

Impacts of a Freshwater Diversion on Wildlife and Fishes in the Maurepas Swamp

David M. Fox, Philip C. Stouffer, D. Allen Rutherford, W. E. Kelso, Megan La Peyre, and Raynie Bamberger

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Prepared for:
U.S. Environmental Protection Agency
Region 6
1445 Ross Avenue
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Prepared by:
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Introduction

The U.S. Environmental Protection Agency, Region 6, and the Louisiana Department of Natural Resources are planning a reintroduction of Mississippi River water into the Maurepas Swamp, Project Number PO-29 (Figure 1). The expected effects of the diversion of approximately 1500cfs of freshwater, and its associated load of nutrients, from the Mississippi River into Maurepas Swamp are decreased stagnation and saltwater intrusion, resulting in increased primary productivity of the freshwater wetland-plant community within the area affected by the diversion (Figure 1) (Shaffer et al. 2003). Increased primary productivity is expected to result in long-term accretion, decreased subsidence, and improved forest health.

As part of the information necessary for the EIS for the diversion of Mississippi River water into the Maurepas Swamp, we provide information on wildlife and fishes of concern within the Maurepas Swamp ecosystem, including those species found within the area of Hope Canal, the preferred diversion route, and several alternative routes. We also provide recommendations for the timing of construction and operation of the diversion canal with respect to critical life-history stages of these species.

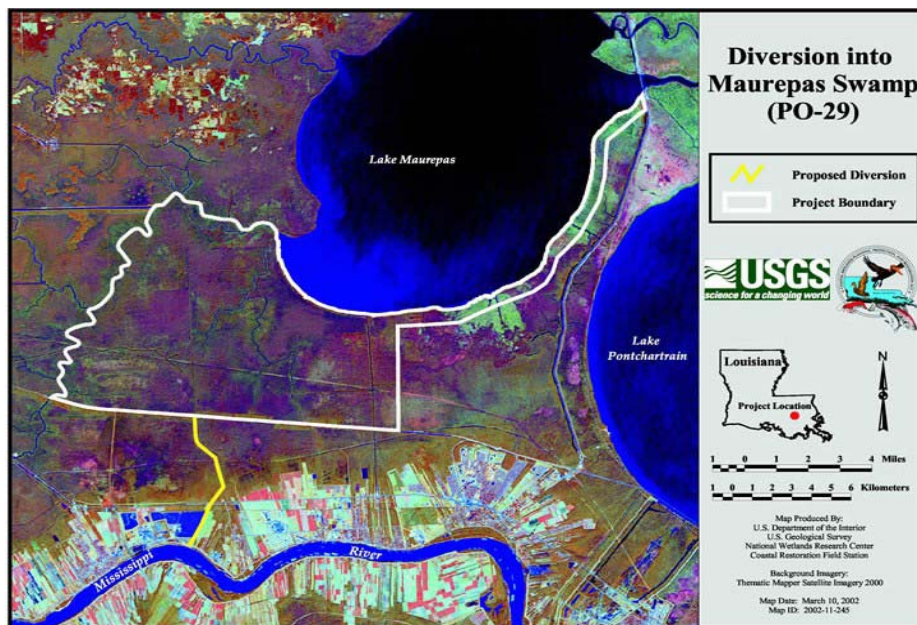


Figure 1. Maurepas Swamp Study Area. Source:
<http://www.epa.gov/Arkansas/6wq/ecopro/em/cwppra/maurepas/index.htm>

Freshwater fishes

William E. Kelso, D. A. Rutherford, Raynie Bamberger

Fish community composition is often a sensitive indicator of habitat quality, and determination of fish abundance and distribution patterns can provide insight into the physicochemical suitability of aquatic habitats for fishes and other biota (Wang et al. 1997; Lammert and Allan 1999; Wang et al. 2001). For this part of the overall assessment, we collected fishes at 20 sites within and adjacent to the projected flow path of the Maurepas diversion project upstream from Lake Maurepas during June 2006 to determine spatial differences in habitat and fish assemblage composition, and to assess possible impacts of the diversion on fishes in the Maurepas swamp.

Methods

Prior to going into the field, we identified 20 sampling sites based on the distribution of canals and bayous in the study area. We were unable to sample sites 07 and 08 because of shallow depths, so two additional sites were added at nearby locations (Figure 2). All fishes were collected with a boat-mounted Smith-Root Model 7.5 GPP electrofishing unit that produced a pulsed-DC current of at least five amps. At each site we collected fishes during duplicate 15-minute electrofishing surveys of all available microhabitats along each shoreline. All sampled fishes were placed in a live-well, and at the end of each 15-minute run, each fish was identified to species, enumerated, and released. At each site, we recorded Secchi depth and measured surface flow velocity (cm/sec) and direction with a Sontek Flowtracker[®] Doppler flowmeter. We also recorded surface, mid-column, and bottom (sites 3, 15, and 18 were less than 0.8 m in depth, so only surface and bottom measurements were taken) temperature (C), pH, specific conductance (mS/cm), salinity (PSU), dissolved oxygen (DO) concentration (mg/l), DO saturation (%), depth (m), and turbidity (NTUs) with a Hydrolab Quanta[®] in situ monitor.

Fish and water quality data were tabulated for each site. For the fish data, we used community richness (number of species), diversity calculated as $e^{(H)} = \exp(-\sum p_i \ln p_i)$, where p_i =proportion of species i at each site; Jost 2006), and evenness calculated as $H/\ln S$, where S =the number of species (Jorgensen et al. 2005), to describe differences in the fish community structure

among the 20 sites. Physicochemical differences among sites were assessed with a principal components analysis (PCA) of surface and bottom water quality characteristics (we eliminated mid-depth readings, as they could not be recorded at sites 3, 15 and 18) after eliminating salinity (collinear with specific conductance) and DO saturation (collinear with DO) from the physicochemical dataset. To visualize trends throughout the study site, we used a biplot of scores for each sampling location on a PC1 and PC2. We also performed a PCA on the fish dataset, and plotted all factor loadings >0.4 for common species (those occurring at more than two sites) on several bi-plots for those PCs exhibiting eigenvalues over 1.0 (i.e., the component explained more variance than the original variables in the analysis). We also included the correlations (vectors) of each of the physicochemical variables with each PC on these plots to help explain the observed fish distributions and species associations. If two physicochemical variables exhibited nearly identical correlations (e.g., surface and bottom specific conductance), then correlations based on surface water quality measurements were used to plot the vector.

Results and Discussion

Physicochemistry. Even within this limited area, there was considerable between-site variation in some of the measured physicochemical characteristics (Table 1). We did not expect much variation in temperature within the study area, and although there was little evidence of vertical stratification in temperature, even at sites with no flow, surface temperatures surprisingly varied over 5C among the 20 sites. Differences were not related to flow, depth, salinity, or Secchi depth, but may have reflected groundwater and source-water inputs at some of the sites. Temperatures below 27C were recorded at sites 3, 9, 19, and 22, which were distributed throughout the lower portion of the study area, whereas the highest surface temperatures were recorded at the periphery of the study site in larger channels along Alligator Bayou, Blind River, and the Reserve Relief Canal (Sites 6, 12, 13, 1, 2, 14, and 17; Figure 2).

All salinity values were relatively low throughout the study site, but higher values (> 3 PSU) were generally found at sites near Lake Maurepas and the Reserve Relief Canal in the eastern portion of the study area, with lower values (< 2 PSU) found at sites associated with Alligator Bayou and Blind River (Table 1; Figure 2). Although dissolved oxygen conditions were typically adequate at surface and mid-depths, many sites (e.g., sites 1, 3, 6, and others)

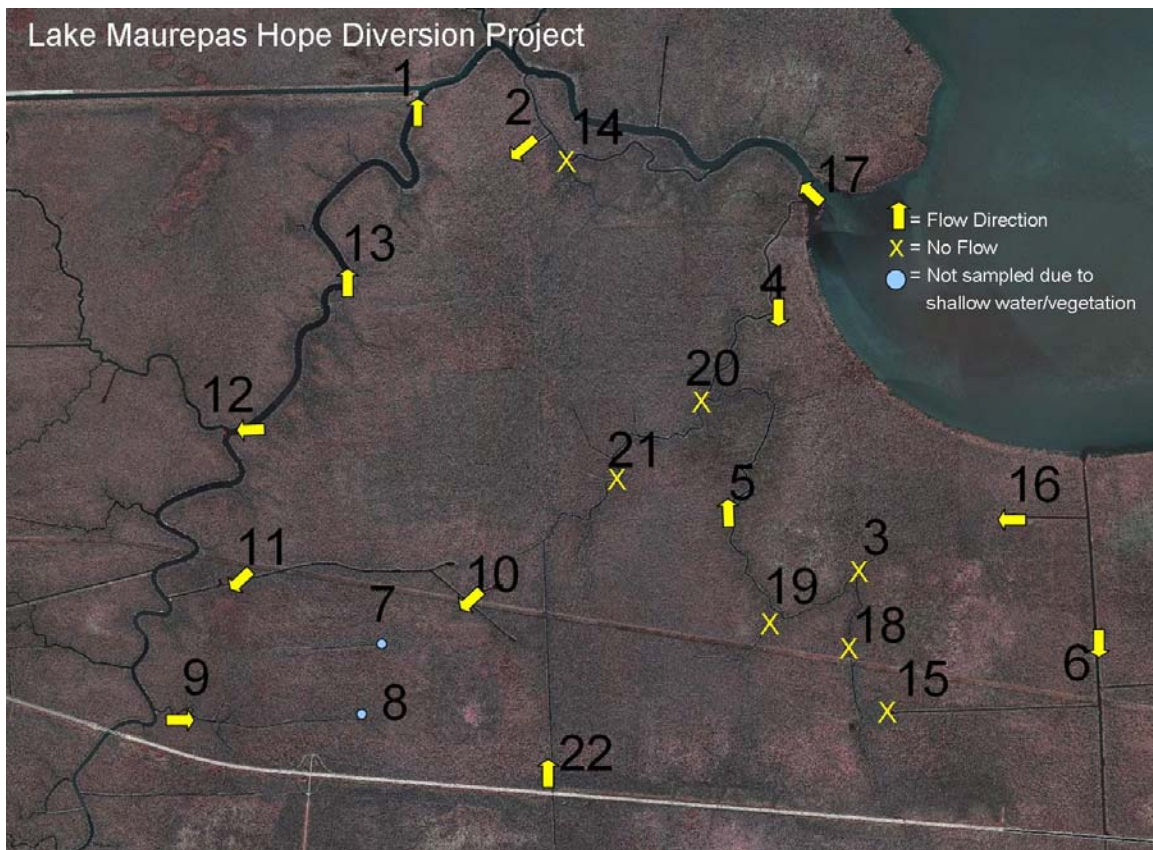


Figure 2. Location of fish sampling sites within the Maurepas swamp study area. Arrows indicate direction of flow for those sites at which current velocities were >0.0 .

exhibited hypoxic conditions ($DO < 2.0$; Fontenot et al. 2001) near the bottom. As might be expected, several (but not all) of these sites typically exhibited little if any flow (e.g., 1, 3, 6, 12). Nonetheless, several stagnant sites (e.g., sites 5, 15, 18, 19, 21) had bottom DO levels over 2.0, possibly related to shallow depths (e.g., site 15) or algal production.

Turbidities generally increased from surface to bottom, and several bottom measurements were extremely high relative to surface levels (Table 1). High bottom turbidities did not seem related to flow (e.g., compare Site 17, 7.3 cm/sec velocity, 252 NTU to Site 22, 9.8 cm/sec velocity, 7.4 NTU, or to Site 19, 0 cm/sec velocity, 219 NTU), and interestingly, neither surface or bottom turbidities were strongly related to Secchi depth ($r < -0.59$). High bottom turbidities could have been caused by contact of the water quality probe with the substrate, but this was not evident during sampling. We did not measure chlorophyll a concentrations, so we do not know the relative contributions of biogenic and abiogenic components to total turbidity, but this parameter may be of interest in future monitoring efforts. Although generally inversely related to turbidity, Secchi depth was also quite variable among sites (28 to 95 cm; Table 1), and also varied substantially as a percentage of total water column depth (7.2 to 100%).

Table 1. Water quality characteristics at the 20 fish collection sites in June 2006. Position refers to bottom (B), mid-depth (M), and surface (S) values.

Site	Position	Temp. (C)	pH	SpC (mS/cm)	Salinity (PSU)	DO% Sat	DO (mg/L)	Depth (m)	Turbidity (NTU)	Secchi (cm)	Flow (cm/sec)	Flow Direction
1	B	29.33	6.27	3.98	2.11	14.6	1.11	5.1	29.3			
1	M	29.65	6.42	3.04	1.59	51.9	3.92	2.6	7.2			
1	S	30.49	6.83	2.24	1.15	100.6	7.52	0.1	9.9	82	1	north
2	B	28.53	6.31	3.06	1.60	45.3	3.49	1.9	15.9			
2	M	29.13	6.36	3.14	1.64	77.7	5.92	1.0	4.9			
2	S	30.57	6.44	3.13	1.64	95.4	7.07	0.0	4.9	57	2.1	southwest
3	B	26.45	6.30	4.11	2.18	15.1	1.20	0.4	129.0			
3	S	26.66	6.27	4.12	2.18	16.8	1.36	0.1	65.6	28	0	no flow
4	B	28.57	6.20	5.90	3.21	46.3	3.53	2.8	40.5			
4	M	28.87	6.15	5.89	3.20	62.0	4.70	1.4	11.6			
4	S	30.33	6.18	5.88	3.20	80.9	6.00	0.1	5.4	66	1.4	south
5	B	27.21	6.25	5.13	2.76	29.6	2.30	1.7				
5	M	27.23	6.22	5.14	2.76	29.1	2.28	0.8	5.8			
5	S	27.36	6.22	5.14	2.76	30.1	2.35	0.1	5.5	91	0.8	north
6	B	28.81	6.58	5.69	3.08	24.1	1.83	3.2	26.0			
6	M	29.14	6.62	5.60	3.03	45.8	3.46	1.5	10.1			
6	S	30.14	6.68	5.61	3.04	73.7	5.47	0.1	10.1	44	2	south
9	B	26.12	6.51	1.54	0.77	25.3	2.04	1.7	4.9			
9	M	26.19	6.48	1.55	0.78	23.6	1.91	0.7	4.4			
9	S	26.29	6.45	1.54	0.77	24.4	1.97	0.2	3.0	95	8.2	east
10	B	26.62	6.48	1.03	0.51	4.3	0.35	1.8	6.8			
10	M	27.57	6.61	1.05	0.52	30.8	2.43	0.9	5.6			
10	S	27.50	6.66	1.05	0.52	25.5	2.01	0.1	3.0	67	3.5	southwest
11	B	27.56	6.45	1.83	0.93	25.6	2.02	2.0	10.4			
11	M	27.89	6.47	1.87	0.95	30.2	2.35	1.0	5.6			
11	S	28.01	6.49	1.85	0.94	31.4	2.42	0.1	5.9	77	3.4	southwest
12	B	28.80	6.50	2.79	1.45	6.0	0.46	4.5	63.0			
12	M	28.44	6.48	1.79	0.91	18.6	1.44	2.2	12.9			
12	S	33.05	7.14	1.18	0.59	98.0	7.04	0.0	7.6	71	1.1	west
13	B	29.31	6.21	4.07	2.16	3.2	0.25	8.3	22.9			
13	M	29.17	6.29	3.15	1.65	20.4	1.56	4.1	15.5			
13	S	29.37	6.42	2.92	1.52	54	4.12	0.1	11.1	60	10.5	north
14	B	28.97	6.54	2.96	1.54	85.8	6.65	1.1	17.8			
14	M	29.17	6.51	2.93	1.52	78	6.02	0.5	8.2			
14	S	30.92	6.66	2.85	1.48	107.7	7.97	0	9	44	0	no flow
15	B	28.29	6.43	6.04	3.29	25.8	1.97	0.8	22.1			
15	S	28.86	6.43	6.03	3.28	39.2	2.97	0.1	0.2	80	0	no flow
16	B	29.80	6.53	6.07	3.31	39.3	2.93	1.5	11.3			
16	M	30.30	6.56	6.11	3.33	55.2	4.08	0.7	6.8			
16	S	30.73	6.58	6.13	3.34	59.1	4.34	0.1	5.6	77	1.3	west
17	B	30.34	6.5	4.27	2.27	20.8	1.56	2.5	252			
17	M	31.70	6.59	4.24	2.26	66.9	4.86	1.3	15.5			
17	S	31.80	6.73	3.38	1.78	94.3	6.87	0	9.3	44	7.3	northwest
18	B	27.30	6.39	4.73	2.53	28.7	2.24	0.7	90.9			
18	S	27.62	6.36	4.73	2.53	31.1	2.42	0.1	7.2	56	0	no flow
19	B	26.48	6.27	4.88	2.61	26.1	2.07	1	219			
19	M	26.49	6.24	4.89	2.62	26.8	2.13	0.5	2.7			
19	S	26.55	6.24	4.89	2.62	27.8	2.2	0.1	2.7	59	0	no flow
20	B	28.23	6.33	5.74	3.11	45.1	3.46	3.4	13.8			
20	M	28.33	6.24	5.7	3.09	42.6	3.27	1.8	13.5			
20	S	28.54	6.22	5.67	3.07	40.5	3.09	0	8.8	67	0	no flow
21	B	27.63	6.05	5.56	3.01	36.3	2.81	1.6	88.7			
21	M	27.79	6.02	5.57	3.01	36.7	2.84	0.8	5.5			
21	S	28.04	6.01	5.57	3.01	40.2	3.1	0.1	5	86	0	no flow
22	B	26.14	6.35	4.34	2.31	22.6	1.81	1.2	7.4			
22	M	26.18	6.31	4.34	2.31	20.7	1.66	0.6	6.4			
22	S	26.31	6.3	4.32	2.3	23.4	1.87	0.1	6.6	70	9.8	north

Flow rates were generally low, ranging from 0 to 10.5 cm/sec, and were spatially inconsistent among sites. The “high flow” category (7.3-10.5 cm/sec) included sites 9, 13, 17, and 22, four sites that were in disjunct bayous at the periphery of the study area (Figure 2). With the exception of site 14 (Alligator Bayou), all of the stagnant sites were in the central portion of the study site in Bayou Bee Croche / Bayou Tent / Mississippi Bayou area. Flow direction indicated a general trend of south to north flow at the periphery of the study area, with variable flow directions at the interior sites (Figure 2).

Principal component analysis of the physicochemical data yielded two PCs that together explained 54% of the variance in the dataset. Principal component 1 (32% of the variance explained) contrasted sites that differed in surface DO, temperature, surface pH, and depth, whereas PC2 (22% of the variance explained) contrasted sites that differed in specific conductance, water column pH, and flow (Figure 3). Although there was considerable spread among sites on the PC biplot, site scores generally reflected spatial relationships among sites, indicating a predictable physicochemical pattern within the study area. Sites 9, 10, 11, and 22 (Group I; Figure 4) in the southwestern part of the study area exhibited lower temperatures and relatively moderate to high flows, but, contrary to what would be expected in moving water, hypoxic or marginally hypoxic surface DO concentrations. Sites 3, 5, 15, 18, 19, 20, and 21 (Group II) in the interior of the study area in Bayou Bee Croche, Bayou Tent, and Mississippi Bayou were also characterized by cooler temperatures and low surface DO levels, but generally exhibited little or no flow. Sites 4, 6, and 16 (Group III) occurred at the eastern periphery of the study area in the Reserve Relief Canal, a canal tributary to the Reserve Relief Canal, and Dutch Bayou, all of which were near Lake Maurepas. These sites exhibited high specific conductance values (and salinities), moderate flows, and high surface DO levels. The remaining sites (Group IV) occurred at the periphery of the study area along Blind River and Alligator Bayou. These sites tended to be deeper with variable flows (0-10.5 cm/sec), higher temperatures, low to moderate specific conductance values, and high surface pH. Site groupings in Figure 4 are admittedly subjective, but it is particularly interesting that, in general, sites tended to group by their position in the study area, even if specific physicochemical characteristics differed among sites within a group.

Fish communities. Fish collections yielded a total of 1,425 fishes from the 20 sites (Table 2). Combining data from all sites, collections the Maurepas swamp study area included 26 fish

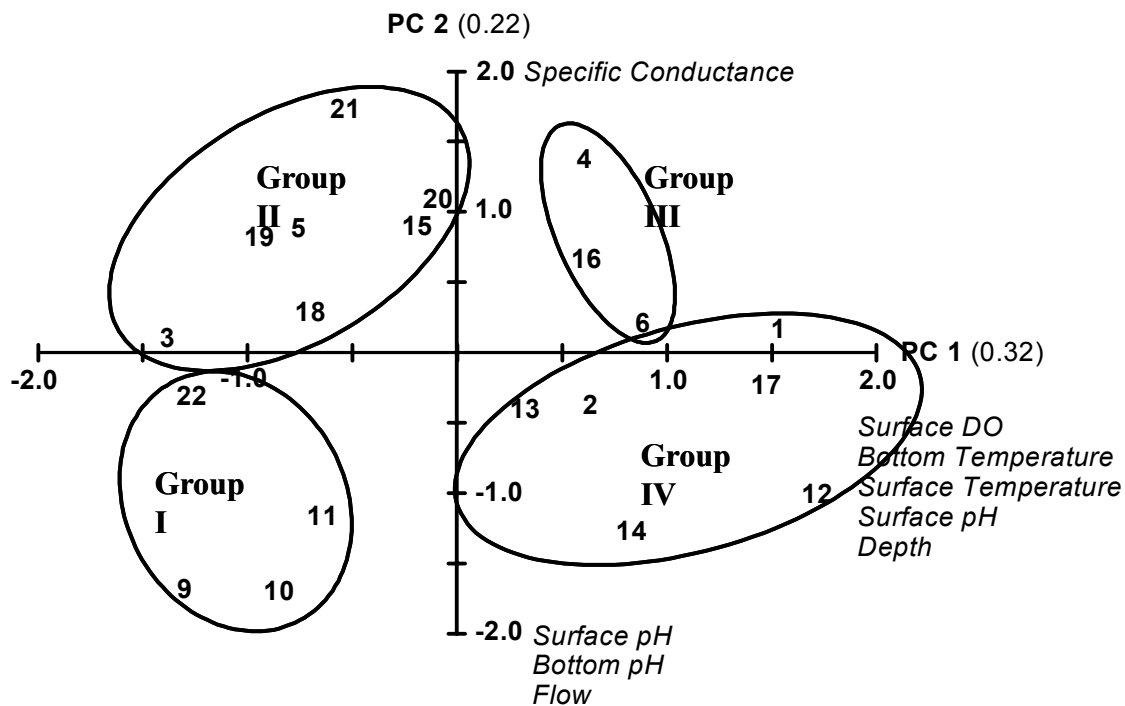


Figure 3. Scores of the 20 Maurepas swamp sites sampled in June 2006 on PC1 and PC2, based on the principal components analysis of physicochemical characteristics recorded at each site. Ellipses are subjective site groupings. Numbers in parentheses are the amount of variance explained by each PC. The physicochemical characteristics shown exhibited loadings $> |0.4|$ with that particular PC.

species, although 16 taxa occurred at four sites or less and were represented by fewer than nine individuals. Collections at each site ranged from 25 to 147 individuals, and most sites were dominated by spotted gar *Lepisosteus oculatus* and striped mullet *Mugil cephalus*, which together comprised 76.5% of all fishes sampled in the study area. In addition to the striped mullet, several other euryhaline species common in the nearshore areas off the Louisiana coastline were also collected, including Alabama shad *Alosa chrysochloris*, Gulf menhaden *Brevoortia patronus*, and ladyfish *Elops saurus*. Samples also included freshwater drum *Aplodinotus grunniens*, blue catfish *Ictalurus furcatus*, and yellow bass *Morone mississippiensis*, three freshwater taxa that are usually more abundant in flowing water systems in the southeastern U.S. (Ross 2001).

Table 2 Fishes collected at each of the 20 sites sampled in June 2006.

Fish species	Site																					
	1	2	3	4	5	6	9	10	11	12	13	14	15	16	17	18	19	20	21	22		
<i>Alosa chrysochloris</i>	5	1									2										8	
<i>Ameiurus melas</i>																			1		1	
<i>Amia calva</i>													1		1				1		4	7
<i>Anguilla rostrata</i>						2								1								3
<i>Aplodinotus grunniens</i>						1		1				1										3
<i>Brevoortia patronus</i>	2					1		1	1	27			2		5							39
<i>Cyprinus carpio</i>		1			1				2													4
<i>Dorosoma cepedianum</i>	7					3		4	1	4	7		7	1	2				1		2	39
<i>Dorosoma petenense</i>								1														1
<i>Elops saurus</i>											1											1
<i>Fundulus chrysotus</i>																				1		1
<i>Ictalurus furcatus</i>	1			1					1	1		1		2	3	1			3			14
<i>Ictalurus punctatus</i>								4					1	2		2						9
<i>Ictiobus cyprinellus</i>		2					1	3	1	3	3		2	3			2	1	1			22
<i>Lepisosteus oculatus</i>	3	9	40	9	22	53	6	11	6	20	12	22	20	16	11	41	37	40	33	11		422
<i>Lepisosteus osseus</i>															1							1
<i>Lepomis gulosus</i>									1							1			1			3
<i>Lepomis humilis</i>																			1			1
<i>Lepomis macrochirus</i>	1	11	2	8	1	1	2	5	1	4	7	9	3	15	1	1	1	6	2			81
<i>Lepomis megalotis</i>									3													3
<i>Lepomis microlophus</i>		1	1		1		2	2			2	1				1		3	1			15
<i>Micropterus punctulatus</i>	1	3		1		2	7	3		1	13	5	1	1	1	1	1			1		42
<i>Micropterus salmoides</i>		3	3	5	1	1		1		1	2	3	1	2	1				5	1		30
<i>Morone mississippiensis</i>		1							1		1	1										4
<i>Mugil cephalus</i>	80	115	1	15	13	48	16	30	47	57	49	43	47	26	21	1			18	33	8	668
<i>Pomoxis nigromaculatus</i>												3										3
Grand Total	100	147	47	39	39	112	34	66	65	118	99	89	85	69	47	49	41	80	74	25		1425

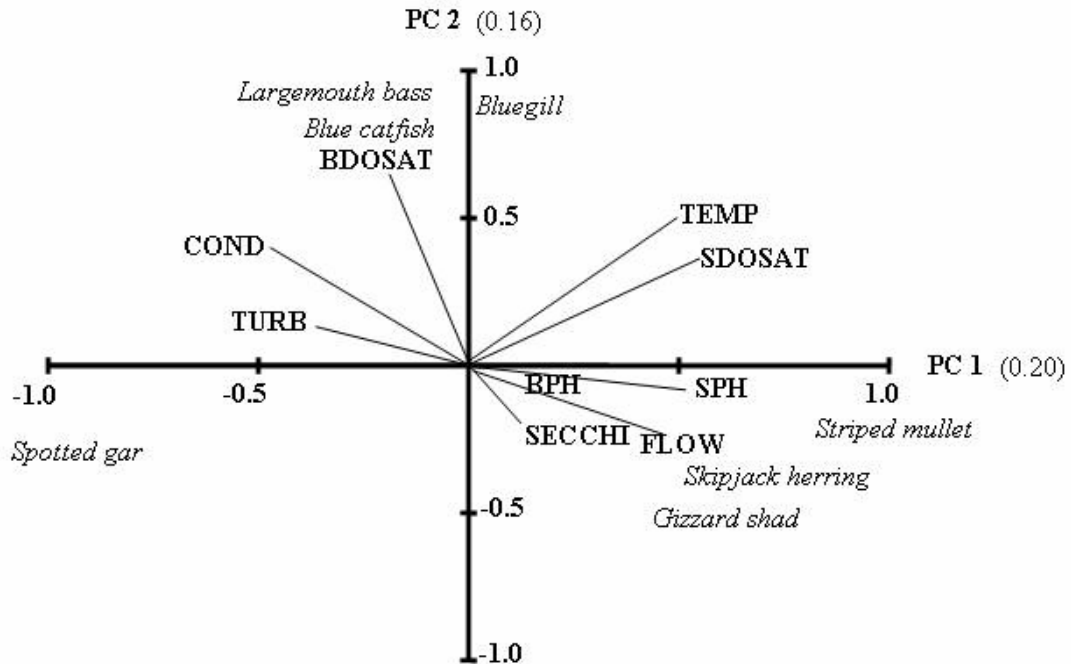


Figure 4. Loadings of Maurepas swamp fishes sampled in June 2006 on PC1 and PC2, based on the principal components analysis of fish abundances at each site. The strength of the correlations of physicochemical variables on each PC is indicated by the direction and length of the associated vectors. Numbers in parentheses are the amount of variance explained by each PC. Fish species on the graph exhibited loadings $> |0.4|$ with at least one PC.

Fish community metrics were quite variable, both among site groupings as well as between sites within groups (Table 3). Species diversity values ranged from 1.52 at site 19 to 6.25 at site 10, richness varied from 4 species at sites 19 and 22 to 12 species at site 10, and evenness ranged from 0.26 at site 1 (80% striped mullet) to 0.88 at site 22 (4 species represented by 2, 4, 8 and 11 individuals). Based on the site groupings identified in Figure 4, lower community diversity, richness, and evenness were exhibited by Group II sites, those in the interior of the study area that were characterized by little flow, lower pH values and lower DO levels. Species richness was generally highest in Group IV (8-11 taxa), although fish communities at these sites exhibited relatively low evenness values. It is interesting that Group I included sites that varied considerably in all three metrics, yet were physicochemically quite similar, with lower temperature and DO levels, higher pH, and some flow.

Comparison of physicochemical characteristics of sites that exhibited low and high fish species richness (Table 4) revealed few differences in many of the measured parameters. However, low-diversity sites showed a trend of slightly lower surface temperatures, slightly lower bottom DO saturation, substantially lower surface DO saturation, higher turbidities, and

higher flows. Most of these measurements were associated with high standard errors, so these data should be interpreted only as trends. However, additional data collection might reveal substantial differences in fish community richness related to these physicochemical characteristics, all of which might be expected to change with significant inputs of Mississippi River water through the Hope Canal (see Management Implications, below).

Table 3. Diversity, richness, and dominance of the fish community at the 20 sites sampled in the Maurepas swamp. Site groupings based on results of the physicochemical PCA.

Site Number	Diversity	Richness	Evenness
Group I			
9	4.15	6	0.79
10	6.25	12	0.74
11	3.17	11	0.48
22	3.39	4	0.88
Group II			
3	1.84	5	0.38
5	2.90	6	0.59
15	3.96	10	0.60
18	2.13	8	0.36
19	1.52	4	0.30
20	4.81	11	0.65
21	3.21	9	0.53
Group III			
4	4.40	6	0.83
6	3.08	9	0.51
16	5.29	10	0.72
Group IV			
1	1.71	8	0.26
2	2.32	10	0.37
12	4.19	9	0.65
13	4.58	11	0.63
14	4.58	10	0.66
17	5.25	10	0.72

Comparison of community metrics and physicochemistry at the 20 sites indicates that there are many potentially important community structuring forces that we did not measure,

emphasizing the role of numerous abiotic and biotic factors (e.g., predator and competitor density, prey abundance, and habitat structural complexity) that ultimately determine species distribution patterns. However, the proposed diversion will hopefully have significant effects on some of the parameters that we measured (e.g., flow, turbidity, salinity, and DO levels, particularly near the bottom), which was the basis for our PCA of the fish dataset and the biplots of species loadings and physicochemical correlates.

Table 4. Comparison of physicochemical characteristics at Maurepas swamp sites with high (>10 species) and low (< 6 species) fish species richness. Numbers in parentheses are standard errors.

	Species Richness	
	High	Low
Richness	10.2 (0.3)	5.2 (0.4)
Bottom Temperature	28.6 (0.3)	26.8 (0.4)
Surface Temperature	29.8 (0.5)	27.3 (0.6)
Bottom pH	6.4 (0.1)	6.3 (0.1)
Surface pH	6.5 (0.1)	6.3 (0.1)
Bottom Specific Conductance	4.0 (0.5)	4.3 (0.6)
Surface Specific Conductance	3.8 (0.6)	4.3 (0.6)
Bottom Salinity	2.2 (0.3)	2.3 (0.3)
Surface Salinity	2.0 (0.3)	2.3 (0.3)
Bottom DO Saturation	31.6 (6.3)	27.5 (4.3)
Surface DO Saturation	63.3 (8.5)	33.9 (9.6)
Bottom Turbidity	45.3 (20.1)	66.8 (36.3)
Surface Turbidity	6.7 (0.9)	14.8 (10.2)
Secchi	643.3 (42.7)	678.3 (99.5)
Flow	2.6 (0.9)	3.4 (1.8)

The fish PCA yielded six PCs that together explained 78% of the variation in the abundance data. We restricted further analyses to the first three PCs (50% of the variance explained) because these were most interpretable given the limited amount of data collected (Table 5). The plot of PC1 versus PC2 yielded species-physicochemistry associations that generally agreed with published reports of fish species habitat preferences (Figure 4). Spotted gar were negatively associated with surface DO saturation, which is not unexpected for a species that can respire aurally (Burlison et al. 1998) and has been found to be abundant in chronically hypoxic habitats in the nearby Atchafalaya River floodplain (Rutherford et al. 2001). Largemouth bass *Micropterus salmoides*, bluegill *Lepomis macrochirus*, and blue catfish were closely associated with high bottom DO saturation, and marginally associated with higher

temperatures, higher surface DO saturation, and low Secchi values. Blue catfish were not associated with high flow areas, which we predicted given their preferences for main channel habitats (Ross 2001). In fact, blue catfish were not abundant at any site, but were found in 9 locations in all four habitat groupings (Table 2; Figure 4). As expected, striped mullet, skipjack herring, and gizzard shad *Dorosoma cepedianum* were closely associated with high flow habitats, as well moderately associated with higher pH values, temperatures, and surface DO levels.

The plot of PC1 versus PC3 included three additional species that exhibited dissimilar distributions. Redear sunfish *Lepomis microlophus* and spotted bass *Micropterus punctulatus* loaded positively on PC3, and exhibited weak positive associations with bottom DO saturation, Secchi depth, and flow (Figure 5). This is not surprising for spotted bass, which are typically abundant in flowing water systems (Layer and Maughan 1987; Tillma and Guy 1998). Bowfin *Amia calva* loaded highly negatively on PC3, and although only seven individuals from four sites were collected, their distribution was weakly associated with lower flows, lower Secchi depths, higher temperatures, and lower bottom DO saturation, which is not atypical for a species well adapted to hypoxic conditions in southeastern swamp ecosystems (Hedrick and Jones 1999). The plot of PC2 versus PC3 yielded no additional information, but again emphasized the opposing distributions of bowfin and the other five species (Figure 6).

Table 5. Loadings of Maurepas swamp fishes on principal components I-III. Only loadings > |0.4| are shown for each species on each PC. Percentages refer to the amount of variance explained by each PC.

Species	PC 1 (20%)	PC 2 (16%)	PC 3 (14%)
<i>Mugil cephalus</i>	86		
<i>Alosa chrysochloris</i>	67		
<i>Dorosoma cepedianum</i>	63		
<i>Lepisosteus oculatus</i>	-91		
<i>Lepomis macrochirus</i>		89	
<i>Micropterus salmoides</i>		87	
<i>Ictalurus furcatus</i>		50	
<i>Micropterus punctulatus</i>			85
<i>Lepomis microlophus</i>			80
<i>Amia calva</i>			-48

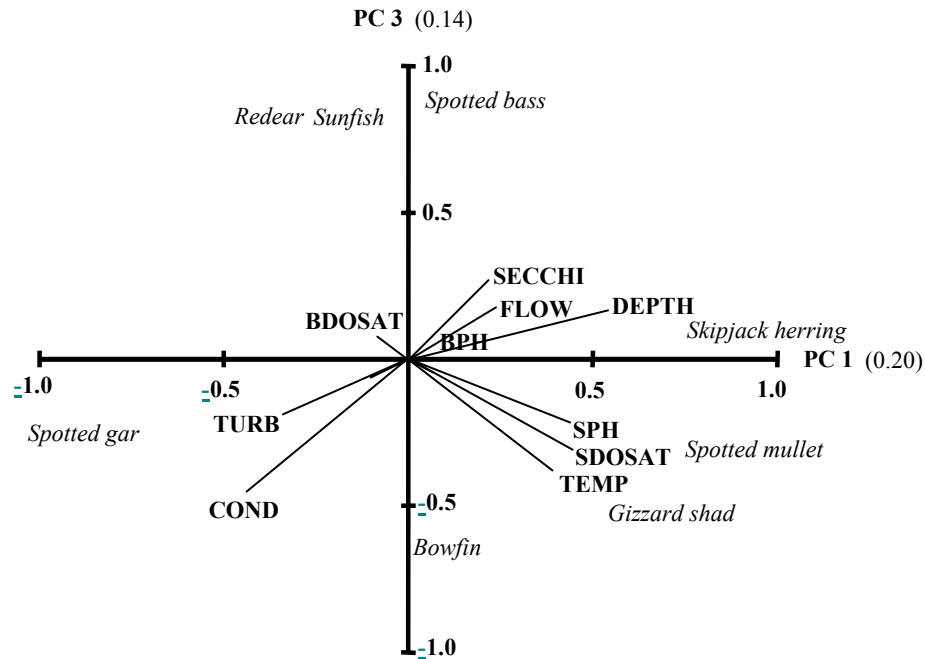


Figure 5. Loadings of Maurepas swamp fishes sampled in June 2006 on PC1 and PC3, based on the principal components analysis of fish abundances at each site. The strength of the correlations of physicochemical variables on each PC is indicated by the direction and length of the associated vectors. Numbers in parentheses are the amount of variance explained by each PC. Fish species on the graph exhibited loadings $> |0.4|$ with at least one PC.

