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Projecting Fine Resolution Land-Cover Dynamics for a Rapidly Changing Terrestrial–Aquatic Transition in Terrebonne Basin, Louisiana, U.S.A.

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ABSTRACT

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Coastal landscapes are in a constant state of flux and continue to change with sea level rise. Past trends in land cover may be useful for predicting future landscapes under different scenarios of change. In this study, models representing land-cover change were created for a coastal forest–open water transition in a rapidly subsiding region in the Mississippi Delta, U.S.A. Land-cover images manually delineated from aerial photography for 1940, 1953, 1978, 1998, and 2004 served as the basis for the models. A combination of Markov chain analysis, a cellular automaton, and suitability images were used to model past trends and to create future land-cover scenarios. Model validation revealed that one of two model runs more closely matched reference images than null (no change) models. Models were generally better at predicting the location of land-cover classes on the landscape than the quantity of each class. Prediction accuracy varied among individual land-cover classes, with forest being the most stable and predictable, and scrub-shrub being the least stable and predictable. Future projections offered a range of outcomes and suggest that coastal stability structures are at least initially effective in promoting marsh replacement of open water. Without management intervention, our models predict dramatic loss of marsh and an increasing interface between water and the relatively resistant forest. These models can be helpful in examining responses of coastal transitions to sea level rise and evaluating the effectiveness of management efforts.

ADDITIONAL INDEX WORDS: *Climate change, coastal wetlands, GIS, Gulf Intracoastal Waterway, Louisiana, sea level rise, subsidence, spatial modeling.*

INTRODUCTION

Knowledge of historical patterns and trends of landscape change can provide insights into processes governing ecosystems (TURNER, 1989). In rapidly changing systems, past trends in land cover are especially valuable because they may provide information for generating ecologically realistic scenarios of future change (VELDKAMP and LAMBIN, 2001). Coastal terrestrial–aquatic transitions are naturally dynamic systems that are in a constant state of flux with changing environmental conditions, including human settlement and corresponding land conversions, hydrologic alterations, and sea level rise (DAY *et al.,* 2000; IPCC, 2001; KENNISH, 2001, 2002; SHIRLEY and BATTAGLIA, 2006; SMALL and NICHOLLS, 2003). These systems are likely to be among the "first responders" to global climate change, particularly sea level rise.

Sea level rise (SLR) is affecting ecosystems of low-lying coastlines around the world (BOURNE, 2000; DAY *et al.,* 1995, 2000; SCAVIA *et al.,* 2002). Relative SLR in the Mississippi Deltaic Region can exceed 10 mm/y, which is nearly 10 times the global average of 1.2 mm/y (PENLAND and RAMSEY, 1990). PARK *et al.* (1991) projected a loss of up to 82% of coastal land in the United States from SLR, the vast majority of which is projected to occur along low-lying coasts in Florida, Texas, and Louisiana (TITUS *et al.,* 1991), where the aquatic–terrestrial transition occurs over small and gradual elevations and distributions of species are tightly linked to flooding and salinity gradients (HOLM and SASSER, 2001; VISSER *et al.,* 1998).

Estuarine ecosystems, nourished by large rivers such as the Mississippi River, differ from other coastal systems in that they have large inputs of fresh water (SHAFFER *et al.,* 1992) that can offset salinity effects associated with SLR (DAY *et al.,* 1995). Coastal communities in the Mississippi River Delta may undergo an extended period of increased freshwater flooding, mortality of flood-intolerant species, and a shift toward more flood tolerant vegetation (CONNER and BRODY, 1989; CONNER and DAY, 1988; DENSLOW and BATTAGLIA, 2002) before salinity effects become apparent. Elevation is strongly correlated with these factors (DENSLOW and BATTAGLIA, 2002) and is therefore used as a surrogate (BATTAGLIA, KEOUGH, and PRITCHETT, 1995; HODGES, 1997) when direct measures of those factors cannot be made. The zonation pattern of vegetation tracks elevation and reflects

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species-specific distributions along flooding and salinity gradients (Visser *et al.,* 1998).

The Mississippi Delta area is experiencing rapid relative SLR due to high rates of subsidence from tectonic downwarping and natural sediment compaction (Day *et al.,* 1995; Penland and Ramsey, 1990). In addition, extensive canal networks facilitate saltwater intrusion into the interiors of coastal communities. Striking changes at the aquatic–terrestrial interface are evident from land-cover images that span even short time sequences (Day *et al.,* 2000; Sasser *et al.,* 1986; Shirley and Battaglia, 2006). Accurate predictions of future changes in these coastal transition communities will require models that incorporate fine-scale spatial data on elevation and land cover in the Delta (Reyes *et al.,* 2000). Further, a sufficiently long record of land cover change is needed to develop and to evaluate future scenarios of change.

The primary objective of this research is to create models based on past land-cover trends that can be used to project future land cover in a rapidly changing coastal landscape in the Mississippi Deltaic Plain. We use a set of aerial photography spanning 60 years to model fine-scale land-cover dynamics and then use these models to project future landscapes based on several hypothetical scenarios.

