# LANDSCAPE MODELING OF COASTAL HABITAT CHANGE IN THE MISSISSIPPI DELTA

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*Abstract.* A landscape model was developed to investigate and predict the environmental factors affecting wetland habitat change within the Barataria and Terrebonne basins of coastal Louisiana, USA. The model linked an overland-flooding hydrodynamic module, using cells of 100 km<sup>2</sup> in size and operating at a 1-h time step, and a spatially articulated ecosystem module, resolving habitat type and change for 1-km<sup>2</sup> cells in daily time steps. Integration across different temporal and spatial scales was accomplished with interpolation routines and averaging algorithms. Forcing functions included dominant regional processes, such as subsidence, sedimentation, and sea-level rise. Hydrologic functions were calibrated against existing climate and hydrologic time series, while habitat information was compared to maps prepared by the United States Fish and Wildlife Service (USFWS) for 1978 and 1988.

Spatial calibration was done by initializing the landscape pattern of the model to a 1978 USFWS habitat map. After a 10-yr simulation, the results were compared against a 1988 USFWS habitat map. Simulated maps had an accuracy of 85–90 (out of a maximum of 100), based on a multiple resolution fit algorithm. For validation, the model was initialized with a 1956 USFWS habitat map, and the results from a 32-yr simulation were compared to the 1988 USFWS habitat map. The landscape model produced reasonable regional agreement, despite the fact that small-scale processes and features were not included. The validation runs produced land-loss rates that matched historical trends with an accuracy fit above 75.

The model simulated 30 years into the future, starting in 1988, testing for long-term climate variability under diverse scenarios. Results indicated that weather variability impacts land-loss rates more than replication of extreme weather years. Even when extreme dry and wet years were repeated, the model predicted lower land loss when compared to historical records. This is indicative of the ability of the simulated plant communities to adapt to repetitive climatic forcing functions.

Key words: Barataria Basin (Louisiana, USA); coastal habitat change; deltaic habitats, response to multiple impacts; landscape model, coastal; Mississippi River Delta (USA); spatial modeling; Terrebonne Basin (Louisiana, USA); watersheds.

#### INTRODUCTION

Coastal habitats in the Mississippi delta (USA) are changing at unprecedented rates with displacement of freshwater vegetation by more salinity-tolerant communities and massive wetland loss resulting in conversion to open water. Within coastal Louisiana, wetland loss rates have ranged from 73 to 102 km<sup>2</sup>/yr (Gagliano et al. 1981). The Barataria–Terrebonne Estuarine Complex (Fig. 1) has the highest wetland loss rates along the Louisiana coastline where, under present conditions, coastal wetland life expectancies range from

Manuscript received 8 September 1998; revised 23 July 1999; accepted 25 July 1999.

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<sup>6</sup> Present address: U.S. EPA Region 5, Mail stop B-19J, 77 West Jackson Boulevard, Chicago, Illinois 60604 USA. 50 to 200 yr (Gagliano et al. 1981). The general pattern of habitat change over the past 60 yr has been attributed to the interaction among several regional factors (Deegan et al. 1984, Walker et al. 1987, Wells 1996, Roberts 1997) including: (1) deltaic lobe abandonment, (2) eustatic sea-level rise and subsidence, (3) changes in the introduction of freshwater and sediments from the Mississippi and Atchafalaya Rivers, and (4) human modification of internal hydrology (Day and Templet 1989, Baumann and Turner 1990, Boesch et al. 1994, Reed 1995, Day et al. 1997, Roberts 1997, Turner 1997). However, these regional factors have been analyzed mostly at small scale (Salinas et al. 1986, Dozier et al. 1983, Deegan et al. 1984, Nyman et al. 1993). In this paper, we use a landscape model to examine the interactions of the cumulative regional impacts of these factors and variations in weather. We combined geo-



FIG. 1. The State of Louisiana (USA) showing location for the Barataria and Terrebonne watersheds. Numbers indicate time-series stations.

logical, hydrological, and meteorological forces to understand the forcing of this climate and its variability on a long-term basis. The deltaic regional factors, such as increased sediment delivery to wetlands and increased marsh flooding with sea-level rise, were mechanistically incorporated into the model.

Wetland elevation gain must equal relative sea-level rise (RSLR) to achieve long-term marsh stability (Chmura et al. 1992, Cahoon et al. 1995). Over the past 6000–7000 yr the Mississippi Deltaic Plain was formed by a series of overlapping delta lobes related to shifts in the course of the river (Penland et al. 1988, Roberts 1997). In abandoned deltaic lobes the land surface loses elevation relative to mean sea level due to compaction and consolidation of sediments (Baumann et al. 1984, Wells 1996). Sea level varies both interannually and long term. Interannual variability can be as high as 10 cm/yr due to changes in circulation patterns in the northern Gulf of Mexico (GoM), variations in river discharge, winter storms, and presence of tropical storms and hurricanes (Penland and Ramsey 1990). Long-term tide readings have shown that RSLR, which accounts for subsidence and eustatic sea-level rise, has averaged about 1.2 cm/yr over the last 30 yr (Penland and Ramsey 1990, Cahoon et al. 1995).

There have been extensive spatial and temporal changes in the hydrology of Mississippi Deltaic Plain. Construction of flood-control levees has isolated most wetlands from the river since the early 1900s (Mossa 1996), eliminating seasonal flooding and the introduction of sediments and nutrients. Currently, the primary source of sediments in most of the delta is resuspended sediment from bay bottoms and the nearshore GoM (Hatton et al. 1983, Baumann et al. 1984). This contribution has not been sufficient to offset RSLR in most of the coastal zone, leading to increased flooding du-

TABLE 1. Area of wetland and aquatic habitats for Barataria and Terrebonne Basins (USA) in 1956, 1978, and 1988 (Reed 1995). Habitat areas are based on U.S. Fish and Wildlife Service maps.

	Area (km <sup>2</sup> )							
	Barataria			Terrebonne				
Habitat	1956	1978	1988	1956	1978	1988		
Swamp Upland All marsh Water	306 256 2137 1735	324 363 1570 2178	319 350 1243 2523	293 83 2769 2458	352 104 2142 2999	316 117 1849 3323		
Total land loss†		442	345		547	316		

<sup>†</sup> Total land loss is the difference of all terrestrial habitats of one period minus the same summation for the following sample year.

ration and vegetation death (Mendelssohn et al. 1981). Recently, river diversions have been proposed and constructed to restore riverine inputs to preserve and maintain areas of deltaic wetlands (Boesch et al. 1994, Day et al. 1997).

Wetland hydrology has been highly modified by the widespread construction of dredged canals and associated spoilbanks (Turner 1997). Spoilbanks decrease the input of materials to adjacent wetlands and make these areas prone to excessive inundation (Swenson and Turner 1987, Boumans and Day 1993, Reed et al. 1997). Canal dredging increases wetland vulnerability to erosion, as shown by several studies relating canal density to wetland loss (Scaife et al. 1983, Bass and Turner 1997, Turner 1997). Additionally, losses have been caused directly by canal dredging and spoil placement. Such human impacts have had a significant impact on land loss. For example, direct land loss due to canal dredging accounted for more than 30% and 10% of the losses in Barataria and Terrebonne basins, respectively, between 1956 and 1990, with indirect effects (e.g., changes in hydroperiod, edge erosion) estimated to cause twice the land loss (Table 1; Reed 1995).

