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Spatial Modeling of Coastal Landscapes: Methodological and Scientific Applications.

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SPATIAL MODELING OF COASTAL LANDSCAPES: METHODOLOGICAL AND
SCIENTIFIC APPLICATIONS

A Dissertation

Submitted to the Graduate Faculty of the
Louisiana State University and
Agricultural and Mechanical College
in partial fulfillment of the
requirements for the degree of
Doctor of Philosophy

in

The Department of Oceanography and Coastal Sciences

by
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B.S., College St. Catherine, 1967
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May 1999

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DEDICATION

This work is dedicated to my partner in life, Henry; and to my children and family who have encouraged me. It is also dedicated to all the women friends who over the years have helped me learn life lessons. Thank you.

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I would first like to acknowledge and thank the members of my graduate committee, Drs. John Day, Jr., Joseph Suhayda, William Wiseman, Robert Costanza, Irving Mendelssohn, and Barry Moser. They have been unfailingly patient and supportive of my work. I acknowledge and thank my husband and children for their patience and support and for the astute awareness that the only thing worse than finishing is not finishing. Several people have assisted with data collection, interpretation, and critical analysis during the BTNEP project, work that was integral and necessary to this dissertation. They are Enrique Reyes, Jennifer Pardue, Marie Newman, Hassan Mashrique, Jay Martin, Paul Kemp, and Vibhas Aravamuthan. Thank you to Jami Donley for her last minute, through editing as well as her moral support. Dad, Mom, Harold, Honnah, Henry, Stephen, Jeffrey, Robert and Jenny, thank you. This work has been funded in part by U.S. Fish and Wildlife Service, Environmental Protection Agency, and the Barataria Terrebonne Estuary Program sponsored by the Louisiana Department of Environmental Quality.

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ABSTRACT

A number of issues related to landscape scale ecological modeling of the wetlands of southern Louisiana are examined in this study. First, using geostatistical methods, a new contour map of the wetland habitats in the Terrebonne basin of southern Louisiana is constructed from data collected in 1994. This map is proposed as the best field verified habitat map of the Terrebonne basin and contains statistical confidence intervals associated with the habitat contours. Second, the problem of how to evaluate the success of a landscape model prediction is investigated. The multiple resolution goodness of fit parameter $F_i(k)$ is evaluated in detail and an alternate formulation, $F_i(\mu, \sigma)$ based on a Gaussian distribution is proposed as an alternative. A perfect simulation model would predict a multiple resolution goodness of fit index of 100, in reality it can only approach 91 - 92 when applied to the base maps available for southern Louisiana.

The unit models that best predict the biomass production and the habitat succession are investigated and tested on independent data from nearby wetland sites. Seasonal patterns of biomass production are well reproduced, biomass values fall within literature values, and predicted habitats match observed field habitats. Sensitivity analysis shows parameterization of these unit models to be most sensitive to the translocation rate of biomass between above and below ground biomass, hours of flooding, temperature, salinity, and photosynthetic production rate, in that order.

Finally, the unit models are inserted into a spatially articulated landscape model framework. The results of the landscape simulations are less successful than the unit model simulations. In order to maximize the fit between the simulated habitat map and the reference habitat map, the rate of photosynthetic production has to be increased by an order of magnitude. Possible reasons for this scale dependent change in parameterization are proposed. This study has an immediate application in the science of wetland restoration because management alternatives can now be analyzed in a scientific and

systematic way to evaluate landscape scale cumulative impacts in the context of global climate change.

CHAPTER 1. INTRODUCTION

The development of an ecological model to predict the succession of a landscape is a useful exercise for a number of reasons. In order to advance the body of knowledge about ecological processes such as productivity, diversity and resilience (Golley, 1994) the processes should be objectively measured, and if possible, predicted. If a process can be modeled, the exercise can shed light on the theories, processes and assumptions that were combined to develop the model. On the pragmatic level, it is essential that resource managers adopt a large-scale ecosystem-level view to environmental problems and abandon the piece-meal approach that has often been the mode of operation in the past (Odum, 1989). There are, however, uncertainties in landscape modeling. Researchers investigate and experiment on the scale, processes, and scope of the landscape to be modeled. These are some of the questions that will be addressed in this dissertation.

The term ecosystem was proposed in 1935 by Sir Arthur Tansley to describe units of the environment in which a stable dynamic equilibrium exists between the organisms and their abiotic environment (Golley, 1994). Ecosystem management is based on the principle that ecosystem integrity should be preserved (or restored) if a landscape is to continue to provide sustainable benefits for human populations (Montgomery et al., 1995; Odum, 1989). This requires expanding the role of science in planning to include evaluating alternative management scenarios against intrinsic landscape capabilities (Montgomery et al., 1995). The science of ecosystem management therefore must consider physical and biological interactions that occur over a variety of relevant spatial scales ranging from the size of individual patches of a particular vegetation assemblage to that of an entire region covering thousands of square

kilometers. Likewise, the time scales involved can range from those affecting hydrology, which may be on the order of seconds, to those of the life span of the longest lived plant species and longer (Odum, 1989).

A landscape has been defined by Urban (1987) as “a mosaic of patches, the components of pattern. The agents of pattern formation on natural landscapes can be categorized as disturbances, biotic processes and environmental constraints.” Landscape ecology began in central Europe in the 1960's as a merging of human geography and holistic ecology, with infusions from landscape architecture, land management and planning, and sociology. The first efforts to integrate information provided by hydrologists, engineers, geomorphologists, vegetation scientists, soil scientists, economists, sociologists, and land use planners were made over three decades ago (Golley, 1994) and were conducted to develop creative solutions to planning and management needs (Jenson et al., 1996). The focus of landscape ecology has been on spatially explicit patterns of landscape mosaics and interactions among their elements, primarily at the scale of kilometers (Wiens, 1993).

In his paper, Wiens (1993) concludes that landscape scale ecosystem science is in a period of formulation and, “Existing theory needs to be reformulated in explicitly spatial terms and new theory must be developed to integrate spatial patterns and processes and to consider scaling functions. Empirical research needs to be focused on carefully selected model systems that occupy key positions in ecological or environmental matrices.”

Clearly it is desirable to be able to predict the future of coastal ecosystems, particularly when human lives and vast sums of money are at risk. These are the stakes that exist when planning the fate and future of the coastal wetlands of Louisiana. Aside from anecdotally documenting history and then forecasting from this into the future, the only method we have at hand to reliably predict changes in land loss and habitat evolution is to develop models of the system. The integration of ecosystem analysis and

landscape ecology provides a promising way to analyze ecosystem management alternatives. By exercising the model with out-of-historical-range or future conditions a model can shed light on the possible responses of the system and point out components of the system that are not adequately studied.

CHAPTER 2. LITERATURE REVIEW OF ECOLOGICAL AND LANDSCAPE MODELING

Ecological Modeling

A model is any abstraction or simplification of a system. Alternately, models can be considered devices for predicting the behavior of a complicated, poorly understood entity from the behavior of parts that are well understood (Hall et al., 1990). A brief review of the types and examples of ecological models is presented in Table 2.1. All of these types of models can be predictive that is, used to extrapolate outside the existing data boundaries (Costanza et al., 1985).

Table 2.1. Types and Examples of Ecological Models

Type of Model	Example	Reference
Conceptual or Diagrammatic	box and arrow, Odum diagram	(Hall et al., 1990)
Budgets	nutrient cycling	(Jorgensen et al., 1988)
Population	predator/prey	(Palladino, 1991)
Statistical	fractal dimension	(Barnsley, 1993)
Energy Flow	trophic	(Wootton et al., 1996)
First Principles	photosynthesis	(Charles-Edwards, 1981)

A distinguishing characteristic of an ecological model is that it integrates effects of atmosphere, hydrosphere, lithosphere, flora and fauna into a simplified representation in order to predict the responses between and/or interactions among the system components. Often these types of models attempt to reproduce the processes occurring at a particular location on a particular species, and are known as dynamic ecological models. There are many examples of this type of model. Sievanen (1988) models above and below ground nitrogen dynamics and photosynthesis. Morris (1984b) models atmospheric gas interactions on the growth of *Spartina alterniflora* while Webb (1991) models the same processes on forest growth. Interactions among bacteria,

phytoplankton, and protozoa in a microenvironmental context are modeled by Azam (1988) to predict organic and inorganic fluxes in pelagic ecosystems. Madden (1996) investigated how the balance of limiting resources controls the growth and productivity of submersed plants.

Ecological models can simulate the dynamics of competition, such as the work by Hanski (1997) that merges two predictive mechanisms to show that the species-area curve theory and the positive relation between species' geographical distribution theory can interact. Roughgarden (1988) constructed a model that combines larval circulation with adult interactions to forecast population fluctuations in rocky marine intertidal zones. All of the models referenced thus far have the common feature of integrating multiple effects (often from varying disciplines) into a simulation of the processes to predict a response. Because they simulate a process at one location, I will refer to this type of model as a "unit model" or "module" in the text of this research. These models predict a process in time, but thus far no models have been referenced that predict in time and space.

Landscape Modeling

Spatially explicit dynamic models attempt to reduce the most important processes of the system into equations that mathematically mimic it, just as a dynamic ecological model would. However, unlike the unit model, they incorporate spatially explicit information and processes and transmit (flux) materials across the landscape. This type of model has been most often associated with the engineering disciplines, and has been applied in hydrodynamics (Casulli, 1990; Cheng et al., 1984) and atmospheric general circulation models (Sellers et al., 1997). In at least one comparative study (Prentice et al., 1987), process-based modeling was found to be more accurate in predicting landscape change than Markovian modeling.

It is only recently that the spatial component has been invoked in process-based ecological models. Turner (1989) maintains that Watt was the first to link time and space into successional stages across a landscape. In a comprehensive review of landscape models, Sklar and Costanza (1991a) define a dynamic spatial model as having feedback and interdependencies between time and spatial variables. This definition of a spatial dynamic ecological model includes the concept that space and time are intertwined and cannot be reduced to two independent components. Nielsen (1992) has called these models structural-dynamic models and argues the case for their utility in describing changes in populations and trophic structures of ecosystems.

A number of process based ecological landscape models have subsequently been developed. Researchers at Louisiana State University (LSU) have developed a spatially articulated landscape model with square cells 1 km on a side for a portion of the western Terrebonne wetlands (Costanza et al., 1990; Sklar et al., 1985). Mitch (1991) modeled the hydrology, productivity, and phosphorus in Lake Erie. Reiche (1994) interfaced a model that simulates the soil water and ground water dynamics, surface runoff, soil heat budget and organic carbon and nitrogen transformation processes with data from a Geographical Information System (GIS). A non-aquatic example of this type of model is the simulation of northern spotted owl nesting habitat (Ribe et al., 1998).

The spatial articulation of systems is commonly thought of in Cartesian coordinates, but polygons (Boumans et al., 1991), hexagons (Hunsaker, 1994), and “patches” (Wu et al., 1994) have been used successfully. Spatially articulated Markovian models are in common use in other disciplines (such as politics and sociology) and have been utilized in landscape modeling.

The increased use of dynamic ecological landscape models gives rise to a number of questions. For example, what is the best grid and scale to represent a system? What are the most important processes? How can the landscape be characterized in a systematic way that is consistent and comparable over many years? What are the

appropriate numerical computational methods to use? What can be used as a measure of success or failure of the model? Because landscape modeling involves complex systems, it is difficult to construct controlled experiments on the landscape scale, there are inadequate or non-existent replications of data, and often there are inadequate resources to collect data as well as to run models. Some of these questions will be addressed in detail in this dissertation, others will be left to later researchers to develop more fully.

The CELSS Model

A model previously mentioned, which was developed for western Terrebonne wetlands, is generically referred to as the CELSS model (Costanza et al., 1990; Sklar et al., 1985). This stands for Coastal Ecological Landscape Spatial Simulation, and it has been described by Sklar (Sklar et al., 1991a) as dynamic spatial interaction models with feedback. It incorporates location-specific algorithms that quantify influences from adjacent cells, and has feedback between the processes and the landscape, so that both the landscape and the intensities of the processes affecting it are allowed to change through time. Algorithms incorporating this type of feedback have been implemented using the CELSS methodology in aquatic modeling, (Reyes et al., 1994) and have since been used in terrestrial simulation programs such as PATCHMOD (Wu et al., 1994), ECOLECON (Liu et al., 1994) and the Frankfurt Biosphere Model, (Kindermann et al., 1996).

In the original CELSS model, above-ground macrophyte growth and within-cell nitrogen interactions were simulated with process-based models, and mass balance was utilized for the movement of water and the constituents that the water carried. The model was calibrated by optimizing the fit of the simulated 1978 habitat map to the actual habitat map for 1978, for the model run of 1955 to 1978. The model was verified by comparing the fit of the 1988 simulation results with the actual habitat map for 1988. The actual maps utilized were the 1km² cell U.S. Fish and Wildlife (USFWS) habitat maps that were classified according to the Cowardin method (Cowardin et al., 1979).

The method used to evaluate the success of the model was a multiple resolution goodness of fit parameter F_i (Costanza, 1989) that employs a sliding window of variable pixel size across the landscape and accumulates the number of correct and incorrect predictions. This accumulation is then weighted by a window size that is appropriate for the degree of detail contained in the landscape to be simulated. In the CELSS Terrebonne model, the fit parameter was weighted for pixel windows from 1x1 to 7x7. The fit for the calibration run was $F_i=89.6$ and the fit for the verification run was $F_i=79.0$ (Costanza et al., 1990; Sklar et al., 1991b).

Dissertation Objectives

Landscape modeling in southern Louisiana is difficult and the problems are the result of many factors. Limitations caused by computational technology will continue to be relaxed as the technology of the computer industry continues to advance. Some uncertainties can be addressed by novel methods of data collection and further refinements in the model. Problems involving prediction require a model to be constructed (conceptually, physically or mathematically) and then the model can be exercised to investigate various responses. However researchers have difficulty in measuring the success of landscape models because metrics are not sufficiently robust to capture the complexity of process and form. I will attempt to investigate some of these problems in this dissertation.

The topics that are of interest to me are: (1) Can we quantitatively evaluate the accuracy of predictions of landscape models? (2) Can we accurately predict the seasonal production of marsh vegetation? and (3) Can we accurately predict the habitat succession? These are important questions to answer because they can provide information about the accuracy of predictions of landscape models as well as the basic processes of primary production and possible interactive effects of primary production at the landscape scale.

The specific objectives to be addressed in this dissertation are:

Objective 1 Evaluation of F_t

What are the spatial and temporal limitations on the use of the multiple resolution goodness of fit index F_t as proposed by Costanza (1989) to quantitatively evaluate the accuracy of predictions of landscape models?

Objective 2 Modeling Biomass Productivity

Can a change in the parameterization of the effects of waterlogging (i.e. duration of flooding) and salinity improve the existing unit and landscape model of primary productivity of macrophytes?

Objective 3 Modeling Habitat Succession

Can changes in the mechanistically based habitat evolution more explicitly reflect wetland habitat succession in the unit and landscape models?

To accomplish this research I will do a number of analyses utilizing a number of techniques. In 1994 the Barataria Terrebonne National Estuary Program (BTNEP) funded the development of a landscape model for use in evaluating the effects of management alternatives on the wetlands of the Barataria and Terrebonne basins. In a collaborative effort, I worked with a number of researchers to develop this new landscape model. I will use the resultant BTNEP model (Reyes et al., 1999; White et al., 1997), which is a variation of the CELSS Terrebonne model, as a method to test the hypotheses proposed above. This model was constructed in unit models that simulate individual processes. The unit models were then assembled into a spatially explicit landscape model and the goodness of fit of the validation simulations was measured by a multiple resolution goodness of fit parameter, F_t .

To explore objective 1, I will apply the F_t to all of the landscape scale habitat maps of the Terrebonne basin that are available. In order to extend the range of maps available, I will construct a new habitat map for the basin from 1994 data. To explore objectives 2 and 3, I will implement algorithm changes in the unit models written in the

STELLA™ modeling language and verify them with data from literature and data collected from different locations of similar marsh types. As will be discussed in detail later, some of the unit models contain unrealistic parameterizations and produce unrealistic predictions, and in this study I will attempt to provide modules that are more robust and scientifically accurate. The new unit models will be incorporated into the spatial model at the landscape scale and the level of improvement in the landscape model will be measured by the fit parameter evaluated in objective 1.

CHAPTER 3. METHODS

The methods that I will use to complete the three objectives include developing new unit models, exercising existing landscape models and evaluating the model fit under various conditions. Individual unit models will be constructed and exercised in the STELLA modeling platform and will allow me to improve the prediction of the biomass and habitat succession of representative marsh types of southern Louisiana. Landscape modeling methodology and techniques will refer to and be compared with the Terrebonne portion of the BTNEP landscape model (Reyes et al., 1999; White et al., 1997). Occasionally model results will also refer to the CELSS landscape model (Sklar et al., 1991b). The study area defined by the Terrebonne basin in the BTNEP landscape model is not coterminous with the Terrebonne study area of the original CELSS landscape model and when that becomes problematic in the analysis, mention will be made of the study area.

BTNEP Model

Study Area

The Terrebonne basin is located in the south central portion of the coastal plain of Louisiana (Figure 3.1). It is bordered by Bayou Lafourche on the east and the Atchafalaya River on the west and occupies approximately 5500 km². Morphological features characterizing the area include natural ridges and artificial levees, bays, lakes and bayous, and coastal island barriers and extensive wetlands. The lower portion of the basin contain typical bar-built estuaries. Water bodies average 1-3 meters in depth with bars at the mouth and a low tide, low-energy coast (Penland et al., 1985). The coastline is primarily a beach-dune system with tidal flats and marshes in protected areas behind the barrier shores (Morgan, 1967). Vegetation zone transitions occur from upland bottomland hardwoods, swamp forest, and fresh, intermediate and salt marsh complexes.

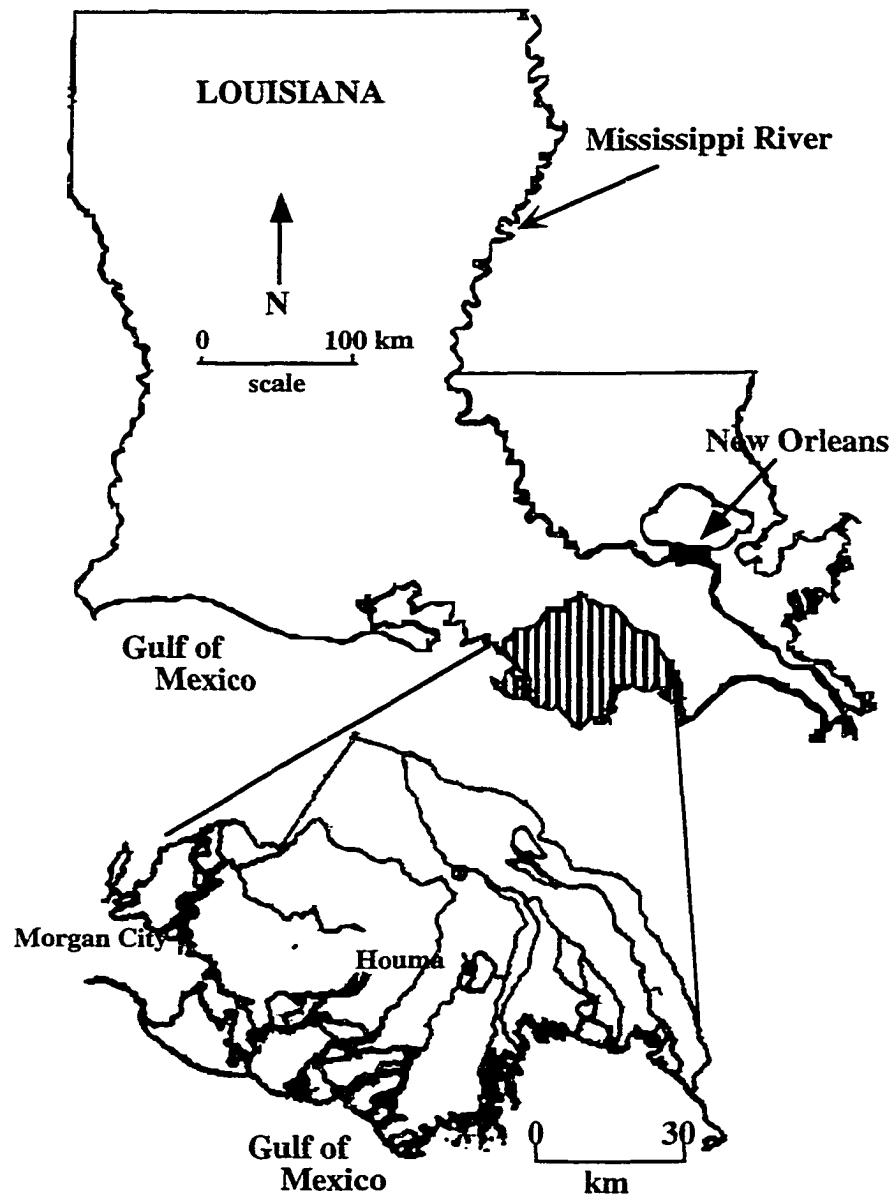


Figure 3.1 Terrebonne portion of the LSU Barataria-Terrebonne National Estuary Program habitat model study area.

The basin is a dynamic system undergoing constant change caused by natural and human processes. The western portion of this basin is directly influenced by the freshwater from the Atchafalaya River discharge and is one of the few locations in southern Louisiana that has experienced net land gain (Roberts, 1997; Roberts et al., 1980). The complex interactions between the enormous volumes of fresh water from the

Mississippi River, Atchafalaya River and the saline waters of the Gulf are controlled and driven by climate events and the shelf topography. In addition, seasonal variations, annual tidal cycles and even decade variations (observed in the adjacent Barataria basin) (Wiseman et al., 1990) have been observed. Recently, (Paille, 1997) noted that Atchafalaya input to the Gulf Intracoastal Waterway has apparently increased in the past decade as stages for a given discharge have risen.

Habitat Distribution

The basin is composed of a number of vegetative communities that reflect a gradient in elevation and in the relative supplies of freshwater derived from the Atchafalaya River, rain, sources of runoff, and higher salinity water from the Gulf of Mexico. Marshes occur as bands of salt, brackish and intermediate vegetation from the Gulf inland. Salt marshes are characterized by an association of *Spartina alterniflora* and *Distichlis spicata* vegetation that gives way to a more diverse assemblage dominated by *Spartina patens* in both the intermediate and brackish marshes. Fresh marshes, whether floating or attached, are more diverse, but most fresh assemblages characteristically include *Panicum hemitomon* and *Sagittaria latifolia*. Fresh marshes give way to swamps and bottomland hardwoods at higher elevations in the most inland reaches of each basin. Deep water swamps are dominated by cypress (*Taxodium distichium*) and water tupelo (*Nyssa aquatica*).

Patterns and rates of land loss and habitat change have been documented by the USFWS from digital maps derived from aerial photography acquired in 1956 and 1978 and from 1988 aerial photography and 1990 satellite imagery (Wicker et al., 1980). These maps are now available in cells or pixels 25 m on a side ($6.25 \times 10^{-4} \text{ km}^2$). This scale was aggregated up to 1 km^2 pixels and the categories were combined to open water, developed fastlands, and four categories of wetlands (Figure 3.2). Each wetland type is characterized in the model by a single dominant species with known responses to salinity

and flood duration. Forested wetlands are characterized by *Taxodium distichium*, fresh marsh by *Sagittaria latifolia*, brackish marsh by *Spartina patens*, and salt marsh by *Spartina alterniflora*.

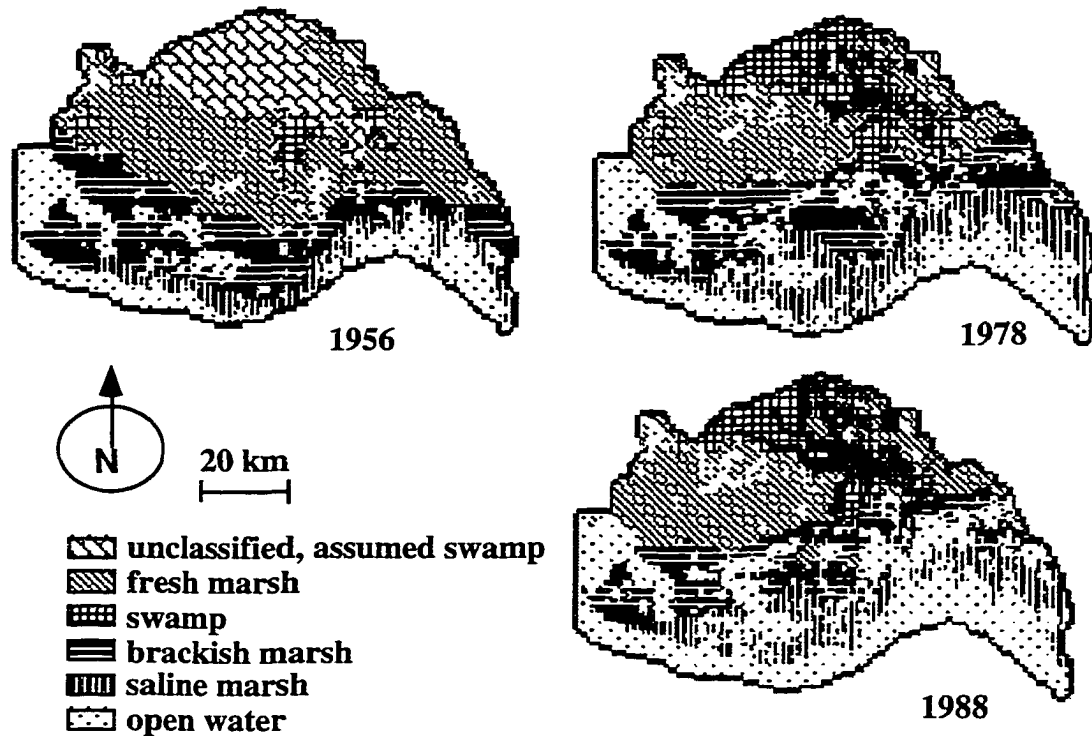


Figure 3.2 Terrebonne habitat basemaps for 1956, 1978 and 1988

Model Characteristics

In 1992 the USFWS expressed interest in expanding the CELSS methodology to the Barataria basin, a wetland hydrologic unit east of the original study area. It was this attempt at wetland modeling that demonstrated the limitations of the mass balance approach of the water component. The Barataria basin does not have the overwhelming influence of a major river to drive water movement. Instead it is a shallow wind dominated basin with excess rainfall as the primary source of fresh water and delayed influence of the Mississippi River that controls the salinity at the Gulf boundary (Conner et al., 1987; Wiseman et al., 1988). The result was hydrologic instability in the model.

One solution, which is neither easily accomplished nor unique, (Baskin, 1993; Lauenroth et al., 1993; Levin et al., 1997; Perestrello de Vasconcelos et al., 1993; Schneider, 1992) is to link modules of different scales in the same model. There are three different time and space scales in the BTNEP application of this technique. The hydrodynamic module uses a 100 km² grid and 1 hour time step, the biological module uses a 1 km² grid and 1 day time step, and in the soil generation and habitat switching module uses a 1 km² grid and 1 year time step. Utilizing scale linking of model components and hydrodynamic equations that conserve mass and energy (rather than mass balance) were the techniques chosen to solve the instability problems. These solutions were applied to the Barataria-Terrebonne National Estuary Program (BTNEP) landscape model for the Barataria basin as well as the Terrebonne basin. Detail of this model can be found in the BTNEP final report (White et al., 1997).

The model is a dynamic spatial landscape model that utilizes a coupling of hydrodynamic, biomass and ecological models. The framework is presented in Figure 3.3 where individual modules are depicted. The hydrodynamic portion is a finite difference, two dimensional, vertically integrated model utilizing a time step of one hour and a spatial cell size of 100 km². The biomass model is of primary productivity and utilizes a time step of one day and a spatial scale of 1 km². The hydrodynamic and biomass results are submitted to a soil generation module and then evaluated by a habitat switching module that allows the landscape to evolve on an annual basis at 1 km² resolution. It is written in FORTRAN modules and runs on the UNIX Cluster at the Louisiana State University System Network Computing Center. At the end of each year of simulation a number of conditions are examined. The habitat conditions are evaluated by a habitat switching routine to see if the habitat has evolved into another habitat type. The daily inorganic deposition is summed and the 1 km² elevation map is updated. The new 1 km² elevation map is averaged to 100 km² for feedback into the hydrodynamic model. Because the Manning coefficient is habitat dependent, the updated 1 km² habitat

map is averaged to produce a new 100 km² Manning coefficient at the end of each year of simulation.

