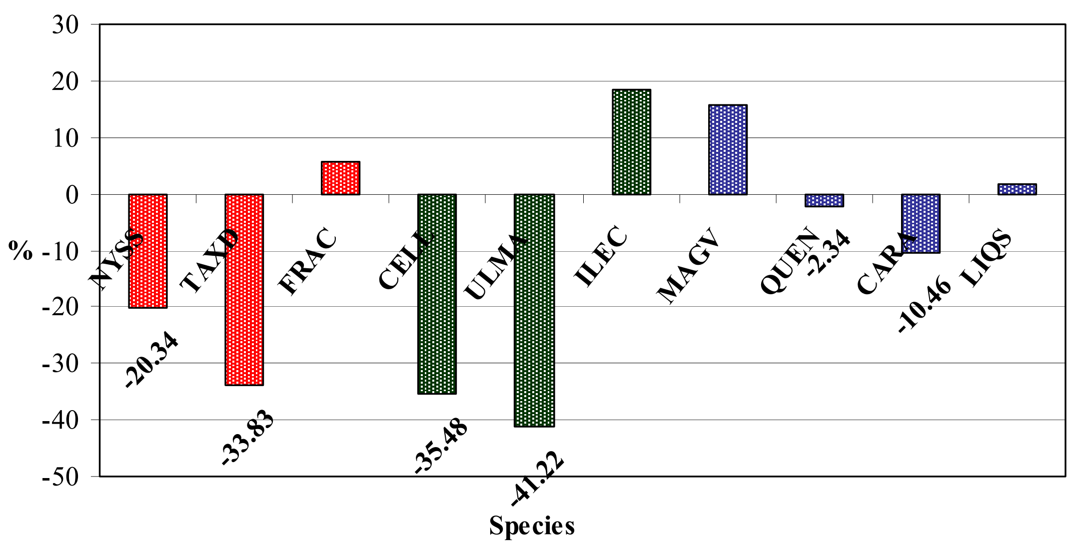


Figure 281. Percent Change (%) in Mean Seedling Height at H1u.

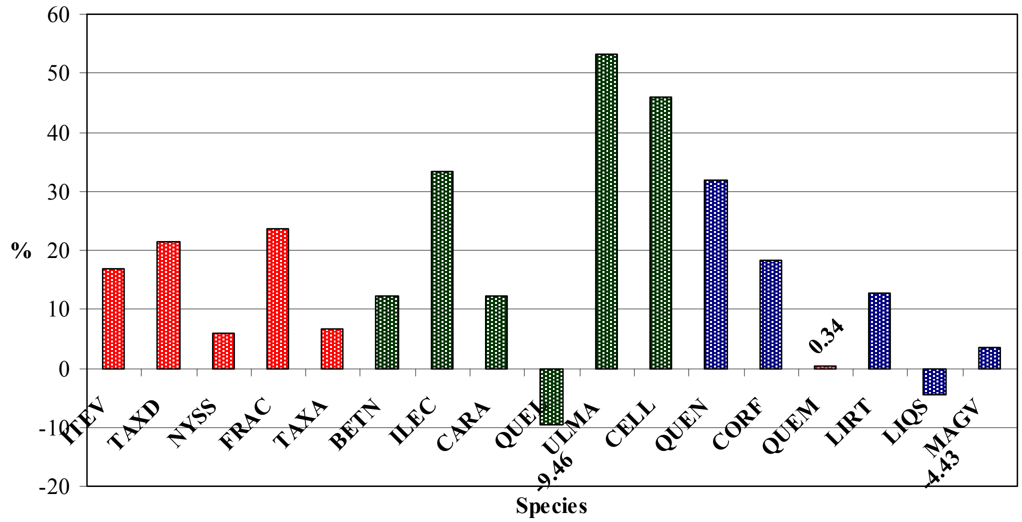


Figure 282. Percent Change (%) in Mean Seedling Height at TEN-1.

Table 114. Percent Change (%) in Mean Seedling Height at SA 10, TEN-1, and H1u.

Species	SA-10	TEN-1	H-1
NYSS	17.73	5.85	-20.34
TAXD	24.24	21.48	-33.83
FRAC	n/a	23.75	5.64
FRAP	24.43	n/a	n/a
CELL	13.52	45.82	-35.48
ULMA	13.00	53.20	-41.22
ILEC	7.66	33.32	18.46
MAGV	22.77	3.58	15.71
QUEN	5.52	31.79	-2.34
CARA	7.65	12.32	-10.46
LIQS	15.27	-4.43	1.61
SABP	n/a	n/a	n/a
SABM	n/a	n/a	n/a
ITEV	6.17	16.81	n/a
TAXA	19.50	6.70	n/a
CEPO	-13.04	n/a	n/a
BETN	-2.31	12.14	n/a
QUEL	4.12	-9.46	n/a
NYSA	21.11	n/a	n/a
PLAO	29.91	n/a	n/a
LIRT	8.61	12.65	n/a
CORF	24.20	18.35	n/a
QUEM	10.17	0.34	n/a
Max	29.91	53.20	18.46
Min	-13.04	-9.46	-41.22

Table 115. Mean Seedling Growth for SA 10, TEN-1, and H1u.

Species	SA-10	TEN-1	H-1u
NYSS	21.13	7.33	-9.5 A
TAXD	29.07	22.67	-26.063 A
FRAC	27.55 A	-6.09	N/A
FRAP	29.56	N/A	N/A
CELL	21.04	41.84	-13.67 A
ULMA	19.29 A	54.09 A	-26.44 A
ILEC	6.37	16.18	6.70
MAGV	19.58	6.05	7.25
QUEN	5.28	20.08 A	0.00
CARA	4.95 A	2.21	-3.60 A
LIQS	20.56 A	-2.92	2.94
ITEV	4.91	16.28	N/A
TAXA	16.53	7.68	N/A
CEPO	-8.06	N/A	N/A
BETN	-2.71	22.53	N/A
QUEL	4.86	-11.54	N/A
NYSA	17.21	N/A	N/A
PLAO	36.68	N/A	N/A
LIRT	10.26	20.13	N/A
CORF	25.04	9.78	N/A
ILEC	8.59	5.47	N/A

A = Statistically different.
alpha = 0.025
tails = 2.00

Seedling Growth and Hydrology

The association between hydrologic conditions and seedling growth, the change in height after one year, was investigated with regression analysis to test for correlation and the strength of association. Since the cause of poor growth was not specifically controlled and tested for, values for seedlings that declined from or maintained their initial height were included. The inclusion of these values weakened the strength of the regression for several species.

SA 10. Results from correlation and regression using distance within the planting zone as the independent variable are presented for SA 10 in Table 116. Nine of the 20 tree species were negatively correlated (r) with decreasing average water levels and root zone inundation, which decreased between 0 m and the endpoint of the planting zone; there was decreased growth with drier conditions.

H1u. Correlation of growth and average water conditions is confounded by tree fall that occurred mainly in Zones 1 and 2 of H1u. Due to this, the relationship between seedling growth and planting zone hydrology was only explored for Zone 3. Results from correlation and regression within Planting Zone 2 are presented for H1u in Table 117. At Zone 3, water levels do not follow a clear hydrologic gradient, fluctuating within the planting zone (Figure 268). A slight negative correlation with decreasing average water levels was found for *Magnolia virginiana* and *Liquidambar styraciflua*.

TEN-1. Most species growth was correlated negatively with distance along the planting zone; there was decreased growth with drier conditions (Table 118). *Ilex cassine* had the strongest association with decreased water availability ($r = -0.733$, $r^2 = 0.54$).

Table 116. Seedling Growth Along the Hydrologic Gradient at SA 10.

Regression Species	ITEV	TAXD	LIQS	TAXA	FRAP	CEPO	NYSS	ULMA	CARA	BETN	QUEL	NYSA	PLAO	LIRT	CELL	MAGV	CORF	QUEM	ILEC	QUEN
r value	-0.3319	-0.5078	0.6785	-0.6091	-0.6767	-0.4747	0.1731	-0.2128	-0.3870	-0.2596	0.1327	0.0656	0.0975	0.0055	0.1049	0.0548	0.2943	-0.3871	0.0907	0.0289
r^2 value	0.1102	0.2578	0.4604	0.3710	0.4579	0.2254	0.0300	0.0453	0.1498	0.0674	0.0176	0.0043	0.0095	0.0000	0.0110	0.0030	0.0866	0.1498	0.0082	0.0008
n	22	30	25	17	25	17	25	24	20	25	22	24	24	23	25	24	26	24	20	25
Δ Height (cm)	4.06	29.56	20.56	18.67	29.31	-9.18	21.32	19.29	4.57	-3.67	4.86	17.94	42.34	10.26	21.04	21.11	29.42	9.78	6.09	5.28

r = Pearson product moment correlation coefficient; measures the direction and strength of a linear relationship between seedling growth and water availability

r2 value = Coefficient of determination; the ratio of variation in growth explained by water availability; the strength of the linear association between growth and water availability.

n = number of individuals used in the regression

Δ Height (cm) = difference in mean height over the period of record

Table 117. Seedling Growth Along the Hydrologic Gradient at H1u.

Regression Species	MAGV	QUEN	CARA	LIQS
r value	-0.29	-0.00152398	0.075922637	-0.145040997
r^2 value	0.08	0.00240459	0.005764247	0.021036891
n	18	24	11	18
Δ Height (cm)	5.25	-1.2	3.7	1.22

r = Pearson product moment correlation coefficient; measures the direction and strength of a linear relationship between seedling growth and water availability

r2 value = Coefficient of determination; the ratio of variation in growth explained by water availability; the strength of the linear association between growth and water availability.

n = number of individuals used in the regression

Δ Height (cm) = difference in mean height over the period of record

Table 118. Seedling Growth Along the Hydrologic Gradient at TEN-1.

Regression Species	ITEV	TAXD	NYSS	FRAC	TAXA	BETN	ILEC	CARA	QUEL
r value	0.13	-0.46	-0.462269	-0.5816479	0.056441676	-0.474776	-0.733324	-0.4172195	-0.05202626
r^2 value	0.02	0.21	0.213693	0.33831424	0.003185663	0.2254127	0.5377638	0.17407211	0.00270673
n									
Δ Height (cm)									

r = Pearson product moment correlation coefficient; measures the direction and strength of a linear relationship between seedling growth and water availability

r2 value = Coefficient of determination; the ratio of variation in growth explained by water availability; the strength of the linear association between growth and water availability.

n = number of individuals used in the regression

Δ Height (cm) = difference in mean height over the period of record

Regression Species	ULMA	CELL	QUEN	CORF	QUEM	LIRT	LIQS	MAGV
r value	-0.46382	-0.611949	-0.600389	-0.5239844	-0.126265005	-0.597398	0.1464629	-0.0276811
r^2 value	0.215125	0.3744817	0.360468	0.27455969	0.015942852	0.3568842	0.0214514	0.00076624
n								
Δ Height (cm)								

Monitoring Sites

This section presents data on wetland tree survival at two wetland sites, PPW-1 and PPW-2, two and four years after planting occurred. Seedling growth is presented for selected species. The effect of hurricanes and severe drought on species survival is also addressed.

Hydrology and Climate. PPW-1 and PPW-2 are in close proximity and thus subject to the same climatic conditions. The first two years after planting (2003-2005) were characterized by Hurricanes Charley, Frances, and Jeanne (Figure 283), whose paths all crossed Polk County, Florida, causing high water events and wind damage to seedlings (personal communication, Kate Himel, FIPR). Those wet conditions were juxtaposed with a severe drought, beginning in the third year after planting and lasting through the 2007 growing season and monitoring. This led to a decrease in cumulative precipitation, as well as quantity and frequency of rainfall events, in 2006 and 2007, compared with average yearly precipitation in Polk County, Florida. Hydrologic conditions at PPW-1 and PPW-2 were comparable due to similar topography and wetland/watershed ratios.

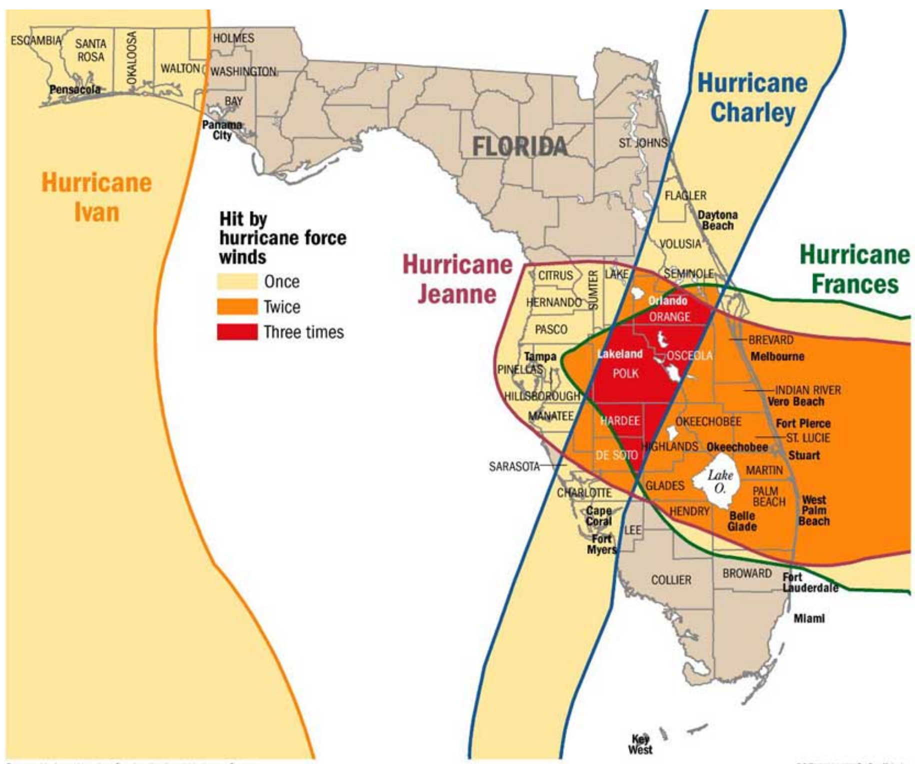


Figure 283. Hurricane Paths Across Polk County, Florida, in 2004.

Survival. Percent survival, in 2005 and 2007, for each species at PPW-1 and PPW-2 is presented in Figures 284 and 285. Species survival rates at PPW-1 are analogous to rates at PPW-2 at two and four years after planting, likely due to similar climatic conditions, wetland hydrology, pre- and post- planting invasive species management, and substrate composition.

Quercus virginiana (live oak) had the best survival after two years at each site (93% PPW-1; 94% PPW-2) with the exception of *Quercus nigra* (water oak) (100% PPW-1; 100% PPW-2). However, sample size for *Quercus nigra* was extremely low compared with other species planted (n = 3 PPW-1; n = 3 PPW-2). *Taxodium distichum* (bald cypress), *Fraxinus caroliniana* (pop ash), *Carya aquatica* (water hickory), and *Myrica cerifera* (wax myrtle) established with over 50% survival at both sites after two years. *Quercus laurifolia* (laurel oak) had the lowest survival at both sites after two years (27% PPW-1; 17% PPW-2).

Quercus virginiana had the best survival at both sites after four years (92% PPW-1 and PPW-2), again with the exception of *Quercus nigra*. *Quercus laurifolia* had the lowest survival after four years (24% PPW-1; 15% PPW-2), but only experienced 3% and 1% declines in survival between 2005 and 2007 at PPW-1 and PPW-2. Declines in seedling survival between 2003 and 2005 were greater than 2005 through 2007 for all species at both sites, with the exception of *Taxodium distichum*, which experienced similar declines over both monitoring periods. This may indicate that hurricane activity immediately after planting affected survival more negatively than drought conditions over 2006-2007. When an aggregate of all species at both PPW-1 and PPW-2 was formed, overall survival was 59% in 2005 and 55% in 2007 (n = 1419).

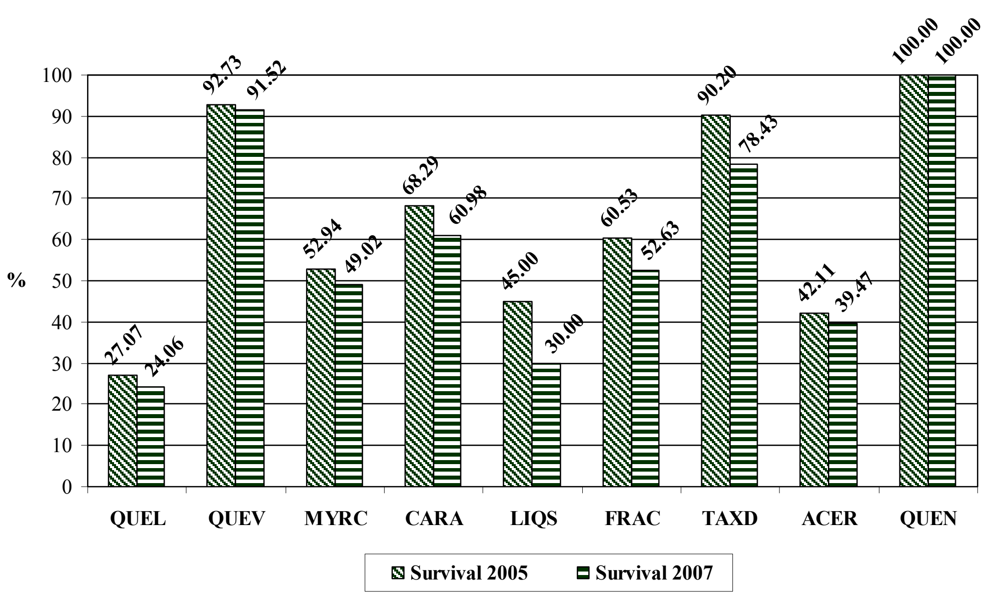


Figure 284. Percent Survival (%) at PPW-1.