Modeling wetland habitat at the landscape level allows testing hypotheses about the dynamics of wetland loss, and is critical for understanding long-term effects of proposed restoration alternatives. This was previously done using spatially articulated landscape models for portions of Terrebonne (Sklar et al. 1985, Costanza et al. 1990, White 1991) and Barataria basins (White et al. 1991). These models are known generically as "coastal ecological landscape spatial simulations" (CELSS), and were defined by Sklar and Costanza (1991) as dynamic spatial interaction models with feedback. They incorporated location-specific algorithms to quantify influences from adjacent cells, such as the existence of sub-grid-scale channels or levees affecting material exchange (Boumans and Sklar 1990, Fitz et al. 1996). The explicit feedback between processes and the landscape allowed both the landscape and the intensity of the effecting processes to change throughout time. These models used a water-balance approach to hydraulic processes. Algorithms incorporating this type of feedback have been used in aquatic modeling programs, such as LAPTER (Reyes et al. 1994), as well as in terrestrial simulation programs such as PATCH-MOD (Wu and Levin 1994), ECOLECON (Liu et al. 1994), and the Frankfurt Biosphere Model (Kindermann et al. 1996).

The objective of this study was to construct a landscape simulation model to predict habitat change in the Mississippi delta for a 30-yr time scale. Using some of the CELSS framework, we incorporated a hydrodynamic module and much enlarged study area (Sklar et al. 1985, Costanza et al. 1990, White 1991), as well as a revision of previous biological algorithms. We investigated the impacts of sea-level changes, Mississippi River discharge, and climate variability on habitat change rates in the Barataria and Terrebonne basins. The Barataria–Terrebonne ecological landscape spatial simulation (BTELSS) model replicated historical trends in land loss and habitat change from 1956 to 1988 for each basin, and then was used to predict trends into the future.

## STUDY AREA

The Barataria–Terrebonne estuarine system includes two of the interdistributary estuarine wetland systems of the Mississippi delta (USA) and is bounded by the Mississippi and Atchafalaya rivers (Fig. 1). The Atchafalaya carries ~ 30% of the total Mississippi River flow. The Barataria Basin is located between the natural levees of the Mississippi River and Bayou Lafourche, encompassing ~ 4100 km<sup>2</sup> of wetlands and waterbodies. The Terrebonne Basin lies to the west of Barataria Basin and occupies ~ 5300 km<sup>2</sup> (Fig. 1). Vegetation zones range from freshwater forested wetlands to fresh, brackish, and salt marsh communities (Chabreck 1972, Chabreck and Condrey 1979).

The Barataria Basin has been closed to direct river inflow since the leveeing of the Mississippi River and the closing of the Bayou Lafourche–Mississippi River connection in 1904. The Mississippi River exerts considerable influence on the lower part of the basin through its effect on salinity in the nearshore Gulf of Mexico (GoM; Walker 1996). Precipitation is the main source of fresh water for the basin; however, a small amount of river water enters the basin through the Gulf Intracoastal Waterway. Additional freshwater enters the basin through water runoff that is pumped from agricultural and urban areas.

In contrast, the Terrebonne Basin is strongly influenced by the fresh water from the Atchafalaya River. The western portion of this basin is one of the few locations in southern Louisiana that has experienced net land gain, with the growth of the Atchafalaya Delta (Adams et al. 1976, Adams and Baumann 1980, Roberts 1997). The complex interactions between the fresh 2334



FIG. 2. Flow of calculations indicating temporal and spatial scales for the BTELSS (Barataria–Terrebonne ecological landscape spatial simulation) model.

water from the Mississippi and Atchafalaya rivers and the saline waters of the GoM are controlled by tides, frontal passages, seasonal sea-level variation, longterm sea-level changes, and shelf topography (Penland et al. 1988, Wiseman et al. 1990, Paille 1997).

# MODEL ORGANIZATION

The BTELSS (Barataria–Terrebonne estuarine landscape spatial simulation) model linked a hydrodynamics, a plant-production, and a soil-dynamics landscape module. The code utilized the same kind of input files and maps for both basins, allowing the use of the same conceptual framework, modules, and algorithms (Fig. 2). The results of the hydrodynamic and productivity modules were linked to the soil module, and then evaluated with a habitat-switching module that allowed the landscape to evolve on a biannual basis.

# Hydrodynamic module

The two-dimensioned, vertically integrated hydrodynamic module used a finite difference scheme with a time step of 1 h and a spatial cell size of 100 km<sup>2</sup>. Freshwater inputs included rainfall, pumping from developed areas, and riverine inputs. Tidal boundary exchanges were a source or sink for salt water. Water was also lost by evaporation, whereas infiltration into groundwater was assumed negligible. The effect of wind stress on water flow was modeled with an exponential relationship (Yeh and Chou 1979) as described in the Appendix.

The effect of friction on water flow was modeled using the standard Manning equation (Martin and Mc-Cutcheon 1999), where the Manning coefficient was a 100-km<sup>2</sup> average of a 1-km<sup>2</sup> habitat-type-dependent Manning coefficient. This 1-km<sup>2</sup> Manning coefficient depended on the different habitats, and was based on their resistance to flow following both published literature values (Wolanski et al. 1980, Burke and Stolzenbach 1983, Wolanski et al. 1992) and knowledge of the hydraulic characteristics of each habitat. For example, swamps were more resistant to flow than grassy marshes (Freeze and Cherry 1979). Salt marshes had a lower Manning value because they are closest to the coast, more dissected by tidal channels than brackish and fresh marshes, and experience more tidal energy.

Standard hydrodynamic equations required a smaller time step than was practical for prediction of long-term effects. Therefore, the model used the diffusion wave approximation of shallow-water equations in the prediction of long-term water level (Singh and Aravamuthan 1995). This approximation considered local acceleration, uniform flow, and Coriolis force to be negligible. This solution has been used in other modeling efforts linking diverse scales (Baskin 1993, USFWS 1995). Full formulation of the finite difference equations (Casulli 1992) is presented in the Appendix.

A 100-km<sup>2</sup> average of land elevation and Manning coefficients was used as input to calculate water height and velocity. These resultant 100-km<sup>2</sup> distributions were then interpolated to 1 km<sup>2</sup> using a binomial interpolation routine (Press 1992). The interpolated water height and land-elevation values were then used to calculate water depth. Spatially distributed suspended sediment and salinity were computed using the same 1-km<sup>2</sup> interpolation algorithms.

Hourly water velocity and sediment concentration determined how much deposited sediment was accumulated each day. Once the 1-km<sup>2</sup> salinity and duration of flooding were computed, these were averaged to daily values. Suspended sediments were distributed in the same way as salinity with two differences: an empirical function (Chmura et al. 1992) was used to resuspend sediments by wind-focused wave action in the bays, and sediments were lost from the water column by deposition on marsh surfaces based on settling velocity.