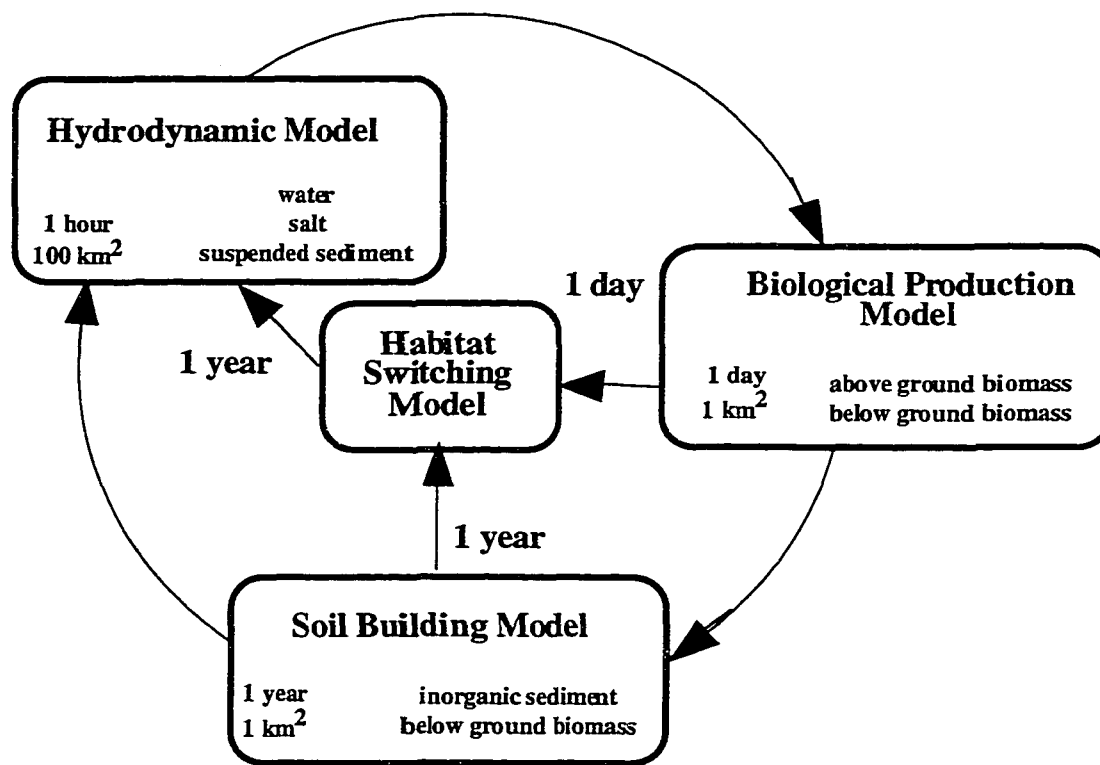


Figure 3.3 Flow of calculations indicating time and spatial scales for the BTNEP habitat model.

Forcing Functions and Boundary Conditions

The forcing functions for the model are wind, rainfall, river discharge and other sources and sinks of water in the basin (i.e. pumping stations). It was difficult to find continuous records of these data for the simulation period for which we had habitat maps (1955-1990), particularly since the hydrodynamic calculations required data at an hourly time step. Data records were investigated, and locations with continuous records closest to the study area were used. The wind record is from Callendar Field south of New Orleans, the closest location that recorded hourly wind observations. Precipitation data is from Houma, temperature maximum and minimum are from Leesville and evaporation is from various southern Louisiana stations. Missing data was reconstructed by

interpolation or by curve fitting (White et al., 1997). A survey contracted by the BTNEP in 1994 (Alawady et al., 1996) supplied land elevation in 134 locations across the two basins. These data were interpolated to provide the land elevation map.

The boundary conditions for the hydrodynamic model were the Gulf of Mexico tide elevation and salinity on the south boundary. The data are from Grand Terre, a data station about 100km. east of the study area. A previous study (Sklar et al., 1991b) shows that there is a high correlation ($r^2=.88$) between the time series at Grand Terre and a station in Terrebonne study area, East Cote Blanche Bay. The boundary conditions at the Gulf for salinity were set using modified salinity records collected from Grand Terre. Salinity was adjusted using seasonal longshore gradients observed in the LATEX-B study (Murray et al., 1995). In general, salinity values were lowest at the Atchafalaya delta and became progressively higher toward the east in the Terrebonne basin. The difference in salinity was seasonal and ranged from 3 ppt. to 9 ppt. The Atchafalaya River discharge and suspended sediment in the Terrebonne basin and various pumping stations and discharge locations at the perimeter of the basins were used as input. In addition, relative sea level rise was imposed separately at the Gulf of Mexico

There are questions of appropriateness when imposing data from outside the study area onto a model. The ideal situation would be to have a number of stations across the basin. Unfortunately, this is not available for the length of records that are required for this type of modeling. The temperature records from Leeville are probably a good approximation to the temperature in the southern Terrebonne basin. This data station is not far removed (50-60 km) and temperature is probably the most gradually changing forcing function over the distances in question. Rainfall is a more spatially variable parameter.

The most suspect data set in the BTNEP model is the wind data. The hydrologic model is quite sensitive to the forcing of wind in such a shallow basin and that data is collected from the location farthest from the study site. The choice to use the wind data

from Calendar Field was made because it was a well documented, long term, and consistently maintained station. The methods used to correct the data from sensor height to sea level and to reproduce missing data are described in detail in the BTNEP final report (White et al., 1997). Effects of the wind are incorporated into the hydrologic portion of the model only. Their only impacts to this dissertation will be in the amount the hydrology contributes to the model, and since the hydrology will be held constant in all landscape simulations, it should not be a factor in the conclusions of this work.

CHAPTER 4. USING GEOSTATISTICS TO CONSTRUCT A 1994 HABITAT MAP OF TERREBONNE BASIN

Introduction

Data of sufficient quality and quantity to parameterize and validate landscape models is one of the most difficult challenges to overcome in the discipline of landscape modeling. In the last chapter, some of the problems of time series records and boundary conditions were mentioned. Even more problematic is obtaining a reliable habitat classification data set that is consistent in scale and vegetation classification over a landscape. The previously referenced USFWS habitat maps are one source of this data and their value lies in the time series (1955, 1978, 1983, 1989-90) that is available. In order to investigate the multiple resolution goodness of fit parameter, F_t (objective 1 of this study) it will be necessary to apply this index to as many landscape scale habitat maps of the Terrebonne basin as are available. To extend the range of maps available, I will construct a new habitat map for the basin from 1994 data collected by the National Resources Conservation Service (NRCS).

Geostatistics allows an ecological researcher to explore data in ways previously unavailable. It is particularly useful and applicable to landscape ecology, where large-scale trends are sought in data that is difficult to collect in a regularly gridded pattern. Geologists were the first to fully develop the concepts and there are many examples of geostatistical applications in the soil science literature (Burgess et al., 1980a; Burgess et al., 1980b; Hill et al., 1995; Matheron, 1963). However the value of geostatistical techniques has been recognized by other disciplines and many recent examples of their application can be found. Fortin (1989) uses this technique to study the spatial structure of sugar maple tree density. Boyer (1997) described the spatial dependence and variation of water quality patterns in southern Florida. Robertson (1988) mapped spatial

variability of nitrogen mineralization, nitrification and denitrification. And Saanderson (1998) mapped water canopy cover in a marsh using satellite data.

There are many pertinent summaries of these techniques to recommend to the reader (Matheron, 1963; Rossi et al., 1992; Ver Hoef et al.,) and a summary of the two used in this analysis follows. They are 1) variography, a method to model spatial dependence using autocorrelation estimates, and 2) kriging, a method to provide estimates, without bias and with minimum and known variance, for unrecorded locations.

NRCS Data Set

In 1994 the Natural Resources Conservation Service (NRCS) surveyed the soils of Terrebonne parish in Louisiana. The BTNEP contracted for additional data collection to take place during this survey. The procedure used for vegetative data acquisition was described by Larry Trahan (personal communication) and can be summarized as:

1. Samples were collected at approximately 1 minute latitude and longitude intervals (approximately 1 km.).

2. Access to the sites was made by helicopter. As the helicopter hovered over a site, an initial percent land/water determination was made. This was described as “green vs. not green”. Heavy stands of floating aquatic vegetation would be characterized as “green” and thus land.

3. Two person teams covered the site. In addition to the soil core, a visual inspection of a 100 foot diameter circle was made to identify the vegetation. The team identified the species and percent coverage of each from a list of 131 common plant names (Appendix A). Total percent coverage for each site summed to 100% that characterized the area was previously defined as “green” or land.

The data presented in Table 4.1 was collected at approximately 1 km. intervals throughout the Terrebonne parish portion of the BTNEP study area (Figure 4.1). The

extreme northern and eastern parts of the basin were not covered as they lie in Lafourche Parish. Using this rich data set I have generated a 1994 habitat map. This will allow verification for the year 1994 and will assist in the verification of some of the BTNEP unit and landscape model parameterizations.

Table 4.1

Parameters reported from soil survey conducted by Natural Resources Conservation Service (NRCS) May 5 - June 16, 1994.

Parameter Recorded	Notes
1. soil series name	
2. record number	
3. USGS quadrangle designation	
4. stop number on quad	
5. latitude	
6. longitude	
7. sample number if lab analysis	7. 28 soil samples were retained for further analysis.
8. percent water area at stop	
9. depth of water	
10. horizon designation of layer #	10 - 21. Up to nine horizons were described in a core of approximately 2 m.
11. upper limit of layer #	Parameters 10 - 21 were reported for each horizon that was described.
12. lower limit of layer #	
13. broken face color layer #	
14. soil texture layer #	
15. fiber content, unrubbed, layer #	
16. fiber content, rubbed, layer #	
17. percent mineral content, layer #	
18. structure, layer #	
19. consistence, layer #	
20. interstitial salinity, layer #	
21. pH layer #	
22. percent occurrence, plant #	22. Surface vegetation was reported as percent occurrence by plant code number. There were 131 possible plant choices.

Map Construction

In order to construct a habitat map from vegetation data, scientific names were assigned to the common names on species list (Appendix A) using Tiner (1993), Materne (1996), Radford (1968), and Godfrey (1981) as references. Each species was then assigned the category of fresh, fresh-intermediate, intermediate, brackish, brackish/saline or saline wetland (there were no instances of intermediate/brackish). This determination was made using the above references and personal communication (Materne, 1997; Mendelssohn, 1997; Trahan, 1997).

Habitat type is a categorical classification, and in order to use the kriging procedure, the data must not only be continuous, but also linear. Assignment of numerical values to categorical data can only be done with the utmost care so that analysis will not be invalid. If the habitat category vs. typical salinity is assigned as in Table 4.2 (Mitsch et al., 1993) the relationship between salinity and habitat type is a continuous relationship only by accident of design of code designation. This relationship is not a linear function; that is, the salinity of habitat three is not three times the salinity of habitat one (Figure 4.2).

Table 4.2 Typical salinity and salinity used in kriging associated with habitat type

Habitat Type	Habitat Code	Typical salinity* (ppt)	Kriging Salinity (ppt)
fresh	1	<0.5	0.02
fresh/intermediate	2	0.5 - 5	2.5
intermediate	3	5.0 - 18	11.5
inter/brackish	4		17.5
brackish	5	18.0 - 30	24.0
brackish/saline	6		29.5
saline	7	30 - 40	35.0

* from Mitch and Gosselink, 1993

To transform the data so that it could be validly used in kriging, two manipulations were done. 1) The relationship between habitat category and salinity was represented as the square of the habitat code (Figure 4.2). This relationship is nearly linear, $r^2 = 97\%$, particularly in the low salinity habitat types. 2) The data for each station consists of a number of species and their percent occurrence. The value for habitat type was used to calculate a weighted average of habitat type (habitat index) for each station.

An autocorrelation analysis was performed on the habitat index and a semi-variogram of this analysis is presented as the data points in Figure 4.3. In practice the

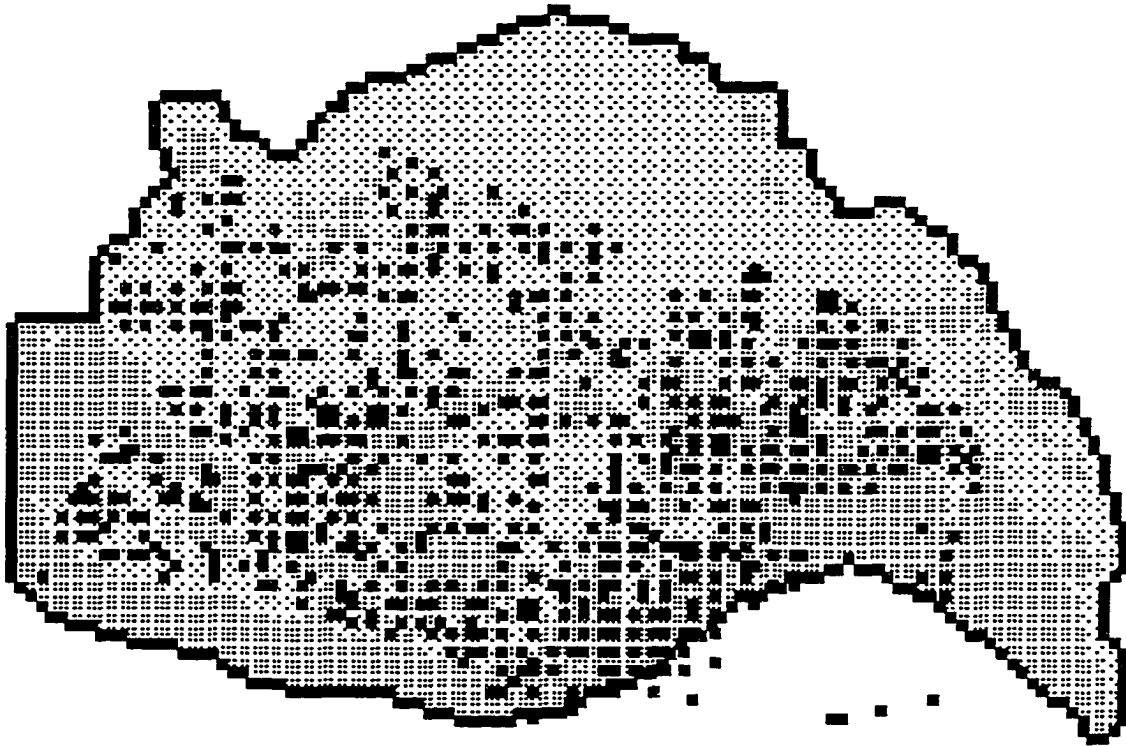


Figure 4.1 Terrebonne BTNEP study area with NRCS soil sample locations and study area boundary are indicated by dark solid cells. Each cell is 1 km² and a solid cell can represent more than one observation. There were 1169 observations made between May 5 and June 16, 1994. Light background pattern indicates land, denser background pattern indicates water. Note that the soil samples do not cover the whole Terrebonne basin study area, but rather stop at the parish (county) border.

calculation of autocorrelation estimates is usually constrained by the computer utilized and software limitations. If this is the case, the number of pairs of autocorrelation estimates is trimmed by some factor. In this analysis, the MGAP software by RockWare Scientific Software (RockWare, 1993) was used, and the program was limited to 32,000 pairs of data. Rossi (1992) states that each lag class must be represented by at least 30-50 pairs of points. In this analysis, 18162 pairs of points were used to construct a semi-variogram with 100 lags, and there are an appropriate number of pairs of points in each lag bin. The model that best represents the variogram distribution is a Gaussian model with a sill of .343, nugget of 0.065 and a range of 43.00 (Figure 4.3). The proportion of the variance of this data that can be modeled as spatially dependent is 81% (sill-

nugget)/sill and the distance at which data is no longer spatially correlated is 43 km. (range) (Rossi et al., 1992).

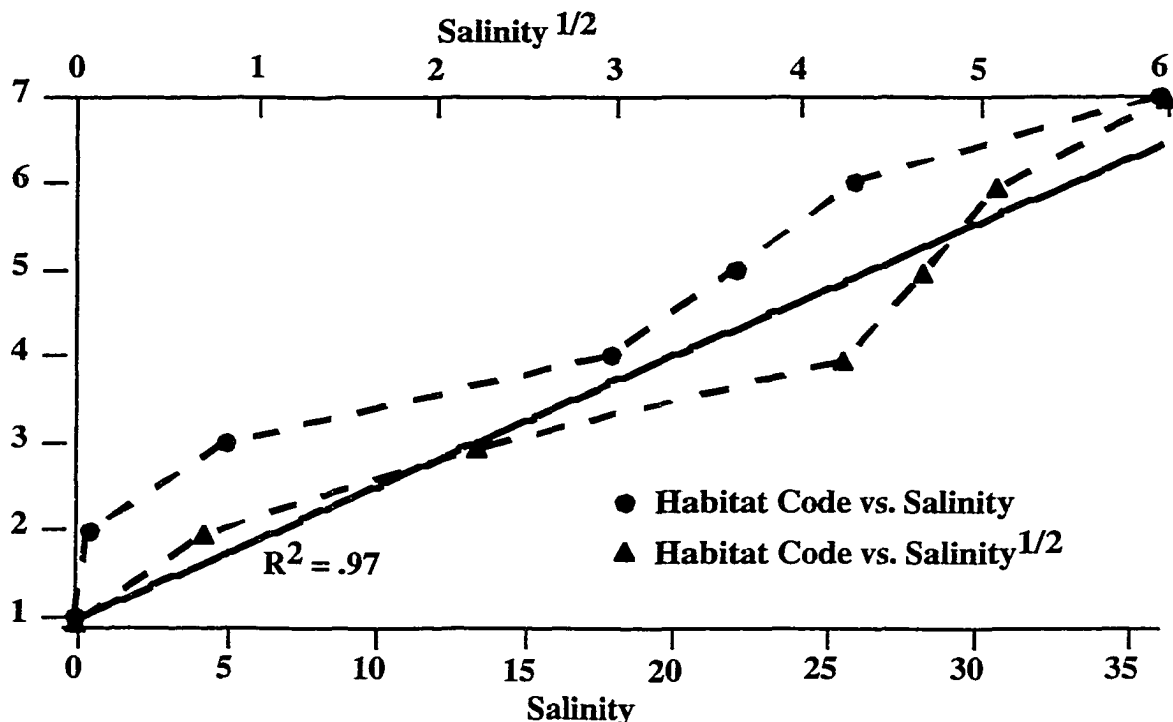
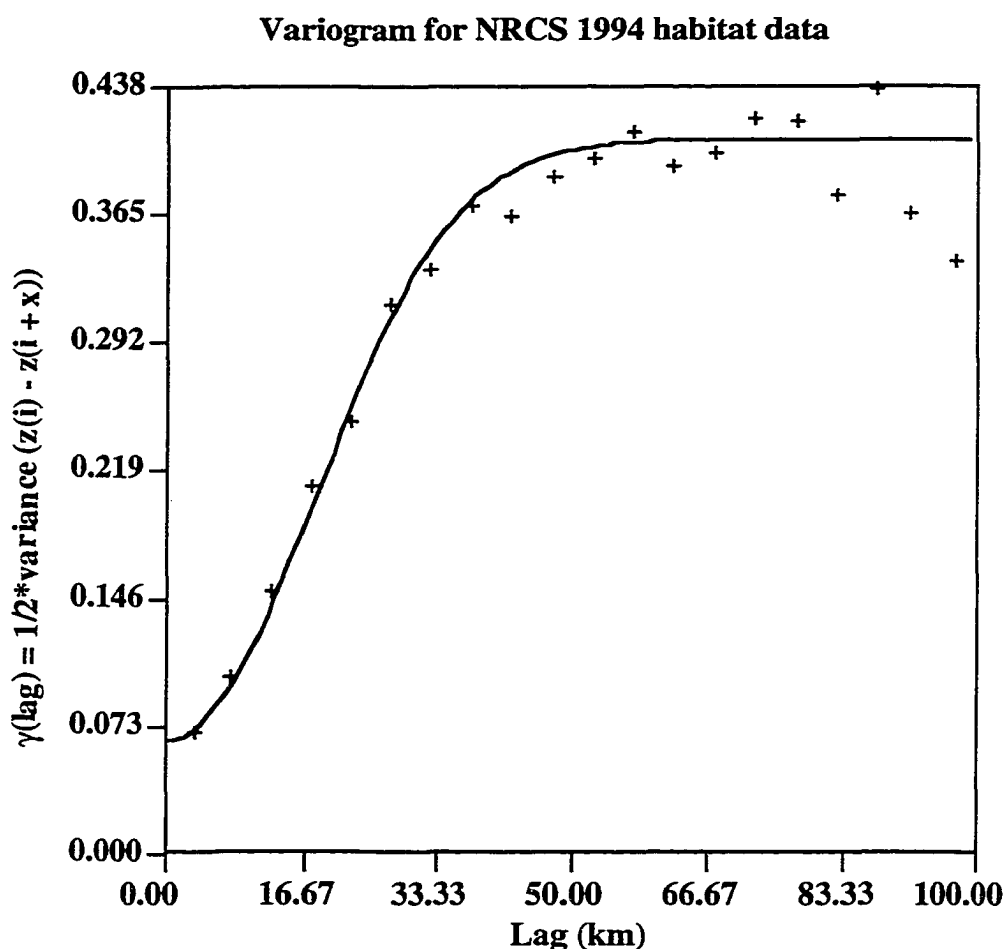


Figure 4.2 Habitat code vs. salinity and salinity^{1/2} for typical habitat types and the corresponding salinities for southern Louisiana. Regression line is for salinity^{1/2} vs salinity.

There are a number of kriging options that are available. The simplest choices are punctual and block kriging. With punctual kriging, values for exact points within the sampling unit are used, while block kriging involves estimating (or averaging) values for areas within the unit. (Robertson, 1987). Simple punctual kriging will produce a map with intricate isograms and fairly large estimation variance, a worse case estimate (Burgess et al., 1980b). Average values over areas rather than point values, obtained by block kriging, yield estimations with variances that are very much smaller (Burgess et al., 1980a).



Parameters	Variable Limits	Model Information
# of pairs: 18162	Min. 0.995	Nugget = 0.065
Direction: 0.0	Max. 2.449	Gaussian: Sill 0.343,
Tolerance: 90.0	Mean: 1.604	Range 43.00
Bandwidth: MAX	Variance: 0.290	

Figure 4.3 Semi-variogram for vegetation data collected by NRCS in Terrebonne parish, LA in 1994. Data was represented as the square root of the habitat index as described in the text.

Co-kriging is another option. With co-kriging, the data analysis is supplemented with another data set that is highly correlated with the first. It could be argued that the NRCS data set contains other variables that could be used in co-kriging the vegetation data. However, the vegetation in a wetland area is the long-term integration of many variables, including of the salinity, water elevation and soil type. It is important not to

confound these effects by including them in the map generation. For this reason, simple kriging, not co-kriging was used.

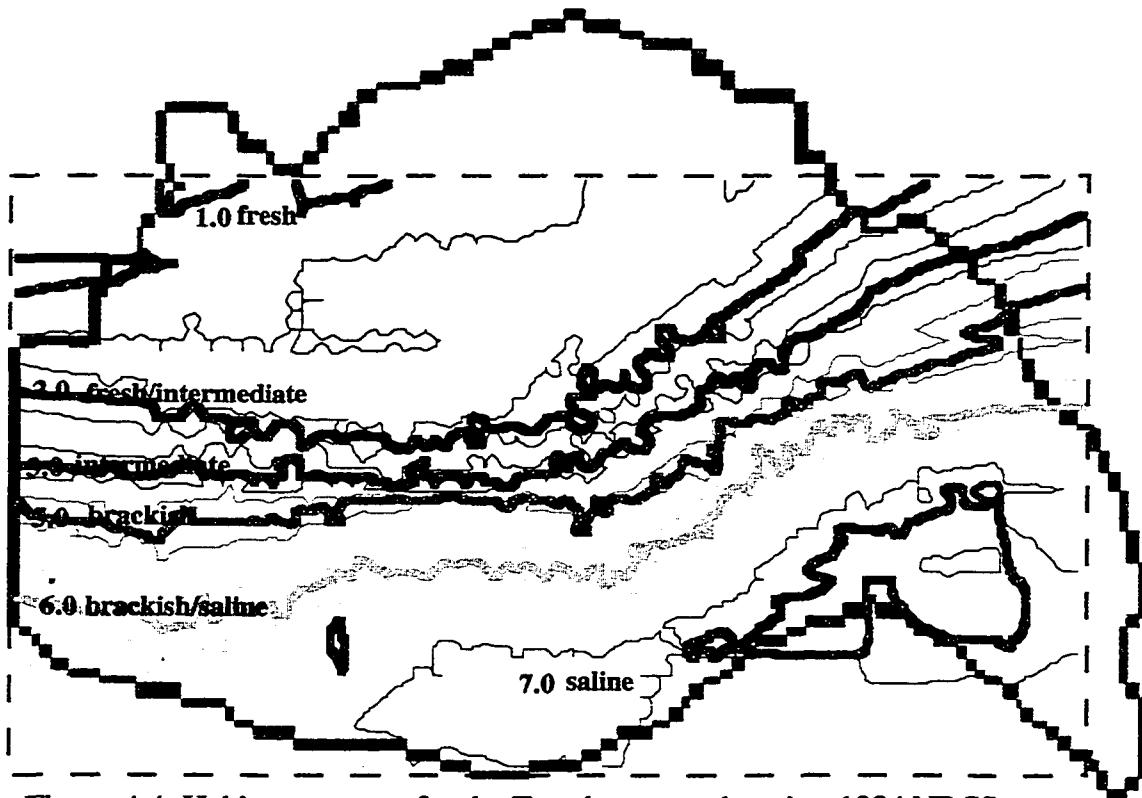


Figure 4.4 Habitat contours for the Terrebonne study using 1994 NRCS species composition data. The study area is bounded by the heavy black border. Dashed line indicates the limit of the NRCS data collection. Heavier color lines indicate contour intervals, lighter color lines are the 95% confidence interval for that contour. Habitat data is transformed such that 1 = fresh marsh through 7 = salt marsh. See text for details.

The map in Figure 4.4 was contoured using data that were punctual krigged with a Gaussian model. The resultant estimates were then inverse-transformed from the habitat index into a habitat code.

A number of features of this map are noteworthy. The resultant vegetation can be considered a proxy for the long-term integration of water mixing patterns. It is interesting to observe the northerly extent of marsh denoted as fresh. Previous analyses report most of the northern part of the basin as fresh marsh, however this analysis shows

the effect of salt intrusion, and consequent limited extent of marsh that can, with 95% statistical confidence, be called fresh.

The effect of the Atchafalaya River is evident and the only area of pure salt marsh (with the exception of one small island) is located in the extreme south east of the basin. The Houma Navigational Canal is located in this high saline area, and the drinking water intake for the city of Houma is located at the northern end of the Houma Navigational Canal. These results suggest that any manipulation of the ratio of waters from the Mississippi and Atchafalaya Rivers will have long term effects on the salt water intrusion for this area.

This map is the first habitat map of the Terrebonne basin (and perhaps in southern Louisiana) that contains statistically significant confidence intervals associated with the habitat types. In addition, the data it was constructed from are all actual observations, not interpretations of habitat. These qualities make it one of the most reliable habitat maps available to date.

CHAPTER 5 ANALYSIS OF MULTIPLE RESOLUTION GOODNESS OF FIT MEASUREMENT

Introduction and Review of Spatial Indices

Before determining whether improvements have been made to a landscape model, it is necessary to investigate how a model's performance can best be measured. In a recent dissertation, Ehlschlaeger (1998) has discussed this topic in detail. He presents the example that the states of Utah and Wisconsin have approximately the same percentage of surface area covered by water, however, Utah's water surfaces comprise several large water bodies, whereas Wisconsin has many smaller water bodies. This simple example illustrates the challenge and importance in choosing a metric that captures heterogeneity. The metric we are seeking would be one that quantifies size, shape and configuration of species structure and distribution by comparing the model results with a reference scene. Fortunately quantification of spatial patterns (which is one result of a landscape model) is an emerging field with a number of spatial indices regularly reported (Turner et al., 1991).

A review of the literature by Downing (1991) indicates that 16%-25% of ecological research is based on ecosystem comparisons and one-third of these comparisons employ some form of regression analysis. Other methods frequently used include the calculation of confidence intervals and one-way to multi-way techniques for performing parametric and non-parametric analysis of variance (Downing, 1991). These methods are inadequate to evaluate from landscape models however, because they do not convey any spatial information. On the contrary, they generally assume that the data is independent of each other and are distributed identically. (Rossi et al., 1992).

Boundaries or shapes can be quantified using fractals, and the fractal dimension can then be used as a measure of the complexity of spatial patterns. It is a useful metric to investigate shapes of boundaries, nested relationships and the scale of processes

creating the pattern (Bellehumeur et al., 1998). Fractal indices have been used most successfully in ecological modeling to study habitat fragmentation (Olsen et al., 1993) (Milne, 1992). But it is not a metric well adapted to evaluate landscapes with more than two categories (inside and outside the boundary). Interface analysis is a better choice if the amount of edge is important, such as for flux relationships or evaluation of shoreline habitat (Turner et al., 1991), but cannot capture shape or adjacency information.

An additional limitation to the indices and statistics described thus far is that they do not directly compare a modeled scene to a reference scene. The comparison of two maps requires the comparison of the derived indices. One method used for the direct comparison of two maps is the confusion matrix, also known as the contingency table or error matrix. Usually, this matrix is used to compare a classified satellite image with a reference data source such as ground-based sampling (Klinkenberg et al., 1994). A deficiency in this metric for our purposes is its inability to include spatial relationships. Another index that directly compares one map to another, and can capture the frequency and spatial distribution of that comparison is the multiple resolution goodness of fit parameter (Costanza, 1989). This index was used to evaluate the results of the CELSS model and the BTNEP model. It is this index that I will evaluate in detail.

Multiple Resolution Goodness of Fit Analysis

An analysis of the multiple resolution goodness of fit parameter is important in understanding the evaluation of modeling results of spatial landscape models in southern Louisiana wetlands. The questions and techniques, however, are applicable to any number of spatial patterns in a temporal framework where one desires a consistent and objective measure of goodness of fit. For example, a spatial model may give somewhat accurate predictions that are mis-registered and the contours of expected results are shifted north-south and/or east-west by a few cells. Likewise, the results might be

temporally mis-registered, i.e. the correct spatial prediction might occur earlier or later than the data that was collected in the base map.