Management Implications

It is anticipated that introduction of Mississippi River water through the Hope Canal will have several positive impacts on the physicochemistry and habitat quality of the lower Maurepas swamp. We predict that water releases will increase flow through the system, increase bottom and surface DO levels, and reduce salinity, all of which have the potential to increase the abundance and diversity of freshwater fishes in the area. Over 100 fish species have been recorded in the Atchafalaya River Basin (unpublished data), which provides a diversity of habitat conditions with regards to flow, turbidity, depth, DO, and aquatic vegetation. Although depth variability in the Maurepas swamp system is limited, inputs of freshwater will hopefully alleviate some of the stagnant conditions in the interior swamp, and should provide a greater diversity of lotic and lentic habitats for resident fishes.

On the negative side, water diversions may also introduce significant turbidity, high sediment deposition in low-flow backwater areas, and high nutrient loads, particularly nitrogen.

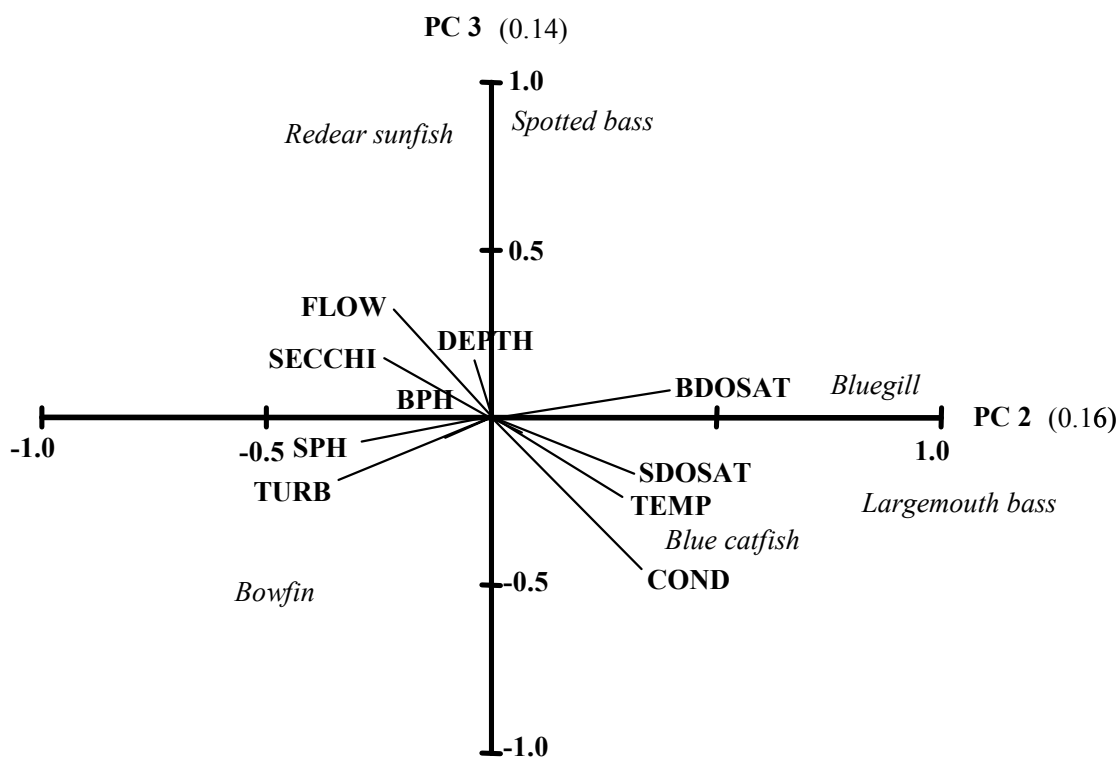


Figure 6. Loadings of Maurepas swamp fishes sampled in June 2006 on PC2 and PC3, based on the principal components analysis of fish abundances at each site. The strength of the correlations of physicochemical variables on each PC is indicated by the direction and length of the associated vectors. Numbers in parentheses are the amount of variance explained by each PC. Fish species on the graph exhibited loadings $> |0.4|$ with at least one PC.

Turbidity was not important in discriminating sites in the physicochemical PCA, but we believe high levels of turbidity would likely favor a fish community dominated by planktivores such as striped mullet and the two shad (*Dorosoma* spp.), species that loaded highly on PC1 in the fish species PCA. High sediment loads would mostly result in reduced floodplain depth and loss of swamp habitats through time, and high spring sediment deposition might also affect reproduction of nesting sunfishes (*Lepomis* spp., *Micropterus* spp., and *Pomoxis* spp.), likely the most popular sportfishes in the area.

Increasing nutrient loads would likely cause significant increases in aquatic macrophyte densities, which could reduce fish habitat and significantly impact water quality, particularly nocturnal DO levels. Many sites sampled in June 2006 (e.g., sites 7 and 8, which were too shallow to sample, 3, 5, and 19) were 100% covered with common salvinia *Salvinia minima*, and hydrilla *Hydrilla verticillata*. Lotus *Nelumbo lutea*, and water hyacinth *Eichhornia crassipes* were also recorded at several other sites. The potential for exotic macrophytes such as hydrilla,

salvinia, and water hyacinth to increase tremendously in distribution and abundance, exclude native plants, and completely alter the ecology of the littoral zone has been readily apparent in the Atchafalaya River Basin over the past two decades (e.g., Colon-Gaud et al. 2004; Poday et al. 2006). Significant increases in nutrient levels would likely increase the distribution and bed density of these plants in the Maurepas swamp as well. Water hyacinth and salvinia are particularly problematic, as they can completely cover the water surface in stagnant areas, precluding photosynthesis and turbulent mixing, and resulting in a hypoxic and essentially uninhabitable water column. Conversely, large stands of macrophytes would be effective in stripping nutrients and sediment from Mississippi River water, increasing the quality of water entering Lake Maurepas. As a consequence, evaluating the positive and negative impacts of macrophyte stands in the diversion area will depend on the overall objectives of the project regarding the role of the swamp in the Lake Maurepas ecosystem.

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Marine Fishery Species

Megan La Peyre, Bryan Gossman, Gary Peterson

Methods

Fish and decapod crustaceans were collected during fall 2005 and 2006 and spring 2006 in the southeastern portion of Lake Maurepas at a total of five habitat types (open water mud bottom, shell bottom, floating vegetation, emergent marsh edge and forested edge). All five habitats were sampled by electrofishing; in addition, shell bottom and open water habitats were sampled by otter trawl. Samples were collected at a total of 10 stations, located along the southern and eastern portions of Lake Maurepas between Blind River and Pass Manchac, although not all stations were sampled in fall 2005 (Figure 7, Table 6). Based on expected differences in salinity in the Lake, we stratified our sampling area into three zones to ensure that we covered the salinity range.



Figure 7. Map of Lake Maurepas, LA showing salinity zones and sampling sites for nekton sampling in 2005 and 2006. Sites 1 and 4 are marsh edge. Sites 2, 3, and 5 are tree stump edge. Site 6 is open water with shell bottom. Site 7 is floating vegetation. Sites 8-10 are open water with mud bottom. Sites C1 and C2 are located at the mouth of canals.

Table 6. Sampling dates, method, stations sampled and effort for Lake Maurepas marine fish sampling

Dates	Method	Stations Sampled	Number of Samples
Nov. 8-9, 2005	Electroshock	3-7, C1, C2	14
	Otter trawl	-	-
June 13-15, 2006	Electroshock	1-7	16
	Otter trawl	7-10	20
Sept. 5 – 6, 2006	Electroshock	1-7	28
	Otter trawl	7-10	10

Electrofishing was conducted using a Smith-Root Model 7.5 GPP Electrofishing system. This unit was mounted in a 20-foot aluminum boat, with the boat hull serving as the cathode, and an 11 inch anode ring (with an attached net) mounted to a hand-held, fiberglass prod-pole. The anode/net operator and a second dip-net person worked from the bow, and a third person operated the boat and the electrofisher controls. This set-up (versus a boom-mounted anode array) allowed the electrical field to focus among stumps, brush-tops, and irregularities in the shorelines. Electric current was pulsed on and off as needed with a foot-operated switch operated by the anode person. Fish were placed in aerated tubs of water for recovery. Sampling time was standardized to 10 minutes of non-continuous “shock time” as measured by the second-counter on the electrofisher unit. Each sample actually took about 15-25 minutes to collect as the shocker would be turned off as we netted fish and placed them in oxygenated containers. Current flow was adjusted, depending on water conductivity, to the minimum level which would effectively stun the fish for easy capture. Most fish and decapod crustaceans were measured, weighed and released live at the conclusion of each sample. Those organisms which could not be accurately measured or identified in the field were immediately placed in an ice-slurry and processed in the lab. Two consecutive independent 10 minute shock time samples were collected at each habitat site. Catch per unit effort (CPUE) was calculated as the number of organisms caught per 10 minute shock event.

In fall 2005, one shell bottom habitat site, one floating vegetation habitat site, one emergent marsh edge habitat site, and three forest edge habitat sites were electroshocked. Furthermore, we sampled at two canal mouths which we dropped in later sampling efforts as canals were covered by the freshwater sampling group. In spring and fall 2006, we sampled one floating vegetation, one shell, two emergent marsh edge and three forest edge habitats. With the

exception of the forest edge habitat type, we sampled all available sites where the other three habitat types occurred. We sampled open water, soft-bottom but did not capture any organisms.

Standard otter trawl sampling consisted of three 10 minute tows with an 8' standard otter trawl. Both open water, soft bottom habitat and shell bottoms were sampled at three habitat sites with a trawl in spring and fall 2006.

Results and Discussion

Environmental variables

Temperature, salinity, dissolved oxygen and turbidity (Secchi) were all typical of an intermediate environment and varied by season (sample event). Mean temperature and salinity were similar between June and September 2006 samples but differed from November 2005 samples. Dissolved oxygen (DO) and Secchi measures did not vary by sample event (Table 7).

Table 7 Mean (SD) of environmental variables collected at each sample event

Variable	Nov 05	June 06	Sept 06
Temperature (°C)	21.0 (1.4)	30.4 (1.9)	29.6 (1.1)
Salinity (PSU)	1.9 (0.4)	3.7 (0.7)	3.3 (1.1)
Dissolved oxygen (mg L ⁻¹)	7.8 (1.3)	5.8 (1.2)	6.2 (0.8)
Secchi depth (cm)	102 (27)	107 (31)	157 (45)

As samples covered a large area of Lake Maurepas with various freshwater and marine influences, we examined the environmental data by zone to determine if there were differences in environmental variables by zone within each sampling period. In June, there was no significant difference by zone for any of the environmental variables. In contrast, in November 2005, temperature ($p=0.0003$) and salinity ($p=0.0013$) differed significantly by zone. In Sept. 2006, dissolved oxygen and Secchi depths also differed significantly ($p=0.0078$ and 0.0005 respectively) with dissolved oxygen means ranging from 5.5 to 6.8 to 6.7 in zones 1 through 3 respectively and Secchi depths ranging from 160 cm to 205 cm to 105 cm in zones 1 through 3 respectively.

Nekton samples

A total of 1,853 individuals representing 41 species were collected during this study (Table 8). There were 36 species of fish, and 5 crustacean species. Overall catch was dominated by striped mullet, *Mugil cephalus* which accounted for 39% of the total number of individuals

caught, and 46% of the total number of fish species caught. White shrimp, *Litopenaeus setiferus* was the second most dominant species accounting for 13% of the total number of individuals caught, and 83% of decapod crustaceans captured. Except for bay anchovy, *Anchoa mitchilli*, which accounted for 13% of total catch, all other species accounted for less than 5% of the total number of individuals collected. Electrofishing accounted for over 99% of individuals collected. Otter trawl sampling collected two species not collected by electrofishing, naked goby, *Gobiosoma bosc* (2 in spring 2006 and 1 in fall 2006) and one gulf pipefish, *Syngnathus scovelli*, in fall 2006. Overall trawl catches were dominated by bay anchovy (81% of total catch).

Sample event effects

Total number of nekton captured, CPUE and biomass did not differ significantly between sample events (Figure 8). The number of species captured was lowest in November 2005 sample event (23 species) as compared to June 2006 and September 2006 (30 and 33 species respectively) (Figure 9).

Habitat type effects

CPUE was highest across all dates for floating vegetation habitat (and canals which were only sampled in Nov. 2006 by the marine fishery group), with marsh and forest edges following closely behind. Catch over shell bottoms and open water tended to be low, likely partly due to the choice of sampling gear, and time (Figure 8, Table 9).

Species of interest results

The effects of the proposed diversion on six estuarine species identified by the National Marine Fisheries Service (NMFS) (white shrimp, red drum, striped mullet, Atlantic croaker, gulf menhaden and blue crab) were of special interest. These species are of special interest to NMFS as they are either federally managed species, or considered to be important estuarine species in the Lake Maurepas system. Of the six species of interest (white shrimp, red drum, striped mullet, Atlantic croaker, gulf menhaden and blue crab), only a few individuals of Atlantic croaker, blue crab, gulf menhaden and red drum were collected during this sampling (Table 10). Striped mullet, another species of interest, were abundant throughout all three sample events and were caught in high numbers particularly along the stump edges, followed by marsh edges and floating vegetation. White shrimp were also abundant, and most were caught along stump and

marsh edges. The total number of white shrimp collected likely underestimates those in the sampling area due to a limited netting efficiency of schooling organisms. White shrimp were caught predominantly in zones 2 and 3, which tended to have lower salinity, and lower visibility (higher turbidity).

Table 8. Total catch table for sampling in Lake Maurepas, LA in 2005 and 2006. Highlighted in grey are species of interest. Gear type s= electroshock, t= otter trawl.

Common Name	Scientific Name	Gear Type	Total Caught	Nov. 2005	Jun. 2006	Aug. 2006
FISHES						
Striped Mullet	<i>Mugil cephalus</i>	s	731	333	199	199
Rough Silverside	<i>Membras martinica</i>	s	78	77	1	0
Spotted Gar	<i>Lepisosteus oculatus</i>	s	78	46	14	18
Bluegill	<i>Lepomis macrochirus</i>	s	62	59	1	2
Largemouth Bass	<i>Micropterus salmoides</i>	s	50	10	14	26
Bay Anchovy	<i>Anchoa mitchilli</i>	s,t	238	39	184	15
Gulf Menhaden	<i>Brevoortia patronus</i>	s,t	43	8	35	0
Yellow Bass	<i>Morone mississippiensis</i>	s	42	0	5	37
Inland Silverside	<i>Menidia beryllina</i>	s,t	31	8	22	1
Redear Sunfish	<i>Lepomis microlophus</i>	s	26	14	1	11
Gizzard Shad	<i>Dorosoma cepedianum</i>	s,t	26	4	16	6
American Eel	<i>Anguilla rostrata</i>	s	22	4	5	13
Freshwater Drum	<i>Aplodinotus grunniens</i>	s	19	6	10	3
Longear Sunfish	<i>Lepomis megalotis</i>	s	19	6	5	8
Spotted Seatrout	<i>Cynoscion nebulosus</i>	s	15	0	0	15
Channel Catfish	<i>Ictalurus punctatus</i>	s,t	13	1	7	5
Hogchoker	<i>Trinectes maculatus</i>	s,t	12	8	2	2
Blue Catfish	<i>Ictalurus furcatus</i>	s,t	10	1	4	5
Red Drum	<i>Sciaenops ocellatus</i>	s	7	0	2	5
Redspotted Sunfish	<i>Lepomis miniatus</i>	s	6	1	0	5
Ladyfish	<i>Elops saurus</i>	s,t	5	0	1	4
Alligator Gar	<i>Atractosteus spatula</i>	s	3	0	1	2
Black Crappie	<i>Pomoxis nigromaculatus</i>	s	3	2	0	1
Southern Flounder	<i>Paralichthys lethostigma</i>	s	3	0	1	2
Atlantic Croaker	<i>Micropogonias undulatus</i>	s	15	0	13	2
Atlantic Needlefish	<i>Strongylura marina</i>	s	2	0	1	1
Spot	<i>Leiostomus xanthurus</i>	s	2	0	0	2
Spotted Bass	<i>Micropterus punctulatus</i>	s	2	2	0	0
Warmouth	<i>Lepomis gulosus</i>	s	2	2	0	0
Bay Whiff	<i>Citharichthys spilopterus</i>	s	1	0	0	1
Common Carp	<i>Cyprinus carpio</i>	s	1	1	0	0
Longnose Gar	<i>Lepisosteus osseus</i>	s	1	0	1	0
Sand Seatrout	<i>Cynoscion arenarius</i>	s	1	0	0	1
Skipjack Herring	<i>Alosa chrysochloris</i>	s	1	0	1	0
Naked Goby	<i>Gobiosoma bosc</i>	t	3	0	2	1
Gulf Pipefish	<i>Syngnathus scovelli</i>	t	1	0	0	1
Total Fishes			1574	632	548	394
CRUSTACEANS						
White Shrimp	<i>Litopenaeus setiferus</i>	s	232	92	2	138
Brown Shrimp	<i>Farfantepenaeus aztecus</i>	s,t	31	0	34	0
Blue Crab	<i>Callinectes sapidus</i>	s,t	11	3	7	4
River Shrimp	<i>Macrobrachium olfersii</i>	s	4	0	3	1
Mud Crab	<i>Xanthidae sp.</i>	s	1	0	0	1
Total Crustaceans			279	95	46	144
Total Catch			1853	727	594	538

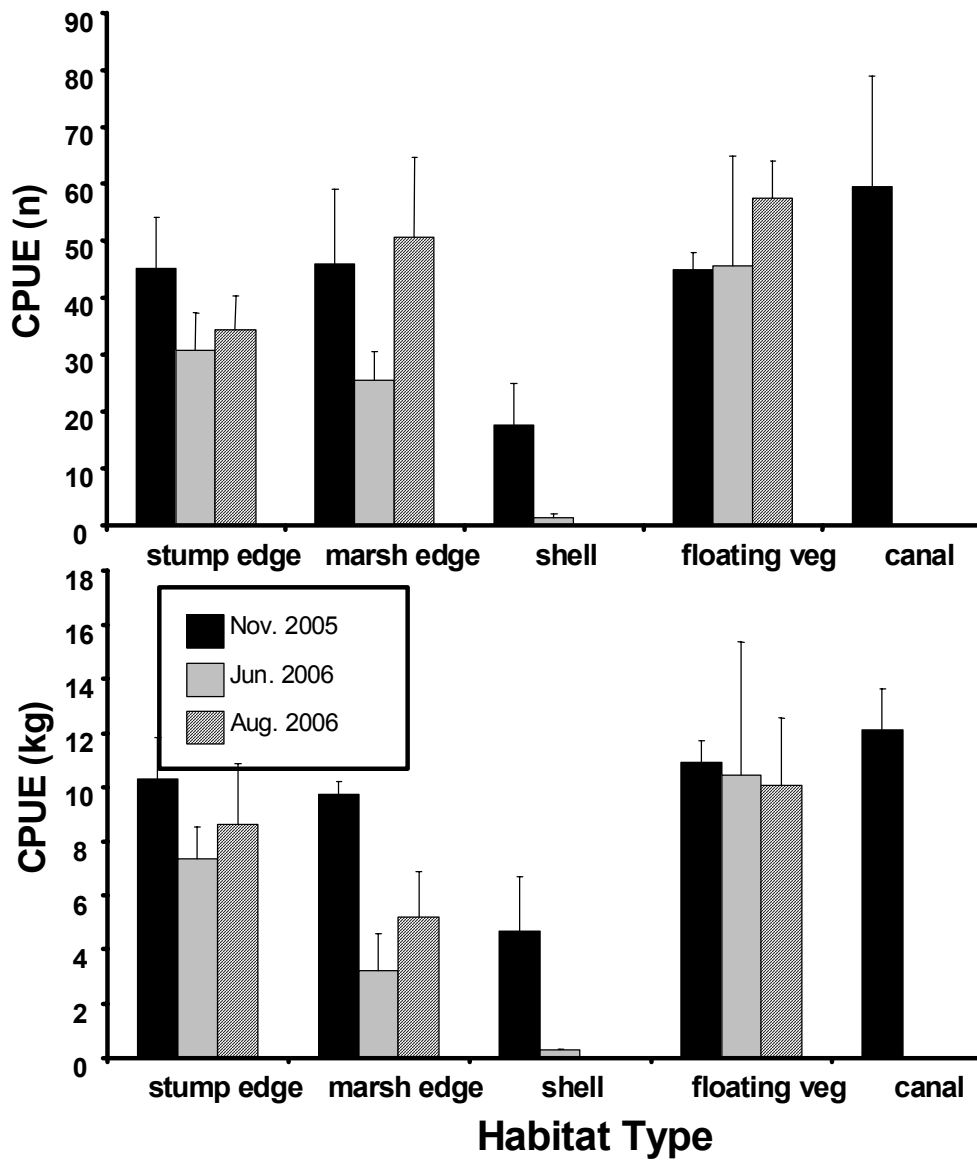


Figure 8. Catch per unit effort (CPUE) by habitat type and sample event in Lake Maurepas, LA. Top graph presents CPUE by the number of individuals caught, bottom graph presents biomass of organisms caught. Mean (SD) are presented. The 'canal' habitat type was only sampled in November 2005.

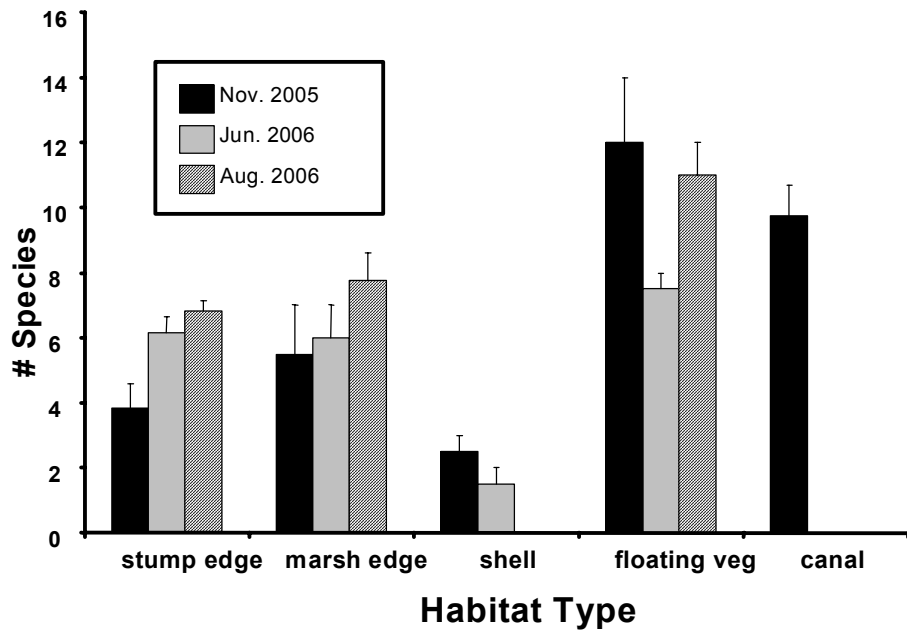


Figure 9. Number of species by habitat type and sample event in Lake Maurepas, LA. Mean (SD) are presented. The 'canal' habitat type was only sampled in November 2005.

Table 9. Total catch by habitat, date and gear type. # = number of individuals collected, spp. = number of species collected. Trawling was not conducted in November 2005, and canal mouth samples were only taken in November 2005.

Date	Electroshocking															Otter Trawl Samples					
	Floating vegetation			Marsh edge			Forest edge			Shell bottom			Canal mouth			Open water			Shell bottom		
	#	CPUE	spp.	#	CPUE	spp.	#	CPUE	spp.	#	CPUE	spp.	#	CPUE	spp.	#	CPUE	spp.	#	CPUE	spp.
Fall 05	90	45	17	92	46	8	271	45	10	36	18	3	238	60	13	-	-	-	-	-	-
Spring 06	91	46	10	102	26	14	185	31	14	3	2	2	-	-	-	174	22	7	40	10	8
Fall 06	115	58	19	203	51	19	206	34	18	0	0	0	-	-	-	13	2	2	3	2	3
Total	296			397			662			39			238			187			43		

Table 10. Catch (# and CPUE) of species of interest by date and habitat type. Shell bottom was sampled with both a shocker and by otter trawl. Trawling only occurred in Spring and Fall 2006. Actual sample dates were November 2005 (Nov. 8-9, 2005), June 2006 (June 13-15, 2006) and September 2006 (Sept. 5-6, 2006). CPUE was calculated for shock and trawl data by dividing the number of each species caught by the number of 10 minute shock or trawl events that occurred for each habitat type for each date.

Species	Date	Shock								Trawl			
		Floating vegetation		Marsh edge		Stump edge		Shell bottom		Open water		Shell bottom	
		#	CPUE	#	CPUE	#	CPUE	#	CPUE	#	CPUE	#	CPUE
Atlantic croaker	Fall 05	0	0	0	0	0	0	0	0	0	-	-	-
	Spring 06	0	0	0	0	0	0	0	0	9	4.5	7	3.5
	Fall 06	2	1	0	0	0	0	0	0	0	0	0	0
Blue crab	Fall 05	0	0	0	0	3	0.5	0	0	0	-	-	-
	Spring 06	0	0	1	0.25	3	0.5	0	0	1	0.25	2	0.5
	Fall 06	0	0	3	0.75	1	0.17	0	0	0	0	0	0
Gulf menhaden	Fall 05	6	3	0	0	0	0	0	0	0	-	-	-
	Spring 06	33	16.5	0	0	0	0	0	0	1	0.25	0	0
	Fall 06	0	0	1	0.25	0	0	0	0	0	0	0	0
Red drum	Fall 05	0	0	0	0	0	0	0	0	0	-	-	-
	Spring 06	0	0	0	0	2	0.33	0	0	0	0	0	0
	Fall 06	1	0.5	1	0.25	3	0.5	0	0	0	0	0	0
Striped mullet	Fall 05	28	14	54	26	195	32.5	32	16	0	-	-	-
	Spring 06	29	14.5	30	7.5	140	23.3	0	0	0	0	0	0
	Fall 06	41	20.5	22	5.5	136	22.7	0	0	0	0	0	0
White shrimp	Fall 05	7	3.5	16	8	57	9.5	1	0.5	0	-	-	-
	Spring 06	0	0	2	0.5	0	0	0	0	0	0	0	0
	Fall 06	2	1	114	28.5	22	3.7	0	0	0	0	0	0

The most recent previous survey of fish fauna in Lake Maurepas was completed in the early 1980s (Hastings et al. 1987). This study only reported catches of gulf menhaden, Atlantic croaker and striped mullet, from collections by trawl, multi-mesh gill net, large mesh gill net and rotenone. An analysis of fish assemblages over 50 years in the Lake Pontchartrain estuary (O'Connell et al. 2004) collected gulf menhaden, (10 % catch), Atlantic croaker (10% catch) and striped mullet (1 % of catch). However, this study focused on catch solely from the saltier Lake Pontchartrain and did not have samples from Lake Maurepas.

Diversion and predicted Lake Maurepas environmental effects

To date, modeling efforts associated with the proposed diversion are limited to impacts on the swamp environment. Salinity effects on Lake Maurepas are predicted as a diversion of 1,500 cfs represents a 45% increase in freshwater input which is estimated to be equivalent to almost two complete turnovers of lake volume over a one year time period (Shaffer et al. 2001, Day et al. 2004). Similarly, a 2,500 cfs is estimated to decrease lake salinity by 30% only after it has been flowed for one month. In contrast, while the diversion water is expected to be cooler than the lake water, current models predict that it will take a minimum of 10 days before any diversion water reaches the edge of the swamp, and hence Lake Maurepas. Similarly, turbidity is not expected to be an issue as most suspended sediments carried in the river water are expected to be retained within the swamp (Shaffer et al. 2001). Early reports predict no adverse water quality effects or extreme or persistent algal blooms in Lake Maurepas (Lane et al. 2003).

Of concern to marine fishery managers are the possible effects of any temperature changes resulting from the proposed diversion on selected fish species. In general, it is believed that many fish species can discriminate small differences in temperature and often have well-defined temperature preferences (Brett 1971, Magnuson et al. 1979). As temperature controls the metabolic rate, and thus the energy available to the fish, temperature can provide an important control on growth rates, reproduction, foraging rates (Jobling 1981, Taylor and Collie 2003, Craig and Crowder 2005). Below is a brief summary of the known temperature and salinity effects on the six species of special management interest. Predictive models of the diversions influence on physiochemical water properties suggest a salinity impact, rather than a temperature impact, therefore salinity preferences for the species of special interest to fisheries managers are included below.

White shrimp - Seasonal variability in abundances of *L. setiferus* have been attributed to temperature, salinity and water level with salinity accounting for the most variability (Zimmerman and Minello 1984). White shrimp have been found in salinities ranging from 0.42 ppt to 43.3 ppt (Gunter and Hildebrand 1954, Muncy 1984) and juveniles are found to grow in all these salinities (Zein-Eldin 1963, Muncy 1984). In Louisiana, white shrimp have been sampled 160 km upriver (Perez-Farfante 1969). Recent work with postlarval shrimp and rapid salinity changes found that growth rate increased with salinity, up to 40 ppt (Rosas et al. 1999). However, spawning takes place in offshore waters with salinities are greater than 27 ppt, and lab experiments have shown that larvae can survive salinities ranging from 18-34 ppt (Perez-Farfante 1969). No other research was located that explicitly examined effects of salinity on growth or survival of white shrimp, particularly in the estuary.

Survival of *L. setiferus* is reduced at low temperatures. A critical thermal minimum has been identified at 8C and 100% mortality at 3C or lower, regardless of salinity (Muncy 1984). Adults tend to be more susceptible to cold temperatures as compared to juveniles (Whitaker 1983). For adults, growth is limited at much higher temperature: *L. setiferus* growth ceases at temperatures below 16C (Etzold and Christmas 1977) and generally occurs only in temperatures above 20C (Etzold and Christmas 1977).

Blue crab - Similar to white shrimp, seasonal variability in abundance of *C. sapidus* has been attributed to change in temperature, salinity and water level, but water level and temperature accounted for the most variability (Zimmerman and Minello 1984). Hatching and larval development occurs under narrow range of salinities and water temperatures offshore (Costlow and Bookout 1959, Costlow 1967), juveniles and adults can tolerate and have been collected within a wide range of salinities (0 to 40 ppt) and water temperatures (0 to 40C) (Tagatz 1969, Mahood et al. 1970, Winget et al. 1976, Copeland and Bechtel 1974). Explicit work on the effects of temperature and salinity suggests that juvenile blue crabs are likely not limited by either condition directly, although relationships between environmental conditions such as timing of temperature changes, rate of temperature change and river discharge have been found to relate to blue crab abundances (Turner 1979, Rogers et al. 1990, Wilbur 1994). The most explicit work on the effects of temperature examined temperature and salinity on the size at maturity of female blue crabs (Fisher 1999). This study found that crabs mature at smaller sizes

as temperature and salinity increase. This study examined salinity in the range of 0 to 30 ppt, and up to 25C.

Atlantic croaker – Atlantic croaker abundances have been shown to be affected by both temperature and salinity. Highest abundances of croaker have been reported in waters ranging from 21 to 25C (Parker 1971, Miglarese et al. 1982). Miglarese et al. (1982) also found no correlation between size and temperature. A number of field and laboratory studies have determined that juveniles are more tolerant of lower temperatures than adults. For example, Parker (1971) reported that juveniles have been collected in waters with temperatures as low as 0.4C, while adults only in waters as low as 5C. Similarly, laboratory tests established lower and upper lethal temperatures of 0.6 and 38 C for juveniles and 3.3 degrees and 36C for adults, respectively. The effects of cold stress have also been tested on croaker with a lab study finding survival at 99.3 % for temperatures held at 7C; the rate of temperature reduction did not affect survival rate (i.e., 1C /day versus 0.2C /day). They also found that cold tolerance increased with increasing salinity (Lankford and Target 2001).

Atlantic croaker are highly euryhaline and have been collected in waters ranging in salinity from 0.2 to 75 ppt (Parker 1971), although survivorship at the higher salinities is unknown. Miglarese et al. (1982) collected croaker in salinities ranging from 0.4 to 34.4 ppt with a high correlation between size and salinity during fall seasons over a two year period. Results of lab studies indicate croaker can withstand salinity fluctuation, but field observations suggest that croaker tend to swim to avoid areas of unstable salinity (Perez 1969). As with other species, it has been suggested that croaker response to salinity variation may not be due to salinity per se, but to some factor correlated with salinity changes (Perez 1969). For example, salinity fluctuations resulting from increased freshwater runoff are probably correlated with a number of other factors, such as altered current regimes, higher turbidity and change in temperature. A recent lab study examined the effects of salinity on short term growth rates of juvenile Atlantic croaker (Peterson et al. 1999). Peterson et al. (1999) found significant differences in body growth at all three salinity treatments (ranging from 5 to 20 ppt) with fish in the 5 ppt treatment growing the fastest. This study was able to demonstrate that salinity could be a key factor causing variability in short-term growth rates in estuarine nursery areas.

Gulf menhaden - Gulf menhaden have been reported in waters ranging from 5 to 35C, although avoidance of water temperatures above 30C has been reported (Galloway and Strawn 1974). In the Mississippi Sound, highest catches of gulf menhaden occurred in waters between 25 and 35C and between salinities of 5 and 25 ppt (Christmas and Waller 1975). Gulf menhaden have been reported in waters ranging from 0 to 67 ppt, although it is likely that salinity ranges are narrower when broken down by life stages. Generally, gravid adults, fertilized eggs, and early larvae are associated with the higher salinity waters. Postlarvae and juveniles appeared to tolerate a wide salinity range, but are most often associated with nearshore, low salinity areas. There are no definitive reports on lower lethal limits for temperature or salinity.

There is limited information on the effects of extreme temperature or salinity on growth or survival of gulf menhaden. One study, in Fourleague Bay, LA found that fish growth was slower and mortality rates were higher during periods of high river discharge combined with cool spring temperatures (Deegan 1990). Similarly, low winter temperatures, high salinities and low turbidity during their stay in estuaries are correlated with poor year classes (Christmas et al. 1982, Guillory et al. 1983); these environmental factors are believed to influence growth and mortality rates of young of the year by affecting food supply.

Red drum - Red drum are euryhaline species and have been reported in salinities ranging from 0 to 50 ppt, although an optimum salinity range of 20-40 ppt has been suggested (Reagan 1985). In general, juveniles are more common at low salinities. Temperature range for red drum is reported to be from 2 to 33C with movement patterns often triggered by changes in temperatures. Adams and Tremain (2000) found that abundance of red drum was negatively correlated with water temperature and water levels in the winter; no correlation with salinity was found. One study examined the effects of salinity and temperature on egg hatch and larval survival and found that low salinity was associated with poor survival (Holt et al. 1981). More importantly, postlarvae, which are more likely to be found inland were found, at two weeks of age to have highest survival at 15C and that growth rates increased with increases in salinity and temperature (Holt et al. 1983).

Striped mullet– There has been several studies on temperature and salinity minima and maxima for different stages of *M. cephalus*. Sylvester et al. (1974, 1975) found that optimal temperature

and salinity conditions for eggs was 19.5-20.5C and 30-32 ppt and that greatest hatching success occurred between 22 and 25C. However, tested at a higher salinity, Walsh et al. (1991) found that hatching time was affected by temperature but not salinity with an optimum at 25.5C and 36.3 ppt. *M. cephalus* juveniles appear to have greater tolerance for extensive diel fluctuations and ranges in both temperature and salinity (Major 1978, Chubb et al. 1981). Pre-juvenile individuals (< 50 mm) have been found in temperatures as high as 42.5C and salinities from 2 to 30 ppt (Major 1978). Major (1978) further found that as body size increased, the temperature range in which individuals were found decreased. A study examining the effects of cold stress on *M. cephalus* juveniles (Kulikova et al. 1986) found that juveniles are active above 10C; below this temperature, juveniles sink toward the bottom and become less active, to where feeding will eventually cease (approximately 2-3⁰C). No long term effects of short cold periods on mullet were detected (temperatures at 7⁰C).

Adults are euryhaline and have been found from freshwater to salinities as high as 90 ppt (Wallace 1975) and can withstand abrupt salinity changes from 0 to 45 ppt (Hossler et al. 1979). However, a lab study of young *M. cephalus* found that salinity affected growth (Murashige et al. 1991) with 22-23 ppt providing optimal growth rates over higher salinities. No testing of lower salinities and growth has been reported.

