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Likely Changes in Habitat Quality for Fish and Wildlife in Coastal Louisiana during the Next Fifty Years

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ABSTRACT

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Louisiana's 2012 Master Plan for a sustainable coast was designed to minimize economic damage from storm surges and to maximize wetland habitat for fish and wildlife. Selecting projects for inclusion in the master plan depended partly on models that simulated the effects of management options on environmental factors that control habitat quality for fish and wildlife. We used 13 models to predict the effects of the master plan on habitat quality for fish and wildlife in coastal Louisiana. Habitat quality was predicted to change more for the Neotropical songbirds and seven other modeled species losing habitat quality with the status quo (37%) than it was predicted to increase for five modeled species gaining habitat quality with the status quo (+18%). The master plan was predicted to slow or negate all changes associated with the status quo. All of the modeled fish and wildlife belong to people of the state of Louisiana, people living in countries bordering the Gulf of Mexico, and to people throughout the Americas. Thus, declining fish and wildlife habitat quality in Louisiana probably will cause market and nonmarket losses, which although concentrated in Louisiana, will extend across the Americas. As funding for Louisiana's master plan is pursued, it is important to consider that almost all of the causes for net wetland losses in Louisiana are external to the owners of these wetlands but that the fish and wildlife that use these wetlands belong to and benefit people throughout the Americas.

ADDITIONAL INDEX WORDS: *Habitat suitability index, American alligator, muskrat, juvenile spotted seatrout, juvenile brown shrimp, juvenile white shrimp, largemouth bass, eastern oyster, gadwall, green-winged teal, mottled duck, roseate spoonbill, and red swamp crawfish.*

INTRODUCTION

In North America, coastal wetlands are most abundant on the southeastern Atlantic coast of the United States and on the northern Gulf of Mexico. Field *et al*. (1988) classified coastal wetlands as salt marsh (39%), fresh marsh (14%), tidal flats (2%), or swamps (45%). Feld *et al*. (1988) estimated that Louisiana contained 39% of coastal salt marshes and 44% of coastal freshwater marshes in the conterminous United States; North Carolina (21%) and Florida (20%) also contained substantial areas of coastal wetlands (Table 1).

Coastal wetland loss is widespread worldwide. Dahl (2011) estimated that estuarine wetlands, *i.e*. salt marsh, in the 48 conterminous United States declined from 23,752 km^2 in 1984 to 23,412 km^2 in 2010 and that most of these losses occurred in Louisiana and Texas. Coastal wetland loss in Louisiana is especially rapid. Couvillion *et al*. (2011) estimated that coastal wetlands in Louisiana declined from 19,543 km^2 in the 1930s to 14,666 km^2 by 2010. Only a tiny fraction of the losses were caused by the active conversion of emergent wetlands into developed areas such as agriculture, ports, *etc*.; instead, the losses resulted from the passive conversion of emergent wetlands into shallow open water following plant death and erosion of the upper 30–100 m of soil. The causes of wetland loss in coastal Louisiana were reviewed by Boesch *et al*. (1994) and Day *et al*. (2001); they include a near total lack of natural

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Table 1. *A list of some of the notable resident and migratory fish and wildlife in coastal Louisiana (Bellrose, 1980; Boesch and Turner, 1984; Herke, 1979; McNease and Joanen, 1978; Palmisano, 1972;* Rogers et al., 1993*; Rozas and Hackney, 1984; Rozas and Reed, 1993).*

Common Name	Scientific Name
American alligator	*Alligator mississippiensis*
Nutria	*Myocastor coypus*
Muskrat	*Ondatra zibethicus*
Raccoon	*Procyon lotor*
Waterfowl	*Anser* spp., *Anas* spp., *Aythya* spp., *Mergus* spp., *etc.*
Woodcock	*Scolopax minor*
River otter	*Lutra canadensis*
White-tailed deer	*Odocoileus virginianus*
Mink	*Mustela vison*
Rabbit	*Sivilagus* spp.
Squirrel	*Sciurus* spp.
Snapping turtle	*Macroclemys temmincki*
Largemouth bass	*Micropterus salmoides*
Blue catfish	*Ictalurus furcatus*
Channel catfish	*Ictalurus punctatus*
Flathead catfish	*Pylodictus olivairs*
Yellow bass	*Morone mississippiensis*
Striped bass	*Morone saxatilis*
Warmouth	*Lepomis gulosus*
Bluegill	*Lepomis macrochirus*
Sac-a-lait	*Pomoxis annularis*
American eel	*Anguilla rostrata*
White shrimp	*Penaeus setiferus*
Brown shrimp	*Penaeus aztecus*
Menhaden	*Brevoortia patronus*
Speckled seatrout	*Cynoscion nebulosus*
Blue crab	*Callinectes sapidus*
Southern flounder	*Paralichthys lethostigma*
Redfish	*Sciaenops ocellatus*
Sheepshead	*Archosargus probatocephalus*
Bay anchovy	*Anchoa mitchilli*
Spot	*Leiostomus xanthurus*
Baitfish	*e.g. Fundulus* spp., *Poecilia* spp., *Notropus* spp.
Atlantic croaker	*Micropogonias undulatus*

wetland creation by the Mississippi River because of maintenance of deep-draft navigation on the Mississippi River, natural wetland loss processes associated with the delta lobe cycle (Coleman, 1988), an acceleration of natural wetland loss processes caused by levees that prevent spring river flooding of existing wetlands, salt-water intrusion into existing wetlands caused by navigation canals that run from oceans to inland, flooding of existing wetlands caused by spoil banks on canals that run parallel to the coastline, and petroleum extraction that enhances subsidence of existing wetlands.

Coastal wetlands deliver various ecosystem services that include providing habitat for fish and wildlife, improving water quality, and reducing storm surge; many of these services are external to the owners of the wetlands (Nyman, 2011). Our focus is on the changes expected to occur during the next 50 years in ecosystem services that result from fish and wildlife that use coastal Louisiana. We are unaware of population estimates for all of the fish and wildlife that are supported by Louisiana's coastal wetlands or even a list of the relevant species. Here we list some of the notable resident and migratory fish and wildlife (Table 2); they include migratory waterfowl, shorebirds, and songbirds, resident fish and wildlife, and fish and crustaceans that depend upon emergent wetlands for only part of their life cycle. It is anticipated that wetland loss will reduce recreational and commercial fish harvests as marsh to water ratios decline (Browder *et al.*, 1989). O'Connell *et al.* (2005) thoroughly assessed wildlife and fish use of Louisiana's barrier islands and concluded that most of Louisiana's threatened and endangered species depended upon barrier islands, which, in addition to providing nesting habitat for shorebirds and sea turtles, also increase the diversity of depth, wave energy, and salinity conditions in an area and thus increase diversity of aquatic organisms that use adjacent areas.

Louisiana's 2012 Master Plan for a sustainable coast (CPRA, 2012) was designed to minimize economic damage to developed areas from storm surges and to maximize the area of land built and/or sustained while maintaining coastwide habitats for fish and wildlife. Selection of storm flood protection and coastal restoration projects for inclusion in the master plan depended partly on models that simulated the effects of protection options, which generally decrease habitat quality for fish and wildlife, and restoration options, which generally increase habitat quality for fish and wildlife, on environmental factors that control habitat quality for fish and wildlife (see other articles in this issue). Here, we describe the results of using these models to predict the effects of flood protection projects and coastal restoration projects, hereafter collectively referred to management, on fish and wildlife that use Louisiana's coastal wetlands.