To quantitatively evaluate the results of the CELSS and LSU BTNEP landscape models, the fit parameter introduced by Costanza and Sklar (Costanza, 1989) was used. Because I will be referring to this analysis extensively, I introduce it to the reader in detail. Suppose map 1 (Figure 5.1), subdivided into individual cells, represents the actual landscape and map 2, subdivided into the same cell structure, represents the simulated landscape. Each cell can be one of four categories. We want to measure how well map 2 matches map 1. At first glance, the maps do not resemble each other.

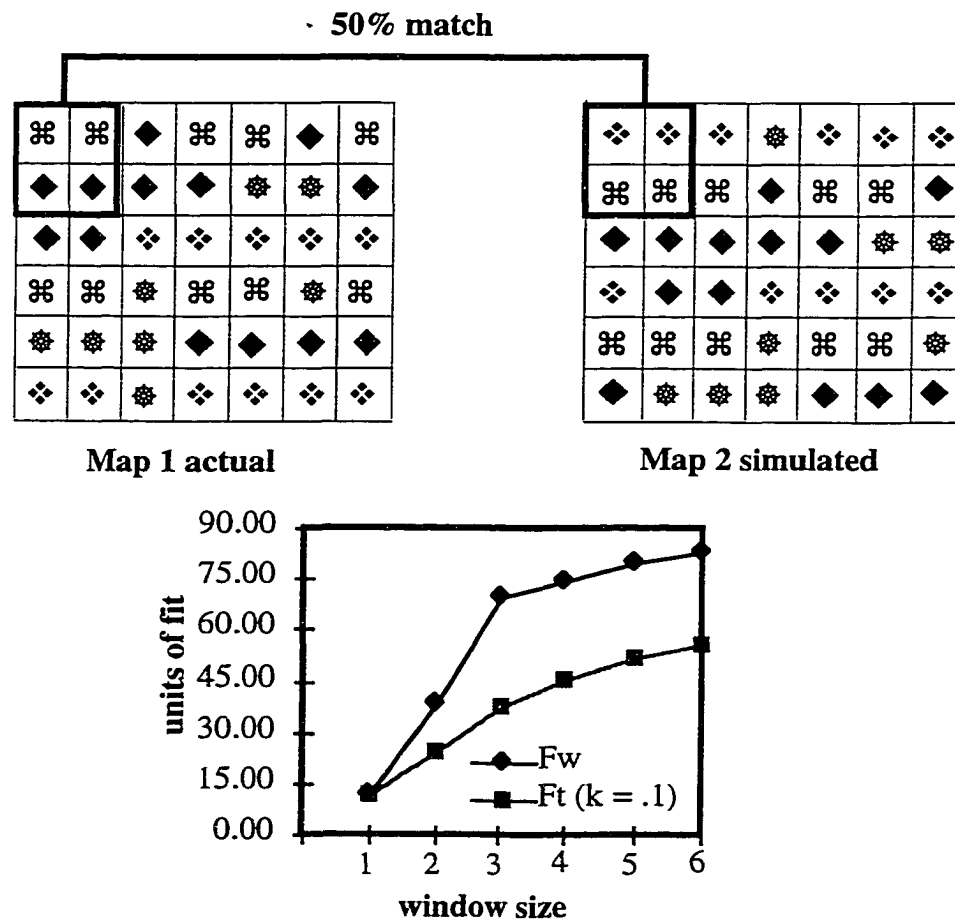


Figure 5.1 Graphical representation of the terms defined for the fit parameters Fw and Ft. Fit was calculated using the sample maps Map 1 (actual) and Map 2 (simulated).

One way to measure the match is to compare cell by cell and to define the measure of the accuracy as a percentage of correct cells. A score of 100% means map 2 exactly duplicates map 1. In this case, the score is 12%. This comparison only focuses on results that are on the scale of one cell. In this example, map 2 exactly matches map 1 except that the maps are mis-registered (map 2 is shifted down and to the right by one cell). The score is very low, even though most of the complicated pattern of the map is well reproduced.

The measure of goodness of fit should incorporate the information about the spatial pattern of the map being investigated. If, instead of comparing cell by cell, one makes a square "window" of cells and calculates matches when moving the outline of the window over the maps, the spatial pattern can be better accounted for. The window does not have to be a square if there is some compelling geographic reason for it to be another shape, but it does facilitate calculation. Using this example and a 2x2 window, the first 4 cells of map 1 are compared with the first four cells of map 2. If there is a correct proportion of each type of habitat, then that will be called a match. The match in this case is 50%. Moving the window outline over one cell and repeating the comparison would yield a fit for a window size of w:

$$F_w = \frac{\sum_{s=1}^{t_w} \left[1 - \frac{\sum_{i=1}^p |a_{1i} - a_{2i}|}{2w^2} \right]}{t_w} \quad \text{equation 5.1}$$

where:

F_w = the goodness of fit for a sampling window size of w

w = the dimension of one side of the square sampling window

a_{1i} = the number of cells of habitat i in map 1 within the window

a_{2i} = the number of cells of habitat i in map 2 within the window

p = the number of different habitat types in the sampling windows

s = the sampling window of dimension w by w that slides through the maps one cells at a time

t_w = the total number of sampling windows in the maps for a window size w .

When $w = 1$, $F_1 = 12\%$ the same cell by cell percent accuracy as was described above. When the sampling window is as large as the smaller dimension of the map, t_{\max} , F_{\max} will be a comparison of the frequency distribution of map 1 with map 2 and there will be no information about the relative positions of the habitat.

There will be as many F_w as the number of sampling windows within the map dimensions. In the sample case, Figure 5.1 contains a graph of the relationship of F_w to window size. To determine an index that gives an overall degree of fit, the F_w 's should be summarized in some manner. For this purpose, the weighted average used by Costanza yields a multiple resolution goodness of fit index, $F_t(k)$.

$$F_t(k) = \frac{\sum_{w=1}^n F_w e^{-k(w-1)}}{\sum_{w=1}^n e^{-k(w-1)}} \quad \text{equation 5.2}$$

The parameter k is a value that determines how much weight is to be given to small sampling windows vs. large sampling windows. When $k = 0$, all window sizes have the same weight. When $k = .1$, only the first few window sizes will have any significant contribution to the $F_t(k)$. In the sample case described above, the fit of map 2 to map 1 is $F_t(k) = 51.23$. When the sample map 1 is compared to randomly generated maps, the average fit is $F_t(k) = 54.37$.

When only the western portion of the Terrebonne basin was modeled in the original CELSS study (Sklar et al., 1985), the calibration run from 1956-1978 had $F_t(k) = 88.2$ and the validation run of 1978-1983 had $F_t(k) = 79.0$. For these model results a $k=0.1$ was chosen, which tends to weigh sampling windows of 1 - 8 cells most heavily

(at window size 8x8, the weight given to F_w is .50). In order to be able to compare the results of $F_t(k)$ from previous analysis to this research, and because we are interested in maximizing the fit at smaller window sizes, the value that will be used for the parameter k will be $k=.10$.

The multiple resolution goodness of fit parameter is dependent on the total map size and the number of categories in the map. Figure 5.2a illustrates the resultant F_w when two randomly generated maps that are 77 x 112, and contain varying numbers of categories, are compared. At the window size 77 x 77, no individual F_w is less than 95. Figure 5.2b illustrates the summation fit, $F_t(k)$, which ranges from 87 and 73. And Figure 5.2c contains the summary $F_t(k)$ versus number of categories. When two random maps with the same number of categories as the BTNEP Terrebonne study area (5 categories) are compared to each other, the $F_t(k)=75.24$. While the differences in the values do not seem to be very great, all categories are statistically significant at $p<.05$. When the 1988 habitat map is compared to randomly generated maps in a Monte Carlo analysis, the fit is 40.31 (significant at 95%, Appendix B).

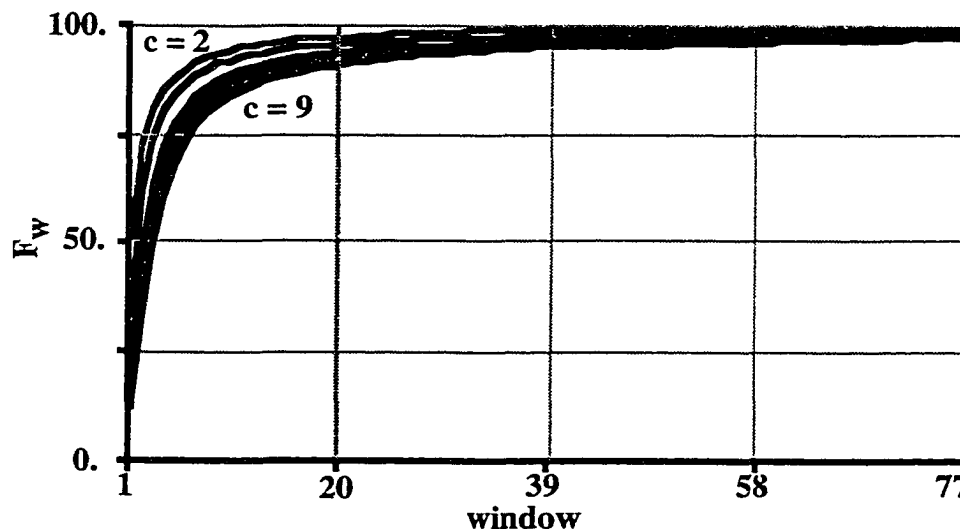


Figure 5.2a Individual F_w calculations for comparison of two randomly generated 77 by 112 pixel maps with varying numbers of categories.

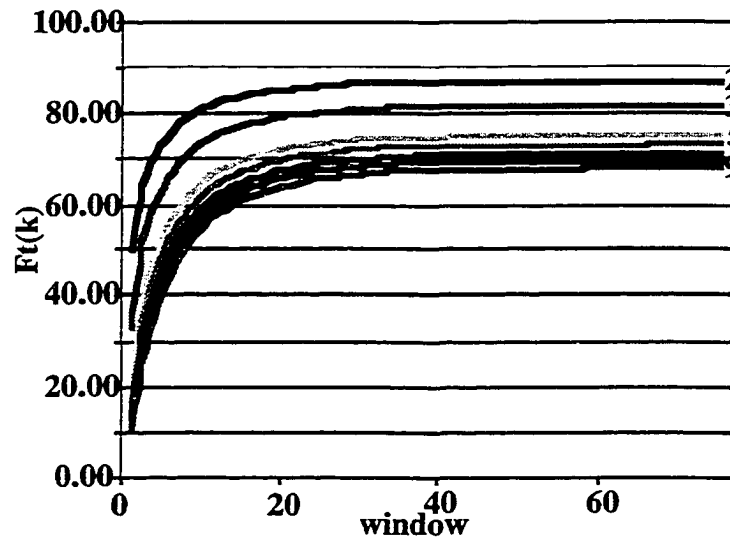


Figure 5.2b The summary index $F_t(k)$ versus window size for the random map analysis described in Figure 5.2a. The number of categories contained in the map analysis ranges from 2 (top curve) to 9 (bottom curve).

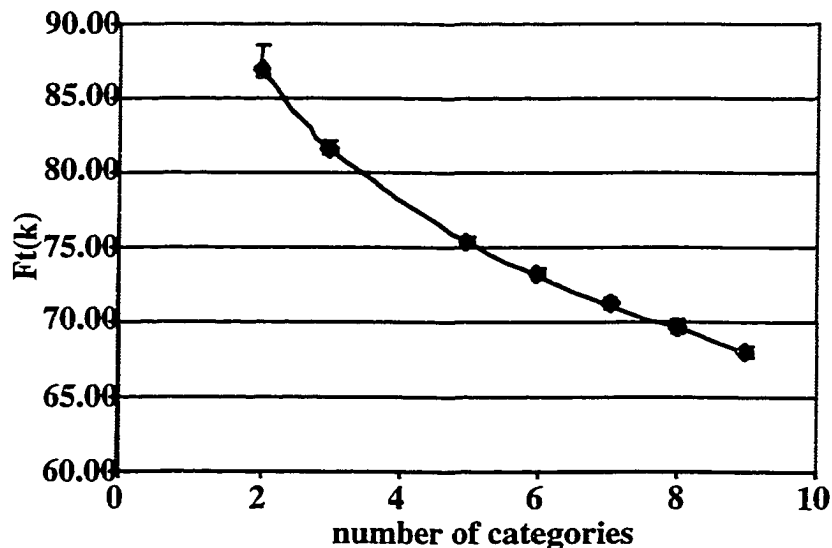


Figure 5.2c Multiple resolution goodness of fit parameter $F_t(k)$, $k = .1$, as a function of the number of categories contained in the randomly generated 77×112 pixel maps. Note that error bars ($n=10$) are can only visually be discerned for category =2 and the differences in the $F_t(k)$ are statistically significant at $p < .05$.

Some interesting considerations emerge when using the fit to evaluate landscape modeling. (1) How does the accuracy of the base map influence $F_t(k)$? (2) How much does $F_t(k)$ vary from year to year? That is, how does the $F_t(k)$ change due to the actual

habitat distribution changes from year to year? (3) How does the summarization routine, in particular the weighting function, influence the $F_t(k)$?

With respect to question 1, the question of base map accuracy, two questions arise a) how variable is the $F_t(k)$ for a habitat map constructed for the same year using two different methodologies and b) what variability is introduced by the scale of the base maps? By answering questions 1 and 2 we can determine an upper limit that can be expected on $F_t(k)$ for spatial landscape models in southern Louisiana. By analyzing question 3, the weighting in the $F_t(k)$, the most appropriate function for use in southern Louisiana landscapes can be determined. This function will best account for the uncertainties that we have identified in questions 1 and 2. Fortunately there are a number of data sets that will allow us to investigate these questions.

Variability of $F_t(k)$ due to Base Maps - Question 1

As was stated previously, one of the most difficult data sets to obtain is a consistent and accurate habitat map of a large landscape. If the reference map is questionable, the index of fit that uses the reference map is also questionable. Uncertainties in the construction of habitat maps can occur as a result of differences in mapping methodology, as well as differences in the scale of the maps. An investigation of these two sources of variability follows.

$F_t(k)$ Variability due to Mapping Methodology - Question 1A

Four data sets from various years in the Terrebonne area were classified by habitat type for the USACOE by Chabreck ((Visser et al., 1996); Chris Brantley, 1997, USACOE, personal communication). Habitat types were mapped on existing USGS quads based on vegetative transects conducted in 1988, 1990, 1992, and 1994. These maps were then scanned, geo-referenced, aggregated into 1 km² cells and cropped to fit the boundaries of the Terrebonne study area. Brackish and intermediate marsh

classifications were aggregated to one class - brackish marsh. The scanned maps and the associated 1 km² data sets are shown in Figure 5.3.a - d.

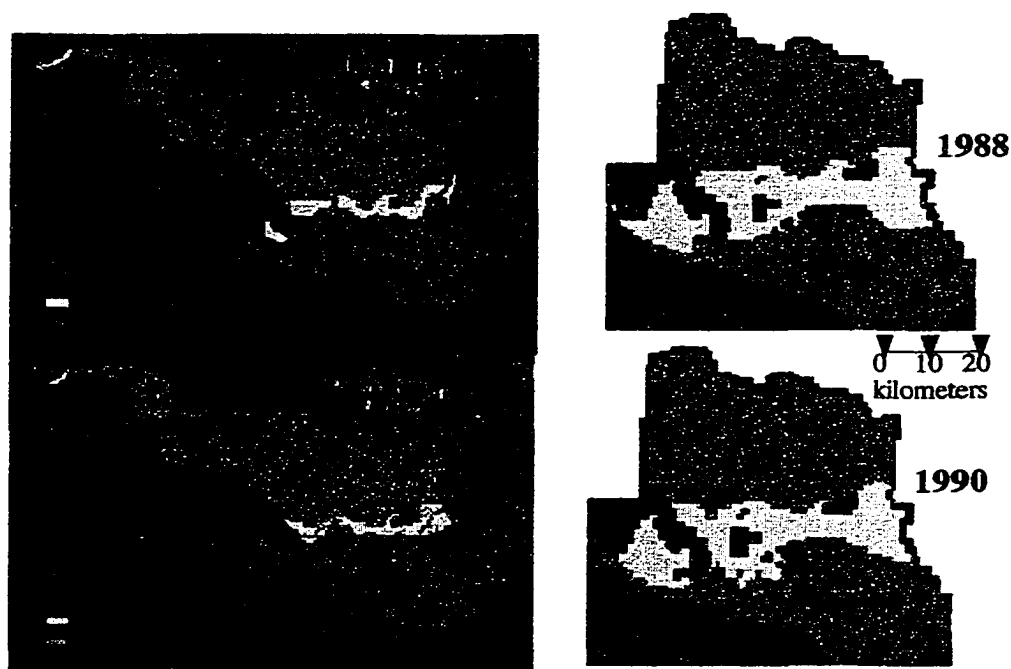


Figure 5.3 a and b: Vegetative contours mapped by Chabreck for (a) 1988 and (b) 1990 and the associated 1 km² digitized map generated from the scanned map. Note that the intermediate and brackish marsh categories in the Chabreck map have been combined to one marsh category in the digitized map.

To answer question 1A, maps containing only the common area (Figure 5.4) of the USFWS 1988 habitat map, described previously, and the Chabreck 1988 habitat map were used. The Chabreck study was concerned with mapping habitat zones, and thus the resultant mapping was done on existing USGS quad sheets. This means that the land/water ratio is inaccurate because no land loss was mapped. In our fit comparisons for this analysis the open water in the interior of both study areas was reclassified to the marsh type that would be present based on the salinity of the area. This was done to make the habitat boundaries the primary criteria used for fit. Comparing USFWS 1988 with Chabreck 1988 gives the fit of $F_t(k) = 91.28$.

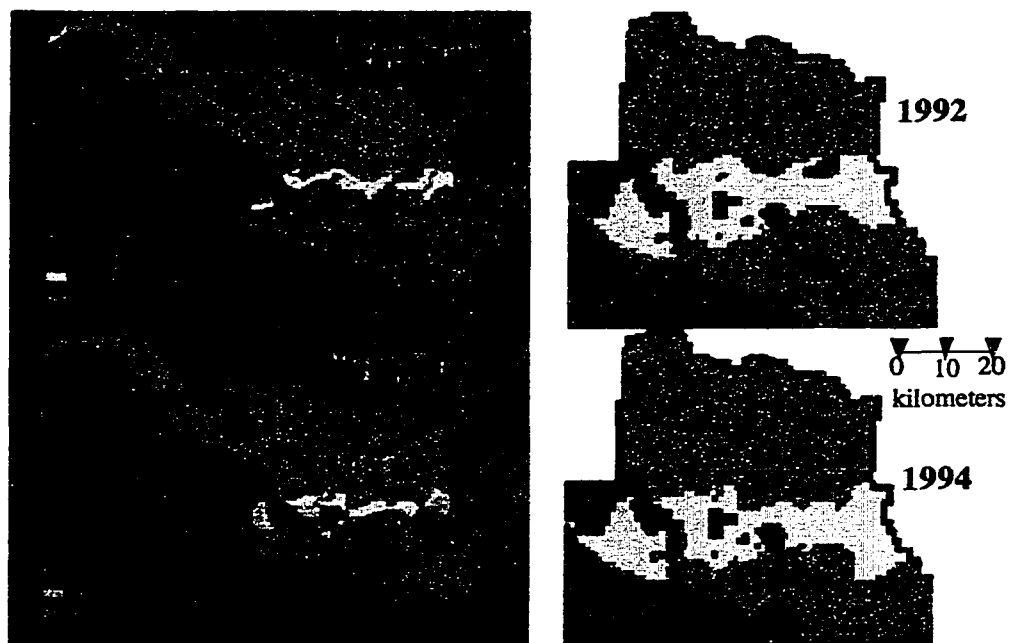


Figure 5.3 c and d: Vegetative contours mapped by Chabreck for (c) 1992 and (d) 1994 and the associated 1 km² digitized map generated from the scanned map. Note that the intermediate and brackish marsh categories in the Chabreck map have been combined to one marsh category in the digitized map.

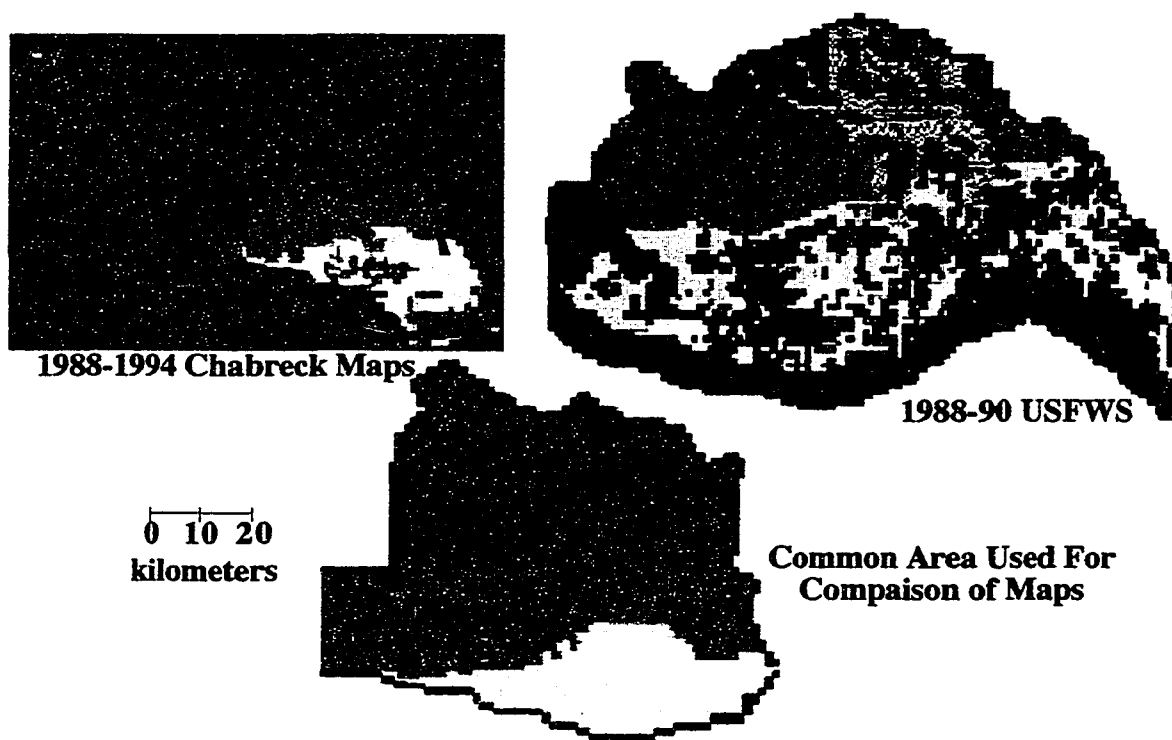


Figure 5.4 Area in common to the Chabreck study area and the Terrebonne basin portion of the BTNEP study area.

In addition to the pair of 1988 habitat maps, we can use the 1994 Chabreck map and the 1994 NRSC maps (Chapter 4) to calculate another comparison fit. Using only the overlapping portions of the two 1994 study areas and removing the uncertainty due to the mapping of water bodies (as described above) the $F_t(k) = 87.70$. A summary of this analysis is contained in Figure 5.5. Calculation of $F_t(k)$ for the same area for the same year using base maps constructed by different methodology yields results of $F_t(k) = 91.28$ and $F_t(k) = 87.70$. These differences cannot be attributed to the patchiness of landscape fragmentation due to land loss. As described above, the water habitat was taken out of this analysis. These differences are due to the mis-alignment of the contours. If data collection and mapping were perfect, we would expect the fit for these maps to be 100. Some sources of error could be in the classification of the remotely sensed data, the large distance between transects of vegetative sampling, or the methods used to contour the data. As stated earlier, the 1994 NRCS habitat map contains the most sampling points as well as the most defensible contouring.

Variability due to Scale - Question 1B

If the base maps are compared at finer and coarser resolution, we might expect the $F_t(k)$ to differ but hopefully the difference would be small enough to be negligible. This is in fact the case. Using the two sets of maps described above, the data were aggregated by majority into 2 km² cell resolution maps and also split into 0.5 km² resolution maps. The results of the fit analysis are presented in Table 5.1. It should be noted that this analysis was performed on maps with very little spatial fragmentation, and the results may not be able to be applied to fragmented landscaping (Marceau et al., 1994; Moody et al., 1994).

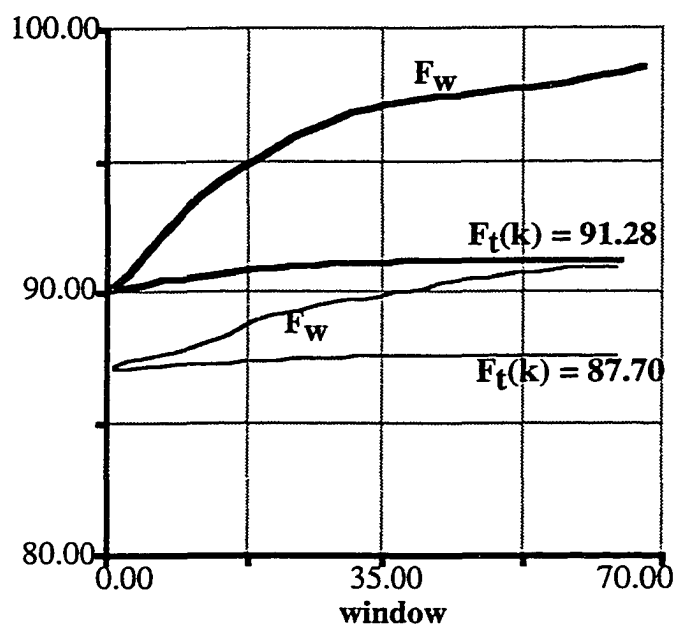


Figure 5.5 Individual window fit F_w and multiple resolution goodness of fit index $F_t(k)$ for analysis comparing mapping methodology. Bold lines are the results of comparing the 1988 USFWS map with the 1988 Chabreck map. Lighter lines are the results of comparing the 1994 NRCS map with the 1994 Chabreck map.

Table 5.1 F_t calculation at various scales for maps of the same region using different mapping methodologies

USFWS vs. Chabreck 1988	
scale	$F_t(k)$
2 km ²	90.77
1 km ²	91.28
.5 km ²	90.67
NRCS vs. Chabreck 1994	
scale	$F_t(k)$
2 km ²	88.10
1 km ²	87.70
.5 km ²	87.43

Temporal Variability - Question 2

To continue this analysis, question 2 poses the problem “how much might the fit parameter be expected to change year to year in an evolving landscape?” Because it has been shown that base maps collected by different methodology and resolution produced differences in the $F_t(k)$, this portion of the analysis will only compare maps that have

been collected using the same methodology. Table 5.2 contains the results of the fit calculation comparing sets of data that have temporal separation. The habitat can remain as similar as $F_t(k) = 97.10$ (1990 vs. 1992) or change by as much as $F_t(k) = 92.00$ (1992 vs. 1994) in 2 years. The data in Table 5.2 are summarized in Figure 5.6.

Table 5.2 Results of fit calculations for various data sets collected by the same methodology.

data set	# years	mid year	$F_t(k)$ $k = .1$
Chabreck 88 vs. 90	2	1989	96.67
Chabreck 90 vs. 92	2	1991	97.10
Chabreck 92 vs. 94	2	1993	92.00
Chabreck 88 vs. 92	4	1990	96.57
Chabreck 88 vs. 94	6	1992	91.67
Chabreck 90 vs. 94	4	1992	91.36
USFWS 56 vs. 78 *	22	1967	82.11
USFWS 78 vs. 83 *	5	1980.5	84.26
USFWS 56 vs. 83 *	28	1970	76.87
USFWS 56 vs. 78 **	22	1967	67.61
USFWS 78 vs. 88 **	10	1984	85.59
USFWS 56 vs. 88 **	32	1973	60.79
DF_t/Dt			$0.95 F_t / \text{year}$
theoretical maximum F_t			96.23

* southwestern portion of the basin only

** total basin from Highway 90 south

No direct cause and effect relationship can be inferred from this analysis. Although every attempt was made to include as many data sets as possible, this analysis, as well as the two preceding, suffer from the lack of data. It is possible that habitat change is occurring at a constant rate, and the top graph of Figure 5.6 represents that rate of change ($0.95 F_t(k)$ per year). In this case, $F_t(k) = 96.23$ (y intercept) is the best fit one could expect, based on data collected with the same methodology. It is also possible that the rate of change of the habitat is not constant, but rather is driven by an unsteady environmental pulsing. If this were the case, the relationship in the bottom graph of

Figure 5.6 would be expected to be non-linear. The data presented in the bottom graph have been fitted to a straight line and the r^2 is .68.