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American Alligator

David M. Fox and Philip C. Stouffer

Introduction

The Louisiana Department of Wildlife and Fisheries estimates American alligator (*Alligator mississippiensis*) populations and sets harvest quotas by extrapolating from aerial nest-counts (McNease and Joanen 1978, Joanen et al. 1984, Ruth Elsey, Louisiana Department of Wildlife and Fisheries, *pers. comm.*). Cypress-tupelo swamp habitat is not surveyed by LADWF because alligator nests are hidden by forest canopy (Lance Campbell, Louisiana Department of Wildlife and Fisheries, *pers. comm.*). Alligator density in swamp habitat was estimated to be 1 alligator per 2.6ha (Joanen et al. 1981, as cited in Joanen et al. 1984), which is similar to densely populated intermediate marsh habitat (1 alligator per 2.1ha, Joanen et al. 1984). However, no studies have been published describing the abundance or natural history of alligators in cypress-tupelo swamp habitat.

Methods

Spotlight surveys are a simple technique for collecting relative abundance and size class distribution data for alligators (Chabreck 1966). We conducted spotlight surveys on waterways in cypress-tupelo swamp along three alternative diversion routes: the route northeast of Romeville, from the drainage canal north and east of LA route-3124 to the Blind River and then to I-10; Hope Canal from Airline Highway to I-10; and Reserve-Relief Canal from Airline Highway to the shore of Lake Maurepas. For comparative purposes, surveys were also conducted in intermediate marsh habitat on the east side of Lake Maurepas on Ruddock Canal and nearby bayous. In addition to these surveys, a single survey of Hope Canal, beginning approximately 2km north of I-10 and ending at the intersection of Hope Canal and Tent Bayou, was conducted in June during a failed attempt to access the southern portion of Hope Canal from Lake Maurepas.

Surveys were conducted once per month in June, July and August 2006 at times when low-tide and low moonlight-intensity coincided. Surveys were conducted after sunset from an outboard motor boat or a canoe. One or two observers conducted the

survey while one person piloted the boat and assisted with spotlighting. Two 1-million candle-power spotlights were used to detect alligator eyeshine. When alligator eyeshine was detected, the alligator was approached and the snout length, from the anterior corner of the eye to the end of the snout, was estimated in inches. Alligator snout length is highly correlated with total length, and 1 inch of snout length is equal to about 1 foot of body length (Chabreck 1966). Alligators for which exact snout length estimates were not obtained were classified as either greater than 4 feet, or less than 4 feet long. Alligators that submerged before we were able to approach were classified as ‘unknown’. Alligator density was calculated as the number of detections per kilometer of surveyed habitat, which included both shorelines of waterways.

Results

With the exception of Hope Canal in August, alligator density was greater than 10 per kilometer across survey routes and months (Table 11, Figure 10). The highest alligator density was observed during a single survey of the northern portion of Hope Canal, but within surveys, density in Ruddock Canal and nearby bayous was higher than in any of the alternative diversion routes (Table 11, Figure 10). Alligator density generally declined with time across survey routes (Table 11, Figure 10).

Table 11. Total alligator detections and alligator density on spotlight survey routes in Maurepas Swamp, June-August 2006.

DATE	ROUTE	TOTAL DETECTIONS	ROUTE LENGTH (km)	DETECTIONS PER km
29-June-06	BLIND RIVER	173	13	13.31
27-July-06	BLIND RIVER	165	13	12.69
23-August-06	BLIND RIVER	153	13	11.77
28-June-06	RESERVE-RELIEF CANAL	97	8.32	11.67
26-July-06	RESERVE-RELIEF CANAL	159	10.04	15.84
24-August-06	RESERVE-RELIEF CANAL	121	10.04	12.05
27-June-06	RUDDOCK CANAL	46	2.9	15.86
24-July-06	RUDDOCK CANAL	103	4.57	22.54
27-Aug-06	RUDDOCK CANAL	89	4.57	19.47
30-June-06	N. HOPE CANAL	118	4.8	24.58
7-July-06	HOPE CANAL	74	5.5	13.46
28-July-06	HOPE CANAL	60	5.5	10.91
21-August-06	HOPE CANAL	28	5.5	5.09

Size class distributions of alligators were similar among survey routes (Figures 2-4), and also similar to pre-Hurricane Rita distributions observed during spotlight counts

by biologists at the Rockefeller National Wildlife Refuge in southwestern Louisiana (Ruth Elsey, *pers. comm.*). Size class distributions within survey routes varied slightly between months, but of the alligators for which a size estimate was made, alligators in size classes less than four feet long were the most frequently detected (Figures 2-4). Few alligators under 1 foot long, or over seven feet long were detected, one 12-13 foot alligator was detected in the Blind River in June and July, but after that night's survey had been completed. Hope Canal consistently had the greatest proportion of alligators of unknown size (Figures 11-13). Although the trend was not documented, alligator density, and the density of large alligators, appeared to increase with proximity to Lake Maurepas.

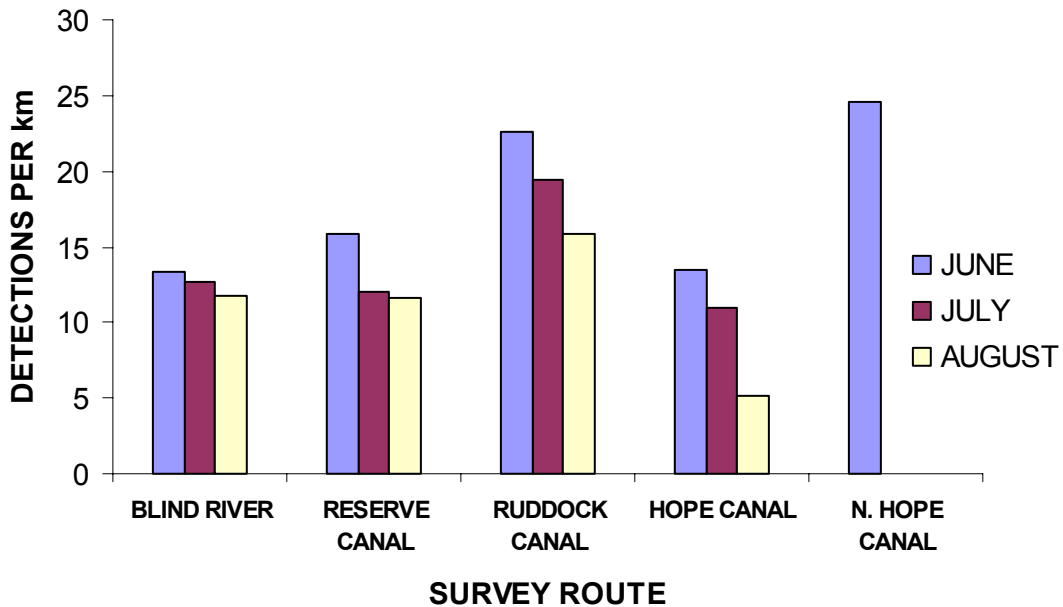


Figure 10. Alligator density in the Blind River, Hope Canal, Reserve-Relief Canal, and Ruddock Canal, June-August 2006. Northern Hope Canal was not surveyed in July and August.

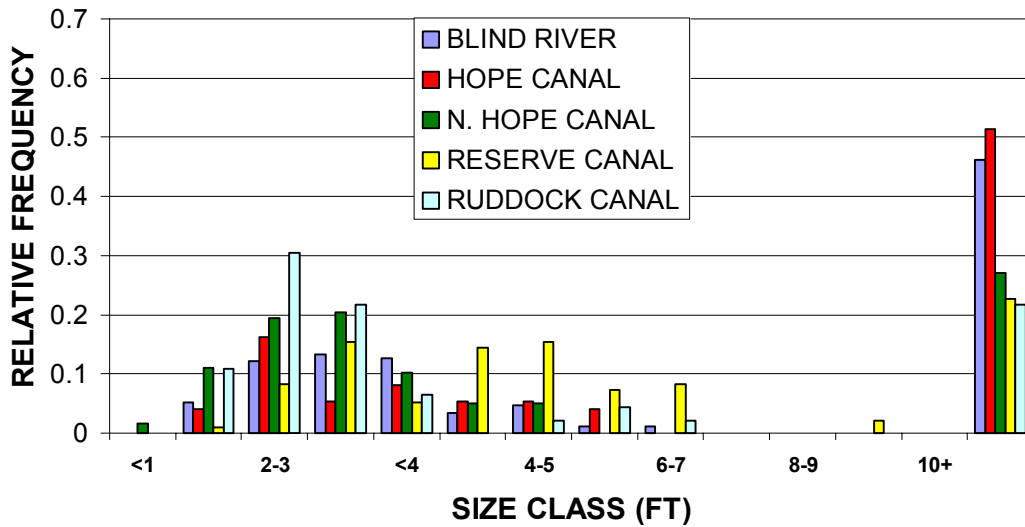


Figure 11. Relative frequency of alligator size classes detected during spotlight surveys in Maurepas Swamp, June 2006.

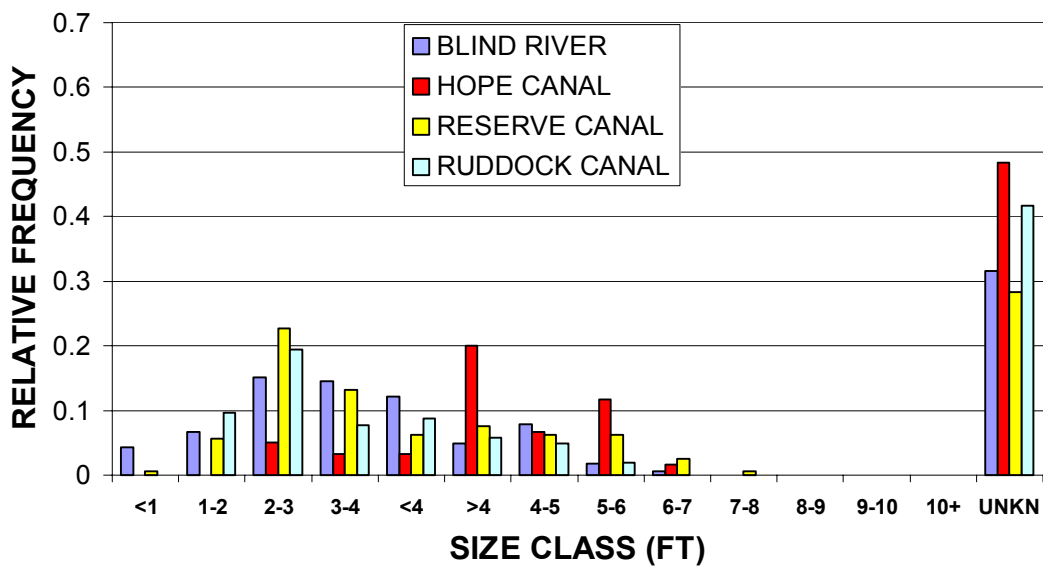


Figure 12. Relative frequency of alligator size classes detected during spotlight surveys in Maurepas Swamp, July 2006.

Discussion

High densities of alligators were observed in both marsh and swamp habitats in Maurepas Swamp. Because our surveys were limited to counts within a single year, trends in abundance and robust conclusions about the relative value of the alternative diversion routes for alligators cannot be determined. Furthermore, our surveys were conducted in the first year after Hurricanes Katrina and Rita impacted coastal Louisiana.

The effects of the hurricanes on alligator populations in Maurepas Swamp are unknown. Hurricane effects on hatching success or alligator survival would be reflected in our relative abundance and size class distribution data, but these effects will remain undetected without data from previous years. However, if hurricanes in 2005 depressed alligator populations in 2006, alligator densities remained high across the study area and may increase with time if the population is recovering.

Our results may have been affected by several factors. Alligator abundance estimates derived from spotlight surveys may vary widely within and between years, and between habitat types (Wood et al. 1985, Woodward et al. 1996, Lance Campbell, *pers. comm.*). Spotlight surveys may detect only 10-25% of alligators (Taylor and Neal 1984, Woodward et al. 1996). We suspect that our detection rates may have been at the high end of this range, because drought conditions in southeastern Louisiana in summer of 2006 probably concentrated alligators in deepwater habitat along our survey routes. However, low water levels may have affected the distribution of alligators along survey routes, and the decline in density with time may have been due to alligators emigrating to areas near the lake with deeper water and less dense aquatic vegetation. Although it was not documented, alligator density in the Blind River was higher than in the canal included in this route, and density was higher in the river north of Airline Highway.

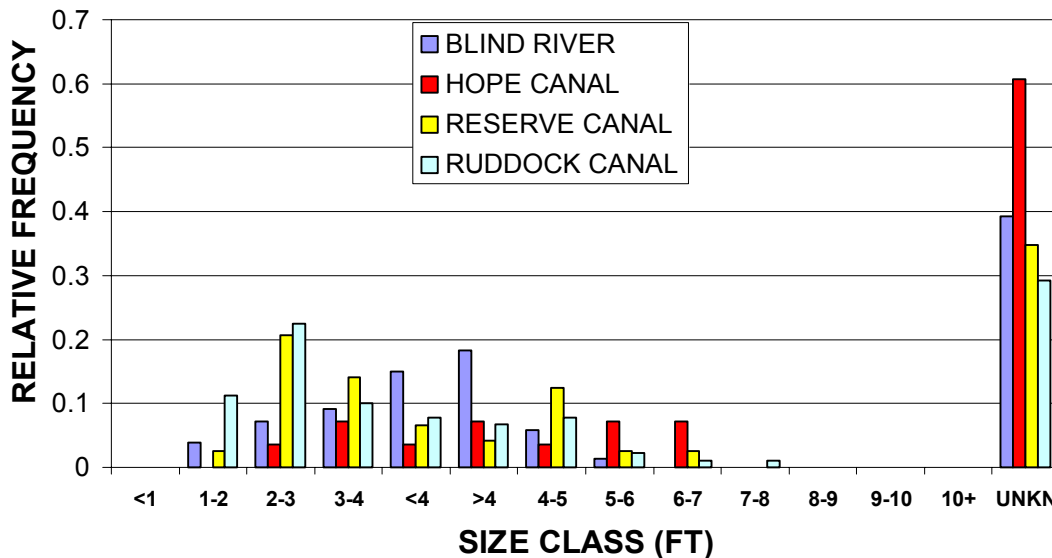


Figure 13. Relative frequency of alligator size classes detected during spotlight surveys in Maurepas Swamp, August 2006.

Alligator abundance estimates from spotlight counts are also affected by season (Chabreck 1966). Alligator nesting behavior in Louisiana peaks in June (Joanen and McNease 1979, Joanen et al. 1984), and nesting females may leave major waterways to nest in more interior habitat (Joanen and McNease 1970). Taylor et al. (1991) estimated that 1.2% of the post-hatch alligator population in a coastal marsh in Louisiana was nesting females, but the yearly proportion of nesting females in swamp habitat is unknown. We began our surveys in late June; therefore, our abundance estimates may be low and biased toward male alligators and non-breeding females, which may also have affected our size class distribution

Size class distributions of alligators estimated from spotlight counts may be affected by differences in vegetation density between habitats, reduced detectability of small alligators that use more interior habitat or densely vegetated areas (Taylor and Neal 1984), and reduced detectability of large, wary alligators (Woodward et al. 1996, personal observations). Alligators off the major waterways were not sampled, and we have no data on the abundance of alligators in the surrounding swamp. However, size class distributions among routes were similar, and alligators between 2-4 feet long were the most commonly detected size classes on all survey routes.

Impacts of Diversion Construction and Operation

Dense populations of alligators were observed in man-made Hope Canal, Reserve-Relief Canal and Ruddock Canal during our surveys, which suggest that the construction of the diversion canal will probably have only temporary effects on alligator abundance. Radio-telemetry studies of mature (Joanen and McNease 1970) and immature (McNease and Joanen 1974) alligators, and recapture of marked alligators (Ruth Elsey, pers. comm.), have documented alligators' ability to disperse long distances under natural conditions. Alligators may leave the construction area, especially if prey populations decline due to construction activities, but will probably return to the area after the disturbance has ended and vegetation is reestablished along the banks of the diversion levee.

Impacts of Diversion Operation

Diversion operation is expected to increase nutrient availability and swamp productivity (Shaffer et al. 2001), and alligator abundance is positively correlated with nitrogen concentration in Florida lakes (Wood et al. 1985). Diversion operation is expected to increase dissolved oxygen levels, fish abundance and species richness (William Kelso, personal communication), which may increase alligator density within the affected area, especially at the south end of Hope Canal. There was a very high density of alligators in the northern portion of Hope Canal, and canal construction may cause these animals to shift their ranges north until construction activities are completed.

Levees along the diversion canal may provide nesting habitat for alligators. In Florida, alligators may actively select levee habitat (Deitz and Heins 1980), but alligators nesting in canal habitat in the Florida Everglades have lower success rates, higher flooding rates, and higher predation rates (Matthew Chopp, unpublished data).

Diversion operation is expected to increase water levels near the outfall area. Increased flooding in this area may jeopardize some alligator nests, because hatching success of alligator eggs is significantly affected by duration and timing of immersion (Joanen et al. 1977). Joanen et al. (1977) found that immersion of six week old eggs for 12 hours killed all embryos, and immersion for 48 hours killed embryos of all ages.

Recommendations

We recommend that the diversion should be operated in June while alligators are nesting to force alligators to build nests above flood level, and to avoid flooding low-lying nests. As part of its alligator management program, the LADWF issues permits for collection of alligator eggs for sale to alligator farms. We recommend that the LADWF be urged to allow increased alligator egg harvest in the diversion canal construction corridor prior to construction activities, and also on a yearly basis in areas that are expected to experience high water levels during diversion operation. This should mitigate for negative impacts of diversion construction and operation on alligator populations in the affected area.

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Bald Eagle

David M. Fox and Philip C. Stouffer

Introduction

Bald Eagles (*Haliaeetus leucocephalus*) prefer to nest away from human disturbances in dominant or co-dominant pine (*Pinus* spp.) or baldcypress (*Taxodium distichum*) trees in forested areas adjacent to large bodies of water with shallow-water areas for foraging (Buehler 2000). In Louisiana in the 1977-1980 breeding seasons, 93% of Bald Eagle nests were in baldcypress trees, of which 68% were live, 18% decadent, and 14% dead (Dugoni 1980). In the same study, the mean height of nest trees was 24.1m (7.9-32.9m), and the mean diameter above the butt-swell was 94.5cm (38.6-174.2cm). In the Southeast, breeding occurs between September and May (Dugoni 1980, USFWS 1984, Buehler 2000). In Louisiana clutches are initiated between October and January (LADWF 2006). In 2006 roughly 88% of Bald Eagle clutches were initiated in November and December (LADWF 2006). Eggs are incubated for 35 days, and young fledge at about three months (USFWS 1984, see Buehler 2000 for literature review). Bald Eagles reach maturity at age five, and pairs generally attempt to hatch one clutch of 1-3 eggs per year, but in southern portions of its range clutches lost early in the breeding season may be replaced (Buehler 2000). Bald Eagles have high nest-site fidelity, and nests are reused for years or decades (Buehler 2000). Nesting territories may contain multiple nests and Bald Eagle pairs may switch nests in succeeding years, especially if a nest fails (Buehler 2000).

Bald Eagles are opportunistic foragers and their food habits vary widely across their range (Buehler 2000), but the majority of food items taken are fish, waterfowl, small mammals and carrion of a variety of vertebrate species (for reviews see Stalmaster 1987, Buehler 2000). Stalmaster (1987) summarized twenty studies of Bald Eagle diet from across its range. Overall, 56% of Bald Eagle prey is fish, 28% birds, 14% mammals and 2% other. In the Southeast, Bald Eagles forage primarily on fish and waterfowl (Dugoni 1980, McEwan et al. 1980). Seventy-nine percent of prey remains in Bald Eagle nests in North-central Florida were fish, mostly catfish (*Ictalurus* spp.), and 17% were birds, mostly American Coot (*Fulica americana*) (McEwan et al. 1980).

Dugoni (1980) examined prey remains in active Bald Eagle nests following the 1978-1979 breeding season in southcentral and southeastern Louisiana. The greatest proportion of remains by percent occurrence was birds (42.4%), of which 49% were American Coot. Among the other bird remains identified were Mottled Duck (*Anas fulvigula*), Blue-winged Teal (*Anas discors*), Common Gallinule (*Gallinula chloropus*), Redhead (*Aythya americana*), and Gadwall (*Anas strepera*). Fish remains were 41.6% of the total, of which 53% were catfish, 10% bowfin (*Amia calva*), 10% striped mullet (*Mugil cephalus*), 10% freshwater drum (*Aplodinotus grunniens*), 5% carp (*Cyprinus carpio*) and 5% largemouth bass (*Micropterus salmoides*). Mammal remains were 15.7% of the total, of which 18% were muskrat (*Ondatra zibethicus*), 14% nutria (*Myocastor coypus*), 4% eastern cottontail (*Sylvilagus floridanus*) and 2% swamp rabbit (*Sylvilagus aquaticus*). In four nests in Louisiana on Avery Island and on Black Bayou in Cameron Parish, Bailey (1919) observed feathers and wings of Northern Pintail (*Anas acuta*) and Mallard (*Anas platyrhynchos*) ducks, remains of catfish and herring (*Alosa* spp.), bird bones including a leg bone of a Great Blue Heron (*Ardea herodias*), and rabbit (*Sylvilagus* spp.) fur. There is an unconfirmed report that a nest in Maurepas Swamp (nest 171, Figure 3) contained several leg-hold traps, suggesting that captured or dead nutria are preyed upon in this area, although live nutria might also be taken. We also have an anecdotal report that small mammal, (probably nutria) bones were common in some Bald Eagle nests near the Davis Pond diversion (Jill Jenkins, USGS personal communication). Catfish appear to be a favored prey species in the Southeast and across its range, even in desert habitat (Haywood and Ohmart 1986), which suggests that shallow water is important foraging habitat (Dugoni 1980).

The population size of Bald Eagles in the lower 48 States before European settlement is unknown, but the population had declined to an estimated 417 nesting pairs by 1963 (Buehler 2000) primarily due to habitat loss, shooting and poisoning, and direct mortality and reduced productivity due to poisoning by environmental contaminants (see Buehler 2000 for a review). Bioaccumulation of DDT by Bald Eagles resulted in eggshell thinning and reduced reproductive success, but eggshell thinning has slowly reversed since the use of DDT was banned in the U.S. in 1972 (USFWS 1984, Buehler

2000). Bald Eagles' habit of consuming carrion may make the birds more vulnerable to poisoning by environmental contaminants.

The combination of Bald Eagles' delayed maturation, low reproductive rate, and relatively small clutch size, contributed to the species' decline (USFWS 1984), but Bald Eagles are long lived and populations are probably influenced more by adult survival rates than factors affecting reproductive success (Buehler 2000). In many areas of the Southeast, populations of Bald Eagle are limited by development of their preferred waterfront habitat (USFWS 1984). Large areas of baldcypress swamp in southeastern Louisiana are unlikely to be developed because they are frequently flooded. However, this second-growth cypress forest has reached maturity in many areas, and is highly attractive to the logging industry. Cypress logging in permanently flooded areas may present a serious threat to Bald Eagle populations in Louisiana, because it is unlikely that baldcypress will regenerate following logging (Chambers et al. 2005).

The Bald Eagle is currently listed as Threatened in the lower 48 States of the United States by the U.S. Fish and Wildlife Service, and Endangered in Louisiana by the Louisiana Department of Wildlife and Fisheries. However, the population of breeding Bald Eagles in the lower 48 States has increased from 417 pairs in 1963 to 7,066 pairs in 2006 (www.fws.gov/migratorybirds/issues/BaldEagle/faqs.htm), and the USFWS proposed to delist the species in 1999. The public comment period on the proposed delisting action was reopened from February-June 2006 and the decision is pending (Federal Register 2006).

Bald Eagle protection and management focus on protecting nesting sites from development and disturbance which might lead to violation of Federal laws protecting Bald Eagles (e.g., the Bald and Golden Eagle Protection Act 1940, the amended Migratory Bird Treaty Act 1972, the Endangered Species Act 1972). Under current Federal management guidelines, tree cutting, logging and development should not take place, at any time, within a primary core area extending to a radius of 750-1500 feet from the nest tree (USFWS 1984). Development and construction within secondary core extending 750 feet to one mile from the primary core should not take place during the nesting season (September-May in the Southeast) (USFWS 1984). Bald Eagles may use alternate nests within their nesting territory that have been unused for several years, and

nests may be taken over by another pair if the resident pair dies (USFWS 1984). For these reasons, nests that have not been used for fewer than five consecutive breeding seasons should be protected as if they are active (USFWS 1984). The exact radii of the management zones depends on the abundance of critical habitat elements in the vicinity of the nest (e.g. nest trees, roost trees), the visibility of the activities from the nest, and the sensitivity of the local population of eagles to disturbance (USFWS 1984, USFWS 2006). LADWF recommends that no activity take place within a 1,500 foot radius of the nest tree at any time, and that no activity take place during the nesting season within a roughly circular, one mile buffer zone around the nest to permit eagles to access feeding areas without being disturbed (Gary Lester, Coordinator Natural Heritage Program, letter to David McQuiddy U.S EPA-Region 6, 14 April 2005). In the event that the Bald Eagle is delisted, the Draft National Bald Eagle Management Guidelines (USFWS 2006) permit most construction activities outside a radius of 660 feet from a nest tree during the non-breeding season. We do not know if LADWF will change the status of the Bald Eagle in Louisiana, or if State management recommendations will change if it is delisted at the Federal level.

The breeding population of Bald Eagles in Louisiana has increased dramatically in the past 22 years. The number of active nests in Louisiana increased from 18 in 1984 (U.S. Fish and Wildlife Service 1984) to 36 occupied breeding territories with 23 successful nests in 1989 (U.S. Fish and Wildlife Service 1989), to 284 active nests in the 2005-2006 breeding season (Louisiana Department of Wildlife and Fisheries 2006). Mean productivity of active Bald Eagle nests in Louisiana over the past five breeding seasons is 1.4 eaglets per nest (LADWF 2006). The largest concentration of active Bald Eagle nests in 2006 was in southeastern Louisiana in Terrebonne (69 nests), St. Martin (31 nests), Assumption (36 nests), St. Charles (29 nests), St. Mary (27 nests), Lafourche (21 nests) and St. John the Baptist (19 nests) Parishes (Figure 14).

Methods

The objectives of this study were to document Bald Eagle recovery in Maurepas Swamp, to determine if the construction corridors of the proposed diversion canal alternatives are within Bald Eagle management buffer zones, and to determine if nesting

Bald Eagles will be negatively impacted by diversion canal construction and operation. The Louisiana Department of Wildlife and Fisheries has conducted annual, state-wide, aerial surveys for nesting Bald Eagles since the 1988-1989 breeding season. LADWF classifies Bald Eagle nests as “active” if there is an adult present at the nest during annual, state-wide, aerial surveys for nesting Bald Eagles. The assumption that nests with an adult in attendance are active, in the sense that there is a pair of adults attempting to raise a brood, probably results in an overestimate of the actual number of active nests, because adult Bald Eagles may be present at nest sites without breeding. However, these data probably provide a reliable index of nesting Bald Eagle populations. In most years LADWF observed active nests twice during the breeding season to determine productivity, but in the 2004-2005 and 2005-2006 breeding seasons a sample of active nests was used to estimate productivity.

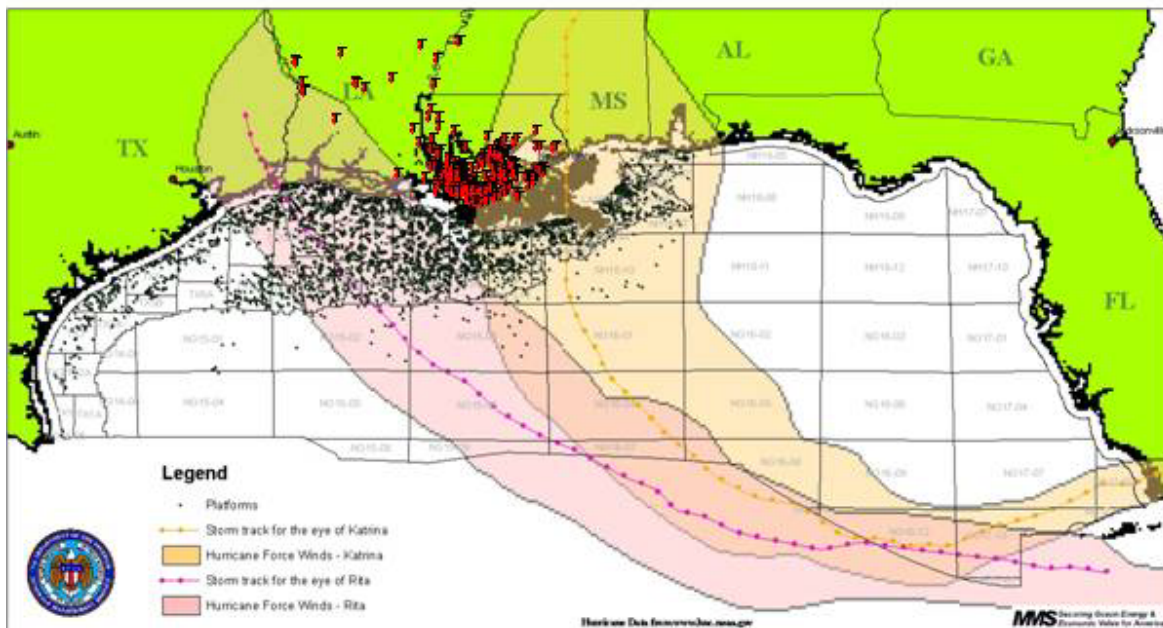


Figure 14. Bald Eagle nest locations 2005-2006 in relation to the tracks of Hurricanes Katrina and Rita (Louisiana Department of Wildlife and Fisheries 2006)

LADWF provided us with nest coordinates and production histories of Bald Eagle nests observed in Maurepas Swamp from 1988-2006 from their annual aerial Bald Eagle nest surveys. Production histories were summarized to determine Bald Eagle population trends in Maurepas Swamp, and areas expected to be affected by diversion canal

construction and operation were examined to determine Bald Eagle nest status and proximity. Anecdotal observations of foraging Bald Eagles were obtained during several hundred field-hours of bird research we conducted in Maurepas Swamp in 2002-2006 in the area expected to be affected by diversion construction around Hope Canal and Reserve-Relief Canal, and in the larger area expected to be affected by diversion operation.

Results

LADWF provided status and production histories for 32 Bald Eagle nests in Maurepas Swamp. Overall, the number of active nests in Maurepas Swamp has increased steadily since 1988 when one active nest was observed, to the 2005-2006 breeding season when there were 17 active Bald Eagle nests, seven nests that had been inactive for fewer than five years, two nests that had been inactive for greater than five years, and six nests that had been destroyed between 1994 and 2006 (Figure 15, Figure 16). There may be some confusion about the status of nest 171 (Figure 16), because it was observed to be inactive from the ground in 2006, and a new, active nest was observed north of nest 171 in 2006 (Gary Shaffer, pers. comm.).

Mean productivity of Bald Eagle nests in Maurepas Swamp has fluctuated during the period in which aerial surveys have been conducted, but over the 18-year survey period, roughly 1.36 eaglets per year were produced per active nest monitored (Figure 17). The overall mean for the last five breeding seasons was 1.16 eaglets per active nest, which is slightly below the range of state-wide averages over the last five breeding seasons (1.4 eaglets per active nest monitored). The difference may be due in part to the small samples of nests for which productivity data was collected in some years in Maurepas Swamp. In 2006 productivity data were obtained for only three nests in Maurepas Swamp.

Two Bald Eagle nests are located within one mile of proposed diversion canal construction corridors. Nest 133 is within one mile of the construction area for the proposed Hope Canal diversion canal (Figure 18). Nest 133 was active, and hatched at least one eaglet every breeding season from 1994-1995 until 2003-2004, but was inactive in the 2004-2005 and 2005-2006 breeding seasons. A new, active nest, designated as

nest 433, was observed approximately one mile southwest of inactive nest 133 in 2006 (Figure 16), which is greater than one mile from the construction area for the Hope Canal alternative. Nest 433 produced no eaglets in 2006. An additional active nest, designated nest 393, is located approximately 400m from the end point of the proposed diversion canal route for the Romeville alternative (Figure 16).

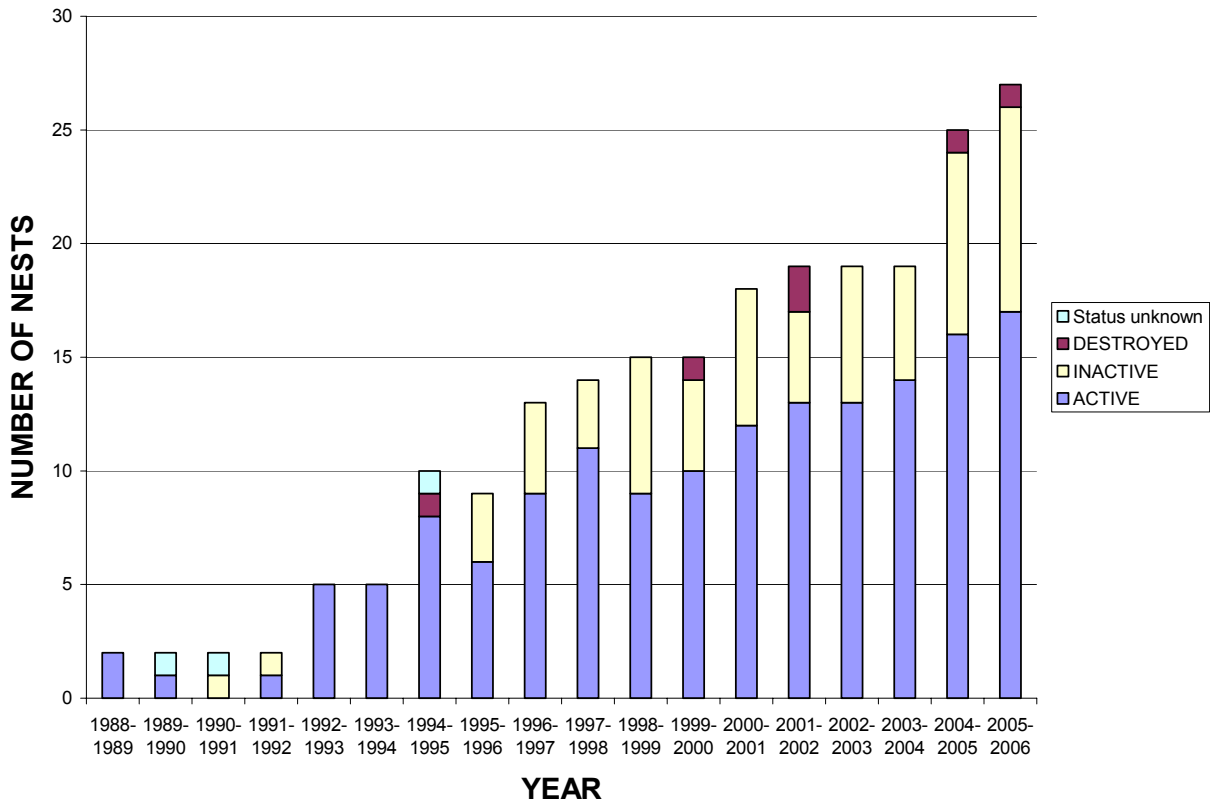


Figure 15. Breeding Bald Eagle population and nest status in Maurepas Swamp 1988-2006. (Source: LADWF Bald Eagle aerial survey data, 2006).

Five Bald Eagle nests that were active in the 2005-2006 breeding season (nests 52C, 171, 147, 88, 434 [Figure 16, Figure 18]), and four nests that have been active within the past five years (nests 170, 170A, 241A, 0399 [Figure 16, Figure 18]) are located in, or within one mile of the area expected to be affected by operation of the preferred alternative diversion route, Hope Canal (Figure 18).

During several hundred of hours of field-work in and around the proposed Hope Canal diversion route in 2002-2006, we have never observed perched or foraging Bald Eagles on or adjacent to Hope Canal. These observations were confirmed by Gary

Shaffer (Southeastern Louisiana University), who has also spent hundreds of hours in the field along this preferred alternative route (Gary Shaffer, personal communication). Several members of the Garyville Hunt Club, which leases the private property surrounding Hope Canal, also have not observed Bald Eagles foraging on Hope Canal. Members of Garyville Hunt Club have observed Bald Eagles flying from Maurepas Swamp toward, and foraging along the Mississippi River. We, and Gary Shaffer, have frequently observed solitary Bald Eagles perched near the mouth of the Blind River in 2002-2006. Gary Shaffer also reported that a solitary Bald Eagle was frequently observed at the mouth of Reserve-Relief Canal in 2000-2006. We observed a solitary, perched adult Bald Eagle at the location where the Mississippi River levee will be breached for the proposed Hope Canal diversion, and solitary, soaring immature eagles near the opposite shore of the Mississippi River in October 2006.