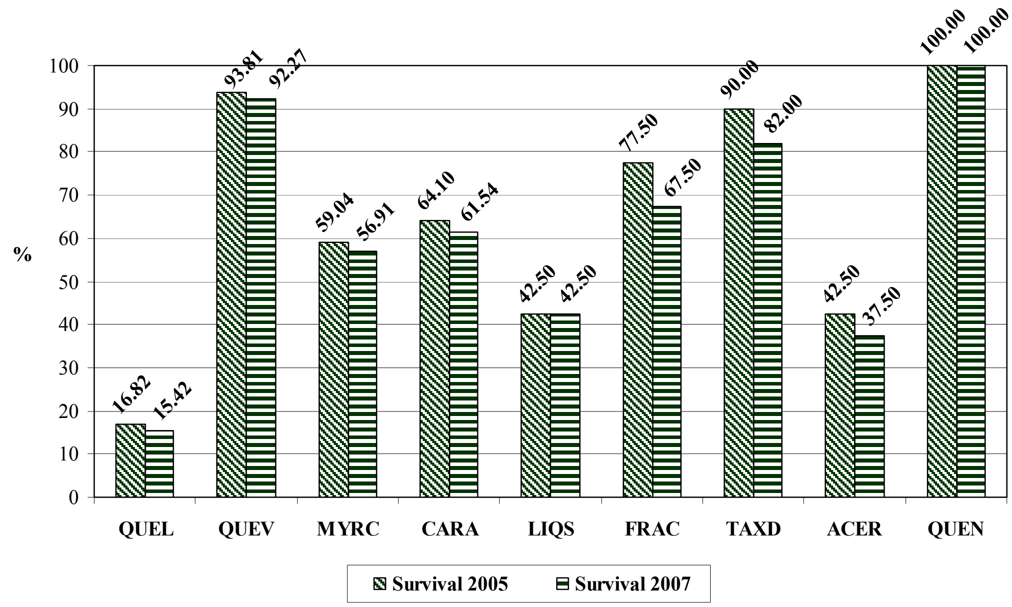


Figure 285. Percent Survival (%) at PPW-2.

DISCUSSION

CHARACTERIZATION OF NATURALLY OCCURING WETLANDS ON CSAs

Naturally established wetlands on CSAs are dominated by a relatively small number of communities, despite their location in two regions of the state with slightly different climate conditions and more proximately, their diverse landscape configurations and constructions. The communities do not change noticeably between 10 and 40 years following abandonment. Differences in elevation profiles, however, do lead to variability in the composition and relative location of these communities.

Although there are natural wetlands systems with only one woody species in the canopy layer, most, including those in Florida, are generally much richer in species (Myers and Ewel 1990). The ground level of CSAs was the most species-rich, but often one community dominated, typically cogongrass in the transitional zones and floating aquatics in the deepest zones of the wetland. More species richness was often seen in areas with moderate depths of flooding, although some canopy layer species can persist in deeper zones.

Standing biomass, as measured by woody basal area, is not as high as it is in natural forested wetland communities. Except when comparing stands of mature willows on CSAs with stands in other locations, basal area is generally lower. This evidence of reduced canopy structure could be due to the species present, which do not have the morphological capacity to grow to match basal areas of species communities in natural and climax wetlands, though caution should be used with such a conclusion since CSAs are typically young systems with accumulating biomass.

Soils in CSAs have been described as heavy clays of industrial origin with high amounts of residual phosphorus (Rushton 1988; Zhang and Albarelli 1995). Tests of CSA wetland soils confirmed high residual P concentrations (mean = 606 ppm TP), which are much higher than soil TP concentration in natural wetlands in the southern Coastal Plain ecoregion, where a survey of TP concentrations in wetland soils averaged 96 ppm (Greco 2004).

CSA soils give indications of expected wetland soil development in some ways, such as the build-up of organic matter with time and the associated increased availability of nitrogen in CSA soils. The increase in available N with ecosystem development and the corresponding stasis in P concentration tend to increase the N:P ratio. However, if one of these nutrients is limiting it would be N, as N is typically limiting when the N:P ratios are less than 14 (Koerselman and Meuleman 1996). Nevertheless, although a nutrient limitation effect was not specifically tested, growth rates and stand densities of vegetation present in CSA wetlands indicated high productivity, which reflects a lack of significant nutrient limitation.

That *Typha spp.* were found in areas with the highest P concentration likely reflects an effect of bioconcentration of P rather than a response to patches of high P content embedded during the clay settling process. *Typha* are able to adapt to low N:P ratios and continue to absorb P whereas other wetlands plants, such as *Cladium jamaicense*, are not (Lorenzen and others 2001). It is likely initial P concentrations are evenly distributed in CSAs, since soils are processed and well mixed upon settling. This effect of increasing P concentration in the top cm of the soil likely serves as a positive feedback to further the growth and expansion of *Typha spp*. *Imperata cylindrica* may have a similar positive effect upon N concentration in the surface soil. In both cases, however, it is unclear whether the plant-soil nutrient effects have an influence over CSA ecosystem development. Overall it might be said that through effects on physical and chemical soil properties, wetland communities are helping direct wetland areas toward maturation, with indications being their effects on the soil N:P ratio, lowering of the pH, and a general increase in biotic influence over ecosystem characteristics (Odum 1964).

HYDROLOGIC ANALYSIS AND MODELING

Hydroperiod Analysis

Observed hydroperiods for surface water features of the eight sites demonstrated differences between sites (Figure 77). Some sites experienced seasonal fluctuations and drying similar to the Green Swamp systems and compared to a typical Florida wetland hydroperiod but with quicker and larger response to rain events and faster rates of decline (Figures 79 and 78). Other sites, however, had surface water features that were more buffered from events, had lower rates of drawdowns, and experienced more ponded conditions compared to other monitored sites and to natural wetland systems. Different hydrologic regimes were also observed at multiple surface water features within one CSA (Figure 69). An understanding of what factors create differences both among and within CSAs and when compared to natural systems is imperative when trying to predict future hydrologic regimes and planning a restoration design.

Spatial Modeling of CSA Hydrology

Depressions within closed basins, such as CSAs, have hydrology that is characteristic of ecosystems typically defined as isolated wetlands. Examples of isolated wetlands in the U.S. include prairie potholes in the Upper Midwest, playas in the Southwest, cypress domes in Florida, and Carolina bays in the South Atlantic Coastal Plain (Tiner 2003). The hydrogeology of CSAs in the phosphate mining region is analogous to these isolated wetland systems, in which water accumulates in topographically low areas. In isolated wetland systems, hydrology is usually dominated by precipitation, ET, and/or groundwater flows, as opposed to channelized inflows and outflows. Also a unique hydrologic phenomenon is evident in some of these systems: at high water levels depressions can become connected (Tiner 2003), thus merging their watershed boundaries. This adds significant complexity to a hydrologic model for a watershed that can have potentially thousands of dynamic subwatersheds. These hydrologic differences between a depressional closed basin system compared to a classic watershed with a dendritic drainage network, cause the choice of watershed delineation scale (z-limit) as specified in GIS to be more important for a spatial hydrology model for CSAs or other flat depressional landscapes. Watershed delineation sets the boundaries of the water feature for water budget calculations within the spatial model. If the watershed delineation scale is too large, some water features may be left out of the model due to the level-pool assumption, but if the scale chosen is too small, then watersheds may quickly become over-filled with water and will not be modeled accurately.

The spatial model results demonstrate that it can accurately model the water level fluctuations in time and space within a CSA. For most CSAs a multiple watershed approach was required for modeling of individual surface water features, since there were many such features located throughout each CSA. The watershed delineation maps show that the six CSAs studied fell into three general groups based on their number, size, and depth of watersheds which likely affect their hydrologic regime. There were CSAs that contained many small, shallow watersheds such as Mosaic H1 and Williams Co., which dried out frequently and had large responses to rain events. There were CSAs that contained one or two large, deep watersheds which dominated the hydrology such as PCS SA 01 and SA 10. In these watersheds the water levels did not fluctuate as much as the other CSAs, probably due to their overall size and geomorphology that concentrated water into one main area. Finally, there were also CSAs dominated by one or two large, shallow watersheds, such as Mosaic K5 and CFI SP-1, which did dry down and fluctuated more than the other large CSAs.

Overall there was quite a range in values of model components used such as runoff coefficients (0.20-0.90), summer daily loss rates (0.70-2.40 cm/day), and winter daily loss rates (0.00-1.40 cm/day). The model fits also had a wide range (R^2 from 0.59 to 0.95). There is no clear reason for this variance between CSAs and within CSAs (Mosaic H1) based on the spatial model results, which leads to the conclusion that individual CSAs have different hydrologic characteristics. The best fit of modeled and measured water level data occurred for Mosaic H1 SW-1 ($R^2 = 0.95$). This was partially due to the inclusion of extra model components such as increased runoff coefficient during large storms (> 4 cm) and changing the daily loss rate on an annual basis. The latter was required for Mosaic H1 SW-1 to apply different rates of decline after an extended dry period. It was suspected that the development of cracks with the drying of the clays may have increased infiltration. This type of change is hard to predict and therefore cannot be explicitly included in the model. While this phenomenon was not observed in the models of the other sites, such extended dry periods in between simulation periods of other sites was not experienced.

The spatial model produced hydropattern maps that could be used for wetland restoration efforts. All watersheds simulated (except CFI SP-1) showed periodic inundation during simulation periods greater than 20% of their area, which ranged from 0.9 ha for H1 SW-1 to 44.9 ha for K5. These areas that were periodically inundated would be greater if the LiDAR maps allowed simulation of the entire monitoring period because many areas that were designated as inundated 100% of the time would actually be dry during part of the monitoring period. The purpose of these hydropattern maps is to guide wetland restoration efforts by identifying areas that have the optimal hydrologic regime for specific plant species. These maps show the frequency of time inundated which could be combined with water depth, soil type, and canopy cover data to increase the potential for wetland plant survival, and therefore successful ecosystem enhancement. Additionally, a spatial model that has been calibrated using a long period of water level data could be used to test scenarios related to CSA design such as outfall height and configuration.

Water Budgets

When attempting to enhance or create wetland ecosystems, an understanding of the hydrology and its major determinants are of primary concern. Both the hydroperiod analysis and spatial model development highlighted differing hydrologic regimes between and within CSAs. This study monitored the internal hydrology of eight CSAs to understand and predict runoff responses, calculate water budgets, observe surface and groundwater interactions, analyze ET in detail, and develop temporal hydrologic models.

In past studies, CSA runoff analysis has primarily focused on runoff events measured at a gauged outflow of the CSA and thus treated the CSA as one watershed (Lewelling and Wylie 1993; Reigner and Winkler 2001; Riekerk and others 1990). As a result, these studies often resulted in runoff amounts lower than what is expected for clayey soils. The focus of this study was on individual surface water features within a CSA, and runoff analysis revealed differences in runoff characteristics between water features and resulted in runoff responses at some sites more typical of clays (Figures 116-123).

The runoff analysis suggested that most sites' runoff depths could be better explained solely from the magnitude of rain than with runoff coefficients and multiple regression models (Table 54). The fact that the multiple regression models, which included upland to wetland areas, were less successful suggests that runoff may have been primarily from a constant contributing area such as the edges of the feature which do not change significantly with water level. Furthermore, because the inclusion of antecedent conditions did not typically result in better models, it appears that the runoff coefficients of these constant contributing areas may be largely determined by magnitude of rain. The results from the modeling of Williams Co. and Mosaic HP-10 were exceptions, and these are sites where both contributing areas and antecedent conditions may play a more important role. Nevertheless, the differences in characteristic runoff responses to rain events among and even within CSAs were clear. While runoff was analyzed differently here than in the spatial model development, the differences in sites that resulted were similar. In their study of three CSAs, Lewelling and Wylie (1993) also observed different characteristics among CSAs regarding runoff responses.

The most pronounced buffering conditions occurred at PCS SA 10 and SA 01, which may be partially explained by the higher conductive soil conditions in portions of their surrounding uplands limiting runoff (Table 62). PCS SA10 had the highest saturated hydraulic conductivity measured and one of lowest runoff contributions to its water balance (Tables 53, 59, and 62). The sand tailings in the upland of PCS SA 01 prevented a hydraulic conductivity from being measured, suggesting a higher conductivity than was measured at the other sites. PCS SA 01 experienced the second lowest runoff contributions. While CFI SP-1 did not experience ponded conditions throughout the study period, the inundated period that occurred from fall 2004 to spring 2006 also experienced a more buffered hydroperiod compared to other systems (Figure 71). The third highest conductivity was measured at CFI SP-1 as a result of its sand-clay mix, and runoff contributions were the lowest (Table 53). Almost 80% of observed event response at CFI SP-1 was explained by direct rainfall, which may account for its extensive dry period.

The runoff contributions at Williams Co. and Mosaic H1 SW-1 were the highest, with the exception of Mosaic HP-10, and explain their flashy nature with rapid response to rain events (Table 53). While Mosaic HP-10 had the largest contribution to its water balance from runoff, it was frequently connected to an outfall system, which minimized flooding depths (Figure 75). In addition to low conductive soils limiting infiltration simply from low permeabilities, they also reduce percolation as a result of the strong capillary forces limiting infiltration capacity and storage capacity (Levine and Salvucci 1999). The latter results from higher initial soil moisture content and minimal depths to the saturated zone that occur with low conductive soils. These effects on infiltration of rain may explain the differences between the runoff contributions of features that were surrounded by uplands with different conductivities. The results from the runoff analysis highlight the utility of upland sand/overburden-capping as a design tool in CSA restoration attempts to buffer otherwise flashy hydroperiods.

Surface water outflow was limited from the monitored CSAs during the study period with only Mosaic K5 and HP-10 experiencing frequent outflow events. While there were outfall systems at all but one CSA, most were inactive during the monitoring period due to drought conditions and/or consolidation. Reigner and Winkler (2001) observed that water storage on the CSAs was greater than anticipated due to underestimation of clay consolidation and crack development. Of the three CSAs studied by Riekerk and others (1990), only one experienced any surface outflow during the three-year study. Lewelling and Wylie (1993) also observed outflow from only one of the three monitored CSAs during a two-year study. Outflow was observed at PCS SA 10 in the early part of the study, but this occurred when surface water levels were above 1.5 m. The extreme depths at PCS SA 10 would have been limited with a lower outfall invert elevation (Figure 63). The depths at PCS SA 10 are much higher than wetland systems typically experience (Figure 78). PCS SA 01 had an outfall that when active successfully limited water depths, but the effects from adding inserts were obvious (Figure 61). The insertion of inserts essentially mimicked continued consolidation, negating connection between surface waters and outfall structures. As recommended by Reigner and Winkler

(2001), flexible outfall designs may help minimize extreme water depths as well as provide baseflow to downstream systems.

The water balances resulted in residuals which represented underestimation of losses to the systems and that varied among surface water features (Table 57). Similar to the runoff analysis results, the differences in the residuals between features corresponded to the different observed hydroperiods. Water balances resulted in the lowest residuals at the more buffered systems of Mosaic H1 SW-3, PCS SA 10, and CFI SP-1 (Figure 77 and Table 57), while the flashier systems with quick declines, such as Williams Co. and Mosaic H1 SW-1, experienced residuals that were four times greater. The water balance of PCS SA 01 resulted in a negative residual, suggesting that the sand tailings in the upland of this site may have provided groundwater inflow.

The residuals of all water features but PCS SA 01 suggested underestimation of an outflow, particularly ET and/or infiltration. Infiltration was defined as surface water loss vertically and as lateral dike seepage. The potential for dike seepage that was demonstrated by the hydraulic gradients across the dikes was limited by the low lateral hydraulic conductivities, and negligible dike seepages were calculated (Tables 60 and 61). Though CSA dikes are typically constructed using overburden material, CSAs are typically bowl-like, resulting in layers of clay that extend to the height of the dikes on the inside sides of the CSAs (Lewelling and Wylie 1993). Also, because clays settle out slowly, the clays are typically the thickest at the lowest portion of the CSAs where surface water accumulates and where dike seepage could be the most probable (Reigner and Winkler 2001). Riekerk and others (1990) used heat-pulse flow meters in groundwater wells along the perimeter of two elevated clay settling areas and determined that there was lateral flow out of the systems ranging from 4.3 to 8.7 inches/year. Murphy and others (2008) measured natural solute and stable isotope signatures of waters within and outside a CSA to evaluate the hydraulic connectivity to the surrounding landscape. The authors determined that shallow CSA water was present in significant concentrations in waters surrounding the CSA, indicating outflow from the CSA likely occurred through the CSA dike. So again, dike seepage may also vary among CSAs depending on water levels and dike construction.

The groundwater analysis demonstrated that, in most cases, water tables in the surrounding uplands were elevated above the surface-water elevations, suggesting the potential for groundwater inflow to the water features (Figures 137-143). Groundwater inflow to the water features, however, was not demonstrated with the water balances, and in fact, the opposite was suggested (Table 57). Again, PCS SA 01 was the exception. The low hydraulic conductivities (Table 62), especially at Mosaic K5, Williams Co., and Tenoroc-4, may have limited groundwater flow. Dingman (2002) gives typical saturated hydraulic conductivities for clayey soils ranging from 1.03 E^{-4} to 2.45 E^{-4} cm/s.

As discussed in Levine and Salvucci (1999), lower conductive soils that limit vertical and lateral distribution of water typically result in a water table with elevations that strongly reflect the surface relief. Dingman (2002) points out that the higher the conductivities, the more subdued the reflection. The clayey soils of the sites in this study often resulted in water table gradients that mimicked surface elevation gradients as observed with the elevated upland groundwater elevations compared to surface water elevations. The reflection occurs as a result of a feedback mechanism provided by the strong coupling that occurs between the vadose zone and water table. This coupling is strongest with shallow water tables and soils with high capillary fringe heights. Higher water tables and capillary forces tend to increase groundwater loss to the vadose zone through ET while increasing runoff. As water tables rise to depths that are essentially tension-saturated, runoff is decreased (as discussed above) and ET is increased, thereby decreasing the water table elevation (Levine and Salvucci 1999). However, as the water table decreases, percolation of rain increases as a result of lower soil moisture and storage capacity (Dingman 2002) and ET becomes more limited. These mechanisms create feedbacks that tend to direct the water table to some depth, where there is a balance between groundwater losses to capillary rise and gains from intermittent percolation (Levine and Salvucci 1999). With low lateral conductivities and minimal lateral movement of groundwater, this feedback mechanism dominates groundwater movement and creates water table relief similar to surface relief.