METHODS

Selection of the particular species of fish and wildlife to model is a key step in evaluating ecosystem services because the new habitat likely will increase the abundance of some species but reduce the abundance of other species. In coastal Louisiana, where shallow open water is replacing emergent wetlands, a focus on species abundant in open water would indicate a net increase in ecosystem services, whereas a focus on those abundant in emergent wetlands would indicate a net decrease in ecosystem services. Prior plans to restore wetlands in coastal Louisiana (USCOE, 2004; CPRA, 2007) also considered output from models of animal habitat but used different procedures to select the animals to model. Foret *et al.* (2004) selected the animals to model for those efforts with the intent of equally balancing animals that depended upon saline and fresher parts of the estuary (two of us were on that team: J.A. Nyman and D.M. Baltz). The fish and wildlife models used during those prior plans (USCOE, 2004; CPRA, 2007) were juvenile Gulf menhaden (*Brevoortia pratronas*), juvenile spotted seatrout (*Cynoscion nebulosus*), juvenile Atlantic croaker (*Micropogon undulatus*), largemouth bass (*Micropterus salmoides*), brown shrimp (*Farfantepenaeus aztecus*), white shrimp (*Litopenaeus setiferus*), eastern oyster (*Crassostrea virginica*), American alligator (*Alligator mississippiensis*), dabbling ducks (*Anas* spp.), mink (*Mustela vison*), muskrat (*Ondatra zibethicus*), and river otter (*Lontra canadensis*) (Foret *et al.*, 2004). We declined to select the animals to model for the 2012 plan and instead requested that restoration planners and stakeholders select the animals to model because we assumed that they would select fish and wildlife most associated with commercial,

Table 2. *Estimates of the area of coastal wetlands in the southeastern United States and the conterminous United States (data from Field* et al.*, 1988).*

Location	Coastal Salt (km^2)	Coastal Fresh (km^2)	Tidal Flats (km^2)	Swamp (km^2)	Total (km^2)
Louisiana	7076.3	2787.5	0	1769.3	11,633.1
North Carolina	642.6	372.3	0	8528.8	9543.7
Florida (Gulf)	1745.4	305.5	0	3928.3	5979.2
Florida (Atlantic)	388.1	1551.6	0	1048.1	2987.8
Georgia	1514.7	127.5	38.4	1157.4	2838.1
Texas	1579.9	318.5	0	163.1	2061.5
South Carolina	1495.3	261.0	0	0	1756.3
Alabama	59.1	42.9	0	612.3	714.3
Mississippi	259.0	16.2	0	307.6	582.7
Conterminous U.S.	17,993.6	6388.8	857.1	20,566.1	45,800.3

recreational, and nonmonetary services. Restoration planners and stakeholders requested 17 models. We created 14 of those models but declined to model black drum (*Pagonias cromis*), Gulf sturgeon (*Acipenser oxyrinchus*), and piping plover (*Charandrius melodus*) because of the current lack of precision in crucial input variables. The 14 models that we created were: American alligator, muskrat, river otter, juvenile spotted seatrout , juvenile brown shrimp, juvenile white shrimp, largemouth bass, eastern oyster, gadwall (*Anas strepera*), green-winged teal (*Anas crecca*), mottled duck (*Anas fulvigula*), roseate spoonbill (*Platalea ajaja*), neotrophic migrants (varied species), and red swamp crawfish (*Procambarus clarkii*). We omitted the model for river otters from this report because its output over time varied in ways that appeared unrelated to input parameters even though its initial output appeared to correlate with otter distributions inferred from recent trapping patterns.

Our models simulated 342,233 cells (500 m by 500 m) that covered coastal Louisiana for 50 years. Our models were not population models; *i.e.* they did not estimate population size in each cell. Instead, each model estimated the capacity of each cell to support each species and thus may be classified as habitat suitability index (HSI) models (USFWS, 1981). HSI models were developed to assess habitat quality based on field measurements of smaller-scale units but now also are used with remotely sensed landscape-level variables (see Tirpak *et al.*, 2009, and literature cited therein). Tirpak *et al.* (2009) recently verified 37 of the 40 models they assessed and concluded that HSI models can be useful tools for conservation planning but more accurately predict habitat quality for abundant species than for rare species. The major caveat of using HSI models is that predicted changes in habitat quality may or may not translate into actual changes in numbers of fish and wildlife because factors other than habitat quality, such as harvest mortality, affect the numbers of fish and wildlife. The same is true for migratory species—predicted changes in habitat quality may not translate into actual changes in numbers of wildlife in Louisiana because of factors on breeding grounds.

Environmental variables that were considered important at causing differences in habitat quality for each species, and that were available, were used as input. These input variables were themselves the output of other models that predicted water depth, water salinity, plant community type, wetland area, and edge habitat (defined as areas of open water within 10 m of the edge of emergent vegetation) for the cells for 50 years. The models for which output was used as input for these fish and wildlife models are described elsewhere in this issue. Input variables for the fish and wildlife models were converted to a unitless index that varied from 0 to 1, with 1 representing ideal conditions for a particular species or species group. All environmental indices then were averaged with either arithmetic means or geometric means to produce a unitless number that ranged from 0 to 1 and that represented habitat quality in each cell for a particular species. Arithmetic means (*i.e.* $x = [a + b]/2$, or $x = [a + b + c]/3$, *etc.*) were used when appropriate, such as when determining the value of a cell that contained more than one habitat type, but the final calculation generally involved geometric means (*i.e.* $x = [a \times \mathrm{b}]^{1/2}$, or $x = [a \times b \times c]^{1/3}$, *etc.*) because the response of fish and wildlife populations to environmental variables often is more similar to the way geometric means respond to input variables than the way arithmetic means respond to input variables. For example, alligators cannot survive long if salinities exceed 10 ppt. If all five environmental variables in the American alligator model were ideal in a cell except that salinity was 15 ppt, then American alligators could not reproduce and persist there. If an arithmetic mean was used for such a cell, then the model incorrectly would predict that the cell could support 80% of the alligators in an ideal cell. If geometric means were used, however, then the model correctly would predict that the cell could support no alligators. Except for when zeroes are present in these models, geometric means and arithmetic means behave similarly (Table 3). Our models thus contained a mix

Table 3. *Comparison of arithmetic and geometric means of environmental indices.*

Index a	Index b	Index c	Index d	Index e	Arithmetic Mean of a through e	Geometric Mean of a through e
0	1.0	1.0	1.0	1.0	0.80	0.00
0.2	1.0	1.0	1.0	1.0	0.84	0.72
0.8	1.0	1.0	1.0	1.0	0.96	0.96
1.0	1.0	1.0	1.0	1.0	1.00	1.00

Table 4. *Projected trends in habitat quality for some fish and wildlife in coastal Louisiana under different various environmental scenarios, with and without the 2012 Louisiana Master Plan. Within each of the three environmental scenarios (moderate, less optimistic, and moderate with high sea-level rise), projections with the master plan are compared to projections without the master plan.*

	Moderate (%)		Less Optimistic (%)		Moderate with High Sea-Level Rise (%)	
Species	Without	With	Without	With	Without	With
Likely to increase with or without the master plan						
Eastern oyster	236	213	115	106	116	107
Largemouth bass	105	113	101	116	102	114
Likely to increase without the master plan but to decline with the master plan						
Brown shrimp	114	97	111	94	119	96
Spotted seatrout	106	87	112	94	115	93
White shrimp	105	93	103	97	106	98
Likely to decline without the master plan but increase with the master plan						
Gadwall (duck)	87	109	81	116	68	99
Crayfish	80	107	72	107	74	115
Likely to decline without the master plan but decline less with the master plan						
Mottled duck	83	96	84	105	67	88
Neotropical birds	83	96	64	80	66	84
Green-winged teal (duck)	80	77	70	77	49	61
Roseate spoonbill	72	86	59	70	54	66
American alligator	78	84	31	42	13	24
Muskrats	64	57	23	27	15	19

of arithmetic and geometric means to account for different relationships between input variables and habitat quality.