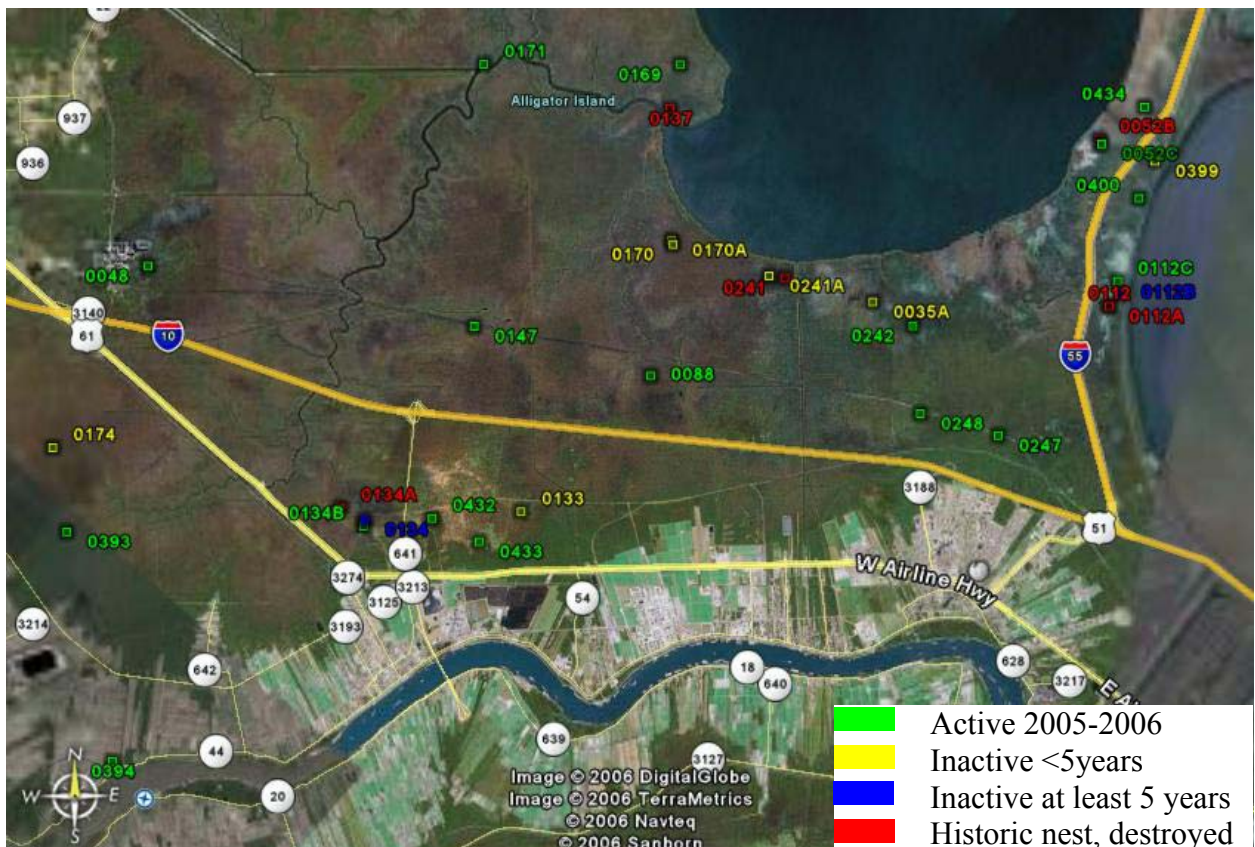


Figure 16. Bald Eagle nest distribution and status in Maurepas Swamp as of February 2006 (Source: LADWF Bald Eagle aerial survey data, 2006).

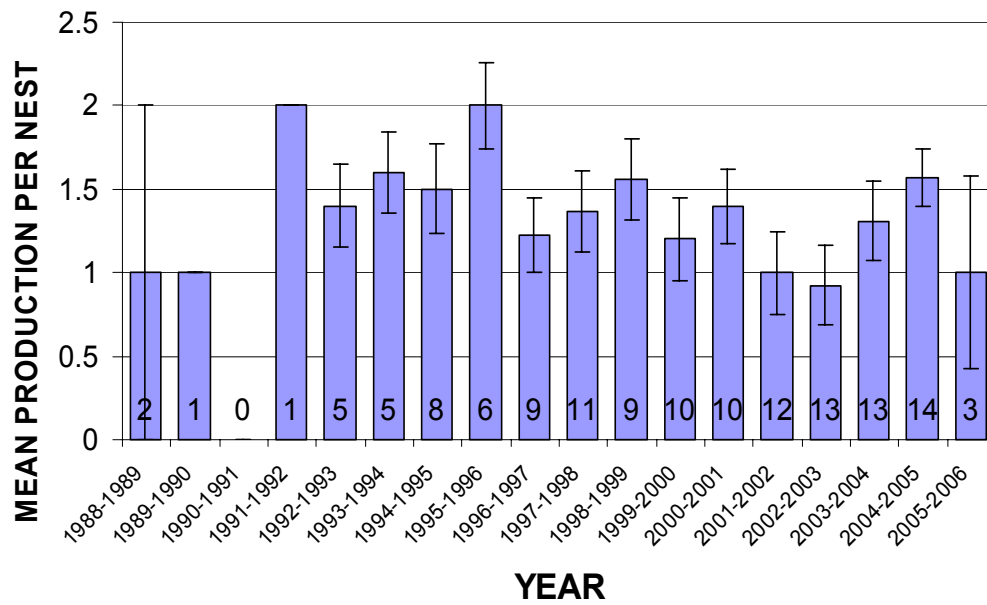


Figure 17. Mean productivity (\pm SE) of active Bald Eagle nests in Maurepas Swamp, 1988-2006. Numbers are the number of active nests for which productivity data were available. (Source: LADWF Bald Eagle survey data)

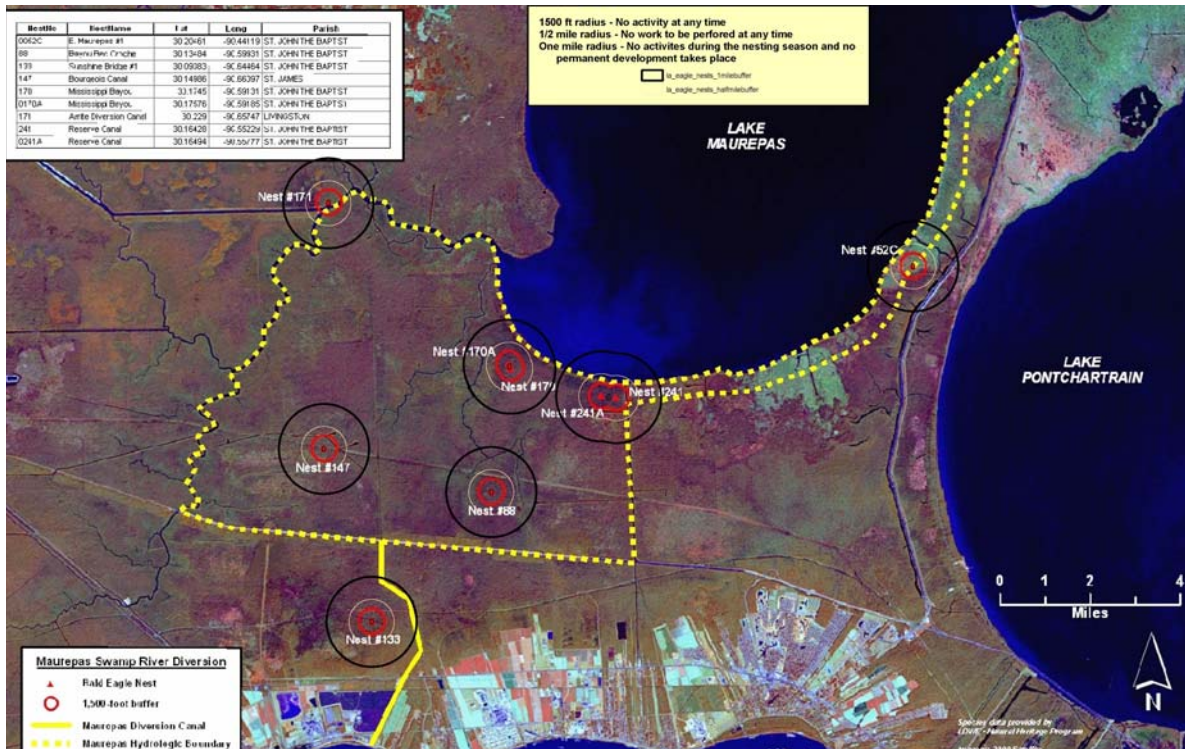


Figure 18. Bald Eagle nests within the area expected to be affected by construction and operation of the Hope Canal diversion alternative (Source: EPA, Region 6). Note that nest 241 in this figure was inactive in every year it was observed (2001-2005), and was destroyed by the 2005-2006 breeding season.

Discussion

The USFWS list of critical Bald Eagle habitat in Maurepas Swamp in 1984 and 1989 included only one inactive nest near Reserve-Relief Canal (U.S. Fish and Wildlife Service 1984, 1989). The population of nesting Bald Eagles in Maurepas Swamp has increased steadily since 1988, and productivity is stable. The 17 active Bald Eagle nests in Maurepas Swamp in 2005-2006 constituted nearly 6% of the active nests in Louisiana in that breeding season. LADWF noted that nests in baldcypress were destroyed less frequently than nests in pines in 2005-2006 despite the fact that many nests in baldcypress swamps in southeastern Louisiana were exposed to hurricane force winds as Hurricane Katrina passed over the area in 2005 (LADWF 2006).

Productivity of active Bald Eagle nests in Maurepas Swamp over the past five breeding seasons was slightly lower than the state-wide average, but stable. Dugoni (1980) estimated that mean productivity in 1977-1980 was 0.94 eaglets per active nest, which suggests that Bald Eagle productivity in Louisiana is increasing with time. We have no information to explain the difference in the five-year average between nests in Maurepas Swamp and nests across the state, but Watts et al. (2006) found that population growth, nesting density and productivity of Bald Eagles was negatively correlated with salinity in Chesapeake Bay, presumably because preferred forage fish species were more abundant in water with lower salinity. Productivity data for many nest attempts on the east side of Lake Maurepas, which is more saline than the west side of the lake, are lacking and we were unable to test the hypothesis that salinity affects population growth and reproduction in this area. Without some additional freshwater input into Lake Maurepas, it is likely that salinity will increase, which may decrease productivity of Bald Eagles.

One Bald Eagle nest, nest 133, is located within one mile of the construction corridor for the preferred Hope Canal diversion route. Nest 133 has been inactive for the past two breeding seasons, but nests that have been inactive for fewer than five consecutive breeding seasons may be used again if the habitat hasn't been degraded during the period of inactivity. Therefore, under the current State and Federal management guidelines, construction activities should not take place within a one mile radius of the nest during the breeding season. In the case of the Federal management

guidelines, the radius of the secondary buffer zone depends on the availability of critical nesting habitat within the buffer, the amount of visual screening of activities from the terrain and forest cover, and on the tolerance of the local eagle population to disturbance (USFWS 2006). We have no information on the tolerance of Bald Eagles in Maurepas Swamp to disturbance, but construction activities may be visible from nest 133 only in the small gap in forest cover where the power line corridor along which nest 133 is located crosses Hope Canal. If the Bald Eagle is delisted, then under proposed Federal Bald Eagle management guidelines the buffer zone will extend only 660 feet from the nest tree, but it is still recommended that construction take place in the non-breeding season.

Under State management guidelines, no construction activity should take place within a one mile radius of the nest tree during the breeding season. However, the State management guidelines suggest that the buffer area be “roughly circular”, and the intent is to limit disturbance along flight paths from the nest tree to foraging areas. In the case of nest 133, the Hope Canal construction corridor lies along the periphery of the one mile buffer zone, and the shape of the buffer might be altered to accommodate construction during the breeding season, especially if the nest is inactive at the time. Furthermore, Bald Eagles have not been observed flying over Hope Canal during our field work, which suggests that the Hope Canal construction corridor is not located along a flight path from the nest tree to a foraging area.

The position of the Bald Eagle at the top of the food chain, and its habit of taking carrion make it vulnerable to bioaccumulation and direct poisoning by a variety of environmental contaminants. Water and sediments in the Mississippi River contain many environmental contaminants, and there is concern that introduction of water from the Mississippi River into Maurepas Swamp may expose Bald Eagles and their forage fish to higher levels of contaminants than they currently experience. A screening-level risk assessment for Bald Eagle’s exposure to Mercury, Nickel and DDT from a reintroduction of Mississippi River water concluded that there is a low magnitude risk to Bald Eagles from exposure to contaminated fish (Battelle 2005). However, Mercury concentrations in fish tissue collected in Lake Maurepas in 2003, and in Lake Pontchartrain in 1997-1999 and 2003-2004, were generally higher than samples collected in the Mississippi River in

1998-1999 and 2001 (Battelle 2005). The authors of the risk-assessment noted that sampling biases and potentially flawed assumptions may have affected their conclusions: the risk-assessment assumed that the maximum concentrations of contaminants detected in water and sediment in the Mississippi River would be present in Maurepas Swamp following diversion operation; many data used in the risk-assessment were several years old, or collected downstream of the Hope Canal diversion site; and water, sediment, and fish tissue samples were not collected in the same areas and/or at the same time (Battelle 2005). Data from a baseline study of contaminants in fish from the Mississippi River and blood from nestling eaglets in the Davis Pond Diversion area, collected by USGS before that diversion was operated at full capacity, were in review at the time that this report was prepared (Jill Jenkins, USGS personal communication).

The long term effects of saltwater intrusion on forest health and nest tree survival may be a serious threat to Bald Eagles nesting in Maurepas Swamp. Some nesting territories on the southern and eastern shores of Lake Maurepas, where salinity and baldcypress mortality are highest, may be completely deforested in less than a decade if current trends continue (Shaffer et al. 2003). At least two of the Bald Eagle nests on the Manchac Landbridge are in standing dead, or nearly dead cypress snags (personal observations). These snags may remain standing for many years, but it is likely that alternative nest trees will be unavailable for breeding Bald Eagles in this area in the near future.

Effects of Diversion Canal Construction on Bald Eagles

Nesting Bald Eagles are most sensitive to disturbance during courtship, laying and incubation (USFWS 2006). Assuming that construction activities take place during the non-breeding season, canal construction on either the proposed Hope Canal or Reserve-Relief Canal diversion routes should have little effect on Bald Eagles. However, nest 393 is located approximately 400m from the outfall area of the proposed diversion canal route near Romeville, Louisiana, which is within the zone in which no construction activities should occur at any time under current State and Federal Bald Eagle management guidelines. The Hope Canal diversion canal construction corridor is situated on the periphery of the one-mile buffer for nest 133, the construction activities may only be

visible from the nest for less than 100-200m along its length, and Bald Eagles do not appear to travel the construction area. Therefore, it might be possible, albeit risky, to allow canal construction activity even if the nest is active, provided that construction takes place during later nesting stages.

The construction corridor width for the Hope Canal diversion route is approximately 100m (Russ Joffrion, Louisiana Department of Natural Resources; personal communication). Some of the healthiest cypress forest is located along Hope Canal (Shaffer et al. 2003), and diversion canal construction may destroy some potential nest trees. However, three Bald Eagle nests are located in trees on the edge of the power line corridor north of I-10 (nests 147, 247, 248; Figure 3). The edge created by the power line may have produced suitable nesting sites for Bald Eagles. We have no information to support this hypothesis, but Bald Eagles usually select dominant trees with sturdy limbs that can support the weight of their nest (Buehler 2000). Trees on newly created edges may produce larger limbs on the open side, and may grow more quickly than trees in the forest interior (Oliver and Larson 1990). Forest clearing for diversion canal construction may produce similar canopy differentiation and provide nesting or perching habitat on the diversion canal.

The most important feature of Bald Eagle foraging habitat is large, open areas for eagles to kill and eat prey (Stalmaster 1987). Bald Eagles cannot become airborne easily in thick forest, and prefer to consume prey in open areas while hunting and eating (Stalmaster 1987). Bald Eagles have not been observed foraging on Hope Canal. This may be because the canopy gap over the canal is only 10-30m wide along most of its length and the canal itself is less than 20m wide, which may not provide enough room for Bald Eagles to perform foraging maneuvers. If this hypothesis is correct, then canal construction may create open foraging habitat along mowed levees where Bald Eagles can access water easily.

Effects of Diversion Operation on Bald Eagles

Diversion canal operation is expected to decrease salinity in Lake Maurepas and Maurepas Swamp, which should have positive effects on Bald Eagles. First, Bald Eagles select large, dominant and co-dominant baldcypress for nesting. Salinity is negatively

correlated with baldcypress productivity and positively correlated with baldcypress mortality. The size and health of the Bald Eagle population in Maurepas Swamp is dependent on the population of suitable nest trees. Suitable snags may persist for decades, but snags will fall or be damaged by hurricanes eventually. Therefore, it is absolutely essential that salinity be reduced to produce replacement nest trees to maintain the Bald Eagle population in Maurepas Swamp. Second, reduced salinity may increase the abundance of preferred, freshwater forage fish, and increase Bald Eagle population growth rates, breeding density and productivity (Watts et al. 2006).

Canal construction and operation may create some foraging habitat for Bald Eagles. Fish that pass through the diversion structure on the Mississippi River and into the diversion canal may be stunned and vulnerable to Bald Eagles. However, high flow-rates on the Colorado River reduced habitat use and foraging effectiveness of Bald Eagles, presumably because deeper water made prey unavailable (Brown et al. 1998). Operation of the proposed Hope Canal diversion at 1500cfs^{-1} is expected to flood the outfall area with up to four feet of water (Bob Jacobsen, personal communication), but most of this area is under dense forest canopy and should not affect foraging Bald Eagles.

Diversion of Mississippi River water into Maurepas Swamp may expose Bald Eagles to elevated levels of a variety of environmental contaminants including several metals, polycyclic aromatic hydrocarbons, pesticides, herbicides and volatile organic compounds. The concentration of many of these contaminants in waters, sediment and fish tissues in Maurepas Swamp and Lake Maurepas relative to concentrations in the Mississippi River is unknown. A screening-level risk assessment for Bald Eagles' suggested that there is a low magnitude risk associated with diverting Mississippi River water into Maurepas Swamp based on maximum concentrations of Mercury, Nickel and DDX in the Mississippi River and Lake Maurepas (Battelle 2005). Mercury concentrations in fish sampled from Lake Maurepas were lower than those in fish from the Mississippi River (Battelle 2005). Therefore, in the case of Mercury exposure, Bald Eagles should not be at greater risk of exposure from a diversion of Mississippi River water into Maurepas Swamp. However, the synergistic effects of multiple contaminants on Bald Eagle health and reproduction are unknown. At least some proportion of the Bald Eagle population has been observed flying out of Maurepas Swamp toward the

Mississippi River, and Bald Eagles have been observed foraging along the Mississippi River near the site where the Mississippi River levee will be breached for the proposed Hope Canal diversion. This suggests that some Bald Eagles in Maurepas Swamp will not experience increased risk of exposure to environmental contaminants from diversion operation.

Recommendations

Several years may pass before diversion canal construction begins. We recommend that LADWF be urged to scrutinize proposed canal construction corridors for new Bald Eagle nests that might be put at risk by canal construction activities. Should the Hope Canal diversion route be selected, we recommend that nest 133 be monitored carefully to determine its status on a yearly basis until construction is complete.

We recommend that canal construction within one mile of active or recently active Bald Eagle nests take place during the non-breeding season. However, we recognize that this may severely restrict construction activities and cause logistical problems and delays because the breeding season of Bald Eagles in Louisiana may last nine months. If nest 133 remains inactive for five consecutive breeding seasons, then restrictions on canal construction on Hope Canal might be relaxed. The radii of State and Federal buffer zones around nest trees may change if the Bald Eagle is delisted at the Federal level. A dialogue with State and Federal biologists should be initiated to determine if the radius of the buffer zone around nest 133 might be relaxed to permit canal construction during late stages of the breeding season if nest 133 becomes active again.

We recommend that large baldcypress trees in the construction corridor be preserved whenever possible to provide potential perching and nesting trees for Bald Eagles. Bald Eagles require areas of shallow water for efficient foraging, and this need is greatest when adults are feeding nestlings. We recommend that, to the extent possible, shallow water areas (e.g. sediment settling ponds, terraced levees) be provided for in the design of the diversion canal. This may provide foraging habitat for Bald Eagles as well as wading birds. Public access to the levee on Reserve-Relief Canal is unrestricted, and large amounts of garbage have been illegally dumped along that canal. Prohibiting public

access to the diversion levees will be controversial with sportsmen, but it will reduce the risk of human disturbance of Bald Eagles and limit illegal dumping of garbage and hazardous waste.

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Colonial Nesting Wading Birds

David M. Fox and Philip C. Stouffer

Introduction

Most of the wading bird species in the orders Ciconiiformes and Pelicaniformes that breed in coastal Louisiana typically nest in colonies. Some of these species were nearly extirpated due to hunting of nesting birds for food or for the millinery trade (U.S. Fish and Wildlife Service 1988). The Lacey Act of 1900, and the Migratory Bird Treaty Act of 1913, contributed to the recovery of many colonial nesting wading birds. Nearly 850,000 birds of 26 species were reported to breed in coastal habitat in Louisiana, Mississippi and Alabama (Portnoy 1977, as cited in Keller et al. 1984). However, recreational, commercial and industrial development of coastal breeding habitat in the United States has caused concern for populations of colonial nesting wading birds (U.S. Fish and Wildlife Service 1988). The purpose of this study was to determine if nesting colonies of wading birds were present in proposed diversion canal construction corridors, and if wading bird colonies would be negatively impacted by diversion canal construction and operation.

Methods

The Louisiana Department of Wildlife and Fisheries conducted aerial surveys of Maurepas Swamp in 1976, 1978, 1983, 1990, 2001, and 2004-2006 for colonies of nesting wading birds. We examined these survey data for nesting colony locations within the corridors of the alternative diversion routes and the larger area expected to be affected by diversion operation on the preferred Hope Canal alternative.

Wading birds were observed foraging on Hope Canal and Reserve-Relief Canal in 2006 during standardized point counts for songbirds (see Breeding Birds and Passage-Migrants sections). Point counts for songbirds are inappropriate for estimating wading bird abundance, but point count detections were used as an index of wading bird activity in the two alternative diversion routes. We also made casual observations of wading bird activity in and around Hope Canal, Reserve-Relief Canal, and the larger area of Maurepas Swamp expected to be impacted by diversion operation in 2002-2006, during several hundred man-hours of bird-research activity. The section of Hope Canal south of Tent Bayou to the pipeline north of I-10 was known to have

nesting wading birds in 2002-2004, and this section of the canal was surveyed specifically for nesting wading birds in May 2006.

Results

Aerial surveys-No active nesting wading bird colonies were detected within areas expected to be directly affected by construction in the alternative diversion routes during aerial surveys in 2005 (Figure 19), and again during aerial surveys conducted in 2006 (Richard DeMay, Barataria-Terrebonne National Estuary Program, pers. comm.). A single active colony, (colony #170, Figure 19), is located on the south shore of Lake Maurepas approximately 2km west of Tobe Canal, which is within the area expected to be affected by operation of a diversion constructed along Hope Canal (Lee Wilson & Associates 2001). In 2005 this rookery contained approximately fifty Great-blue Heron (*Ardea herodias*) nests, 275 Great Egret (*Ardea alba*) nests and 11 Anhinga (*Anhinga anhinga*) nests, and in 2006 it contained one Great Egret nest, two Anhinga nests and 37 empty nests (Richard DeMay, pers. comm.).

Colony 282 is the closest historic colony to any alternative diversion route, approximately 400m west of the preferred alternative diversion route, Hope Canal (Lee Wilson and Associates 2001). Colony 282 was inactive in 2001 and 2004-2006. Colony 282 contained a combined total of >3800 nests of Little Blue Heron (*Egretta caerulea*), Snowy Egret (*Egretta thula*), Tricolored Heron (*Egretta tricolor*), and Cattle Egret (*Bubulcus ibis*) in 1976 and <500 nests of the same species composition in 1978. It was inactive in 1983, but in 1990 it contained <1000 nests of Little Blue Heron, Snowy Egret and Cattle Egret (Michael Green, Texas State University, personal communication).

Field observations-In 2002-2004 a loose colony of 10-15 nesting Yellow-crowned Night-herons (*Nyctanassa violacea*) was observed on the east shore of Hope Canal between the intersection of Hope Canal and Tent Bayou, and the pipeline located north of I-10. We did not observe this colony in 2005. In May 2006, three active and three inactive Yellow-crowned Night-heron nests were found during a survey of this section of Hope Canal for nesting wading birds.

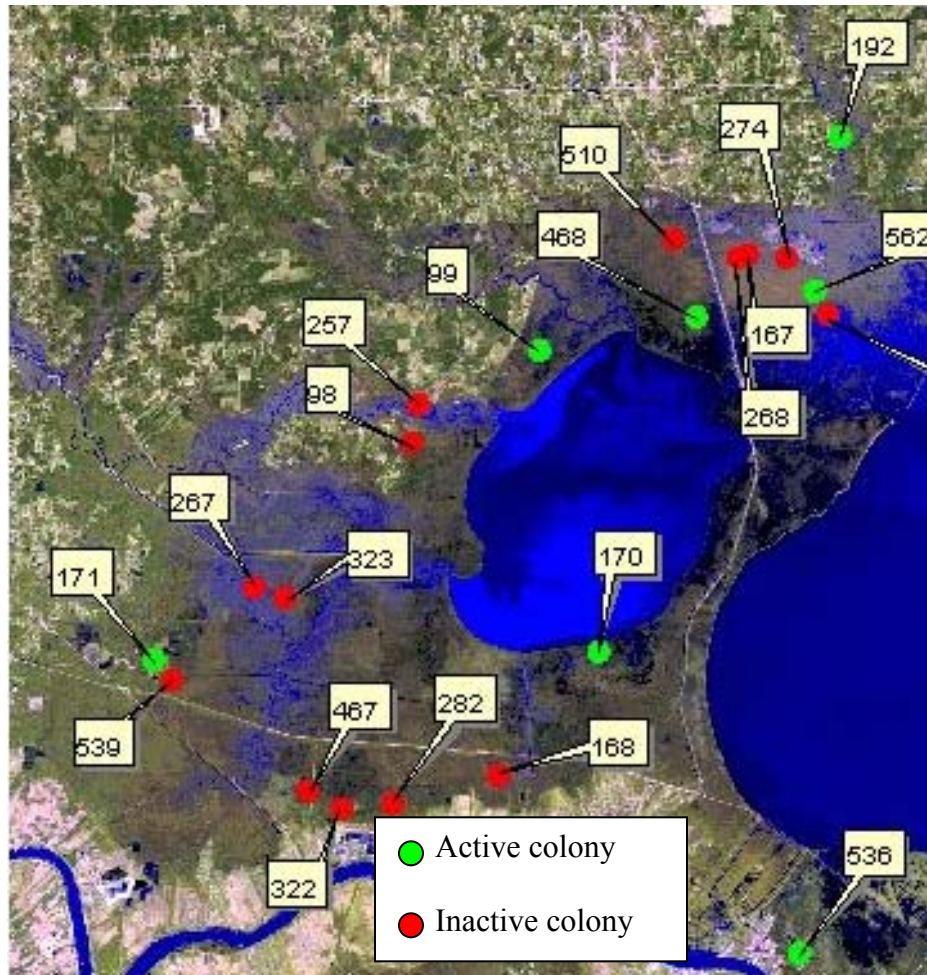


Figure 19. Active and historic colonial nesting wading bird colonies detected during aerial surveys of Maurepas Swamp in 2005 (after Green et al. 2006).

In 2004 a small mixed-species colony of about five Yellow-crowned Night-heron nests and one Great Egret nest was observed on the abandoned railroad grade about 1500m north of the intersection Hope Canal and Tent Bayou. In 2004 a few (<3) solitary Yellow-crowned Night-heron nests and one Great Egret nest were observed in interior forest north of I-10 on the east side of Hope Canal, and in 2006 a solitary Yellow-crowned Night-heron nest was observed on the east side of Hope Canal approximately 3km north of Airline Highway.

Point counts-Relatively few wading birds were detected during point counts, but after correcting for survey effort between canals, nearly twice as many wading birds were detected on Hope Canal than on Reserve-Relief Canal (Table 12). All wading bird species, with the exception of

Great Blue Heron and Great Egret, were detected more frequently on Hope Canal relative to Reserve-Relief Canal (Table 12).

Table 12. Total point count detections of colonial nesting wading birds on Hope Canal and Reserve-Relief Canal, corrected for survey effort, in April-May 2006.

Species	Hope Canal	Reserve-Relief Canal
Anhinga ^a	2.00	0
Black-crowned Night-heron	0.67	0
Great Blue Heron	0.67	3
Great Egret	0.67	3
Green Heron	16.67	7
Little Blue Heron	9.33	0
Snowy Egret	4.67	3
Tri-colored Heron	6.00	2
White Ibis	0.67	0
Yellow-crowned Night-Heron	9.33	8
Grand Total	50.67	26

^aAnhinga is not a wading bird, but nests with other wading birds, and is included in aerial survey counts of colonial nesting wading birds.

Discussion

Populations of common wading birds are generally increasing in Louisiana relative to some other Gulf States (Fleury and Sherry 1995, Sauer et al. 2005), but even before Hurricanes Katrina and Rita in 2005, few large colonies of nesting wading birds were detected in Maurepas Swamp. During aerial surveys in 2005 and 2006, no active colonies were detected in areas that would be impacted by diversion canal construction on any of the alternative routes. One colony in the area expected to be influenced by freshwater from a diversion constructed on Hope Canal was detected on the south shore of Lake Maurepas, but this colony had few nests in 2006 and may be in decline. However, these aerial surveys were conducted by fixed-wing aircraft traveling at relatively high speeds, and may not have detected single nests, small colonies or colonies of dark colored wading birds (e.g. Yellow-crowned Night-heron, Black-crowned Night-heron [*Nycticorax nycticorax*], Green Heron [*Butorides virescens*], Little Blue Heron [*Egretta caerulea*]) (Frederick et al. 1996, Michael Green, unpublished data), or colonies hidden by forest canopy, as demonstrated by their failure to detect the small Yellow-crowned Night-heron colony on Hope Canal in 2004.

Relatively few wading birds were detected during songbird point counts because wading birds usually flushed from their foraging or perching locations as we approached our sampling

points. However, twice as many wading birds were detected during point counts on Hope Canal as compared to Reserve-Relief Canal. This result was probably obtained because Reserve-Relief Canal has a public boat launch at its terminus at Airline Highway, and it is heavily traveled relative to Hope Canal, which is located on private property. Frequent boat passage on Reserve-Relief Canal probably caused many wading birds to forage in less disturbed areas. All species of wading birds except Great Blue Heron and Great Egret were detected more frequently on Hope Canal than on Reserve-Relief Canal. We are unsure of the reason for these results, but possible reasons include: there may have been undetected nests of Great Blue Heron and Great Egret near Reserve-Relief Canal or nests of smaller waders near Hope Canal; individual birds may have become acclimated to boat traffic on Reserve-Relief Canal; the size distribution or species composition of prey species may have favored large waders in Reserve-Relief Canal and smaller wading birds on Hope Canal; and/or more shallow water habitat may have been available for smaller wading birds on Hope Canal.

We detected no large nesting colonies of wading birds in Maurepas Swamp during our field activities in 2002-2006. However, our surveys along Hope Canal and Reserve-Relief Canal were conducted, at most, 100m from the shoreline of major waterways in these areas. Large colonies of wading birds may be detected from great distances by noise from squabbling adults or begging nestlings, or the smell of guano and/or dead nestlings (Michael Green, personal communication). There was no such evidence of large colonies of wading birds near Hope Canal or Reserve-Relief Canal in 2002-2006, but it is doubtful that we would have detected these cues from small nesting wading bird colonies. No large flights of wading birds, other than relatively high-altitude flights of White Ibis (*Eudocimus albus*), were observed near these canals during the breeding seasons in these years. Although we did not document wading bird abundance during our research activities in 2002-2005, we observed more wading birds foraging near Lake Maurepas on Dutch Bayou than we observed on Hope Canal in 2006.

Effects of Diversion Construction on Wading Birds

Diversion construction is not expected to negatively impact any active nesting wading bird colonies on any alternative diversion route, assuming that those areas remain uncolonized until construction activities end. During our songbird surveys in 2006 we observed only one wading bird nest that would be directly impacted by forest clearing for diversion canal

construction. The isolated nests that we detected in interior swamp habitat in 2002-2005 occurred in small, interior forest gaps that may be limited to areas with relatively large logging canals or railroad lines. The lack of large colonies of nesting wading birds in densely forested habitat surrounding the alternative diversion routes suggests that suitable nesting habitat may be scarce in this area, or that better habitat exists outside the areas expected to be impacted by construction. However, nesting wading birds may change the locations of their colonies over time (Bancroft et al. 1988, Michael Green, pers. comm.), and isolated nests and small colonies of wading birds have gone undetected by aerial surveys of Maurepas Swamp. Because nesting colonies have not been observed near the alternative diversion routes, canal and levee construction will probably not directly affect the distribution of nesting colonies. In other words, it is unlikely that forest clearing will destroy valuable nesting habitat for wading birds, because rookeries are not historically known from these areas.

It is possible that undetected nesting wading bird colonies are present in interior locations along the alternative diversion routes within the recommended 1500ft buffer zone for nesting colonies (Russell Watson, Acting Supervisor Louisiana Field Office USFWS, letter to David McQuiddy, 28 May 2002). In Oregon, Great Blue Heron nesting colony abandonment increased and nest density decreased due to logging and road construction within 500m. Direct impacts (e.g. forest clearing, dredging) and indirect impacts (e.g. construction noise, decreased water clarity) of construction activities on nesting wading birds may be avoided by conducting construction activities in the non-breeding season. The breeding season for the suite of nine species of wading birds commonly detected in Hope Canal and Reserve-Relief Canal (Table 1) spans the period from 15 February-1 September (see literature reviews by Butler 1992, Kushlan and Bildstein 1992, Davis 1993, Davis and Kushlan 1994, Rodgers and Smith 1995, Watts 1995, Frederick 1997, Parsons and Master 2000, McCrimmon et al. 2001), but actual dates depend on the weather, species present, and whether they are successful in rearing their first attempted brood.

Diversion canal and levee construction will probably only temporarily affect the distribution of foraging wading birds. Foraging wading birds will probably abandon the construction area during construction activity, but will most likely return after construction and after water clarity improves for sighting prey. Most wading bird species are able to disperse several kilometers from nesting sites to foraging areas (see literature reviews by Butler 1992,

Kushlan and Bildstein 1992, Davis 1993, Davis and Kushlan 1994, Rodgers and Smith 1995, Watts 1995, Frederick 1997, Parsons and Master 2000, McCrimmon et al. 2001), so the temporary displacement of birds from the construction area should not negatively impact foraging wading birds.

Assuming that construction takes place during the non-breeding season, the greatest effect of diversion canal construction may be the alteration of the profile of the cross section of an existing channel. Wading birds forage in shallow water, and a large canal with steep banks will provide only narrow zones of shallow water along the banks of the levees. Design plans for the diversion canal had not been finalized at the time that this report was prepared, so the magnitude of this effect could not be estimated. However, we observed very few wading birds on the east bank of the southern portion of Reserve-Relief Canal which had an extremely steep slope. The cross sectional area of the diversion channel will also determine the velocity at which water flows through the canal, which may affect prey abundance and availability for wading birds (see Expected Effects of Diversion Operation on Wading Birds, below).

Diversion canal construction will create some wetland habitat for foraging wading birds between the Mississippi River the existing channel chosen for the diversion. The amount and quality of habitat created by diversion canal construction will depend on the length of the alternative route selected and the profile of the cross section of the channel.

Effects of Diversion Operation on Wading Birds

Diversion operation is expected to increase primary productivity of the Maurepas Swamp ecosystem (Shaffer et al. 2003), and this should benefit wading bird populations in two ways. First, without some restorative measure, such as the planned diversion, the area of swamp forest within the Pontchartrain Basin is expected to decrease by approximately 50% by 2050 (Barras et al. 2003). Wading birds may establish nesting colonies on isolated tree islands within marsh habitat (e.g. the Florida Everglades), but contiguous forest in Maurepas Swamp may provide a greater number of alternative nesting sites and/or nesting habitat for a more diffuse population of isolated wading bird nests across the area. Second, diversion operation is expected to increase abundance of forage fish and crayfish (*Procambarus* spp.) in the affected area. Nutrient enrichment in some areas of the Florida Everglades has resulted in increased abundance of some wading bird species, presumably due to increased food abundance (Crozier and Gawlick 2002).

Increased food availability should increase the carrying capacity of habitat within Maurepas Swamp for breeding and wintering wading birds.