Levine and Salvucci (1999) further discuss the relationship between the capillary forces and groundwater movements. Water tables which are fairly shallow and experience large capillary fringes cause groundwater discharge to be primarily via ET versus discharge to lower surface water features. This further explains why elevated surrounding water tables were observed in this study while groundwater inflow to the water features was often not. This phenomenon, however, may develop groundwater troughs that create gradients that affect flow between surface water systems and extremely local groundwater. Elevated transpiration from phreatophytic vegetation surrounding a water feature may intercept groundwater flow into a surface water system while creating a cone of depression (Sophocleous 2002). The cone of depression surrounding a water feature develops a gradient to provide outflow from the surface water system through its bed.

Rosenberry and Winter (1997) performed a detailed study of groundwater connection between two nearby wetlands, one elevated relative to the other. The authors measured hydraulic conductivities using slug tests within groundwater wells that ranged from 2 E^{-4} to 6 E^{-4} cm/s and observed significant capillary fringe heights. The conspicuous gradient from the elevated wetland to the other often did not result in net groundwater movement with the gradient. Observations from twelve groundwater wells instrumented within the 150 m that separated the two systems revealed the formation of groundwater mounds and troughs. The former interrupted groundwater flow between the systems and resulted from shallow water tables and strong capillary forces creating large groundwater responses to rain events. Rosenberry and Winter (1997) found groundwater troughs, as discussed in Sophocleous (2002), surrounding the wetlands that interrupted groundwater inflow to the wetland and created outflow from the wetland to the trough. In summary, the strong capillary forces determined with the moisture release curves (Figures 169-171) and low conductivities of the clay uplands (Table 62) support mechanisms to: (1) create strong coupling between water table and vadose zone; (2)

limit discharge to downstream water features; (3) increase ET flux, limit infiltration, and increase runoff; (4) create mounds and troughs; and (5) limit groundwater circulation.

While the residuals from the water balance and the above discussion suggest zero groundwater inflow to most water features studied, an analysis of the diurnal surface water fluctuations with the White (1932) method often indicated otherwise. The analysis calculated positive exfiltration into most sites and often rates over one cm/day (Tables 71-74). Applying the White method also resulted in extremely high ET rates, with over one cm/day frequently calculated for the winter months. A closer examination of surface water increases at night, which were experienced by most sites, revealed a declining rate of groundwater inflow at night (Figures 146 and 148). The White method assumes constant groundwater flow during the entire day and this assumption was not met. Hill and Neary (2007) also observed changing groundwater inflows at night, which resulted in high calculated ET rates. Though the groundwater rates here were conservatively calculated by not including the initial increases immediately after sundown, ET was still likely overestimated. The calculated ET rates for winter were well over typical values and maximum ET calculated with incoming radiation and latent heat of vaporization (Jacobs and others 2002; Mao and others 2002). The same held true for the calculated summer rates, just not to the same degree.

The most conservative ET estimate was calculated with the average decline over 24 hours, assuming zero net groundwater flow to the surface water system. This assumes a transient groundwater flow where groundwater inflow at night is balanced by groundwater outflow during the day, and a hypothesis is offered referring back to the discussion of groundwater troughs. As ET is diurnal, so may be the trough that develops around the water feature. The low specific yields of the clay soils (Dingman 2002) and strong capillary forces may significantly increase the cone of depression during the day and create gradients for groundwater outflow. As ET ceases at night, the forces to create the trough would be relaxed, allowing recovery of both the trough and surface water system. Dingman (2002) suggests the majority of groundwater to surface water connections occur through the littoral zone. The good connection between the surface water and immediately adjacent local groundwater was observed with the Groundwater-1 wells (Figures 137, 140, 142, and 143), and such a connection may provide diurnal and transient exchange between the surface water system and extremely local groundwater. An increased number of wells around and from a water feature would provide more information to test this hypothesis. Nevertheless, the most conservative ET rates during the summer were often 1.5 to 2 times greater than typical rates, while winter rates were more comparable (Tables 71-75) (Jacobs and others 2002; Mao and others 2002, Lu and others 2005). Again, this is the most conservative estimate and was calculated without including the observed groundwater inflow at night as done in the White method, which significantly increases the estimates.

The water balances and the diurnal curve analysis of surface water levels at PCS SA 01 demonstrated that exfiltration to the surface water feature occurred. The higher conductive soils that surround the water feature at PCS SA 01 may limit the extent of trough development due to lower capillary forces (shown for sandy soils with the moisture release curve for the SP-1 upland in Figure 172). Also, the lower capillary fringe likely decreased water table and vadose coupling that promoted better groundwater connection, which along with a minimal trough, resulted in net groundwater inflow to the water feature. While assuming the most conservative estimate of ET negates net groundwater inflow to the other monitored features, zero groundwater inflow should not be a definitive conclusion of all conditions. Murphy and others (2008) observed lateral transport of groundwater on a CSA with significant inflows to an isolated surface water feature on the site. The diurnal analysis was performed using surface water levels from 2007, which was a fairly dry year. Rosenberry and Winter (1997) suggest that trough development is much more significant during drier years when water levels are low. The buffered regime at PCS SA 10 and its higher conductivity relative to the other sites suggests that it, too, could experience intermittent groundwater inflow.

The water balance residuals for Williams Co. and Mosaic H1 SW-1 were much higher compared to other sites and to another feature on Mosaic H1 (SW-3), demonstrating differences both among and within sites (Table 57). The analysis of diurnal fluctuations of these sites indicated that net infiltration had occurred from their systems (Figures 149 and 150). The pathway of the groundwater outflow was unknown, but speculated to be vertical, as these features are low areas in their respective watersheds. Reigner and Winkler (2001) calculated negligible infiltration rates using measured permeabilities, whereas Lewelling and Wylie (1993) witnessed the possibility of higher infiltration rates through underlying overburden piles.

Temporal Hydrologic Modeling

The temporal modeling with Penman-estimated ET, the runoff models, and the omission of infiltration overestimated stage and supported the water balance results. PCS SA 01 was an exception as stage was underestimated. Inclusion of infiltration terms resulted in end-stage matching between predicted and actual but caused summer decline to be underestimated and winter decline to be overestimated. The application of seasonal coefficients improved the models' predictions (Table 76). Without inclusion of infiltration, seasonal coefficients during the non-growing season for PCS SA 10, Mosaic H1 SW-3, CFI SP-1, and Mosaic K5 were near one, indicating that winter decline could be explained by Penman-estimated ET and without infiltration rates. The required growing season coefficients to accurately predict decline without infiltration for these sites ranged from 1.3 to 1.5. The need for seasonal coefficients to adjust ET rates and since the inclusion of infiltration rates alone did not create accurate models, again supports the underestimation of ET for these systems using empirically derived, climatic-based methods. To keep coefficients for Williams Co. and Mosaic H1 SW-1 within this range, inclusion of significant infiltration rates to the models was required, supporting the results from the diurnal curve analysis (Table 76). The model for PCS SA 01 required an exfiltration rate of 1.2 mm/day to avoid under-predicting stage, which agrees with the results from the water balance and drawdown analyses (Table 76**.** This rate was determined when both seasonal coefficients were 1.0 and would increase if coefficients required for other models were applied.

The temporal models' predictive abilities were demonstrated, and high non-linear correlation coefficients were obtained (Table 76). Similar to the spatial models, however, sites required different decline rates and characteristic runoff responses, supporting the observation of differing hydrologic regimes among and within CSAs. Infiltration rates, either positive, negative, or zero, were applied to the models. Different upland soil types control the runoff characteristics and surface and groundwater interactions, significantly affecting hydrological regimes and demanding site-specific analysis when predicting hydrology.

Ecohydrology

Wetland boundaries are not limited to areas of flooding and include areas of permanent and temporary soil saturation. Therefore, an understanding of groundwater dynamics and its effects on soil moisture levels and plant behavior is also important when wetland restoration includes, as it should, transitional zones between upland and inundated conditions. To address these interactions, a study of the ecohydrologic relationships between *Salix caroliniana* and CSA soils was included. The strong capillary forces of the clayey soils, which increase the coupling between the water table and vadose zone as discussed above, were calculated with the moisture release curves (Figures 161-171). Fringe heights over a meter resulted and were greater than typical heights of clayey soils, which range from 50 to 65 cm (Dingman 2002). The moisture release curves also indicated saturation levels of 57 to 60% VWC, which are slightly higher than typical clays (Dingman 2002). Saturation of typical clays occurs at approximately 50% VWC, and the higher saturation levels found here demonstrate the higher porosity of CSA soils compared to typical clays. Permanent wilting may be induced when soil moisture in the clay decreases to approximately 40% VWC (Figures 161-171). The sand-dominated upland at CFI SP-1 resulted in a much different moisture release curve and one typical of sandy soils (Figure 172) (Dingman 2002). Saturation levels occurred at 33% VWC, capillary fringe heights were near 20 cm, and PWP corresponded to a VWC of 10%.

Coupling the moisture release curves with the recorded soil moisture and groundwater regimes further demonstrated the role of the capillary forces from the clays. Though not observed in all cases, groundwater levels as deep as one to two meters maintained constant soil moisture near saturation levels well above the water table (Figures 174 and 179). It was also observed, however, that the top 10 and 25 cm of the soil profiles in upland areas had fluctuating soil moisture levels and ones frequently below soil VWCs that may induce PWP. The soil moisture levels collected at the saturated stations of Williams Co. and CFI SP-1 revealed that high soil moisture levels can be maintained up to ground surface in a frequently inundated area during an extended dry period (Figures 183 and 189). The results exhibit the ability of clayey soil to extend saturation zones beyond inundated areas and to maintain suitable saturation for wetland species in a historically flooded area during a dry period.

The upland and transitional stations at CFI SP-1 demonstrated the effect of a sand-clay mix fill (Figures 185 and 187). Saturation levels, as indicated by the moisture release curves, were never reached throughout the entire soil profile that was monitored at the upland. Furthermore, recorded soil moisture levels frequently reached values which may induce PWP, despite the VWC for PWP in sands being substantially lower than that for clay (Figures 185 and 172). The differences of soil moisture regime in the top 30 cm of the soil profile at the transitional station compared to regimes at greater depths were apparent, exhibiting the soil heterogeneity with depth that is created by sand-clay fill (Figure 187). So while a higher conductive upland such as a sand cap may buffer hydrologic regime of water features, it reduces the saturated zone that extends from the feature.

An analysis of the root biomass allocation revealed *Salix caroliniana*'s ability to penetrate the clays without preclusion and to depths of saturation and into the water table (Figures 190-196). The rather deep rooting that was observed is consistent with other studies which found rooting depths greater in clays and humid climates (Snyder and Williams 2000; Schenk and Jackson 2002; Porporato 2000). Such conditions create a constant source of available water through capillary forces that can be relied on rather than periodic rain events and a shallow rooting system. Increased rooting depths also act to increase the coupling between the water table and vadose zone and its effects, as discussed above (Rosenberry and Winter 1997).

The combination of fairly extensive rooting depths and the strong capillary forces of the clays help explain the success of *Salix caroliniana* across large, including never flooded, areas on CSAs. These results demonstrate the potential for including large transitional zones at distance from surface-water features in wetland restoration plans. The results also heed caution to planting methods, however, as initial rooting depths of seedlings should be sufficiently deep enough to reach wetter zones unless they receive supplemental irrigation. So while the clay may increase the flashy nature of water features, it may also act to increase wetland environment beyond the typically flooded areas.

Transpiration studies on *Salix caroliniana* supported the conclusions from coupling rooting depths with soil moisture profiles that trees are able to root deeply and thus reach available soil water. In terms of stem flow, trees did not experience stress conditions in the upland locations, where higher rates of stem flow occurred compared to inundated conditions (Figures 198 and 201). When transpiration was expressed as stand level transpiration, however, this trend was reversed, suggesting that stress conditions were also not caused by inundated conditions, and that *Salix caroliniana* biomass controls its contribution to total ET (Figures 199 and 202). In the wetter regions of Mosaic K-5 and CFI SP-1, stand level transpiration from *Salix caroliniana* alone was 1.7 and 1.2, respectively, times greater than Penman-estimated total ET using on-site climatic data. The high stand transpiration supports the results from the water balance, drawdown, and modeling analyses as well as further supporting the possibility of trough development around water features.

The analyses of ET with surface-water levels and measurements of transpiration resulted in high, yet consistent rates, and some explanation is offered. CSAs are eutrophic systems, where water availability is more than sufficient through capillary forces and pure stands of *Salix caroliniana* occur. These species are phreatophytes which are typically pioneer and limited to small disturbed, wet areas. Most CSAs are in arrested succession because of limited seed source (Odum and others 1991) and, thus, allow *Salix caroliniana* to establish at higher densities and across larger areas than would be typically found. Transpiration is proportional to primary productivity and the extremely high rates observed in this study provide evidence of the potential for high ecosystem productivity on CSAs. It should be noted that the evapotranspiration rates determined in the diurnal analysis were only for the surface water features, and since the transpiration analysis exclusively considered *Salix caroliniana*, no rates for total ET from the upland areas were reported. Therefore, these results should be used with caution for entire CSA evapotranspiration, as rates determined from the wetter regions should not be applied system-wide.

EVALUATION OF B. RUSHTON FIELD TRIALS

Summary

Evaluating the progress of a created wetland after a long period of time (20 years in this study) is valuable for determining which species are appropriately adapted for site conditions and what the role of these species may be in the development of an ecosystem. Though tree survival was higher on the sand-clay mix soil than on pure clay or sand, hydrology and site disturbances were more important factors than soil type in determination of tree survival. Each wetland tree species survived in positions along a hydrologic gradient that fit a species-specific tolerance range for inundation. This positional range was more apparent for a species after 20 years than it was after 1 or 3 years. This information provides a good indicator of long-term hydrology within these plots and would be valuable for future planting efforts on these sites.

Tree growth among surviving individuals was just as high on pure clay soils as on the sand-clay surface. By this measure, established trees were successful on clay. Nevertheless, the sustainability of planted tree populations on CSAs is uncertain. In most cases offspring of the planted trees were scarce after twenty years. Models showed that the size of tree populations on two sites, each with few offspring, will not grow significantly or will possibly decline after 50 years, assuming high survival of current mature trees. The cause(s) of the low numbers of new seedlings still needs to be clarified. The presence of a high number of seedlings on one clay site proved, however, that the clay soils alone do not prohibit seedling establishment.

Planted plots are more structurally mature, as measured in total woody basal area, than non-planted areas, and this is promoting the accumulation of soil organic matter, but the rate of accumulation does not always exceed accumulation under other CSA

communities, as volunteer vegetation also contributes to organic matter accumulation. No strong relationship between planted plots and understory vegetation has yet emerged on the selected CSAs. The assemblage of understory vegetation appears to be more strongly determined by the site surroundings and the plot hydrology. The influence of the trees may become stronger as they continue to mature. For sites planted with trees, the intentional introduction of additional species in the understory could provide the means for creating a more diverse community.

Tree Populations in Relation to Environmental Factors

Tree Survival by Site and Species

The presence of wetland trees planted 20 years ago on multiple CSAs is an indication that conditions are similar to those in which these wetland trees have evolved to persist. *Taxodium distichum*, both *Fraxinus caroliniana* and *pennsylvanica*, and *Nyssa aquatica* survived on all sites chosen for the study, though not in equal percentages. Though the typical lifespan of trees of these species is much greater than 20 years, their growth and healthy condition on some sites herald continued persistence. For the species that did not survive at any of the sites, questions remain as to the site factors that they were unable to tolerate. Mature trees of some of the other species planted by Rushton were present on one or more of the study sites. *Acer rubrum*, which survived in small numbers at some sites, was dominant in the understory under canopies with many mature individuals at OHW, and to a lesser extent at TEN. This species also occurs in high densities in some areas of CFI, from which they have recruited from an adjacent floodplain. *Ulmus americana* has also been recruited on some of these same sites, though to a lesser extent than *Acer rubrum*. *Quercus laurifolia* is not uncommon at CFI and OHW. Isolated individuals of *Persea palustris* were found outside the sampled area at OHW. The failure of these species to persist in these planted plots does not preclude their capacity to survive on CSAs, but does indicate a relatively poorer survival capacity in the conditions to which the plots were subjected.

Overall, *Fraxinus caroliniana* had a very high survival rate, though it was planted in a limited range of water depth and a smaller number of individuals were planted. *Taxodium distichum* was planted more than any of the aforementioned four species and over a range of water depths, most of which it tolerated. In terms of survival, it was the most successful species of the three in the cypress-gum plots.

One- and three-year survival was a good predictor of 20-year survival for these four species. Though tree populations continued to decline in years 3 to 20, albeit more slowly than in the early years, the relative proportions of species were consistent across sites. In other words, a similar survival trend was present for these species, and the species with the highest survival after three years was in almost all cases the species with the highest survival after 20 years. This perhaps indicates a similar response to environmental stresses among the species.

Tree Survival and Hydrology

Time allows for a clear determination of a suitable landscape position of a wetland species relative to its period of exposure to saturated conditions and the depth of inundation. At CFI, water depth did not preclude 20-year survival among the trees living after year 1; however, hydrological factors may have had an effect on the likelihood of survival of *Nyssa aquatica* and *Fraxinus pennsylvanica*.