METHODS

Study Area

Mandalay National Wildlife Refuge (MDNWR) is located in the Terrebonne Basin (11.3 km southwest of Houma), which is part of the Mississippi Deltaic Region in southern Louisiana, U.S.A. (Figure 1). The refuge is managed by the U.S. Fish and Wildlife Service. Mandalay was chosen for modeling because it contains an intact coastal forest-to-marsh transition, and it is in a region that is losing land to water rapidly (Sasser *et al.,* 1986). Relative SLR in this area is estimated at >1 cm/y (Penland and Ramsey, 1990) and is the result of tectonic downwarping, sediment compaction, and anthropogenic alterations such as levees and canals (Day *et al.,* 1995; Penland and Ramsey, 1990; Visser *et al.,* 1999), in addition to background eustatic SLR. Vertical marsh accretion usually cannot offset the rising water levels (DeLaune *et al.,* 1989), and coastal communities are threatened with inundation, erosion, and saltwater intrusion (Baumann, Day, and Miller, 1984; Day *et al.,* 1995; Martin *et al.,* 2002). MDNWR is bisected by the Gulf Intracoastal Waterway (GIWW), which was dredged in 1935; smaller canals followed in later years. Erosion control structures, including earthen plugs and submerged concrete revetment mats, were installed along the GIWW in 2003 to prevent further erosion resulting from wave action and salinity encroachment (USFWS, 2003).

The transition of coastal vegetation assemblages at MDNWR is characterized by expansive floating freshwater marshes (Sasser *et al.,* 1996; Swarzenski *et al.,* 1991); common species include *Panicum hemitomon* J.A. Schultes and *Sagittaria lancifolia* L. interspersed with shrub patches of *Morella cerifera* (L.) Small. The forest–marsh ecotone has scattered flood-tolerant woody species and freshwater marsh species in the groundcover and grades into forest, composed of *Taxodium distichum* (L.) L.C. Rich and bottomland hardwood species, that sits upon a former natural levee of the Mississippi River.

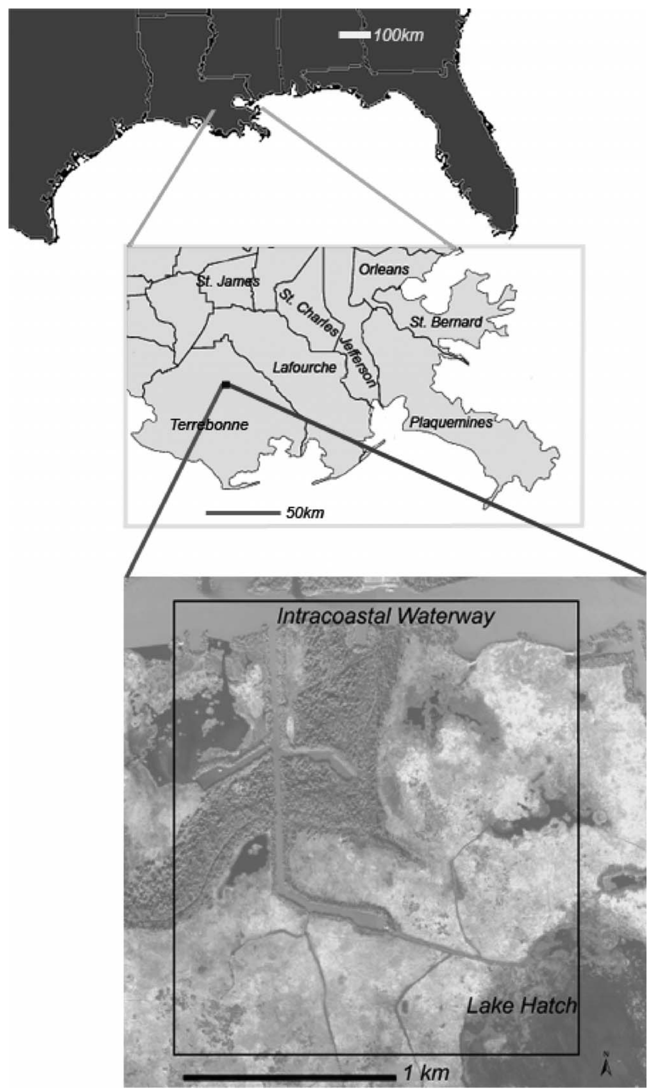

Figure 1. Location of Mandalay National Wildlife Refuge (Lat: 29.52° N; Long: 90.48° W) in southeast United States.

Land-Cover Data

Land-cover data based on aerial photographs of MDNWR (Figure 2) were used to build models because photographs provide the longest record of landscape changes of all remotely sensed imagery, and they have been used effectively for assessing change in forest-herbaceous ecotones (Bowman, Walsh, and Milne, 2001; Mast, Veblen, and Hodgson, 1997). The aerial photography was manually delineated into forest, scrub-shrub, marsh, and open water for 1940, 1953, 1978, 1998, and 2004 using a stereoscope; data were georeferenced to a Universal Transverse Mercator coordinate system. Manual interpretation affords the opportunity to distin-