Diversion operation is expected to raise water levels in the outfall area by as much as several feet. Common wading birds in Maurepas Swamp generally require shallow water for foraging habitat. Although the maximum water depth used for foraging varies depending on the height of the species in question (see literature reviews by Butler 1992, Kushlan and Bildstein 1992, Davis 1993, Davis and Kushlan 1994, Rodgers and Smith 1995, Watts 1995, Frederick 1997, Parsons and Master 2000, McCrimmon et al. 2001), in general, the maximum water depth used by the largest birds, Great Blue heron and Great Egret, is 30cm (Butler 1992, McCrimmon et al. 2001). The deeply flooded area near the diversion outfall will be unavailable for foraging for most wading birds during diversion operation, and the amount of area affected will vary by the height of the species in question. However, most wading birds are able to disperse many kilometers from nesting areas to foraging areas, and the outfall area will be unavailable only during diversion operation. Foraging wading birds are able to respond to changing water levels in areas with artificial water cycles (e.g., crayfish ponds, Fleury and Sherry 1995). Therefore, negative effects of increased water levels in the outfall area on foraging habitat availability should be partially ameliorated by pulsed operation of the diversion. Forage fish species may be concentrated in interior pools as water levels fall after diversion operation has ceased, which may increase food availability for wading birds.

Diversion operation is expected to increase productivity of herbaceous vegetation in interior swamp resulting in a gradual increase in elevation of the affected area due to the accumulation of organic matter. Eventually, this may eventually reduce the amount of area with standing water in Maurepas Swamp for foraging wading birds. In the short term, increased herbaceous vegetation may reduce foraging habitat for wading birds that use open, interior sites. During extensive field work in interior sites in Maurepas Swamp, we only observed White Ibis foraging in interior sites regularly.

Wading bird colonies are frequently situated over water, which provides a security zone against predation (Kushlan and Hafner 2000). Therefore, increased flooding may generate more suitable nesting habitat and/or reduce nest predation rates of nesting wading birds in Maurepas Swamp. If for some reason the outfall area is unsuitable for nesting wading birds during

diversion operation, their ability to shift nesting location should prevent diversion operation from negatively impacting wading bird populations in the area.

The abundance and availability of foraging habitat and forage fish for wading birds in the diversion canal will depend on the velocity of the current in the canal and the turbidity of the water (Allen Rutherford, Louisiana State University, personal communication). The design plans for the diversion canal had not been finalized at the time that this report was prepared, so the magnitude of these effects could not be estimated. However, it is expected that foraging habitat and prey availability will be maximized during times when the diversion is operated at low flow volume.

Chemical contaminants in water diverted into Maurepas Swamp from the Mississippi River are of concern for predators that may bioaccumulate these toxins. A preliminary study of contaminant levels in the Mississippi River and Maurepas Swamp suggests that DDX levels are higher in the swamp than in the river (Battelle 2005). Eggshell thinning and nest success of common wading bird species in Maurepas Swamp do not appear to be significantly affected by DDX (see literature reviews by Butler 1992, Kushlan and Bildstein 1992, Davis 1993, Davis and Kushlan 1994, Rodgers and Smith 1995, Watts 1995, Frederick 1997, Parsons and Master 2000, McCrimmon et al. 2001). This may be due to the fact that most of these species forage upon smaller fish than top avian predators (e.g., Bald Eagle, Osprey [*Pandion haliaetus*]). Great Blue Herons prey upon larger, predatory fish (e.g. largemouth bass [*Micropterus salmoides*]) more frequently than other wading birds, but DDX may not affect Blue Heron nest success (see literature review by Butler 1992). Great Blue Heron populations in the area affected by diversion operation should not be negatively affected because DDX concentrations appear to be lower in the Mississippi River than in Maurepas Swamp.

Recommendations

Wading bird abundance and species richness was greater on Hope Canal than on Reserve-Relief Canal, but we recommend that the preferred alternative route, Hope Canal, be selected for diversion construction, because it is the alternative expected to benefit the greatest area of swamp (Lee Wilson and Associates 2001).

Diversion canal construction is not expected to begin until after hydrological modeling and diversion design is complete, which may be several years after the preparation of this report.

Because the distribution of wading bird nesting colonies is likely to change in the interim, and because aerial surveys may miss nesting colonies under the forest canopy, the recommended 1500' buffer zone for wading bird nesting colonies along the selected diversion route should be surveyed on foot to be certain that no colonies are present. However, if diversion construction occurs during the non-breeding season (September-January), then construction should not negatively affect wading bird populations in the area. If the Hope Canal alternative is selected, then we recommend that colony #282 (Figure 1) be closely monitored to determine its status on a yearly basis, and to determine the duration of the breeding season in this area, which may expand the window of opportunity for construction activities.

Diversion operation is expected to flood the outfall area by as much as 4 feet (Bob Jacobsen, personal communication, which will make this area unsuitable for wading bird foraging during diversion operation. We recommend that the diversion be pulsed as frequently as possible during the wading bird breeding season, at least when hydrological constraints, restoration goals and other wildlife considerations coincide, to allow wading birds to forage in the outfall area as frequently as possible.

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Breeding Forest-Songbirds

David M. Fox and Philip C. Stouffer

Introduction

We conducted several studies of the breeding songbird community within Maurepas Swamp in 2002-2005 to describe songbird community composition with respect to habitat type, and the functional role of birds in the swamp ecosystem. These studies described the effects of forest degradation in Maurepas Swamp on the songbird community, and the effects of songbirds on caterpillar herbivory in the swamp. Data collected on proposed diversion canal routes in 2006 were analyzed to determine if effects of diversion canal construction and operation on breeding forest-songbirds might differ between the proposed routes.

Between 1883 and 1991, as much as 77% of southern bottomland hardwood forest was lost, primarily due to logging and levee construction along major rivers (The Nature Conservancy, as cited in Mitsch and Gosselink 2000). The Lower MAV currently contains 2 million ha of forested wetland (Twedt and Loesch 1999), which is the largest area of forested wetland remaining in the United States (Mitsch and Gosselink 2000). Greater than 50% of the remaining forested wetland in the Lower MAV is located in Louisiana (Twedt and Loesch 1999), but much of the cypress-tupelo swamp forest in coastal Louisiana is at risk of converting forested wetland into open marsh (Chambers et al. 2005). It is estimated that 42% of the remaining swamp in the Louisiana Deltaic Plain will be lost by 2050 (Louisiana Coastal Wetlands Conservation and Restoration Task Force 1998). Loss of coastal wetland forests had not been addressed specifically until recently, most likely because the transition of swamp to open marsh results in little wetland loss per se (Conner and Day 1988, Chambers et al. 2005).

Southern bottomland hardwood forests support a variety of birds during the wintering and breeding seasons (Kennedy 1977, Dickson 1978), and also during migration (Gauthreaux 1992, Barrow et al. 2005). Many bird species associated with bottomland forests in the Mississippi River Alluvial Valley (MAV) have experienced long-term population declines (James et al. 1992, Twedt et al. 1999, Sauer et al. 2003). Analyses of Breeding Bird Survey data, from 1966–2002, identified 37 bird species that

utilize the MAV that had declining population trends (Sauer et al. 2003). In 2006, the Neotropical Migratory Bird Conservation Program, also known as “Partners in Flight” listed 29 bird species that breed in the MAV as being of regional conservation concern (www.rmbo.org/pif/scores/scores.html), nine of which may require immediate management action to maintain their current populations.

Seventy species of birds are known to breed regularly in bottomland hardwood forests and 48%-65% of individuals within site-specific breeding bird communities in BLH may be Neotropical migrants (Pashley and Barrow 1990). Swamps may have lower bird species richness than bottomland hardwood forest (Kennedy 1977, Harris and Vickers 1984, Wakeley and Roberts 1996), because Neotropical migratory bird abundance and richness may be lower in cypress-tupelo swamp forest than in bottomland hardwood forest (Kennedy 1977), and/or because ground-nesting bird species may be absent from swamp bird communities because there is little dry nesting or foraging habitat. However, Louisiana’s coastal cypress-tupelo swamps support dense populations of some Neotropical migratory birds of high conservation concern. Prothonotary Warbler (PROW; see Appendix A for scientific names and four-letter codes for birds), Northern Parula (NOPA) and Yellow-throated Warbler (YTWA) are common, breeding Neotropical migratory forest-songbirds in Louisiana swamp habitat (Kennedy 1977, Sallabanks et al. 2000, Zoller 2004, Fox 2006). PROW (Figure 20) is among the birds of greatest conservation concern in the MAV, where 25% of the global population of PROW breeds (www.rmbo.org/pif/scores/scores.html). In 2006 Partners in Flight recommended immediate management action to maintain populations of PROW in the MAV, and also listed Northern Parula (NOPA) as a species of high management concern (www.rmbo.org/pif/scores/scores.html).

Maurepas Swamp is currently one of the largest contiguous tracts of wetland forest remaining in the MAV. Like most coastal forests in Louisiana, Maurepas Swamp was almost completely deforested by logging operations between 1900 and 1930 (Mancil 1980, A. Dranguet, personal communication). Much of the cypress-tupelo forest in Maurepas Swamp regenerated following logging operations, but forest regeneration and productivity of baldcypress (*Taxodium distichum*) and tupelo (*Nyssa* spp.) trees in Maurepas Swamp are inhibited by increased flooding depth and duration, salt water

intrusion, nutria damage to small trees, and defoliation by caterpillars (Shaffer et al. 2003). These stressors have generated a mosaic of habitats in Maurepas Swamp, ranging from natural and anthropogenic marsh to closed-canopy cypress-tupelo swamp and bottomland hardwood forest (Figure 21). Salinity stress and forest loss has been greatest on the east and south sides of Lake Maurepas (Shaffer et al. 2003; Figure 21, Figure 26).



Figure 20. Prothonotary Warbler at nest cavity during nestling provisioning trip.

During our studies of the breeding songbird community in Maurepas Swamp, we found that this swamp supports large populations of some migratory and resident songbirds during the breeding season, including extremely dense populations of PROW and NOPA. PROW and NOPA were the most abundant bird species in relatively healthy swamp that receives freshwater run-off from canals (hereafter ‘throughput’ swamp), and sites beginning to transition to marsh habitat (Zoller 2004, Fox 2006), hereafter referred to as ‘relict’ swamp due to the fact that these sites may not regenerate into forest if they are logged or if trees die for other reasons (Chambers et al. 2005). YTWA was the third most abundant breeding Neotropical migratory bird species in swamp forest in Maurepas Swamp (Zoller 2004, Fox 2006). The Audubon Society is likely to list Maurepas Swamp as an Important Bird Area at the global scale because of its large populations of PROW

and NOPA, which are estimated to be approximately 240,000 pairs each (M. Driscoll, Louisiana Important Bird Area Coordinator, personal communication).

Höppner (2002) and Shaffer et al. (2003) described the vegetative characteristics of the wetland habitats in Maurepas Swamp. Shaffer et al. (2003) used vegetative characteristics, hydrological regime, and primary productivity levels to broadly group swamp habitats into three categories, which we refer to as throughput swamp; relict swamp; and degraded sites.

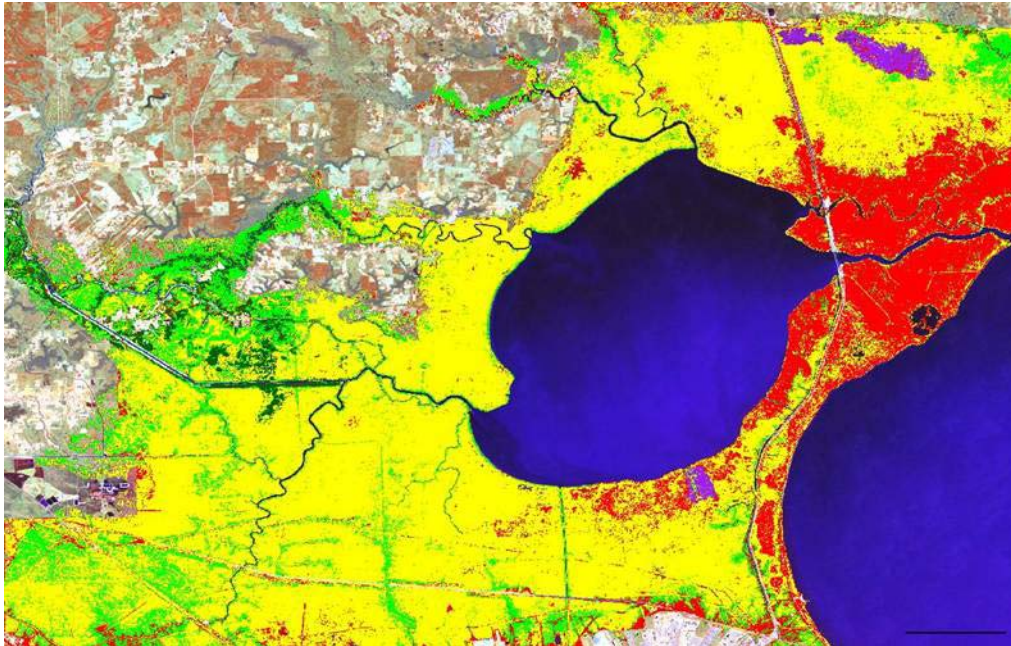


Figure 21. Habitat classification of wetlands in Maurepas Swamp. Red=degraded, Yellow=relict swamp, Light green=throughput swamp, Dark green=bottomland hardwood forest or pine forest (Jason Zoller and Gary Shaffer, unpublished).

Throughput swamp habitat (Figure 22, Figure 23) has a closed canopy dominated by baldcypress and water tupelo (*Nyssa aquatica*), little herbaceous vegetation, and a sparse mid-story canopy (Höppner 2002, Shaffer et al. 2003), although historic photographs of logging operations in Maurepas Swamp show a dense understory layer of saplings (DMF, personal observations). Relict swamp (Figure 24, Figure 25) has an open canopy with widely spaced, dominant baldcypress, and intermediate and co-dominant baldcypress, water tupelo and swamp tupelo (*Nyssa sylvatica* var. *biflora*). Most large trees are concentrated along ditches created during logging operations in the early 1900's. The mid-story of relict swamp is dominated by swamp red-maple (*Acer rubrum* var. *drummondii*), pumpkin ash (*Fraxinus tomentosa*) and green ash (*Fraxinus*

pennsylvanica), and waxmyrtle (*Myrica cerifera*). The herbaceous layer is dense, continuous, and dominated by arrow arum (*Peltandra virginica*). Most tupelo in relict swamp exhibit crown dieback to some degree, and their boles are frequently broken 3-10m above the ground (Shaffer et al. 2003, personal observations).



Figure 22. Aerial view of throughput forest near Hope Canal in February 2006. The southwestern shore of Lake Maurepas is visible in the upper right of the photograph.



Figure 23. Throughput forest site adjacent to Hope Canal. Note closed canopy and sparse understory. Photograph by Jason Zoller.

Degraded sites (Figure 26, Figure 27) are in close proximity to Lake Maurepas, which allows water exchange and salt-water intrusion (Höppner 2002, Shaffer et al. 2003). Most degraded sites were formerly forested, but are now characterized by a high abundance of herbaceous marsh vegetation and low abundance of woody vegetation. Most living trees are concentrated along levees or in small cypress domes.



Figure 24. Aerial view of relict swamp near Dutch Bayou in February 2006. Note reduced basal area relative to throughput forest.



Figure 25. Ground view of relict swamp in February. Note abundant shrubs, emerging herbaceous vegetation, broken topped tupelo at far left and lower right, with Prothonotary Warbler nest cavity visible at lower right.



Figure 26. Aerial view of degraded area on the southern shore of Lake Maurepas in February 2006.



Figure 27. Ground view of a degraded site near Reserve-Relief Canal in winter. Note sparse canopy and shrub cover, and abundant herbaceous vegetation. Photo by Jason Zoller.

Most of Maurepas Swamp is continuously flooded or the soils are saturated with water, and forested areas are dominated by baldcypress and tupelo. However, there is a small area of bottomland hardwood forest at the southern periphery of Maurepas Swamp near Airline Highway, and areas with a mixture of throughput swamp and bottomland hardwood forest where bottomland hardwood forest on artificial levees extends north into the swamp in narrow (approximately 30m wide) strips along Hope Canal and Reserve-Relief Canal. There have been no studies of the vegetative characteristics of this bottomland hardwood forest, but it has a closed canopy of large trees of a mixture of species, dominated by oaks (*Quercus* spp.). Black willows (*Salix nigra*) were common on shorelines of canals. Understory trees include swamp red-maple, ash (*Fraxinus* spp.) mulberry (*Morus* spp.), and persimmon (*Diospyros virginiana*) on Hope Canal (DMF, personal observations). Understory vegetation on levees includes dense thickets of *Rubus* spp., and the understory is generally thicker and more contiguous on Hope Canal because the crest of the eastern levee of Reserve-Relief Canal, south of Interstate 10, is disturbed by a well-traveled dirt road along much of its length.

Zoller (2004) found that breeding forest-bird (i.e. small-medium sized birds with moderate-sized home ranges that utilize forested habitat, including only representatives from the Cuculidae, Picidae, and Passeriformes) species-richness and abundance were significantly lower at sites which were transitioning to open marsh relative to relict and throughput sites, which had similar forest songbird species-richness and abundance. Zoller (2004) also reported that mean migrant bird species-richness was nearly four times higher in relict and throughput swamp relative to degraded sites. Zoller (2004) found that species richness and abundance of forest birds in Maurepas Swamp was highly correlated with foliage structural complexity, and he concluded that high forest-bird abundance in relict swamp may have been due to its high horizontal habitat complexity in the lowest canopy layer, which compensated for its reduced vertical canopy complexity relative to throughput swamp.

Methods

Breeding forest songbird-species richness and abundance were measured by performing 10-minute point counts between April-June following slightly modified

versions of the Lower Mississippi Valley Joint Venture Program Protocol for Monitoring Forest Interior Bird Populations (Lower Mississippi Valley Joint Venture 2004) during our studies of the breeding bird community in throughput swamp, relict swamp and degraded sites in Maurepas Swamp in 2002-2005. Sampling points included in analyses for this report were >200m apart, and most point count locations in swamp were <300m from shorelines of major waterways (Figure 9) due to the difficulty of walking through these habitats. Point counts were done only on days with low wind speeds and light, or no precipitation. Point count sampling is usually begun when it is light enough to see 200m, and sampling ends before 10am, but because it was necessary for us to boat to sampling locations, point counts were generally begun well after sunrise and some ended after 10am. Point counts used in our analyses for this report included only those counts which were completed by 10:05am. All birds seen and heard during the 10-minute sampling period were recorded, and the distance from the observer to the location of the bird was estimated at the time it was first detected for estimating bird density with program DISTANCE (Thomas et al. 2005). Movements of birds were noted to avoid double-counting of individuals. Birds flying over point count locations were recorded, but not included in analyses. Zoller (2004) performed fixed-radius point counts in which only detections within 50m were recorded, while in our other studies in 2003-2005 we performed unlimited-radius point counts in which detections of all birds, from all radii, were recorded. To increase sample size, data from all studies were truncated to include only detections within 50m of the observer. Data were pooled to test effects of habitat type on relative abundances of forest songbirds as a group and the three most common, breeding Neotropical migratory birds, PROW, NOPA and YTWA. Relative abundance of birds in this report is expressed as the mean number of birds detected per point count within 50m of the observer.

To address specific questions of potential impacts of diversion canal construction on breeding forest-songbirds on two of the proposed diversion routes, Hope Canal and Reserve-Relief Canal, we conducted additional point count surveys of breeding birds from 9 April-24 May 2006 on these canals. Sampling points were accessed with canoes and kayaks, and point counts were performed on the water. Sampling points were located 500m apart, starting 500m north of Airline Highway on Hope Canal and approximately

750m north of Airline Highway on Reserve-Relief Canal. Six sampling points were surveyed on Reserve-Relief Canal and nine points were surveyed on Hope Canal. Vegetative characteristics of habitat at sampling points were evaluated subjectively during counts, and observations were confirmed with aerial photographs. Sampling points were broadly classified as bottomland hardwood forest (hereafter BLH), throughput swamp, or 'mixed' habitat in areas where narrow strips of BLH on levees were embedded in a matrix of throughput swamp. Data were truncated to detections within 50m for inclusion with and comparison to data collected in 2002-2005.

Degraded, relict swamp and throughput swamp were sampled in 2002-2004 in April-June. Only relict swamp habitat was sampled in 2005, and only in the early part of the breeding season, in April and May, when birds sing more frequently. Therefore, relative abundance and species richness data from 2005 were not included in analyses, but were included in figures displaying test results for 2002-2004 for visual comparison of means. Surveys on Hope Canal were also conducted in April and May, and included point counts in throughput swamp, mixed and BLH habitat, but there was no throughput habitat within the survey route on Reserve-Relief Canal. Survey periods and habitats sampled were not similar between the 2002-2004 and 2006 data sets. Therefore, these data were analyzed separately.

All statistical analyses were performed with SAS (SAS Institute 2002). Unless otherwise stated, point count data were log-transformed to closer approximate assumptions of normality for parametric statistics, and significance levels were set at 0.05.

To test for significant differences in forest-songbird species richness in 2002-2004 among habitats (throughput swamp, relict swamp and degraded sites), and year*habitat levels, we performed a one-way ANOVA with a randomized block design, blocked on year, using PROC MIXED. Neotropical migrant bird species-richness was tested in a similar manner, with a protected alpha level of 0.025. To test for significant differences in species richness in 2006 between the two alternative diversion routes sampled, Hope Canal and Reserve Canal, and among habitats (BLH, mixed forest and throughput swamp) and habitat*canal levels, we performed a two-way ANOVA with a 2x3 factorial

design. Neotropical migrant bird species-richness on canals was tested in a similar manner, with a protected alpha level of 0.025.

To test for significant differences in total forest-songbird relative abundance in 2002-2004 among habitat types we performed a one-way ANOVA with a randomized block design, blocked on year, using PROC MIXED. Neotropical migrant bird abundance was tested in a similar manner, with a protected alpha level of 0.025. To test for significant differences in relative abundance of PROW, NOPA, and YTWA among habitats in 2002-2004, we performed a subsequent MANOVA with a randomized block design, blocked on year, using PROC GLM, with a protected alpha level of 0.0125. To test for significant differences in total forest bird relative abundance between alternative diversion canals and among habitats and habitat*canal levels in 2006, we performed a two-way ANOVA with a 2x3 factorial design using PROC MIXED. Neotropical migrant bird abundance on canals was tested in a similar manner, with a protected alpha level of 0.025. To test for significant differences in relative abundance of PROW, NOPA, and YTWA between canals and among habitats and canal*habitat levels, we performed a subsequent MANOVA with a 2x3 factorial design, using PROC GLM, with a protected alpha level of 0.025.

Non-significant variables with F-values <1.7 were removed with a backwards selection process, pooled with the error term, and reduced models were tested. Multiple pair wise comparisons were performed to determine significant differences among significant main effects and interactions using least-squared means with Tukey-Kramer adjustments. Letter groupings for least significant means were generated with Saxton's macro 'pdmix800.sas' (Saxton 1998) in PROC MIXED. Although Red-winged Blackbirds were frequently detected during point counts in relict swamp and on canal edges in throughput swamp, and were observed breeding in herbaceous vegetation and shrubs in relict habitat, they are more typical of a marsh bird community, and large flocks of Red-winged Blackbirds created significant outliers in point count datasets. Therefore, we removed Red-winged Blackbirds from analyses of habitat effects on relative abundance of forest songbirds.

Density estimates were generated for PROW, NOPA and YTWA among the different habitats sampled in Maurepas Swamp with program DISTANCE (Thomas et al.

2005). Point count data were pooled across years and data included in analyses were restricted to counts completed by 10AM. Only detections of singing males, or detections without sex assigned and assumed to be detections of singing males, were included in the final data set for DISTANCE analyses. DISTANCE analyses were performed by Jennifer Norris (Louisiana State University, School of Renewable Natural Resources).

Results

A total of 117 bird species, of 14 orders and 38 families, were observed in Maurepas Swamp during all research activities (Appendix A). A total of 576 individual point counts were conducted before 10:05AM at 116 unique point count locations (Figure 28, Appendix B), during which a total of 6728 point count detections of forest songbirds were recorded (Table 13). Overall mean relative abundances indicated that the bird community at degraded sites was dominated by Red-winged Blackbirds, and PROW and NOPA were the most abundant birds in forested habitats (Table 14, Appendix A). PROW was the most abundant bird in relict swamp, and NOPA was the most abundant bird in Throughput, BLH and Mixed habitats (Table 14).



Figure 28. Point count locations within Maurepas Swamp. Note that all points were not surveyed in all years (see Table 1, Appendix 1). Some points or labels in intensively surveyed areas are not displayed for figure clarity.

A total of 29 species of forest songbirds were detected across years (Table 13). Forest-songbirds species richness in 2002-2004 was significantly affected by habitat type ($F_{2,396}=166.97$, $p<0.0001$) and the interaction between habitat type and year ($F_{4,396}=2.40$, $p<0.0499$). Forest-songbird species richness was not significantly different between throughput and relict swamp habitat, but was significantly lower at degraded sites (Figure 29). Neotropical migrant bird species-richness was significantly affected by habitat type ($F_{2,396}=210.84$, $p<0.0001$). Neotropical migrant bird species-richness was highest in throughput swamp, intermediate in relict swamp and lowest at degraded sites (Figure 30).

Table 13 Summary of point count survey effort by year and habitat type in Maurepas Swamp.

	Year	BLH	Mixed	Throughput	Relict	Degraded	Total ^a
Number of points surveyed	2002	0	0	9	13	13	35
	2003	0	0	9	37	13	59
	2004	0	0	15	11	15	41
	2005	0	0	0	12	0	12
	2006	6	5	4	0	0	15
	Total^a	6	5	28	49	28	116
Number of point counts completed before 10:05AM	2002	n/a	n/a	27	39	39	105
	2003	n/a	n/a	27	104	39	170
	2004	n/a	n/a	52	35	40	127
	2005	n/a	n/a	n/a	34	n/a	34
	2006	61	52	27	n/a	n/a	140
	Total	61	52	133	212	118	576
Total number of forest songbird species observed within 50m during point counts	2002	n/a	n/a	16	22	20	25
	2003	n/a	n/a	21	26	16	27
	2004	n/a	n/a	23	20	13	24
	2005	n/a	n/a	n/a	18	n/a	18
	2006	22	22	18	n/a	n/a	22
	Total^a	22	22	23	27	22	29
Total number of forest songbird detections recorded within 50m of survey points	2002	n/a	n/a	357	489	454	1300
	2003	n/a	n/a	305	1116	333	1754
	2004	n/a	n/a	778	451	476	1705
	2005	n/a	n/a	n/a	318	n/a	318
	2006	599	670	382	n/a	n/a	1651
	Total	599	670	1822	2374	1263	6728

^a Totals reflect the number of unique points surveyed, or species detected, across habitats and/or years.

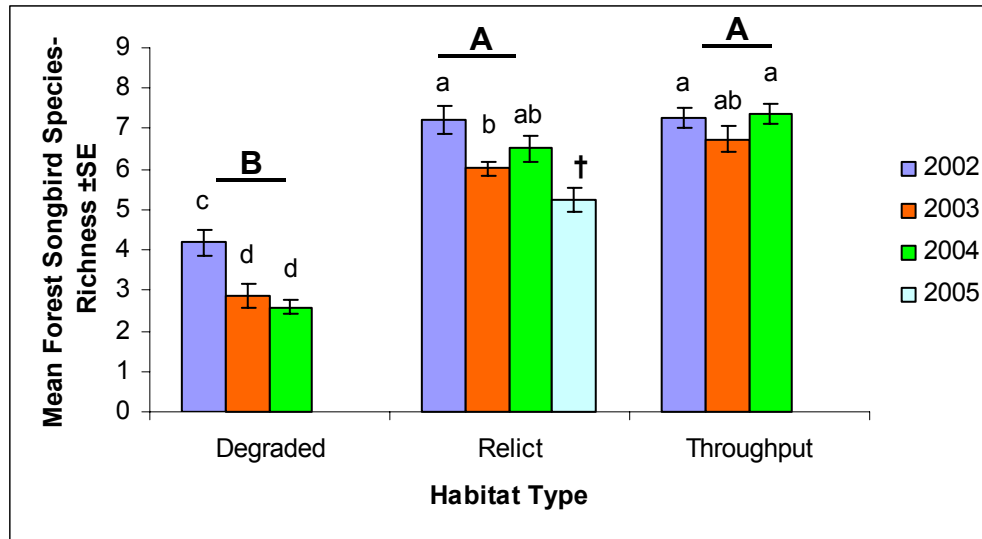


Figure 29. Mean forest-songbird species richness per count in 2002-2004. Capital letters indicate significant differences among habitats. Lower case letters indicate significant differences in habitat*year levels. † Only relict habitat was sampled in 2005; data displayed for visual comparison.

Table 14. Overall relative abundance of the five most frequently detected songbirds in each habitat sampled in Maurepas Swamp in 2002-2006. See Appendix A for four-letter alpha codes.

Degraded		Relict		Throughput		Mixed		BLH	
Species Code	Relative Abundance	Species Code	Relative Abundance	Species Code	Relative Abundance	Species Code	Relative Abundance	Species Code	Relative Abundance
RWBL	6.50	PROW	3.40	NOPA	2.81	NOPA	3.08	NOPA	1.78
NOPA	0.51	NOPA	2.51	PROW	2.68	PROW	2.31	PROW	1.65
NOCA	0.48	CACH	1.23	CACH	1.66	NOCA	1.43	CACH	1.30
COGR	0.41	CARW	1.02	RBWO	1.02	CACH	1.09	NOCA	1.07
YTWA	0.40	RBWO	0.74	YTWA	0.99	BGGN	0.88	BGGN	0.83

Mean relative abundance of total forest songbirds in 2002-2004 was significantly affected by habitat type ($F_{2,395}=300.70$, $p<0.0001$) and the interaction of habitat type and year ($F_{4,395}=2.61$, $p<0.0355$). Relative abundance of total forest songbirds was significantly lower at degraded sites relative to throughput and relict swamp (Figure 31), and lowest at degraded sites in 2003 and 2004.

Mean relative abundance of breeding Neotropical migrant songbirds was significantly affected by habitat type ($F_{2,395}=205.05$, $p<0.0001$) and the interaction of habitat type and year ($F_{4,395}=4.80$, $p<0.0009$). Mean relative abundance of Neotropical migrant songbirds was highest in throughput sites, intermediate in relict sites, and lowest at degraded sites, and mean relative abundance generally declined in degraded and throughput sites, and increased in relict sites (Figure 32).

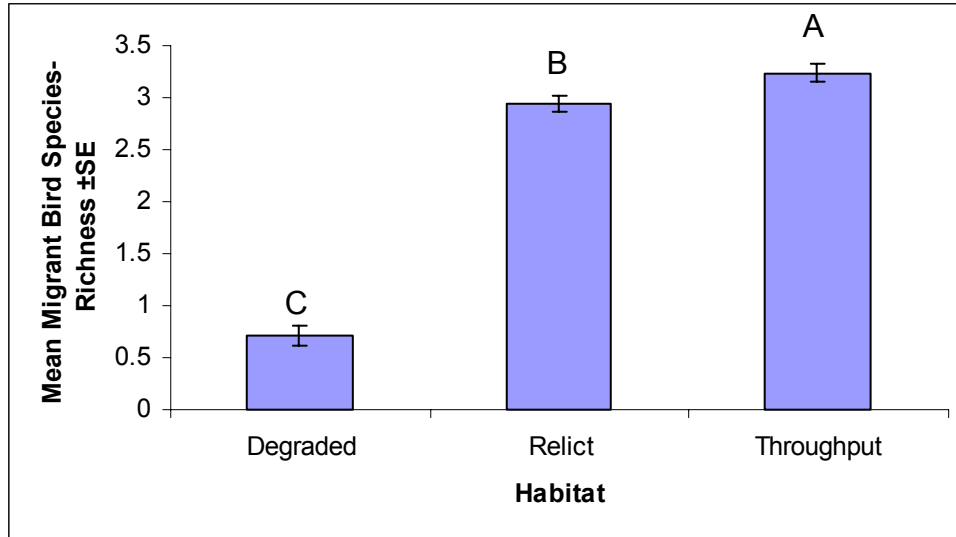


Figure 30. Mean Neotropical migrant bird species-richness in habitats sampled in 2002-2004. Letters indicate significant differences among habitats.

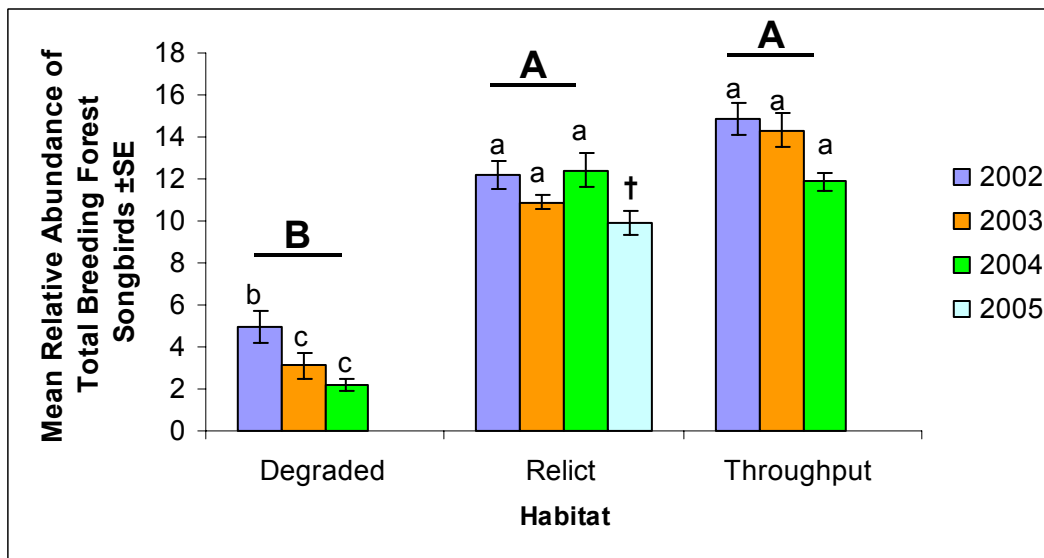


Figure 31. Mean relative abundance of total breeding forest songbirds among habitats sampled in Maurepas Swamp in 2002-2005. Capital letters indicate significant differences among habitats. Lower case letters indicate significant differences in habitat*year levels. † Only relict habitat was sampled in 2005; data displayed for visual comparison.

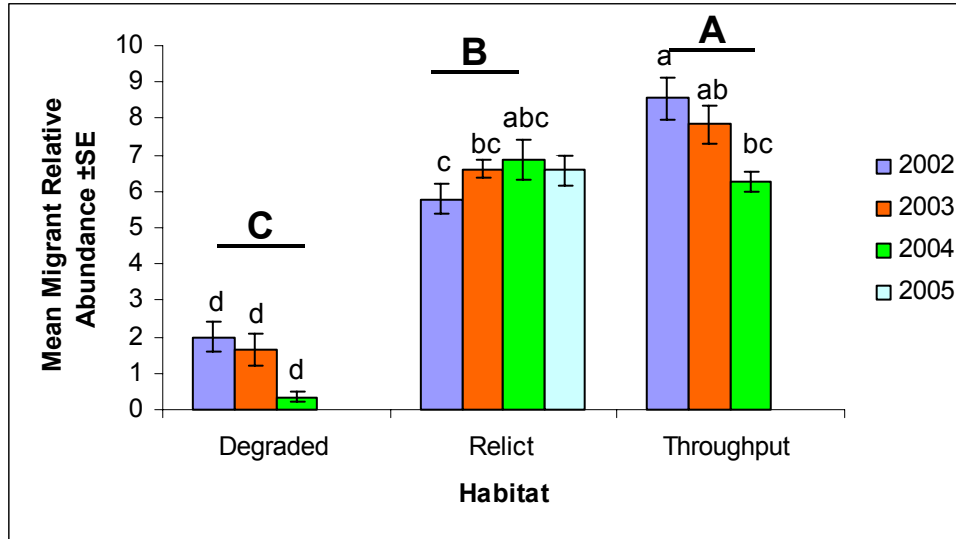


Figure 32. Mean relative abundance of breeding Neotropical migrant songbirds among habitats sampled in Maurepas Swamp in 2002-2005. Capital letters indicate significant differences among habitats. Lower case letters indicate significant differences in habitat*year levels. † Only relict habitat was sampled in 2005; data displayed for visual comparison.

PROW, NOPA and YTWA relative abundances in 2002-2004 were significantly affected by habitat type and the interaction between habitat type and year (Table 15). Relative abundances of PROW in 2002-2004 were not significantly different between throughput and relict swamp, but were significantly lower at degraded sites (Figure 33). Relative abundances of NOPA (Figure 34) and YTWA swamp (Figure 35) in 2002-2004 were highest in relict swamp, intermediate in relict swamp and lowest at degraded sites. Relative abundances of the three warbler species generally declined across years in degraded and throughput sites, and increased in relict sites.

Table 15. Results from MANOVA tests of effects of habitat type and habitat*year interaction on relative abundance of PROW, NOPA and YTWA in 2002-2004.