	Mimima		Maxima			
Habitat type	Salinity	Biomass (kg OM/m <sup>2</sup> )	Salinity	Biomass (kg OM/m <sup>2</sup> )		
Freshwater marsh	0.0	0.9	<4.5	4.6		
Swamp	0.0	20.3	9.0	45.2		
Brackish marsh	4.5	0.4	<12.5	2.2		
Saltwater marsh	12.5	1.2	40.0	6.0		

TABLE 2. Habitat-type definition by salinity and biomass for the habitat-switching algorithm.

*Notes:* Salinity is reported in terms of the practical salinity scale, a conductivity ratio. For biomass, OM = organic matter (dry mass). The open-water habitat type resulted when any habitat was inundated for 24 h each day and biomass was less than minimum.

#### Productivity module

The biological module simulated net productivity of the macrophyte community for each 1-km<sup>2</sup> cell on a daily time step. Ecological parameters, such as production and respiration rates, varied according to the cell wetland type. The vegetative communities in each basin reflected the present gradients in elevation, salinity, and soil type. Each wetland type was characterized with algorithms representative of a single dominant species with well-known responses to salinity and flood duration. Salt marshes were characterized by Spartina alterniflora, brackish marshes by Spartina patens, fresh marshes by Panicum hemitomon, and forested wetlands by Taxodium distichium (Chabreck 1972, Conner et al. 1987, Tiner 1993, Visser et al. 1996). Equations of the productivity module are presented in the Appendix.

Macrophytes were modeled with two state variables: belowground and aboveground biomass (in kilograms of organic matter [OM] per square meter, kg OM/m<sup>2</sup>). Gross production was a function of aboveground biomass, maximum gross production rate (kg  $OM \cdot m^{-2} \cdot d^{-1}$ ), and a limitation function. This limiting factor integrated functional responses to water level, salinity, and temperature as a factor ranging from 0 to 1, depending on the synergistic effect of the total environmental conditions (Phipps 1979, Hopkinson et al. 1988, Mitsch 1988). Salinity stress was determined by plant tolerances, depending on habitat types (Howes et al. 1986, Pezeshki et al. 1987). Waterlogging constrained the rate of growth representing different habitat-type tolerances to flooding conditions. Respiration rates were held constant (Pomeroy et al. 1976, Cronk and Mitsch 1994, Dai and Wiegert 1996). Respiration and mortality were differentiated for aboveground biomass and belowground biomass (Pomeroy et al. 1976). Excess fixed carbon calculated from the aboveground biomass was translocated to the belowground storage (Gosselink and Kirby 1974, Howes et al. 1985).

## Soil module

The soil module included cumulative storage of inorganic sediments (in kilograms per square meter) and dead belowground organic sediments. Live belowground biomass from the productivity module was added to dead organic sediments for total belowground organic sediments. Inorganic and organic components were divided by bulk density (2.65 and 1.14 g/cm<sup>3</sup> for inorganic and organic, respectively) and pore-space volume set at 90% (Nyman et al. 1990), and summed to calculate marsh elevation (in meters). While inorganic sediments were conserved, decomposition, calculated as the storage multiplied by a decompositionrate constant, was lost from belowground organic sediments.

Relative sea-level rise (RSLR) has proven a critical factor in determining relative marsh elevation and habitat changes in the Louisiana coast (Day and Templet 1989, Cahoon 1994, Day et al. 1997). RSLR in Louisiana averages about 1.2 cm/yr with subsidence accounting for 0.84 cm/yr (Swanson and Thurlow 1973, Trahn 1982, Penland and Ramsey 1990). While subsidence was not explicitly included in the soil module, decomposition losses from the organic stock partially simulate shallow subsidence. Deep subsidence of the Holocene layer underlying the study area, which has been identified as the dominant factor contributing to RSLR in Louisiana (Penland and Ramsey 1990), was accounted for by increased rates of eustatic sea-level rise and included in the tidal forcing at the Gulf of Mexico boundary.

#### Habitat-switching module

A feature of the model was its capacity to keep track of habitat characteristics for each land parcel throughout time. The program not only recognized what type of habitat existed in each 1-km<sup>2</sup> cell, but also recorded a suite of environmental parameters, such as salinity and duration of flooding, that characterized the cell. The determination of habitat type based on biotic and abiotic factors has been extensively documented (Mitsch 1988, Laurenroth et al. 1993). Each day the module queried the biomass density, salinity, and duration of flooding for each 1-km<sup>2</sup> cell, and determined what habitat type characterized those values (Table 2). A tally for each daily habitat type was kept per cell, and this habitat counter added one unit to the type registered each day and subtracted one from the previous habitat type.

Habitat counters for each cell were queried by a hab-

itat-switcher algorithm at the end of every two years of simulation. This algorithm evaluated if the environmental conditions of the past two years for each cell had resulted in a habitat change. If more than half of the counts were for open-water conditions, then the cell was assigned an open-water type. If more than half of the counts were for marsh or swamp, the habitat with the highest count was assigned. New habitat-specific productivity rates were then used. Environmental stress was represented by daily salinity values and flooding duration, biomass density as an indicator of vegetation coverage, and the presence or absence of water as an indicator for open-water classification.

# Spatial implementation

The boundary conditions for the 100-km<sup>2</sup> hydrologic model were GoM tide elevation and salinity, Atchafalaya River discharge and suspended-sediment concentration, and various pumping stations and discharge locations at the perimeter of the two basins. A rainfalldependent discharge, located at the northern-most line of grid cells to account for the effects of runoff from the upper Terrebonne Basin, was set proportional to the actual rainfall for the upper basin.

The model incorporates nine forcing functions, including wind speed and direction, rainfall and evaporation, and tide in hourly time series, while salinity, temperature, river discharge, and inorganic sediment concentrations are daily time series. Wind velocity and direction records started in 1964 for the area. A canonical correlation (SAS Institute 1990) was performed, and missing years were replaced with the highest correlated years to reconstruct records for 1955-1963. Data originally given as daily values (e.g., rainfall and evaporation) were divided by 24 to compute hourly values. Tide stages were acquired from the National Ocean Service (NOS) at Bayou Rigard, Grand Isle, for 1955-1979 and from East Point, Grand Isle, from 1980-1988. To isolate the effects of RSLR all of the tide records were collated, where two distinct longterm trends of mean water level emerged. The period 1955 through 1974 had a linear trend of 1.31 cm/yr RSLR and the period 1975 through 1992 had a linear trend of 1.19 cm/yr. For the future climate-variability simulations, a constant rate of 1.2 cm/yr was applied.

Salinity values from Grand Terre Laboratory (near Grand Isle) collected by Louisiana Department of Wildlife and Fisheries were used as the boundary condition for the Barataria Basin. Because salinity was recorded as conductivity, the practical salinity scale (a conductivity ratio) is assumed. All of the data were missing for 1956 to 1958 and were replaced by the years 1977, 1987, and 1988 respectively, after a canonical correlation analysis (SAS Institute 1990). Hourly time series were comprised of 24 values of the daily salinity. Boundary conditions for Terrebonne Basin were set using the salinity distribution reported by Murray and Donley (1994). Salinity values were lowest at the Atchafalaya Delta and became progressively higher toward the east in the Terrebonne Basin. The daily difference between the two source points ranged from 3 to 9 depending on the season.