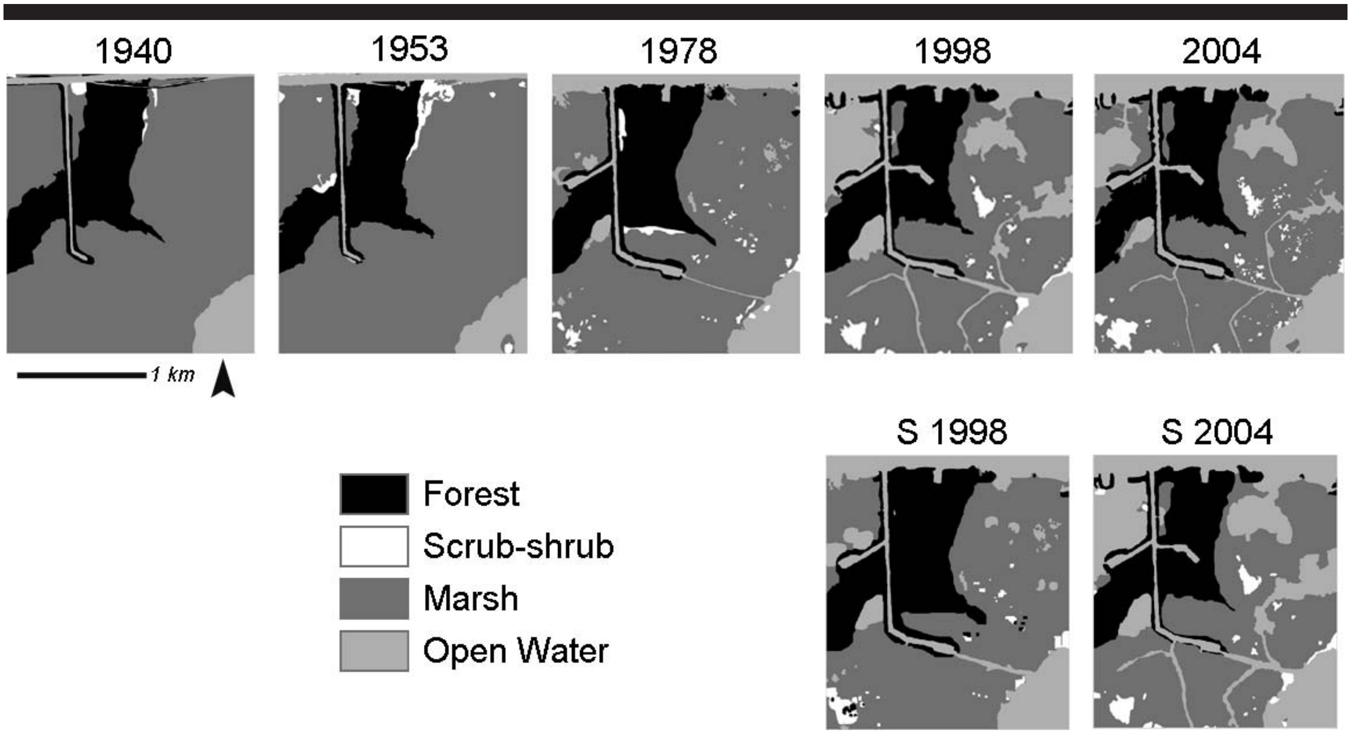

Figure 2. Land cover for Mandalay National Wildlife Refuge in 1940, 1953, 1978, 1998, and 2004 and simulated images (S) of 1998 and 2004 for model validation.

guish different vegetation types at a fine scale (KADMON and HARARI-KREMER, 1999), which is critical in systems where species turnover occurs spatially over a small range of elevation and temporally with relative sea level rise. Sparse land-cover categories such as small shrub patches were captured in the delineation process.

Model Overview

We used a combination of Markov chain analysis (MCA) and a cellular automaton (CA) to project future land cover (HOUET and HUBERT-MOY, 2006; PAEGELOW and CAMACHO OLMEDO, 2005). Markov chain analysis is a stochastic process model that predicts quantities of each land-cover category based upon the probability of state change (BAKER, 1989). Markov chain analysis does not include associated geographical information (WEAVER and PERERA, 2004), so the spatial distribution of cover quantities is established by a CA and designated suitability image for each class. CA can model many types of ecological changes governed by spatial processes (HOUET and HUBERT-MOY, 2006; PAEGELOW and CAMACHO OLMEDO, 2005). Known or suspected areas that either provide stable habitat or are vulnerable to change were incorporated into the model with the third model input, the multicriteria–multiobjective land allocation (MOLA), which is a decision-optimization process that produces a single output map based upon multiple suitability layers created for each state (EASTMAN, 2006).

Model Creation

Models were created using the CA_Markov module in IDRISI 15.0, which combines MCA, CA, and MOLA (EASTMAN, 2006). The CA_Markov module requires the input of the area predicted to change from one state to every other state, CA neighborhood specifications (we used the default 5 × 5 cell neighborhood), and suitability images for each class. The MCA probabilities were created using the MARKOV module in IDRISI 15.0. A single suitability layer was produced for each land-cover class. Suitability images were created using multicriteria evaluation (MCE) in IDRISI. Multicriteria evaluation combines two main types of inputs, factors, and constraints by implementing weighting and exclusion factors to produce risk values for each cell (EASTMAN, 2006). Factors are continuous variables that allow for a range of suitability values, and constraints exclude certain areas. Constraints were not used in this analysis because transitions to and from every state were observed in the input data.