The effects of flood control and coastal restoration projects on fish and wildlife will depend partly on environmental conditions that cannot be precisely predicted. Therefore, model runs were made using a variety of environmental conditions, including sea-level rise, subsidence, hurricane frequency, hurricane intensity, Mississippi River discharge, rainfall, evapotranspiration, Mississippi River nutrient concentration, and marsh collapse threshold (*i.e*. the amounts of flooding and salinity stress required to convert emergent wetlands into open water), which were described by Peyronnin *et al*. (2013). The effects of flood control and coastal restoration projects on wildlife also will depend partly on ecological relationships not yet fully described. We are fairly certain that future conditions will not be outside the highest and lowest model input conditions. We also are fairly certain that predictions of differences among model runs probably are more certain than predictions of similarities among model runs because of the simplicity and sensitivity of these models. Our models are described in detail elsewhere (Baltz, 2012a, b, c; Kaller, 2012; Leberg, 2012a, b, c, d, e; Nyman, 2012 a, b; Romaire, 2012; Soniat, 2012) and are summarized here in the Appendix.

RESULTS AND DISCUSSION

Examination of the model output led us to develop two unrelated classification systems. One system classified modeled species based on habitat quality with and without the master plan after 50 years (Table 4). The other system classified modeled species based on the way in which habitat quality responded to environmental variability and the master plan over time (Figures 1, 2, and 3). Both systems are discussed here.

The models predicted that if the status quo (or future without action) continues (*i.e*. the master plan is not implemented), habitat quality will decline during the next 50 years for Neotropical migrant songbirds and eight other modeled species (Table 4). Altering the status quo by implementing the flood protection and coastal restoration projects included in Louisiana's 2012 Master Plan was predicted to reverse declines for two of these species, crawfish and gadwall, but merely slow those declines for Neotropical migrant songbirds and the other five modeled species (Table 4). High-quality habitat for most of these species depends partly upon emergent wetlands dominated by *Spartina patens*, or fresher marsh, which are predicted to decline, but at a slower rate if the master plan is implemented. We attributed the negative effects of the status quo on habitat quality for these species on the relatively slow rate at which the Mississippi River is creating new lower-salinity wetlands (Roberts, 1997), the ongoing conversion of emergent wetlands into shallow, open water (Couvillion *et al*., 2011), and the ongoing declines in area of less saline wetlands and increases in area of more saline wetlands (Chabreck and Linscombe, 1982). We attributed the beneficial effects of the master plan on habitat quality for these species to result from the effects of the master plan in increasing the rate at which the Mississippi River builds new lower-salinity wetlands, in reducing water salinity in existing emergent wetlands, and in slowing conversion of existing emergent wetlands into shallow, open-water areas.

Habitat quality for five modeled species was predicted to increase with the status quo (Table 4). Habitat quality for two of these species, the eastern oyster and the largemouth bass, was predicted to increase regardless of whether or not the master plan is implemented (Table 4). We are confident that the master plan could increase habitat for eastern oyster even more if some adjustments were made. Two of the five oyster

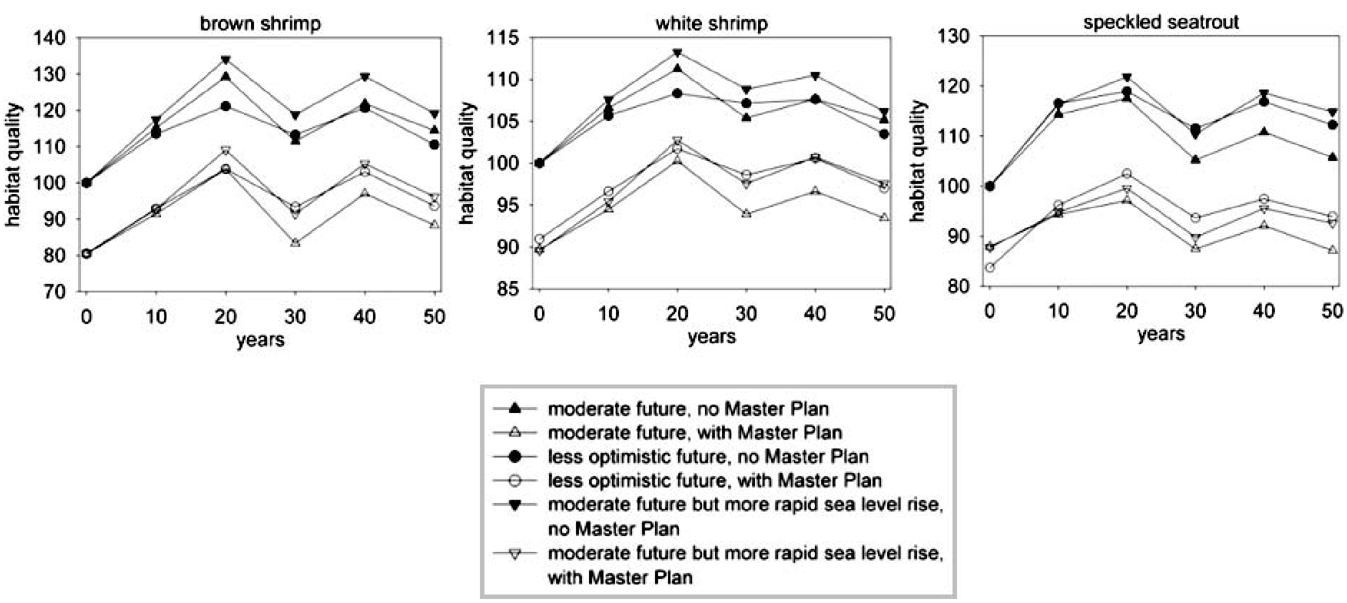

Figure 1. Predicted changes in brown shrimp, white shrimp, and spotted seatrout.

reef projects in the master plan (CPRA, 2012, p. 22) are sited in an area too fresh to support oysters, *i.e*. within 50 km of the mouth of the Atchafalaya River. These would be better located in terms of oyster habitat if they were in areas farther from the Atchafalaya River with higher salinity. Model output showed that the oyster reef projects between Vermilion Bay and Atchafalaya Bay failed to increase habitat quality for eastern oysters. CPRA now recognizes that those oyster reef projects