OHW was likely affected by a disturbance event that affected the drier end of plots R2A and R2B (see Figure 209), thus water was not likely the key factor in mortality of the trees in the drier area. *Fraxinus pennsylvanica* did not tolerate the wetter locations of this transect, though it appears to have tolerated the same average depth at CFI. That *Fraxinus pennsylvanica* did not tolerate locations where the average depth was 0.2 and 0.3 m at OHW, though it did tolerate those depths at CFI, could be interpreted as a greater tolerance for standing water in sand-clay than in clay. But there are likely differences in the hydrologic regime between the two sites that could have affected tree survival. In both plots, trees are growing on the fringe of a pond where surface-water outfall occurs at a given depth. Though data are not available to determine at what water level relative to the trees that surface outflow occurs in the two basins, it is possible that water could be retained longer at the same average water level depths at OHW, increasing the period of inundation. This case exemplifies the difficulty of inferring hydrologic similarity from monthly measurements over a single growing season. Inside the plots perhaps the location of surviving trees relative to one another is a better indication of hydrology than monthly water level measurements, but this was an assumption that could not be made within this study.

The shallower water depth distribution of surviving trees in the TEN basin (Figure 210) likely does not represent the average depth of water that trees were exposed to before the basin was ditched in 2001, when average seasonal depths for all trees were likely greater. Hydrologic factors may have impacted mortality, as the less tolerant *Fraxinus pennsylvanica* did not survive in the deeper part of the range where *Taxodium distichum* did, but the animal grazing (see Table 82) noted during the initial years of establishment was likely also a major factor in the high tree mortality in this basin.

The long-term change in the hydrology on CSAs due to the continuing settling of the clays is a challenge to long-term wetland creation unique to CSAs. But this study only revealed anecdotal indications of an effect of clay consolidation and resultant hydrologic alteration on planted trees. At TEN, laterally branching roots of *Taxodium distichum* and *Fraxinus caroliniana* with rigid epidermal cells not typically found above the ground surface were found in two basins. Faint clay stains were present on these roots, which were as much as 3.5 feet above the ground surface. These root features are potentially signs of clay consolidation, but since the basin hydrology was altered by ditching in 2001, they could also be remnants of a dramatic decrease in water levels.

Tree Growth Comparison between Sand-Clay and Clay Sites

Comparison of the effects of soil medium on tree growth could not include sand-capped sites because there was no control for the effect of water level on tree growth. The data clearly indicate that trees survived in greater numbers after 20 years on the sand-clay site than in the clay, despite similar survival after one year for *Taxodium distichum* and *Fraxinus pennsylvanica*. For all three species planted in cypress-gum plots, survival was better on the sand-clay site. Nevertheless, the soil medium is not the most probable explanation for this difference. Initial growth after one year was similar for cypress-gum plots on CFI and OHW, on which no notable growth occurred. Twenty-year survival on OHW R2A and R2B was affected by the death of all trees in 1/3 of the plot. Because all species died and because the area experienced similar water-level conditions to part of CFI (sand-clay) on which individuals of all species survived, it is probable that one or multiple disturbance events, likely fire, caused the mortality rather than the water level or the clay soil. The domination of that area now by a fire-adapted species, *Imperata cylindrica*, and reports of fires that consumed trees in nearby plots provide further evidence of this mortality hypothesis on OHW.

Where *Taxodium distichum* and *Fraxinus pennsylvanica* survived, the halves of the populations in areas with higher average water tables showed little or no advantage in growth over those in shallower water. This could be partially confounded by the tendency of these species to buttress in response to prolonged water conditions. Richardson and Kluson (2000) found that trees in saturated conditions, but with infrequent standing water, grew faster than trees in standing water. However, it should be noted that these results were found on non-CSA sites.

On TEN, the poor initial survival of some of the trees in the basin used in the survival figure (Figure 210) was reported to be partially due to heavy grazing. Grazing significantly reduced initial tree growth, an important indicator of future survival, and thus likely was the principle cause of the high mortality in the following years. However, water levels possibly resulted in *Fraxinus pennsylvanica* death in the deeper areas and all species in the extreme dry areas. Though *Fraxinus pennsylvanica* survived into a much deeper average water depth on other sites, it is likely that this basin stored more water before the hydrology was altered in 2001 and that these trees then earlier were subjected to more frequent inundation. High survival percentages in other plots in clay where the same trees were planted, like TEN H6, is further evidence that, given appropriate hydroperiods and freedom from devastating disturbance, viability of *Taxodium distichum* and *Fraxinus spp.* species on clay is good.

Across the board, *Nyssa aquatica* had poor initial survival in the clay sites, as the clay may have impeded the establishment process, as it is the most likely explanation considering that the hydrology was similar across soil types. Yet once the species established, it grew as well or better and survived at a similar rate on the clay.

Recruited Trees

The scarcity of recruited trees in the periphery of most plots made it impossible to make broad inferences about the conditions appropriate for seedling establishment on CSAs. Lack of data on seed production, germination success, and seedling survival did not enable a determination of the causes of absence of recruits.

First-year *Taxodium spp.* seedlings cannot tolerate long periods underwater (Wilhite and Toliver 1990). On TEN H3, a particular abundance of new seedlings emerged in the spring and early summer of 2005, where in May water levels dropped below ground but remained close enough to the surface to maintain saturated conditions appropriate for germination. However, water levels rose and likely remained high enough to completely inundate 72 of 85 of these seedlings. This rise in water level is the most likely explanation for the high mortality among these first-year seedlings. If those seedlings that were inundated are assumed to have died during the period, the survival rate for the remaining seedlings up to 100 cm would be close to 90%. Though unique in the density of seedlings in this study, this plot provides evidence that given the presence of viable seed and appropriate water levels, *Taxodium distichum* can germinate and establish on a CSA, and that water levels are of critical importance in the establishment process.

The source of seedlings present on some of the sites was impossible to establish when other mature trees had been planted by other parties. At CFI, more than 500 *Taxodium distichum* seedlings had been planted on the site since the Rushton planting. Mature trees not planted by Rushton are present just off the deeper margin of the plots and in between plots in cases. It could not be determined with certainty that the recruited trees found inside the plots were offspring of the planted trees. At HOM *Taxodium distichum* trees had been planted in the same basin a few years before the Rushton plantings. The recruited trees found at HOM were perhaps planted or offspring of trees from the previous planting.

Seedling establishment of wetland tree species depends heavily on a gentle rise in the relative topography of the landscape. In natural floodplain systems, the extent of the spread of the population is determined by the extent of the flood zone. In some CSAs the flood zone is restricted due to a steep elevation gradient, often a residual of the mine cut-spoil pile pre-fill topography. This topography may restrict the area favorable to wetland tree seed establishment, which requires fluctuating water level conditions for adequate but tolerable moisture.

The size class distributions show normal to left-skewed shape distributions for most sites. A right-skewed or inverse-J shape distribution is a sign of a growing population dominated by smaller individuals (Manabe and others 2000). Overall scarcity of new seedlings at the sites poses challenges to future population success. In all species of planted trees monitored in the study, at least some individuals had reached a maturity to produce seed based on what is reported for individuals of those species (USDA 2004). Though there was no formal collection of seed production data, there were records of seeds present on trees or floating in water for each of the species present. If the trees continue to survive it would be natural that they would become more fecund as they grew.

Though this study shows that failure of seedling establishment is not endemic to CSAs, studies need to be conducted to show if establishment presents any particular challenges. Further study into seedling establishment and growth could reveal any obstacles that exist on CSAs related to soil clay content or vegetation cover. But in order to be conclusive, any such study needs to take into account all stages of seedling establishment, including seed production, dispersion, viability, germination and initial survival along a variety of environmental gradients typical of CSAs.

Tree Population Model

Because this was a young population there was not good data on survival of older trees. Reclamation of phosphatic clay settling areas did not begin until the early 1980s and therefore there is no reference for longevity of *Taxodium distichum* in these areas. To fill in the data gap, survival probability of larger trees was assumed to continue to increase in larger size classes. The estimated survival probability of the largest size class of *Taxodium distichum* in the models was consistent with the survival probability of the largest size class in models of other woody species (Zuidema and Zagt 2000). Since the mortality of the largest size classes was the most sensitive parameter in the model, the confidence of the model could be improved by real data of large tree mortality.

The probabilities of growth, survival, and reproduction are affected by the hydrologic conditions. Incorporating the effect of different hydrologic regimes in the transition probabilities of multiple transition matrices is one technique for implicitly accounting for the effect of hydrology on a wetland tree population (Lytle and Merritt 2004). For these models, a time series of data and a hydrologic record would be necessary to build this model.

The small changes in population size predicted by the models for trees on CFI and OHW are a consequence of both high survival probabilities of larger trees and low reproductive probabilities of mature trees. These same trends would likely have been present in models of a number of the other tree populations in this study, but such trends cannot yet be generalized for *Taxodium distichum* or other tree populations on CSAs.

Characteristics of Successful Species on CSAs

A common trait among the tree species that survived on multiple sites after 20 years (*Fraxinus caroliniana, Fraxinus pennsylvanica, Nyssa aquatica, Taxodium distichum*) is the ability to tolerate anaerobic conditions for an extended period of time during the growing season. The least tolerant, *Fraxinus pennsylvanica*, can tolerate inundation for up to 40% of the growing season (Fowells 1965). Each of these species has special adaptations that permit extended survival in periods when the root zone is saturated, including adventitious rooting and buttressing.

Also common to these species is the ability to resprout from the root stock and to coppice (resprout from a stump) following disturbance. For environments that may be frequently exposed to disturbances, especially fire, resprout ability could be important for long-term survival (Pausas and others 2004). Evidence of resprouting was present in each of the four species.

These four species naturally occur in riverine swamps (Myers and Ewel 1990). *Fraxinus caroliniana* and *Taxodium distichum* are also naturally present in a number of other forested wetland types, such as cypress stands and lake fringe swamps. Two of the species, *Fraxinus pennsylvanica* and *Nyssa aquatica*, do not natively occur in Polk County. The southern extent of the range of these species is in the Big Bend region. Among natural forested wetlands in Florida, these two species are typically restricted to riverine swamps.

The similarity of the natural habitat of these species and the CSA environment may help to further explain their success on CSAs. Characteristics of riverine swamps, a common habitat of these species, include a short hydroperiod and mineral soils, typically containing clays. Plots in the study had a mix of hydroperiods during the 2005 growing season, but *Fraxinus pennsylvanica* and *Nyssa aquatica* were more successful in plots that had a short to moderate hydroperiod. *Fraxinus caroliniana* and *Taxodium distichum* naturally occur in areas with a range of hydroperiod and, on CSAs, were successful in areas with longer hydroperiods. Clay, sand-clay, and sand-capped sites in this study all had low organic matter contents at the time of planting that would fit a mineral soil characterization. Other species found surviving or volunteering in transitional areas, including *Acer rubrum*, *Quercus laurifolia*, and *Ulmus americana*, are also naturally found in riverine swamps. Two species that did not survive, *Gordonia lasianthus* and *Sabal palmetto,* are more often found in ecosystems with sandier soils and less extensive periods of inundation.

Species characteristics are important in determining capacity to survive in the new anthropogenic environment of CSAs, and copying species assemblages that exist in natural wetlands with similar characteristics is a potential method for finding appropriate species. Yet because the CSA conditions are unique, there is no perfect correlate ecosystem from which to select appropriate species. The species that were most successful after 20 years in these plots were those that not only occurred naturally in riverine wetlands, but also were those with the most tolerance for anaerobic conditions and the ability to resprout. A species' biological characteristics and the similarity of its native habitat are more important to tree success in CSAs than native range, confirming an earlier finding by Paulic and Rushton (1991b).

In a 2005 survey of Homeland FM-07, another CSA where trees were planted in 1988 [but not included in this study; see Paulic and Rushton (1991b) for details], a similar assemblage of surviving species was found. *Quercus lyrata*, *Fraxinus* *pennsylvanica* and *Taxodium distichum* were the only species found. *Quercus lyrata* is another native of north Florida riverine swamps adapted to anaerobic environments.

The hydric swamp plots in less wet to more transitional conditions at PRP, OHW, and TEN were mostly devoid of trees or any plot boundary markings. Fire was a likely cause of death at PRP and OHW, whereas circumstances are unclear at TEN. On PRP the transitional areas are dominated by *Imperata cylindrica*. On another site mentioned in the previous paragraph (FM-07) no trace of plots set up in transitional areas was available and these areas were also dominated by *Imperata cylindrica*. At OHW, a mixed forested canopy is now present over transitional plots H2 and H3. *Schinus terebinthifolius* was dominant in the remnants of two TEN transitional plots.

Drier areas are more susceptible to fire and post-fire colonization, and overall had poorer survival after 20 years, leaving the long-term viability of transitional tree species on CSAs uncertain.

Ecosystem Development in Rushton and Reference Plots

Plot Selection and Comparison

Although the study intended to examine whether the surviving Rushton trees have played a role in ecosystem development, there were no clear minimum values of the quantity of trees, tree biomass, or tree cover necessary to reveal an effect. It was not the purpose of this study to find a minimum level of some quantitative measure of the trees at which an effect could be detected, but whether or not an effect on ecosystem development could be detected under the conditions in which trees were present. Because the measurements of ecosystem development had different degrees of spatial precision, it was safer to assume common influence on a plot or subplot when survival of trees was higher and thus spatially more homogenous.

Adequate descriptions of the vegetation composition in reference plots and Rushton plots at the time of planting (1985-1986) were not available to determine if the composition was identical. By selecting areas adjacent to the same water feature with similar hydrology it was assumed that: (1) the vegetation in the areas at the time of planting was similar; (2) the depth and duration of flooding for the Rushton and reference[*] plots was similar; and (3) no significant disturbances that would radically alter the vegetation and/or soil affected the plots unevenly since the time of planting. The comparison of Rushton and reference plots rests on these assumptions, and plots or subplots were eliminated from the comparison if they violated one of these assumptions. Hydrology is perhaps the primary driver of wetland ecosystem development (Mitsch and

---

[*]As a reminder, "reference" plots, as the term is used in this report, refer to non-planted plots on the same CSA selected to represent hydrologic and topographic conditions identical to a given planted plot; "reference" plots are not plots in undisturbed ecosystems, as the term is frequently used in ecological literature.

Gosselink 1993). Thus the most important criterion for selection of a reference plot within the site was its hydrology. Though it was impossible to establish a reference plot in the same water feature at Homeland, the reference plot was within 100 m of the closest Rushton plot and had a similar minimum, maximum, and average depth, and average change in elevation.

In some cases, there was considerable variation of water depth and percent inundation within a group of Rushton plots. Mean water depths at CFI ranged from 0.36 m at R1 to -0.36 m at R6. The mean water depth of the reference plot was appropriately exactly in the middle at 0.0 m, but the difference in depth and percent inundation within Rushton plots was large enough to lead to detectable differences in ecosystem development parameters among the Rushton plots. The R1, R2, R3, and R5 understories were dominated by floating aquatic vegetation, whereas R4 and R6 were dominated by ferns. Yet the understories in these plots were still more similar to one another than they were to the reference plot (see Figure 91). There was also a -0.70 correlation (Table 93) between the water depth and organic matter on CFI, indicating a difference in soil OM within Rushton plots. Hydrologic variation within Rushton plots made delineation of differences from reference plots more difficult.

Structural Differences

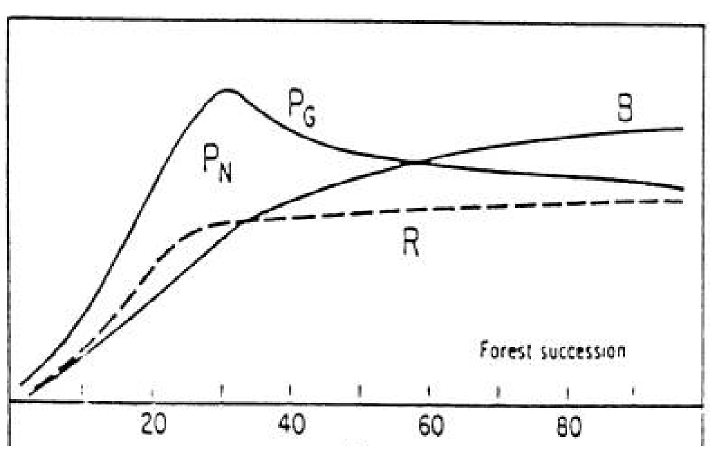
The clearest distinction between Rushton and reference plots was present in the tree and shrub strata. In planted areas with moderate to high survival of planted trees there was significantly more structure at these levels in the plots. Rushton plots had in 9 of 10 cases a more developed shrub and canopy layer. In plots on CFI, in TEN R2A and R2B, and in TEN H6, plot basal area was more than twice as high as what has been found in natural forested wetland systems, including mixed hardwood forest and cypress domes, but this difference is confined to the narrow boundaries of the Rushton plots. However, the estimates of canopy cover interpreted from the canopy photos showed little difference between Rushton and reference plots. A possible explanation is the trend that occurs with the estimation of canopy cover as plot basal area increases (Figure 224). Estimated canopy cover increases very rapidly and then levels off as basal area continues to increase. Generally the Rushton plots had enough structure so that all were near that asymptotic 'level' of canopy cover.

The canopy photo technique was used to estimate the proportion of light blocked by the tree and shrub layers from reaching the understory. Because of the proximity of the shrub level to the camera lens, also true of the understory, the shrub layer potentially had a more significant effect on this estimation. The technique does not estimate layering in the canopy, nor the opacity differences in different vegetative structures. Because there is more opaque, woody structure in Rushton plots and likely more frequent overlap of structure in different strata, the differences in the light reaching the understory could be greater than estimated in Rushton and reference plots.

Soil Organic Matter

Woody vegetation is an important contributor of litter that becomes incorporated into soil organic matter. At TEN, higher percent soil organic matter was found in reference plots dominated by *Salix caroliniana* than in corresponding Rushton plots (pairs 7, 8, 9), though this was not the case at PRP or CFI, where *Salix caroliniana* dominated reference plots. At OHW, two reference plots dominated by *Ludwigia peruviana* had higher organic matter than corresponding Rushton plots. Both of these species are characteristic of wetlands on CSAs, and may result in faster organic matter buildup than planted species, but this trend is not consistent across all sites. Other factors, such as fire frequency, also were important. At Peace Park, frequent fire and high tree mortality likely caused high deposition of woody particulate matter in Rushton plots that led to high soil organic matter. The presence of floating woody debris and burn scars on dead stumps was qualitative evidence of this effect. Surprisingly, correlation of water depth with soil organic matter was negative at most sites (see Table 93), which contradicts what is commonly found in wetland systems, where sediment deposition is greater in lower areas (Hupp and Bazemore 1993). This could be due to lack of vegetative colonization of deeper areas.