Daily maximum temperature records were collected from the National Weather Service tables for New Orleans Airport. Daily river discharge and sediment-load data were obtained from the U.S. Army Corps of Engineers, New Orleans District for the Tabert Landing Station. For continuous data, such as inorganic suspended sediments, the daily reading was used for 24 hours.

The U.S. Fish and Wildlife Service (USFWS) produced digital maps for coastal Louisiana derived from 1956 to 1978 aerial photography and from satellite imagery for 1988 habitat classification and 1990 land– water boundaries. The original map pixels, 25 m on a side, were aggregated to 1 km<sup>2</sup> by majority rule, then reclassified to four wetland categories (1, swamp; 2, fresh; 3, brackish; and 4, salt marshes), and two other habitats (open water and developed lands).

To evaluate how any simulated map compared to the USFWS maps, we used a multiple resolution fit index (Costanza 1989), as previously used for similar analyzes (Sklar and Costanza 1991). The index calculation begins with a comparison on a one-to-one cell basis between two maps, and computes the total number of matches. Then, the comparison window increases by a cell per side and recalculates the total number of matches, stopping when the window size is of the same size as the map itself. The multiple resolution fit index ( $F_i$ ) is the sum of total matches within the window size, and varies from 0, or no match, to 100, or perfect match.

An elevation base map was developed from a 1994 survey (Alawady and Al-Taha 1996). These measurements were interpolated to derive continuous land surfaces. These values provided information on the degree of variability of the elevation of each habitat, and were used for the 1988 elevation map. Initial 1978 elevations were estimated by incrementally adding elevation to the 1994 land surface and 1978 USFWS habitat map.

# CALIBRATION

### Individual module calibration

A calibration strategy was implemented to consider the different scales and modules of the model. Each module was first tested independently, and then combined into the final landscape model. Modules were tested using 1-yr forcing functions repeatedly and, later, with the forcing for the 1978–1988 period.

The hydrodynamic module was tested using the 1994 elevation map with only predicted tidal effects derived from the Naval Ocean Services (NOS) tidal constituents (Shureman 1994). Next, the module was run with the 1988 actual tide. NOS time series from the two basins were extracted where available. These time se-



FIG. 3. Calibration time series for salinity and water elevation for various stations in the Barataria and Terrebonne basins. The actual time series is the thick gray line, and the simulated time series is the thin line. Salinity is reported in terms of the practical salinity scale. Note different *x*-axis time scales.

ries were detrended and demeaned, and finally decomposed using a Fourier analysis (Dennis and Long 1978), to compare the magnitude and phase of the primary tidal constituents with the NOS observations (Fig. 3).

Salinity and suspended sediment calibrations were done by iteratively manipulating boundary conditions, such as water inflow and diffusion rates. Salinity results closely matched available data (Murray and Donley 1994) for the lower portions of the basins (Fig. 3). However, the averaging and interpolation procedures distributed salinity more smoothly and farther inland (by about 10 km<sup>2</sup>) than was generally observed. The actual pattern for these inland reaches is one of uniformly low or zero salinity punctuated by short-lived peaks of higher salinity.

Calibration of the macrophyte module was done by repeatedly running the four wetland habitats using daily time steps for a year, while varying forcing functions within observed limits. Some forcing functions, such as duration of flooding and salinity limits, varied with habitat while others, such as temperature, were held constant for all habitat types.

# Spatial calibration

The landscape calibration was done in three steps: First, the model was run repeatedly until matches were produced for land/water ratio for each basin; then, for the habitat-type proportions; finally, for the habitat distribution. For land/water ratios, the BTELSS (Barataria–Terrebonne ecological landscape spatial simulation) model was run using the 1988 forcing functions repeatedly until stable conditions were reached with all modules running concurrently. Then, observed sea-level rise was introduced, and the model was run from 1978 to 1988. A concurrent sensitivity analysis identified those parameters (e.g., Manning's coefficient and initial elevation) most critical to the model output.

To match habitat proportions, the landscape models for each basin were tested by varying several of the spatially distributed parameters (salinity limits, elevation, and Manning's coefficient). For example, changing salinity yielded fluctuations in the total number of cells per habitat (Table 3). These calibration runs indicated a need to obtain more accurate information about the mechanics of vegetation processes, as discussed below (see *Sensitivity analysis*).

Wetland elevation relative to sea level has been shown to be an important factor affecting productivity and health of vegetation (Nyman et al. 1993, Visser et al. 1996, Venterink and Wassen 1997). However, in both basins wetland elevation relative to mean sea level

Salinity upper limit	Brackish cells (km²)	Habitat fit index, $F_t$
12.5	865	85.08
13.0	851	84.52
13.5	896	84.60

TABLE 3. Salinity tolerance used for brackish marshes during the habitat-switcher calibration for Barataria Basin.

*Note:* Salinity is reported in terms of the practical salinity scale, a conductivity ratio.

prior to 1994 was unknown. The measured 1994 surface was used to calibrate the 1988 hydrodynamics (Table 4).

The third step in the spatial calibration was goodness-of-fit analysis (Costanza 1989) between the 1988 model and USFWS maps. The model was repeatedly run, varying the initial spatial parameters (e.g., initial elevation) until the overall fit improved to 85 or better for both basins (Table 4). The 1978–1988 calibrated base case simulations yielded a fit of 89.3 for the Barataria Basin and 85.08 for the Terrebonne Basin (Figs. 4 and 5, Table 5). There was also agreement for total wetland and water areas for the two watersheds ( $F_t =$ 96 for Barataria and  $F_t =$  94 for Terrebonne).

The BTELSS model was designed to simulate ecological processes that produce broad habitat patterns, and calibrated to match these landscape patterns so that all land-loss processes would be implicitly included. By using the USFWS maps for 1978 and 1988, the model incorporated all land-loss factors, since these maps reflect all of the effects impacting the landscape. The BTELSS model only incorporated large-scale factors (salinity, RSLR, and sediment transport) and their impacts having regional effects. The model did not explicitly incorporate local processes (<1-km<sup>2</sup> cell), and thus it did not accurately recreate historical land changes of a particular cell. Insofar as these processes contributed to overall land loss, they were included implicitly or indirectly since the actual land-loss rates were used for calibration.

#### VALIDATION

To validate the BTELSS (Barataria–Terrebonne ecological landscape spatial simulation) model, simulations were run for 1956–1987 with all parameters set to the 1978–1988 (base case) values, and predicted open-water area from 1956 to 1990 was output annually. The first derivative (wetland loss, in square kilometers per year) of these values was computed to illustrate the variation in wetland loss rates (Fig. 6).