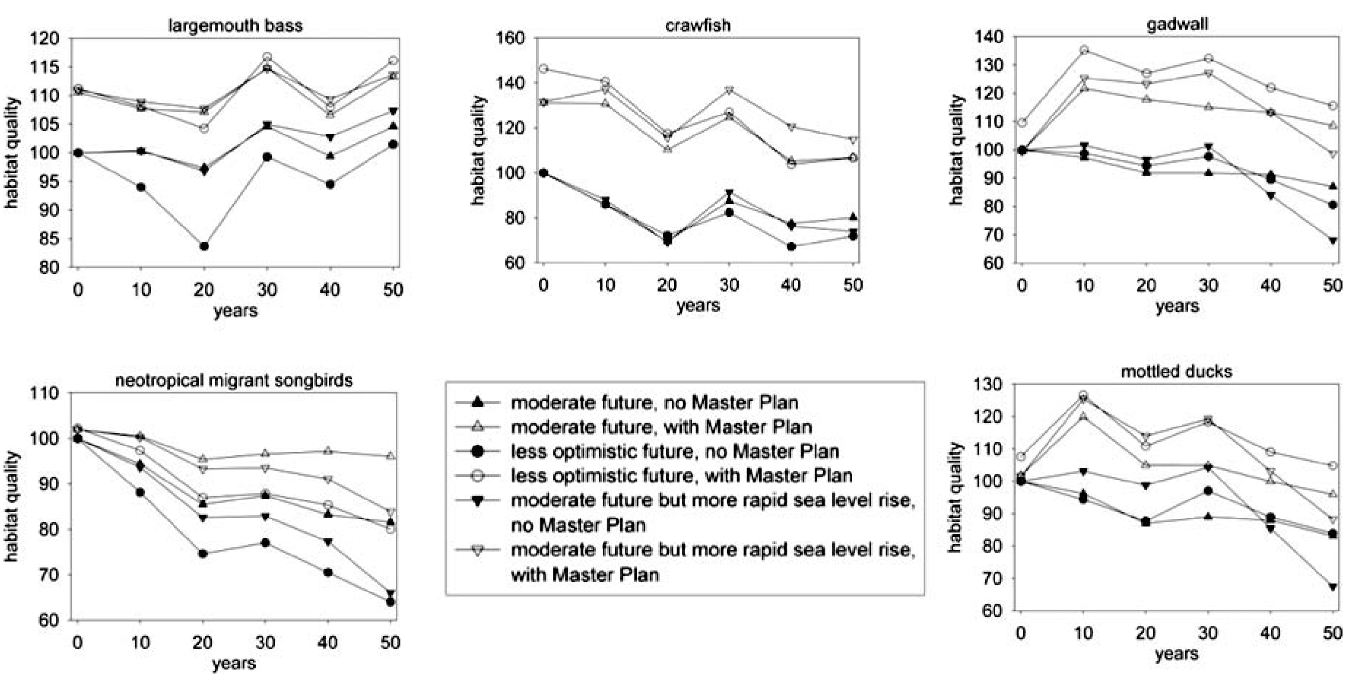

Figure 2. Predicted changes in bass, crawfish, Neotropical migrants, and two waterfowl species.

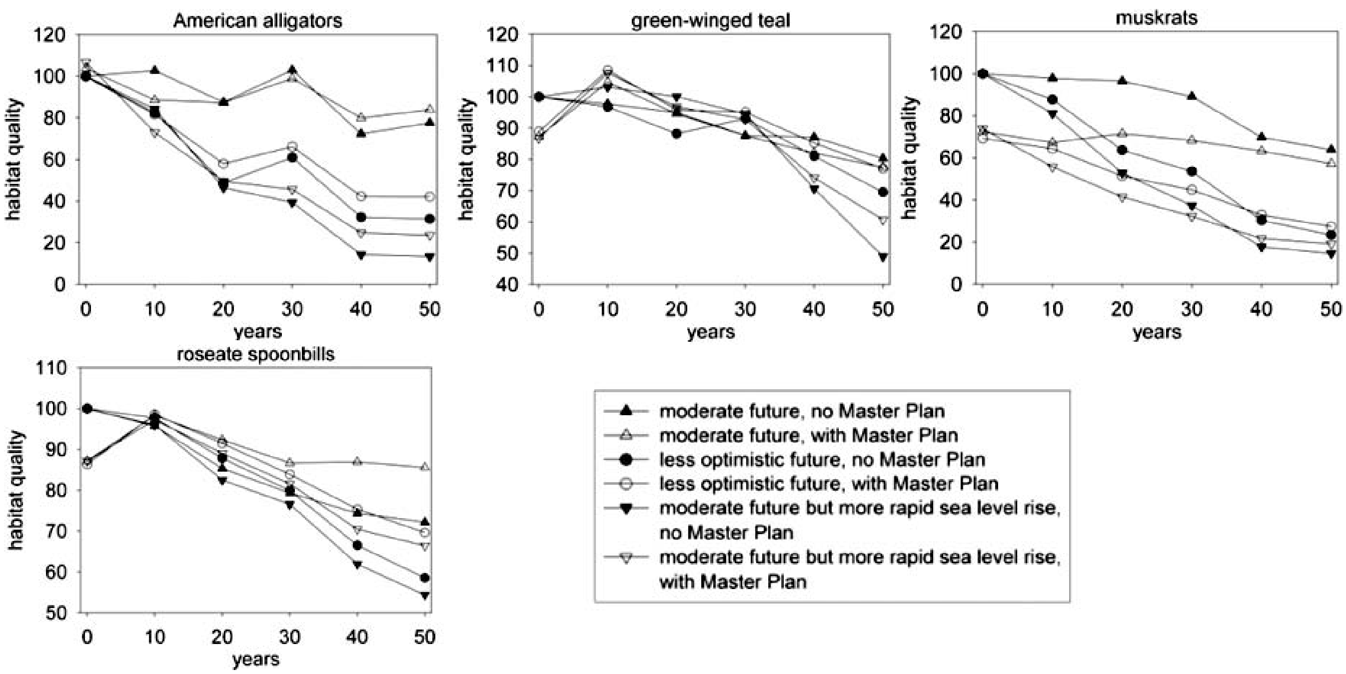

Figure 3. Predicted changes in alligators, green-winged teal, muskrats, and roseate spoonbills.

are not in a location that will foster oyster production and will revise those projects to a proper location. Habitat quality for the other three modeled species was predicted to increase with the status quo but not if the master plan is implemented (Table 4). All five of these modeled species depend less upon emergent wetlands than the other species modeled and are considered to be recreationally or commercially valuable seafood. All, except the largemouth bass, depend upon habitats saline enough to be dominated by *Spartina alterniflora*. We attributed the positive effects of the status quo on habitat quality for these species to ongoing conversion of emergent wetlands into shallow, open water (Couvillion *et al*., 2011) and to ongoing declines in area of less saline wetlands and increases in area of more saline wetlands (Chabreck and Linscombe, 1982). We attributed the negative effects of the master plan on habitat quality for three of these species to result from the master plan reducing water salinity in existing emergent wetlands and at slowing conversion of existing emergent wetlands into shallow, open-water areas.

Habitat quality is predicted to change more for the modeled species losing habitat quality with the status quo, *i.e*. 37% (49% to 22% depending upon environmental conditions; averaged from data in Table 3), than it is predicted to increase for the modeled species gaining habitat quality with the status quo, +18% (+8% to +33% depending upon environmental conditions; averaged from data in Table 3). With management, our models predict that these losses can be slowed to 21% (30% to 11% depending upon environmental conditions; averaged from data in Table 3).

The classification of the modeled species as gaining or losing habitat quality provides a useful summary for communicating large-scale patterns, but, alone, that type of classification only grossly describes likely changes in habitat quality for fish and wildlife. When the changes over time were examined for each model, a different scheme emerged that consisted of three broad patterns among the modeled species. In one pattern that was reflected by three modeled species, the master plan resulted in less habitat quality regardless of the environmental conditions that were used (Figure 1). In all three models, the master plan appeared to be responsible for more variation in habitat quality than environmental conditions (Figure 1). All three of these species, the spotted seatrout and both shrimp species, depend upon saline conditions for high-quality habitat.

In a second pattern that was reflected by Neotropical migrant songbirds and four other modeled species, the master plan resulted in more habitat quality regardless of the environmental conditions that were used (Figure 2). In all five models, the master plan appeared to be responsible for more variation in habitat quality than environmental conditions (Figure 1). For largemouth bass, gadwall, crawfish, and mottled ducks, the increases in habitat quality were substantial, with some approaching 50% (Figure 2). All four of these species depend upon fresh or low-salinity conditions. The trend for Neotropical migrant songbirds was similar in that all of the futures with the master plan produced higher-quality habitat than without the Mater Plan, but there were declines nonetheless with the master plan (Figure 2).