Factors were based upon known or hypothesized processes governing site dynamics. In wetlands such as these, plants are distributed across flooding and salinity gradients according to the physiological tolerance of the species in question (BATTAGLIA and SHARITZ, 2006; JONES *et al.,* 1994; KEDDY, 2000; SHARITZ and PENNINGS, 2006). Plants in low-lying areas are typically more vulnerable to changes in flooding regimes and saltwater intrusion, especially relatively short-lived marsh species (BRINSON, CHRISTIAN, and BLUM, 1995). Standard 10- to 30-m digital elevation models (DEMs) cannot distinguish the small changes in elevation and related hydrology that influence species at this site, so we used a DEM from 2002 sampled with LIDAR techniques at a 5-m grid. It is possible that the elevation changed slightly during the study period with land subsidence, a component of relative sea level rise, which may have influenced the ability of the model to predict certain land-cover classes (see Discussion). The DEM was reclassified for each category by assigning higher suitability values to elevation ranges that were more likely to contain that particular land-cover category. The likelihood of a category falling into each elevation range was determined by calculating the frequency of occurrence of classes within each elevation range in the 2004 land-cover data. The 2004 data set was selected for calculation because it was the closest to the year of the DEM (2002).

The reclassified DEMs served as the basis for the suitability images for each vegetation class, and other factors were included on a case-by-case basis. As observed from past land-cover images, forest tends not to readily expand into areas it did not previously occupy, so areas near existing forest were assigned higher suitability values. This layer was created using the Multiple Ring Buffer Tool in ArcMap 9.1; the image was converted to a 2-m raster, imported into IDRISI 15.0, and stretched and reclassified to meet the requirements of the MCE procedure input. This technique was also used for the marsh and open water suitability images to account for the rapid marsh to open water conversion near the Gulf Intracoastal Waterway (GIWW) (Figure 2). We assigned decreasing suitability values for marsh and increasing suitability values for open water within 500-m of the 1940 GIWW outline. Unlike forest, scrub-shrub is a highly dynamic class that appears in a variety of locations. For this reason, no factors besides elevation were added to prevent or decrease the suitability of certain areas. Scrub-shrub occupied a relatively large range of elevations, and so the corresponding suitability image rendered most areas suitable for scrub-shrub (Figure 3). All factors were combined using the MCE module.

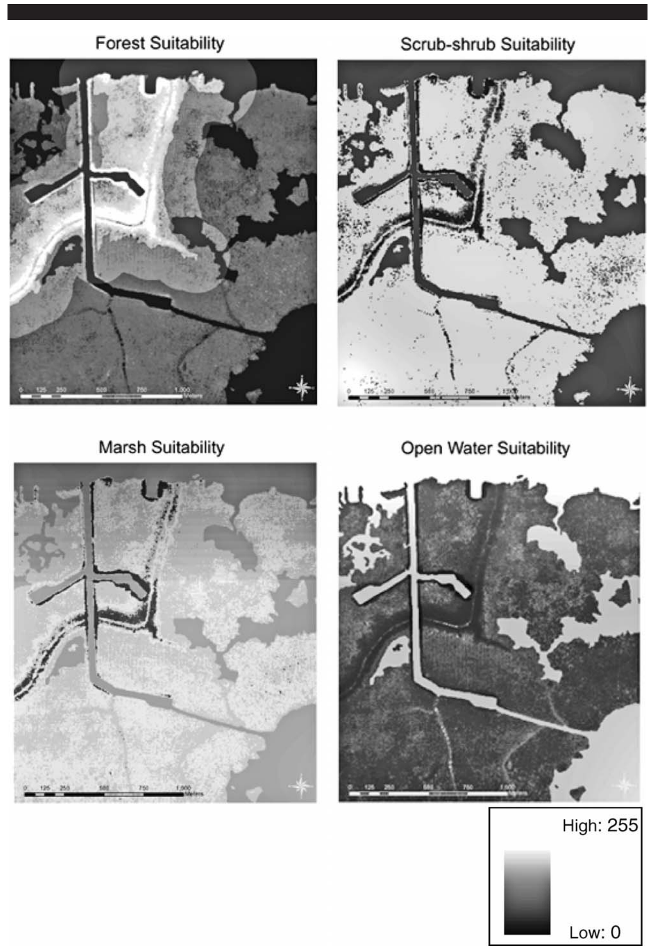

Figure 3. Suitability images for each land-cover class at Mandalay National Wildlife Refuge. Suitability values range from low (0) to high (255) suitability.

Model Validation

To evaluate model accuracy, land cover was simulated for 1998 and 2004, and each simulation was compared with its reference image and to a null, no change model (Pontius and Malanson, 2005). Data from the two previous study dates were used as model inputs for each simulation. The amount of land cover for each category in the 1998 simulation was projected from 1978 based on the probability of state transition between 1952 and 1978. For the 2004 simulation, land-cover quantities were estimated from state transition probabilities between 1978 and 1998. The null model in each case was the land-cover data from the study date preceding the date of the relevant simulated image. Thus, the null model was the 1978 image for the 1998 simulation, and the 1998 image for the 2004 simulation. Agreement between the null model and the reference image was compared to agreement between the simulated and reference images. If the simulated image more closely matches the reference image than the null, then having the model is better than having no model.