The model predicted annual wetland loss fluctuations of 0-65 km<sup>2</sup> for Barataria and 0-85 km<sup>2</sup> for Terrebonne, similar to the values reported by Gagliano and colleagues (1981) of 73 km<sup>2</sup>/yr. Both basins had a pattern of moderate loss rates in the 1950s and 1960s and high rates in the 1970s (Fig. 6, Table 6). Wetland loss dropped through the 1980s to nearly 0 by 1990. Wetland-loss rates from the model compared favorably to values reported by the U.S. Fish and Wildlife Service (USFWS) and the U.S. Army Corps of Engineers (Table 6, Dunbar et al. 1992). Simulated wetland loss was similar to actual wetland loss for the 32-yr validation period (Table 6). Simulated open water was higher in both basins, indicating that the BTELSS model calculations, based on the 1978-1987 land-loss rate, were around 3% higher than the rates reported for earlier maps (Table 6). The calibration fit,  $F_{t}$ , was over 70 for both basins when comparing 1988 habitat distribution result to the 1988 USFWS map (Table 5).

#### SENSITIVITY ANALYSIS

For the sensitivity analysis, two parameters—habitat-dependent Manning coefficient  $(M_h)$  and initial el-

Initial	Habitat-type cell counts (km <sup>2</sup> )					Fit	0	
elevation (m)	Freshwater marsh	Swamp	Brackish marsh	Saltwater marsh	Open water	index, $F_t^{\dagger}$	$Sensitivity S_j$ ‡	cv§
-0.2	1284	990	357	150	3154	80.52	-460	105
-0.15	1103	990	580	340	2922	83.35	237	98
-0.1	1172	990	222	500	3051	83.71	-232	102
-0.05	1257	955	250	430	3043	83.63	-250	102
-0.01	1250	990	252	520	2923	83.75	438	98
0.0	1259	990	259	530	2897	83.72	640	97
0.05	887	991	671	267	3119	88.16	-1260	104
0.15	842	995	719	386	2993	88.99		
0.2	843	995	753	391	2953	88.94	-800	99
$0.4\ $	1002	781	894	175	3083	78.16	nc¶	nc¶

TABLE 4. Sensitivity analysis results for changes in the 1992 initial elevation base map for Barataria Basin.

*Notes:* The elevation base value (0.15 m) is boldfaced. All indices use the results from the base run for comparison purposes. All runs are for the 1987 weather repeated 10 times except as noted; nc = not calculated.

† See Costanza (1989).

‡ Jorgensen's (1988) sensitivity index,  $S_j = W_i - W_{0.15}/E_i - E_{0.15}$  where  $W_i$  is the resulting water area and  $E_i$  the initial elevation of each run.

§ Coefficient of variation (Steel and Torrie 1980).



FIG. 4. Comparison of Barataria 1988 U.S. Fish and Wildlife Service habitat map and calibrated BTELSS model output.

evation—were chosen because of the scarcity of empirical data and their likely contribution to model instability. Each parameter was varied  $\pm 1$  sD from its mean value to examine the effect on habitat composition and fit index, used as indicators of performance. Two sensitivity indices were used to evaluate the response: the relative change of the total land area by the relative change of the tested parameter ( $S_j$ , Jorgensen 1988), and the coefficient of variation between the land area base case and the sensitivity run (CV, Steel and Torrie 1980).

The first sensitivity experiment increased  $M_h$  for each habitat type as a measure of how fast water flows through the 100-km<sup>2</sup>-grid landscape. Using different  $M_h$ 



FIG. 5. Comparison of Terrebonne 1988 U.S. Fish and Wildlife Service habitat map and calibrated BTELSS model output.

	Resulting habitat coverage (km <sup>2</sup> )						
	Freshwater		Brackish		Open	Calibration fit <sup>†</sup>	
Scenario name	marsh	Swamp	marsh	marsh	water	Land/water	Habitat
Barataria Basin							
USFWS 1988 map	755	1022	734	460	2952		
Base case (calibration, 1977–1988)	723	1002	722	634	2854	95.97	89.32
Base case (validation, 1956–1988)	1191	795	577	288	3084	86.48	74.39
Normal conditions (1988–2018)	396	1017	236	217	4057		
Forward climate variability (1988–2018)	363	1017	294	247	4002	97.00	96.69
Mean Gulf and mean river (1988–2018)	682	1022	520	312	3387	89.78	88.07
High Gulf and high river (1988–2018)	1022	1022	293	159	3427	88.75	85.63
Low Gulf and low river (1988–2018)	859	817	773	303	3183	86.08	76.74
High Gulf and low river (1988–2018)	1099	817	770	279	2970	83.56	73.35
Terrebonne Basin							
USFWS 1988 map	1170	432	828	576	2106		
Base case (calibration, 1977–1987)	1100	516	865	551	2080	94.33	85.08
Base case (validation, 1956–1987)	847	657	596	842	2170	86.64	72.74
Normal conditions (1988–2018)	510	428	499	365	3310		
Forward climate variability (1988–2018)	496	428	535	356	3297	98.55	98.30
Mean Gulf and mean river (1988–2018)	737	426	600	360	2989	96.85	95.93
High Gulf and high river (1988–2018)	817	429	538	278	3050	94.37	92.63
Low Gulf and low river (1988–2018)	640	609	629	627	2607	90.40	88.76
High Gulf and low river (1988–2018)	847	429	840	425	2571	91.52	86.20

TABLE 5. Summary of scenario results performed in each basin.

*Notes:* Scenarios assumed different sea-level elevations for the Gulf of Mexico and different Mississippi River discharge amounts. "Forward climate variability" means taking the 1956–1992 climate records and repeating them beginning in 1993. † Fit values for base-case scenarios were computed against 1988 USFWS habitat map; fit values for weather patterns were computed against the 2018 normal-conditions habitat map.

revealed the zone with highest sensitivity as the transition area between fresh and saline environments (Table 7). There was a positive response between  $M_h$  and cv. The  $S_j$  had a negative relationship with  $M_h$  without a clear indication of causality.



FIG. 6. Model simulation output for annual land-loss rates in Barataria and Terrebonne basins. Literature estimates are shown for comparison, and horizontal bars indicate length of interval (yr).

For the second sensitivity analysis, the initial elevation was incrementally varied for 10-yr simulations (1978–1987, Table 4). Elevation determined the amount of flooding and how water was distributed throughout the landscape. Varying elevation did not always result in a steady increase of open water, and neither of the two sensitivity indices showed a correlation (Table 4). However, the best fit occurred with an elevation 15 cm above the 1992 datum. Starting below this elevation (negative numbers in Table 4) resulted in an unrealistic increase in land loss due to flooding, while starting above this elevation resulted in land loss due to lack of salinity in brackish and salt-marsh habitats. Running a 30-yr simulation with initial elevation 40 cm above datum showed that the relationship between sea-level rise, subsidence, and land elevation was not linear (last line of Table 4).

# FUTURE SCENARIOS

Our objective was to understand how climate variability influenced landscape habitat distribution and determine the resilience of Barataria and Terrebonne wetland habitats to diverse weather conditions as surrogates for potential conditions of global change. The Barataria–Terrebonne ecological landscape spatial simulation (BTELSS) model simulations analyzed the impact of different climate-variability scenarios for both basins from 1988 to 2018. All future simulations included a relative sea-level rise (RSLR) of 1.2 cm/yr over the mean sea-level conditions at the Gulf of Mexico, a value in the middle of the range reported in the

	Barataria				Terrebonne			
Interval	Dunbar et al. (1992)	Reed (1995)	USFWS (1988)	Model output	Dunbar et al. (1992)	Reed (1995)	USFWS (1988)	Model output
1931-1958	7				5			
1956-1978		20.09	21.55	25.50	•••	24.86	21.27	26.73
1974-1983	22				20			
1978–1988	•••	34.5	35.40	27.11		31.6	33.10	33.89

TABLE 6. Comparison of estimates of land-loss rates (km²/yr) for several intervals in Barataria and Terrebonne basins.