The third pattern was reflected by four modeled species. Unlike the previous two patterns, which appeared to be dominated by whether or not the master plan was implemented, this pattern appeared to be dominated by the selection of the likely future environmental conditions (Figure 3). The master plan produced higher-quality habitat within a given future for three of these modeled species. We attributed these responses to these species' dependence on lower-salinity wetlands. Of these four modeled species, only the muskrat was not benefited by the master plan (Figure 3). We attributed this response to the muskrat's dependence upon unbroken, brackish marsh.

CONCLUSIONS

Habitat quality for fish and wildlife is declining in coastal Louisiana because of the slow rate at which the Mississippi River is creating new lower-salinity wetlands and because existing wetlands are becoming more saline and/or are converting to open water. All of the modeled species for which habitat is predicted to decline belong to the public, *i.e*. to people of the state of Louisiana in the case of resident animals, to people living in countries bordering the Gulf of Mexico in the case of the estuarine-dependent fish, and to the people of the Americas in the case of the migratory birds. It would be difficult to monetize the costs of a 37% decline in habitat quality for these modeled species in coastal Louisiana. While some are only indirectly important economically (by contributing to the food webs that support economically important species), the costs are likely significant given that Louisiana's coastal wetlands represent 39% of coastal salt marshes and 44% of coastal freshwater marshes in the conterminous United States. Thus, declining habitat quality in Louisiana probably will cause market and nonmarket losses, which although concentrated in Louisiana, will extend across the Americas. Almost all of the causes for net wetland losses in Louisiana are external to the owners of these wetlands. These causes, reviewed elsewhere (see Boesch *et al*., 1994; Day *et al*., 2001), are associated primarily with dams in the upper watershed (for irrigation, flood control, and power generation) and with deep-draft navigation in the final 300 km of river (primarily to export agricultural products, coal, refined petroleum products, *etc*., *via* the Mississippi River or to import crude oil *via* the Calcasieu Ship Channel). Dams in the upper watershed affect habitat quality in coastal Louisiana by decreasing the availability of mineral sediments in the lower river (Kesel, 2003), which reduces the capacity of the river to build new wetlands *via* the delta lobe cycle (Roberts, 1997).

Management of the lower 300 km of the Mississippi River for deep-draft navigation also reduces habitat quality, primarily through a dependence upon the Old River Control Structure, which prevents the Mississippi River from switching the bulk of its flow down the Atchafalaya River (Winer, 2011). Moreover, river levees have prevented the spring flood of the river from reaching existing, adjacent wetlands and precluded the river from building new wetlands adjacent to its channel (Boesch *et al*., 1994; Day *et al*., 2001). In southeast Louisiana, natural subsidence of relict delta lobes contributes greatly to wetland loss, but such natural wetland loss would be slower in existing wetlands and offset by natural creation of new wetlands by the river if the river were not managed by people. In southwestern Louisiana, a deep-draft navigation channel from the Gulf of Mexico through Calcasieu Lake to Lake Charles increased tidal range (Zhang *et al*., 2011) and salinity, which led to wetland loss in adjacent wetlands either because the changes were too rapid (Gosselink, Cordes, and Parsons, 1979), or because the soils lacked enough iron to support higher-salinity wetlands (Nyman, DeLaune, and Patrick, 1990). Some of the costs of dams and deep-draft navigation are thus external to the economic beneficiaries of these practices; instead, the costs are borne primarily by people in south Louisiana, but also throughout the Americas. Addressing such market failures requires learning how natural ecosystems sustain valued services, communicating this information to the public and decision-makers, developing effective economic incentives, and fostering cooperation among different public and private interests (McNeely, 1992). Mechanisms for addressing such market failures in a largely voluntary, Pareto-improving manner include restoration cost-share and grant programs such as the Prairie CARE program of the North American Waterfowl Management Plan (Taylor, Gray, and Rosaasen, 1992) and the CRP program of U.S. Department of Agriculture (Lant *et al*., 2005). If funding for the coastal restoration aspects of Louisiana's 2012 Master Plan can be secured and implemented, then we predict that declines in habitat quality for fish and wildlife can be reduced from a 37% decline to a 21% decline (30% to 11% decline depending upon environmental conditions). As funding for these projects are pursued, it is important to consider how dams and deep-draft navigation have impacted the habitat quality and migratory patterns of fish and wildlife species, and to consider who owns and benefits from the fish and wildlife that use coastal Louisiana. Such information is vital for facilitating the cooperation of different economic and governmental players and for securing funding to implement Louisiana's 2012 Master Plan for a sustainable coast.

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APPENDIX

Eastern Oyster Habitat Suitability

Soniat (2012) described the eastern oyster model in detail; here we briefly summarize the model. The overarching assumption is that oyster habitat quality can be modeled as suitable salinity over suitable substrate. Suitable cultch (SI_1) was expressed as the percentage of the bottom covered with hard substrate (*e.g*. oyster shell), with optimal conditions occurring when the percentage of bottom covered with hard substrate was 50%. Suitable salinity was resolved into three salinity-based variables that addressed different aspects of the oyster's dependency on salinity. SI_2 was optimal when mean salinity during the spawning season (May through September) was between 18 and 22 ppt. SI_3 was assumed to be optimal when the lowest annual salinity exceeded 8 ppt. This relationship between minimum salinity and SI did not include any potential positive benefits of floods, such as reducing predators and disease (*e.g*. La Peyre, Gossman, and La Peyre, 2009). Annual mean salinity (SI_4) was assumed to be ideal when it was between 10 and 15 ppt. The final variable was percent land (SI_5), which was assumed to be ideal when it was 0%. Our model was:

$$\mathrm{HSI} = (\mathrm{SI}_1 \times \mathrm{SI}_2 \times \mathrm{SI}_3 \times \mathrm{SI}_4 \times \mathrm{SI}_5)^{1/5}.$$

We were unable to validate our oyster model as we've validated an earlier oyster model (Soniat and Brody, 1988), but we compared known spatial patterns in eastern oyster abundance with predicted spatial patterns in habitat suitability for eastern oysters and concluded that the model output was valid.

Largemouth Bass Habitat Suitability

Kaller (2012) described the largemouth bass model in detail; here we briefly summarize the model. We modified the Stuber, Gebhart, and Maughan (1982) largemouth bass HSI, which predicted habitat suitability from 22 variables that described the habitat's ability to provide food and cover, support appropriate water quality, and provide opportunity for reproduction. Not all 22 original variables were available to us; however, variables were available that were appropriate to describe food, cover, water quality, and opportunity for reproduction.