An essential measure of a model's utility is its ability to discriminate between errors resulting from inaccurate quantities of land cover (determined by MCA) and those affecting the placement of land-cover cells (determined by CA and suitability images) (Pontius, 2000, 2002). With this information, the model can be improved by isolating the model input that is responsible for more errors. Quantitative and spatial errors were separated by quantifying agreement and disagreement at varying resolutions. Starting with 1,027,580 fine resolution grid cells (the resolution of the input data), cells were aggregated using a geometric sequence until only one cell remained. Cells retain information on the abundance of each class by allowing partial agreement of cells as they are aggregated (Pontius, 2002). At the resolution of the raw data, disagreement results from both errors in location and quantity. With the entire landscape aggregated into only one cell, disagreement results only from quantitative error, and quantitative error remains constant across all resolutions (Pontius, 2002). Spatial error can then be calculated by subtracting the quantitative error from the total disagreement at each resolution. If error due to location shrinks quickly with

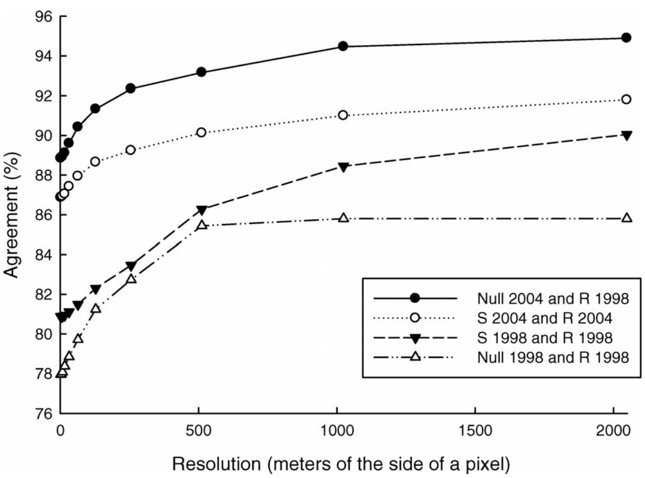

Figure 4. Comparison of the percentage total agreement between simulated and reference models, and null and reference models for MDNWR at varying resolutions beginning with the resolution of the raw data and ending with only one coarse cell remaining. Total agreement includes agreement due to chance, quantity, and location. Total disagreement includes disagreement due to quantity and location at all resolutions except the coarsest resolution, at which all of the remaining disagreement is disagreement due to quantity. Disagreement due to quantity remains constant across all resolutions.

increasingly coarse resolutions, then although the cell is classified incorrectly, it was not far from the correct location. With the entire landscape aggregated into only one cell, all remaining error is due to quantity (Pontius, 2002).

Projecting Future Change

After model validation, land cover was projected from 2004 to 2005, 2015, and 2050 using three different scenarios of different rates and types of change. Future land-cover images produced for each scenario represent what future land cover might look like if trends continued as they had during the periods from which the probabilities of state change were calculated. Scenario 1, which was based on probabilities of change from 1998 to 2004, was selected because it was the most recent period, and therefore representative of current conditions, which appeared to be influenced by the installation of shoreline protection structures along the GIWW in 2003. Specifically, the trend of marsh replacement by open water near the GIWW was reversed (Figure 4). For scenario 1 only, the factors that increased the likelihood of marsh loss and open water gain near the GIWW were removed from the suitability images to allow for potential marsh expansion in this area. Scenario 2 was based on the probability of change between 1978 and 1998, when the most marsh to open water conversion occurred. Scenario 2 would more closely represent future conditions if the shoreline protection structures were only effective initially. Scenario 3, based on trends from 1940–2004, offers a long-term perspective.

RESULTS

Model Validation

The comparison of the simulated 1998 and 2004 images with their respective null models yielded mixed results in that only the 1998 simulation was more accurate than its null at all resolutions (Figure 4). The 2004 simulation was less accurate than its null at all resolutions (Figure 4). Disagreement due to quantity was smaller for the 1998 simulation and greater for the 2004 simulation than the respective null models. Disagreement due to location was greater for the 1998 simulation and smaller for the 2004 simulation compared with the null models. At the resolution of the raw data, 81% and 87% of the landscape was correctly classified in the simulated 1998 and 2004 images, respectively. Agreement increased with increasing resolution for both simulations because there was some location disagreement.

Agreement between the simulated and reference images for individual classes ranged from 33% for scrub-shrub in both simulations to 86% for forest in the 2004 simulation. Of the individual land-cover classes, scrub-shrub was the only category to be misclassified more often than correctly classified in the simulated images (Table 1). Most of the disagreement in scrub-shrub resulted from marsh being incorrectly classi-

Table 1. *Matrices comparing reference images to simulated images and null models used in model validation. Columns show percent of the landscape for simulated and null categories, and rows show reference values. Bold cells along the diagonal contain the amount of agreement between images.*

		1998 Simulation					1998 Null				
		FO	SS	M	OW	Total	FO	SS	M	OW	Total
1998 Reference	FO	**19**	0	1	0	20	**19**	0	1	0	20
	SS	0	**1**	1	0	2	0	**0**	2	0	2
	M	4	1	**43**	1	49	2	1	**45**	1	49
	OW	1	0	9	**18**	28	1	0	13	**14**	28
	Total	25	2	55	19	100	22	2	61	15	100
		2004 Simulation					2004 Null				
		FO	SS	M	OW	Total	FO	SS	M	OW	Total
2004 Reference	FO	**18**	0	1	0	19	**18**	0	0	0	19
	SS	0	**1**	1	0	2	0	**1**	1	0	2
	M	1	1	**44**	8	54	1	1	**47**	6	54
	OW	1	0	0	**23**	24	1	0	1	**22**	24
	Total	19	2	46	32	100	20	2	49	28	100

fied as scrub-shrub and *vice versa.* Misplaced scrub-shrub cells in the simulated image tended to be nearer to their correct location in the 2004 simulation than in the 1998 simulation (Figure 2). In the 2004 simulation, scrub-shrub patches that were incorrectly classified as marsh were located near scrub-shrub patches from the 1998 data. The model did not predict the loss of scrub-shrub near Lake Hatch between 1998 and 2004 (Figure 2).