Note: USFWS (1988) = U.S. Fish and Wildlife Service 1988 habitat map.

literature (Gornitz 1995). Although there had been simulations of global-change effects of Mississippi River discharge (Miller and Russell 1992, Knox 1993) and coastal waters of Louisiana (Justic et al. 1996), none existed for Mississippi Delta wetlands.

The BTELSS model allowed treatment of long-term sea-level rise separate from interannual variation in mean water levels as factors potentially driving habitat change. The first simulation of future conditions was labeled the "normal-conditions scenario" (NCS). Climate variability, including the interactions among weather, river discharge, and changes in mean sea level that produce the observed variation in land loss, was highly dependent on the time series used to drive the model. To test how climate variability influenced the landscape dynamics, experiments were done using various permutations of the time series to generate alternative futures. To evaluate these interactions, comparisons with the NCS 2018 habitat map were made using the goodness-of-fit index and the overall percentage of land change (OLC-the percentage of predicted total land counts divided by 1988 land counts). Five alternative scenarios were simulated: (1) repetition of weather time series, (2) yearly mean sea level and mean river discharge conditions, (3) high sea level and high

discharge, (4) low sea level and low discharge, and (5) high sea level and low discharge (Table 5).

# Normal conditions

The normal-conditions scenario consisted of a 30-yr (1988–2018) simulation for each basin using theoretical time series and boundary conditions. Climate tends to be cyclic in nature (Latif and Barnett 1994, Thomson 1995), and to simulate future conditions we ran the original time series in reverse order. That is, the forcing functions and boundary conditions used data for the years 1955–1992, but when the year 1993 was simulated the weather from 1991 was applied, when 1994 was simulated the weather from 1990 was used.

The resulting habitat-change maps showed largescale marsh deterioration. In Barataria 1,105 km<sup>2</sup> of marsh were converted to open water (Fig. 7), compared to 1204 km<sup>2</sup> for the Terrebonne Basin (Fig. 8). The water/land ratio increased from 0.94 in 1988 to 1.99 and 0.62 to 1.57 for Barataria and Terrebonne, respectively. The largest decline in the Barataria Basin was for brackish marsh (498 km<sup>2</sup>), while fresh marsh loss was greater (660 km<sup>2</sup>) for the Terrebonne Basin. In both basins there were large contiguous losses in the midand upper basins, and fragmentation in the lower, saline

TABLE 7. Sensitivity analysis results for the habitat-dependent Manning coefficient,  $M_{\rm h}$ .

	Habitat-type cell counts (km <sup>2</sup> )						
Freshwate		hwater		Saltwater	Open	Sensitivity index	
$M_{ m h}$	marsh	Swamp	marsh	marsh	water	$S_j$ †	CV‡
Barataria Basi	n						
0.0125	767	1001	745	653	2769	3680	98
0.025	750	1001	722	647	2815		
0.05	732	1001	696	622	2884	2760	102
0.1	726	1001	662	603	2943	1707	105
0.2	721	1002	632	573	3007	1097	107
Terrebonne Ba	isin						
0.0125	1103	516	917	629	1947	5920	96
0.025	1101	516	877	597	2021		
0.05	1094	516	832	564	2106	3400	104
0.1	1096	513	756	505	2242	2947	111
0.2	1105	512	701	458	2336	1800	116

*Notes:* Higher coefficients indicate higher friction. The base value is 0.025 (boldfaced). All indices use the results from the base run for comparison purposes.

† Jorgensen's (1988) sensitivity index,  $S_j = W_i - W_{0.025}/M_i - M_{0.025}$  where  $W_i$  is the resulting water area and  $M_i$  is the Manning coefficient of each run.

 $\ddagger CV = coefficient of variation.$ 



FIG. 7. Simulated output under the normal-conditions scenario and comparison with Barataria 1988 U.S. Fish and Wildlife Service habitat map. The output map is for a 30-yr run from 1988–2018.

areas (Figs. 7 and 8, Table 5). These spatial differential losses were due to differences in initial elevation, salinity, and water levels between the upper and lower basin. However, model predictions for land-loss rates (Table 8) were similar to current estimates (Table 4).

Increasing water levels and flood duration led to lower marsh productivity and marsh conversion to open water at several sites (Table 8). For example, at Bayou L'Ours (Fig. 1), increasing water depth led to habitat change by 2017 (Fig. 9a), while at Bayou Perot (Fig. 9b) the habitat change and consequently land loss occurred in 2015, probably because of the proximity of this site to the coast. The conversion of a freshwater marsh north of Lake de Cade (Fig. 9c) in 2011 was also due to increased flooding and salinity. At the Dularge site (Fig. 9d) flooding duration increased to >20 h/d, similar to what has been measured in deteriorating coastal marshes in Louisiana (Wang 1997).



FIG. 8. Simulated output under the normal-conditions scenario and comparison with Terrebonne 1988 U.S. Fish and Wildlife Service habitat map. The output map is for a 30-yr run from 1988–2018.

	Land-loss rates (km <sup>2</sup> /yr)			
Scenario name	Barataria	Terrebonne		
Normal conditions, 1988–2018	36.83	40.13		
Forward climate variability, 1988–2018 <sup>†</sup>	29.67	38.03		
Mean Gulf sea level and river discharge,				
1988–2018	15.33	32.23		
High Gulf sea level and river discharge, 1988–2018	15.83	31.83		
Low Gulf sea level and river discharge,				
1988–2018	14.10	16.87		
High Gulf sea level and low river discharge,				
1988–2018	15.23	15.50		

TABLE 8. Predicted rates of land loss for the 1988–2018 interval in Barataria and Terrebonne basins of coastal Louisiana, USA, with varying conditions in the Gulf of Mexico and the Mississippi River.

<sup>†</sup> The 1956–1992 climate records were repeated beginning in 1993.

### Climate variability scenarios

The first experiment ran the weather records in a forward manner (Forward Climate Variability Scenario in Table 5), where the 1956–1992 climate was repeated beginning in 1993. The results for the Barataria and Terrebonne Basins showed 1.3% and 0.4% dissimilarity, respectively, with the 1988–2018 NCS. Both of these values were less than the calibration (1978–1988) variation (3.3% and 1.2%, respectively) and, thus, considered indicators of negligible change.

Model stability and the capability of the marsh communities to endure long-term extreme conditions were tested by repetitively running four representative years with mean and extreme values for sea level and river discharge (Fig. 10). The year 1986 had mean values, in 1983 high levels of both forcing functions occurred, 1964 had low values, and in 1956 a combination of high Gulf of Mexico sea level and low river discharge occurred. The model was run for 30 yr with each of these weather years repeatedly to gauge the overall effect of extreme and average conditions on habitat change.