V_1 percent emergent vegetation per 500 m^2
V_2 average water temperature for April to August
V_3 maximum yearly salinity for June to August
V_6 percent of cell that is Submersed Aquatic Vegetation (SAV) per 1 km^2
V_7 index value of primary productivity in open waters

Variables based on water depth, SI_4, and habitat similarity, V_5, initially were proposed; however, through an iterative process in cooperation with personnel at the Coastal Protection and Restoration Authority of Louisiana during model output quality review, the variables were removed. The resulting model was:

$$\mathrm{HSI} = (\mathrm{SI}_1 \times \mathrm{SI}_7)^{1/8} \times (\mathrm{SI}_1 \times \mathrm{SI}_6)^{1/8} \times (\mathrm{SI}_2 \times \mathrm{SI}_3 \times \mathrm{SI}_7)^{1/12} \times (\mathrm{SI}_2)^{1/4}.$$

Food was described as being dependent upon emergent vegetation (SI_1) or chlorophyll *a* (SI_7). The % emergent vegetation variable was used as a surrogate for Stuber, Gebhart, and Maughan's (1982) % bottom cover variable, because largemouth bass in these habitats forage in and around vegetation rather than off the bottom (Maceina, 1996; Hoyer *et al.*, 2008). SI_1 was assumed to be optimal when an area had 30%–50% emergent vegetation. The specific relationship between % emergent vegetation and largemouth bass suitability was based on Sammons, Maceina, and Partridge (2005); Hoyer *et al*. (2008); Reed and Pereira (2009); and Fries (2010). Chlorophyll *a* was included as the second food variable because it was highly associated with better Louisiana largemouth bass conditions (Fries, 2010). SI_7 was assumed to approach optimal conditions as chlorophyll *a* exceeded 50 μg L ^{1}. Stuber, Gebhart, and Maughan (1982) described cover by % bottom cover, pool, and backwater area, and water-level fluctuation. In our model, cover was described by % emergent vegetation and % SAV because the other variables were not relevant (pool and backwater) or available (water-level fluctuation). The SAV variable, SI_6, was a logical addition because in the absence of other hard structures (*e.g*. rocks and cobble or woody debris) in these habitats, vegetation serves as predator avoidance and resting habitats (Maceina, 1996; Hoyer *et al*., 2008).

Water quality was described by April–August average water temperature (SI_2), June–August maximum yearly salinity (SI_3), and the indexed values of primary production (SI_7). SI_2 was assumed to be optimal when April–August water temperatures averaged between 18 and 30 C. SI_3 was assumed to be optimal when salinity averaged less than 8 ppt annually. Primary production, SI_7, was assumed to be optimal as conditions as chlorophyll *a* exceeded 50 μg L ^{1}. Stuber, Gebhart, and Maughan (1982) used pH and dissolved oxygen as input, but these values were not available to us. Primary productivity was included to replace dissolved oxygen because of the close association between the two variables in Louisiana largemouth bass studies (Constant, 1990; Fries, 2010). We modified Stuber, Gebhart, and Maughan's relationship between temperature and salinity based on recent studies (Meador and Kelso, 1989, 1990; Peer, DeVries, and Wright, 2006; Neal and Noble, 2006; Rehage and Loftus, 2007; Buisson, Blanc, and Grenouillet, 2008; Hayer and Irwin, 2008; Rogers and Allen, 2009; Rypel, 2009; Fries, 2010). Lastly, opportunity for reproduction was based on April–August average water temperature, SI_2, which was assumed to be optimal when April–August water temperatures averaged between 18 and 30 C. Stuber, Gebhart, and Maughan (1982) also included temperature, but added salinity, pool and backwater area, substrate type, water-level fluctuation, and current velocity. Only salinity was available to us, but we did not include it in the reproduction component because we used it as input in the water-quality component. The largemouth bass model was validated by comparing model outputs with unpublished field data from ongoing and completed sampling by the School of Renewable Natural Resources (Kelso, Harlan, and Kaller, 2008; Kelso *et al*., 2010) and the Louisiana Department of Wildlife and Fisheries.

Brown Shrimp Habitat Suitability

Baltz (2012a) described the brown shrimp model in detail; here we briefly summarize the model. An HSI for the brown shrimp was developed in 1983 for the U.S. Fish and Wildlife Service (Turner and Brody, 1983). The index was then modified for the 2004 planning effort (Foret *et al*., 2004). For the 2012 Coastal Master Plan modeling effort, the model was further modified based on analysis of Louisiana Department of Wildlife and Fisheries data for juveniles. Our brown shrimp model used three input variables: percent of area covered by emergent wetland vegetation (SI_1), mean water salinity during spring (SI_2), and mean water temperature during spring (SI_3). The model was:

$$\mathrm{HSI} = (\mathrm{SI}_1^2 \times \mathrm{SI}_2 \times \mathrm{SI}_3)^{1/4}.$$

In consultation with Louisiana Department of Wildlife and Fisheries biologists, the initial curves from the 2004 model were compared to newly developed curves from trawl and seine data sets to develop generalized salinity and temperature curves. New data led us to model optimum salinity in the new model as slightly lower and narrower than in the 2004 model, such that SI_2 was optimal when mean salinity from February through May averaged between 10 and 20 ppt, and that SI_3 was optimal when water temperature from February through May averaged between 20 and 30 C. We compared known spatial patterns in brown shrimp abundance with predicted spatial patterns in habitat suitability for brown shrimp and concluded that the model output was valid.

Spotted Seatrout Habitat Suitability

Baltz (2012b) described the spotted seatrout model in detail; here we briefly summarize the model. We previously modeled spotted seatrout for the 2004 planning effort (Foret *et al*., 2004). We analyzed newer data provided by the Louisiana Department of Wildlife and Fisheries and concluded that no revision to this model was justified. Our spotted seatrout model used four input variables: percent of marsh vegetation in a 500 m^2 cell (SI_1), highest monthly mean summer salinity (SI_2), lowest monthly mean winter water temperature (SI_3), and highest monthly mean summer water temperature (SI_4):

$$\mathrm{HSI} = (\mathrm{SI}_1 \times \mathrm{SI}_2 \times \mathrm{SI}_3 \times \mathrm{SI}_4)^{1/4}.$$

Percent of area covered by emergent wetland vegetation (SI_1) was optimum when it was between 25% and 80%. Highest monthly mean summer salinity (SI_2) was optimal when it was between 10 ppt and 25 ppt. Lowest monthly mean water temperature (SI_3) was optimal when it was between 20 ppt and 30 ppt. Highest monthly mean summer water temperature (SI_4) was optimal when it was between 20 C and 30 C. We compared known spatial patterns in spotted seatrout abundance with predicted spatial patterns in habitat suitability for spotted seatrout and concluded that the model output was valid.

White Shrimp Habitat Suitability

Baltz (2012c) described the white shrimp model in detail; here we briefly summarize the model. We previously modeled white shrimp for the 2004 planning effort (Foret *et al*., 2004). We analyzed newer data provided by the Louisiana Department of Wildlife and Fisheries and concluded that no revision to this model was justified. Our white shrimp model used three input variables: percent of marsh vegetation in a 500 m^2 cell (SI_1), mean summer salinity (SI_2), and mean summer water temperature (SI_3):

$$\mathrm{HSI} = (\mathrm{SI}_1^2 \times \mathrm{SI}_2 \times \mathrm{SI}_3)^{1/4}.$$

Percent of area covered by emergent wetland vegetation (SI_1) was optimum when it was between 25% and 80%. Mean summer salinity for summer (SI2) was optimal when it was between 5 ppt and 15 ppt. Monthly mean summer water temperature (SI3) was optimal when it was between 20 C and 30 C. We compared known spatial patterns in white shrimp abundance with predicted spatial patterns in habitat suitability for white shrimp and concluded that the model output was valid.

Gadwall Habitat Suitability

Leberg (2012a) described the gadwall model in detail; here we briefly summarize the model. There is a gadwall habitat suitability model (HSI) on its nesting ground (Sousa, 1985) but not its wintering grounds such as Louisiana. Our model is based on three relationships. First, on winter grounds, gadwalls use intermediate marsh more than fresh or brackish marsh (Bolduc, 2002). Saline marsh is used less frequently than other marsh types (Gray, 2010). Gadwalls also use flooded forested areas (Fredrickson and Heitmeyer, 1987), but the use is more limited than that of marshes. Second, compared to the other ducks being modeled, the gadwall tends to forage on submerged aquatic vegetation (White, 1975; Leschack, Mckinght, and Hepp, 1997). Finally, gadwalls tend to require fairly deep water, compared to other dabbling ducks, for foraging (Bolduc, 2002). Our gadwall model used three input variables: habitat type (SI1), percent of the area with water that supports SAV (SI_2), and water depths during winter (SI_3).

$$\mathrm{HSI} = (\mathrm{SI}_1 \times \mathrm{SI}_2 \times \mathrm{SI}_3)^{1/3}.$$

Habitat type (SI_1) was assumed to be ideal when vegetation led to classification as an intermediate marsh. SI_2 was assumed to be ideal when 70% or more of the area was occupied by water that supported SAV; SI_2 declined to 0 when less than 30% of the area was occupied by water that supported SAV. SI_3 was assumed to be ideal when monthly water depth from September through March averaged between 18 cm and 32 cm depth. We validated the model by comparing known spatial patterns in gadwall abundance to predicted spatial patterns in gadwall habitat suitability.