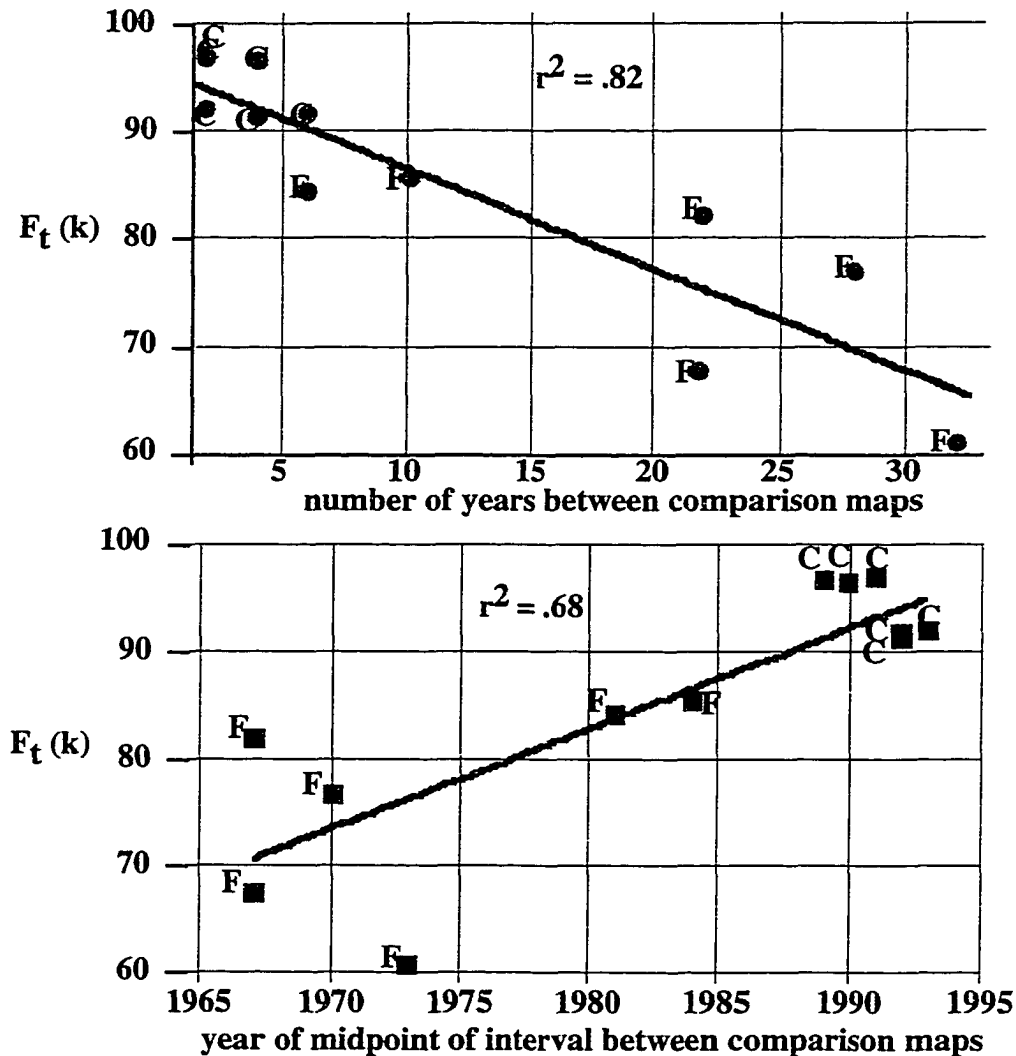


Figure 5.6 The fit parameter $F_t(k)$ plotted against the number of years in the comparison map (top) and the midpoint year of the interval between comparison maps (bottom). C indicates map comparisons based on Chabrecks maps, F indicates USFWL maps.

To summarize this analysis, there are three possible values for a practical upper limit on the $F_t(k)$ for landscape predictions in the wetlands of southern Louisiana. Comparing data from the same location and year, which were collected and mapped by different methodologies, yields limits of 91.28 and 87.70. Extending the slope of the

line of $\Delta F_t(k)/\Delta t$ to the origin yields a limit of 96.23. This may be the more reliable number since it captures data from the most data sources. The $F_t(k)$ for a landscape change by .95/year and the average difference due to scale aggregation from 0.5 to 2.0 km² is 0.43.

Investigation of Weighting Factor - Question 3

The factor used to weight the individual F_w in this analysis thus far is the exponential function $e^{-k(w-1)}$. As stated previously, this function weights most heavily for a window size of 1, regardless the value of k chosen. The results of the analysis on the previous maps leads one to question whether this window size should be so heavily weighted when the method of mapping is different for the two maps that are being compared.

An alternate weighting function that retains the decreasing weight at large window size, but allows for a broader range of heavily weighted window sizes is the Gaussian distribution. In particular this distribution is attractive because the shape of the curve can be modified based on window and study area parameters rather than the coefficient, k , which cannot be immediately related to a physical parameter in the modeling setup. Figure 5.7 illustrates the comparison of the weighting function $e^{-k(w-1)}$ and the alternative Gaussian function

$$e^{-1/2\left[\frac{(w-\mu)}{\sigma}\right]^2} \quad \text{equation 5.3}$$

where μ = center of the curve (window size of maximum weight) and σ = sigma, the standard deviation (width of window sizes to be considered in summation). Figure 5.8 compares the original exponent with the results of adjusting σ by various amounts for a window size of maximum weight of 3x3 cells.

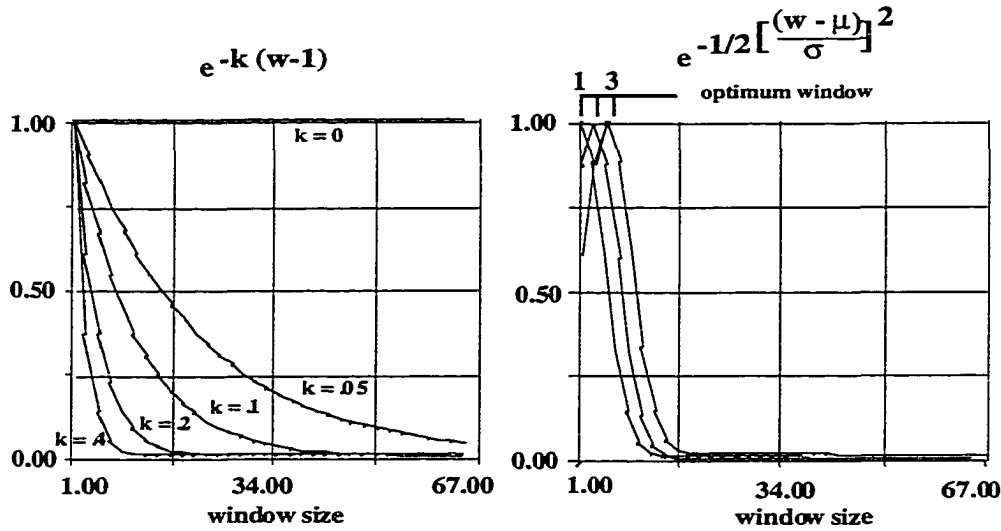


Figure 5.7 The weighting function on the left is the exponential utilized in the CELSS and BTNEP modeling. The factor $k = 0$ weights all window sizes equally and increasing k reduces the effects large windows will have on the calculation of $F_t(k)$. A window size of 1×1 will always be given the highest weight. The gaussian function shown on the right allows the optimum window size to be adjusted.

The multiple resolution goodness of fit parameter for this weighting factor is

$$F_t(\mu, \sigma) = \frac{\sum_{w=1}^n F_w e^{-1/2[(w-\mu)/\sigma]^2}}{\sum_{w=1}^n e^{-1/2[(w-\mu)/\sigma]^2}} \quad \text{equation 5.4}$$

This new function is only a better choice if there is some way to determine the appropriate window size (μ) and spread (σ) for the weighting function. To determine these, the Chabreck vs. USFWS 1988 maps and the Chabreck vs. NRCS 1994 maps were used. The assumption is that the actual landscape for the same year should be identical regardless of the mapping methodology. As stated before, any differences in the comparison of the maps from year are the result of sampling or contouring error.

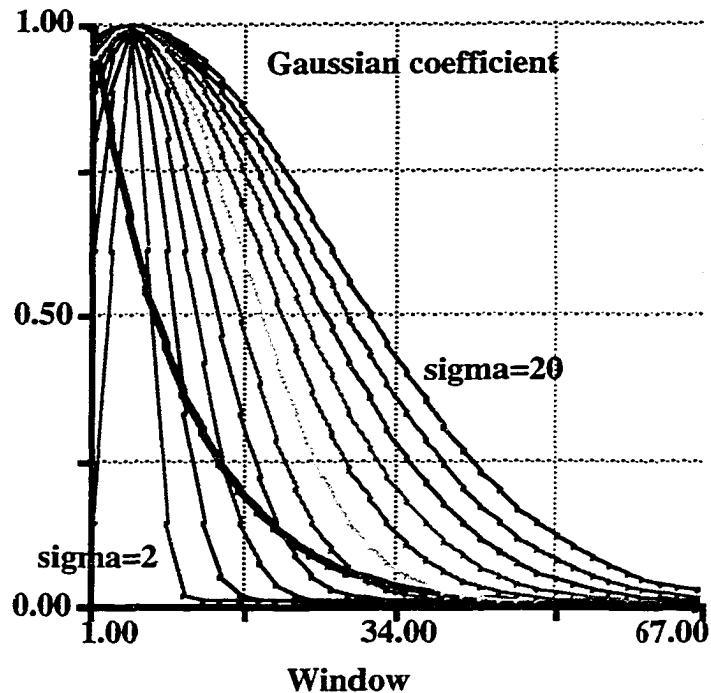


Figure 5.8 Comparison of exponential fit coefficient (bold line) and Gaussian fit coefficient with sigma varying from 2 to 20. Fit analysis performed on CELSS and LSU BTNEP habitat model results utilized the exponential coefficient with $k=1$.

If it is necessary to use these maps, the amount of error may be accounted for by adjusting μ and σ in weighting function. Fit calculations, $F_t(\mu, \sigma)$ were made on both sets of maps varying the optimum window size (μ) from 1x1 to 10x10 and the spread of the curve (σ) from .5 to 10.0. The highest value of $F_t(\mu, \sigma)$ is not necessarily the best. Selecting an optimal window size of 10x10 and allowing the spread of the weighting curve to be 10.0 produces the largest value of $F_t(\mu, \sigma)$ but is an unsatisfactorily vague weighting because we are hoping to be able to discern habitat responses on a smaller scale than 100 km². To determine the value at which the greatest rate of change in $F_t(\mu, \sigma)$ is taking place, the first derivative with respect to window size and sigma were calculated. The fit calculation, when weighted with a Gaussian distribution and applied to the 1988 and 1994 maps showed the most sensitivity at μ of 2 (i.e. a 2x2 or 4km²

window) and $\sigma=2.0$. The fit calculations using these parameters with a Gaussian weighting factor for the maps collected by the same methodology are shown in Table 5.3

Table 5.3 Results of fit calculations for various data sets collected by the same methodology. Values for $F_t(k)$ are the same as in Table 5.2 and are presented again here for ease of comparison.

data set	# years	mid year	$F_t(k)$	$F_t(\mu, \sigma)$
Chabreck 88 vs. 90	2	1989	96.67	96.30
Chabreck 90 vs. 92	2	1991	97.10	96.93
Chabreck 92 vs. 94	2	1993	92.00	91.74
Chabreck 88 vs. 92	4	1990	96.57	96.24
Chabreck 88 vs. 94	6	1992	91.67	91.34
Chabreck 90 vs. 94	4	1992	91.36	90.94
USFWS 56 vs. 78 *	22	1967	82.11	80.69
USFWS 78 vs. 83 *	5	1980.5	84.26	82.43
USFWS 56 vs. 83 *	28	1970	76.87	75.16
USFWS 56 vs. 78 **	22	1967	67.61	64.20
USFWS 78 vs. 88 **	10	1984	85.59	81.44
USFWS 56 vs. 88 **	32	1973	60.79	56.15
DF_t/Dt			0.95 Ft /year	0.90 Ft /year
theoretical maximum F_t			96.23	95.82

* southwestern portion of the basin only

** total basin from Highway 90 south

The upper limit on the $F_t(\mu, \sigma)$ for landscape predictions comparing two maps made in the same year is 88.71 ± 1.49 ($n = 2$). The theoretical upper limit based on the intercept of the line of $\Delta F_t(\mu, \sigma) / \Delta t$ is 95.82. The $F_t(\mu, \sigma)$ for a landscape would be expected to change by 0.90/year and the average difference due to scale aggregation is 0.24. The results for $F_t(k)$ and $F_t(\mu, \sigma)$ are summarized in Table 5.4.

A number of recommendations can be made from this analysis. While the theoretical upper limit on the multiple resolution goodness of fit is 100, in reality it is only 91 - 92 when applied to the processes and habitat maps of southern Louisiana that are available today.

Table 5.4 Comparison of results of analysis using the exponential vs. Gaussian weighting function

Analysis	$F_t(k)$ $k=.1$	$F_t(\mu,\sigma)$ $\mu=2,\sigma=2$
Upper limit on fit parameter		
Chabreck 1988 vs. USFWS 1988	87.70	87.22
Chabreck 1994 Vs. NRCS 1994	91.28	90.20
x intercept	96.23	95.82
average	91.74	91.08
average difference due to scale	.43	.24
expected change per year	.95	.90
suggested minimum years simulation	8.7	9.9

This upper limit gives rise to a suggested minimum simulation run based on these limitations:

$$\text{Minimum simulation} = \frac{(100. - \text{average upper limit})}{\text{expected change per year}} \quad \text{equation 5.5}$$

As the accuracy of habitat mapping changes, this minimum simulation length should change.

The choice of whether to use $F_t(k)$ or $F_t(\mu,\sigma)$ can now be made keeping the benefits and limitations of each in mind. The choice of $F_t(k)$ allows a slightly higher average upper limit, a larger expected rate of change per year and consequently a shorter minimum simulation. The choice of $F_t(\mu,\sigma)$ allows the user to choose an optimum window and spread for the analysis and reduces the difference due to aggregation, however, it requires a longer minimum simulation run. An additional advantage of the $F_t(\mu,\sigma)$ fit parameter is that it can be used to evaluate fit in cases where the resolution of one of the maps does not match the other.

CHAPTER 6. PRODUCTIVITY PARAMETERIZATION AND HABITAT SWITCHING

Introduction

Thus far this study has reviewed and evaluated a number of methods to measure the comparison of a habitat map to a reference map (objective 1 in dissertation objectives). This index has been used as a measure of success for a number of spatial landscape models, including the BTNEP landscape simulations. As will be shown, in the BTNEP landscape model the productivity unit model that produced the best overall spatial fit for the landscape over predicted the primary production of the plants at a 1 m² scale. This does not imply that the *final* results of the landscape model are in error. The landscape model predicts habitats, and those habitats are decided by the habitat unit module. If that module required an unrealistically high value of biomass in order to keep from becoming open water, the results might well be correct due to one module compensating for another. Realistic predictions of biomass production and habitat succession, dissertation objectives 2 and 3, will be investigated and tested in the unit models in this chapter. I will attempt to parameterize the primary production module and investigate the behavior of the habitat switching module. In the next chapter these newly developed and tested unit models will be incorporated into the landscape model and evaluated at the landscape scale.

Biomass Productivity Unit Model

Literature review

Because the effects of salinity and waterlogging stresses were empirically included in the primary production module of the BTNEP landscape model, I have chosen to isolate them for further investigation. The second objective of this study is can we accurately predict the seasonal production of marsh vegetation? I suggest that there

will be a range of salinity and flooding data by species that will produce an optimum productivity as well as extreme values of those variables that will produce mortality.

There is a wealth of literature investigating the stresses on wetland vegetation due to waterlogging and salinity (Ewing, 1997; Flowers et al., 1986; Josselyn et al., 1990; Latham et al., 1991; Nixon, 1980; Turner, 1976). As would be expected, fresh species communities experience the most stress from salinity (Feijtel et al., 1989; Latham et al., 1991; McKee et al., 1989; Mitsch et al., 1993). However, recent evidence indicates that pulses of high salinity waters can be tolerated by fresh species that are not simultaneously experiencing other stresses (Grace et al., 1996; Howard et al., 1993).

Even hydrophytic vegetation can experience stress from extreme waterlogging (Burdick et al., 1990; Mendelssohn et al., 1981; Naidoo et al., 1992; Wilsey et al., 1992). Recent research suggests it is not lack of nutrient availability, but toxicity that produces the stress (Koch et al., 1989; Koch et al., 1990; Mendelssohn et al., 1988; Portnoy et al., 1997). In addition, waterlogging affects roots more than aboveground biomass and thus leaves the plant vulnerable to drought (Kozlowski, 1984). Evidence indicates that multiple stresses have a synergistic negative effect on photosynthetic production (Burdick et al., 1989; Howard et al., 1993). The literature review for the pertinent values are presented in Appendix C.

Biomass unit model

The macrophyte unit model for the BTNEP landscape model was originally developed in STELLA™, a simulation language that facilitates model development and modification. The model runs on daily time steps and the forcing functions are the hydrodynamically controlled features such as duration of flooding and salinity, that were generated by the hydrodynamic module of the landscape model, as well as time series of temperature. The seasonal tendency of plant production to peak during the summer and senesce during winter, the maximum and minimum values of primary production and the ratio of below to above ground biomass are all used to evaluate the success of the

simulation. A satisfactory unit model will be one that reproduces the seasonal above and below ground productivity values for a number of years utilizing observed forcing functions. Primary production is generally reported in the literature as grams of biomass (or carbon) per square meter per time, so the processes in the unit module are scaled to 1 m². This seems to be a valid approach, as Lechowicz reports that for forest plants, predicting multiple processes at distances greater than 2 meters, the individual processes are negligible (Lechowicz et al., 1991).

Equations of state for the primary production unit model are presented in Table 6.1 and are represented diagrammatically in Figure 6.1. Macrophytes are modeled using two state variables: above ground photosynthetic carbon biomass, $B(t)$, which aggregates leaves and photosynthetic herbaceous stems and below ground non-photosynthetic carbon biomass, $G(t)$, that aggregates roots and rhizomes. The above ground biomass gains mass by photosynthesis (Nielsen et al., 1996). The net production is a function of its biomass, the species specific maximum gross production rate and a limitation function (Hopkinson et al., 1988; Mitsch, 1988; Phipps, 1979). This limiting function includes empirical responses to flooding, salinity and temperature via a coefficient ranging from 0.0 to 1.0. This coefficient will reduce the maximum specific production rate depending on the synergistic effect of the total environmental conditions. Water temperature is estimated as a linear function of air temperature. Salinity stress is determined by plant tolerances depending on their habitat (Howes et al., 1986; Pezeshki et al., 1987a). The rate of growth is further constrained by a water logging function, based on duration of flooding, to represent the different tolerances to flooding conditions. To estimate the effects of metabolic stresses on vegetation, respiration rates are increased as a function of increases in stress factors (Cronk et al., 1994; Dai et al., 1996; Howes et al., 1986; Mitsch et al., 1982; Nyman et al., 1991b; Pomeroy et al., 1976).

Table 6.1 Equations of state for the primary production unit model

Formula/Symbol	Process	Value of coefficient	Reference
<u>Above ground macrophyte production</u>			
$B(t) = B(t - dt) + dB$	Photosynthetic activity.		(1)
$dB/dt = P - (T+H+D+ Rb)$			
$P = (\mu B) * (N*S*L*(C/C_{max}))$			(1)
Where:			
B	Above ground biomass in gOM		
$T = \phi * P$	Biomass translocated to below ground biomass.		
ϕ	translocation rate from above ground biomass (B) proportional to photosynthetic activity (P).	$k_{trans} = .6$	(14)
$H = k * B$	Herbivory consumption in gOM.		
$k = 0.00$	Herbivory consumption rate in gOM/d.		(3)
$D = l * B$	biomass lost as detritus in gOM.		
l	detritus production rate in gOM/d, habitat dependent	$l_{fresh} = 0.00619$ $l_{brack} = 0.00619$ $l_{salt} = 0.00414$	(4; 5)
$Rb = r * B$	Above ground respiration losses in gOM		
r	respiration rates in gOM, habitat dependent	$r_{fresh} = 0.00619$ $r_{brack} = 0.00619$ $r_{salt} = 0.00414$	(6; 7; 8; 9)
μ	maximum gross production rate, habitat determined in gOM/d	$\mu_{fresh} = 60$ $\mu_{brack} = 60$ $\mu_{salt} = 43$ $\mu_{swamp} = 22.1$	(6)
N	Minimum nutrient accumulation rate (kg /m ² d), habitat dependent	$N_{fresh} = 7.67e-4$ $N_{brack} = 1.31e-3$ $N_{salt} = 4.77e-3$	(10)
S	Optimal salinity range in ppt, habitat dependent	$S_{fresh} = 0.0$ to 3.0 $S_{brack} = 4.50$ to 9.0 $S_{salt} = 10.0$ to 35.0	(4; 11)

(Table 6.1 continued)

Formula/Symbol	Process	Value of coefficient	Reference
L	water level stress tolerance in hours, habitat dependent flooding tolerance for brackish and saline marshes flooding tolerance for swamp and fresh = 0.00 to 24.00 hours		
C	air temperature in centigrade		
Cmax	mean maximum air temperature for a 30 yr. record in centigrade		(12)
<u>Below ground macrophyte production</u>			
$G(t) = G(t - dt) + dG$			
$dG/dt = T - (M + Rg)$			
Where:			
G	Below ground biomass determined by habitat in gOM.		(2; 13; 16)
$M = h * G$	Below ground mortality in gOM.		
h	Below ground mortality rate in gOM per day, habitat dependent	hfresh = 0.00619 hbrack = 0.00619 hsalt = 0.00414 hswamp = 0.000475	
$Rg = s * G$	Below ground respiration losses in gOM.		
s	Below ground respiration rate in gOM per day, habitat depend	sfresh = 0.00619 sbrack = 0.00619 ssalt = 0.00414 sswamp = 0.000475	(15)
$T = \phi * P$	Biomass translocated to below ground biomass.		
ϕ	translocation rate from above ground biomass (B) proportional to photosynthetic activity (P).		
Sources:			
(1) (Conner & Day, 1976)	(2) (Childers & Day, 1990)		
(3) (Mann, 1982)	(4) (Turner, 1976)		
(5) (Reimold, 1972)	(6) (Dai & Wiegert, 1996)		
(7) (Mitsch & Reeder, 1991)	(8) (Blum, Seneca & Stroud, 1978)		
(9) (Hopkinson, Day & Gael, 1978)	(10) (Nyman & DeLaune, 1991)		
(11) (Pezeshki et al., 1987)	(12) (Morris, Houghton & Botkin, 1984)		
(13) (Marinucci, 1982)	(14) (Howes et al., 1985)		
(15) (Gleason & Dunn, 1982)	(16) (Kirby & Gosselink, 1976)		

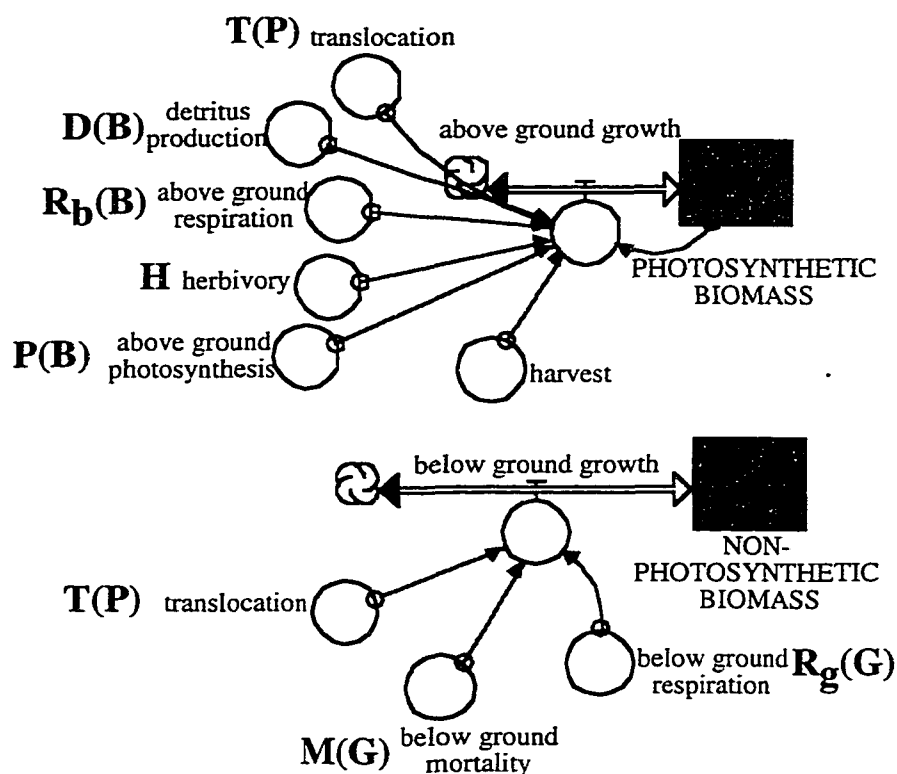


Figure 6.1 Diagrammatic representation of the primary production unit model. Functional relationships are contained in Table 6.1. See text for details.

Carbon that is fixed in excess of leaf growth requirements is translocated to the non-photosynthetic storage (Gosselink et al., 1974; Howes et al., 1985; Howes et al., 1986). If leaf growth requires more carbon, it is translocated from the non-photosynthetic reserve carbohydrate pool. By separating the two macrophyte components, annual losses as litterfall do not decrease the non-photosynthetic biomass values that are used to calculate root depth. This allows for a computation of the organic component of soil that is used by the soil building sector of the landscape model. Separate flows for respiration and mortality exist for photosynthetic biomass and non-photosynthetic biomass (Pomeroy et al., 1976).

The module was calibrated for the three representative species occurring in the three marsh habitat types of Terrebonne basin (Figure 6.2). Literature values were used for some of the parameters (Table 6.1) and sensitivity analysis was used to determine the limitation and waterlogging functions described above.

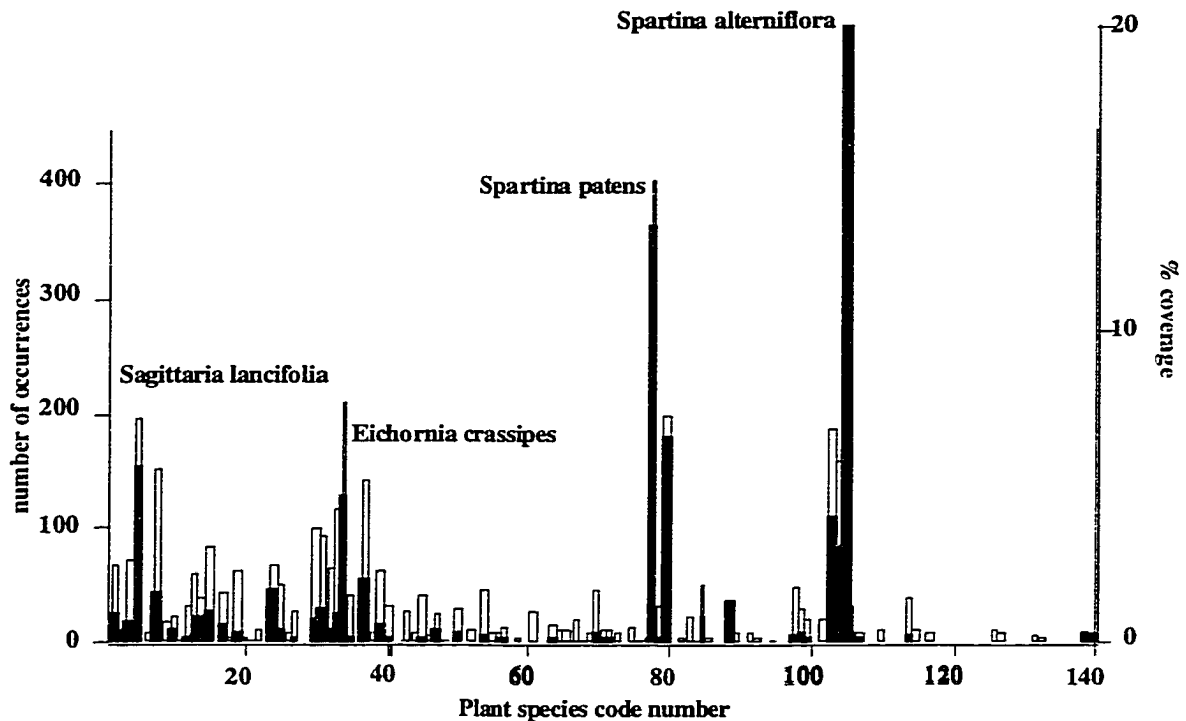


Figure 6.2. Frequency of occurrence (left scale, hollow bars) and total percent coverage (right scale, solid bar) for plant species observed by NRSC personnel during soil survey conducted May 5 - June 16, 1994. Details of data collection are presented in Chapter 4 and plant species list and code numbers are presented in Appendix A. The dominant species observed in each habitat type are *S. lancifolia* (fresh), *S. patens* (brackish) and *S. alterniflora* (saline).

In all three marsh species, the productivity values that produced the best overall spatial fit for the landscape over-predicted the primary production of the plants at a 1 m² scale (Callaway et al., 1992; Dai et al., 1994; Dai et al., 1996; Howard et al., 1995; Kirby et al., 1976; Kludze et al., 1994; Pezeshki et al., 1991). In the review of literature in Appendix C, the highest density of *S. alterniflora* was 4.50 kg/m²; the highest density of *S. lancifolia* was 3.6 kg/m² and the highest density of *S. patens* was 2.8 kg/m². The correction used for this problem was to “harvest” the plant when the values of biomass reached the unrealistically high value of 10 kg/m². Sample output of primary production for landscape simulations at Oyster Bayou (brackish marsh), Turtle Bayou (fresh marsh) and Cocodrie (saline marsh) are shown in Figure 6.3. Although relative values are reasonable, in general, primary production is seriously over-predicted by the unit module as it was parameterized for the B-TNEP landscape model.