In wetland systems, wood biomass and soil organic matter often represent the largest storages of organic matter (Megongial and Day 1988). In Rushton plots, a larger amount of total basal area and smaller amount of a soil organic matter relative to non-planted plots indicate that relatively more organic matter is bound up in living biomass. A high percentage of the organic matter pool tied up by living organisms has been proposed as an indicator of a more mature ecosystem (Odum 1969). In a transition period, the net production of organic matter theoretically peaks and declines as biomass continues to increase (Figure 286). Although gross production is likely still increasing in these systems, as indicated by continual tree growth and a greater total basal area in older sites, a greater proportion of the organic matter is being tied up in woody biomass and less deposition to the soil is occurring.



Note: From Odum (1969). PG = gross production; PN = net production; R = respiration; B = total biomass.

Figure 286. Succession in a Forested System.

Understory Vegetation

For most plots, the coverage of plants in the understory, the species richness, and the species evenness was similar among Rushton and reference plots. The similarity among Rushton and corresponding reference plots was made apparent by the NMDS (Figure 225). A distinct site-based grouping of understory assemblages emerged in this plot. CFI (without the reference plot), OWH, and TEN are clustered by themselves, and PRP and HOM overlap. This finding demonstrates the importance of site surroundings on understory composition. The dispersal of propagules from outside is the only plant source in CSAs, as there is no seed bank in the clay from which plants can emerge. Seeds must be carried in by wind or animals, and this process is limited by the distance to the nearest seed source. Interestingly, the HOM and PRP sites, which overlap on the NMDS, are within a mile of one another and likely share the same source (the Peace River floodplain) of propagules.

Alternatively, propagules of wetland species other than trees could be brought in during the reclamation process. This was done at CFI, where *Nephrolepis spp.* were planted under the canopy of Rushton trees.

There was some similarity in the understory across sites based on plot hydrology. Floating aquatics, primarily duckweed (*Lemna minor* and *Spirodella polyrhiza*) and *Salvinia minima*, were often the most prevalent vegetation on wetter transects. Where they occurred they often accounted for the majority of cover. Though these species have limited to medium shade tolerance, they were present in Rushton and reference plots, without a clear trend in a relationship between basal area or canopy cover in their occurrence, except in Pair 5 at OHW and Pair 6 at PRP.

An exception to the trend of similarity among species assemblages may exist at HOM, where the species present in the reference plot were more typical of a freshwater marsh than a shrub or tree-dominated system. The Rushton trees planted at HOM may be directing succession toward a forested wetland, whereas it otherwise might be developing into a marsh.

Relationships Among Measures of Ecosystem Development

The correlation matrices presented by site show some across-site similarity in relationship between causal and response variables for OHW and TEN, and also for PRP and HOM. Generally weak correlations are present between Rushton basal area and the response variables. This could be because of they are older sites abutted on one side by a source of propagules, and because the ecosystems' reference plots are more developed on these sites, dampening the effect of planted trees. Still, there are large differences in total basal area and thus more organic material stored in the living biomass in the Rushton plots on these sites, so differences do exist.

On both PRP and HOM, the Rushton plots stand out more in their structural differences with reference plots than at other sites. These structural differences appear to have a strong effect upon the understory vegetation, and clearly contribute to increased organic matter buildup. Planted trees may have more detectable influence on ecosystems development on less vegetated sites.

WETLAND REVEGETATION FIELD TRIALS

Marsh Revegetation

Several planted herbaceous species were able to establish within the Mosaic H1 and PPW-3 marsh revegetation sites, and either maintained or increased in frequency after revegetation in 2005 and 2006. Likewise, several trees species were able to survive and increase in height over the period of record at each site. Lack of success by other planted species, however, should not indicate inappropriateness of these species on CSA wetlands in general. Drought conditions over the period of record for both sites strongly influenced wetland hydroperiod and the dynamics of planted and volunteer vegetation. Volunteer species recruited heavily to both sites over the period of record. Different pre- and post-volunteer species management and more controllable hydrologic conditions could have yielded a different set of results for both planted and volunteer species.

Due to their isolated position on the landscape and negligible offsite infiltration, wetland hydrology on CSAs is largely dictated by precipitation and ET (Callahan and others 1991, Reigner and Winkler 2001). Therefore, climatic conditions strongly affect site vegetation dynamics. Outfall structures are sometimes installed on CSAs for site drainage after filling; however, settling of clay substrate typically results in ground elevations much lower than outfall structures at the perimeter of the CSA, making them inactive for the majority of the year or only when extreme climatic conditions occur, depending on the site's topography and age (Reigner and Winkler 2001). Even if outfall structures exist and are actively adjusted for settling, wetland features may become hydrologically disconnected from the portion of the site controlled by the outfall, rendering them useless in controlling the wetland feature's water levels. The H1 marsh site has no working outlet structure that would function to control water levels by dampening spikes in water levels following storm events. Likewise, the PPW-3 marsh site is isolated from other existing wetland features on the CSA, as well as any relic outfall structures.

Mosaic H1 Marsh

High water conditions that recede only through ET, especially immediately after planting occurs, may eliminate the success of species adapted to more shallow water and transitional wetland environments. This may have been the case with *Peltandra virginica*, *Bacopa caroliniana*, *Muhlenbergia capillaris,* and *Panicum hemitomon,* which were planted within the graminoid planting zone at the H1 marsh, and experienced large declines in species frequency in the first year after planting. Spikes in high water levels can also negatively affect plantings by causing immediate mortality through dislodging and floating of individual plants, as seen with the *Scirpus californicus* planting, and most likely the lily marsh planting of *Nymphaea odorata* and *Nuphar polysepala* at H1 in 2006. Outfall structures and site topography, if properly designed prior to planting and maintained throughout the life of the revegetation, could help stabilize wetland hydrology on CSAs, allowing greater control and confidence in plantings, both in deepwater and shallow wetland environments.

Since many wetland features on CSAs are strongly affected by variable precipitation, drought conditions can cause drastic changes in wetland hydroperiod from year to year. Lack of rainfall during the 2006 winter and growing season decreased the extent and length of flooding at the marsh and several planted species responded positively. The three species within the flag marsh planting zone, *Saggitaria lancifolia*, *Thalia geniculata*, and *Pontederia cordata,* were qualitatively observed flowering, producing new leaves and increasing in stem count over the 2006 growing season. This is reflected in the species' frequency data, with the exception of *Thalia geniculata*. Other species also performed well, including *Scirpus californicus*, *Spartina bakeri*, *Eleocharis cellulosa*, and *Juncus effusus*. These species were able to persist through high water conditions, drought, and on clay substrate. Despite the initial spike in water levels followed by drier than normal growing season conditions, most tree species planted at the periphery of the marsh survived well. *Taxodium distichum* and *Cephalanthus occidentalis* had the best survival after one year, and *Persea palustris* had the worst, possibly affected by prolonged high water conditions immediately after planting. Most tree species grew and survived along the entire gradient of hydrologic conditions over which they were planted within the first year; however, sample size for each species was entirely too small to draw any sound conclusions about species zonation. *Gleditsia aquatica* and *Taxodium distichum* had the highest growth of all planted species, as well as high percent survival after one year.

Eupatorium capillifolium responded positively to drought conditions, by volunteering across the H1 marsh over the 2006 growing season. Neither this species nor *Polygonum hydropiperoides*, which volunteered across the majority of the site, appeared to out-compete nor physically damage planted species, although experiments to test this hypothesis were not performed in this study. *Typha latifolia*, which dominated the marsh prior to planting, only reestablished in the wettest areas. *Ludwigia peruviana* and *Baccharis halimifolia* that occurred within monitoring plots were observed to be small sprouts, only several centimeters in height. Drier conditions, available seed source, and lack of post-planting species management allowed a thick regrowth of *Ludwigia peruviana* along the northern and western periphery of the H1 site, which would continue to thrive during the second year after planting.

As drought conditions intensified over the 2007 winter and growing season, the dynamic nature of both planted and volunteer vegetation became evident at H1. The wetter, western portion of the marsh, including the bulrush and flag marsh planting zones, were not dominated by the shrubbier species found across the rest of the site, although *Polygonum hydropiperoides* was present throughout. *Scirpus californicus* continued to thrive and reproduce both within and outside of its original planting zone. *Eleocharis* *cellulosa* also survived and reproduced well as an understory species to the shrubby canopy of *Ludwigia peruviana* and *Eupatorium serotinum* that established over its entire planting zone. Individuals recruited to the northern edge of the saw-grass planting zone, where wetland hydroperiod was similar, but volunteer vegetation was less dense and was composed of *Polygonum hydropiperoides*, *Pluchea odorata*, and *Eupatorium capillifolium*. It seems *Scirpus californicus* and *Elecoharis cellulosa* were able to survive high water conditions with partial to full submergence, and then reproduce vegetatively and possibly through seeding, when drier conditions occurred over the 2006 and 2007 growing seasons, regardless of volunteer vegetation density, although this may have impeded growth. *Saggitaria lancifolia* and *Pontederia cordata* were the most successful flag marsh species planted, and seemed to grow best at the drier areas of the planting zone. Neither species survived well at FM4, the wettest monitoring plot, located in the western planting zone. However, both species were present further west of FM4 in the drier portion of the western flag marsh planting zone, indicating a specific range of success for both species on CSA wetlands. While the western zone was not occupied by shrubby species, the eastern zone was almost completely overtaken, and yet these two species were able to survive and reproduce along a gradient of pressure by volunteer vegetation. The population of *Cladium jamaicense* survived poorly at the H1 marsh, but was able to maintain its presence between 2006 and 2007 in the drier portions of the planting zone. Where *Cladium jamaicense* was able to establish, individuals grew prodigiously. *Juncus effusus* and *Spartina bakerii*, which either maintained or declined in frequency during the first year after planting, experienced better growth as the site dried down between 2006 and 2007, despite overgrowth by shrubby species. Regrowth by other planted species within the graminoid planting zone did not occur.

The overall decline in seedling survival was greater in the second year after planting, when seedlings experienced drier conditions and pressure from volunteer, invasive shrub species, and all but two individual species experienced greater survival between 2005 and 2006. Still, after two years, all planted species had at least 50% survival, with the exceptions of *Persea palustris* and *Hypericum fasciculatum*, which both had 0% survival in 2007. *Taxodium distichum* and *Cephalanthus occidentalis* by far survived best over the period of record. Most planted species were able to grow, as well as survive, under varying site conditions, in terms of hydroperiod and volunteer vegetation; however, growth was relatively low for *Cephalanthus occidentalis* despite its high survival. Alternately, *Itea virginica*, *Nyssa sylvatica var. biflora*, and *Gleditsia aquatica* grew better in the first year after planting. Dominance by shrub vegetation in terms of height and density was greatest at the tree planting zone, which is the driest portion of the H1 marsh site. *Annona glabra*, planted in 2006, survived and grew well within its planting area at H1. This species is able to quickly create shade and is highly tolerant of flooding, and therefore may be ideal for planting within flooded portions of CSAs. The area of planting occurred on a swale which connected as a surface-water feature with wetlands north of H1 during wet years. Due to wetter conditions, the planting zone was mainly occupied by *Eupatorium serotinum* and *Polygonum hydropiperoides*, as well as recruited *Eleocharis cellulosa.*

Pre-planting management of all types of volunteer vegetation was included in the revegetation scheme to ensure less competition for planted species as they established, and while species did volunteer over the 2006 growing season, water levels were deep and prolonged to such an extent that drier volunteer species such as *Ludwigia peruviana*, *Baccharis halimifolia*, and *Myrica cerifera* did not establish as dominant species within the site. As the H1 marsh continued to dry over the 2007 winter and growing season, smaller individuals of drier species present in 2006 were able to flourish, and the moving front of *Ludwigia peruviana* to the north, south, and west of the planting area encroached heavily. However, it cannot be determined if higher water levels and increased inundation during 2007 would have excluded *Ludwigia peruviana* or the other shrubby species completely or at all. Also unclear is the long-term effect of invasive, volunteer species encroachment on the success of freshwater marsh plantings under current field conditions or in a more hydrologically controlled wetland setting.

While *Baccharis halimifolia*, *Myrica cerifera,* and *Eupatorium serotinum* are all species that normally occur in transitional wetland areas (USDA 2004), *Ludwigia peruviana* is a pioneer species able to form dense, monotypic stands with tremendous seeding potential at moist sites along streams and swamps and also areas with shallow, standing water (Jacobs and others 1994, ISSG 2004). Since this species seems to thrive even when flooded, Richardson and Kluson (2000) experimented with planting wetland tree seedlings within *Ludwigia peruviana* stands, and found *Taxodium distichum, Carya aquatica*, and *Fraxinus caroliniana* could survive and eventually shade out *Ludwigia peruviana* at wetland sites. Carstenn (2002, cited in Brown and others 2002) tested the effect of shade on *Ludwigia peruviana* "robustness" to conclude if a maturing forest canopy could eventually thin, or possibly exclude, this species from an ecosystem. It was found that a reduction in available light (30%) significantly decreased species height, leaf area, percent cover, and productivity; however, the study also found that species such as *Ludwigia peruviana* can significantly contribute to organic matter and nutrient accumulation in newly constructed wetlands, indicating their usefulness in ecosystem development on CSAs.

PPW-3 Marsh

The establishment of planted herbaceous species at PPW-3 was negatively affected by drought conditions and the subsequent lack of available soil moisture in the later part of the 2006 growing season, and during the 2007 growing season. Despite standing water present at the PPW-3 marsh in 2003, 2004, and 2005, hydrologic conditions at the majority of the site during 2006 and 2007 resembled an upland environment, with only fleeting inundation at the wettest portions of the marsh and water levels well below the ground surface for the majority of the year. Even without the shortfall in annual and growing season precipitation, the PPW-3 marsh site is more shallow and ephemeral in terms of water storage when compared with the H1 marsh site. PPW-3 is completely isolated from other existing wetland features on site, even during the most extreme climatic events, unlike the H1 marsh, which has in the past connected with wetland areas to the northwest as a surface-water feature. This fact, in combination with several herbicidal treatments prior to planting to control *Imperata cylindrica* invasion within the wetland, has led to little organic matter accumulation on site, which has been shown to retain soil moisture in wetlands (Stauffer and Brooks 1996).

Dry conditions caused complete mortality of planted *Eleocharis cellulosa* within the first three months of planting, even though the wettest conditions over the period of record occurred approximately one month after planting. Graminoid planting zone species also survived poorly, most likely due to extremely dry conditions. Of the 120 individuals, composed of four species, only one *Muhlenbergia capillaris* was observed in the entire planting zone in 2007. Although declines in frequency occurred within the planting zones, *Cladium jamaicense* and the species within the flag marsh planting zone were able to establish over the period of record, although reproduction only occurred for one individual of both *Thalia geniculata* and *Saggitaria lancifolia*.

The lack of above-ground inundation and saturation of the immediate root zone of herbaceous plantings did not allow for real comparison of species frequency along a hydrologic gradient at PPW-3. While several species were able to survive drought conditions at PPW-3 and H1, the sustained presence of the planted herbaceous species is in question, due to the disconnected nature of the revegetation sites, especially if drought conditions persist. Since herbaceous wetland vegetation is integrally tied to hydrologic conditions and may be more sensitive to decreased water availability than woody species, and thus more susceptible to die off, an understanding of the effects of drought on planted herbaceous species survival and the invasion of revegetation projects by volunteer species, more tolerant to extreme fluctuations in wetland water levels, is crucial.

Dry site conditions negatively affected tree seedlings as well, especially *Annona glabra*, which was unable to survive even at the wettest areas in which it was planted. Survival for *Annona glabra* at H1 was markedly greater where average water levels were much higher. Van der Valk and others (2007) found *Annona glabra* seedlings survived and grew at both “wet” and “dry” locations on constructed tree islands after almost two years, while investigating flooding tolerance of tree species typical of these formations within the Everglades. However, water levels at dry locations at H1 where the species survived well ranged between 18 cm and -35 cm at least 88% of the time, with consistent flooding between August and December. These conditions are still much wetter than those experienced at the wettest planting location for *Annona glabra* at PPW-3. Many *Taxodium distichum* seedlings experienced losses in height, possibly due to pinning and breakage by *Indigofera hirsuta* during the 2006 growing season, since *Taxodium distichum* is known to survive and grow well in well-drained soils. Like the H1 marsh site during 2006, *Eupatorium capillifolium* volunteered to the majority of the site over the 2007 growing season, with the highest density and height at the tree planting zone, shading most tree seedlings and outcompeting *Indigofera hirsuta*. It is unclear at this time whether the establishment of this species with such dominance will hinder seedling survival and growth in the future, as the species seemed to grow with more vigor than the population of individuals at H1 in 2006.

Overall first-year seedling survival at the PPW-3 site was lower than first-year survival at the H1 marsh, although the sample size for H1 was considerably lower. *Cephalanthus occidentalis* and *Gleditsia aquatica* survived well under the hydrologic regime of PPW-3, which was unusually dry, and also at H1 where the species experienced high water conditions. Interestingly, *Persea palustris* survived well (95%) across the PPW-3 planting zone, despite the drought conditions. The species had poorer survival after one year at the H1 marsh, where average water levels ranged between -0.25 m and -1.07 m and transects were inundated for several weeks in the 2005 and 2006 growing seasons. All other planted species had higher survival after one year at the H1 marsh, possibly due to greater available moisture, but even so, species survival was relatively high for the planted species at PPW-3, with the exceptions of *Nyssa sylvatica var. biflora* and *Hypericum fasciculatum*. *Hypericum fasciculatum* performed poorly at both sites during dry years.