Crawfish Habitat Suitability

Here we summarize our crawfish model, which Romaire (2012) described in detail. The only other published crawfish HSI model that we are aware of was developed to rank stream habitat refugia sites for relocating populations of white-clawed crawfish (crayfish) *Austropotamobius pallipes* from endangered stream habitats in England (Watson and Rogers, 2003). The red swamp crawfish (*Procambarus clarkii*) and southern white river crawfish (*Procambarus zonangulus*) dominate species in commercial and recreation harvests in Louisiana (Huner, 2002; Walls, 2009). Our crawfish model assumed that the habitat quality for red swamp crawfish and southern white river crawfish is affected by five variables: salinity, water temperature, water depth, vegetative habitat type, and water-level fluctuations during summer–early fall and late fall–winter–spring. The variables were combined to represent water quality, habitat, and reproduction:

$$\mathrm{HSI} = (\mathrm{CI}_{\text{water quality}} \times \mathrm{CI}_{\text{habitat}} \times \mathrm{CI}_{\text{reproduction}})^{1/3}.$$

Water quality was assumed to vary with water salinity and temperature:

$$\mathrm{CI}_{\text{water quality}} = (\mathrm{SI}_1 \times \mathrm{SI}_2)^{1/2}.$$

SI_1 was assumed to be ideal when salinity was less than or equal to 1 ppt; SI_1 declined to zero when water salinity exceeded 5 ppt. SI_2 was assumed to be ideal when water temperature was between 20 C and 26 C. Habitat was assumed to vary with water depth (SI_3), habitat class (SI_4), and water depth variability (SI_5):

$$\mathrm{CI}_{\text{habitat}} = (\mathrm{SI}_3 \times \mathrm{SI}_4 \times \mathrm{SI}_5)^{1/3}.$$

SI_3 was assumed to be ideal when water depth was between 30 cm and 61 cm deep. SI_4 was assumed to be ideal when vegetation led to a classification as swamp. SI_5 was assumed to be ideal when the monthly variability exceeded 3 m from June through November and when monthly variability was less than 1 m from December through May. Reproduction was assumed to depend upon water depth (SI_3) and water depth variability (SI_5):

$$\mathrm{CI}_{\text{reproduction}} = (\mathrm{SI}_3 \times \mathrm{SI}_5)^{1/2}.$$

Water depth and water-depth variability thus were more important in the model than the other variables. We validated the model by comparing known spatial patterns in crawfish abundance to predicted spatial patterns in crawfish habitat suitability. For example, habitat quality for crawfishes was projected to be low (ranging from 0 to 0.2) in areas modeled as saline marsh..

Mottled Duck Habitat Suitability

Here we summarize our mottled duck model, which Leberg (2012b) described in detail. The mottled duck differs from the other modeled waterfowl (gadwall and green-winged teal) in that it is a year-round resident of coastal Louisiana. There is an existing HSI that has been developed for this species (Rorabaugh and Zwank, 1983). In that effort, habitat was evaluated for either its reproductive value or its foraging value. We intended to continue that approach, but the available input data limited us to modeling only foraging habitat. We modified the HSI model of Rorabaugh and Zwank (1983) to make it useful with the available model inputs and to reflect knowledge gained from more recent studies (*e.g*. Zwank, McKenzie, and Moser, 1989; Moorman and Gray, 1994; Bolduc, 2002; Bolduc and Afton, 2004). Our model of habitat quality for mottled duck foraging used two variables as in input: habitat type (SI_1) and the proportion of days of the year with water depths suitable for foraging (SI_2):

$$HSI = (SI_1 \times SI_2)^{1/2}.$$

SI_1 was assumed to be ideal when vegetation led to classification as fresh marsh. SI_2 was assumed to be ideal when water depth in a cell was between 6 cm and 34 cm deep 100% of the days in a year. We validated the model by comparing known spatial patterns in mottled duck abundance to predicted spatial patterns in mottled duck habitat suitability.

Neotropical Migrant Songbird Habitat Suitability

Here we summarize our model of habitat quality for Neotropical migrants, which Leberg (2012c) described in detail. This model was an attempt to model species richness, weighted by rarity for Neotropical migrant passerines in the wetlands of southern Louisiana. We defined Neotropical migrants as a passerines (order Passeriformes) that breed in the United States or Canada and winter primarily in the Neotropics (south of the tropic of Cancer). Published surveys of relative abundance of Neotropical migrant species, conducted across the habitats of coastal Louisiana, could not be located. Based on migratory patterns, we determined that 67 species clearly fit a definition of passerines that were Neotropical migrants (Leberg, 2012c). We developed an index of habitat importance for this diverse group based on reports of birds observed during birding trips to specific sites in southern Louisiana as recorded in a database maintained by the Cornell Laboratory of Ornithology and the National Audubon Society (Sullivan *et al.,* 2009). There are a number of shortcomings to this approach, such as different levels of observer ability, reporting accuracy, and site homogeneity, but the benefits of having a large number of observations are assumed to minimize the bias and imprecision in individual surveys. We used the hierarchical richness (HRI) index of French (1994) because it incorporated abundance and diversity and could be used with the available data (reports of numbers of individuals for a large number of species). Rarer species have higher ranks, so a site that has a large number of species that are typically rare in Louisiana will have a high habitat value score, especially if those species are abundant. Average HRI values were determined for each habitat type, based on all the sites where that habitat was dominant. Once we determined the vegetation type that had the highest average HRI value, we divided the HRI values of the other habitat types by that average value, setting the highest relative value as 1, and setting the other values relative to the highest value. The relative values of the HRI are referred to as the habitat importance index (HI). The original model included relative importance values for ridges, maritime forests, and bottomland hardwood forests. These forests had higher HRI, and thus HI, values than the wetland types included in the final model. However, ridges, maritime forests, and bottomland hardwood forests were removed from the final model because they were not modeled in the vegetation model due to lack of data on these habitat types. Thus, the relative values of HI were determined based on the most modeled vegetation type (swamp) that had the highest HRI. Other habitats evaluated were fresh, intermediate, brackish, and saline marsh. Habitat type was the only input variable for this model. We validated the model by comparing known spatial patterns in Neotropical bird richness to predicted spatial patterns in habitat quality for Neotropical birds.

Green-Winged Teal Habitat Quality

Here we summarize our model of habitat quality for green-winged teal, which Leberg (2012d) described in detail. The green-winged teal (*Anas crecca*) forages in Louisiana's coastal marshes during the fall, winter, and spring. To our knowledge, a habitat suitability index model has not been developed for this species. This model is based on two variables: habitat type (Fredrickson and Heitmeyer, 1987; Johnson, 1995; Bolduc, 2002) and water depth during migration and winter (Bolduc and Afton, 2004). The suitability of a cell to provide habitat for green-winged teal was computed as:

$$HSI = (SI_1 \times SI_2)^{1/2}.$$

SI_1 was assumed to be ideal when vegetation led to classification as fresh marsh. SI_2 was assumed to be ideal when 100% of the days from September through March had water 8 to 18 cm deep. We validated the model by comparing known spatial patterns in green-winged teal abundance to predicted spatial patterns in habitat quality for green-winged teal.