The effects of river discharge and mean sea level were different for each basin. Barataria Basin had the highest percentage of change (OLC = 15.5%) with the high Gulf–high river scenario. For the same scenario in Terrebonne, the OLC was only 7.8%. Open-water cell counts were minimal for low Gulf–low river, and high Gulf–low river conditions in both basins. The percentage change for low Gulf–low river was 21.5% in Barataria and 21.2% in Terrebonne. The high Gulf–low river scenario yielded 26.8% change for Barataria and 26.4% for Terrebonne.

To measure the influence of the climate variability in a spatial context the multiple resolution fit index was also used (Table 5). Mean Gulf and river-discharge fit indices were the highest (or more similar to the NCS) from all the climate variation simulations, showing that even typical conditions of mean sea level and river discharge did not reduce the current land-loss rate trends. The lowest indices for both basins resulted from the two low-river simulations. These lower indices were consequence of less open-water cell counts on each basin.

#### DISCUSSION

Landscape models are one of the few tools that can be used to predict the effects of complex interactions and cumulative, long-term effects of global change both spatially and temporally. The response of deltaic habitats to multiple impacts cannot be simply extrapolated. Historical trends of these responses often do not accurately predict future conditions (Dale and Rauscher 1994), mainly because of the accelerated rates for sea-level rise and global warming, and changes in the composition of the plant assemblages (Dale and Rauscher 1994). A process-based, spatially explicit model can cope with the inherent complexity of such future scenarios (Ruth and Pieper 1994).

The Barataria–Terrebonne ecological landscape spatial simulation (BTELSS) model presented here, was based on the existing CELSS (coastal ecological landscape spatial simulations) methodology (Sklar et al. 1985, Costanza et al. 1988, Costanza et al. 1990). However, this new landscape simulation model included the implementation of an explicit hydrodynamic module, improved ecological algorithms for primary production and habitat switching, and three times the original spatial extent. Much of the present effort was dedicated to refine the algorithms from earlier CELSS versions, and to obtain a similar or better resolution fit than the original CELSS model (CELSS fit was 86, Costanza et al. 1987, Sklar et al. 1991).

The BTELSS model was designed to be forced by and respond to dominant regional coastal processes. However, the model did not simulate plant and soil processes at less than a 1-km<sup>2</sup> scale, or hydrologic processes at less than 100 km<sup>2</sup>, which can produce localized changes (Salinas et al. 1986, Turner and Rao 1990). Therefore, the calibration method had to compensate for the presence of local effects. Increasing habitat response to weather effects and regional forcing



FIG. 9. Time series of water depth and photosynthetic biomass for four sample locations for the normal-conditions scenario. Arrows indicate when habitat change occurred.

functions resulted in a predicted habitat map that resembled the real 1988 coastal habitat distribution of each basin (Figs. 4 and 5, Table 5).

The validation runs further tested the effects of regional factors, such as subsidence and salt intrusion, on wetland habitat response. The fit index,  $F_t$ , for this run (Table 5) demonstrated the degree of uncertainty of the habitat response algorithms. The  $F_t$  difference between the maps for the historical 30-yr runs and the 1988 U.S. Fish and Wildlife Service maps was about 30 points (i.e., a variation of about 10 points every 10 years). This decadal 10-point discrepancy may be attributed to interannual variations of sea-level rise (a constant rate for the 30 yr in our model) and other effects such as canal dredging, and marsh impoundment.

A primary objective of this study was to evaluate the usefulness the BTELSS model for predicting landscape responses under varying environmental conditions (Tables 5 and 8). The overall highest fit achieved was under mean sea level and mean discharge conditions. This can be interpreted as a result of maintaining conditions within the normal range of weather variability. Minimum influences (i.e., low Gulf of Mexico



FIG. 10. Mississippi River discharge and mean sea level for historical records (1956–1988) and simulated annual landloss rates per basin. Shaded bars indicate selected years for weather experiments. Note that land loss in the 1960s and late 1980s coincides with low sea levels.

sea level-low river-discharge scenario) resulted in less land loss and thus greater differences in habitat distribution when compared to the normal-conditions (NC) scenario. As the weather conditions exerted stress (i.e., high Gulf-high river-discharge scenario) land loss increased, demonstrating further the coupled response of coastal regional change and wetland loss (Table 8).

The model predicted relatively high loss rates, on the order of 50-80 km<sup>2</sup>/yr for the mid-1990s for all but the high Gulf-high river-discharge scenario, where values nearly double this amount took place (Fig. 6). These results, along with the NC and forward-climatevariability scenarios, suggested interannual variability as responsible for the largest changes in marsh stability. The importance of yearly sea-level rise and its effect on land loss for the two basins was evident (Figs. 6 and 10). Accretion, vegetation productivity, and sediment inputs alone did not compensate for the effects of increased sea-level rise (as high as 10 cm interannually; Penland and Ramsey 1990), acute weather conditions (hurricanes and winter storms), and natural subsidence (Baumann et al. 1984, Coleman 1988, Day and Templet 1989, Cahoon 1994, Dale and Rauscher 1994).

# Model limitations

The accuracy of model functioning and predictions could be improved with better input and validation data. For example, elevation is one of the most sensitive parameters affecting marsh survival (Day and Templet 1989, Cahoon 1994, Reed et al. 1997). Yet, there are accurate elevation data for only a few locations and lack of historical records. Elevation of more locations gathered at regular intervals would prove invaluable for model improvement, as well as for coastal management in general (Wells 1996, Reed et al. 1997). Better monitoring of salinity and water level at a number of stations would also allow much better calibration and validation of the model.

The ecological and habitat-switching modules focused on those factors that directly and predictably influence land elevation and habitat type. One of the most important factors for vegetation production is nutrient availability (Howes et al. 1986, Childers and Day 1990, Nyman et al. 1990). In a landscape context, the influences of river-borne nutrients could not be isolated and distinguished from the effects of freshwater and sediment. A lack of landscape-level nutrient information made it difficult to predict availability, rates of transformations within the estuary, or exchange with the atmosphere, much less the response of plant communities to all of these factors. While nutrient influences affect land elevation, inclusion of nutrients would call for a great deal of data. The productivity module should include nutrient influences to make the model a much more useful tool in predicting eutrophication and inshore nutrient cycling.

# Future directions

Although the present model satisfactorily represented by Barataria and Terrebonne systems at a large scale, there are some improvements in the structure that would result in better spatial and temporal resolution. Two of the most important changes would be reducing the scale of the hydrodynamic module and adding a land-building component. The scale of the hydrodynamic module should be reduced to 1 km<sup>2</sup>, the same scale as the ecological and soil modules, to account for smaller-scale features. Inclusion of land-building capability in the model would allow for vegetation colonization. For much of the present effort, this was not a problem because most of the area is not directly affected by riverine sediment input. In the future, modeling river diversions and the influence of the Atchafalaya River will be more accurate with a land-building component.