Species	Habitat Effect	Habitat*Year Effect
PROW	$F_{2,395}=222.05, p<0.0001$	$F_{4,395}=6.45, p<0.0001$
NOPA	$F_{2,395}=212.96, p<0.0001$	$F_{4,395}=4.40, p=0.0017$
YTWA	$F_{2,395}=16.13, p<0.0001$	$F_{4,395}=3.73, p=0.0054$

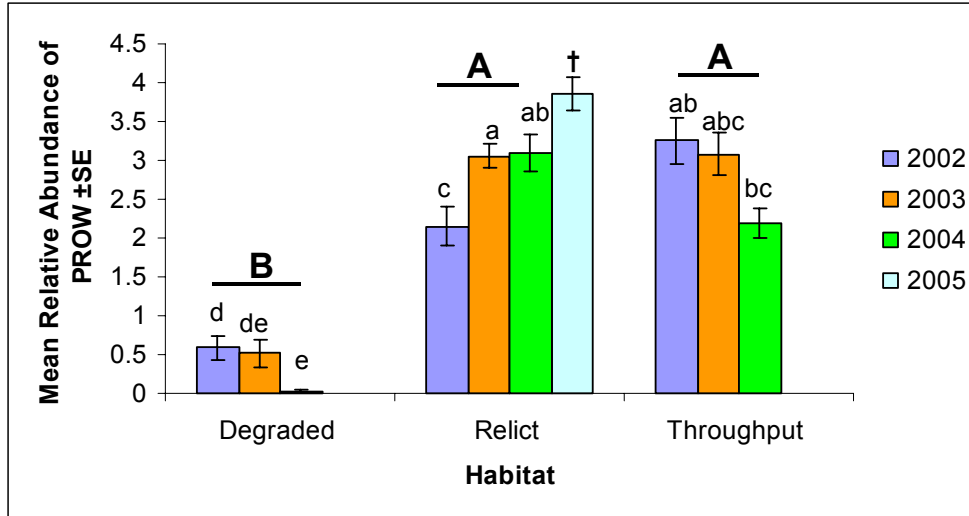


Figure 33. Mean relative abundance of Prothonotary Warbler among habitats sampled in Maurepas Swamp in 2002-2005. Capital letters indicate significant differences among habitats. Lower case letters indicate significant differences in habitat*year levels. † Only relict habitat was sampled in 2005; data displayed for visual comparison.

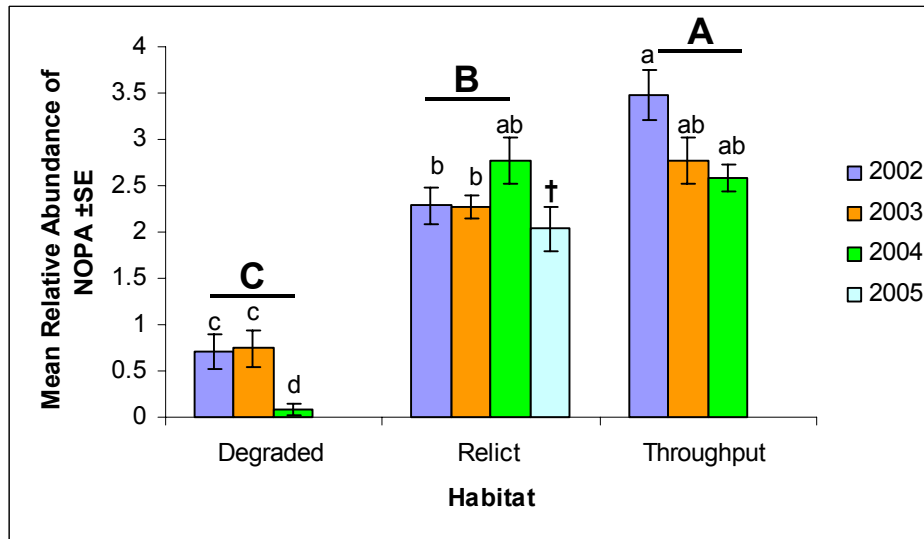


Figure 34. Mean relative abundance of Northern Parula among habitats sampled in Maurepas Swamp in 2002-2006. Capital letters indicate significant differences among habitats. Lower case letters indicate significant differences in habitat*year levels. † Only relict habitat was sampled in 2005; data displayed for visual comparison.

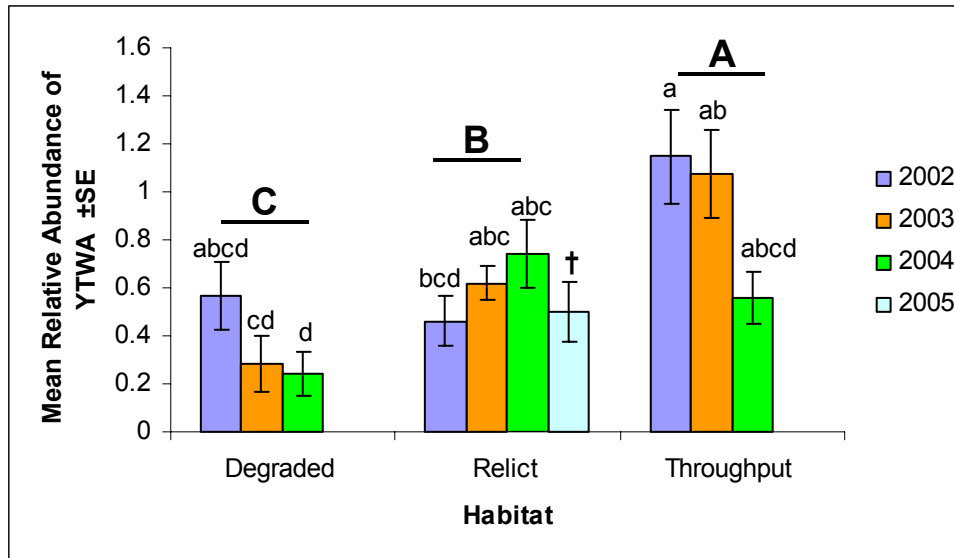


Figure 35. Mean relative abundance of Yellow-throated Warbler among habitats sampled in Maurepas Swamp in 2002-2006. Capital letters indicate significant differences among habitats. Lower case letters indicate significant differences in habitat*year levels. † Only relict habitat was sampled in 2005; data displayed for visual comparison.

Ten point count surveys were conducted on Hope Canal, during which 77 point counts were completed by 10:05AM, and 11 point count surveys were conducted on Reserve-Relief Canal, during which 77 and 63 point counts, respectively, were completed by 10:05AM. A total of 1651 forest songbird detections were recorded during point counts on the two alternative diversion routes in 2006 (Table 13). Twenty-three species of forest songbirds were detected on Reserve-Relief Canal, and 25 species were detected on Hope Canal in 2006. The two species detected on Hope Canal that were not detected on Reserve-Relief Canal were Blue Jay (a single detection) and Fish Crow, but several detections of unidentified crows on Reserve-Relief Canal may have been Fish Crows. Mean forest-songbird species-richness was 6.52 (± 0.24) on Hope Canal and 7.02 (± 0.27) on Reserve-Relief Canal, and these means were not significantly different ($F_{1,135}=2.82$, $p=0.0956$). Habitat type had a significant effect on forest-songbird species richness on canals ($F_{2,135}=5.09$, $p=0.0074$), but the habitat*canal interaction was not significant ($F_{1,135}=2.02$, $p=0.1574$). Forest-songbird species richness was not significantly different between mixed forest and throughput swamp, but was significantly lower in BLH (Figure 36). Species richness of breeding Neotropical migrant forest-songbirds was not significantly different between Hope Canal and Reserve-Relief Canal ($F_{1,136}=3.56$, $p=0.0612$), but was significantly affected by habitat type ($F_{2,136}=10.24$, $p<0.0001$).

Breeding Neotropical migrant bird species-richness was not significantly different between mixed forest and throughput swamp, but was significantly lower in BLH (Figure 37).

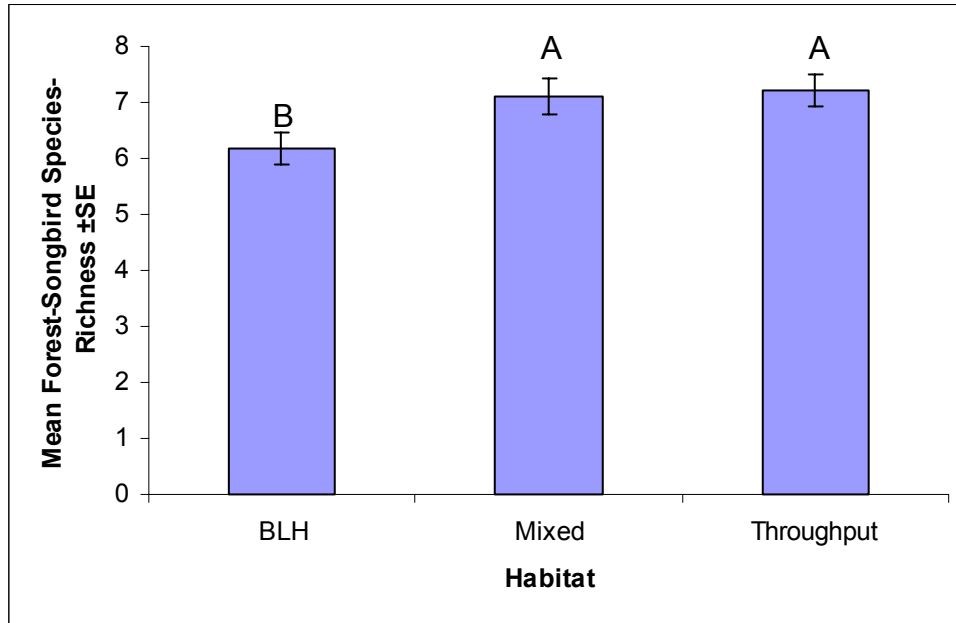


Figure 36. Mean forest-songbird species-richness among habitats on Hope and Reserve-Relief Canals. Letters indicate significant differences between habitats.

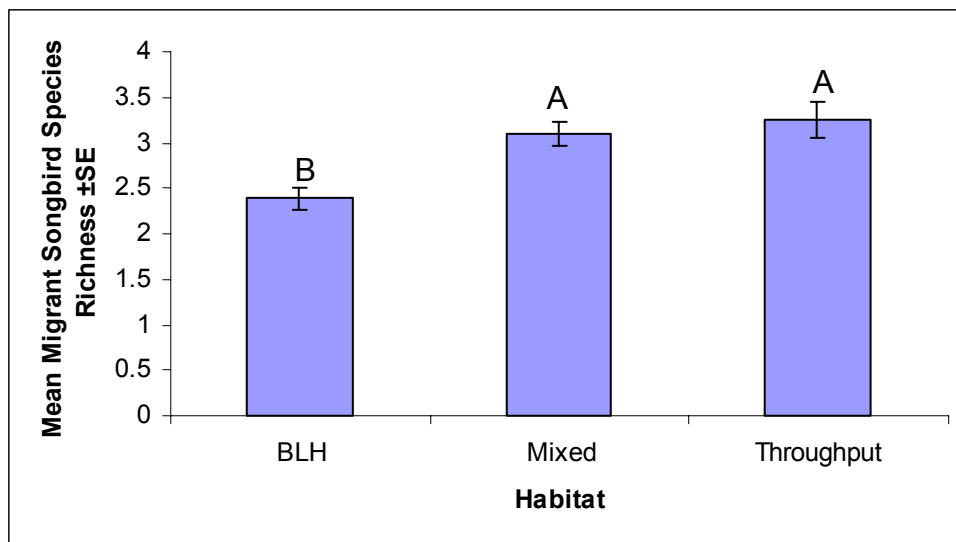


Figure 37. Mean breeding Neotropical migrant songbird species-richness among habitats on canal survey routes in 2006.

Mean total forest-songbird relative abundance was 11.5 ± 0.49 on Hope Canal and 11.4 ± 0.44 on Reserve-Relief Canal. Mean total forest-songbird relative abundance was

not significantly different between Hope and Reserve-Relief Canals ($F_{1,135}=0.91$, $p=0.3429$). Results from the reduced model indicated that mean total forest-songbird relative abundance varied significantly by habitat ($F_{2,135}=12.91$, $p<0.0001$) and habitat*canal ($F_{2,135}=6.07$, $p=0.003$). Mean relative abundance of total forest songbirds was not significantly different between mixed forest and throughput swamp, but was significantly lower in BLH habitat and lowest in BLH on Hope Canal (Figure 38).

In the test of the full model, mean relative abundance of breeding Neotropical migrant songbirds was not significantly affected by canal ($F_{1,135}=0.01$, $p=0.9320$) or the canal*habitat interaction ($F_{1,135}=0.26$, $p=0.6085$). In the test of the reduced model, with the canal effect removed, habitat had a significant effect on mean relative abundance of breeding Neotropical migrant songbirds ($F_{2,137}=27.21$, $p<0.0001$). Mean abundance of breeding Neotropical migrant songbirds was not significantly different between mixed forest and throughput swamp, but was significantly lower in BLH (Figure 39).

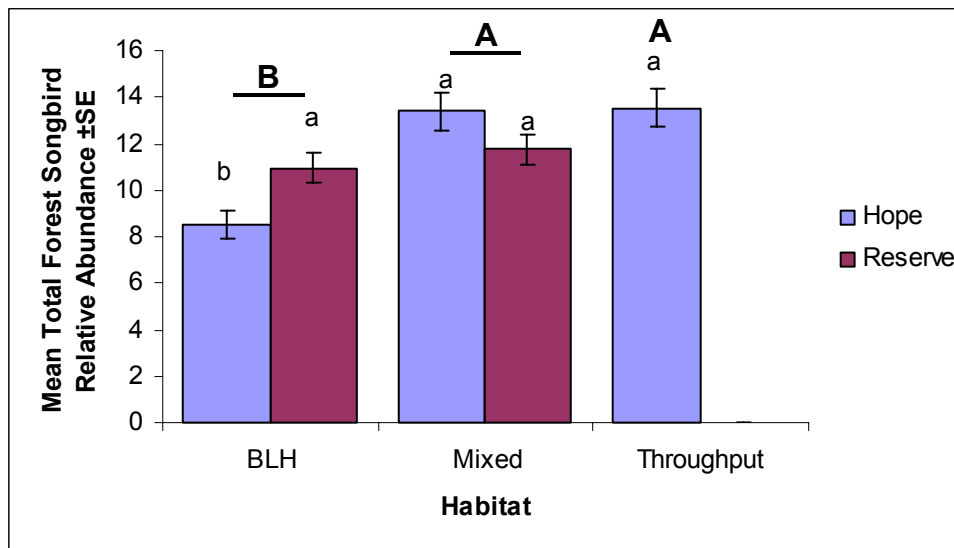


Figure 38. Mean total breeding forest-songbird relative abundance on Hope and Reserve-Relief Canals by habitat. Capital letters indicate significant differences among habitats; lower case letters indicate significant differences in habitat*canal levels.

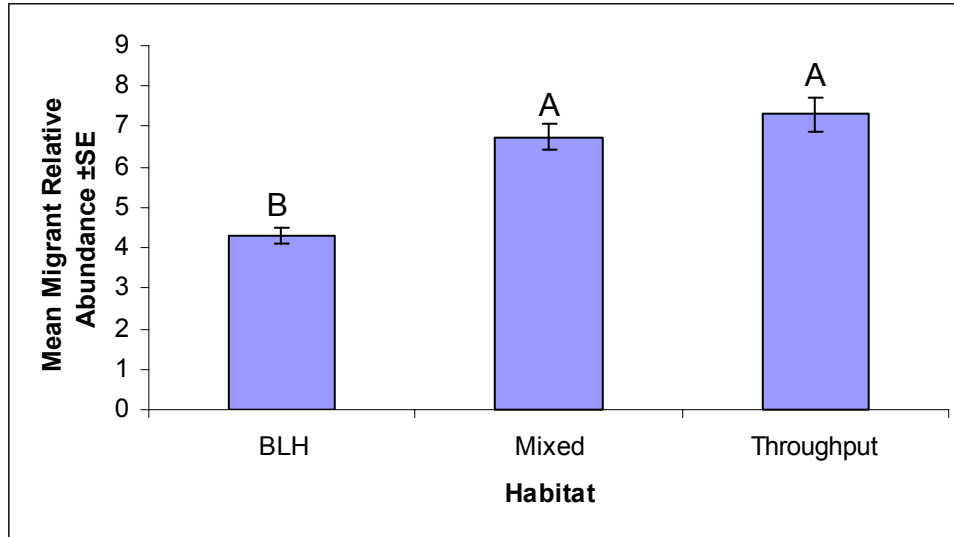


Figure 39. Mean relative abundance of breeding Neotropical migrant songbirds among habitats sampled on canals in 2006. Letters indicate significant differences among habitats.

In the MANOVA test of effects of canals and habitat types on relative abundances of PROW, NOPA and YTWA, only YTWA point-count detection data required log-transformation to approximate a normal distribution. In the full-model MANOVA, the overall effect of the canal*habitat interaction (Wilks' lambda= 0.9963, $F_{2,133}=0.17$, $p=0.9189$) was not a significant source of variation, and it was removed from the model. In the reduced-model MANOVA, the overall effect of canal on relative abundance of the three warbler species was not significant (Wilks' lambda= 0.9615, $F_{3,134}=1.79$, $p=0.1521$), but habitat type had significant effects on relative abundances of all three warbler species (Table 16). Relative abundance of PROW and NOPA were not significantly different between mixed forest and throughput swamp, but were significantly lower in BLH (Figure 40). Relative abundance of YTWA was highest in mixed forest, intermediate in throughput swamp, and lowest in BLH (Figure 40).

Table 16. Results from MANOVA test of effects of habitat type on relative abundance of PROW, NOPA and YTWA on potential diversion canals in 2006.

PROW	$F_{2,136}=9.84$, $p=0.0001$
NOPA	$F_{2,136}=16.55$, $p<0.0001$
YTWA	$F_{2,136}=39.25$, $p<0.0001$

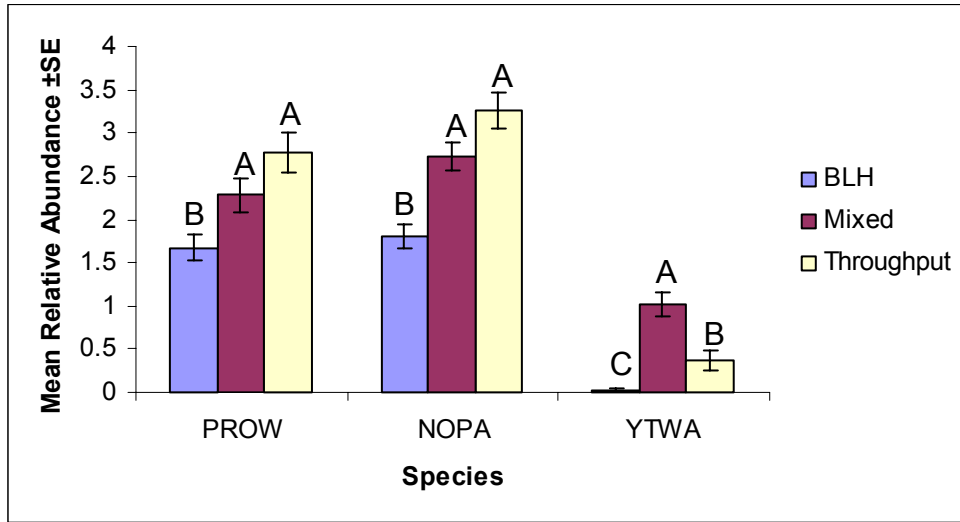


Figure 40. Mean relative abundance of PROW, NOPA and YTWA in habitats sampled on canals in 2006. Letters indicate significant differences within species, only.

Density estimates for YTWA, generated with DISTANCE using pooled data from 2002-2006, were not significantly different among habitat types and, overall, their density was estimated to be 1.2 singing males per hectare (90% confidence interval 0.931-1.55; Source: Jennifer Norris, Louisiana State University, personal communication). Densities of PROW and NOPA varied among habitats (Figure 41), but habitat effects generally followed patterns similar to those observed with analyses of relative abundance data. PROW density was highest in throughput and relict swamp, and was significantly lower in mixed and BLH habitat, which were similar to degraded sites. With the exception of BLH, NOPA density was significantly higher in all forested habitats relative to degraded sites, and significantly higher in throughput swamp relative to relict swamp. However, 90% confidence limits for mixed forest were extremely broad and the upper bound (13.32 singing males per hectare) was higher than upper bounds for all other forested habitats. Ninety percent confidence intervals for PROW and NOPA density estimates overlapped within all habitat types.

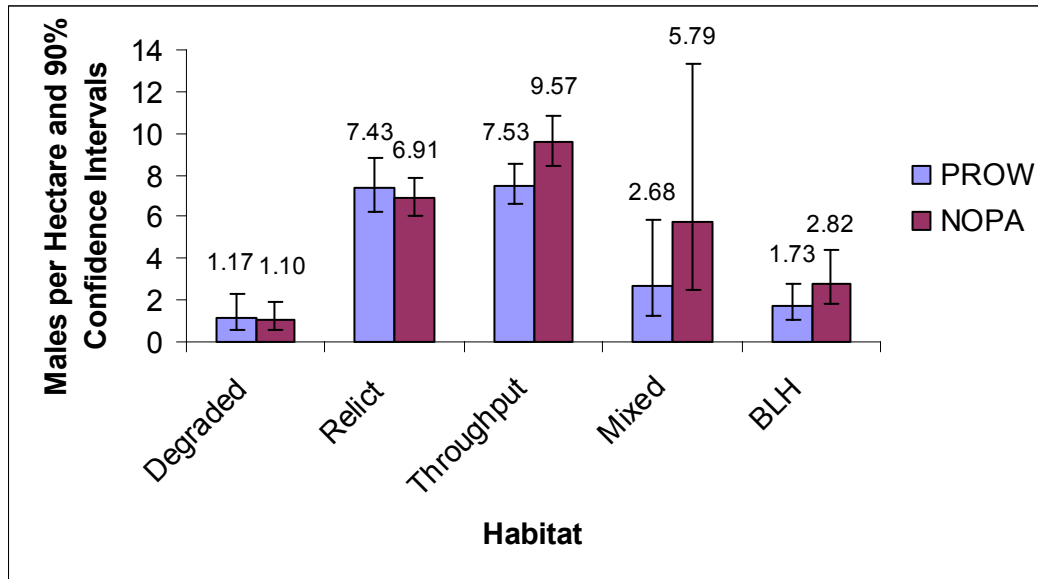


Figure 41. Density estimates for singing, male Prothonotary Warbler and Northern Parula in habitats sampled in Maurepas Swamp, 2002-2006. Values above bars are density estimates of singing males per hectare.

Discussion

Breeding forest birds on proposed diversion routes

Although caution should be used when interpreting results from a single season of point count data collection, because birds are highly mobile and populations may fluctuate on a yearly basis, in our study of the breeding forest-songbird community on two proposed diversion canal routes, Hope Canal and Reserve-Relief Canal, we found that there was no significant difference between canals with respect to mean total forest-songbird species-richness or relative abundance, mean breeding Neotropical migrant species-richness or relative abundance, and mean relative abundance of PROW, NOPA and YTTA. Habitat types within canals had significant effects on all of the aforementioned variables. Mean total forest-songbird species-richness and relative abundance, and mean breeding Neotropical migrant species-richness and relative abundance were not significantly different between throughput and mixed forest habitats, but they were all significantly lower in BLH habitat. We did not collect vegetation data on our survey routes on canals, so we are unable to determine if forest structure in BLH adjacent to canals at the southern end of Maurepas Swamp was responsible for these results. However, subjectively speaking, much of the shoreline of the canals in BLH habitat was dominated by woody vegetation with a relatively sparse understory, and

foliage complexity was relatively simple compared to more northerly sections of the canals where thickets of *Rubus*, understory trees, and vine tangles were more abundant. Therefore, differences in foliage complexity among habitats might have been responsible for differences in relative abundance and species richness of forest songbirds. It should be noted that we did not survey diversion canal routes near their juncture with the Mississippi River, including BLH forest on the Mississippi River levee and a small (~1ha) patch of forest that would be destroyed by canal construction on the Hope Canal alternative, between River Road and Airline Highway. However, these stands appear to be unremarkable, young BLH, and a small gap, relative to the total area of levee forest in the area, would be created for the diversion control structure.

We did not detect any ground-nesting bird species during the breeding season on either canal (Appendix A). Zoller (2004) noted the absence of ground nesting birds in relict and throughput swamp, and he hypothesized that these birds were absent due to the persistently or frequently flooded condition of relict and throughput forest. However, this does not explain the absence of ground nesting birds from drier BLH habitat at the southern periphery of Maurepas Swamp. Species richness of forest songbird communities in BLH in other regions may be greater than that in BLH at the southern periphery of Maurepas Swamp (Pashley and Barrow 1993, personal observations). In addition to possible flooding and habitat structure effects on forest-songbird species richness, habitat patch size in general (van Dorp and Opdam 1987, Boulinier et al. 1998), and the width of BLH strips embedded in other habitats (Hodges and Krementz 1996, Kilgo et al. 1998) have been shown to be positively associated with species richness. Therefore, species richness in BLH in Maurepas Swamp may be lower relative to other areas due to the size and width of the patches of BLH along Hope Canal and Reserve-Relief Canal.

Effects of swamp degradation on breeding bird communities

We examined the effects of swamp degradation on the breeding forest-songbird community using pooled data collected in 2002-2005 in throughput, relict and degraded swamp habitats. Zoller (2004) found that species richness of forest birds and Neotropical migratory birds in the breeding season were not significantly different between

throughput and relict swamp, but were significantly lower in degraded sites in Maurepas Swamp in 2002 and 2003. Our results, using a larger data set collected in 2002-2004, largely confirmed results obtained by Zoller in his study of the bird community of Maurepas Swamp (2004). Species richness of forest songbirds as a group was not significantly different between throughput and relict swamp, but was significantly lower at degraded sites. However, we found that species richness of breeding Neotropical migrant songbirds was significantly different among all habitats sampled. Breeding Neotropical migrant bird species-richness was highest in throughput sites, intermediate in relict sites and lowest in degraded sites. The difference between our results for breeding Neotropical migrant bird species richness and those obtained by Zoller (2004) is most likely due to the fact that Zoller used maximum species richness observed per count location in his analyses, while we used the mean species-richness per point count location. However, the difference in breeding Neotropical migrant bird species-richness between throughput and relict sites in our analysis, although significant, was relatively small, and the overall pattern in breeding Neotropical migrant species-richness is similar between the two analyses; species richness of forest birds, be they resident or migrant species, declines sharply as swamp habitat transitions to marsh habitat.

Zoller (2004) found that relative abundances of forest songbirds, as a group, and migrant songbirds were not significantly different between throughput and relict sites, but were significantly lower in degraded sites. Our results were similar for relative abundance of total forest songbirds in 2002-2004, but we found that breeding Neotropical migrant songbird relative-abundance was significantly lower in relict sites relative to throughput sites. Trends in mean relative abundances of NOPA and YTWA, and density estimates for NOPA, followed, and probably caused, trends observed in mean relative abundance of breeding Neotropical migrants. However, mean relative abundance of total forest songbirds and PROW, and density estimates for PROW, were not significantly different between throughput and relict sites, suggesting that PROW, resident birds, or both, compensated for decreased relative abundance of NOPA and YTWA in relict sites. Zoller (2004) and Fox (2006) found that the relative abundance of PROW in relict swamp could be increased by adding nest boxes, which suggests that PROW populations in relict habitat have not reached their maximum potential density.

Although 90% confidence intervals overlapped widely, patterns of PROW and NOPA density across habitat types were similar to those observed with relative abundance data. The main difference between relative abundance data and density estimates generated with DISTANCE was the non-significant effect of habitat type on YTWA density. However, DISTANCE analyses were performed with data pooled across years, and high relative abundance of YTWA at degraded sites in 2002, and low relative abundance in throughput swamp in 2004 may have masked habitat effects in the analysis. Data included in DISTANCE analyses were limited to singing males and detections without sex assigned, and assumed to be singing males. However, PROW and NOPA were frequently detected by call notes, from which sex cannot be determined for NOPA, and from which sex may be determined for PROW only by observers with great familiarity with PROW. Therefore, density estimates should probably be viewed as conservative density estimates for birds of both sexes, rather than strictly singing males.

Lower mean species richness and relative abundance of forest songbirds were expected at degraded sites that are dominated by marsh vegetation and have little woody vegetation. However, although mean relative abundance and species richness of breeding Neotropical migratory warblers and total forest songbirds in degraded sites were only about one quarter that found in relict sites, these estimates may have been positively biased because some of these point count locations were near cypress domes or forested levees that may have supported more forest songbirds than typical degraded habitat. In 2004 a greater proportion of point count locations at degraded sites were located more than 200m from levee edges than in 2002 and 2003, and this may explain why PROW and NOPA relative abundances at degraded sites were lower in 2004 than in other years. Alternatively, observer efficiency has been shown to increase with experience (Kendall et al. 1996), and the observer who conducted point counts at the degraded and throughput sites in 2004 was less experienced than observers at all other sites in 2002-2005. Although we did not explore effects of observer efficiency in our relative abundance analyses, this might partly explain why mean relative abundances of PROW and YTWA were lower in throughput habitat in 2004 compared to other years. Observer efficiency, or bias, may also explain the generally increasing trends in relative abundance data in relict sites, where more experienced observers conducted point counts in all study years.

However, observers in 2004 and 2005 used electronic range finders to measure distances to birds during point counts, and these data may have been collected more accurately than data collected in previous years. While observer efficiency might explain some variation in relative abundance and species richness data, we observed significant declines in mean relative abundances of total forest songbirds, PROW and YTWA, and a significant decline in mean forest songbird species-richness within degraded habitat in 2002-2003 when only experienced observers collected data.

Zoller (2004) found that non-significant differences in forest songbird abundance between throughcut and relict swamp were explained by foliage structure complexity in the shrub layer in relict swamp. We hypothesize that in addition to foliage structure, high PROW and NOPA abundance in relict swamp may also be due to the availability of suitable nesting habitat for these species. PROW nest in tree cavities and many of the dead or broken-topped tupelo and swamp red-maple in relict swamp contain suitable nesting cavities. NOPA nest in Spanish moss (*Tillandsia usneoides*), and many mid-story trees in relict swamp, especially swamp red-maple, supported clumps of Spanish moss large enough to contain a NOPA nest, which may account for relatively high densities of NOPA in relict swamp.

Partners in Flight characterized Maurepas Swamp as being a large forested area of low restoration priority and of only moderate protection priority (Twedt et al. 1999), because it is an exceptionally large tract of wetland forest in the MAV, and because, at the time of their evaluation, a large percentage of the area was in private ownership. However, the Richard King Mellon Foundation donated two tracts of swamp forest within Maurepas Swamp, totaling more than 25,300 hectares, to the Louisiana Department of Wildlife and Fisheries in 2001, which increased public ownership in Maurepas Swamp to approximately 50%. It is unclear whether Partners in Flight considered the rate at which forest is being lost in Maurepas Swamp, or the populations of PROW and NOPA there, when they prioritized areas in the MAV for forest restoration. Our data suggest that the dense populations of PROW and NOPA in Maurepas Swamp are an important source population of these birds.

A fortuitous situation may exist for breeding PROW in relict areas of Maurepas Swamp where numerous suitable nest-cavities are available, and the diversity and

abundance of nest predators is relatively low. Apparent nest success of 137 PROW nests in nestboxes in 2002-2003 was greater than 70% (Zoller 2004, Fox 2006), and apparent nest success rates of early-season natural nests, determined from fledgling surveys of PROW and NOPA territories in 2005, were approximately 50% and 79.2%, respectively (Fox 2006). Apparent nest success of PROW in nest boxes in Maurepas Swamp was higher than apparent nest success rates of PROW in Arkansas (57%) (Drumtra 2006) and Tennessee (32-57%) (Petit and Petit 1996), and apparent nest success of natural nests in Maurepas Swamp was higher than natural nest success in Arkansas (22%) (Drumtra 2006) and Wisconsin (38%) (Flaspohler 1996).

Canopy cover in relict swamp is not continuous, and the site is frequently saturated or flooded, which may decrease the suitability of the habitat for some arboreal animals (e.g., rat snakes [*Elaphe spp.*], raccoons, flying squirrels [*Glaucomys volans*], eastern gray squirrels [*Sciurus carolinensis*]) which are known to depredate PROW nests in bottomland forests with more contiguous canopy cover (Petit 1999, Drumtra 2006, DMF personal observations). During several hundred field-hours in Maurepas Swamp, we observed only one Texas rat snake (*Elaphe obsoleta lindheimerii*), a potential nest-predator of forest songbirds and their eggs. Red-bellied Woodpeckers (RBWO) were the most abundant avian nest-predator in Maurepas Swamp (Appendix A) and the fifth most abundant forest bird in relict swamp, and on two occasions RBWO were observed depredating eggs from PROW nests (Fox 2006). Raccoons (*Procyon lotor*) were common in Maurepas Swamp, and were assumed to have been responsible for most depredations of PROW nests in nest box studies conducted in 2002-2003 (Zoller 2004, Fox 2006).

Brown-headed Cowbirds are obligate nest parasites, meaning that they do not build nests, and females lay their eggs in host nests and allow the host to rear the young until they fledge. Brown-headed cowbird nest parasitism may reduce clutch size, hatching success, and nestling mass and survival of PROW (Hoover 2003), and other forest songbird species (Lorenzana and Sealy 1999, Siegle and Ahlers 2004). Brown-headed Cowbirds accounted for only 0.005% of the total forest-songbird point count detections in 2002-2006. PROW nests are frequently parasitized by Brown-headed Cowbirds (Petit 1989, Petit 1999, Hoover 2003, Drumtra 2006) and PROW are the only

SCNB in North America that is frequently parasitized by Brown-headed Cowbirds (Petit 1999). Reported parasitism rates of PROW nests in nest boxes range from 0% (Petit 1989) to 50% (Hoover 2003), and from 12.3% (Goertz 1977) to 41% (Hoover 2003) of nests in natural cavities. However, Brown-headed Cowbird nest-parasitism was not observed in any PROW nests in nest boxes in 2002-2003 (Zoller 2004, Fox 2006), or in PROW nests in natural cavities in Maurepas Swamp in 2002-2006 (Fox 2006; DMF, personal observations; Zoller, personal communication). Only five Brown-headed Cowbird fledglings were detected during five years of research in Maurepas Swamp, and NOPA were feeding the BHCO fledglings in the three cases where the host species was observed.

In 2002-2006, 38.2% of the total forest-songbird point count detections collected in Maurepas Swamp was of bird species that use cavities in trees, made by rot or woodpeckers, for their nesting sites. These secondary cavity-nesting bird (SCNB) species included PROW, Carolina Chickadee, Eastern Tufted-Titmouse, Carolina Wren, Great-crested Flycatcher and Eastern Bluebird. The lack of tupelo regeneration in Maurepas Swamp and the vulnerability of swamp red-maple to wind-throw are of concern for SCNB because naturally created cavities are more abundant in hardwood tree species than in conifers (baldcypress), due to differing physiological responses to limb death and rot (Peace 1962, as cited in Waters et al. 1990 and Newton 1994). As tupelo and swamp red-maple die or fall without replacement, naturally occurring cavities will become rarer, leading to increased cavity limitation of breeding populations of SCNB in the swamp. Although mean relative abundances of total forest songbirds and PROW in relict swamp were not significantly lower than that in throughput swamp, approximately two-thirds of the Maurepas Swamp is currently classified as relict swamp habitat (Chambers et al. 2005), and overall tree mortality in relict swamp is approximately 2% per year (Shaffer et al. 2003). Furthermore, 15-100% of midstory trees in study plots in degraded swamp and relict swamp near bayou edges were wind thrown during Hurricane Katrina (Gary Shaffer, unpublished data). Since mid-story trees provide nesting habitat for PROW and NOPA, it is likely that recent hurricane damage will cause immediate population declines of PROW and NOPA, and probably total forest songbird abundance at degraded sites and relict swamp near degraded areas. It was estimated, before

Hurricane Katrina, that 42,460 ha of swamp will be lost in the Pontchartrain Basin alone by 2050 (Coast 2050). Therefore, as trees die due to various stressors, and formerly forested sites transition to open marsh, populations of forest songbirds will decline over an increasing area.

While declines of populations of forest songbirds *per se* are of concern, the ecological consequences of a 75% decline in forest songbird abundance between relict and degraded sites should also be considered. Most breeding passerine birds in temperate forests feed primarily on insects (Holmes and Schultz 1988), and caterpillars are the most important insect taxon in the diets of breeding passerine birds in temperate forests (Holmes and Schultz 1988). The water tupelo component of coastal baldcypress-tupelo swamps has been subject to severe spring defoliation by forest tent caterpillar (*Malacosoma disstria*) outbreaks on about a five-year cycle, since 1948 (Nachod and Kucera 1971 as cited in Chambers et al. 2005), and baldcypress leafroller caterpillars, an emerging insect pest of baldcypress, have moderately to severely defoliated the baldcypress component of Maurepas Swamp since about 1993 (Goyer and Chambers 1997). Caterpillar populations in Maurepas Swamp may not be regulated by insect predators and parasites in a density dependent manner, because some important insect predators and parasites of caterpillars pupate in soil, which is permanently saturated in Maurepas Swamp. Therefore, the major caterpillar population controls in Maurepas Swamp include starvation, disease, and vertebrate predators, especially by birds. Birds have the potential to consume large numbers of forest insects due to their mobility and high metabolic rates (Kirk et al. 1996). While birds may not have significant impacts on epidemic populations of caterpillars, birds may increase the time it takes for caterpillar populations to reach epidemic populations, increase the rate at which caterpillar populations decline following outbreaks, and increase the periodicity of caterpillar outbreaks (Holmes 1990).