Overall, forest was the most correctly classified, with agreement ranging from 76% to 86%. Some disagreement occurred as a result of forest being overestimated in the 1998 simulation by ~20 ha (19% of the total forest area in 1998). The 1998 simulation incorrectly extended the forest edge up to 130-m beyond the observed forest–marsh boundary in some places (Figure 2). The model also incorrectly placed a few patches of forest scattered throughout the marsh. These forest patches were located in cells that were scrub-shrub in 1978. Also, the model did not predict the canal extension that occurred between 1978 and 1998. The percentage of the landscape occupied by forest was the same in the 2004 simulation and reference images, although a few misplaced cells occurred (Table 1). Errors occurred along the forest edge where forest was incorrectly classified as marsh and *vice versa,* and in a narrow band along the GIWW where open water should have been forest (Figure 2).

The major discrepancies between simulated images and their respective reference images were in the marsh and open water categories. In the 1998 simulation, marsh was overestimated and open water was underestimated. The opposite was true of the 2004 simulation. Areas of incorrectly classified marsh and open water were mainly located near the GIWW but also in small waterways extending from canals and in small patches in the internal marsh matrix (Figure 2).

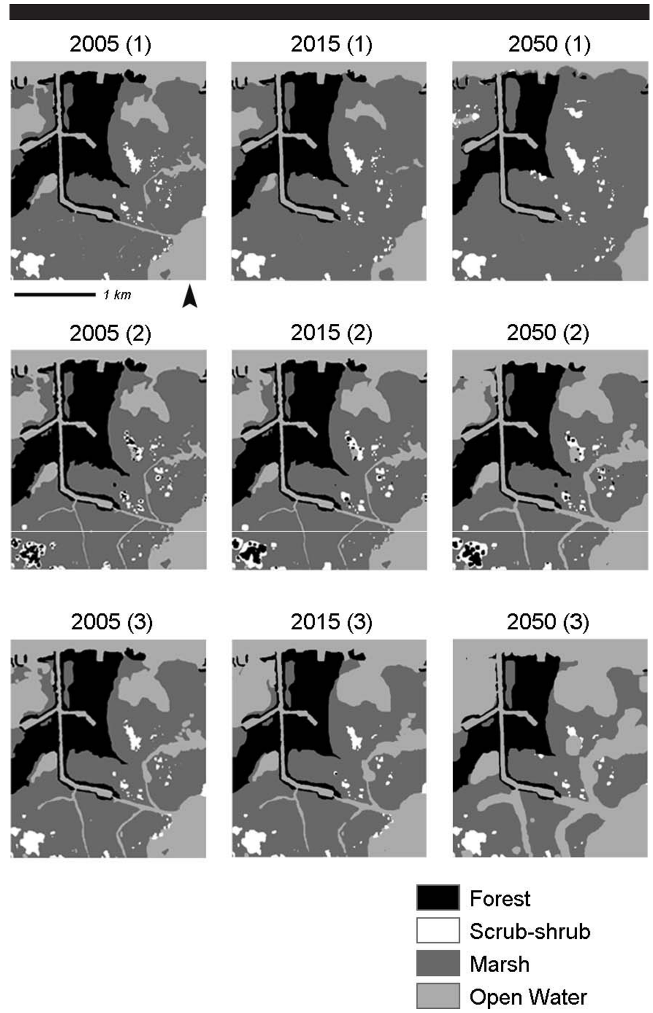

Figure 5. Images of potential future land cover for the three scenarios.

Future Projections

Scenario 1 represents a continuation of the most recent land-cover dynamics. The most obvious changes were the shrinking of the forest and expansion and amalgamation of the marsh. By 2050, marsh had even reclaimed some of the GIWW and had begun to encroach into Lake Hatch (Figure 5). Scenario 2 differed from scenario 1. In scenario 2, open water increased and marsh decreased. Scrub-shrub declined slightly, as did forest. By 2050, large expanses of marsh had converted to open water. These transitions occurred mostly along the GIWW but also along Lake Hatch and other waterways bordered by marsh. Scenario 3 offered a landscape that was most similar to the 2004 image in terms of open water and marsh. The amount of change in scenario 3 predicted for each class was intermediate between the amount predicted in scenarios 1 and 2, except for forest, which increased only in scenario 3.

DISCUSSION

Evaluation of Models

Pontius, Huffaker, and Denman (2004) determined that their model did not have predictive ability beyond that of a null model at the raw image resolution. Pontius *et al.* (2007) found that most modeling applications that demonstrated accuracy greater than the corresponding null model used information from the validation map. Our results show that one of our two model runs was better than a null model at all resolutions using no information from the reference image.