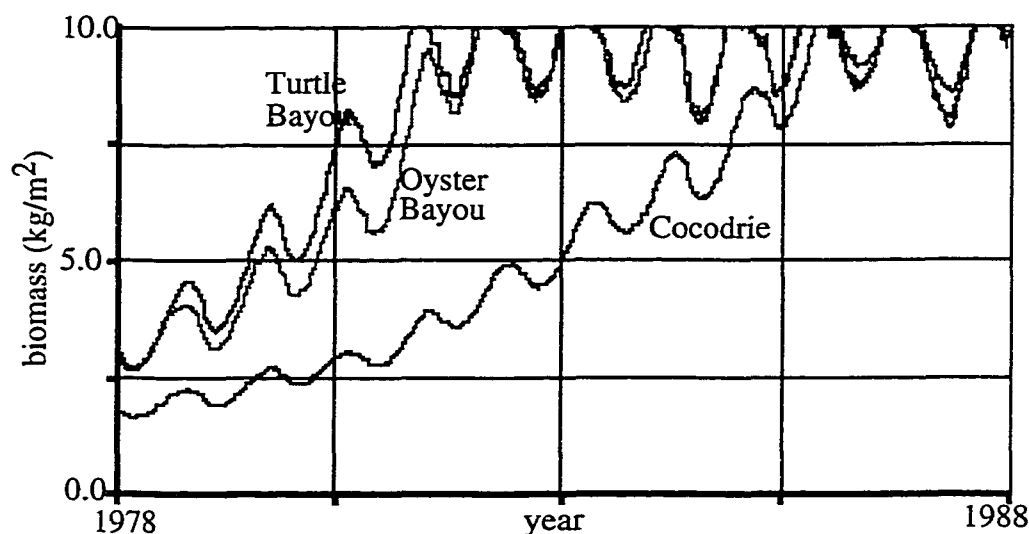


Figure 6.3 Predicted above ground biomass at three stations in the Terrebonne basin using the BTNEP Landscape model. Turtle Bayou results simulate fresh marsh, Oyster Bayou results simulate brackish marsh and Cocodrie results simulate saline marsh. Biomass production is constrained so that it cannot exceed 10.0 kg/m^2 .

If biomass is being over-predicted, it could be due to a primary production rate that is too high, respiration rates too low, or lack of sufficient photosynthetic limitation due to stresses. It is my assertion that the parameters most likely in need of changing are the biomass production response to the stresses of salinity and waterlogging. To provide guidance in this investigation, a literature review was conducted and the data collected from a number of studies is summarized in Appendix C. Blum (1978) reports that ecosystem respiration amounts to 71% of gross annual photosynthesis. Dai (1994) reports the net highest rate of new growth was about $15 \text{ g/m}^2 \text{ day}$ in *S. alterniflora*. The coefficient for photosynthetic activity for *S. alterniflora* is $43 \text{ g/m}^2 \text{ day}$ (Table 6.1), well within the range of net primary productivity reported by Dai ($43 \text{ g/m}^2 \text{ day} \times (100\% - 71\%) = 12.5 \text{ g/m}^2 \text{ day}$). For this reason I turn my attention to the effects of the stresses of waterlogging and salinity on the biomass production.

Hours of flooding can be used as a surrogate for redox or flooding stress. Photosynthetic activity would be expected to decrease as soon as duration of flood lasted long enough to reduce the oxidation-reduction potential, Eh to below 300 millivolts (mv)

since soil phytotoxins begin to accumulate when Eh decreases to +220 mv and sulfide production begins when Eh reaches -150 mv. The water depth is not a good surrogate for duration of flooding and the relationship between Eh and hours of flooding of the soil was not experimentally addressed in any of the literature that I found. A number of research products do allude to the relationship. Howes states that variation in plant biomass was closely correlated with sediment Eh and accounted for 62% of the variation in plant biomass, however, water table depth at low tide was not as well correlated (Howes et al., 1986). In the laboratory portion of the same study, Eh increased immediately as a result of draw down of water (on sediments obtained from non-creek bank portions of a New England salt marsh) and Eh decrease lagged flooding by only a few hours. The data derived from the literature review in Appendix C show Eh and water depth are slightly inversely correlated (-.43). Arenovshi (1992) reported that redox reached its stable value within days of initial flooding. Eh in brackish marsh vegetation is higher than in fresh or saline marsh vegetation when experiments were done holding soil type and flooding regime constant (Nyman et al., 1991a).

The water elevation, duration of flooding, and salinity that were predicted by the hydrologic component of the BTNEP landscape model for the simulation shown in Figure 6.3 (the forcing functions for the biomass predictions) were copied into a STELLA™ unit model. The exercise of calibrating the unit model was then repeated. In the calibration phase of the unit model development, the production limitation coefficients were then manipulated until a realistic value for above ground and below ground biomass was obtained for each marsh type. The results of the simulation are shown in Figure 6.4 and the productivity limitation coefficients, that produced the simulation, are shown in Table 6.2 and Figure 6.5

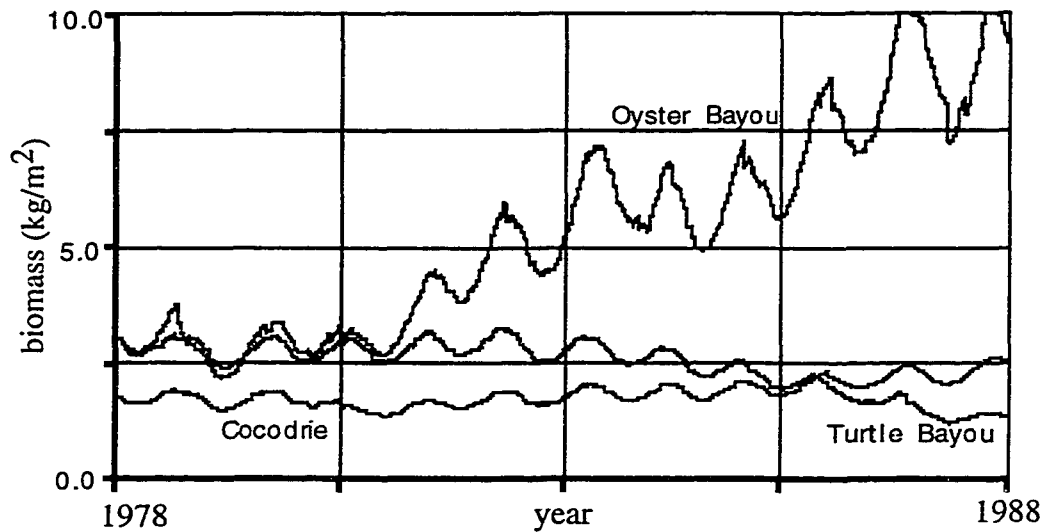
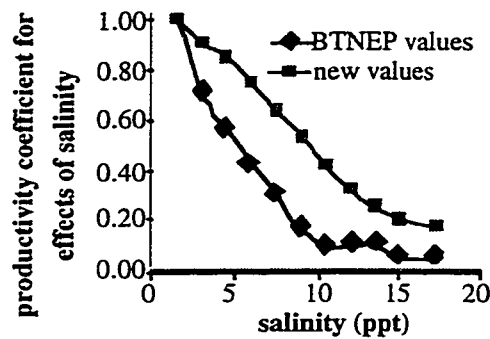


Figure 6.4 Predicted above ground biomass using the BTNEP productivity module with new parameterization for the effects of water logging and salinity stress. Turtle Bayou results simulate fresh marsh, Oyster Bayou results simulate brackish marsh and Cocodrie results simulate saline marsh. Biomass production is constrained so that it cannot exceed 10.0 kg/m^2 .

Table 6.2 Comparison of parameterization for productivity model
Coefficients that have been changed are listed. See Table 4.1 for equations and coefficient definitions.

original unit model	modified unit model	
ktrans(fresh) = .6	ktrans(fresh) = .707	translocation rate from above ground biomass (B) proportional to photosynthetic activity (P).
ktrans(brackish) = .6	ktrans(brackish) = .6	
ktrans(saline) = .6	ktrans(saline) = .615	
sfresh = 0.00619	sfresh = 0.020	Below ground respiration rate in gOM per day, habitat determined
sbrack = 0.00619	sbrack = 0.022	
ssalt = 0.00414	ssalt = 0.022	
μ fresh = 60	μ fresh = 55	maximum gross production rate, habitat determined in gOM/d
lfresh = 0.00619	lfresh = 0.0051	
Sfresh = 0.0 to 3.0	see graph in Figure 6.5	
f	see graph in Figure 6.5	detritus production rate in gOM/d, habitat determined
		Optimal salinity range in ppt, habitat determined.
		flooding tolerance for marshes

Effects of salinity on fresh marsh production



Effects of flooding duration on marsh production

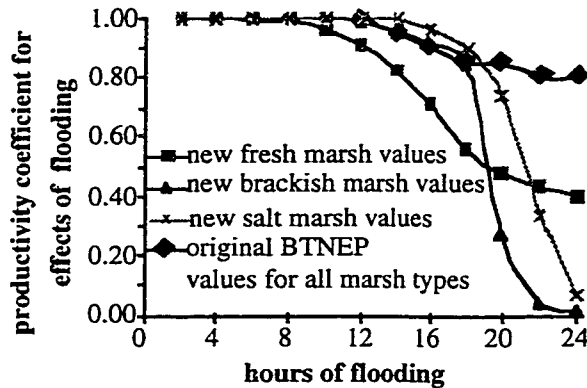


Figure 6.5 Limitation coefficients for the effects of salinity and flooding for the primary production module of the BTNEP Landscape model.

In the previous parameterization, (Figure 6.5) all marsh types exhibited the same response to flooding. In the new parameterization, the three marsh types respond differently, and all are more sensitive to flooding. In addition, fresh marsh is less sensitive to salinity. Biomass production for brackish marsh is still over-predicted. This simulation uses the values predicated by the hydrodynamic model from the Oyster Bayou site. This is arguably the healthiest part of the whole Terrebonne basin as it experiences regular tidal flushing and is flooded less than most of the rest of the basin (Morris et al., 1984b) a factor accounted for in the unit model. Biomass predictions for fresh and saline marsh, however, are very stable and do not show the tendency to gradually increase as is seen in Figure 6.3. Detritus production for each of the species was in the range expected as reported in the literature, (Hopkinson et al., 1978; Kirby et al., 1976). The next step in model development is to validate the model with an independent data set.

Validation - Caernarvon Data

To validate the newly parameterized unit model it should be applied to an area where detailed water elevation, salinity and vegetation data exists. The vegetation data sets for the Terrebonne marsh are insufficient for this, because no long-term simultaneous salinity and water elevation data exists.

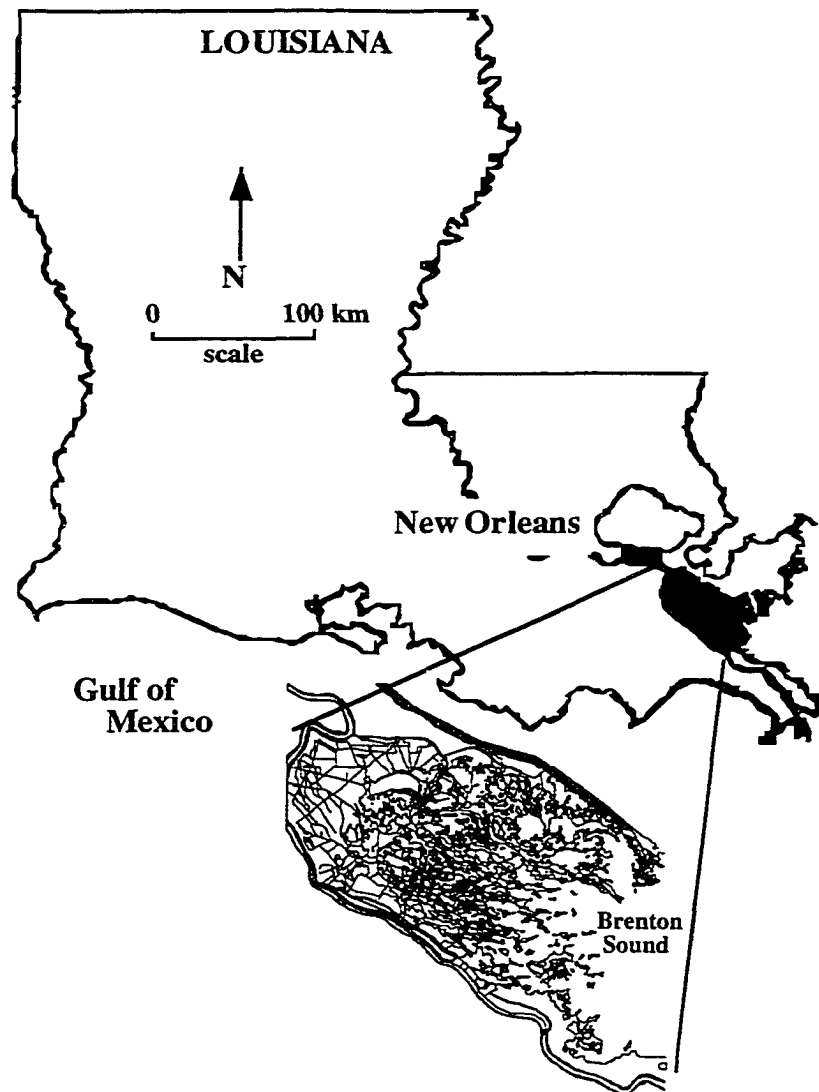


Figure 6.6 Caernarvon fresh water diversion structure and receiving basin.

A nearby study area does exist with sufficient monitoring to validate the module. The Caernarvon fresh water diversion project, located on the east bank of the Mississippi River (Figure 4.6) was completed in February 1991. The purpose of the water diversion

project is to freshen the water, and consequently the vegetation of the receiving basin. Prior to its construction and in the subsequent years of operation, habitat monitoring was conducted by the Louisiana Department of Natural Resources (LDNR). The water diversion project is designed to discharge up to 8000 cfs into Breton Sound according to an operational plan implemented by the Corps of Engineers and the LDNR.

Water elevation and salinity were recorded at three sites in the outfall area, Bay Gardene, California Bay and Black Bay. Salinity means of the total data records were Bay Gardene, 7.6 ppt; California Bay, 10.1 ppt; and Black Bay, 9.8 ppt. The Bay Gardene site contained the longest continuous record, 35 months of data from January 1992 through December 1994, with a time step of 1 hour. There was a period of missing salinity data 209 points long that constituted 0.8% of the total record. To reconstruct this missing data, a regression was performed between the salinity in Bay Gardene and Black Bay ($r^2 = .89$) and the missing data was generated as a function of the Black Bay data. Hours of flooding were simulated by taking the mean water elevation for the three year period and incrementing a simulated hours of flooding whenever the water elevation was greater than the mean. Deviations from the mean water level do not necessarily imply hours of flooding. However in this study there were no elevation benchmarks to tie the water level data to and this approximation should be kept in mind in any discussion of possible error.

The data collected at the Caernarvon study area does not contain biomass measurements with which to compare the results of the unit model (this data has recently become available, and should be included in subsequent analysis). However, vegetation transects were conducted annually during the study and results were reported in percent of area covered by type of species. In general, vegetative cover increased at a rate of about 6% per year (Gammill, 1998) and thus a satisfactory model should predict either a steady state or a slight increase in biomass that does not exceed realistic bounds.

The Bay Gardene salinity and water elevation data was used to force the newly parameterized primary production unit model. As is seen in Figure 6.7, the unit module predicts the above ground biomass (and the corresponding below ground biomass) of all three marsh types.

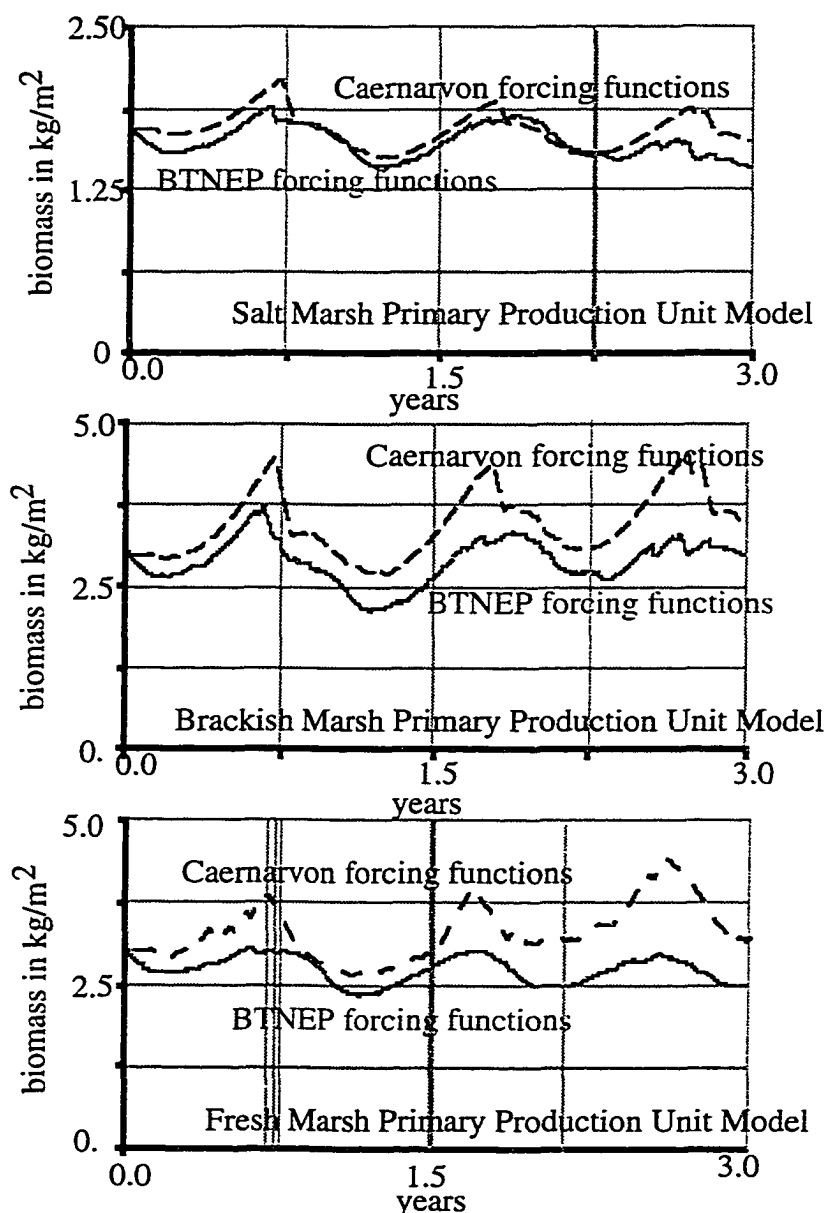


Figure 6.7 Above ground biomass predictions for primary production module with new limitation coefficients for salinity and duration of flooding. Solid line represents predictions made with forcing functions derived from the hydrodynamics of the BTNEP landscape model. Broken line represents predictions made with forcing functions measured in Bay Gardene, Louisiana. Note different scales for the y axis.

As stated earlier, a successful unit model would reproduce the seasonal above and below ground productivity values using observed forcing functions. This is in fact the case. The seasonal pattern of biomass peak in the end of the summer and corresponding die-back in the winter is predicted (Cramer et al., 1981; Kirby et al., 1976). The winter minimum is higher than it should be for all three marsh types, but the summer peak is within acceptable limits. The peak biomass increases 13% for fresh marsh, 2% for brackish marsh and 12% for salt marsh, compared to the reported 6% vegetation increase reported previously for the Caernarvon area.

In order to investigate more fully the response of the unit model, a sensitivity analysis was done on selected parameters (Singh, 1988). Each parameter was exercised independently using the unit model forced with the same time series data that were predicted in the BTNEP landscape model. The value of the parameter being tested was increased and decreased in incremental steps (up to 10% of the optimum value of the parameter) and the percent change in the response of the photosynthetic and nonphotosynthetic biomass was recorded. Table 6.3 shows the average of ten sensitivity runs for each of the parameters listed.

As the unit model is now structured, the biomass production is most sensitive to the translocation rate from above to below ground biomass. In general, non photosynthetic biomass is slightly more sensitive than photosynthetic biomass, to parameter manipulation. All marsh types are more sensitive to flooding parameterization than to changes in salinity parameterization. This would explain why the fresh marsh biomass simulation with the Caernarvon data responds so well. The area is frequently flushed and the salinity response is not as toxic to the simulated vegetation as would be hours of continuous flooding. The unit model does not contain a subsurface component that would allow salt to infiltrate as would happen in reality, and this limitation should be kept in mind if the model is applied to fresh areas that are inundated by salt water. Finally, it is interesting to note that the model is more sensitive to changes in temperature

than to most other parameters. If the unit model is correctly framed, vegetation in southern Louisiana is most susceptible to the forcing functions of temperature, then flooding, and finally salinity changes.

Table 6.3 Sensitivity analysis on parameterization of unit productivity module. Values are expressed as percent change for each parameter and are the average of 10 simulations

parameter	photosynthetic biomass response	non photosynthetic biomass response
photosynthetic respiration rate	-16.552	-16.458
non photosynthetic respiration rate	0.000	0.085
translocation rate	47.274	45.444
photosynthetic production rate	26.626	28.846
temperature	29.689	30.729
salt sensitivity - fresh	6.882	7.094
salt sensitivity - brackish	5.241	6.031
salt sensitivity - saline	7.677	7.722
flooding sensitivity - fresh	41.189	42.357
flooding sensitivity - brackish	10.702	11.499
flooding sensitivity - saline	34.232	35.359

The higher sensitivity of biomass to temperature and hydrology rather than salinity is an important finding. Other ecological modelers have reported the same relationship. Long-term temperature cycles result in significantly lower predictions of forest biomass than observed in the control case for a forest on a biome transition (northern hardwoods/boreal forest (Yeakley et al., 1994). Poiani (1995) found hydrology and temperature to be the most sensitive forcing function in a model of a prairie wetland. In a field investigation, Sasser (1995) found that temperature and the hydrologically related variables of precipitation, evaporation, and water level in floating marshes account for 99% of the variation total aboveground biomass. Van Wambeke (1986) proposes a hypothesis of why temperature may be so influential. The air and soil temperatures are important to the growth of plants but in addition, the temperature is exceedingly important in the rate of chemical processes and, therefore, in the rate of weathering of the primary minerals of the soil. As convincing as this hypothesis might

be, it cannot explain the response of this unit model because temperature is a factor only in the photosynthetic calculations of primary production.

Habitat Succession Unit Model

One of the unique features of the BTNEP landscape model is its capacity to keep track of habitat characteristics for each land parcel throughout time. The program not only recognizes what type of habitat exists in each 1 km² cell but also records a suite of environmental parameters such as salinity, water elevation and productivity. These parameters are then accumulated and evaluated annually to determine if the environmental conditions are characteristic of another type of habitat. This bookkeeping of environmental parameters is used as the basis for the habitat succession algorithm and has been refined from earlier CELSS versions (Costanza et al., 1988; Sklar et al., 1985). As unit models, the productivity model and the habitat succession model can be independent. But when used in a spatial landscape model, they are interdependent, and a change in one requires a reevaluation of the parameters that are used to characterize the other. Recall that objective 3 is “can changes in the mechanistically based habitat evolution more explicitly reflect wetland habitat succession in the unit and landscape models?” In this section I will investigate the habitat succession unit model.

Literature Review

Because plants cannot migrate, they must either adapt to changing conditions or die and make way for those who can. In primary succession, community development accompanies the development of the habitat (Dobson et al., 1997). Studies on the rates of species re-establishment following the last de-glaciation suggest that communities of plants colonize at the rate of 25 to 40 km per century, with a maximum rate of 200 km. per century (Aber, 1992). Although the primary characteristics of habitats are physicochemical, the biological processes are important for the development of a habitat that can support a properly functioning ecosystem. In competition models, a spatial

component can explain the coexistence of numerous plant species (Tilman, 1994). In a non-spatial version of a productivity model, Pacala possible (Pacala et al., 1994) predicted a single species will out-compete all others while the spatial version predicted that coexistence is.

The choice of which parameters to use to characterize the succession of habitats of a plant is problematic due to lack of controlled studies. Temperature is one parameter that can be used, particularly with aquatic organisms. Long term temperature changes of 1.5 degrees have been shown to reduce the zooplankton community by 80% (Roemmich et al., 1995). Chmura (Chmura et al., 1997) reports that temperature (ice formation) can be the factor controlling competition in marsh vegetation. Moisture regime in sagebrush terrestrial community was used to investigate response to global warming (Harte et al., 1995). Salinity, soil organic matter and elevation were the parameters identified by Latham as necessary to characterize a *Scirpus* marsh (Latham et al., 1991). Transition from an aquatic system to a terrestrial system of vegetation was simulated by (Brinson et al., 1995) using water elevation (sea level rise) first and then salinity, sediments and reduced solar insulation as the factors in forcing the system.

Simulated biomass might be used as a surrogate for overall ecosystem health. Underwood suggests that biomass is the variable to track when attempting to evaluate the response of a system to stress (Underwood, 1989). However, salinity has been reported as the primary determining factor in vegetation stratification in southern Louisiana (Visser et al., 1996), while principal component analysis showed five zones in mid-Atlantic tidal wetlands based on salinity (Bulger et al., 1993). Table 6.4 summarizes the ranges of salinities reported for various marsh types, as well as the salinity ranges utilized in determining marsh types the CELSS model (Sklar et al., 1991b) and the BTNEP landscape model (White et al., 1997).

Table 6.4 Characteristic salinity reported in literature for various marsh types. Range is reported in brackets.

Reference	Marsh type				
	fresh	fresh/ intermediate	intermediate	intermediate/ brackish	brackish saline
(Visser et al., 1996)	0.0 (0.0 – 3.0)		4.0 (2.0 – 8.0)		10.0 (4.0 – 18.0) 18.0 (8.0 – 29.0)
(Bulger et al., 1993)	(0.0 – 4.0)		(2.0 – 14.0)	(11.0 – 18.0)	(16.0 – 27.0) (24.0 – 36.0)
(Mitsch et al., 1993)	(0.0 – 0.5)	(0.5 – 5.0)	(5.0 – 18.0)	(18.0 – 30.0)	(30.0 – 40.0)
(White et al., 1997)	(0.0 – 4.5)				(>4.5 – 12.0) (>12.0 – 40.0)
(Sklar et al., 1991b)	(0.0 – 4.5)				(4.2–11.0) (>11.0–36.0)

Habitat Succession Unit Model

The habitat succession unit model is composed of two parts, a counter and a switcher. The counter checks daily the values of salinity and biomass, compares them to the classification criteria and then increments the habitat type counter of a summation matrix. The initial value assigned to a habitat type is a year's worth of daily values. At the end of every year of simulation, the habitat switcher algorithm queries the daily habitat counts and by simple majority assigns a habitat type to each cell. If the habitat type has changed, the appropriate new parameters are assigned to the productivity subroutine. The classification criterion that define a habitat type are biomass (kg OM/m^2) and salinity (ppt). Salinity affects the classification on any given day only if the wetland experiences flooding on that day, thus water elevation is also necessary in the evaluation. The biomass and salinity limits used in the BTNEP landscape model for each habitat type are defined in the top portion of Table 6.5.

As stated earlier, choosing which state variables to use to determine habitat succession, and determining the ranges of those variables, can be difficult because of the lack of data. The limits of these variables were determined in trial and error runs of the landscape model. While they do produce the simulation with the highest fit value, they

are not in agreement with salinity ranges reported in the literature (Table 6.4 and Appendix C). In particular, the lower limit of salinity for brackish marsh is low and the upper limit for fresh marsh is high. Again we can turn to the Caernarvon study area to validate a unit model with these values.

Table 6.5 Biomass and salinity limits used in the BTNEP landscape model (top) and limits suggested as more realistic limits to test with Caernarvon study area data.

BTNEP limits		
Biomass (kg OM/m²)	Minimum	Maximum
Freshwater Marsh	0.92	10.0
Brackish Marsh	0.44	10.0
Salt Marsh	1.2	10.0
Salinity (ppt)	Minimum	Maximum
Freshwater Marsh	0.00	4.5
Brackish Marsh	4.5	12.0
Salt Marsh	12.0	40.0
Proposed limits		
Biomass (kg OM/m²)	Minimum	Maximum
Freshwater Marsh	0.25	10.0
Brackish Marsh	0.25	10.0
Salt Marsh	0.25	10.0
Salinity (ppt)	Minimum	Maximum
Freshwater Marsh	0.00	2.0
Brackish Marsh	2.0	19.0
Salt Marsh	19.0	40.0

Validation - Caernarvon Habitat Analysis

The Caernarvon study was specifically monitored to detect habitat change caused by changing environmental conditions. Vegetation transects were conducted from 1988 through 1994 along the transect grid shown in Figure 6.8. Sites were identified as marsh, levee, natural bayou or lake bank. Emergent vegetation was recorded as the percent of each of the 29 plant species or as unvegetated. The species included are indicated in the table in Appendix A by the inclusion of an asterisk. When only the sites that contained marsh for all 6 years of the monitoring are included, there are 198 marsh locations. The observations for the 6 years of monitoring were transformed into a weighted index of habitat using the same transformation described in Chapter 4. During the six years of observation, 31 sites became saltier, 14 sites did not change, and 153 sites became fresher (Figure 6.8).