Roseate Spoonbill Model

Here we summarize our model of habitat quality for roseate spoonbills, which Leberg (2012e) described in detail. We modified an earlier model of habitat quality for roseate spoonbill (Lewis *et al*., 1983) because some of the input variables they used were not available to us and to account for new information, including a model of habitat quality for other wading birds (Draugelis-Dale, 2007). The model simulates nesting habitat quality and foraging habitat quality separately and assigns the higher of the two values to an area. Lack of suitable foraging habitat may be one of the primary reasons for colony abandonment (Leberg *et al*., 2007). Foraging habitat depended upon edge habitat and water depth:

$$HSI_{foraging} = (SI_6 \times SI_7)^{1/2}.$$

SI_6 was assumed to be ideal when water was 1 to 12 cm deep in 100% of a cell. We modified the foraging habitat component of Draugelis-Dale (2007), which focused on different wading birds, to focus on water depths when roseate spoonbills breed (see Dumas, 2000). SI_7 was assumed to be ideal when the proportion of the cell that is edge (SI_7), *i.e*. the area of water projecting 10 m from the land-water interface, was 100%. We focused on a 10 km radius around nesting colonies, as did Draugelis-Dale (2007), but we used a threshold of 50% suitable water depths rather than the proportion of suitable water depths because we had more information on average water depths and because personal observations suggest that reproductive success decreases only when foraging habitat becomes relatively uncommon.

Nesting habitat was calculated differently for islands and for other wetlands. Both models depended upon vegetation type (SI_2), woody vegetation (SI_3), and the availability of foraging habitat with 1 km of a cell (SI_4); the island model also depended upon island size (SI_1). The models for islands and wetlands were:

$$HSI_{island} = (SI_1 \times SI_2 \times SI_3 \times SI_4)^{1/4}$$

and

$$HSI_{wetlands} = (SI_2 \times SI_3 \times SI_4)^{1/3}.$$

Island size (SI_1) was assumed to be ideal when islands were less than 100 ha. We modified the island size relationship from Lewis (1983) to account for the spatial resolution available to us. Ideal vegetation (SI_2) type was assumed to be swamp; ideal wood vegetation (SI_3) was assumed to be 100%. The relative value of different plant communities was based on data from Michot *et al*. (2003) and Green *et al*. (2006). SI_4 depended upon available foraging habitat, which was defined as the proportion of daily water depths for the period of February through July where the water depth was between 1 and 12 cm. SI_4 was assumed to be ideal when water was 1 to 12 cm deep in 50% of the adjacent cells. SI_4 and SI_6 were similar, but SI_6 applied to the cell being modeled, whereas SI_4 applied to the adjacent cells. We validated the model by comparing known spatial patterns in roseate spoonbill abundance to predicted spatial patterns in habitat quality for roseate spoonbills. For instance, we incorporated a geographical information system (GIS) mask to exclude barrier islands when they were too far offshore for regular use by roseate spoonbills.

American Alligator Habitat Suitability

Here we summarize our model of habitat quality for American alligators, which Nyman (2012a) described in detail. The 2004 Louisiana Coastal Area Study (LCA Study; USCOE, 2004) and 2007 Louisiana Coastal Master Plan (CPRA, 2007) used an American alligator model (Foret *et al*., 2004) based on the HSI model prepared by Newsom *et al*. (1987). We updated the 2004 model with new information regarding habitat distribution, flooding, salinity, and wetland edge effects. The 2004 model also was the subject of an in-depth review and comparison with the American alligator model used by the South Florida Management District in planning Everglades' restoration (Draugelis-Dale, 2007). That review concluded that the Louisiana model would benefit from using seasonal rather than yearly water levels, as the Florida model did, and that the Florida model would benefit from incorporating a variable accounting for percent open water, as the Louisiana model did (Drauglis-Dale, 2007). As suggested by Draugelis-Dale (2007), our model used monthly water-level estimates, which were unavailable for the 2004 and 2007 models. Draugelis-Dale (2007) also suggested that the Louisiana model should use a different habitat classification, such as the Cowardin *et al*. (1987) system, but that suggestion appeared illogical to us because the Cowardin *et al*. (1987) system combines all *Spartina*-dominated marshes into a single class, but nest density of American alligators declines greatly from low-salinity *Spartina patens* marshes through high-salinity *Spartina patens* marshes to no nests in *Spartina alterniflora* marshes (Nymana, 2012). Foret *et al*. (2004) also suggested that future American alligator models incorporate edge effects, which concentrate many wildlife prey species near edges of open water and emergent vegetation. Edge effects were unavailable as input for the 2007 model (Foret *et al*., 2004) but were incorporated into our model. Our model of alligator habitat suitability used five input variables: percent emergent vegetation (SI_1), water depth (SI_2), habitat type (SI_3), edge habitat (SI_4), *i.e*. open water within 10 m of emergent vegetation, and water salinity (SI_5). It was computed as:

$$HSI = (SI_1 \times SI_2 \times SI_3 \times SI_4 \times SI_5)^{1/5}.$$

Emergent vegetation (SI_1) was assumed to be ideal when it covered between 60% and 80% of a cell. Water depths (SI_2) were assumed to be optimal when they averaged 0.15 cm below marsh soil elevation. Habitat type (SI_3) was ideal when the vegetation led to a classification as fresh marsh. The edge input (SI_4) was simulated by the Wetland Morphology modeling group working on the 2012 Coastal Master Plan (Couvillion *et al*., 2013); the distribution of edge was used to scale the relationship between edge and SI_4 such that values less than the 50th percentile produced an SI_4 of approximately 0.5 and such that values greater than the 90th percentile produced an SI_4 of 1.0. Water salinity (SI_5) was assumed to be ideal when it averaged 0 ppt. We validated our model by comparing known spatial patterns in American alligators to predicted spatial patterns in habitat quality for American alligators. For instance, initial output predicted no American alligators at the mouth of the Mississippi River, which led the hydrologic modelers (Meselhe *et al*., 2013) to reduce the area of the boxes in their grid to more accurately estimate the zonation of lower-salinity and higher-salinity wetlands there.

Muskrat Habitat Suitability

Here we summarize our model of habitat quality for muskrats, which Nyman (2012b) described in detail. Allen and Hoffman (1984) prepared an HSI for muskrats, which served as a basis for a muskrat HSI model in coastal marshes that was developed for the 2004 Louisiana Coastal Area (LCA) study and also used in the 2007 Louisiana Master Plan (Foret *et al*., 2004). The model developed for the 2012 Coastal Master Plan was based on the 2004 model but was modified to account for better information regarding average hydrologic conditions and to take advantage of hydrologic models capable of providing monthly, rather than annual, estimates of average water level. Our model of habitat suitability for the muskrat uses three input variables: percent of emergent vegetation (SI_1), average water depth (SI_2), and habitat type (SI_3):

$$HSI = (SI_1 \times SI_2 \times SI_3)^{1/3}.$$

Emergent vegetation (SI_1) was assumed to be optimal when it was between 50% and 80%. Water depths (SI_2) were assumed to be optimal when they averaged 0.15 cm below marsh soil elevation. Habitat type (SI_3) was ideal when the vegetation led to a classification as brackish marsh. We validated the model by comparing known spatial patterns in muskrat abundance to predicted spatial patterns in muskrat habitat suitability.