#### Acknowledgments

Financial support for this study was provided by the Barataria–Terrebonne National Estuary Program (BTNEP) through the Louisiana Department of Environmental Quality. The authors would like to acknowledge the assistance of the BTNEP Scientific and Technical Committee members. We are grateful to Hasan Mashriqui, Phillip Atkinson, and James Hyfield for their time and effort with midnight runs and graphics preparation, as well as to Jennifer Purdue for data assembly. Emily Hyfield proved to be invaluable at the word processor and bibliographic research. We also appreciate the comments and suggestions of two anonymous reviewers and W. L. Baker.

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#### APPENDIX

Here, we present equations for several modules of the BTELSS (Barataria–Terrebonne ecological-landscape spatial simulation) model.

#### HYDRODYNAMIC MODULE EQUATIONS

The governing equations for two-dimensional water movement are the continuity equation

$$\frac{\delta z}{\delta t} + \frac{\delta}{\delta x} [(h+z)U] + \frac{\delta}{\delta y} [(h+z)V] = \text{sources} - \text{sinks}$$

where h = mean water elevation (m), t = time (s), U = depthaveraged velocity in x direction (m/s), V = depth-averaged velocity in y direction (m/s), and z = water level variation (m) and the x and y momentum equations

$$\frac{\delta u}{\delta t} + u \frac{\delta u}{\delta x} + v \frac{\delta u}{\delta y} - fv = -g \frac{\delta z}{\delta x} - \gamma_x u + \frac{\tau_x^w}{\rho_w H} \text{ and}$$
$$\frac{\delta v}{\delta t} + u \frac{\delta v}{\delta x} + v \frac{\delta v}{\delta y} + fv = -g \frac{\delta z}{\delta y} - \gamma_y v + \frac{\tau_y^w}{\rho_w H}$$

where f = Coriolis parameter (dimensionless),  $g = \text{acceler$  $ation due to gravity (m/s<sup>2</sup>)}$ , H = h + z total water elevation (m),  $\gamma = \text{friction coefficients (dimensionless)}$ ,  $\tau = \text{wind stress}$ in horizontal direction (N/m<sup>2</sup>), and  $\rho_w = \text{density of water (kg/m<sup>3</sup>)}$ .

The effect of friction is accounted for by the relationship

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$$\gamma = \frac{gM^2}{H^{4/3}}(u^2 + v^2)^{1/2}$$

where M = Manning coefficient averaged at a 10 km<sup>2</sup> scale. The effect of wind stress is:

$$\tau = \rho_a C_d W$$

where

$$C_d$$
 = wind-sheer stress coefficient based on wind speed

$$= 1.25 \times 10^{-6} + 1.75 \times 10^{-6} \sin\left(\frac{\pi_{W-W_1}}{2W_2 - W_1}\right)$$
  
for  $W_1 < W \le W_2$ 

$$= 3.0 \times 10^{-6}$$
 for  $W > W_2$ 

and where  $W_1 = 5.1$  m/s,  $W_2 = 15.0$  m/s, W = wind speed (m/s), and  $\rho_a =$  density of air (kg/m<sup>3</sup>).

These equations in finite difference form require a smaller time step than is practically useful in the prediction of longterm effects (Casulli 1992). The use of these equations in the prediction of long-term water level requires some simplification. Diffusion wave approximation of shallow-water equations (Singh and Aravamuthan 1995) were used, requiring that (1) local acceleration, (2) uniform flow, and (3) Coriolis force are negligible—i.e., for (1) local acceleration,  $\delta u/\delta t =$  $\delta v/\delta t \cong 0$ ; (2) uniform flow,  $\delta u/\delta x = \delta u/\delta y = \delta v/\delta x = \delta v/\delta x$ = 0); and (3) Coriolis force,  $fv \cong fu \cong 0$ ).

CONSERVATIVE AND NONCONSERVATIVE MATERIALS

The continuity equation for salinity is

$$\frac{\delta S}{\delta t} + u \frac{\delta S}{\delta x} + v \frac{\delta S}{\delta y} + D_s \frac{\delta^2 S}{\delta t^2} = 0$$

where  $D_s$  = diffusion coefficient for salinity and inorganic sediment (m/s<sup>2</sup>) and S = salinity (in terms of the practical salinity scale, a conductivity ratio).

The continuity equation for inorganic suspended sediments is

$$\frac{\delta I}{\delta t} + u \frac{\delta I}{\delta x} + v \frac{\delta I}{\delta y} + D_s \frac{\delta^2 I}{\delta t^2} + \frac{dz_i}{dt} A = 0$$

where  $A = \text{area of cell } (m^2)$ , I = inorganic suspended sediment concentration (kg/m<sup>3</sup>), and  $z_i = \text{depth of deposited inorganic sediment (m)}$ .

#### **PRODUCTIVITY MODULE EQUATIONS**

Aboveground macrophyte production is given by

$$\frac{dB}{dt} = pE$$

where B is above ground biomass in grams of organic matter (gOM), and p is the actual production rate, with

$$p = P - (\phi B + \lambda B + \gamma B)$$

where the coefficients  $\phi$  = translocation rate from aboveground biomass,  $\lambda$  = detritus or litterfall rate (Turner 1976), and  $\gamma$  = the habitat-specific aboveground respiration rate (Blum et al. 1978, Hopkinson et al. 1978, Mitsch and Reeder 1991, Dai and Wiegert 1996), all in g OM/d, and the specific gross production, *P*, is

$$P = \mu P \times F(S \times J \times [C/C_{max}])$$

where  $\mu P$  = the maximum gross production rate per habitat (Dai and Wiegert 1996). Environmental external factors include: salinity tolerance ( $S_i$ ) as an empirical function per habitat (Pezeshki et al. 1987), water-level stress tolerance (J) in hours tolerance also as an empirical function where tolerances for brackish and salt marshes are from 0.00 to 11.00 h and for swamp and fresh marsh are from 0.00 to 24.00 h (Nyman et al. 1993). The last term of the production equation is a function of temperature (C in centigrade) and C<sub>max</sub> is the air temperature maximum for a 30-yr record (Morris et al. 1984).

The belowground macrophyte production function (g OM) is defined as

$$\frac{dG}{dt} = kG$$

with initial biomass values being habitat determined (Kirby and Gosselink 1976, Childers and Day 1990), and where

$$k = T - (\eta G + \sigma G)$$

i.e., the belowground increment rate (*k*), per habitat, is a function of the translocated aboveground biomass (T = - øB, Howes et al. 1985) minus belowground mortality rate ( $\eta$ ; in g OM/d) and  $\sigma$  is the belowground respiration rate (g OM/d, Gleason and Dunn 1982).

#### SOIL MODULE EQUATIONS

The contribution of soil pore space to elevation was calculated using the following equation

poreht = 
$$\frac{\text{pore }\%}{1 - \text{pore }\%} \times (\text{ht. inorg + ht. org})$$

Total sediment elevation (poreht) results from adding belowground inorganic height (ht. inorg), organic storage (ht. org), and percentage of pore space (pore %). Total sediment height is the sum of organic, inorganic, and pore height for each 1-km<sup>2</sup> cell. Annual inorganic deposition (kg/m<sup>2</sup>) from the hydrodynamic module and belowground mortality (kg/ m<sup>2</sup>) are inputs to inorganic sediments and dead belowground organic sediments, respectively.