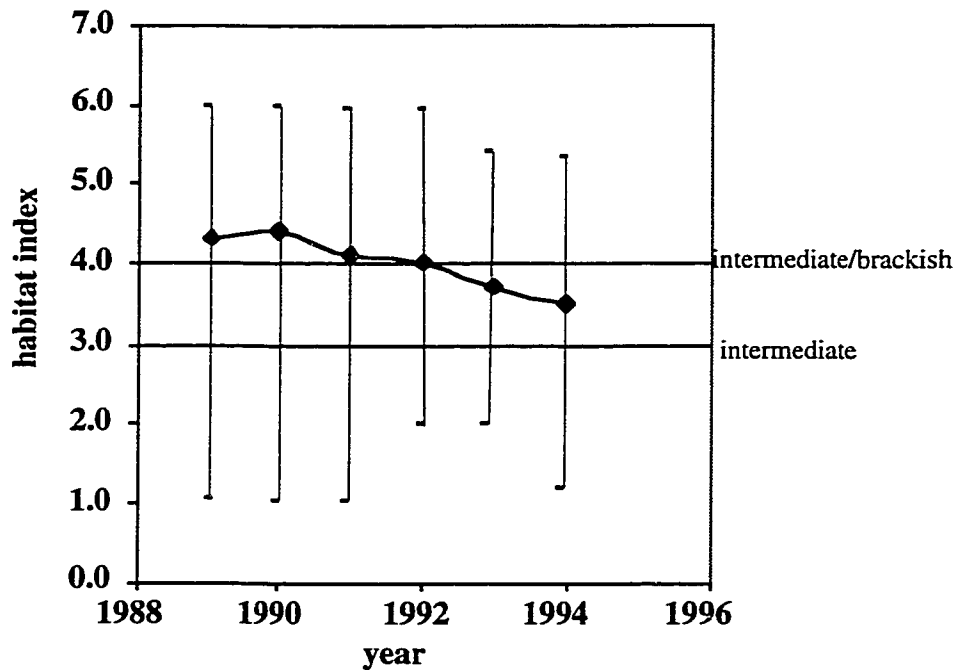


Figure 6.9 Average habitat index (weighted sum of habitat type as described in Chapter 4) for 198 marsh sites in the Caernarvon study area. The freshwater diversion structure became operational in 1991. Habitat limits at 4.0 (intermediate/brackish) and 3.0 (intermediate) marsh were taken from Mitch and Gosselink, 1993.

The primary production values predicted by the unit model for Caernarvon vegetation in the first part of this chapter can be used with the measured environmental conditions to test the habitat succession model. The model can be run with the BTNEP limits and the proposed new biomass and salinity to see if they are, in fact, appropriate for southern Louisiana marshes. The habitat succession unit model was run using data from Bay Gardene, biomass predictions, and the salinity limits from the BTNEP landscape habitat switcher. Fresh conditions were predicted 7.3% of the time, brackish conditions 72.2% of the time, and saline conditions 20.5% of the time. If the more saline conditions of California Bay and biomass predictions are used, fresh conditions exist 0.2%, brackish conditions exist 54%, and saline conditions exist 45.8% of the time.

The biomass predictions for fresh, brackish, and saline marsh were used (biomass results shown in Figure 6.7). Since the biomass predictions never exceed the

limits, all three simulations produce the same habitat predictions. The model chooses the habitat annually, on a majority basis, and thus the existing BTNEP parameterization would yield accurate results for this data.

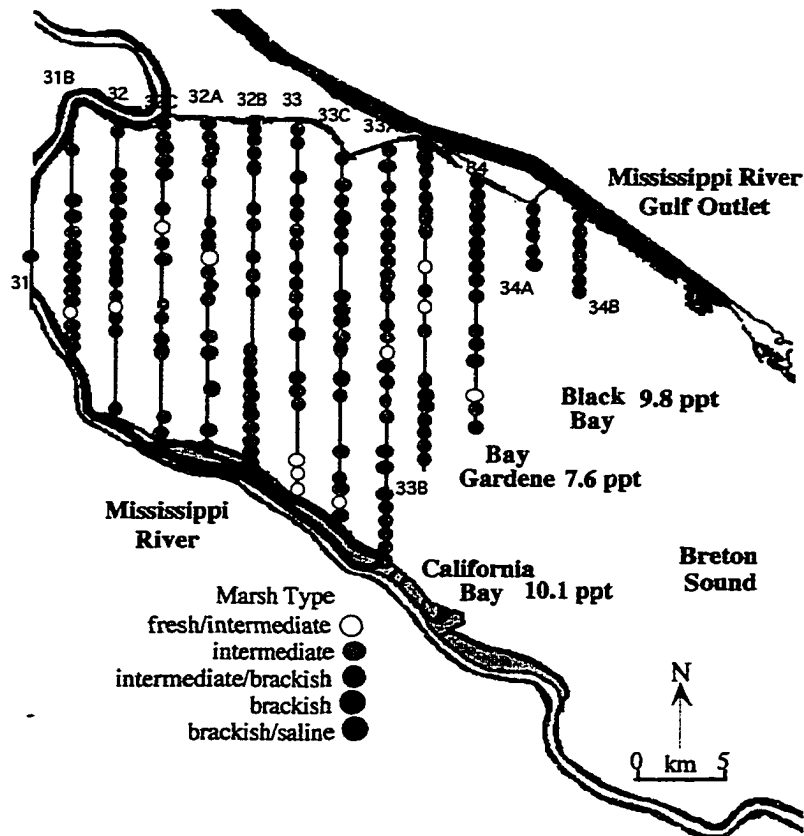


Figure 6.10 Marsh locations at the Caernarvon fresh water diversion study area. Color density indicates marsh type in 1994 based on a weighted average of vegetation observed. The average salinity for 1991 - 1993 for the three stations in Breton sound are noted beside the station.

Figure 6.10 shows predominantly brackish vegetation that had previously been more saline vegetation (Figure 6.8). Yet the saline counter was incremented 20% of the time in Bay Gardene and 46% of the time in California Bay, when in fact the data indicates a more brackish marsh. When the habitat unit model was run using salinity limits (Mitsch et al., 1993) that are consistent with the habitat type, brackish conditions are predicted 99.9% of the time for Bay Gardene and 98.7% of the time for California

Bay. These predictions are much more in keeping with the observed habitat in the study area and are the values that I recommend for use in the determination of salt and brackish marsh in an improved landscape model.

It remains now to use the new parameterization of productivity and habitat succession in a landscape setting and evaluate the results of the ecological landscape model.

CHAPTER 7. RESULTS AND DISCUSSION

The final step in this research project is to take the newly parameterized unit models for biomass production and habitat succession and insert them into the framework of the ecological landscape model. The success or failure of the landscape model to reproduce the Terrebonne basin base maps can be evaluated with the multiple resolution goodness of fit index that has been investigated in detail. In this chapter I will first review salient details of the multiple resolution goodness of fit index, then present the results of various landscape simulations and finally, discuss features of the landscape model results.

Review of F_t

The multiple resolution goodness of fit index should be presented in context of the map size and number of categories of comparison. Figure 7.1 is one attempt to visualize this context. Because the relative order of the analysis does not change with the use of the $F_t(k)$ or $F_t(\mu, \sigma)$ version of the metric, Figure 7.1 only contains analysis of the $F_t(k)$ for $k=1$. The comparison of two identical maps would yield a multiple resolution goodness of fit index of 100, but in reality the expected value can approach only 91 or 92 when applied to the base maps available for southern Louisiana. The expected change in the landscape caused by the active processes is approximately 1 point of fit per year given the mapping accuracy that is available at this time. Thus the minimum model simulation run that can validly use the fit index is between 9 and 10 years. One problem with using this index with this data is the relatively small range within which model improvements can be made. When the actual 1978 habitat map is compared to the actual 1988 habitat map, the $F_t = 85.74$. At most, the index can be improved by only about 14 points, and realistically that value is about 7 points. Keeping

these limitations in mind, the fit index will be the method used to measure the goodness of fit of the ecological landscape model results.

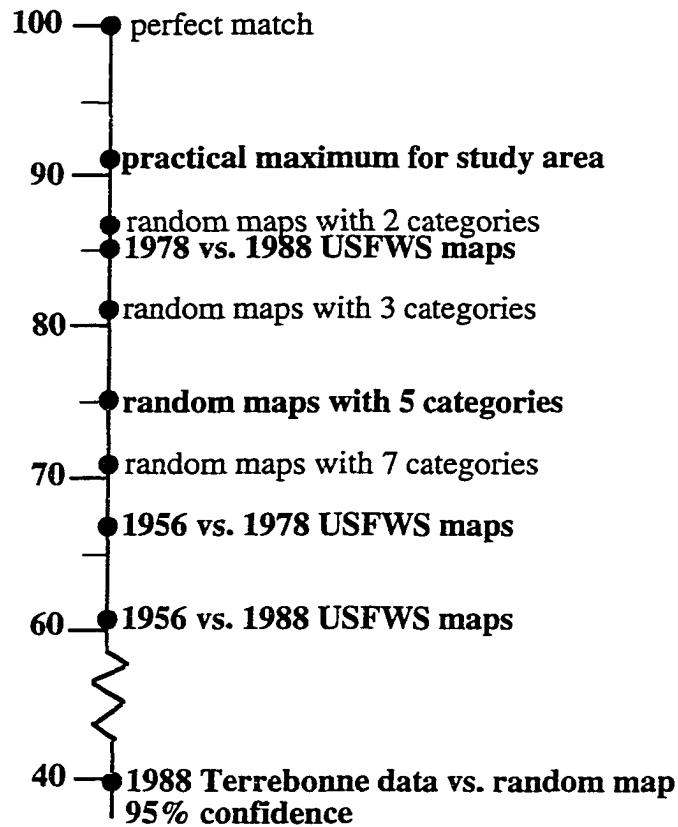


Figure 7.1 Scale of F_t ($k=.1$) for various comparisons of 77 x 112 maps. Simulations pertaining to the BTNEP landscape model are shown in bold.

Landscape Model Simulations with New Unit Modules

The primary production unit module as it is presented in Chapter 6 can be successfully parameterized with coefficients derived from literature values. The unit module can predict reasonable above and below ground seasonal production for 10 years using hydrodynamic forcing functions that were simulated from the BTNEP landscape model (Figure 6.4). In addition, it can successfully simulate above and below ground seasonal production for three years using hydrodynamic forcing functions collected at a controlled experimental location (Figure 6.7). This unit model is most sensitive to the

parameterization of translocation, temperature, and photosynthetic production rates, in that order. In general, non-photosynthetic biomass is more sensitive to rate manipulation than photosynthetic biomass. And finally, the wetlands, as modeled, are more sensitive to flooding stresses than to salt stresses.

When this unit model is used in the BTNEP landscape model and a simulation is run, the results are poor. The results are shown in Table 7.1 and are labeled “new unit productivity, old hab”. Figure 7.2 shows the resultant habitat map and the difference map for the simulation from 1978 to 1988. The difference map contains four categories of data. First are cells that are either predicted accurately, or cells that contain urban or swamp habitat. These are denoted as “no change” (the spread of urban habitat and the behavior of swamp habitat are not the focus of this study and thus those changes are not relevant). Second are cells that were marsh in 1988 but were predicted to be open water resulting from biomass death (blue). Third are cells that were marsh in 1988 and were predicted to be marsh, but were classified incorrectly by the habitat succession routine (orange). And finally, there are cells that were open water in 1988 but were predicted to be marsh (pink).

Table 7.1 Results of various parameterizations of the landscape model for the Terrebonne marsh

Analysis	Ft (k)	Ft (μ, σ)	cells fresh	cells brackish	cells saline	cells water	land/water
actual 1988 values			1170	828	576	2106	0.619
BTNEP landscape base case	85.40	81.44	1100	865	551	2080	0.607
new unit productivity old hab	77.05	71.63	773	625	324	2874	1.092
new unit productivity new hab	73.25	67.90	603	932	222	2836	1.062
best prod old hab	87.07	81.38	1102	844	557	2093	0.613
best prod new hab	86.77	81.05	1077	878	536	2100	0.616
56 vs. 78 USFWS	67.70	64.27					
78 vs. 88 USFWS	85.74	81.77					
56 vs. 88 USFWS	60.92	56.33					

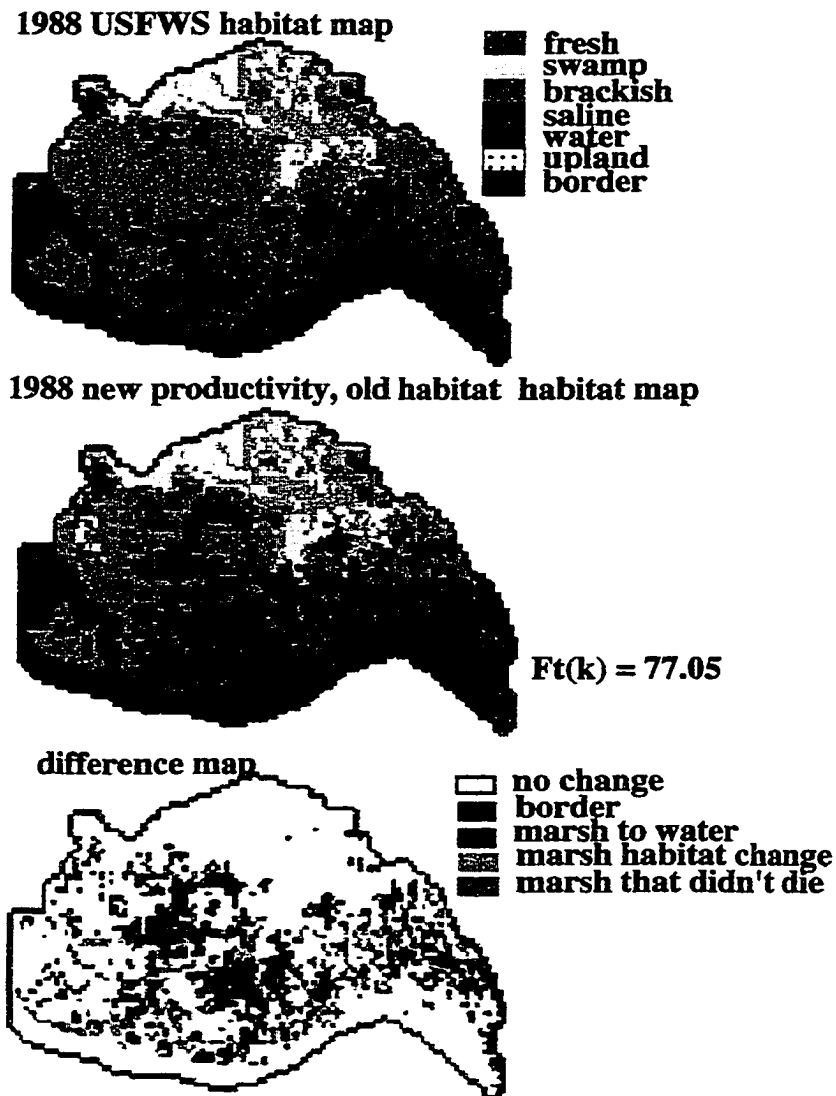


Figure 7.2 Results of ecological landscape simulation using the old habitat succession routine and the new biomass productivity routine. The difference map shows the cells of the landscape simulation that did not match the 1988 USFWS base map. See text for details of the legend for the difference maps.

This simulation seriously over-predicts the amount of open water and the $F_t(k)=77.05$. The time series of the state variables from various locations in the basin indicate that the productivity steadily decreases until the habitat succession routine considers the marsh dead (less than 0.25 kg.m^2 for the majority of the year). The habitat then reverts to open water. The newly parameterized habitat succession routine described in Chapter 6 was then incorporated into the landscape model. These results are labeled “new unit productivity, new hab” in Table 7.1. Figure 7.3 shows the corresponding

habitat and difference maps for this simulation. The $F_t(k)$ is 73.25, an even worse case than the first. The change from marsh to open water occurs as in the first simulation, but in addition, the open water allows for more salt water intrusion. The elevated salinity, and a new habitat succession routine that is more sensitive to salinity in the fresh marsh, result in a large portion of the center of the basin being categorized incorrectly. The unit models as they are formulated cannot be translated directly into the landscape model.

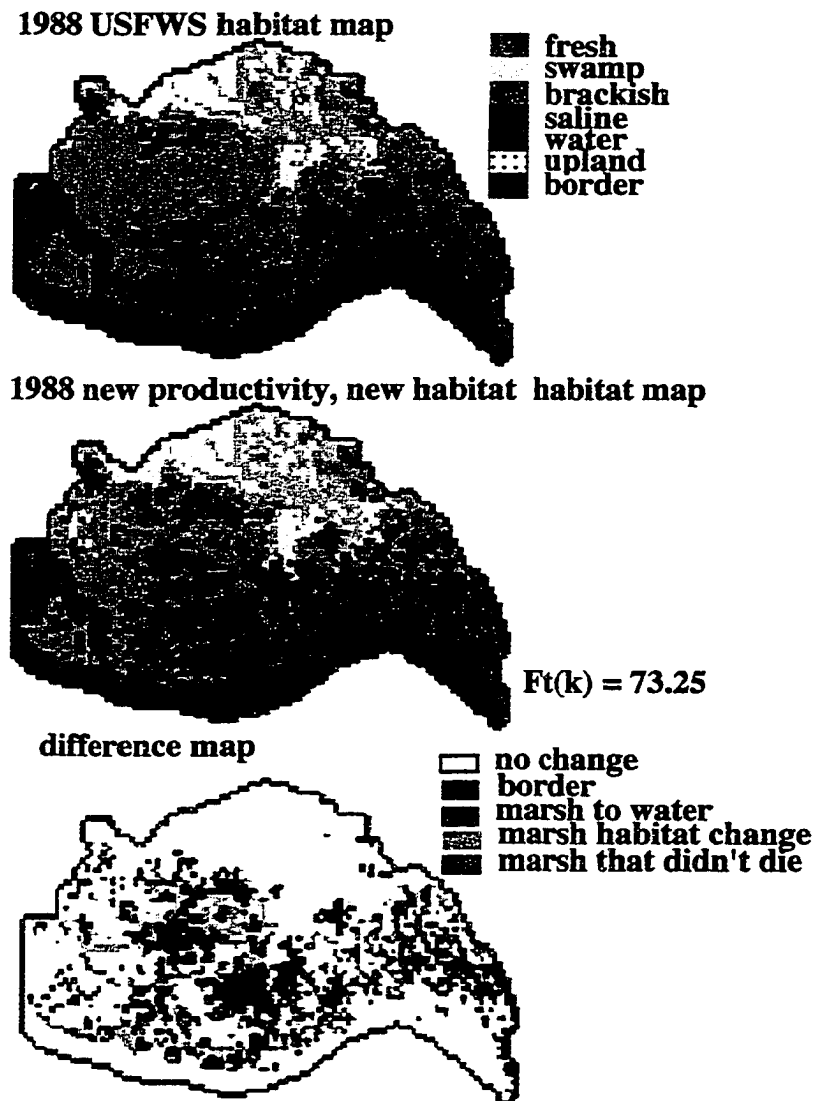


Figure 7.3 Results of ecological landscape simulation using the new habitat succession routine and the new biomass productivity routine. The difference map shows the cells of the landscape simulation that did not match the 1988 USFWS base map. See text for details of the legend for the difference maps.

The parameters that produce the optimum values for a unit module may not be the same as those for a landscape. The scale dependence of the parameterization of spatial models has been recognized with species distribution. They are not a linear function of fine-scale movement rates, but rather are controlled by different processes operating at different scales (Johnson et al., 1992; With et al., 1996). The spatial dependence of productivity rates is even more difficult to quantify, however, there are some examples of this in recent ecological literature. Band (Band et al., 1991) has identified potential bias in distributed modeling that is introduced by employing landscape mean values for input variables when using a model with significant nonlinear responses. Turner (Turner et al., 1995) reported that net primary production simulated over a landscape at 50 m² grid size is 11% higher than at 1 km² grid size and concludes that there is no benefit to simulating coupled hydrodynamic and biological processes at a scale finer than 1 km². In a non-spatial version of a productivity model, a single species will out-compete all others, while the spatial version predicts that coexistence is possible (Pacala et al., 1994). And finally, it may be that the assumption of 2 m spatial uniformity of forest plant processes is not appropriate for marsh plants (Lechowicz et al., 1991).

The question that is unanswered then, is what rates or parameterizations should be adjusted to bring the simulations of the landscape model into best agreement with the base maps? An investigation of some of the results of the simulations may provide some guidance. Figure 7.4 contains the results of the multiple resolution goodness of fit index for the simulations (plots labeled 1 and 2). There are no obvious discontinuities or jumps in values that would make one look at a particular window size as the scale of the problem. Not enough is known about the response of F_w and F_t to the fragmentation of landscape. The only distinguishing characteristic of the plot of F_w is that it remains low over the whole range of window sizes. This implies the problem is system wide and not confined to one scale or habitat type.

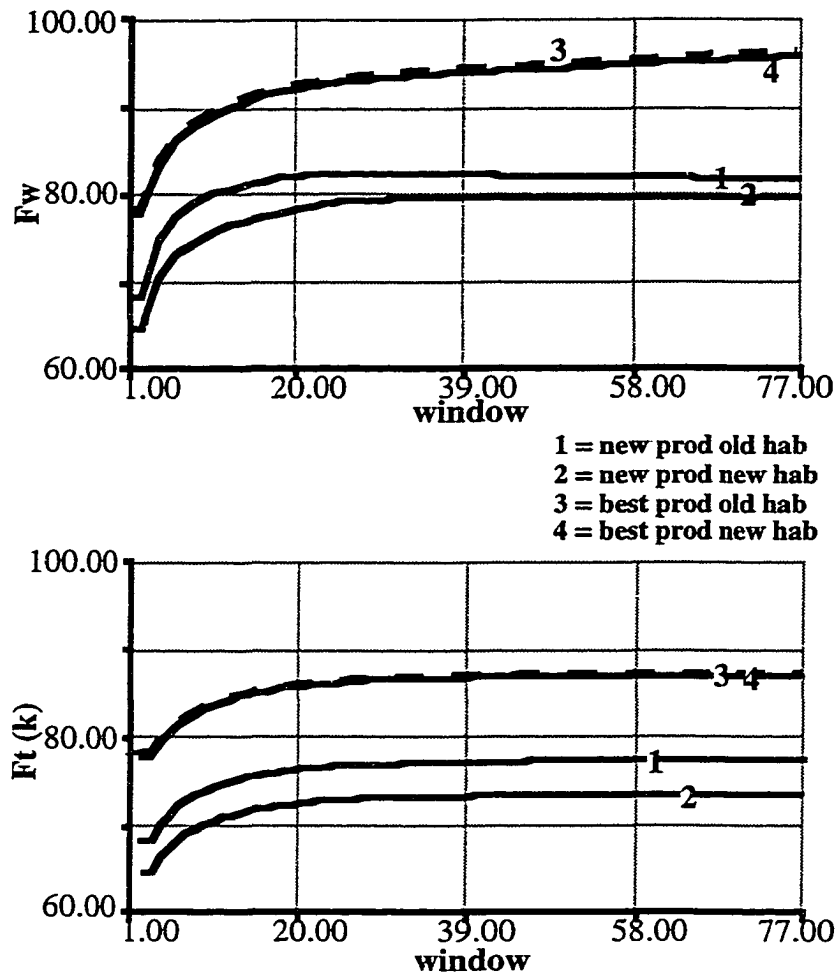


Figure 7.4 Individual window weights F_w (top graph) and multiple resolution goodness of fit index $F_t(k)$ (bottom graph) for the four landscape simulations described in this chapter.

Successive manipulations with various coefficients as they are applied in the landscape model yielded better results only when the photosynthetic production rate was increased. In the unit model parameterization, only the rates for fresh marsh were modified (Table 6.2). When transferring the unit model into the spatial landscape model, all three marsh photosynthetic production rates had to be changed. The saline rate was increased from 43 to 115, the brackish rate was increased from 60 to 115, and the fresh rate was increased from 55 to 98 (all rates are expressed as grams of biomass/m² per day). These increases represent a 170 to 270% increase in the photosynthetic production rate. When these rates were utilized in the landscape model, (best prod, old hab), the

highest fit thus far was obtained, $F_t(k)=87.07$. The habitat maps and difference maps are shown in Figure 7.5. When the new habitat succession values were coupled with the best productivity (best prod, new hab) the results were slightly lower, $F_t(k)=86.77$. These results are shown in Figure 7.6.

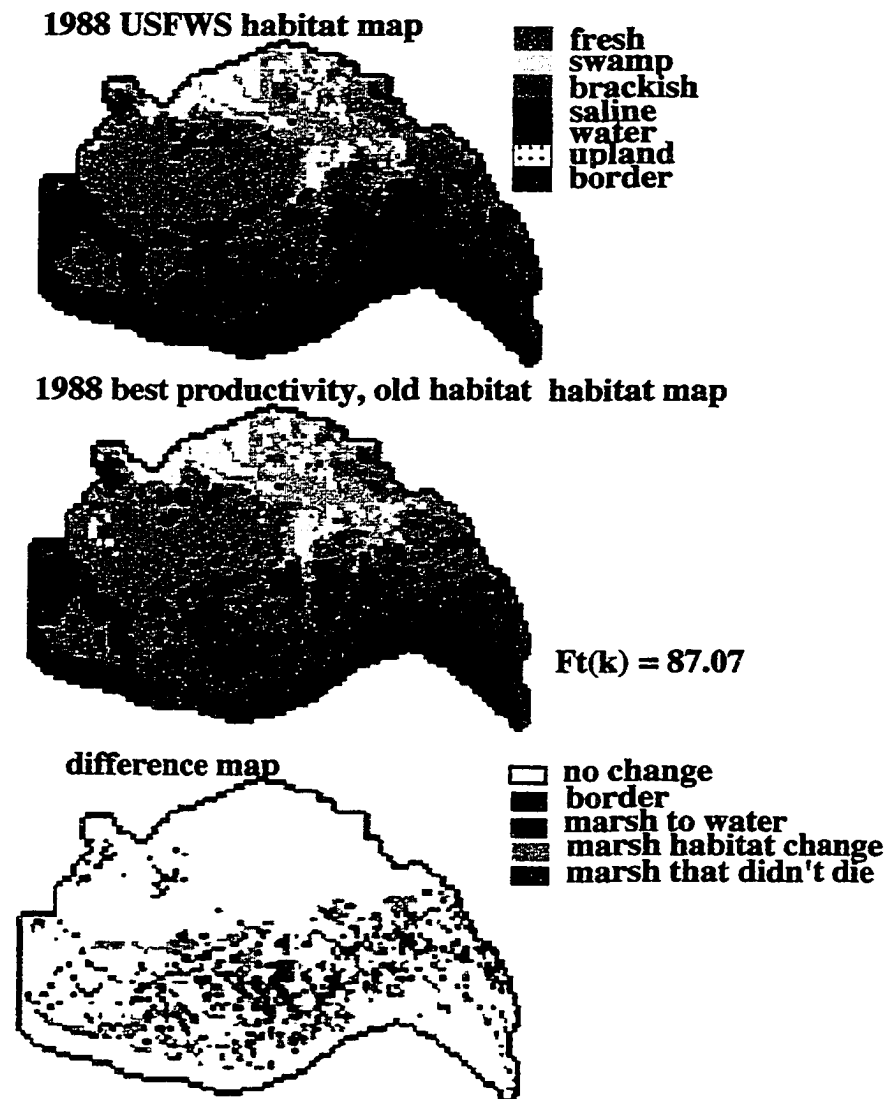


Figure 7.5 Results of ecological landscape simulation using the old habitat succession routine and the best biomass productivity routine. The difference map shows the cells of the landscape simulation that did not match the 1988 USFWS base map. See text for details of the legend for the difference maps.

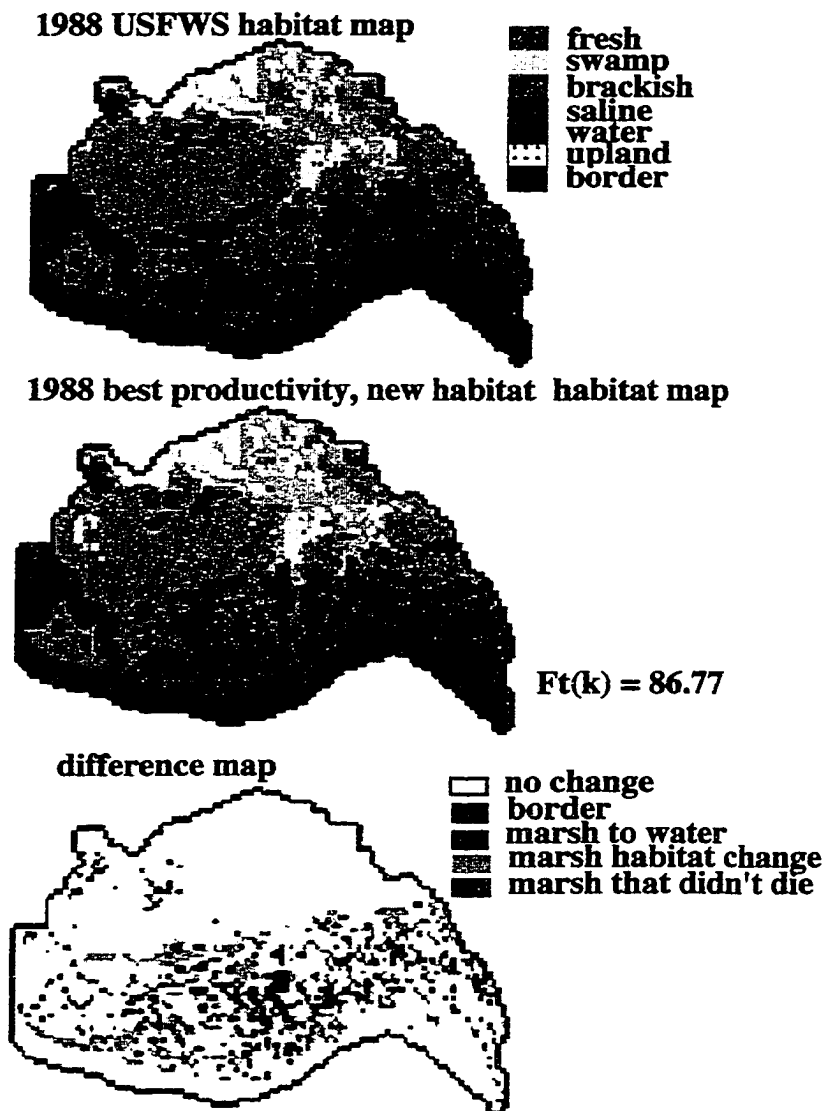


Figure 7.6 Results of ecological landscape simulation using the new habitat succession routine and the best biomass productivity routine. The difference map shows the cells of the landscape simulation that did not match the 1988 USFWS base map. See text for details of the legend for the difference maps.