Declines in bird species richness and abundance may result in decreased functionality of the insectivorous bird guild, increase caterpillar herbivory on remaining trees, and accelerate the rate at which relict swamp transitions into open marsh. Baldcypress and water tupelo are among the few 'foundation' tree species upon which swamp ecosystems are based. Outbreaks or introductions of defoliating insects can

destabilize forest ecosystems (Ellison et al. 2005, Scheffer et al. 2001), and effects of insect outbreaks on ecosystem stability may be especially severe in ecosystems in which resilience has been reduced by gradual anthropogenic changes in abiotic conditions, such as the Maurepas Swamp ecosystem (for reviews see Scheffer et al. 2001, Folke et al. 2004). Forest ecosystems with few foundation tree species, such as cypress-tupelo swamps, are inherently less resilient to perturbations due to their lack of functional redundancy (Ellison et al. 2005). Therefore declines in forest songbird populations may have positive feedback on the rate of tree mortality in Maurepas Swamp.

Regenerating trees, whether planted or naturally regenerated, must survive and grow in order to restore the forest in Maurepas Swamp. Regenerating seedlings and saplings under larger trees may be particularly vulnerable to caterpillar herbivory, because caterpillars may be concentrated on these trees after dispersing, or falling from larger trees (Goyer and Chambers 1997). Fox (2006) studied the effects of insectivorous birds on planted baldcypress and water tupelo seedling growth in relict stands in Maurepas Swamp by excluding birds from trees. Fox (2006) found that there was no difference between measures of defoliation on seedlings in the bird exclosure treatment and controls, and he also observed that common, insectivorous birds did not forage on trees less than 1.5m tall. This suggests that regenerating trees must be able to survive and grow to an 'escape size' of 1.5m before birds begin to act to control caterpillar populations on small trees. Fox (2006) videotaped adult PROW provisioning nestlings, and found that baldcypress leafrollers were the food item most frequently delivered to nestlings between late April and early May. Although PROW, and other insectivorous birds, did not forage on small trees, Fox (2006) conservatively calculated that a pair of adult PROW feed enough baldcypress leafroller caterpillars to their nestlings during early nest attempts to protect approximately 76g dry weight of baldcypress foliage from caterpillar herbivory, and these calculations did not include estimates for the amount of foliage protected by the adults' depredation of baldcypress leafroller caterpillars. Fox (2006) also hypothesized that NOPA, which were not studied intensively, may have an even greater protective value to baldcypress than PROW. NOPA began eating baldcypress leafrollers at an earlier caterpillar instar stage, less than 30% of observed NOPA foraging maneuvers were on trees other than baldcypress, and less than 5% were

in the herbaceous layer , while nearly 50% of observed PROW foraging maneuvers were in the herbaceous layer. Furthermore, fledgling surveys conducted by Fox (2006) suggested that NOPA are approximately 25% more likely to fledge a brood of young than PROW during the time of baldcypress leafroller activity.

Without restoration and protection of Maurepas Swamp, the dense populations of PROW and NOPA that breed there will decline over an increasing area as trees die due to flooding, salinity and herbivory stressors. Protecting the ecological services provided by these bird populations is further justification for, and perhaps an integral part of, efforts to restore forested wetlands in Maurepas Swamp.

Effects of Diversion Canal Construction on Breeding Songbirds

Diversion canal construction will require clearing forest from land along an existing canal. Based on our results from statistical tests of relative abundance of birds between proposed diversion routes, there would probably be little difference in the number of birds displaced per kilometer of diversion construction between canals, but more birds would be displaced on Hope Canal because that route is longer, and because it contains more throughput forest area, which has higher relative abundances of common breeding Neotropical migrants, than Reserve-Relief Canal. Specific dimensions of the canal construction area were not available at the time this report was prepared, but the number of birds affected by canal construction might be estimated by multiplying the area of each habitat type cleared by density estimates of singing males. For example, diversion canal construction on Hope Canal would require forest clearing along at least 5.5km of the existing canal, and if the width of the construction zone is 100m, then approximately 55 hectares of forest will be cleared during construction. Assuming that BLH, mixed forest and throughput swamp each account for one third of the area to be cleared, and that density estimates presented herein are actually conservative estimates of total numbers of birds (rather than singing males), then the number of PROW displaced by construction on Hope Canal would be calculated as: $(18.33\text{ha BLH}) \times (1.73 \text{ PROW/ha of BLH}) + (18.33\text{ha mixed forest}) \times (2.68 \text{ PROW/ha of mixed forest}) + (18.33\text{ha throughput swamp}) \times (7.53 \text{ PROW/ha of throughput swamp}) \approx 219 \text{ PROW}$. If forest clearing is done before migrant arrival in spring, then direct impacts of construction on

migrants should be minimal. Resident forest birds (e.g. Carolina Chickadee, Northern Cardinal, etc., see Appendix A for additional examples), are unlikely to suffer direct mortality as a result of forest clearing, but birds that defend territories in the non-breeding season will be forced to compete with conspecifics in other areas to establish new territories. Forest clearing after 1 July coincides well with the window of construction opportunity for Bald Eagles and wading birds (see sections on Bald Eagles and wading birds, this report)

We hypothesize that an indirect impact of diversion canal construction and maintenance may be increased Brown-headed Cowbird nest-parasitism of NOPA, PROW and other breeding forest songbirds. The diversion canal levees will probably be planted with grasses, and kept mowed in order to monitor the levees for damage, and Brown-headed Cowbirds may preferentially select corridors within forested habitat that have mowed grass (Rich et al. 1994). PROW nests close to grazed levees in the White River National Wildlife Refuge in Arkansas were parasitized more heavily than interior nests (Gannon 2005, Larry Wood unpublished manuscript, personal observations). BHCO may travel up to 7km from foraging areas in search of nests (Rothstein et al. 1984). Therefore, levees of the diversion canal may increase BHCO parasitism across tens of square kilometers. Forest will be cleared along a minimum of 5.5km of the existing channel of Hope Canal, and this may provide foraging and staging areas for Brown-headed Cowbirds to foray deeply into Maurepas Swamp over an area of nearly 39km² in search of host nests to parasitize. However, populations of PROW and NOPA in Maurepas Swamp appear to be reproducing with relatively high success rates, and it is unlikely that increased Brown-headed Cowbird nest parasitism in a portion of the swamp will cause major population declines of host species. Construction of levees and maintenance roads will also provide people, and mammalian nest predators (e.g. feral cats and dogs, raccoons, opossums [*Didelphis virginianus*]) easy access to interior swamp habitat. Mammalian nest predators may depredate birds and/or nests adjacent to levees, and unless access to the levees is restricted, people are likely to dump trash and garbage along maintenance roads on levees as they have on Reserve-Relief Canal.

Effects of Diversion Canal Operation on Breeding Songbirds

During diversion operation, the area at the outfall is expected to be flooded with greater than 1m of water (Russ Joffrion, Louisiana Department of Natural Resources, personal communication). PROW frequently nest in cavities less than 1.5m off the ground (Petit 1999, personal observations). Therefore, flooding will reduce the availability of nesting cavities for PROW near the outfall area, and destroy nests of PROW built in low cavities before the operation period and also nests of bird species that may nest low in shrubs (e.g. Northern Cardinal, White-eyed Vireo, and Red-eyed Vireo). Flaspohler (1996) reported that 36% of Prothonotary Warbler nests that were active during record flooding of the Mississippi River in 1993 were destroyed. However, PROW are adapted to wetland forest environments in which water levels may fluctuate during the nesting season, and they usually renest in higher cavities, if they are available following nest failures (DMF personal observations). PROW prefer to nest over water, presumably because it reduces nest predation (Petit 1999). Therefore, increased flooding may increase nest success of PROW in deeply flooded areas. Most resident and breeding Neotropical migrant songbirds that breed in Maurepas Swamp, including PROW, are able to renest multiple times after nest failures.

As many as 50% of PROW foraging maneuvers may be performed in the herbaceous layer in relict swamp (Fox 2006). Diversion canal operation will temporarily result in reduced available foraging area for PROW and other birds that have been observed to forage in the herbaceous layer (e.g. Red-winged Blackbirds, NOPA). But, if the herbaceous layer of the swamp responds to freshwater reintroduction as expected, then it should quickly grow above maximum flood stage and restore, and eventually increase, foraging surface area in the outfall area. Furthermore, PROW were frequently observed gleaning insects from the water's surface and floating debris in relict swamp, and most insectivorous birds that breed in Maurepas Swamp have been observed foraging on abundant baldcypress leafrollers in spring at heights of 1.5m and greater (Fox 2006). Diversion operation is expected to increase primary productivity in affected areas, and increased primary productivity may result in increased productivity of birds' insect prey (Souther-Effler 2004).

The expected improvement in forest health following diversion operation should reduce tree mortality, including mortality of trees containing suitable nest cavities for PROW, and increase primary productivity. Increased primary productivity should increase foliage complexity, which Zoller (2004) showed was positively correlated with relative abundance and species richness of the forest songbird community in Maurepas Swamp.

Recommendations

Breeding activity of some resident forest songbirds (e.g. Carolina Chickadee, Tufted Titmouse) may begin in February, and breeding behavior of some larger passerines and non-passerines (e.g. woodpeckers, owls, hawks) may begin in December or January. The earliest arrival dates for Neotropical migrant birds that breed in Maurepas Swamp are approximately 1 March for Northern Parula and Yellow-throated Warbler, and 9 March for Prothonotary Warbler. Breeding activity of most songbirds in Maurepas Swamp ends about 1 July, although we observed one active PROW nest that was assumed to have fledged on or about 4 July. Therefore, we recommend that forest clearing for canal construction begin after 1 July and end before 1 January. Fortunately, these dates largely coincide with the window of opportunity for construction activities to avoid disturbing nesting Bald Eagles. We recommend that the diversion be operated at high flow rates in late March and early April to prevent forest songbirds from building nests in flood-prone areas. We also recommend that access, or at least vehicular access to levees, be restricted to reduce human impacts along the diversion canal.

The diversion canal levees will probably be planted in grass, which will probably create foraging and staging areas for breeding Brown-headed Cowbirds. We recommend that Brown-headed Cowbird use of levees, and nest success of nearby songbird nests, be monitored to determine if Brown-headed Cowbird populations increase to the point where active control might be warranted.

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Passage-Migrant Songbirds

David M. Fox and Philip C. Stouffer

Introduction

Millions of Neotropical migrant birds pass through southeastern Louisiana in spring and fall as they move between their breeding grounds in North America and wintering grounds in the Caribbean, Central America, and South America (Lincoln 1935, Barrow et al. 2005). Coastal forests provide stopover habitat for many of these Neotropical migrant birds (Barrow et al. 2005) and Maurepas Swamp is one of the largest remaining, contiguous forest patches near the Gulf Coast. In southeastern Louisiana, bottomland hardwood forest in the Pearl River drainage appears to be preferred to swamp forest as stopover habitat for many migrants (Stouffer and Zoller 2006). However, Neotropical migrant birds may use a variety of less suitable habitats as stopover habitat depending on weather and their energetic condition (Lowery 1945). Destruction of much of the bottomland hardwood forest in the Pearl River drainage by Hurricane Katrina in fall 2005, and extensive coastal development in many areas of the Gulf coast, might increase the relative importance of Maurepas Swamp as stopover habitat for passage migrant birds. The goal of this study was to determine the species composition, species richness and abundance of passage-migrants along proposed diversion canal routes and the surrounding swamp during spring and fall migration.

Methods

Spring passage-migrants

In April and May 2006 we conducted ten-minute variable-radius point counts of birds on Hope Canal and Reserve Canal (see Breeding Forest-Songbirds, this report, Figure 9). Detections of Neotropical passage-migrant birds from these counts were used in our analyses of spring passage-migrant bird abundance and species richness. Some Neotropical migrant birds' breeding distributions include Maurepas Swamp (e.g., Northern Parula, Prothonotary Warbler, Red-eyed Vireo) and also forested areas in more northerly locations. In order to estimate the numbers of these birds in-transit through the swamp, we compared mean relative abundance per point count of these species from

counts conducted in April when both passage migrants and breeding birds should be present, and counts conducted in May, by which time most passage migrants should have passed through the area.

Birds may respond readily to playback recordings of predators and/or passerine alarm calls during the non-breeding season (Johnson et al. 1981, Sliwa and Sherry 1992, Zoller 2004) and the breeding season (Zimmerling and Ankney 2000). In attempt to increase detection rates of passage-migrants, we conducted 5-minute vocalization playback surveys (Johnson et al. 1981, Sliwa and Sherry 1992, Zoller 2004) for passage migrant birds. Five-minute playback surveys for passage-migrants were conducted by playing recorded calls of Eastern Screech Owl (*Megascops asio*) (Zoller 2004) or Ferruginous Pygmy Owl (*Glaucidium brasilianum*), or recorded “pishing” (Zimmerling and Ankney 2000), for 30 seconds at regular intervals during the five-minute sampling period, and listening and looking for responding birds. Playback surveys were conducted at each point count location immediately following ten-minute point counts, and were repeated, in reverse order along the point count route, beginning one half-hour after the last point count was conducted, and generally beginning after 10AM. Only new species and new individuals of passage migrant birds were recorded during playback surveys to avoid double counting individuals recorded during ten-minute point counts.

In order to determine if spring passage-migrants that continued flying past the Gulf Coast “fell out” along Hope Canal and Reserve-Relief Canal and used these forested areas as stopover habitat, we conducted a second set of ten-minute point counts followed by five-minute playback surveys at point count locations, beginning at 2pm. Trans-Gulf migrants leave their southern “jumping-off” points near the Gulf of Mexico at dusk, and fly non-stop to the U.S. Gulf-coast, or beyond (Lowery 1945). With normal, favorable wind conditions, migrating birds arrive at the U.S. Gulf-coast the following day, mostly between noon and 2pm, and most birds have stopped moving by 6pm (Gauthreaux 1971).

Passage-migrants that were detected between point count locations were recorded and assigned to the most recently surveyed point count location. Only new individuals and species of passage-migrants detected on days in which playback surveys were conducted immediately following each point count, and sampled again after point counts were completed on the entire route, were included in analyses to avoid double counting

of birds. All detections recorded at point count locations within a day were pooled across survey methods for estimates of passage-migrant relative abundance and species richness. Mean relative abundance and species richness of spring passage-migrants were defined as the mean per point count location. Migrant bird species that are known to winter in Louisiana in full or in part (e.g. Blue-Gray Gnatcatcher; see Appendix A for migratory status of birds) were not included in analyses of passage-migrants.

To test for significant differences in total spring passage-migrant mean relative abundance per point, and mean species-richness per point, between Hope Canal and Reserve-Relief Canal, and among habitats (throughput, mixed, and BLH; see Breeding Forest-Songbirds section, this report, for habitat descriptions) and canal*habitat levels, data were analyzed as two-way ANOVAs, with 2X3 factorial designs and alpha levels of 0.05, using PROC MIXED in SAS (SAS Institute 2002). Abundance data were log-transformed to closer approximate assumptions of parametric statistics.

Fall passage-migrants

We conducted 30-minute transect surveys for fall passage-migrants following a modified version of the sampling protocol used by the USGS National Wetlands Research Center (Wylie Barrow, personal communication). Transect surveys were conducted by walking slowly along 300m transects on roads near Hope Canal and Reserve Canal while recording detections of fall passage-migrants, or observers used kayaks or canoes and surveyed 600m of one shoreline of the Blind and Tangipahoa Rivers. Transects on Hope Canal and Reserve-Relief Canal were located more than 250m north of Airline Highway, and separated by 250m. Transects on Hope Canal and Reserve-Relief Canal were surveyed on the same day, with the order in which canals were surveyed alternated between survey days. Transects on Blind River were located at least 100m from Airline Highway.

With the exception of transects on the Blind River, and two of five transects on the Tangipahoa River, transects were located in bottomland hardwood forest. This resulted in missing cells for analyses of habitat effects on fall passage-migrant abundance and species richness. Therefore, fall passage-migrant data were analyzed to detect significant differences in abundance and species richness among sites. Fall passage-

migrant abundance and species richness data were log-transformed to closer approximate assumptions of parametric statistics, and data were analyzed as one-way ANOVAs with alpha levels of 0.05 with PROC MIXED in SAS (The SAS Institute 2002). Migrant bird species that are known to winter in Louisiana in full or in part (e.g., Yellow-rumped Warbler; see Appendix A for migratory status of birds) were not included in analyses of passage-migrants.

In all statistical tests, non-significant variables with F-values <1.7 were removed with a backwards selection process, pooled with the error term, and reduced models were tested. Multiple pairwise comparisons were performed to determine significant differences among significant main effects and interactions using least-squared means with Tukey-Kramer adjustments. Letter groupings for least significant means were generated with Saxton's macro 'pdmix800.sas' (Saxton 1998) in PROC MIXED. In all figures, raw means and standard errors are displayed with least significant means letter-groupings from statistical tests.

Results

Spring Passage Migrants

Logistical constraints precluded standardization of data collection techniques across the duration of the study. On 9-20 April 2006 (hereafter referred to as sampling period one) we conducted ten-minute point counts and five-minute playback surveys immediately following point counts and after point counts were completed on the entire route. On 23 April-10 May 2006 (hereafter referred to as sampling period two) we conducted five-minute playback surveys at point count locations beginning at 2pm, and on 12-20 May 2006 (hereafter referred to as sampling period three) we conducted ten-minute point counts. Data were analyzed separately for the three distinct sampling periods.

Mean detections of spring passage migrants (Table 17) were not comparable between sampling periods, because survey effort and methodology varied among sampling periods. In sampling period one we conducted ten-minute point counts immediately followed by a five-minute playback survey, and then repeated playback surveys at point count locations after all point counts were completed. In sampling

period two we conducted afternoon point counts immediately followed by five-minute playback surveys. In sampling period three we conducted only ten-minute point counts in the morning.

Proportions of Neotropical migrant bird species known to breed in Maurepas Swamp (Great-crested Flycatcher, Northern Parula, Prothonotary Warbler, Red-eyed Vireo, Ruby-throated Hummingbird, Yellow-billed Cuckoo, and Yellow-throated Warbler, hereafter referred to as breeding migrants) that were in passage through the area were assumed to have completed passage by the beginning of the third sampling period, 12 May. Therefore, these species were not included in analyses of species richness for the third sampling period. Mean detections per point count of these breeding migrants were frequently greater in period three relative to period one (Table 17), suggesting that these species did not reach their maximum densities until after the first sampling period. Differences in relative abundances of breeding passage-migrants between periods one and three were generally within one standard deviation of the means of period one. Because we could not determine how many individuals of these breeding species may have been in passage, we excluded these species (Great-crested Flycatcher, Northern Parula, Prothonotary Warbler, Red-eyed Vireo, Ruby-throated Hummingbird, Yellow-billed Cuckoo, and Yellow-throated Warbler) entirely from analyses of passage migrants.

A total of 282 individual counts were conducted across sampling periods and canals in 2006 (

Table 18), during which 2,992 detections of locally breeding and in-transit migrant birds were recorded. A total of 35 species of Neotropical migrant birds were detected on Hope Canal and Reserve Canal in spring 2006 (Table 17, Table 2), ten of which were known to breed in Maurepas Swamp (see Appendix A). Although we did not test the effects of playback surveys on estimates of total species richness and relative abundance, playback generated few responses from migrants, including locally breeding migrants.

Species richness of spring passage-migrants was significantly greater on Hope Canal than on Reserve-Relief Canal in sampling period one, but was not significantly different between canals during sampling periods two and three (Table 19, Figure 42). Habitat type had a significant effect on species richness of spring passage-migrants in sampling periods one and two (Table 19, Figure 42). Species richness was significantly lower in BLH relative to mixed and throughcut forest in sampling periods one and two, and species richness was lower in BLH, although not significantly so, in sampling period three (Table 19, Figure 42).

Mean total spring passage-migrant abundance was significantly greater on Hope Canal than on Reserve-Relief Canal in sampling period one, but was not significantly different between canals during sampling periods two and three (Table 19, Figure 43, Figure 44). Mean total spring passage-migrant abundance was significantly lower in BLH relative to mixed and throughput forest in sampling period one, but was not significantly different among habitats in sampling periods two and three (Table 19, Figure 43).

Total abundance and species richness of spring passage-migrants on Hope Canal and Reserve Canal peaked in the last week of April (Figure 44), although relatively large numbers of migrants passed through forest along Hope Canal in mid-April. The most commonly detected non-breeding spring passage-migrant bird species on Hope Canal and Reserve-Relief Canal was Indigo Bunting, followed by Gray Catbird (Table 17, Figure 44). We observed no major fallouts of spring passage-migrants on Hope Canal and Reserve-Relief Canal, even when storm fronts had passed through during the previous evening, or were active north of the survey routes during counts.

Table 17. Mean abundance of spring passage-migrant birds per sampling date (\pm SE) on Hope Canal and Reserve Relief Canal in 2006, by sampling period. Latest detection dates are not applicable for breeding migrants.

Species	Hope Canal			Reserve-Relief Canal			Earliest Detection Date	Latest Detection Date
	Sampling Period 1	Sampling Period 2	Sampling Period 3	Sampling Period 1	Sampling Period 2	Sampling Period 3		
Acadian Flycatcher	0	0.037 \pm 0.0211	0	0	0	0	8-May	10-May
Baltimore Oriole	0.037 \pm 0.0259	0	0	0	0.146 \pm 0.1263	0	20-Apr	28-Apr
Barn Swallow	0.093 \pm 0.0478	0	0	0	0.042 \pm 0.0291	0	13-Apr	NA
Blue Grosbeak	0.019 \pm 0.0185	0.012 \pm 0.0123	0	0.025 \pm 0.0250	0	0	9-Apr	8-May
Blue-winged Warbler	0.056 \pm 0.0315	0.012 \pm 0.0123	0	0	0	0	9-Apr	26-Apr
Chimney Swift	0.037 \pm 0.0259	0.049 \pm 0.0389	0.028 \pm 0.0278	0	0.021 \pm 0.0208	0.042 \pm 0.0417	13-Apr	NA
Common Nighthawk		0.025 \pm 0.0174	0	0	0	0	25-Apr	25-Apr
Unkn Nightjar	0.037 \pm 0.0370	0	0	0	0	0	19-Apr	19-Apr
Chestnut-sided Warbler	0.019 \pm 0.0185	0.062 \pm 0.0321	0	0	0.021 \pm 0.0208	0	20-Apr	28-Apr
Eastern Kingbird	0.019 \pm 0.0185	0.049 \pm 0.0347	0.028 \pm 0.0278	0.025 \pm 0.0250	0.104 \pm 0.0744	0	19-Apr	NA
Great Crested Flycatcher	0.074 \pm 0.0446	0.173 \pm 0.0550	0.278 \pm 0.0856	0.325 \pm 0.1154	0.500 \pm 0.1191	0.458 \pm 0.1201	9-Apr	NA
Gray Catbird	0.093 \pm 0.0398	0.160 \pm 0.0480	0.028 \pm 0.0278	0.025 \pm 0.0250	0.250 \pm 0.1209	0.042 \pm 0.0417	13-Apr	20-May
Hooded Warbler	0.037 \pm 0.0259	0	0	0	0	0	9-Apr	13-Apr
Indigo Bunting	1.889 \pm 0.3453	1.025 \pm 0.3333	0.111 \pm 0.0531	0.475 \pm 0.1290	0.188 \pm 0.0925	0	9-Apr	NA
Louisiana Waterthrush	0.019 \pm 0.0185	0.012 \pm 0.0123		0	0	0	17-Apr	25-Apr
Magnolia Warbler	0	0.062 \pm 0.0321	0.028 \pm 0.0278	0	0.021 \pm 0.0208	0	28-Apr	15-May
Mississippi Kite	0.130 \pm 0.0594	0.444 \pm 0.1204	0.306 \pm 0.0961	0.075 \pm 0.0422	0.792 \pm 0.3407	0.250 \pm 0.0903	9-Apr	NA
Northern Parula	3.537 \pm 0.2404	3.506 \pm 0.2050	3.472 \pm 0.2271	3.675 \pm 0.3004	3.125 \pm 0.2623	3.750 \pm 0.3142	9-Apr	NA
Northern Waterthrush	0	0.012 \pm 0.0123	0	0	0.042 \pm 0.0291	0	28-Apr	28-Apr
Orchard Oriole	0.278 \pm 0.0932	0.037 \pm 0.0211	0	0.275 \pm 0.1790	0	0	9-Apr	28-Apr
Ovenbird	0	0	0	0.025 \pm 0.0250	0	0	11-Apr	11-Apr
Painted Bunting	0	0	0	0.025 \pm 0.0250	0.167 \pm 0.0544	0.208 \pm 0.0847	18-Apr	NA
Prothonotary Warbler	3.204 \pm 0.2622	3.469 \pm 0.1933	3.639 \pm 0.3064	3.725 \pm 0.2727	3.125 \pm 0.2771	4.167 \pm 0.2990	9-Apr	NA
Purple Martin	0.074 \pm 0.0360	0.025 \pm 0.0174	0.389 \pm 0.0998	0.025 \pm 0.0250	0.625 \pm 0.2260	0.500 \pm 0.1346	17-Apr	NA
Rose-breasted Grosbeak	0	0.136 \pm 0.0909	0.000 \pm 0.0000	0	0.042 \pm 0.0417	0	25-Apr	28-Apr
Red-eyed Vireo	0.463 \pm 0.1014	0.309 \pm 0.0779	0.278 \pm 0.0757	0.125 \pm 0.0639	0.146 \pm 0.0595	0.042 \pm 0.0417	9-Apr	NA
Ruby-throated Hummingbird	0.222 \pm 0.0571	0.420 \pm 0.0579	0.500 \pm 0.0845	0.175 \pm 0.0706	0.188 \pm 0.0569	0.250 \pm 0.1085	9-Apr	NA
Scarlet Tanager	0	0.000 \pm 0.0000	0	0	0.021 \pm 0.0208	0	7-May	7-May
Summer Tanager	0.019 \pm 0.0185	0.025 \pm 0.0247	0	0	0.208 \pm 0.0840	0	20-Apr	3-May
Tennessee Warbler	0.296 \pm 0.0681	0.049 \pm 0.0242	0	0.075 \pm 0.0422	0	0	9-Apr	28-Apr
Tree Swallow	0.037 \pm 0.0259	0.012 \pm 0.0123	0	0.025 \pm 0.0250	0.021 \pm 0.0208	0	13-Apr	4-May
Wood Thrush	0	0.049 \pm 0.0389	0	0	0	0	25-Apr	25-Apr
Yellow-breasted Chat	0.093 \pm 0.0398	0.012 \pm 0.0123	0	0	0.021 \pm 0.0208	0	13-Apr	25-Apr
Yellow-billed Cuckoo	0.185 \pm 0.0595	0.111 \pm 0.0351	0.111 \pm 0.0531	0.075 \pm 0.0422	0.042 \pm 0.0291	0.125 \pm 0.0690	9-Apr	NA
Yellow Warbler	0	0	0.028 \pm 0.0278	0.025 \pm 0.0250	0.063 \pm 0.0462	0.042 \pm 0.0417	11-Apr	15-May
Yellow-throated Warbler	0.481 \pm 0.0944	0.765 \pm 0.1058	0.917 \pm 0.1885	0.350 \pm 0.1318	0.208 \pm 0.0663	0.333 \pm 0.1153	9-Apr	NA

Table 18. Summary of spring passage-migrant survey effort on Hope Canal and Reserve-Relief Canal in 2006.

	Hope Canal			Reserve-Relief Canal			Total
	Sampling Period 1	Sampling Period 2	Sampling Period 3	Sampling Period 1	Sampling Period 2	Sampling Period 3	
Number of Survey Points	9	9	9	6	6	6	15
Number of Survey Days	6	9	4	7	8	4	38
Counts	53	81	36	40	48	24	282
Number of Detections	618	896	365	382	486	245	2992

Table 19. Results from ANOVA tests of effects of habitat type and canal on species richness and mean total relative abundance of spring passage-migrants per point, by sampling period.

Variable	Sampling Period	Effect	F	p
Species Richness	1	Canal	$F_{1,90}=12.8$	0.0006
		Habitat	$F_{2,90}=4.13$	0.0191
	2	Habitat	$F_{2,126}=12.38$	<0.0001
		3	Habitat	$F_{2,57}=2.08$
Relative Abundance	1	Canal	$F_{1,90}=8.64$	0.0042
		Habitat	$F_{2,90}=10.64$	<0.0001
	2	Habitat	$F_{2,125}=0.03$	0.9747
		3	Habitat	$F_{2,57}=2.54$

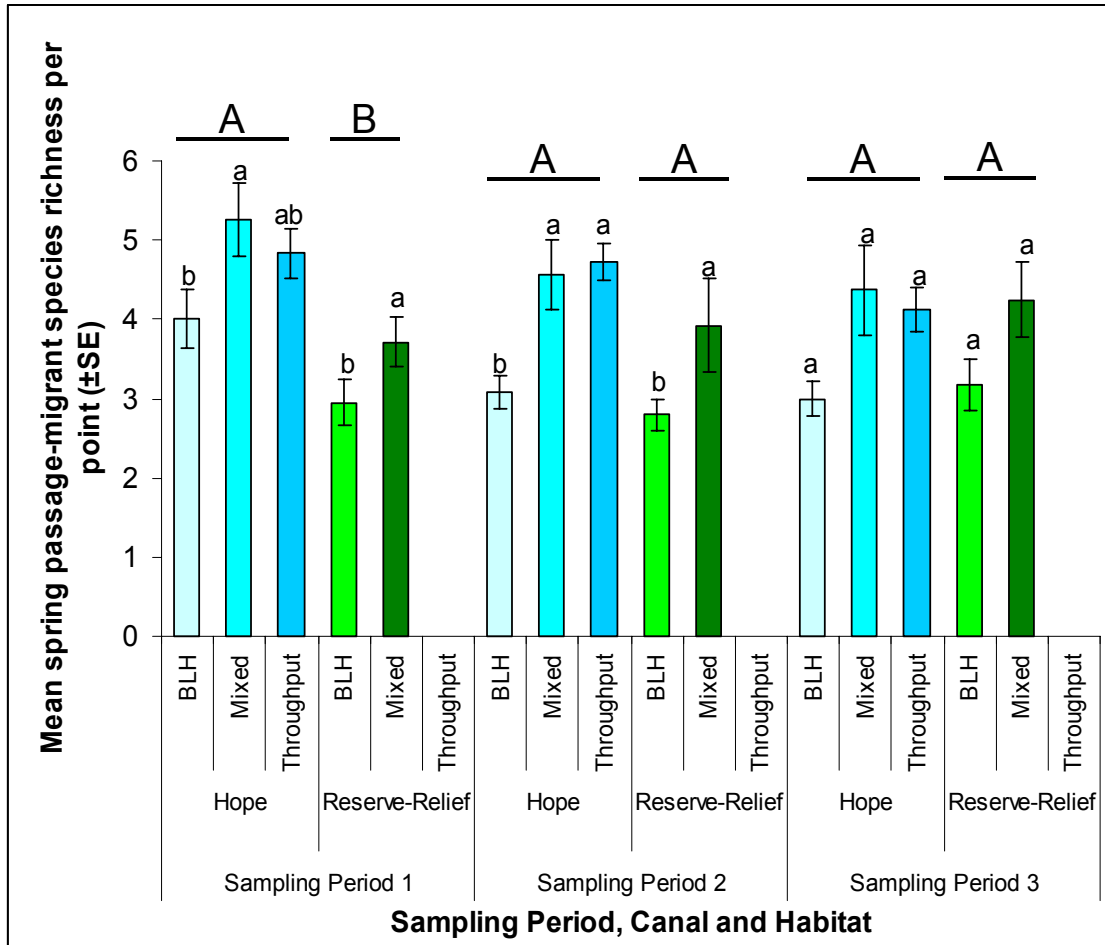


Figure 42. Mean spring passage-migrant species richness per point (\pm SE) on Hope Canal and Reserve-Relief Canal in 2006, by sampling period and habitat. Letters indicate significant differences within sampling periods, only. Capital letters indicate significant differences between canals, and lower case letters indicate significant differences between habitats

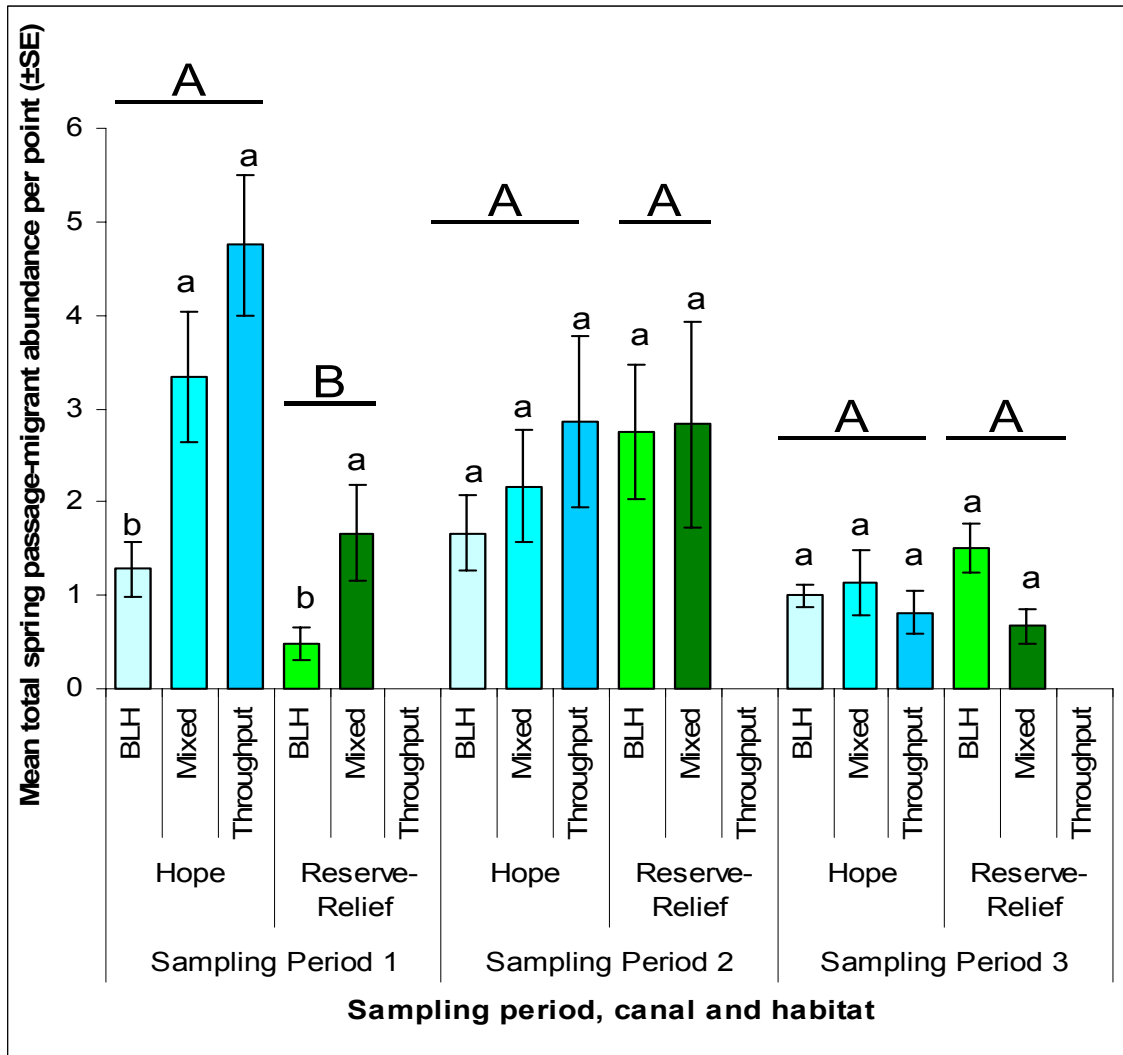


Figure 43. Mean total relative abundance of spring passage-migrants per point (\pm SE) on Hope Canal and Reserve-Relief Canal in 2006, by sampling period and habitat. Letters indicate significant differences within sampling periods, only. Capital letters indicate significant differences between canals, and lower case letters indicate significant differences between habitats.