Percent change in mean seedling height per species was greater after one year at PPW-3 for seven tree species, although hydrologic conditions differed dramatically between the two planting sites, PPW-3 ('06-'07) and H1 ('05-'06). If this same species attribute is compared between PPW-3 and the second year of growth at H1, when available moisture is more comparable, growth for the same seven species as well as *Annona glabra* is still higher. This compared the sample of *Annona glabra* from the PPW-3 tree planting zone with growth from H1 ('06-'07). *Nyssa sylvatica var. biflora* and *Taxodium distichum* had lower growth at PPW-3 compared with both years at the H1 marsh. This data indicates the variability of wetland tree survival and growth on CSA wetlands that encounter different hydrologic conditions and competitive pressure from volunteer vegetation. No strong correlations or associations were found between seedling survival and growth and average water levels at PPW-3, meaning certain species can successfully establish over a range of hydrologic conditions present at CSA wetlands.

Comparison with Previous Findings

Several tree species planted as part of the marsh revegetations were planted on CSA wetland sites as part of the Rushton 1988 study. As was the case with H1 and PPW-3 in 2006 and 2007, *Fraxinus caroliniana* had high survival and growth along a gradient of hydrologic conditions and overstory and understory vegetation after one growing season. *Taxodium distichum* also had high survival and growth, especially at hydric swamp plots with wetter conditions (Rushton 1988). Since such a large number of *Taxodium distichum* seedlings experienced breakage and dieback at the drier marsh site, PPW-3, Rushton plots where the species experienced negative growth were investigated. Negative change occurred at sites where average water levels were at or below -0.3 m and usually *Salix caroliniana* canopy was present. *Persea palustris* was planted in the wettest plots within Rushton's hydric swamp experimental plots on CSAs as part of her dissertation and the associated FIPR study. The species was planted at ten hydric swamp experimental plots in the wet species grouping. Plots differed in canopy cover and understory species, but *Persea palustris* had consistently higher growth after one year at drier plots, with average water levels below the ground surface or sites only flooded for brief periods of time (Rushton 1988). Similarly, the highest survival for *Persea palustris* occurred at PPW-3, the drier revegetation site. *Nyssa sylvatica var. biflora* survived the worst of the cohort planted at PPW-3, while survival and growth were slightly higher at H1. *Nyssa sylvatica var. biflora* was also planted at Rushton's hydric swamp plots in the wet and transitional groupings and had an average survival of 39% after one year. This is consistent with the low first-year survival at the H1 and PPW-3 sites. The three sites where the species performed well in the Rushton study had average water level ranging between the ground surface and -0.36 m (Rushton 1988). When hydric swamp plots with at least 50% first-year survival were monitored 19 years later, no individuals of *Nyssa sylvatica var. biflora* or *Persea palustris* were monitored (Ingwersen 2006), while surviving *Taxodium distichum* and *Fraxinus caroliniana* trees were still present. If survival within the first few years after planting is a good predictor of long-term survival (Ingwersen 2006), excluding the occurrence of fire or other catastrophic events, seedlings of successful tree species planted at the two revegetation sites will most likely survive over time if current hydrologic conditions persist and sustain the wetland features.

Recommendations for Marsh Revegetation on CSAs

Establishing marsh wetland ecosystems on CSA wetland features presents challenges. Wetland size, topography, wetland:watershed ratio, connectivity to other wetland features, and outfall control are all variable at wetlands occurring on CSAs, as shown in this project's wetland watershed and hydrologic analysis. These features are also subject to inavailability of a desirable freshwater marsh seedbank on the CSAs (Rushton, in Odum and others 1983), the variability in wetland hydroperiod due to clay settling and climatic conditions, and the presence and pressure of both upland and wetland pioneer species, which may be considered non-native or invasive (Odum 1983 and 1991; Cates 1991).

To effectively restore freshwater marsh ecosystems on CSAs, several considerations must be employed. Control over the initial site topography may be crucial to both achieve an appropriate wetland size and shape, as well as ensure more predictable hydrologic conditions. Although not specifically studied, cracks that form in the clay substrate of CSA wetlands, disconnected from surrounding wetland systems and subject to drought conditions, may decrease their long-term viability as water storage and movement internal to the CSA may shift away from that wetland feature. Cracks in the substrate at the H1 marsh site became visible after dry-down in 2007, although there is no way of knowing when these cracks formed. If these formations allow water to drain away from the wetland feature, even in years with average and above-average precipitation, planted vegetation may not persist, and drier, more transitional species may invade and dominate those areas.

While desired marsh species can survive and expand in appropriate conditions, volunteer species will likely always recruit to CSAs both in wetland and upland areas. Controlling water levels at wetland features, and intensive spraying of any undesired species within and at the periphery of sites is possible, but seems inefficient in a long-term sense and does not address the need for whole system restoration including upland, transitional, and wetland areas. Planting the upland and transitional areas of a site with tree species may establish shade, which over time can exclude shade-intolerant pioneer invasives offering an alternative to yearly spraying (Richardson and Kluson 2000). The facultative species *Quercus virginiana* and *Myrica cerifera* seem to survive and grow well on CSAs despite very wet and very dry conditions experienced over the period of record at PPW-1 and PPW-2. Since survival in the first year after planting was high, and both species seemed to create shade quickly, these are obvious choices for planting in transitional and upland areas adjacent to CSA wetlands. Herbiciding, manual clearing, and other methods of species management can certainly aid in wetland establishment success, especially prior to planting of wetland sites. Selective species treatment, if timed correctly with seasonal rainfall and performed repeatedly, can give planted species the opportunity to establish and reproduce without invasive competition after planting occurs. Since species such as *Ludwigia peruviana* can still persist in wet conditions, it may be necessary to eliminate the species manually both in transitional and wetland areas through active species management using herbicide.

Seedling Underplanting

Underplanting wetland features with trees that are able to establish underneath an existing canopy, as well as appropriate placement of species within the wetland feature in terms of water tolerance, has been studied in the past as a way of accelerating natural succession of forested wetlands on CSAs (Harrell 1987; Rushton 1988; Ingwersen 2006). This concept, in terms of planted species assemblage, placement, and success, was further explored on three CSA wetlands differing in canopy composition and structure, substrate, and hydroperiod, although a greater diversity of tree species were planted, compared with this past work. As with the freshwater marsh revegetations, drought conditions greatly affected seedling inundation and moisture availability, which in turn affected the survival and growth of certain planted species. The proximity of planting to dike features and the stability of the underplanting canopy influenced planted seedling success.

Survival and Growth

Twenty-three wetland tree species from differing forested wetland community types were planted across three sites located in northern (SA 10) and central Florida (H1u and TEN-1). Planted species were representative canopy and subcanopy species from forested wetland communities common to southwest Florida, including bay swamps, floodplain forests (river swamps), cypress domes, hydric hammocks, and mixed hardwood swamps (Myers and Ewel 1990). Most planted tree species survived well, although site-specific differences caused first-year survival rates for certain species to differ between underplanting sites. Habitat generalists *Taxodium distichum*, *Ulmus americana*, *Quercus nigra*, and *Fraxinus spp.* survived well at all underplanting sites, which was also the case in Rushton (1988). *Celtis laevigata* and *Cornus foemina* both found in hydric hammocks and floodplain forests also survived well. Both *Sabal* *palmetto* and *Sabal minor* had high first-year survival in the drier planting areas of TEN-1 and H1u, respectively. Results, both from this study and past work, may indicate a variety of forested wetland ecosystems can be represented through species on CSA wetlands if hydrologic, canopy, and understory conditions remain stable, as was the case with SA 10. However, when species encounter more extreme fluctuations in water level, as is the case with hurricane and drought years, as well as pressure from invasive understory species, their range for successful planting on CSA wetlands begins to shrink.

SA 10. Survival after one year was high for all species planted at the SA 10 underplanting site. The high clay content of the substrate and low slope wetland edge, extending south from the site's permanently ponded feature, ensured available soil moisture along the entire gradient that species were planted across and buffered against drydown during the drought conditions experienced over the 2007 growing season. This type of wetland feature, with its closed canopy, was the most successful for wetland tree planting of the three sites studied, with 94% overall survival. *Taxodium distichum*, *Liquidambar styraciflua*, *Cephalanthus occidentalis*, *Nyssa aquatica*, *Ulmus americana*, *Carya aquatica*, *Betula nigra*, *Quercus lyrata*, *Nyssa sylvatica var. biflora*, *Platanus occidentalis*, *Liriodendron tulipifera*, *Celtis laevigata*, *Cornus foemina*, *Ilex cassine*, and *Quercus nigra* all had 100% survival after one year. Survival for the remaining species, *Itea virginica*, *Taxodium ascendens*, *Fraxinus pennsylvanica*, *Magnolia virginiana*, and *Quercus michauxii*, ranged between 88% and 96%, and did not seem to be strongly correlated with wet or drier areas within the species' planting zone. Growth for species was equally good, with positive percent change in height ranging between 4% and 30%. Dieback, or a negative percent change in height, occurred for *Cephelanthus occidentalis* and *Betula nigra*, although the cause of this is unknown. Neither species seemed particularly affected by growth of the vine, *Lygodium japonicum*, which was seen especially covering *Carya aquatica* and *Plantanus occidentis*. Most species had larger variation in seedling height after one year due to seedling dieback and regrowth. Many species had low r and r^2 values, implying there is little association of growth and the hydrologic gradient over which certain species were planted, although the growth of many of the more obligate wetland tree species was negatively correlated with decreasing average water levels, or distance away from the surface-water feature. The buffering effect of the site's low slope, high clay content, and adjacent wet feature is the likely cause of mostly equal growth along the hydrologic gradient.

An active outfall weir, present in the northeast corner of the site, ensures that water levels will not spike and submerge tree seedlings in wetter years, if maintained and adjusted for settling at the CSA. However, years which receive average or above-average precipitation will certainly inundate the wettest portions of all three planting zones, possibly affecting growth and survival. The outfall also functions to retain the permanently ponded feature, which has provided the planting site with a stable source of moisture and water table despite drought conditions, leading to less drastic drydown conditions in the early part of the growing season, as seen at the Tenoroc-1 site.

Tenoroc-1. Overall survival at TEN-1 was lower than that at SA 10, and 100% survival after one growing season only occurred for *Taxodium distichum* and *Sabal palmetto*, a palm species not included at the SA 10 planting. *Fraxinus caroliniana, Ulmus americana*, *Quercus nigra*, *Cornus foemina*, and *Magnolia virginiana* also survived well along the entire gradient over which they were planted, indicating usefulness for planting during drier years and along dike or overburden areas present near CSA wetlands. Other species were negatively affected by drought conditions that were exacerbated by the planting location. The site, located at the edge of the CSA dike, had substrate with higher sand content leading to faster drainage within planting areas after precipitation occurred, and was planted along a much steeper slope than SA 10 or H1. The deeper wetland feature adjacent to the planting is not regulated by outfall structures and experienced dry conditions for the majority of the 2006 and 2007 growing seasons, providing little buffer against dry-down at the planting site. Several species were able to survive at the most downslope areas of the planting, but were unable to do so at the drier areas along the dike's slope and planting of those species in these areas should be avoided. Likewise, many species' growth was negatively correlated with decreasing average water levels, or distance along the gradient from wet to dry, although associations were weak, most likely due to small sample size. The range in percent change in mean seedling height at TEN-1 was broader than SA 10, with the populations of *Ulmus americana* and *Celtis laevigata* experiencing greater increases at TEN-1, although most of this growth was concentrated in the wetter areas at the species' planting zone.

Unlike the canopy at SA 10, which was stable over the period of record and composed of native wetland tree species, the majority of the canopy in the drier areas of TEN-1 was occupied by *Sapium seriferum*, and the canopy of *Salix caroliniana* in the wetter areas experienced some tree fall. This allowed several species of shade-intolerant pioneers to volunteer where the canopy opened. Tree fall is a likely occurrence associated with planting under *Salix caroliniana* as the canopy ages and dies back or is negatively affected by storm events, and although invasive herbaceous species may volunteer to areas where the canopy has opened, this also offers the opportunity for growth over the existing canopy by planted species. The tree species *Sapium sebiferum* is non-native and considered by some to be detrimental to Florida's ecosystems (Burks 1996; Bruce and others 1997). Removal of this species from understory planting sites, however, may negatively affect underplanted tree species. Manual removal of the canopy, or burning and herbicidal removal which may cause extensive tree fall that may damage seedlings as seen at H1u, will encourage shade-intolerant herbaceous species to copiously volunteer, thus eliminating the advantage of planting under an existing canopy. A more effective approach may be to leave the existing canopy and underplant with species that will eventually shade out *Sabium sebiferum*, with removal of recruited seedlings at regular intervals either manually or with herbicide.

H1u. Overall survival was lowest at H1u of all the three sites, and no species was able to maintain 100% survival over the period of record. However, first-year survival was still above 50% for all planted species except *Nyssa sylvatica var. biflora* and *Ilex cassine.* Only three species, *Celtis laevigata*, *Quercus nigra*, and *Liquidambar* *styraciflua*, had higher survival than populations of the same species planted at TEN-1, and two of those were present in an area of the underplanting that was least affected by tree fall. While hydrologic conditions at H1u were similar to those at SA 10, the majority of the canopy present in 2006 had fallen by 2007, allowing an increased presence by volunteer understory and vine species. Tree fall was extremely damaging to seedlings, causing breakage of many seedlings below their initial planting height. Animal herbivory by rabbits may have also affected seedling survival and growth. Six of the ten planted species, monitored for height, experienced a negative percent change in mean seedling height, and also significantly lower mean growth for several species at H1u compared with SA 10 and TEN-1. If dead limbs had not fallen, water levels were such that higher species survival and growth would most likely have occurred.

Underplanting versus Planting in Full Sun

Although hydrology most closely controls where seedlings will establish and persist within restored and created wetland environments, light availability and the control of competitive weedy species are also major determinants of planted seedling success (Clewell and Lea 1990). Survival and growth were compared between underplanting and marsh revegetation sites to explore any advantages or disadvantages to planting wetland tree seedlings under an existing canopy. As previously discussed, the instability of the canopy at H1u caused unusually high mortality and seedling breakage, and thus provides little useful data for this comparison with either type of site. Five species were planted at both marsh and underplanting sites. Results were mixed when survival and growth of those species was compared between sites with a stable canopy and those without. Growth was highest for three of the five species where light was most available and site conditions were the driest, at PPW-3. *Itea virginica* survived well at sites with full sun and underplanting sites with adequate moisture (SA 10, H1 '05-'06), but grew best at the driest planting location, PPW-3. *Taxodium distichum* was able to survive well at all four sites, and after two years at the H1 marsh, but experienced its lowest growth at the marsh sites, PPW-3 and H1 '06-'07), possibly due to breakage by *Indigofera hirsuta* and *Ludwigia peruviana*, respectively. *Cephalanthus occidentalis* survived and grew worst underneath canopy despite adequate moisture (SA 10). Survival was high after one year for *Fraxinus spp.* seedlings at all four planting locations, while *Nyssa sylvatica var. biflora* survived worst sites at the driest planting sites (TEN-1, PPW-3) in the first year after planting, regardless of canopy cover. Although results were inconclusive on a species basis and hydrologic conditions differed, overall first-year survival was higher at the two sites with canopy versus the two without, and the aggressive overgrowth of shrubby invasive species and subsequent declines in overall survival at the H1 marsh in the second year after planting, indicate the lack of control over planting success in full sun environments without volunteer invasive species management. Underplanting with an existing canopy composed of early successional species such as *Salix caroliniana* and *Acer rubrum* can likely provide an advantage to planting tree species in full sun, where long-term management of volunteer species through herbicidal and manual clearing are likely necessary components to any revegetation scheme in order to ensure the success of planted seedlings. Wetland sites without an existing canopy can also be invaded by *Imperata cylindrica,* which may increase the planting site's susceptibility to fire and limit the success of planted seedlings (Ingwersen 2006; Lippencott 2000). This species was observed recruiting to the drier areas of H1u in 2007, after canopy fall occurred in 2007.

Nine species of wetland and transitional tree species were planted in full sun at PPW-1 and PPW-2. Declines in survival were lower during the first two years after planting as seedlings established and were also encountered with a disproportionate amount of hurricane activity. In general, tree species survived well in the third and fourth years after planting despite drought conditions, although all volunteer species were routinely eliminated around the periphery of each wetland. *Quercus laurifolia* had poor survival in the full sun environment, although this may be mainly due to hurricane damage and high water conditions, as the remaining populations maintained survival rates in the third and fourth years after planting. *Fraxinus caroliniana*, *Carya aquatica*, and *Taxodium distichum* were planted in the wetter areas of the wetland and all survived well after four years. Survival for these three species was also high at the SA 10 and TEN-1 underplanting sites after only one year. Likewise, *Quercus virginiana* was located at the drier planting periphery and also survived well. Although not quantitatively measured, shade was greater and groundcover beneath these individuals was less than in other areas of the wetland, thus this species may serve to establish structure and shade in transitional wetland environments over time. Results from these two wetlands prove a variety of wetland tree species can survive in a full-sun environment while experiencing hydrologic extremes with accompanying management of upland vegetation. Since no effort was made to plant the upland areas surrounding PPW-1 and PPW-2, these areas will have to be managed well into the future to prevent the spread of *Imperata cylindrica* into the wetland, especially in drier years.

Appropriate Wetland Community Types for CSAs

CSA wetland features can be revegetated with species occurring in both forested wetland and freshwater marsh communities; however, the long-term viability of these species additions remains in question. This is especially true for the restoration of herbaceous marsh ecosystems through revegetation. While establishment may be possible, the initial site design and invasive species management are both energy-intensive aspects of restoration, and do not ensure the long-term success of planted species to the exclusion of more invasive, "undesirable" species, and so the unmanaged restoration of a freshwater marsh ecosystem as it occurs naturally may never be achieved. However, further research and more extensive marsh plantings must take place during years without extreme drought to conclude the success of revegetating freshwater marsh ecosystems on CSAs. Since many wetland features on CSAs self-organize and succeed to a forested wetland dominated by *Salix caroliniana* (Odum and others 1983), typical of primary succession in some floodplain and bottomland hardwood wetlands, restoration through proper placement of forested wetland species beneath this canopy may be a natural fit for CSA wetlands.