Model evaluation revealed that some land-cover types were harder to predict than others. Forest was the most consistently classified category, probably because of its relatively stable nature and the longevity of tree species (Brinson, Christian, and Blum, 1995). In addition to its stability in terms of quantity, forest, for the most part, existed as one large patch. The spatial contiguity of the forest enhanced the model's ability to predict the location of any forest transitions (*i.e.,* the model did not have to account for numerous and scattered small patches). Although it was the most unpredictable class according to agreement between the simulated and reference images (Table 1), scrub-shrub was better predicted by our models than the respective null models. Most of the unpredictability in terms of location was because scrub-shrub was usually taking over or being overtaken by the

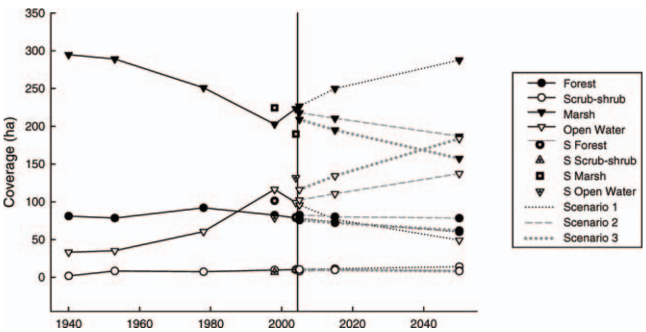

Figure 6. Observed and simulated land-cover quantities for each category through time at MDNWR. Simulated (S) values (1998 and 2004) for model validation have a dot in the symbol. Points to the right of the line are extrapolated values of the three scenarios.

floating marsh (BATTAGLIA, DENSLOW, and HARGIS, 2007; SHIRLEY and BATTAGLIA, 2006; WILLIAMSON, BARKER, and LONGSTRETH, 1984). Marsh was the most expansive category, so determining the location of a particular marsh to scrub-shrub transition proved very challenging and the scrub-shrub suitability layer we created was of limited utility because scrub-shrub occurs in a relatively wide range of elevations. Marsh was estimated slightly better by the null models, although agreement was over 65% for all null and simulated models.

Both simulations predicted open water better than the corresponding null models. Predicting locations where open water would occur benefited from the incorporation of topographic data. Unlike other classes, open water occurred only at a narrow range of elevations, rendering the suitability images very effective for predicting appropriate locations of open water in a low-lying system where relative elevation ranges from 0 to ~3 m. However, the use of elevation data from 2002 could have affected the model results differently over the 60-year study period. If elevation changes occurred between the dates of the input data and 2002, then the suitability images could have enhanced the model's ability to predict open water in years closer to 2002, and possibly diminished its ability to predict earlier trends in land cover or ones beyond 2002. Even if elevation did vary over the 60-year time span, low-lying areas should be first to be converted into open water; therefore, the suitability images should still be useful because they reflect *relative* elevation. The canal extension was not incorporated into the model, but planned canal extensions in the future could easily be taken into account.

The main limitation in the modeling process was the prediction of land-cover quantities, which were calculated using MCA. Simulated land-cover quantities varied depending on transition probabilities of the particular period used as a model input. In the past, some boundaries between the vegetation states shifted in one direction and then back in the reverse direction (Figure 6). These short-term dynamics in reference land-cover data were not captured in the simulated images. For example, the 1998 simulation overestimated forest because forest expanded in the 1953–1978 period and then receded from 1978 to 1998 (Figure 6).

The nonlinearity in forest dynamics may reflect species tracking natural and anthropogenically induced fluctuations in hydrology and other environmental factors. Initially, the marsh was converting to scrub-shrub (and eventually to forest) near existing forest. Those trends appear to reverse post-1978, coincidental with continued expansion of the Gulf Intracoastal Waterway (GIWW) and the addition of a new canal carved into the existing large forest patch. The wetter, more saline conditions could have prevented new establishment and persistence of forest and shrub species (WILLIAMS *et al.,* 1999a, 1999b). Lags in biological responses, including delayed mortality after exposure to abiotic stresses (MICHENER *et al.,* 1997) and recovery with reduction in stress, can influence onset and duration of such trends. The Markov probability overestimated the transition of scrub-shrub to forest from 1978 to 1998 because scrub-shrub along the forest edge did change to forest between 1953 and 1978. Instead of scrub-shrub being replaced by forest along the forest edge as it had been from 1953–1978, it began expanding into interior portions of the marsh according to the reference data. Internal scrub-shrub patches had only begun to appear in the marsh interior from 1953–1978, and the 1998 simulation began with the 1978 image. Because such a large proportion of scrub-shrub changed to forest in the reference images from 1953–1978, the model inevitably replaced some of the scrub-shrub in the marsh with forest, even though these areas had a low suitability for forest cover.

Similarly, the discrepancy in quantities for simulated and reference land cover for 2004 most likely resulted from the nonlinear trend characterizing the marsh and open water (Figure 6). Shoreline protection structures were constructed along the GIWW in 2003, primarily to reduce wave activity and concomitant erosion of the marsh. The land-cover changes suggest that these structures are promoting expansion of marsh into open water, particularly along the GIWW. Other studies have shown marsh increases following measures such as these designed to counteract land loss in the Mississippi Delta (COSTANZA, SKLAR, and WHITE, 1990; MARTIN *et al.,* 2002). Future modeling attempts might benefit from employing alternatives to MCA for predictions of land-cover quantities.

Future Scenarios

Although the shoreline protection structures appear to be effective at facilitating marsh reestablishment in recent years, the likelihood of marsh recovery continuing as projected in scenario 1 is probably low without additional management efforts because marsh is unlikely to expand beyond the structures. Additionally, the observed postmanagement marsh building occurred near the GIWW and the protection structures, so the filling in of Lake Hatch in scenario 1 is unlikely. These results are still encouraging for Mandalay and other similar low-lying coastal areas because they illustrate that these efforts were, at least initially, successful. If these structures cease to function, the outcome could be much worse, as indicated by the scenario 2 images.