Discussion of Productivity Parameterization

Only the spatial scale of the individual cells was modified when inserting the unit model into the landscape model, in this case by an order of 10^6 . Yet this required a change in the parameterization of the photosynthetic rate by an order of 10^2 . There are theoretical reasons why this may be necessary. Heuvelink (1998) has looked at this question in detail and suggests three factors to consider.

The first consideration is that different processes dominate at different scales, and so different processes are ignored in the simplification step of the model development. An example of this might be the positive interactions among marsh plants that buffer one another from potentially limiting physical stresses. Bertness (1994) showed that distribution patterns of New England salt-marsh plants are strongly influenced by facilitative associations among neighboring plants. Positive associations such as these are likely common but unappreciated forces in harsh environments that have been largely overlooked. Hacker (1995) showed that the presence of *Juncus* with its ability to withstand waterlogging and salt stress can create a hospitable environment for *Iva*. But in another modeling study, daily photosynthesis could be predictably estimated between modeling scales; it was the hydrologic outflow that was not highly correlated between them. (White et al., 1994).

Second, input data are often absent or of a much lower quality at larger scales, which results in a tendency to use simpler, empirical models at the larger scale. In fact, data limitations of the forcing functions have been discussed previously in this study. Another example might be the assumption that biomass per square kilometer can be extended linearly from measurements of biomass per square meter. The various mix of plant species and the association of vegetated versus unvegetated area within that square kilometer are factors that we do not have the data to verify.

And finally, Heuvelink introduces a concept called "support". Support is similar to "level of aggregation" and "sample volume" that changes with change of scale, and thus affects the relationships between them. Moving from small to large scale implies that the model input and output have become a kind of averaging of point values within the larger spatial unit or block. A change of support may require a change in the model because the relationships that exist between variables at the point support need not extend to the block support.

Applying this concept, the productivity problem may lie in the interaction of a combination of biological processes. In addition to the photosynthetic rate, the biomass depends on the rate of translocation and respiration of above and below ground biomass. The values of above ground biomass, below ground biomass and respiration are within the ranges for *S. alterniflora* reported by Morris in a field study in a Sapelo Island marsh in Georgia (Morris et al., 1984b). This leaves the translocation rate, the parameterization the unit model is most sensitive to, unverified.

This discussion has produced a number of basic research topics that would be valuable to have as supporting evidence in the study of wetland vegetative modeling. To what extent does positive (or negative) interaction play in the primary production of marsh biomass? Is there any relationship in the orders of magnitude of rates of various processes at varying scales? How do the rates of processes vary with habitat type and species? And what role does translocation of biomass between above and below ground biomass play in this scaling problem?

CHAPTER 8. CONCLUSIONS

In this study I have attempted to investigate questions about ecological landscape modeling in detail. As often happens in research, the answers to these questions have led to many others. Despite the scaling uncertainties, an ecological landscape model now exists for the Terrebonne wetlands which is successful at predicting habitat succession. This is important because a serious question that can now be addressed is, when humans make changes to the system, what are the consequences on the habitat at the landscape scale?

This study has an immediate application in the science of wetland restoration (Dobson et al., 1997). Wetland loss, water quality, hydrologic isolation, and saltwater intrusion are all the problems identified by the BTNEP as most likely to affect the long-term productivity and function of Barataria–Terrebonne estuary. Alternatives such as freshwater diversion projects, barrier island restoration, levee construction or degradation, and various structures to influence water routing (Soileau, 1990) have been devised to prolong the sustainability of these wetlands. These solutions can now be evaluated in a scientific and systematic way.

The management plans proposed by the BTNEP have already been modeled with the BTNEP landscape model (Reyes et al., 1999; White et al., 1997). The results of this study can shed light on the usefulness of those model results, especially as they pertain to decisions that rely on the multiple resolution goodness of fit measurement. This study has found that while a perfect simulation model would predict a multiple resolution goodness of fit index of 100, in reality it can only approach 91 - 92 when applied to the base maps available for southern Louisiana. The expected change in the landscape due to the active processes is approximately 1 point of fit per year, thus the minimum model simulation run that can validly use the fit index is between 9 and 10 years. The choice of whether to use $F_t(k)$ or the alternate formulation, $F_t(\mu, \sigma)$, can be made keeping the

benefits and limitations of each in mind. Choosing $F_t(k)$ allows for a slightly higher average upper limit, a larger expected rate of change per year and consequently a shorter minimum simulation. Choosing $F_t(\mu, \sigma)$ allows the user to choose an optimum window and spread for their analysis and reduces the difference due to aggregation and to unequal resolutions of the base maps, however it requires a slightly longer minimum simulation run.

A limitation in the use of this index is the small range over which improvements can be measured. Fit results higher than 40.31 are significant at the 95% level (Figure 8.1). But the practical lower limit is the value of the index when the beginning map is compared to the ending map, in this case 85.74. Even small improvements in the fit are important because the useful spread of values is between 85 and 92.

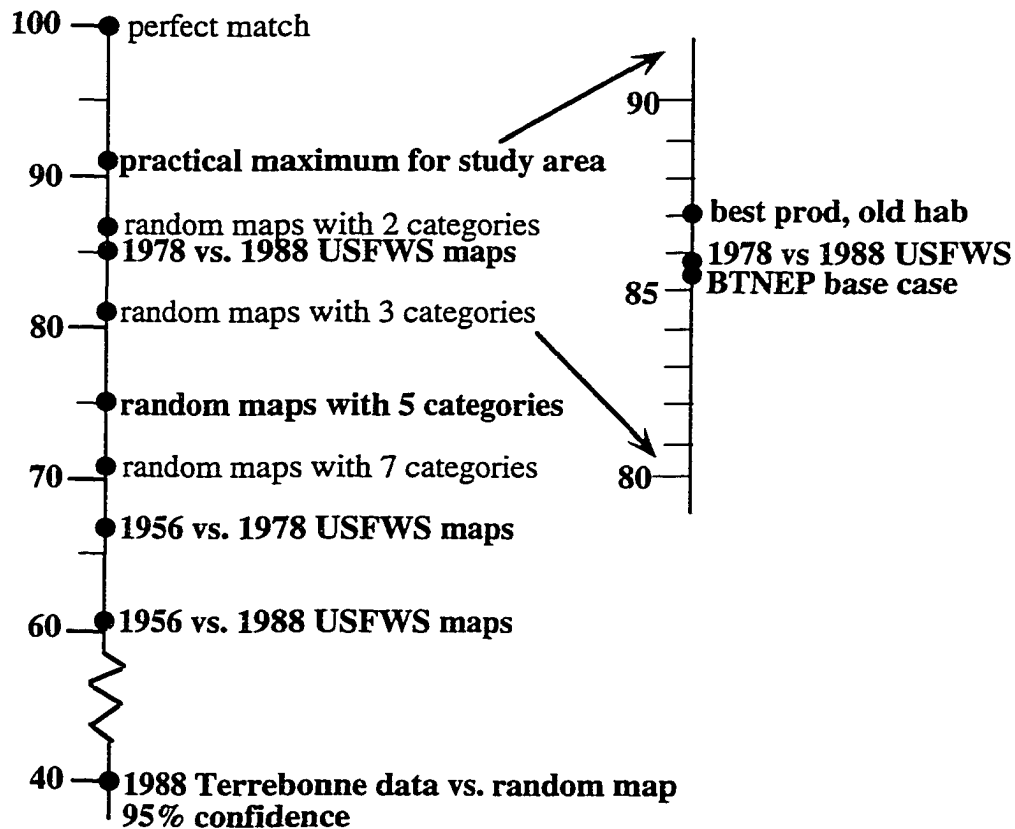


Figure 8.1 Scale of $F_t(k)$ for various comparisons of 77 x 112 maps. Simulations pertaining to the BTNEP landscape model are shown in bold. Refer to the text and Table 7.1 for details of the simulations.

The unit models that are being used in the landscape model are most sensitive to the parameterization of the translocation, temperature, and photosynthetic production rates, in that order. In general, non-photosynthetic biomass is more sensitive to rate manipulation than photosynthetic biomass, and the wetlands are more sensitive to flooding stresses than to salt stresses. Because the productivity is sensitive to temperature changes as well as flooding regimes, sea level rise and global warming scenarios can be run alone and in combination with human activities. There are very few, if any, objective mechanisms that can be used to evaluate landscape scale cumulative impacts in the context of global climate change.

The most pressing question that remains unanswered is the role that scale manipulation plays in the parameterization of rates of the processes. Further research needs fall into three types of work: (1) collection and evaluation of data, (2) unit modeling of processes from first principles, and (3) rigorous investigation of rates and processes as they are translated from one scale to another.

There are several examples of the type of research for item 1. Climate analysis should be done to analyze the options and suitability of using time series from diverse locations, as is often required in landscape models. Habitat data should be assembled from as many sources as possible and the techniques of geostatistics should be used to compile a more complete time series of habitat maps. Relationships, such as Eh vs. duration of flood or the effects of translocation vs. temperature vs. photosynthetic rates for various marsh types are needed to parameterize productivity models.

The development of unit models using first principles will allow us to tease out the interdependencies of photosynthesis, translocation and respiration in the prediction of biomass. An example of a stress that should be included explicitly in the productivity unit model is the effect of the salinity in the soils on the above and below ground biomass. An example of a process that should be included explicitly in the habitat succession unit model is the re-vegetation of bare soil.

Finally, the scaling factors required to transfer the unit model to the landscape model need to be investigated. Are these relationships universal to all habitat types and scales of models? What is the relationship between scaling temporal rates and scaling spatial rates? There is much interesting and exciting work to be done.

This study has shown the value of the iterative process of model development, evaluation, and refinement to predict the productivity, diversity and resilience of ecosystems. It has suggested areas for further research to enhance our understanding of ecosystem processes while providing tools and guidance to natural resource managers in exercising a landscape view of natural resources.

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APPENDIX A LIST OF SPECIES USED IN MAP GENERATION

This list of species was used to identify flora observed in the NRCS survey May-June 1994. The scientific names and habitat were assigned using references from (Tiner, 1993), (Materne, 1996), (Radford et al., 1968), (Godfrey et al., 1981) and personal communications with Materne, Mendelssohn, Trahan (1997). Species that were determined to be duplications are cross-referenced with the species code of the duplicate.

code	common name	scientific name	habitat
1	Alligatorweed	<i>Alternanthera philoxeroides</i>	fresh
2	White waterlily	<i>Nymphaea odorata</i>	aq/fresh
3	Arrowhead *	<i>Sagittaria latifolia</i>	fresh
4	Bulltongue *	<i>Sagittaria lancifolia</i>	fresh
5	Bladderwort	<i>Utricularia vulgaris</i>	aq/fresh
6	Carolina water hyssop	<i>Bacopa carolinians</i>	fresh
7	Cattail	<i>Typha</i> sp.	fresh
8	Buttonbush (#41)	<i>Cephalanthus occidentalis</i>	fresh
9	Common rush	<i>Juncus effusus</i>	fresh
10	Delta duckpotato *	<i>Sagittaria platyphylla</i>	fresh
11	Duckweed	<i>Lemna major</i>	aq/fresh
12	Eastern baccharis *	<i>Baccharis halimifolia</i>	fresh
13	Elephant ear (#40)	<i>Colocasia esculenta</i> L.	fresh
14	Floating pennywort	<i>Hydrocotyle ranunculoides</i>	aq/fresh
15	Giant bristlegrass	<i>Setaria magna</i>	fresh
16	Giant cutgrass	<i>Zizaniopsis miliaceae</i>	fresh
17	Giant ragweed	<i>Ambrosia trifida</i>	fresh
18	Hemp sesbania (#140) *	<i>Hemp sesbania</i>	fresh
19	Jamaica sawgrass	<i>Cladium jamaicense</i>	fresh
20	Lizards Tail	<i>Saururus cernuus</i>	fresh
21	Lotus	<i>Nelumbo lutea</i>	aq/fresh
22	Louisiana palmetto (#136)	<i>Sabal minor</i>	fresh
23	Maidencane	<i>Panicum hemitomon</i>	fresh
24	Marshfern	<i>Thelypteris thelypteroides</i>	fresh
25	Pickereelweed	<i>Pontederia cordata</i>	fresh
26	Rattlebox *	<i>Sesbania Drummondii</i>	fresh
27	Royal fern	<i>Osmunda regalis</i>	fresh
28	Smooth beggarticks	<i>Bidens laevis</i>	fresh
29	Sedge	<i>Carex fissa</i> / <i>Carex folliculata</i>	fresh
30	Spikesedge (rush)	<i>Eleocharis</i> sp.	fresh
31	Swamp smartweed *	<i>Polygonum amphibium</i>	fresh
32	Umbrella pennywort	<i>Hydrocotyle umbellata</i>	fresh

33	Water hyacinth	<i>Eichornia crassipes</i>	aq/fresh
34	Waterwillow	<i>Decodon verticillatus</i>	fresh
36	Water primrose	<i>Ludwigia octovalis</i>	fresh
37	Spiderlilly	<i>Hymenocaulis occidentalis</i>	fresh
38	Wax Myrtle *	<i>Myrica cerifera</i>	fresh
39	Bushy bluestem	<i>Andropogon glomeratus</i>	fresh
40	Elephant ear(#13)	<i>Colocasia esculenta L.</i>	fresh
41	Buttonbush (#8)	<i>Cephalanthus occidentalis</i>	fresh
42	Wild iris	<i>Iris virginica</i>	fresh
43	Cypress tree	<i>Taxodium distichum</i>	fresh
44	Cypress weed (dog fennel)	<i>Eupatorium capillifolium</i>	fresh
45	Marsh mallow	<i>Hibiscus coccineus</i>	fresh
46	Hairy rice grass (cutgrass)	<i>Leersia oryzoides</i>	fresh
47	American bulrush	<i>Scirpus americanus</i>	fresh
48	Bearded sprangletop *	<i>Leptochloa fascicularis</i>	fresh
49	California bulrush (bulwhip)	<i>Scirpus californicus</i>	fresh
50	Coast cockspur	<i>Echinochloa sp.</i>	fresh
51	Fragrant flatsedge *	<i>Cyperus odoratus</i>	fresh
52	Gulf cordgrass	<i>Spartina spartinae</i>	intermediate
53	Hairypod cowpea *	<i>Vigna luteola</i>	fresh/inter
54	Purple pluchea	<i>Pluchea</i>	fresh/inter
55	Seashore paspalum	<i>Paspalum vaginatum</i>	intermediate
56	Softstem bulrush (bulwhip)	<i>Scirpus validus</i>	fresh
57	Southern naiad	<i>Najas guadalupensis</i>	fresh/inter
58	Virginia saltmarshmallow	<i>Hibiscus lasiocarpus</i>	intermediate
59	Woolly rosemallow	<i>Kosteletzkya virginiana</i>	fresh/inter
60	False loose strife	<i>Ludwigia leptocarpa</i>	fresh
61	Mosses	<i>Mayaca spp., Lycopodium spp.</i>	fresh
62	Hackberry *	<i>Celtis laevigata</i>	fresh
63	Sedge white top (#113)	<i>Dichromena cololrata</i>	fresh
64	Water willow - black	<i>Salix nigra</i>	fresh
65	Tallow tree - Chinese Tallow	<i>Sapium sebiferum</i>	spoil/fresh
66	Dillweed (mock bishopweed)	<i>Ptilimnium</i>	fresh
67	Walters millet *	<i>Echinochloa walteri</i>	fresh
68	Elderberry	<i>Sambucus canadensis</i>	spoil/fresh
69	Blue stem *	<i>Andropogon sp.</i>	fresh
70	Morning glory (#78) *	<i>Ipomoea sagittata</i>	fresh/inter
71	Big cordgrass	<i>Spartina cynosuroides</i>	brackish
72	Camphor pluchea *	<i>Pluchea camphorata</i>	brackish
73	Coast hyssop	<i>Bacopa Monnieri</i>	fresh/inter
74	Common reed (roseau cane) *	<i>Phragmites australis</i>	brackish
75	Dwarf spikesedge *	<i>Eleocharis parvula</i>	fresh/inter
76	Gulfcoast waterhemp	<i>Acnida spp.</i>	brackish
77	Marshhay cordgrass *	<i>Spartina patens</i>	brackish
78	Marsh morningglory (#70) *	<i>Ipomoea sagittata</i>	fresh/inter

79	Olney bulrush *	<i>Scirpus olneyi</i>	brackish
80	Parrotfeather	<i>Myriophyllum aquaticum</i>	aq/fresh
81	Sago pondweed	<i>Potamogeton pectinatus</i>	brackish
82	Saline aster *	<i>Aster tenuifolius</i>	brackish/salt
83	Salt heliotrope	<i>Heliotropium</i> spp.	brackish/salt
84	Saltmarsh bulrush	<i>Scirpus maritimus</i> , <i>Scirpus robustus</i>	brackish
85	Showy dodder	<i>Cuscuta</i> spp.	brackish
86	Wand lythrum	<i>Lythrum</i> spp.	brackish
87	Widgeongrass	<i>Ruppia maritima</i> L.	aq/brackish
88	Paspalum	<i>Paspalum</i> spp.	brackish
89	Thistle	<i>Cirsium nuttallii</i>	fresh
90	Begger Lice	<i>Desmodium</i> sp.	spoil/fresh
91	Red maple (#129)	<i>Acer rubrum</i>	fresh
92	Sweet bay	<i>Magnolia virginiana</i>	fresh
93	Blue vervain	<i>Verbena hastata</i>	fresh
94	Bermuda grass	<i>Cynodon dactylon</i>	spoil/fresh
95	Ammannia, purple	<i>Ammannia coccinea</i>	saline
96	Beach morningglory	<i>Ipomoea stonifera</i>	saline
97	Bigleaf sumpweed	<i>Iva frutescens</i>	brackish/salt
98	Black mangrove	<i>Avicennia germinans</i>	saline
99	Bush sea-oxeye	<i>Borrchia frutescens</i>	saline
100	Gulf croton	<i>Croton</i> spp.	saline
101	Maritime saltwort	<i>Batis maritima</i>	saline
102	Needlegrass rush *	<i>Juncus roemerianus</i>	saline
103	Seashore saltgrass *	<i>Distichlis spicata</i>	saline
104	Smooth cordgrass *	<i>Spartina alterniflora</i>	saline
105	Woody glasswort	<i>Salicornia virginica</i>	saline
106	Iva (Marsh Elder)	<i>Iva Frutescens</i>	brackish/saline
107	Dodder	<i>Cuscuta</i> sp.	saline
108	Seaside lavender	<i>Limonium carolinianum</i>	saline
109	Seaside goldenrod (#137) *	<i>Salidago sempervirens</i>	saline
110	Pennl smartweed	<i>Polygonum</i> spp.	saline
111	Buttercup	<i>Ranunculus</i> spp.	fresh
112	Bitterweed	<i>Helenium amarum</i>	spoil/fresh
113	Whitetop sedge (#63)	<i>Dichromena cololrata</i>	fresh
114	Hibiscus	<i>Hibiscus moscheutos</i>	fresh
115	Dandelion	<i>Taraxacum</i> spp.	spoil/fresh
116	Bacopa *	<i>Bacopa monnieri</i>	fresh
117	Panicum *	<i>P. repens</i> , <i>P. virgatum</i> , <i>P. hemitomon</i>	fresh/inter
118	Ironweed	<i>Vernonia noveboracensis</i>	fresh
119	Bitter pecan	<i>Carya</i>	fresh
120	Leafy three square *	<i>Scirpus robustus</i>	brackish
122	St. Augustine	<i>Stenotaphrum secundatum</i>	fresh
124	Dew berry vines	<i>Rubus</i> spp.	fresh

125	Salvinia	Salvinia rotundifolia	fresh
126	Bagscale	Sacciolepis striata	fresh
127	Sycamore	Platanus occidentalis	fresh
128	Water oak	Quercus nigra	fresh
129	Red maple (#91)	Acer rubrum	fresh
130	Pig weed	Amaranthus	fresh
131	Marsh St. Johns Wort	Hypericum mutilum	saline
132	Sugarcane plumegrass	Erianthus giganteus	fresh
133	Green Ash	Fraxinus pensylvanica	fresh
134	Zig Zag grass	Panicum dichotomiflorens	fresh/inter
135	Water lettuce	Pistia stratiotes	aq/fresh
136	Palmetto (#22)	Sabal minor	fresh
137	Goldenrod (#109)	Salidago sempervirens	saline
138	Black needle grass rush	Juncus roemerianus	saline
139	Open sand		saline
140	Coffee weed (#18) *	Hemp sespania	fresh

* these items also appear on the species list for vegetation identified by the Louisiana Department of Natural Resources at the Caernarvon fresh water diversion site (Chapter 5).

APPENDIX B MONTE CARLO ANALYSIS FOR MULTIPLE RESOLUTION GOODNESS OF FIT PARAMETER

In order to determine if the multiple resolution goodness of fit parameter produces an index that is statistically significant, a Monte Carlo analysis was performed. One hundred randomly generated maps were constructed with five habitat types arranged in the same boundary as the 1988 USFWS habitat map. The results of this analysis are shown graphically in Figure B.1

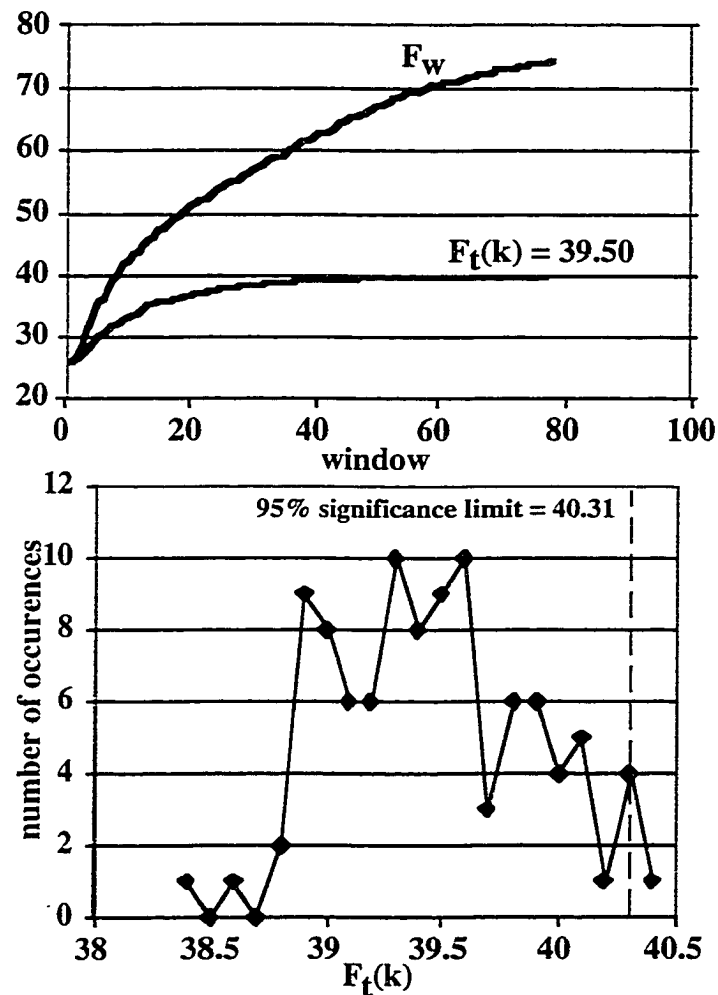


Figure B.1 Top: plot of the average F_w and summary $F_t(k)$ $k=1$ for the Monte Carlo analysis of 1988 USFWS habitat map with 100 random maps containing the same number of categories. Bottom: frequency distribution of the scores of the 100 fit calculations indicating the 95% significance level of $F_t(k) = 40.31$.

APPENDIX C SUMMARY OF LITERATURE REVIEW FOR BIOMASS PRODUCTION VALUES

The following data were compiled from the references listed at the end of the table. In cases where data were not presented numerically in the text or in tables, values were estimated from graphs.

Ref	Location	Species	water salinity (ppt)	water level (m)	above biomass (kg)	below biomass (kg)	root/shoot	sulfide (ppm)	redox Eh	oxygen (ppm)
1	Glasshouse	S.alterniflora	15	0.1	0.155	0.23	1.5	0.299	-393	0.66
1	Glasshouse	S.alterniflora	15	-0.05	0.464	0.46	1	0.00122	-289	7.4
1	Glasshouse	S.alterniflora	15	0.1	1.08	0.84	0.77	3.44E-08	-320	7.9
1	Glasshouse	S.alterniflora	15	0.1	0.34	0.37	1.09	0.00863	-191	7.9
2	Glasshouse	S.alterniflora	4	-0.13	0.52	0.54	1.04		260	
2	Glasshouse	S.alterniflora	4	-0.05	1.06	0.64	0.6		66	
2	Glasshouse	S.alterniflora	4	0.03	1.14	0.88	0.77		-150	
2	Glasshouse	S.alterniflora	4	-0.13	0.4	0.28	0.7		432	
2	Glasshouse	S.alterniflora	4	-0.05	0.84	0.64	0.76		125	
2	Glasshouse	S.alterniflora	4	0.03	0.94	0.98	1.04		101	
2	Glasshouse	S.alterniflora	4	-0.13	0.62	0.58	0.94		694	
2	Glasshouse	S.alterniflora	4	-0.05	0.52	0.36	0.69		352	
2	Glasshouse	S.alterniflora	4	0.03	0.72	0.46	0.64		-42	
2	N. Carolina	S.alterniflora			2.6	1.16	0.45		156	
2	N. Carolina	S.alterniflora			0.74	0.28	0.38		11	
2	N. Carolina	S.alterniflora			2.44	1.78	0.73		112	
2	N. Carolina	S.alterniflora			0.28	0.46	1.64		2	
2	N. Carolina	S.alterniflora		0.01	1.72	1.52	0.88		-4	
2	N. Carolina	S.alterniflora		0.01	0.64	0.52	0.81		-69	
3	Barataria Bay, LA	S.alterniflora		-0.09	1.768					
3	Barataria Bay, LA	S.alterniflora		-0.09	2.178					
3	Barataria Bay, LA	S.alterniflora		-0.09	1.562					
3	Barataria Bay, LA	S.alterniflora		-0.09	1.906					
3	Barataria Bay, LA	S.alterniflora		-0.09	1.158					
3	Barataria Bay, LA	S.alterniflora		-0.09	1.501					
3	Barataria Bay, LA	S.alterniflora		0	0.945					
3	Barataria Bay, LA	S.alterniflora		0	0.71					
3	Barataria Bay, LA	S.alterniflora		0	0.376					
3	Barataria Bay, LA	S.alterniflora		0	1.137					
3	Barataria Bay, LA	S.alterniflora		0	1.216					
3	Barataria Bay, LA	S.alterniflora		0	1.077					
4	N. Carolina	S.alterniflora	22.9	-0.3	0.243	4.878	20.07	10.0E-2.1	-389	
4	N. Carolina	S.alterniflora	18.4	-0.3	0.22	4.878	22.17		-373	
4	N. Carolina	S.alterniflora	23.3	-0.2	0.192	4.601	23.96	10.0E-1.2	-354	
4	N. Carolina	S.alterniflora	20.1	-0.2	0.309	4.601	14.89		-288	
4	N. Carolina	S.alterniflora	22.4	-0.1	0.158	4.613	29.2	10.0E-13.1	-187	
4	N. Carolina	S.alterniflora	18.6	-0.1	0.243	4.613	18.98		-78	
4	N. Carolina	S.alterniflora	18.2	0	0.374	4.521	12.09	10.0E-16.1	-102	
4	N. Carolina	S.alterniflora	12.9	0	0.372	4.521	12.15		-69	
4	N. Carolina	S.alterniflora	9.6	0.1	0.405	4.533	11.19	10.0E-0.6	-130	
4	N. Carolina	S.alterniflora	9.1	0.1	0.455	4.533	9.96		-77	
5	Glasshouse	S.alterniflora		-0.04	3.949	3.057	0.77		0	
5	Glasshouse	S.alterniflora		-0.04	3.949	3.057	0.77		49	
5	Glasshouse	S.alterniflora		-0.04	3.949	3.057	0.77		49	
5	Glasshouse	S.alterniflora		-0.04	3.949	3.057	0.77		20	
5	Glasshouse	S.alterniflora		-0.04	4.459	4.586	1.03		25	
5	Glasshouse	S.alterniflora		-0.04	4.459	4.586	1.03		-10	
5	Glasshouse	S.alterniflora		-0.04	4.459	4.586	1.03		-20	
5	Glasshouse	S.alterniflora		-0.04	4.459	4.586	1.03		-35	
5	Glasshouse	S.alterniflora		-0.04	4.076	1.911	0.47		-110	
5	Glasshouse	S.alterniflora		-0.04	4.076	1.911	0.47		-65	
5	Glasshouse	S.alterniflora		-0.04	4.076	1.911	0.47		-70	
5	Glasshouse	S.alterniflora		-0.04	4.076	1.911	0.47		-70	