CSAs and the wetland features they support may be thought of as "emergent" or novel ecosystems resulting from a human-induced change of the abiotic environment and the limited seed dispersal from native ecosystems in the post-mining environment (Hobbs and others 2006). As a result, restoration goals for herbaceous marsh and forested wetlands on CSAs, in terms of community composition and structure, may need to be adapted to align with the specific biotic and abiotic characteristics of each wetland feature. Although possessing physical characteristics distinct from natural wetlands, these ecosystems may still provide valuable wetland functions and mitigate for regional wetland loss in the post-mining landscape.

CONCLUSIONS AND RECOMMENDATIONS

Over the course of this project, a thorough investigation of naturally established wetlands and their hydrology led to a much improved understanding of the CSA wetland environment. Evaluations of previous plantings of wetland species on CSAs and recent field trials with a greater number of species in more diverse and representative CSA settings increased our competency in selecting species appropriate for introduction to CSA wetland areas. These components forged together were the basis for guidelines for wetland restoration in which site-specific abiotic and biotic characteristics are taken into account. Numerous models to aid in understanding CSA dynamics were developed. Specific recommendations for future research are made regarding CSA wetlands that would build further upon the knowledge generated in this research and aid in future efforts to understand and create wetlands on CSAs.

CONCLUSIONS

Findings regarding naturally established CSA wetlands and their hydrology include the following:

- Areas that meet hydrologic, soil, and plant community definitions for wetlands occupy substantial but variable portions of abandoned CSAs;
- Naturally established wetlands on CSAs are isolated depressional, eutrophic systems which are often dominated by woody pioneer species in the shrub and canopy layers and a more diverse mix of obligate and facultative species in the ground layer. Marshes exist but in lesser abundance than forested wetlands;
- The landscape positions of wetland communities on CSAs are defined more by physical gradients, in particular the hydrologic regime, than by chemical gradients;
- While communities are organized along ecotone-wetland gradients, there are large spatial ranges and considerable overlap exists among communities, which may reflect tolerance of colonizing species and changing environmental conditions;
- Naturally establishing communities on CSAs are rapidly increasing soil organic matter, becoming less alkaline, and increasing the soil N:P ratios to ones similar to those of natural and climax systems;
- CSA wetlands are highly productive systems, as evidenced by the high transpiration of one of the dominant species (*Salix caroliniana*), the fast accretion of organic matter, and the high density and spread of volunteer vegetation;
- CSA wetlands are unlike reference wetlands in soil chemistry with less total nitrogen and organic matter and higher total phosphorus concentrations, and those not planted have less standing biomass and woody species diversity than reference wetlands;

- Hydroperiods of surface water features on 8 CSAs demonstrated substantial differences in hydrologic regime among and within CSAs;
- Wetland features on CSAs are typically characterized by one of two hydrologic regimes: (1) flashy; those with a quick response to precipitation causing immediate flooding and subsequent quick drawdowns or (2) ponded; those with areas with permanent or near-permanent flooding and high-water depths with sharp transitions into areas that rarely experience flooding;
- Precipitation is the primary source of water for CSAs due to their isolated nature. The response of wetland features to rain events varies among and within CSAs as a function of hydraulic conductivity of surrounding uplands and can be accurately predicted solely as a function of rain amount with little effect from antecedent conditions;
- CSAs often contain multiple isolated watersheds. A range of 2-200 watersheds was delineated on six CSAs studied;
- Three dominant watershed types were identified on CSAs: (1) small, shallow watersheds that are least frequently flooded, (2) large watersheds with broad shallow areas that intermittently flood and dry out, and (3) large watersheds with concentrated deep areas that rarely dry out;
- ET represented the dominant pathway of water loss from CSA wetlands and was high relative to typical rates or climatic-based estimations as demonstrated by the temporal model, diurnal curve analysis, and the transpiration study;
- Loss of water out of CSAs through dikes was negligible at the three sites studied;
- Though outfalls existed at most sites where surface water could exit CSAs, only two were frequently active, likely due to consolidation;
- Local groundwater exchange between surrounding uplands and water features varied among and within CSAs. Often groundwater exchange was negligible, but both groundwater flow into and out of water features was observed. The magnitude of flow was dependent upon upland soil type;
- Uplands that had more conductive soils due to sand and/or overburden additions resulted in decreased runoff amounts and potentially facilitated groundwater flow to wetland features, while the lower capillary forces of such soils limited the extent of saturated zones beyond inundated features;
- The strong capillary forces of clay-dominated substrates extended the transitional zone from the wetland while the lower hydraulic conductivities of such soils increased the runoff volume and reduced surface-to-groundwater interactions;
- Capillary forces in the clays maintained saturation levels near the surface even at considerable depth to water table. As a result, water features are often surrounded by broad clay-dominated reaches with adequate available soil water for planting wetland tree species, particularly those whose roots can penetrate 50 cm below ground into areas of nearly permanent saturation. In the case of *Salix caroliniana*, rooting occurred throughout saturation depths and extended to the water table, and transpiration from *Salix caroliniana* is high, even in the wettest and driest regimes studied.

The evaluation of previous and new field trials led to the following findings:

- Five planted tree species, typical of various forested wetland types, have survived for over 20 years on multiple CSAs. Based on results from recent plantings, a larger number of species that can persist in periodically flooded to transitional areas have been demonstrated to survive in the short term;
- No distinguishable differences in tree growth in the long term have been associated with differences in CSA fill-type, namely sand-clay mix versus clay fill;
- Long-term tree survival was generally higher in wetland areas than ecotone areas, likely due to fire and drought;
- An overstory of *Salix caroliniana* or other pioneer species may have beneficial microclimatic effects that aid in tree survival in the early years;
- Trees planted in full-sun environments are often subject to overgrowth by volunteer vegetation and greater drought stress than those planted under an existing canopy, likely affecting survival;
- Seedlings of mature planted species were not present in sufficient numbers to expand populations, likely due to the lack of areas buffered from quick rises in water levels or drought;
- The planted trees surviving greater than 20 years added significantly more structure in terms of biomass and stem density to CSAs than what was provided by pioneer species;
- The amount of water available to planted trees goes beyond proximity to water features, and includes factors related to soil moisture such as soil conductivity, watershed configuration, and feedbacks from existing vegetation;
- Woody and herbaceous species for CSAs may be more appropriately identified using their tolerance of a given hydrologic regime and general capacity for resistance to hydrologic swings rather than by their association with a given state or geographic region, soil type, nutrient profile or association with other species;
- The best survival of tree seedlings planted during the recent field trials occurred under a healthy canopy along a gentle, saturated slope buffered against extreme depths and drawdowns;
- Several herbaceous species from freshwater marsh ecosystems were able to establish and reproduce in deepwater, moderately flooded, shallow, and saturated wetland environments on CSAs;
- Freshwater marsh wetlands on CSAs are vulnerable to invasion by herbaceous and woody volunteer vegetation from surrounding transitional and upland areas during prolonged periods of drought;
- Composition and structure of volunteer and recruited vegetation in planting areas were strongly affected by hydrologic conditions, shade, and management treatments.

PRODUCTS OF THIS RESEARCH

In addition to the knowledge of CSA wetlands gained, products of this research include:

- Highly accurate temporal hydrologic models developed for 7 water features on 6 CSAs that predict daily water levels based on the results of empirical studies in this project on CSA hydrology;
- A spatial model that can divide CSAs into a realistic array of watersheds that accurately distribute water across a CSA. The model produces maps that spatially illustrate water depths and frequency and duration of flooding. With sufficient datasets provided for surface water hydrology and topography, the spatial model can be adapted and applied to large- or small-scale restoration projects;
- A tree species population model that can predict long-term population change of a given tree species based on current population size structure;
- A large list of both wetland and facultative trees and wetland herbaceous plants with demonstrated survival on CSAs; optimal landscape species positioning in relation to hydrologic and ecological factors; expectations for survival and growth of planted species and for the persistence or appearance of volunteer vegetation in planted areas;
- General guidelines for future wetland creation or enhancement on CSAs.

GUIDELINES FOR DESIGNING WETLANDS ON CSAs

- Use site-specific analysis of watersheds (configuration and characteristics) and hydrologic models to identify appropriate areas or designs for wetlands using the following insights:
- Steeper gradients will create ponded systems less conducive to wetland development. Including a flexible outfall invert design to handle continued consolidation can be used to indefinitely control surface water storage;
- Flatter CSAs will have a high number of flashy wetland features. Connecting and grading features will create more buffered hydrologic regimes, which are more conducive to the survival and recruitment of desirable species. Creating islands, increasing shoreline length or other means of adding more micro-relief to lower-lying features will have similar positive effects on establishment and survival;
- Acknowledge and utilize the differences of clay-dominated versus mixed-soil uplands:
    - Mixed-soil uplands will buffer hydroperiods by decreasing immediate surface runoff and potentially increasing groundwater inflow but will dramatically reduce transitional zone areas;
    - Clay-dominated uplands will increase flashiness of wetland hydroperiods, but will also increase transitional areas and support wetland tree species well beyond periodically flooded zones;

- Adopt a holistic ecosystem approach – plant upland, transitional, and wetland areas with the following goals:
    - Shade out competitors that will otherwise encroach on wetlands during periods of drought;
    - Introduce plants to occupy all vertical strata to increase structural complexity and diversity;
    - Create connectivity between ecosystems to promote viable faunal habitat and provide seed dispersal via animal and hydrologic vectors;
- Carefully weigh the potential benefits versus the inconveniences of planting among existing vegetation. Canopy species may aid in ecosystems development and planting success through:
    - Increasing organic matter with positive effects on water storage and nutrient availability to planted species;
    - Creating microclimates through shading that can reduce water loss through soil evaporation and shade out competitors;
- Plant larger seedlings since they compete better with pioneer species, tolerate deeper flooding, and have a more developed root structure, enabling access to available water below the ground surface;
- Employ species that proved successful in previous field trials; and select new species based predominantly on existing community structure, species hardiness, and hydrologic characteristics of species and less upon local geography, soil type, and natural community type associations.

RECOMMENDATIONS FOR FUTURE STUDY

- Document rooting depths and transpiration rates of desirable tree species on CSAs to understand their role in ecohydrology by employing a similar approach to that taken for *Salix caroliniana*. This will help to determine appropriate soil moisture regimes on a species-specific basis and allow for observation and prediction of any changes in CSA hydrology caused by feedbacks from more desirable species;
- Build upon and draw directly from existing data through continued monitoring of hydrology and vegetation on CSAs studied in this project, particularly at the field trial sites;
- Apply design guidelines from this study for restoration at the site-scale and on relatively new CSAs;
- Continue to investigate CSA hydrology with particular interest in understanding long-term feedbacks of vegetation and hydrology and predicting long-term changes in water budgets.

REFERENCES

Akilan K, Considine J, Marshell J. 1994. Water use by Geraldton wax (*Chamelaucium uncinatum* Schauer) as measured by heat balance stem flow gauges. New Zealand Journal of Crop and Horticultural Science 22: 285-94.

Alexander TR. 1955. Observations on the ecology of the low hammocks of southern Florida. Journal of the Florida Academy of Sciences 18: 21-7.

Anand M, Desrochers R. 2004. Quantification of restoration success using complex systems concepts and models. Restoration Ecology 12: 117-23.

Bardi E, Brown MT, Reiss KC, Cohen MJ. 2005. UMAM training manual: web based training manual for Chapter 62-345, FAC for wetlands permitting. Gainesville (FL): University of Florida, Center for Wetlands. Available online at: http://publicfiles.dep.state.fl.us/dear/labs/sas/library/docs/umam/UMAM%20Training%20Manual.pdf

Beatty SW. 1984. Influence of microtopography and canopy species on spatial patterns of forest understory plants. Ecology 65: 1406-19.

Bell FG. 2004. Engineering geology and construction. London: Taylor & Francis. p 797.

Brooks J, Schulte P, Bond B, Coulombe JD, Hinckley T, McDowell N, Phillips N. 2003. Does foliage on the same branch compete for the same water? Trees 17: 101-8.

Brown MT, Tighe RE. 1991. Techniques and guidelines for reclamation of phosphate mined lands. Bartow (FL): Florida Institute of Phosphate Research. FIPR publication nr 03-044-095.

Brown MT, Best GR, Clayton D, Bower T, Kendall A, Prado A, Wiggington J. 1997. Vegetation. Chapter 6. In: Erwin KL, Doherty SJ, Brown MT, Best GR. Evaluation of constructed wetlands on phosphate mined lands in Florida. Volume III. Bartow (FL): Florida Institute of Phosphate Research. FIPR publication nr 03-103-139.

Brown MT, Carstenn S, Baker J, Bukata BJ, Gysan T, Jackson K, Chinners-Reiss K, Sloan M. 2002. Successional development of forested wetlands on reclaimed phosphate mined lands in Florida. Volume 2. Bartow (FL): Florida Institute for Phosphate Research. FIPR publication nr 03-131-193.

Bruce KA, Cameron GN, Harcombe PA, Jubinsky G. 1997. Introduction, impact on native habitats, and management of a woody invader, the Chinese tallow tree, *Sapium sebiferum* (L.) Roxb. Natural Areas Journal 17: 255-60.

Burks KC. 1996. Adverse effects of invasive exotic plants on Florida's rare native flora. Resource Manage. Notes 8: 15-16.

Butner JR, Best GR. 1981. Plant community study in the clay settling areas of Central Florida. In: Odum HT (ed.). Studies on phosphate mining, reclamation and energy. Gainesville (FL): University of Florida, Center for Wetlands.

Callahan J, Rivera O, Cates B. 1991. Status assessment of reclaimed clay settling areas with forested and herbaceous wetlands. Tallahassee (FL): Florida Department of Environmental Protection, Bureau of Mine Reclamation.

Carrier WD. 1982. Waste clay disposal in mine cuts. Bartow (FL): Florida Institute of Phosphate Research. FIPR publication nr 02-006-009.

Carstenn S. 2000. Self-organization and successional trajectories of constructed forested wetlands [DPhil dissertation]. Gainesville: University of Florida. 224 leaves.

Caswell H. 1989. Matrix population models: construction, analysis, and interpretation. Sunderland (MA): Sinauer Associates, Inc. 328 p.

Cates B. 2001. Guidance in the reclamation of forested and herbaceous wetlands on phosphatic clay settling areas. Tallahassee (FL): Florida Department of Environmental Protection, Bureau of Mine Reclamation.

Chambliss CG, Ezenwa IV. 2002. Minor use summer annual forage legumes. Florida Cooperative Extension Service SS-AGR-79. Gainesville (FL): University of Florida, Institute of Food and Agricultural Sciences. p 1.

Clewell A. 1999. Restoration of riverine forest at Halls Branch on phosphate-mined land, Florida. Restoration Ecology 7: 1-14.

Clewell AF, Lea R. 1990. Creation and restoration of forested wetland vegetation in the southeastern United States. In: Kuslar JA, Kentula ME (eds.). Wetland creation and restoration: the status of the science. Washington (DC): Island Press. p 195-231.

Connell JH, Slayter RO. 1977. Mechanisms of succession in natural communities and their role in community stability and organization. American Naturalist 111: 1119-44.

Cowardin LM, Carter V, Golet FC, LaRoe ET. 1979. Classification of wetlands and deepwater habitats of the United States. Washington (DC): U.S. Department of the Interior, Fish and Wildlife Service. p 131.

DeLotelle RS, Fletcher SW, Arcuri, AN. 1981. Patterns of wading bird utilization of natural and altered freshwater marshes: causes and management implications. In: McCaffrey PM, Beemer T, Gatewood SE, editors. Proceedings of a Symposium on Progress in Wetlands Utilization and Management; 1981 Jun 9-12; Orlando, FL.

Tallahassee (FL): Coordinating Council on the Restoration of the Kissimmee River Valley and Taylor Creek-Nubbin Slough Basin. p 335-44.

Dingman L. 2002. Physical hydrology. 2nd edition. Upper Saddle River (NJ): Prentice-Hall, Inc. 646 p.

Drexler JZ, Snyder RL, Spano D, Tha Paw K. 2004. A review of models and micrometeorological methods used to estimate wetland evapotranspiration. Hydrological Processes 18: 2071-2101.

Dulohery CJ, Kolka RK, McKevlin MR. 2000. Effects of a willow overstory on planted seedlings in a bottomland restoration. Ecological Engineering 15(S-1): 57-66.

Egler FE. 1954. Vegetation concepts. I. Initial floristic competition. A factor in old field vegetation development. Vegetatio 4: 412-17.

ESRI. 2008. ArcGIS® Spatial Analyst, version 9.2. Redlands (CA): Environmental System Research Institute.

Everett S. 1991. Growth of bald cypress and pond cypress seedlings and the effect of nutrient tables. In: Odum HT, ed. Evaluation of alternatives for restoration of soil and vegetation on phosphate clay settling ponds. Bartow (FL): Florida Institute of Phosphate Research. p 25-42.

Faith DP, Minchin PR, Belbin L. 1987. Compositional dissimilarity as a robust measure of ecological distance. Vegetatio 69: 57-68.

Farmer EE, Blue WG. 1978. Reclamation of lands mined for phosphate. In: Schaller FW, Sutton P, eds. Reclamation of drastically disturbed lands. Madison (WI): ASA, CSSA, SSSA.

Florida Department of Environmental Protection. 1994. Wetland status: delineation of extent of wetlands and surface waters. Florida Administrative Code Ch. 62-340. Tallahassee (FL): Florida Department of State.

Fowells HA. 1965. Silvics of forest trees of the United States. Washington (DC): USDA Forest Service.