Scenario 3 offered an intermediate picture of future land-cover change in terms of open water and marsh. However, this scenario is probably too optimistic because it places forest within scrub-shrub patches far from existing forest. This anomalous result stems from the input data used in the MCA (1940 and 2004). In 1940, some scrub-shrub patches that were located on the forest edge changed to forest in later years, and correspondingly, the Markov analysis predicted some scrub-shrub–forest transition. However, in the base image (2004), scrub-shrub patches are in a different location (and most likely consist of different species) than in 1940 (*i.e.,* scrub-shrub located in the marsh is more likely composed of true shrubs and scrub-shrub patches near the forest are probably juvenile trees). In the land-cover classification, these are not distinguishable, and so the MCA-specified scrub-shrub–forest conversions had to be located in the marsh where scrub-shrub patches occurred in 2004. A more accurate model could be produced if these two life forms in the scrub-shrub category could be separately delineated.

Ecological Implications

The model validations showed that forest was the most consistently and accurately classified land-cover type with 19% and 18% of all cells in the landscape correctly classified in the 1998 and 2004 models, compared to 6% and 3% misclassified (Table 1). The relative ease with which forest was modeled can be attributed to its stability. Past land-cover images revealed that the forest–marsh boundary, although not completely unchanged, shifted less than that of boundaries between other classes. Forest vegetation was relatively slow to change because the constituent species are generally long-lived and are able to survive for some period, and perhaps hinder encroachment of herbaceous marsh species, even after being exposed to physiologically untenable conditions (Brinson, Christian, and Blum, 1995). In recent years, however, trees near the forest–marsh ecotone have been unable to persist, and the forest has been slowly receding. The rate of forest loss, contingent upon coastal flooding and saltwater intrusion, may slow when water levels reach a certain elevation, but rising water will continue to gradually encroach upon the higher elevation areas of the natural levee that presently contain forest. Retreat of this coastal forest will likely continue with SLR (Williams *et al.,* 1999a), as illustrated by scenarios 1 and 2.

Unlike forest, scrub-shrub was the most difficult to predict. Even though our models better predicted occurrence compared with null models, only 1% of the landscape was classified correctly for every 2% that was misclassified for the 1998 and 2004 simulation. Scrub-shrub dynamics may be tied to the fluctuating environmental conditions of the floating marsh. Because the floating marsh is not subjected to overland flooding, establishment of shrub and small stature tree species can occur (Battaglia, Denslow, and Hargis, 2007; Swarzenski *et al.,* 1991). Once established, however, the trees and shrubs may weigh down the mat, causing mat flooding and eventually leading to their mortality (Williamson, Barker, and Longstreth, 1984). If this is the case, future models could take into account this cyclic trend and perhaps better predict scrub-shrub dynamics. Obviously if there is no longer a marsh mat, as is the case for much of the landscape in scenario 2, then shrub species would not have suitable habitat in which to establish and persist.

The largest transitions occurred between the marsh and open water categories between 1940 and 2004. These shifts likely reflect erosion of the floating marsh near the GIWW and other canals as well as chronic effects of SLR including coastal flooding and changes in salinity. Conversion from marsh to open water happened within the larger marsh matrix, but also along the shore of Lake Hatch. Historically, this was not the case. Russell (1942) used Lake Hatch as a "type" example of a lake that was shrinking as water was replaced by floating aquatics and eventually by emergent marsh. He predicted that Lake Hatch would be solid marsh in the near future. Although Russell's prediction held from 1940–1953, Lake Hatch expanded in the following years. The initial expansion of Lake Hatch coincided with canals being extended into the lake. With the canals present, it is unlikely that marsh species will be able to establish on the lake's edge (as in scenario 1). Similarly, development of many of the open water patches within the marsh seemed to correspond to GIWW widening. If the widening of the GIWW is the primary reason for open water encroachment, and if the shoreline protection structures continue to function, then marsh reestablishment could lead to a landscape with similar past marsh coverage (as shown in scenario 1). If management efforts are not used or become ineffective, the marsh could be greatly affected and, in some places, the coastal transition would be reduced to forest–open water, with marsh missing entirely (scenario 2).

Our study provides a bleak outlook for Mandalay National Wildlife Refuge and similar areas in the Mississippi Delta and other deltaic systems. Rapid marsh to open water conversion illustrated in scenario 2 could be a realistic outcome, considering premanagement changes observed at Mandalay. Although the shoreline stabilization structures are promising tools to counter erosion, it is unclear how long these measures will remain effective. As survival and regeneration become more difficult for species at their seaward limits, their ability to persist in the landscape will require them to establish successfully in areas further inland (Brinson, Christian, and Blum, 1995). Thus, inland areas suitable for plant establishment, such as reserve buffers, are critical for persistence of intact coastal communities.

CONCLUSIONS

Our validation results show that one of our two model runs predicted land cover better than a null model at all resolutions. Future land-cover images vary widely depending on alternative scenarios. If the shoreline protection structures installed in 2003 continue to be as effective as they were initially, then marsh loss may be reversed. However, if management ceases to be as effective and trends continue as they had in recent years, then dramatic marsh losses and some forest loss would be expected. Eventually, marsh may be lost altogether, creating a transition directly from forest to open water.

The methods used in this research can be applied to a variety of ecological systems and are especially appropriate for modeling dynamics of coastal vegetation and other wetland systems where the vegetation is closely linked to hydrology and elevation. Our models are designed for examining community responses to a range of scenarios and could be used to evaluate the effectiveness of management decisions.

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