Ref	Location	Species	water salinity (ppt)	water level (m)	above biomass (kg)	below biomass (kg)	root/ shoot	sulfide (ppm)	redox Eh	oxygen (ppm)
5	Glasshouse	P. hemitomon		-0.04	5.478	4.586	0.84		200	
5	Glasshouse	P. hemitomon		-0.04	5.478	4.586	0.84		140	
5	Glasshouse	P. hemitomon		-0.04	5.478	4.586	0.84		25	
5	Glasshouse	P. hemitomon		-0.04	5.478	4.586	0.84		200	
5	Glasshouse	P. hemitomon		-0.04	6.242	4.331	0.69		20	
5	Glasshouse	P. hemitomon		-0.04	6.242	4.331	0.69		20	
5	Glasshouse	P. hemitomon		-0.04	6.242	4.331	0.69		35	
5	Glasshouse	P. hemitomon		-0.04	0.637	0.382	0.6		-45	
5	Glasshouse	P. hemitomon		-0.04	0.637	0.382	0.6		-85	
5	Glasshouse	P. hemitomon		-0.04	0.637	0.382	0.6		-65	
5	Glasshouse	P. hemitomon		-0.04	0.637	0.382	0.6		-40	
6	Barataria Bay	S.alterniflora	24.5		0.696			0.015mM	60	
6	Barataria Bay	S.alterniflora	24.3		0.696			0.04mM	20	
6	Barataria Bay	S.alterniflora	27.5		0.696			0.0mM	55	
6	Barataria Bay	S.alterniflora	28		0.696			0.01mM	95	
6	Barataria Bay	S.alterniflora	21.3		0.728			0.95mM	-150	
6	Barataria Bay	S.alterniflora	22.3		0.728			0.30mM	-110	
6	Barataria Bay	S.alterniflora	23.8		0.728			0.10mM	330	
6	Barataria Bay	S.alterniflora	26.3		0.728			0.01	130	
6	Barataria Bay	S.alterniflora	24.5		0.427			0.02	60	
6	Barataria Bay	S.alterniflora	26		0.427			0.015mM	-100	
6	Barataria Bay	S.alterniflora	25.8		0.427			0.04mM	-90	
6	Barataria Bay	S.alterniflora	21.8		0.427			0.41mM	-200	
6	Barataria Bay	S.alterniflora	21.3		0.178			0.95mM	-150	
6	Barataria Bay	S.alterniflora	22.3		0.178			0.39mM	-130	
6	Barataria Bay	S.alterniflora	21.3		0.178			0.40mM	-140	
6	Barataria Bay	S.alterniflora	22		0.178			0.75mM	-190	
8	Caminada Bay, LA	S.alterniflora	19.9	-0.3	0.36	0.2	0.56	1.10mM	170	
8	Caminada Bay, LA	S.alterniflora	20	-0.3	0.35	0.2	0.57	0.70mM	168	
8	Caminada Bay, LA	S.alterniflora	19.5	-0.3	0.28	0.185	0.66	1.25mM	230	
8	Caminada Bay, LA	S.alterniflora	19.5	-0.3	0.23	0.12	0.52	1.20mM	130	
8	Caminada Bay, LA	S.alterniflora	19.3	0	0.16	0.095	0.59	1.18mM	-125	
8	Caminada Bay, LA	S.alterniflora	20.3	0	0.15	0.08	0.53	1.19mM	-130	
8	Caminada Bay, LA	S.alterniflora	19.7	0	0.15	0.09	0.6	1.10mM	-125	
8	Caminada Bay, LA	S.alterniflora	19	0	0.14	0.09	0.64	1.18mM	-105	
9	Glasshouse	S.alterniflora	0	-0.02	0.001		0.36			
9	Glasshouse	S.alterniflora	4	-0.02			0.47			
9	Glasshouse	S.alterniflora	8	-0.02			0.48			
9	Glasshouse	S.alterniflora	16	-0.02			0.5			
9	Glasshouse	S.alterniflora	32	-0.02			0.51			
9	Glasshouse	S. cynosuroides	0				0.24			
9	Glasshouse	S. cynosuroides	4				0.25			
9	Glasshouse	S. cynosuroides	8				0.3			
9	Glasshouse	S. cynosuroides	16				0.29			
9	Glasshouse	S. cynosuroides	32				0.29			
9	Glasshouse	D. spicata	0				0.25			
9	Glasshouse	D. spicata	4				0.27			
9	Glasshouse	D. spicata	8				0.28			
9	Glasshouse	D. spicata	16				0.3			
9	Glasshouse	D. spicata	32				0.32			
11	Barataria Bay, LA	S.alterniflora			1.29				220	
11	Barataria Bay, LA	S.alterniflora			1.2				215	
11	Barataria Bay, LA	S.alterniflora			1.4				275	
11	Barataria Bay, LA	S.alterniflora			1.5				40	
11	Barataria Bay, LA	S.alterniflora			1.49				175	
11	Barataria Bay, LA	S.alterniflora			1.2				-140	

Ref	Location	Species	water salinity (ppt)	water level (m)	above biomass (kg)	below biomass (kg)	root/ shoot	sulfide (ppm)	redox Eh	oxygen (ppm)
11	Barataria Bay, LA.	S.alterniflora			0.9				-100	
11	Barataria Bay, LA.	S.alterniflora			0.7				-220	
11	Barataria Bay, LA.	S.alterniflora			0.59				-45	
11	Barataria Bay, LA.	S.alterniflora			0.25				-145	
11	Barataria Bay, LA.	S.alterniflora			0.3				-175	
11	Barataria Bay, LA.	S.alterniflora			0.57				-60	
11	Barataria Bay, LA.	S.alterniflora			0.18				-190	
12	Glasshouse	S. foliosa	0				0.51			
12	Glasshouse	S. foliosa	15				0.53			
12	Glasshouse	S. foliosa	30				0.63			
12	Glasshouse	S. foliosa	40							
12	Glasshouse	Scripus robustus	0				0.65			
12	Glasshouse	Scripus robustus	15				0.56			
12	Glasshouse	Scripus robustus	30				1.21			
12	Glasshouse	Scripus robustus	45				1.92			
12	Glasshouse	Salicornia virginica	0				0.49			
12	Glasshouse	Salicornia virginica	15				0.25			
12	Glasshouse	Salicornia virginica	30				0.33			
12	Glasshouse	Salicornia virginica	45				0.33			
13	Glasshouse	S. patens	6				0.08		475	20.6(%)
13	Glasshouse	S. patens	0				0.1		475	20.6
13	Glasshouse	S. patens	6				0.07		235	1
13	Glasshouse	S. patens	0				0.06		235	1
13	Glasshouse	S. patens	6				0.08		-115	0
13	Glasshouse	S. patens	0				0.05		-115	0
15	Barataria Bay, LA.	S.alterniflora	15	0.1					180	
15	Barataria Bay, LA.	S.alterniflora	20	0					50	
16	Glasshouse	Avicennia germinans	36	0			0.83		157	
16	Glasshouse	Avicennia germinans	36	0.01			0.98		-73	
16	Glasshouse	Avicennia germinans	36	0.15			0.5		-39	
16	Glasshouse	Avicennia germinans	36	0.01			1.09		-143	
16	Glasshouse	Avicennia germinans	36	0			0.92		159	
16	Glasshouse	Avicennia germinans	36	0.01			1.18		-26	
16	Glasshouse	Avicennia germinans	36	0.15			1.08		-66	
16	Glasshouse	Avicennia germinans	36	0.01			1.68		-167	
17	N. Carolina	S.alterniflora	27		1.414					
17	N. Carolina	S.alterniflora	27		0.815					
17	N. Carolina	S.alterniflora	27		1.13	2.121	1.88			
17	N. Carolina	S.alterniflora	27		0.915	2.311	2.53			
17	N. Carolina	S.alterniflora	27		0.701	2.204	3.14			
17	N. Carolina	S.alterniflora	27		0.799	2.232	2.79			
17	N. Carolina	S.alterniflora	27		0.827	2.19	2.65			
17	N. Carolina	S.alterniflora	27		0.719	2.067	2.87			
17	N. Carolina	S.alterniflora	27		0.886	2.795	3.15			
17	N. Carolina	S.alterniflora	27		0.898	2.256	2.51			
17	N. Carolina	S.alterniflora	27		0.748	2.796	3.74			
17	N. Carolina	S.alterniflora	27		1.124	1.858	1.65			
17	N. Carolina	S.alterniflora	27		0.873	1.876	2.15			
17	N. Carolina	S.alterniflora	27		0.6	1.5	2.5			
17	N. Carolina	S.alterniflora	27		0.537	1.437	2.68			
17	N. Carolina	S.alterniflora	27		0.931	1.586	1.7			
17	N. Carolina	S.alterniflora	27		0.52	1.997	3.84			
17	N. Carolina	S.alterniflora	27		0.663	1.819	2.74			
17	N. Carolina	S.alterniflora	27		1.033	2.446	2.37			

Ref	Location	Species	water salinity (ppt)	water level (m)	above biomass (kg)	below biomass (kg)	root/ shoot	sulfide (ppm)	redox Eh	oxygen (ppm)
18	Glasshouse	S.alterniflora					14.8			
18	Glasshouse	S.alterniflora					15.8			
18	Glasshouse	S.alterniflora					7.1			
18	Glasshouse	S.alterniflora					18.1			
18	Glasshouse	S.alterniflora					15.7			
18	Glasshouse	S.alterniflora					13.1			
19	Glasshouse	S.alterniflora	24		0.112	0.143	1.28			
19	Glasshouse	S.alterniflora	24		1.131	0.773	0.68			
19	Glasshouse	S.alterniflora	24		3.056	1.614	0.53			
19	Glasshouse	D.spicata	12		0.124	0.11	0.89			
19	Glasshouse	D.spicata	12		0.749	0.298	0.4			
19	Glasshouse	D.spicata	12		1.466	0.512	0.35			
19	Glasshouse	S.foliosa	24		0.036	0.109	3.03			
19	Glasshouse	S.foliosa	24		0.083	0.112	1.35			
19	Glasshouse	S.foliosa	24		0.39	0.353	0.91			
19	Glasshouse	S.patens	12		0.227	0.146	0.64			
19	Glasshouse	S.patens	12		0.171	0.112	0.65			
19	Glasshouse	S.patens	12		1.227	0.477	0.39			
20	Bayou Rigolettes	P.hemitomon	2.1	0.1					30	
20	Bayou Rigolettes	P.hemitomon	2.2	0					147	
20	Bayou Rigolettes	P.hemitomon	2.2	0					24	
20	Bayou Rigolettes	S.Lancifolia	2.1	0.1					30	
20	Bayou Rigolettes	S.Lancifolia	2.2	0					147	
20	Bayou Rigolettes	S.Lancifolia	2.2	0					24	
20	Bayou Rigolettes	L.oryzoides	2.1	0.1					30	
20	Bayou Rigolettes	L.oryzoides	2.2	0					147	
20	Bayou Rigolettes	L.oryzoides	2.2	0					24	
20	Bayou Rigolettes	P.dichotomflorum	12.3	0.1					14	
20	Bayou Rigolettes	P.dichotomflorum	12.2	0					127	
20	Bayou Rigolettes	P.dichotomflorum	12	-0.1					292	
20	Glasshouse	P.hemitomon	0	-0.1					197	
20	Glasshouse	P.hemitomon	0	0					-23	
20	Glasshouse	P.hemitomon	0	0.1					-34	
20	Glasshouse	P.hemitomon	1.2	-0.1					136	
20	Glasshouse	P.hemitomon	1.2	0					10	
20	Glasshouse	P.hemitomon	1.2	0.1					-15	
20	Glasshouse	P.hemitomon	2.4	-0.1					48	
20	Glasshouse	P.hemitomon	2.4	0					-119	
20	Glasshouse	P.hemitomon	2.4	0.1					-47	
20	Glasshouse	P.hemitomon	4.8	-0.1					155	
20	Glasshouse	P.hemitomon	4.8	0					-23	
20	Glasshouse	P.hemitomon	4.8	0.1					-127	
20	Glasshouse	P.hemitomon	9.4	-0.1					254	
20	Glasshouse	P.hemitomon	9.4	0					-89	
20	Glasshouse	P.hemitomon	9.4	0.1					-132	
21	Glasshouse	S.alterniflora								0
21	Glasshouse	S.alterniflora								2
21	Glasshouse	S.alterniflora								4
21	Glasshouse	S.alterniflora								6
21	Glasshouse	S.alterniflora								8
21	Glasshouse	S.alterniflora								10
21	Glasshouse	S.alterniflora								12
22	Glasshouse	S.lancifolia	0	0.01	3.241					
22	Glasshouse	S.lancifolia	0	0.01	3.602					
22	Glasshouse	S.lancifolia	0	0.01	1.801					
22	Glasshouse	S.lancifolia	6	0.01	1.981					
22	Glasshouse	S.lancifolia	6	0.01	1.585					
22	Glasshouse	S.lancifolia	6	0.01	0.648					
22	Glasshouse	S.lancifolia	12	0.01	0.72					
22	Glasshouse	S.lancifolia	12	0.01	0.36					
22	Glasshouse	S.lancifolia	12	0.01	0.18					
22	Glasshouse	S.lancifolia	0	0.01	3.241	3.061	0.94			
22	Glasshouse	S.lancifolia	6	0.01	2.341	2.701	1.15			
22	Glasshouse	S.lancifolia	6	0.01	1.621	1.261	0.78			
22	Glasshouse	S.lancifolia	12	0.01	1.441	1.441	1			
22	Glasshouse	S.lancifolia	12	0.01	0.18	1.261	7			
22	Glasshouse	S.lancifolia	0	0.01	3.602	2.881	0.8			

Ref	Location	Species	water salinity (ppt)	water level (m)	above biomass (kg)	below biomass (kg)	root/ shoot	sulfide (ppm)	redox Eh	oxygen (ppm)
22	Glasshouse	<i>S.lancifolia</i>	6	0.01	1.873	2.989	1.6			
22	Glasshouse	<i>S.lancifolia</i>	6	0.01	1.441	2.161	1.5			
22	Glasshouse	<i>S.lancifolia</i>	12	0.01	0.648	0.972	1.5			
22	Glasshouse	<i>S.lancifolia</i>	12	0.01	0.144	2.017	14			
22	Glasshouse	<i>S.lancifolia</i>	0	0.01	2.125	5.799	2.73			
22	Glasshouse	<i>S.lancifolia</i>	6	0.01	0.612	2.341	3.82			
22	Glasshouse	<i>S.lancifolia</i>	6	0.01	0.72	2.701	3.75			
22	Glasshouse	<i>S.lancifolia</i>	12	0.01	0.018	2.431	135			
22	Glasshouse	<i>S.lancifolia</i>	12	0.01	0.036	1.405	39			
24	Jean Lafitte, LA	<i>S.lancifolia</i>	1.212		1.766	6.249	1.56	1.9	-50	
24	Jean Lafitte, LA	<i>S.lancifolia</i>	1.126		1.236	4.862	1.12	2.2	-70	
24	Jean Lafitte, LA	<i>S.lancifolia</i>	1.181	0.075	1.176	4.707	1.16	4.8	-110	
24	Jean Lafitte, LA	<i>S.lancifolia</i>	1.25	0.15	1.351	5.474	1.28	5.9	-130	
25	Pearl River, MS.	<i>Panicum virgatum</i>	0		0.771					
25	Pearl River, MS.	<i>Aster subulatus</i>	0		0.032					
25	Pearl River, MS.	<i>Spartina</i>	0		0.381					
		<i>synosuroides</i>								
25	Pearl River, MS.	<i>Vigna luteola</i>	0		0.023					
25	Pearl River, MS.	<i>S.patens</i>	4		0.501					
25	Pearl River, MS.	<i>Mikania scandens</i>	4		0.168					
25	Pearl River, MS.	<i>Pnadium virgatum</i>	4		0.011					
25	Pearl River, MS.	<i>S.lancifolia</i>	4		0.108					
25	Pearl River, MS.	<i>Vigna luteola</i>	4		0.037					
25	Pearl River, MS.	<i>S.alterniflora</i>	6		0.993					
25	Pearl River, MS.	<i>Panicum virgatum</i>	0		0.517					
25	Pearl River, MS.	<i>Aster subulatus</i>	0		0.011					
25	Pearl River, MS.	<i>Spartina</i>	0		0.355					
		<i>synosuroides</i>								
25	Pearl River, MS.	<i>Vigna luteola</i>	0		0.032					
25	Pearl River, MS.	<i>S.patens</i>	4		0.29					
25	Pearl River, MS.	<i>Mikania scandens</i>	4		0.096					
25	Pearl River, MS.	<i>Pnadium virgatum</i>	4		0.059					
25	Pearl River, MS.	<i>S.lancifolia</i>	4		0.12					
25	Pearl River, MS.	<i>Vigna luteola</i>	4		0.065					
25	Pearl River, MS.	<i>S.alterniflora</i>	6		0.713					
28	California								60	6
28	California								150	12
28	California								210	12
28	California								240	20
28	California								400	20.5
28	California								400	21
28	California								400	21
28	California								400	21
28	California								400	22
28	California								400	22
28	California								375	22
28	California								175	8
28	California								180	11
28	California								220	10
28	California								250	18
28	California								450	20
28	California								420	21
28	California								420	21
28	California								420	21
28	California								400	20
28	California								375	17
28	California								125	6
28	California								100	3
28	California								20	1
28	California								-40	1
28	California								400	19
28	California								320	20
28	California								350	20
28	California								310	21
28	California								320	20
28	California								150	18
28	California								20	8
28	California								-45	1
28	California								-60	1.5

Ref	Location	Species	water salinity (ppt)	water level (m)	above biomass (kg)	below biomass (kg)	root/ shoot	sulfide (ppm)	redox Eh	oxygen (ppm)
28	California								-100	0
28	California								-100	0.5
28	California								-120	2
28	California								-120	2
28	California								-120	0
28	California								-120	1
28	California								-75	1
29	Georgia	S.alterniflora	20							
29	Georgia	S.alterniflora	20							
29	Georgia	S.alterniflora	22							
29	Georgia	S.alterniflora	27							
29	Georgia	S.alterniflora	29							
29	Georgia	S.alterniflora	31							
29	Georgia	S.alterniflora	35							
29	Georgia	S.alterniflora	35.5							
29	Georgia	S.alterniflora	36							
29	Georgia	S.alterniflora	36.5							
29	Georgia	S.alterniflora	39.5							
29	Georgia	S.alterniflora	40							
29	Georgia	S.alterniflora	43							
30	Louisiana			-0.42					300	175
30	Louisiana			-0.2					400	140
30	Louisiana			0					20	175
30	Louisiana			0					80	180
30	Louisiana			0					100	120
30	Louisiana			0.1					100	110
30	Louisiana			0.1					100	50
30	Louisiana			0.1					100	40
30	Louisiana			0.1					100	0
30	Louisiana			-1.2					500	200
30	Louisiana			-1.2					520	200
30	Louisiana			-1.2					600	200
30	Louisiana			-1.2					600	200
30	Louisiana			-0.8					450	180
30	Louisiana			-0.2					80	135
30	Louisiana			-0.1					20	100
30	Louisiana			0					100	25
30	Louisiana			0.1					100	30
30	Louisiana			0					90	20
30	Louisiana			-0.2					110	60
30	Louisiana			-0.8					80	180
30	Louisiana			-0.8					350	200
30	Louisiana			-0.6					400	175
30	Louisiana			-1.2					380	200
30	Louisiana			0					100	50
30	Louisiana			0.1					80	10
30	Louisiana			0.1					220	10
30	Louisiana			0.15					220	10
30	Louisiana			0.2					230	0
30	Louisiana			0.15					220	10
30	Louisiana			0.1					250	0
30	Louisiana			0					230	10
30	Louisiana			-0.6					580	180
30	Louisiana			-0.8					250	200
30	Louisiana			-0.5					350	170
30	Louisiana			-0.8					380	175
30	Louisiana			-0.6					650	175
30	Louisiana			0.1					100	130
30	Louisiana			0.8					0	120
30	Louisiana			0.2					80	25
30	Louisiana			0.2					20	35
30	Louisiana			0.4					200	40
30	Louisiana			1.2					200	50
30	Louisiana			-0.3					0	10
30	Louisiana			-0.4					0	10
30	Louisiana			-0.4					40	10
30	Louisiana			0					120	100

Ref	Location	Species	water salinity (ppt)	water level (m)	above biomass (kg)	below biomass (kg)	root/ shoot	sulfide (ppm)	redox Eh	oxygen (ppm)
30	Louisiana			0					500	50
30	Louisiana			0					60	40
30	Louisiana			0.1					80	20
30	Louisiana			0.2					40	20
30	Louisiana			0.2					50	40
30	Louisiana			0.2					100	10
30	Louisiana			0.3					80	20
30	Louisiana			-0.3					60	20
30	Louisiana			-0.2					120	20
30	Louisiana			-0.4					100	0
30	Louisiana			0.2					-200	0
30	Louisiana			0.1					200	0
30	Louisiana			0.1					0	0
30	Louisiana			0.1					0	0
30	Louisiana			0.05					200	0
30	Louisiana			0.1					200	0
30	Louisiana			-0.4					160	0
30	Louisiana			-0.8					120	140
30	Louisiana			-1					400	150
31	Glasshouse	<i>S.alterniflora</i>		0.05						
32	Louisiana	<i>S.alterniflora</i>			1018					
32	Louisiana	<i>S.alterniflora</i>			788					
33	Louisiana	<i>S. lancifolia</i>		0	0.857	1.24				
33	Louisiana	<i>S. lancifolia</i>		0	0.755	0.965				
33	Louisiana	<i>S. lancifolia</i>		0.075	0.619	0.934				
33	Louisiana	<i>S. lancifolia</i>		0.15	0.759	1.086				
34	Glasshouse	<i>S.patens</i>	0		0.111					
34	Glasshouse	<i>S.patens</i>	5		0.116					
34	Glasshouse	<i>S.patens</i>	10		0.102					
34	Glasshouse	<i>S.patens</i>	15		0.089					
34	Glasshouse	<i>S.patens</i>	20		0.084					
34	Glasshouse	<i>S.patens</i>		-0.1	0.133					
34	Glasshouse	<i>S.patens</i>		0.1	0.122					
34	Glasshouse	<i>S.patens</i>		0.3	0.036					
34	Louisiana		12.4	0					343	
34	Louisiana		12.4	0.1					-116	
34	Louisiana		4						-87	
34	Louisiana		6.9						-79	
34	Louisiana		11						-107	
34	Louisiana		14.3						-104	
34	Louisiana		17.6						-103	
35	Massachusetts	<i>S.alternaflora</i>			0.2	1.524			-168	
35	Massachusetts	<i>S.alternaflora</i>			0.2	1.524			-172	
35	Massachusetts	<i>S.alternaflora</i>			0.304	1.697			-98	
35	Massachusetts	<i>S.alternaflora</i>			0.304	1.697			-127	
35	Massachusetts	<i>S.alternaflora</i>			0.37	1.527			28	
35	Massachusetts	<i>S.alternaflora</i>			0.37	1.527			-47	
36	Louisiana	<i>S.patens</i>	2.2		1.85					
36	Louisiana	<i>S.patens</i>	6		2.2					
36	Louisiana	<i>S.patens</i>	1.7		2.8					
36	Louisiana	<i>S.patens</i>	4.5		2.2					
37	Glasshouse	<i>S.alternaflora</i>	0	0	1.16					
37	Glasshouse	<i>S.alternaflora</i>	0	0.05	0.74					

Ref #	Reference
1	(Linthurst, 1979)
2	(Mendelssohn et al., 1980)
3	(DeLaune et al., 1979)
4	(Linthurst et al., 1980)
5	(Koch et al., 1989)
6	(Mendelssohn et al., 1988)
8	(Wilsey et al., 1992)
9	(Parrondo et al., 1978)
10	(Pezeshki et al., 1987b)
11	(Mendelssohn et al., 1981)
12	(Pearcy et al., 1984)
13	(Bandyopadhyay et al., 1993)
14	(Flanagan et al., 1988)
15	(DeLaune et al., 1983)
16	(McKee, 1993)
17	(Broome et al., 1986)
18	(Pezeshki et al., 1995)
19	(Smart et al., 1978)
20	(McKee et al., 1989)
21	(Morris et al., 1984a)
22	(Howard et al., 1993)
28	(Josselyn et al., 1990)
29	(Nestler, 1977)
30	(Faulkner et al., 1992)
31	(Mendelssohn et al., 1992)
32	(Kirby et al., 1976)
33	(Howard et al., 1995)
34	(Broome et al., 1995)
35	(Arenovski et al., 1992)
36	(Cramer et al., 1981)
37	(Portnoy et al., 1997)

VITA

Mary Louise White received her bachelor of science degree in 1971 from the College of St. Catherine, St. Paul, Minnesota. Her major field of study was physics and her minor was in mathematics and education. She was awarded the master of science degree in 1977 from Louisiana State University in Baton Rouge, Louisiana. For that degree her major field of study was marine science and her minor field of study was computer science. She has worked as a research associate at Louisiana State University in the Coastal Studies Institute and Coastal Ecology Institute. She currently is employed as an ecologist at the United States Environmental Protection Agency as a staff member on the Critical Ecosystem Team. She will receive the degree of Doctor of Philosophy in May, 1999.

DOCTORAL EXAMINATION AND DISSERTATION REPORT

Candidate: Mary Louise White

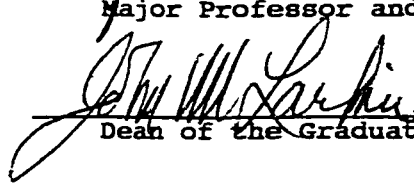
Major Field: Oceanography and Coastal Sciences

Title of Dissertation: Spatial Modeling of Coastal Landscapes:
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Approved:

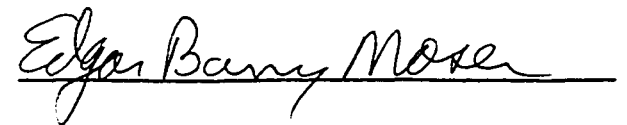
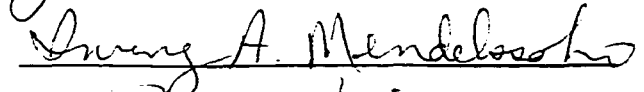
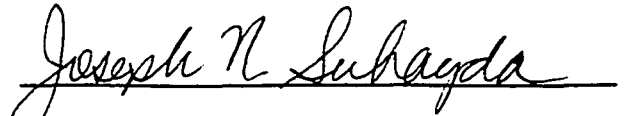


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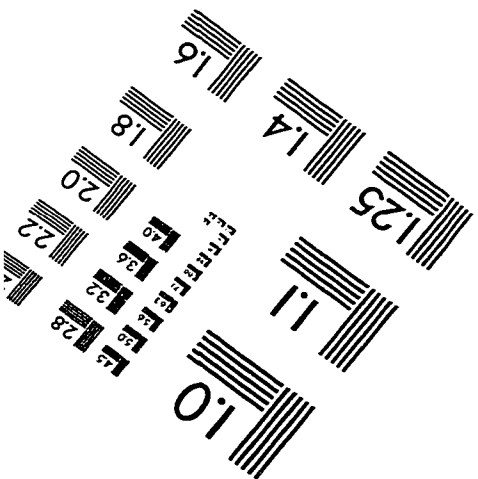
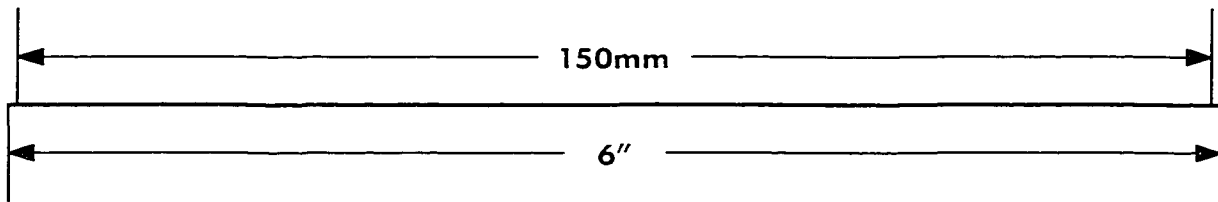
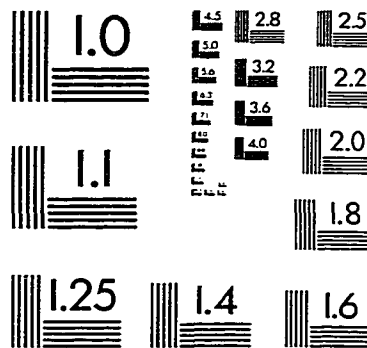
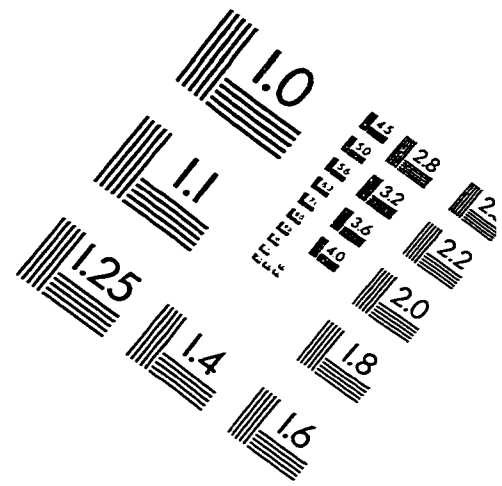
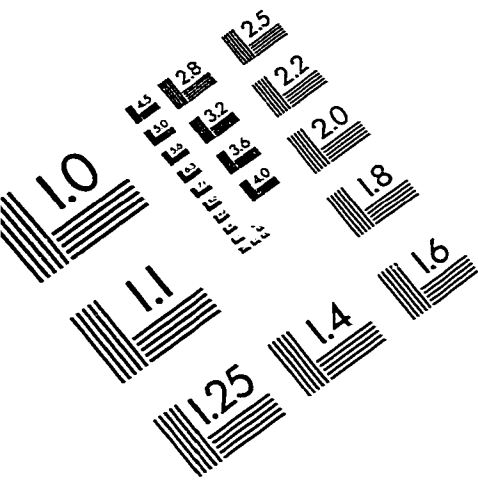
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