Gansart D. 2003. Xylem sap flow as a major pathway for oxygen supply to the sapwood of Birch (*Betula pubescens* Ehr.). Plant, Cell and Environment 26: 1803-14.

Garbrecht J, Martz LW. 2000. Digital elevation model issues in water resources modeling. In: Maidment D, Djokic D, eds. Hydrologic and hydraulic modeling support with Geographic Information Systems. Redlands (CA): ESRI Press.

Garlanger JE, Babcock JW. 1987. Field evaluation of sand-clay mix reclamation – Phase II. Bartow (FL): Florida Institute of Phosphate Research. FIPR publication nr 03-006-007.

Gilbert T, King T, Barnett B. 1981. An assessment of wetland habitat establishment at a Central Florida phosphate mine site. Washington: U.S. Dept. of the Interior, Fish and Wildlife Service. Report nr FWS/OBS-81/38.

Gilman EF. 1999. Sabal minor. Fact Sheet FPS 518. Gainesville (FL): University of Florida Cooperative Extension Institute of Food and Agricultural Sciences. p 1-3.

Godfrey RK, Wooten JW. 1979. Aquatic and wetland plants of Southeastern United States. Athens (GA): University of Georgia Press. p 933.

Greco S. 2004. A biogeochemical survey of wetlands in the United States [MS thesis]. Gainesville: University of Florida. 122 p.

Hill AJ, Neary VS. 2007. Estimating evapotranspiration and seepage for a sinkhole wetland from diurnal surface-water cycles. Journal of the American Water Resources Association 43(6): 1373-82.

Hobbs RJ, Arico S, Aronson J, Baron JS, Bridgewater P, Cramer VA, Epstein PR, Ewel JJ, Klink CA, Lugo AE, Norton D, Ojima D, Richardson DM, Sanderson EW, Valladares F, Vila M, Zamora R, Zobel M. 2006. Novel ecosystems: theoretical and management aspects of the new ecological world order. Global Ecology and Biogeography 15: 1-7.

Hook DD. 1984. Waterlogging tolerance of lowland tree species of the South. Southern Journal of Applied Forestry 8: 136-49.

Hvorslev MJ. 1951. Time lag and soil permeability in ground water observations. Bulletin nr 36. Vicksburg (MS): Waterways Experiment Station, U.S. Army Corps of Engineers.

Hupp CR, Bazemore DE. 1993. Temporal and spatial patterns of wetland sedimentation, West Tennessee. Journal of Hydrology 141: 179-96.

Ingwersen W. 2006. Viability of wetland trees after twenty years on phosphatic clay settling areas and their role in ecosystem development [MS thesis]. Gainesville (FL): University of Florida. 127 p.

Invasive Species Specialist Group. 2004. Global Invasive Species Database. Nairobi (Kenya): Global Invasive Species Programme. Available online at: http://www.issg.org/database/.

Jacobs JM, Mergelsberg SL, Lopera AF, Myers DA. 2002. Evapotranspiration from a wet prairie wetland under drought conditions: Paynes Prairie, Florida, USA. Wetlands 22(2): 374-85.

Kangas PC. 1981. Succession as an alternative for reclaiming phosphate spoil mounds. In: Odum HT, ed. Studies on phosphate mining, reclamation and energy. Gainesville (FL): University of Florida, Center for Wetlands.

King TL, Hord T, Gilbert F, Montablo F, Allen JN. 1980. An evaluation of wetland habitat establishment and wildlife utilization in phosphate clay settling areas. In: Lewis RR, Cole DP, eds. Seventh Annual Conference: The Restoration and Creation of Wetlands. Tampa (FL): Hillsborough Community College.

Kirk JA, Wise WR, Delfino JJ. 2004. Water budget and cost-effectiveness analysis of wetland restoration alternatives: a case study of Levy Prairie, Alachua County, Florida. Ecological Engineering 22: 43-60.

Koerselman W, Meuleman AFM. 1996. The vegetation N:P ratio: a new tool to detect the nature of nutrient limitation. Journal of Applied Ecology 33: 1441-50.

Kuzovkina YA, Quigley MF. 2005. Willows beyond wetlands: uses of *Salix* L. species for environmental projects. Water, Air, & Soil Pollution 162(1-4): 183-204.

Lamont WE, McLendon JT, Clements LW, Field IL. 1975. Characterization studies of Florida phosphate slimes. Washington (DC): USDI, Bureau of Mines. Report nr RI-8089.

Levine JB, Salvucci GD. 1999. Equilibrium analysis of groundwater-vadose zone interactions and the resulting spatial distribution of hydrologic fluxes across a Canadian prairie. Water Resources Research 35(5): 1369-83.

Lewelling BR, Wylie RW. 1993. Hydrology and water quality of unmined and reclaimed basins in phosphate-mining areas, West-Central Florida. Tallahassee (FL): U.S. Geological Survey. Water Resources Investigation Report nr 93-4002.

Lippencott CL. 2000. Effects of *Imperata Cylindrica* (L.) Beauv. (cogongrass) invasion on fire regime in Florida sandhill (USA). Natural Areas Journal 20: 140-9.

Lorenzen B, Brix H, Mendelssohn IA, McKee KL, Miao SL. 2001. Growth, biomass allocation and nutrient use efficiency in *Cladium jamaicense* and *Typha domingensis* as affected by phosphorus and oxygen availability. Aquatic Botany 70(2): 117-33.

Lu J, Sun G, McNulty SG, Amatya DM. 2005. A comparison of six potential evapotranspiration methods for regional use in the Southeastern United States. Journal of AWRA 39(4): 621-33.

Lugo A, Brown S, Brinson MM. 1988. Forested wetlands in freshwater and salt-water environments. Limnology and Oceanography 33(4): 894-909.

Lytle DA, Merritt DM. 2004. Hydrologic regime and riparian forests: a structured population model for cottonwood. Ecology 85: 2493-2503.

Manabe T, Nishimura N, Miura M, Yamamoto S. 2000. Population structure and spatial patterns for trees in a temperate old-growth evergreen broad-leaved forest in Japan. Plant Ecology 151: 181-97.

Mao LM, Bergman MJ, Tai CC. 2002. Evapotranspiration measurement and estimation of three wetland environments in the Upper St. Johns River Basin, Florida. Journal of AWRA 38(5): 1271-85.

Matthews JD. 1989. Silvicultural systems. New York: Oxford University Press.

McCune B, Grace JB. 2002. Analysis of ecological communities. Gleneden Beach (OR): MjM Software Design.

McKevlin MR. 1992. Guide to regeneration of bottomland hardwoods. Asheville (NC): U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station. General Technical Report nr SE-76.

McLanahan TR. 1986. The effect of seed source on primary succession in a forest ecosystem. Vegetatio 65: 175-78.

McLeod KW, Reed MR, Nelson EA. 2001. Influence of a willow canopy on tree seedling establishment for wetland restoration. Wetlands 21(3): 395-402.

Meeks DW, Hatfield, JC, Howell TA, Idso SB, Reginato RJ. 1984. A generalized relationship between photosynthetically active radiation and solar radiation. Agronomy 76: 939-45.

Megongial JP, Day FP. 1988. Organic matter dynamics in four seasonally flooded forest communities of the dismal swamp. American Journal of Botany 75(9): 1334-43.

Mitsch WJ, Gosselink JG. 1993. Wetlands. New York (NY): Van Nostrand Reinhold.

Murphy KE, Rains MC, Kittridge MG, Stewart MT, Ross MA. 2008. Hydrology of clay settling areas and surrounding landscapes in the phosphate mining district, peninsular Florida. JAWRA 44: 980-95.

Myers RL, Ewel JJ. 1990. Ecosystems of Florida. Orlando (FL): University of Central Florida Press.

Odum EP. 1969. The strategy of ecosystem development. Science 164(3877): 262-70.

Odum HT. 1994. Ecological and general systems. Niwot (CO): University Press of Colorado.

Odum HT, Best GR, Miller MA, Rushton BT, Wolfe R, Bersok C, Feiertag J. 1990. Accelerating natural processes for wetland restoration after phosphate mining. Bartow (FL): Florida Institute of Phosphate Research. FIPR Publication nr 03-041-086.

Odum HT, Miller M, Rushton B, McClanahan T, Best TR. 1983. Interaction of wetlands with the phosphate industry. Bartow (FL): Florida Institute of Phosphate Research. FIPR Publication nr 03-041-086.

Odum HT, Rushton BT, Paulic M, Everett S, McClanahan T, Munroe M, Wolfe RW. 1991. Evaluation of alternatives for restoration of soil and vegetation on phosphate clay settling ponds. Bartow (FL): Florida Institute of Phosphate Research. FIPR Publication nr 03-076-094.

Paulic M. 1991. Factors influencing the establishment of hardwood swamps on clay settling ponds. In: Odum HT, ed. Evaluation of alternatives for restoration of soil and vegetation on phosphate clay settling ponds. Bartow (FL): Florida Institute of Phosphate Research. p 3-24.

Paulic M, Rushton B. 1991a. Success of seedlings planted under a mature canopy. In: Odum HT, editor. Evaluation of alternatives for restoration of soil and vegetation on phosphate clay settling ponds. Bartow (FL): Florida Institute of Phosphate Research. FIPR Publication nr 03-076-094.

Paulic M, Rushton B. 1991b. Success of seedlings planted under a mature canopy. In: Odum HT, ed. Evaluation of alternatives for restoration of soil and vegetation on phosphate clay settling ponds. Bartow (FL): Florida Institute of Phosphate Research. p 43-74.

Pausas JG, Bradstock RA, Keith DA, Keeley JE. 2004. Plant functional traits in relation to fire in crown-fire ecosystems. Ecology 85: 1085-1110.

Pittman WE, Sweeney JW. 1983. State-of-the-art of phosphatic clay dewatering and disposal techniques. Vol. 1, A review of phosphatic clay dewatering research. Bartow (FL): Florida Institute of Phosphate Research. FIPR Publication nr 02-017-021.

Poiani KA, Johnson WC. 1993. A spatial simulation model of hydrology and vegetation dynamics in semi-permanent prairie wetlands. Ecological Applications 3: 279-93.

Porporato A, Laio F, Ridolfi L, Rodriquez-Iturbe I. 2000. Plants in water-controlled ecosystems: active role in hydrologic processes and response to water stress. III: Vegetation water stress. Advances in Water Resources 24: 725-44.

Reigner WR, Winkler W. 2001. Reclaimed phosphate clay settling area investigation: hydrologic model calibration and ultimate clay elevation prediction. Bartow (FL): Florida Institute of Phosphate Research. FIPR Publication nr 03-109-176.

Rhoades CC, Eckert GE, Coleman DC. 1998. Effect of pasture trees on soil nitrogen and organic matter: implications for tropical montane forest restoration. Restoration Ecology 6: 262-70.

Richardson SG, Kluson RA. 2000. Managing nuisance plant species in forested wetlands on reclaimed phosphate mined lands in Florida. In: Cannizzaro PJ, ed. Proceedings of the 26th Annual Conference on Ecosystems Restoration and Creation. Tampa (FL): Hillsborough Community College. p 104-18.

Richardson S. 2005. Reclamation of phosphate lands in Florida. Florida Institute of Phosphate Research. Available online at:
http://www.fipr.state.fl.us/research-area-reclamation.htm.

Riekerk H, Korhnak LV, Brown MT. 1990. The hydrology of reclaimed phosphate-mined wetlands. Chapter 7. In: Brown MT, Tighe RE. 1991. Techniques and guidelines for reclamation of phosphate mined lands. Bartow (FL): Florida Institute of Phosphate Research. FIPR Publication nr 03-044-095.

Rosenberry DO, Winter TC. 1997. Dynamics of water-table fluctuations in an upland between two prairie-pothole wetlands in North Dakota. Journal of Hydrology 191: 266-89.

Rushton BT. 2005. Personal communication with Wes Ingwersen. Gainesville (FL): University of Florida, Center for Wetlands.

Rushton BT. 1983. Natural wetland regeneration on clay settling ponds. In: Proceedings of the Eleventh Annual Conference on Wetland Restoration and Creation. Tampa (FL): Hillsborough Community College Environmental Studies Center. p 152-67.

Rushton BT. 1988. Wetland reclamation by accelerating succession [DPhil dissertation]. Gainesville (FL): University of Florida. 283 p.

Schenk HJ, Jackson RB. 2002. Rooting depths, lateral root spreads and below-ground/above-ground allometries of plants in water-limited ecosystems. Journal of Ecology 90: 480-94.

Schneider RL, Sharitz RR. 1986. Seed bank dynamics in a southeastern riverine swamp. American Journal of Botany 73: 1022-30.

Schnoes RS, Humphrey SR. 1980. Terrestrial plant and wildlife communities on phosphate mined lands in Central Florida. Gainesville (FL): Florida State Museum. Special Scientific Report nr 3. 189 p.

SERCC. 2008. Southeast regional climate center period of record monthly climate summary. Chapel Hill (NC): Southeast Regional Climate Center. Available online at: http://www.sercc.com.

Snowdon P, Raison J, Keith H, Ritson P, Grierson P, Adams M, Montagu K, Bi H, Burrows W, Eamus D. 2002. Protocol for sampling tree and stand biomass. Canberra (Australia): National Carbon Accounting System, Australian Greenhouse Office. Technical Report nr 31.

Snyder KA, Williams, DG. 2000. Water sources used by riparian trees varies among stream types on the San Pedro River, Arizona. Agricultural and Forest Meteorology 105(1-3): 227-40.

Sophocleous M. 2002. Interactions between groundwater and surface water: the state of the science. Hydrogeology Journal 10(1): 52-67.

Stauffer AL, Brooks RP. 1996. Salvaged marsh surface and organic matter amendments at a created wetland in central Pennsylvania. Wetlands 17: 90-105.

Theriot R. 1993. Flood tolerance of plant species in bottomland forests of the Southeastern United States. Vicksburg (MS): Waterways Experiment Station, U.S. Army Corps of Engineers. Report nr WRP-DE-6.

Tiner RW. 2003. Geographically isolated wetlands of the United States. Wetlands 23: 494-516.

Tiner RW. 1999. Wetland indicators: a guide to wetland identification, delineation, classification, and mapping. Orlando: CRC Press.

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

USDA. 2004. The PLANTS database, version 3.5 (http://plants.usda.gov). Baton Rouge (LA): National Plants Data Center.

Van der Valk A, Wetzel GP, Cline E, Sklar FH. 2007. Restoring tree islands in the Everglades: experimental studies of tree seedling survival and growth. Restoration Ecology 3: 370-6.

White WN. 1932. A method of estimating ground-water supplies based on discharge by plants and evaporation from soil: Results of investigation in Escalante Valley, Utah. Reston (VA): U.S. Geological Survey. Water Supply Paper nr 659-A.

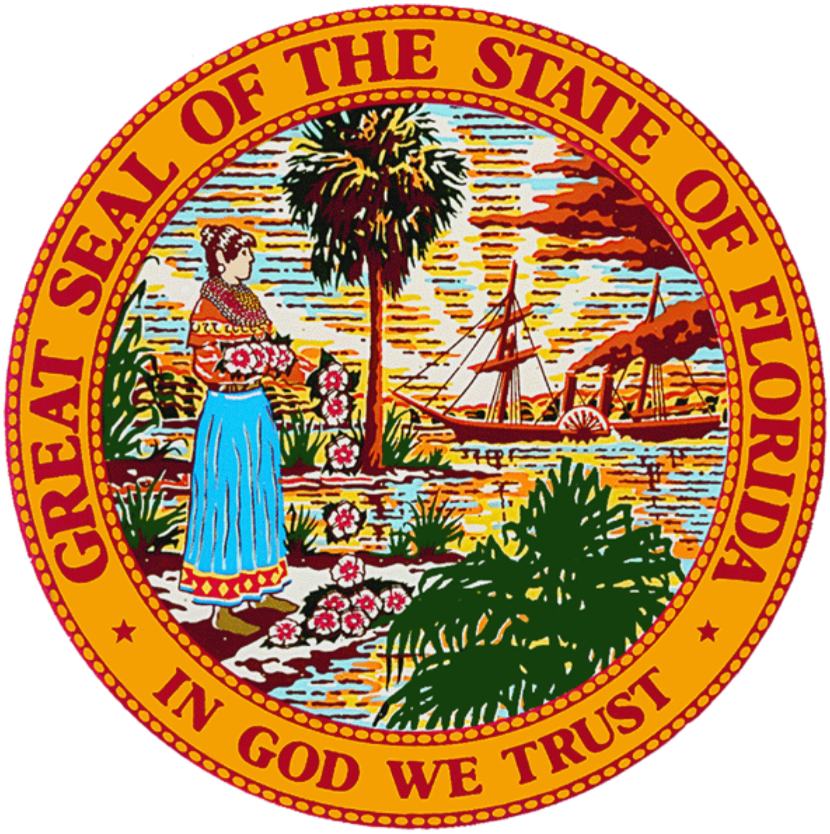
Wilhite LP, Toliver JR. 1990. *Taxodium distichum (l.)* Rich baldcypress. In: Burns RM, Honkala BH, eds. Silvics of North America. Washington (DC): USDA Forest Service. p 563-72.

Woods R, Rowe L. 1996. The changing variability of subsurface flow across a hillslope. Journal of Hydrology 35: 51-86.

Zellars-Williams and Conservation Consultants. 1980. Evaluation of pre-July 1, 1975 disturbed phosphate lands. Tallahassee (FL): Department of Natural Resources, Bureau of Geology.

Zhang P, Albarelli GR. 1995. Phosphatic clay bibliography. Bartow (FL): Florida Institute of Phosphate Research. FIPR Publication nr 02-097-114.

Zuidema PA, Zagt RJ. 2000. Using population matrices for long-lived species: a review of published models for 35 woody plants. In: Zuidema PA, ed. Demography of exploited tree species in the Bolivian Amazon. Rio de Janeiro: PROMAB.


GREAT SEAL OF THE STATE OF FLORIDA
IN GOD WE TRUST