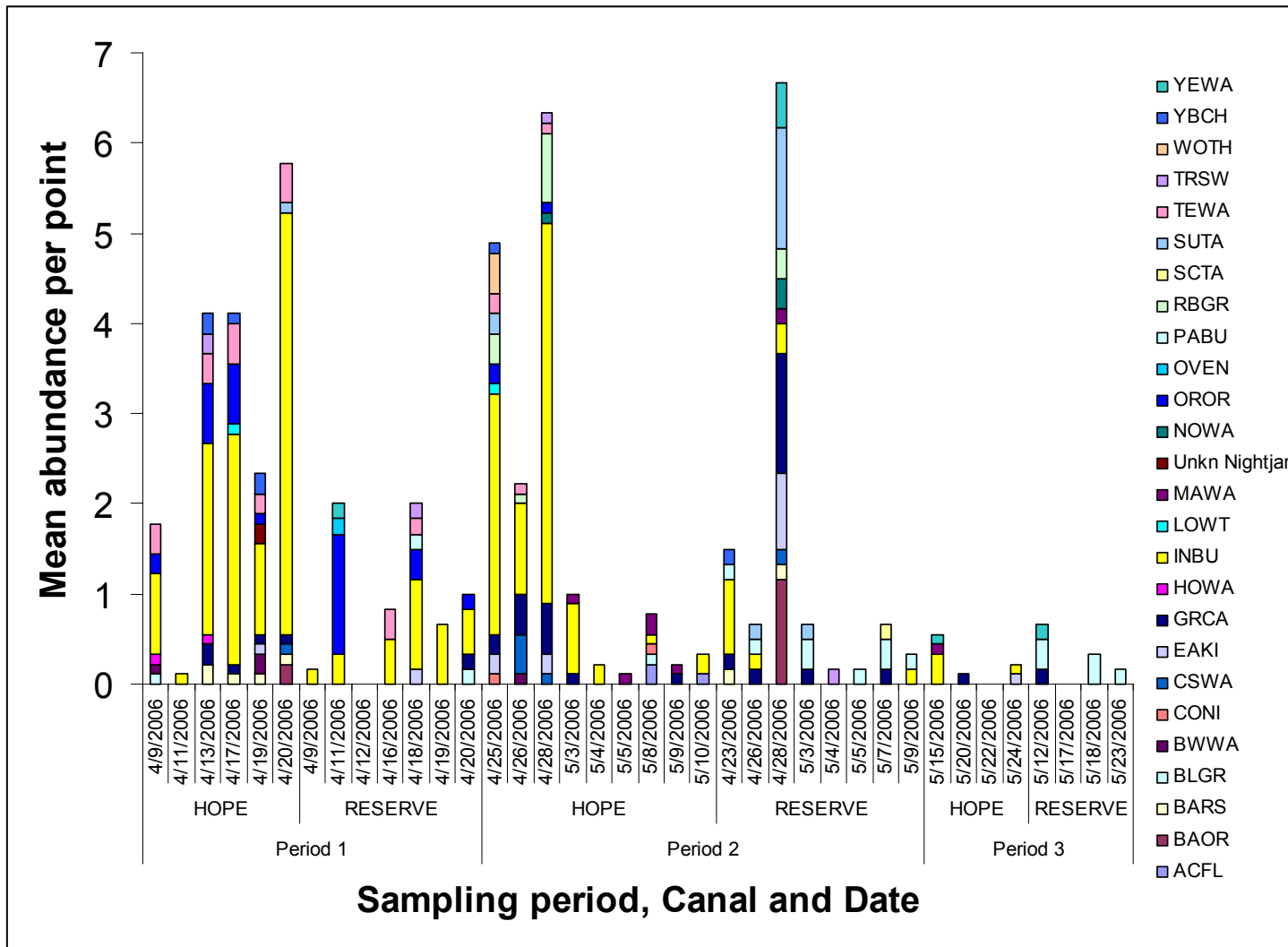


Figure 44. Mean abundance per point of spring passage-migrant species on Hope Canal and Reserve-Relief Canal in 2006, by sampling period and date.

Fall Passage Migrants

We conducted fall passage-migrant surveys on fewer days on Hope Canal than on Reserve-Relief Canal because access to the Hope Canal site was not permitted by the landowner after 1 October 2006 due to the start of the deer hunting season. Few surveys were conducted on Blind River because the lower portion of the river was made impassable by floating aquatic vegetation soon after surveys were begun. We conducted 82 transect surveys, across sites, between 3 September-15 October 2006, during which we recorded a total of 973 detections of fall passage-migrants (Table 20).

Table 20. Summary of fall passage-migrant survey effort in 2006.

	Blind River	Hope Canal	Reserve-Relief Canal	Tangipahoa River	Total
Transects	5	2	2	5	14
Transect-Days	10	24	29	19	82
Detections	50	312	254	249	973

A total of 46 species of fall passage-migrants was detected across sites, of which 10 were detected on Blind River, 32 on Hope Canal, 35 on Reserve Canal, and 25 on the Tangipahoa River (Table 5). Mean species richness per transect-day was significantly affected by site ($F_{3,79}=5.86$, $p=0.0012$), and was highest on Hope Canal, lowest on Blind River and Reserve-Relief Canal, and intermediate on Tangipahoa River (Figure 45).

Mean total abundance of fall passage-migrants was significantly affected by site ($F_{3,79}=3.39$, $p=0.022$), and was highest on Hope Canal, lowest on Blind River, and intermediate on Reserve-Relief Canal and Tangipahoa River (Figure 46). Peaks in fall passage-migrant species richness in 2006 were observed on 8 September on Blind River (seven species), 14 and 26 September on Hope Canal (14 and 13 species, respectively), 14 and 19 September on Reserve-Relief Canal (nine species), and 26 September on the Tangipahoa River (14 species) (Figure 47). Peaks in mean relative abundance of fall passage-migrants generally coincided with peak species richness, except on Reserve-Relief Canal where peak abundance was observed on 14-15 October when large numbers of Indigo Bunting were observed. The most frequently detected fall passage-migrant

species were Chimney Swift on the Blind and Tangipahoa Rivers, Ruby-throated Hummingbird on Hope Canal, and Indigo Bunting on Reserve-Relief Canal (Figure 6).

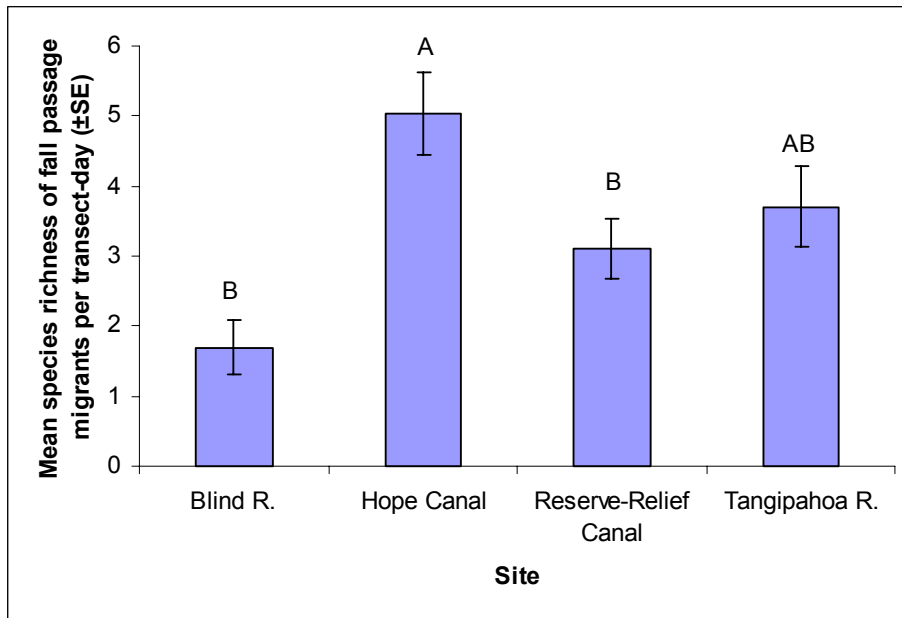


Figure 45. Mean species richness of fall passage-migrants per transect-day (\pm SE) in 2006.

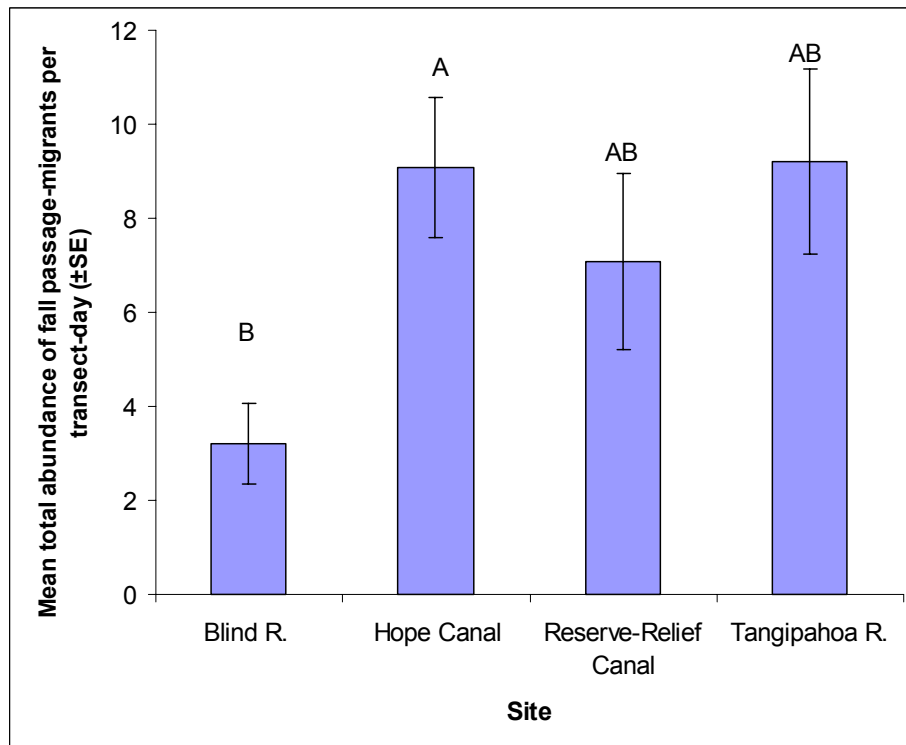


Figure 46. Mean total abundance of fall passage-migrants per transect-day (\pm SE) in 2006.

Table 21. Mean detections per transect-day (\pm SE) of fall passage-migrant species on survey routes in fall 2006.

Common Name	Blind R.	Hope Canal	Reserve-Relief Canal	Tangipahoa River	Overall Mean	Earliest Sample Date	Latest Sample Date
Yellow-billed Cuckoo	0	0.042 \pm 0.042	0.069 \pm 0.048	0.05 \pm 0.05	0.048 \pm 0.024	NA	16-Sep
Chuck-will's-widow	0	0.042 \pm 0.042	0	0	0.012 \pm 0.012	9-Sep	9-Sep
Chimney Swift	0.8 \pm 0.554	0	0.034 \pm 0.034	1.5 \pm 0.467	0.47 \pm 0.146	7-Sep	14-Oct
Ruby-throated Hummingbird	0.4 \pm 0.221	2.292 \pm 0.588	0.241 \pm 0.081	1 \pm 0.290	1.036 \pm 0.207	NA	3-Oct
Unknown Hummingbird	0	0	0	0.25 \pm 0.204	0.060.050	NA	12-Sep
Eastern Wood-Pewee	0	0.417 \pm 0.180	0.034 \pm 0.034	0.05 \pm 0.05	0.145 \pm 0.057	3-Sep	15-Sep
Yellow-bellied Flycatcher	0	0.083 \pm 0.058	0.034 \pm 0.034	0	0.036 \pm 0.021	14-Sep	26-Sep
Acadian Flycatcher	0	0.042 \pm 0.042	0.172 \pm 0.112	0.1 \pm 0.069	0.096 \pm 0.044	14-Sep	26-Sep
Alder Flycatcher	0	0.167 \pm 0.098	0.103 \pm 0.076	0	0.084 \pm 0.039	3-Sep	19-Sep
Least Flycatcher	0	0.083 \pm 0.058	0.034 \pm 0.034	0	0.036 \pm 0.021	14-Sep	17-Sep
Great Crested Flycatcher	0.3 \pm 0.213	0.042 \pm 0.042	0.069 \pm 0.048	0.65 \pm 0.357	0.229 \pm 0.094	NA	26-Sep
Eastern Kingbird	0	0	0	0.15 \pm 0.109	0.036 \pm 0.027	7-Sep	12-Sep
Unknown Empidonax Flycatcher	0	0.083 \pm 0.058	0.172 \pm 0.112	0.35 \pm 0.131	0.169 \pm 0.054	7-Sep	29-Sep
Yellow-throated Vireo	0.3 \pm 0.213	0.208 \pm 0.104	0	0.4 \pm 0.266	0.193 \pm 0.076	NA	26-Sep
Red-eyed Vireo	0	0.5 \pm 0.170	0.345 \pm 0.278	0	0.265 \pm 0.110	NA	26-Sep
Philadelphia Vireo	0	0	0.034 \pm 0.034	0	0.012 \pm 0.012	15-Oct	15-Oct
Barn Swallow	0	0	0	0.15 \pm 0.082	0.036 \pm 0.021	NA	12-Sep
Tree Swallow	0	0	0.069 \pm 0.069	0	0.024 \pm 0.024	15-Oct	15-Oct
Bank Swallow	0	0	0	0.15 \pm 0.15	0.036 \pm 0.036	7-Sep	7-Sep
Unknown Swallow	0	0	0.103 \pm 0.103	0.15 \pm 0.109	0.072 \pm 0.045	NA	31-Oct
Veery	0	0	0.034 \pm 0.034	0.05 \pm 0.05	0.024 \pm 0.017	23-Sep	26-Sep
Gray Catbird	0	0	0.380.230	1.1 \pm 0.557	0.398 \pm 0.161	26-Sep	15-Oct
Golden-winged Warbler.	0	0.042 \pm 0.042	0	0	0.012 \pm 0.012	14-Sep	14-Sep
Magnolia Warbler	0	0.625 \pm 0.198	0.448 \pm 0.190	0.05 \pm 0.05	0.349 \pm 0.091	3-Sep	15-Oct
Chestnut-sided Warbler	0	0.042 \pm 0.042	0	0	0.012 \pm 0.012	7-Sep	14-Oct
Yellow Warbler	0	0.333 \pm 0.167	0.138 \pm 0.065	0	0.145 \pm 0.057	3-Sep	26-Sep
Black-throated Green Warbler	0	0	0.069 \pm 0.048	0	0.024 \pm 0.017	14-Oct	14-Oct
American Redstart	0.1 \pm 0.1	0.583 \pm 0.208	0.069 \pm 0.048	0	0.205 \pm 0.068	14-Sep	14-Oct
Black-and-white Warbler	0	0.292 \pm 0.112	0.103 \pm 0.058	0.3 \pm 0.164	0.193 \pm 0.055	3-Sep	15-Oct
Northern Parula	0.1 \pm 0.1	1.167 \pm 0.384	0.172 \pm 0.071	0.25 \pm 0.160	0.470.129	NA	15-Oct
Prothonotary Warbler	0	0.083 \pm 0.058	0	0.2 \pm 0.092	0.072 \pm 0.029	NA	26-Sep
Worm-eating Warbler	0	0.042 \pm 0.042	0	0	0.012 \pm 0.012	26-Sep	26-Sep
Ovenbird	0	0	0.034 \pm 0.034	0	0.012 \pm 0.012	25-Sep	25-Sep
Northern Waterthrush	0	0.167 \pm 0.078	0.310.112	0.1 \pm 0.069	0.181 \pm 0.049	7-Sep	5-Oct
Kentucky Warbler	0	0.167 \pm 0.130	0.069 \pm 0.048	0	0.072 \pm 0.041	7-Sep	14-Sep
Tennessee Warbler	0	0.167 \pm 0.098	0.069 \pm 0.048	0.05 \pm 0.05	0.084 \pm 0.035	16-Sep	15-Oct
Blue-winged Warbler	0	0.167 \pm 0.098	0.034 \pm 0.034	0	0.060.031	3-Sep	19-Sep
Hooded Warbler	0.1 \pm 0.1	0.125 \pm 0.069	0.069 \pm 0.048	0	0.072 \pm 0.029	7-Sep	26-Sep
Canada Warbler	0	0.042 \pm 0.042	0.034 \pm 0.034	0	0.024 \pm 0.017	7-Sep	19-Sep
Yellow-breasted Chat	0	0.083 \pm 0.058	0	0	0.024 \pm 0.017	19-Sep	21-Sep
Unknown Warbler	0	0.042 \pm 0.042	0.069 \pm 0.048	0.25 \pm 0.099	0.096 \pm 0.033	NA	29-Sep
Scarlet Tanager	0	0	0.034 \pm 0.034	0.1 \pm 0.1	0.036 \pm 0.027	29-Sep	5-Oct
Summer Tanager	0	0.458 \pm 0.159	0.138 \pm 0.082	0	0.181 \pm 0.057	3-Sep	26-Sep
Blue Grosbeak	0	0.042 \pm 0.042	0.034 \pm 0.034	0	0.024 \pm 0.017	16-Sep	26-Sep
Indigo Bunting	0.5 \pm 0.307	0.420.150	3.207 \pm 1.308	1.4 \pm 0.642	1.639 \pm 0.498	7-Sep	15-Oct
Rose-breasted Grosbeak	0	0	0.034 \pm 0.034	0	0.012 \pm 0.012	15-Oct	15-Oct
Dickcissel	0	0	0	0.1 \pm 0.069	0.024 \pm 0.017	21-Sep	26-Sep
Baltimore Oriole	0.1 \pm 0.1	0	0	0.1 \pm 0.069	0.036 \pm 0.021	7-Sep	19-Sep
Orchard Oriole	0.1 \pm 0.1	0	0	0	0.012 \pm 0.012	8-Sep	8-Sep
Unknown	0.4 \pm 0.306	0	0	0.15 \pm 0.082	0.084 \pm 0.043	NA	NA
Overall Mean	3.2 \pm 0.854	9.083 \pm 1.50	7.069 \pm 1.869	9.2 \pm 1.969	7.699 \pm 0.933	NA	NA

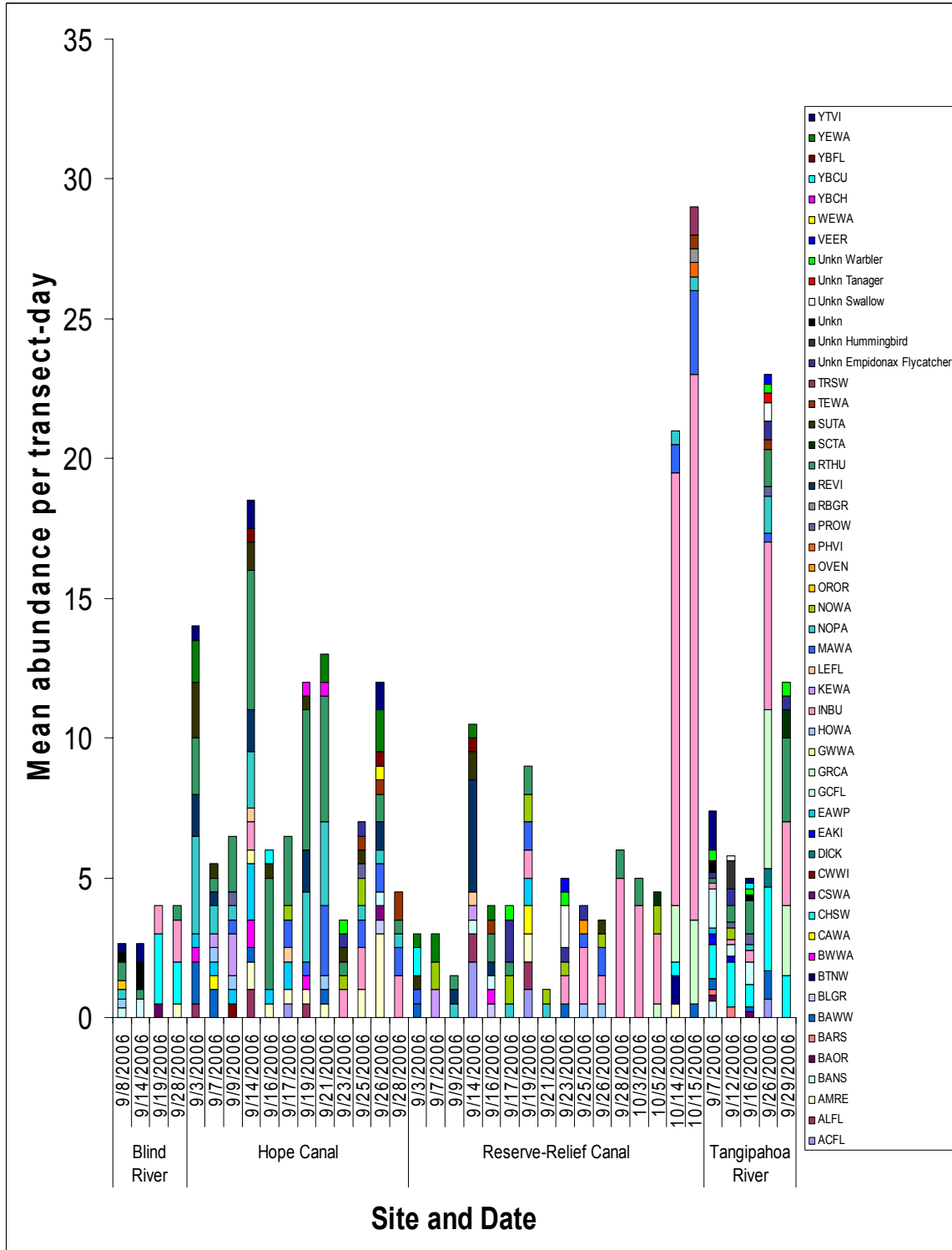


Figure 47. Mean abundance per transect-day of fall passage-migrant species in 2006, by site and date.

Discussion

Spring Passage Migrants

Spectacular trans-Gulf migrant fall-out events typically occur when birds encounter unfavorable weather conditions over the Gulf or near the coast (Lowery 1945). During flights with typical, favorable wind conditions many trans-Gulf migrant birds flying over Louisiana will fly past coastal forests to latitudes north of Baton Rouge, producing what is known as the “coastal hiatus” (Lowery 1945). The apparent preference for BLH in areas such as the Pearl River drainage, and the coastal hiatus are the most likely explanations for the relatively low relative abundance and species richness of spring passage-migrants on Hope Canal and Reserve-Relief Canal in 2006.

It is interesting to note that baldcypress swamp has been the dominant forest type in coastal Louisiana for thousands of years and that baldcypress was known for its lack of insect herbivores until the baldcypress leafroller began outbreaking in 1983. Perhaps the passage-migrant coastal hiatus is an adaptive behavior for avoiding stopover habitat with limited food resources. Most spring passage-migrants are in-transit through Maurepas Swamp during baldcypress leafroller activity, and several species of Neotropical migrant birds have been observed depredating baldcypress leafrollers (DMF personal observations). An important question to ask is whether spring passage-migrants might eventually alter their stopover habits in response to this novel and abundant food source.

Stouffer and Zoller (2006) found that densities of departing spring passage-migrants around Lake Maurepas were greater in swamp habitat than in marsh habitat, and that densities were greater in swamp on the southern rim of the lake than in more southerly areas of Maurepas Swamp. Stouffer and Zoller (2006) suggested that spring passage-migrants might stop-over on the south and west edges of the lake, before flying over that barrier. Our most northerly survey points were several kilometers south of Lake Maurepas, so we did not sample in areas which might have held the greatest densities of spring passage-migrants. However, during our field studies in more northerly areas of the swamp in 2002-2005, we detected passage-migrant species (e.g. Bobolink, Veery) that were not detected during our spring surveys of diversion canal construction corridors in 2006, which might indicate that spring migrant species may use areas of the swamp differentially.

We were unable to determine proportions of breeding migrants that used Maurepas Swamp in transit. Northern Parula, Yellow-throated Warbler and Prothonotary Warbler arrive in the first half of March. We probably missed a substantial proportion of these birds that were in-transit through Maurepas Swamp, but positive differences between their mean abundances per point in sampling period one and sampling period three suggest that some of these birds had not reached their maximum densities in late April. Alternatively, breeding migrants in-transit through Maurepas Swamp might not have been detected, because advertising their presence with song within a breeding bird's territory would most likely result in an aggressive encounter with the territorial male.

Spring passage-migrant species richness and relative abundance peaked during the second sampling period, even though less sampling effort per point was exerted in sampling period two than in sampling period one. Spring passage-migrant species richness and relative abundance were significantly higher on Hope Canal relative to Reserve-Relief Canal in sampling period one, but were not significantly different between canals in sampling period two when spring passage-migrant species richness and relative abundance peaked. Spring passage-migrant relative abundance in sampling period one, and species richness in sampling periods one and two were significantly lower in BLH relative to other forest types. These results are similar to those obtained for breeding forest birds (see Breeding Forest Songbird section, this report). As we noted in the section of this report dealing with breeding forest songbirds, we did not collect vegetation measurements in the habitats surveyed on the alternative diversion routes, so we are unable to determine if passage-migrant species richness and abundance were affected by vegetation characteristics of the various habitats.

Fall Passage Migrants

We observed few individuals of PROW and NOPA and YTWA after mid-July during our field studies of breeding birds in 2002-2005, and most of these common, breeding warblers were assumed to have left the region by the time we began fall passage-migrant surveys in 2006.

Results of fall passage-migrant surveys were probably affected by differences in observer experience. The technicians hired to assist with fall passage-migrant data

collection, Fred Benham (LSU) and Justin Eby-Bosler (LSU), were highly experienced birders, and they conducted most surveys on Hope Canal and Reserve-Relief Canal. DMF was less experienced, and he conducted all surveys on the Tangipahoa River and Blind River. Therefore, estimates of species richness for the Tangipahoa River are probably low. Comparisons of transects surveyed by foot on Hope Canal and Reserve-Relief Canal and those survey by kayak on the Blind and Tangipahoa Rivers should be interpreted with caution. Transects surveyed from the water were more difficult to conduct because the observer needed to deal with currents, snags and recreational boaters while conducting the survey. The observer was also observing from a sitting position, which may have affected the number of visually detected birds relative to transects surveyed by foot.

Species richness of fall passage-migrants was highest on Hope Canal, and flycatchers and warblers were noticeably more abundant on Hope Canal relative to Reserve-Relief Canal. Many fall passage-migrants were detected in willows (*Salix* spp.), especially on the more northerly of the two transects on the road near Hope Canal, which had pure strips of willows lining the road. Although we did not test effects of habitat type on relative abundance and species richness of fall passage-migrants, there was a noticeable decline in their abundance and species richness on the Tangipahoa River as one proceeded from BLH to swamp habitat.

The area with the densest concentration, and often the greatest species richness of fall passage-migrants was the area near the boat ramp at the south end of Reserve-Relief Canal. We did not include this area in our transect surveys of Reserve-Relief Canal because the vegetation in this area was different from other transects, and because noise from traffic on Airline Highway would have interfered with the survey. The vegetation in this area was dominated by giant ragweed (*Ambrosia trifida*), and Yellow Warblers and Indigo Buntings were highly abundant in this growth.

Effects of Diversion Canal Construction on Passage Migrant Songbirds

Diversion canal construction will probably have few, if any, negative effects on passage-migrant Neotropical birds regardless of which diversion canal route is selected. There are substantial areas of habitat comparable to that which will be destroyed by

diversion canal construction, so the few birds that might have stopped in the construction area can be accommodated elsewhere. Passage-migrant birds are able to adjust their stopover locations for barriers as large as the Great Lakes (Diehl et al. 2003), so the relatively small area of forest cleared for the diversion canal should be easily avoided by passage-migrants that prefer to stop in forested habitat. In the short term, logging debris and disturbed soil may improve foraging habitat for some ground-foraging species, but conversion of tall herbaceous (e.g. giant ragweed) cover to mowed grass will probably reduce passage-migrant use of the area.

Effects of Diversion Canal Operation on Passage Migrant Birds

Most passage-migrant species detected along Hope Canal and Reserve-Relief Canal forage in shrubs or trees, and the most frequently detected migrant species that forage on or near the ground, Louisiana Waterthrush and Northern Waterthrush, are adapted to foraging in riparian habitats. Therefore, diversion canal operation should have little effect on passage-migrant activity, and preferential consideration for the timing of operation may be given to other groups of species.

Recommendations

Forest clearing activities should take place between July and September, after migrant and resident songbirds have finished breeding and before most fall passage-migrant birds begin to arrive, which coincides with recommended timing of construction for Bald Eagles (see Bald Eagle section, this report). Diversion canal operation should have little effect on passage-migrant activity. Individuals of several wading bird species migrate through southeastern Louisiana, and some of these birds will probably use the diversion canal for foraging habitat. Therefore, diversion canal design and operation recommendations for wading bird foraging habitat (see Colonial Nesting Wading Birds section, this report) should take precedence over any considerations for Neotropical passage-migrant songbirds.

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Appendix A. List of bird species detected, their resident and breeding status, and overall relative abundance of forest songbirds in various habitats in Maurepas Swamp in 2002-2006.

Order	Family	Common Name	Species Code	Scientific Name	Resident Status ^a	Breeding Status ^b	Overall relative abundance					
							Degraded	Relict	Throughput	BLH	Mixed	
Anseriformes	Anatidae	Wood Duck	WODU	<i>Aix sponsa</i>	R	Y	NA	NA	NA	NA	NA	
		Mottled Duck	MODU	<i>Anas fulvigula</i>	R	Y	NA	NA	NA	NA	NA	
		Mallard	MADU	<i>Anas platyrhynchos</i>	W	N	NA	NA	NA	NA	NA	
Pelicaniformes	Pelecanidae	Black-bellied Whistling-Duck	BBWD	<i>Dendrocygna autumnalis</i>	R	U	NA	NA	NA	NA	NA	
		Brown Pelican	BRPE	<i>Pelecanus occidentalis</i>	R	N	NA	NA	NA	NA	NA	
		Double-crested Cormorant	DCCO	<i>Phalacrocorax auritus</i>	R	U	NA	NA	NA	NA	NA	
Ciconiiformes	Anhingidae	Anhinga	ANHI	<i>Anhinga anhinga</i>	R	Y	NA	NA	NA	NA	NA	
		Ardeidae	Great Egret	GREG	<i>Ardea alba</i>	R	Y	NA	NA	NA	NA	NA
			Great Blue Heron	GBHE	<i>Ardea herodias</i>	R	Y	NA	NA	NA	NA	NA
American Bittern	AMBI		<i>Botaurus lentiginosus</i>	M	N	NA	NA	NA	NA	NA		
Falconiformes	Cathartidae	Cattle Egret	CAEG	<i>Bubulcus ibis</i>	R	U	NA	NA	NA	NA	NA	
		Green Heron	GRHE	<i>Butorides virescens</i>	R	Y	NA	NA	NA	NA	NA	
		Little Blue Heron	LBHE	<i>Egretta caerulea</i>	M	Y	NA	NA	NA	NA	NA	
		Snowy Egret	SNEG	<i>Egretta thula</i>	R	Y	NA	NA	NA	NA	NA	
		Tricolored Heron	TRHE	<i>Egretta tricolor</i>	M	N	NA	NA	NA	NA	NA	
		Least Bittern	LEBI	<i>Ixobrychus exilis</i>	M	N	NA	NA	NA	NA	NA	
		Yellow-crowned Night-Heron	YCNH	<i>Nyctanassa violacea</i>	M	N	NA	NA	NA	NA	NA	
		Black-crowned Night-Heron	BCNH	<i>Nycticorax nycticorax</i>	R	Y	NA	NA	NA	NA	NA	
		Threskiornithidae	White Ibis	WHIB	<i>Eudocimus albus</i>	M	N	NA	NA	NA	NA	NA
			Roseate Spoonbill	ROSP	<i>Platalea ajaja</i>	R	N	NA	NA	NA	NA	NA
Falconiformes	Accipitridae	Turkey Vulture	TUVU	<i>Cathartes aura</i>	R	Y	NA	NA	NA	NA	NA	
		Black Vulture	BLVU	<i>Coragyps atratus</i>	RM	Y	NA	NA	NA	NA	NA	
		Red-shouldered Hawk	RSHA	<i>Buteo lineatus</i>	R	Y	NA	NA	NA	NA	NA	
		Bald Eagle	BAEA	<i>Haliaeetus leucocephalus</i>	R	Y	NA	NA	NA	NA	NA	
Gruiformes	Rallidae	Mississippi Kite	MIKI	<i>Ictinia mississippiensis</i>	M	U	NA	NA	NA	NA	NA	
		Osprey	OSPR	<i>Pandion haliaetus</i>	R	Y	NA	NA	NA	NA	NA	
		American Coot	AMCO	<i>Fulica americana</i>	W	N	NA	NA	NA	NA	NA	
		Purple Gallinule	PUGA	<i>Porphyrio martinica</i>	M	Y	NA	NA	NA	NA	NA	
Charadriiformes	Scolopacidae	Sora	SORA	<i>Porzana carolina</i>	M	N	NA	NA	NA	NA	NA	
		Virginia Rail	VIRA	<i>Rallus limicola</i>	M	N	NA	NA	NA	NA	NA	
		Killdeer	KILL	<i>Charadrius vociferus</i>	M	U	NA	NA	NA	NA	NA	
Charadriiformes	Recurvirostridae	Black-necked Stilt	BNST	<i>Himantopus mexicanus</i>	M	N	NA	NA	NA	NA	NA	
		Greater Yellowlegs	GRYE	<i>Tringa melanoleuca</i>	M	N	NA	NA	NA	NA	NA	

^a R=resident, M=migrant, W=wintering.

^b: Y=known breeding activity in Maurepas Swamp, N=no breeding activity, U=present in breeding season, but no breeding activity observed.

Appendix A. List of bird species detected, their resident and breeding status, and overall relative abundance of forest songbirds in various habitats in Maurepas Swamp in 2002-2006.

Order	Family	Common Name	Species Code	Scientific Name	Resident Status ^a	Breeding Status ^b	Overall relative abundance				
							Degraded	Relict	Throughput	BLH	Mixed
Charadriiformes	Scolopacidae	Solitary Sandpiper	SOSA	<i>Tringa solitaria</i>	M	N	NA	NA	NA	NA	NA
	Laridae	Laughing Gull	LAGU	<i>Larus atricilla</i>	R	N	NA	NA	NA	NA	NA
		Ring-billed Gull	RBGU	<i>Larus delawarensis</i>	M	N	NA	NA	NA	NA	NA
Columbiformes	Columbidae	Rock Pigeon	ROPI	<i>Columba livia</i>	R	N	NA	NA	NA	NA	NA
		Mourning Dove	MODO	<i>Zenaida macroura</i>	R	N	NA	NA	NA	NA	NA
Cuculiformes	Cuculidae	Yellow-billed Cuckoo	YBCU	<i>Coccyzus americanus</i>	M	N	0.024	0.136	0.080	0.015	0.040
Strigiformes	Strigidae	Great Horned Owl	GHOW	<i>Bubo virginianus</i>	R	Y	NA	NA	NA	NA	NA
		Barred Owl	BAOW	<i>Strix varia</i>	R	Y	NA	NA	NA	NA	NA
Caprimulgiformes	Caprimulgidae	Common Nighthawk	CONI	<i>Chordeiles minor</i>	M	N	NA	NA	NA	NA	NA
Apodiformes	Apodidae	Chimney Swift	CHSW	<i>Chaetura pelagica</i>	M	Y	NA	NA	NA	NA	NA
	Trochilidae	Ruby-throated Hummingbird	RTHU	<i>Archilochus colubris</i>	M	Y	0.000	0.049	0.077	0.231	0.211
Coraciiformes	Alcedinidae	Belted Kingfisher	BEKI	<i>Ceryle alcyon</i>	W	N	NA	NA	NA	NA	NA
Piciformes	Picidae	Northern Flicker	NOFL	<i>Colaptes auratus</i>	R	N	0.008	0.000	0.000	0.000	0.000
		Pileated Woodpecker	PIWO	<i>Dryocopus pileatus</i>	R	Y	0.020	0.073	0.147	0.067	0.113
		Red-bellied Woodpecker	RBWO	<i>Melanerpes carolinus</i>	R	Y	0.297	0.736	1.021	0.353	0.380
		Red-headed Woodpecker	RHOW	<i>Melanerpes erythrocephalus</i>	R	N	NA	NA	NA	NA	NA
		Downy Woodpecker	DOWO	<i>Picoides pubescens</i>	R	Y	0.041	0.213	0.387	0.216	0.133
		Hairy Woodpecker	HAWO	<i>Picoides villosus</i>	R	Y	0.000	0.008	0.000	0.115	0.075
		Eastern Wood-Pewee	EAWP	<i>Contopus virens</i>	M	N	NA	NA	NA	NA	NA
Passeriformes	Tyrannidae	Acadian Flycatcher	ACFL	<i>Empidonax virescens</i>	M	U	0.000	0.009	0.036	0.000	0.000
		Great Crested Flycatcher	GCFL	<i>Myiarchus crinitus</i>	R	Y	0.024	0.127	0.153	0.030	0.118
		Eastern Kingbird	EAKI	<i>Tyrannus tyrannus</i>	M	Y	0.073	0.000	0.000	0.000	0.000
	Vireonidae	Yellow-throated Vireo	YTVI	<i>Vireo flavifrons</i>	RM	U	0.008	0.053	0.054	0.100	0.038
		Warbling Vireo	WAVI	<i>Vireo gilvus</i>	M	N	NA	NA	NA	NA	NA
		White-eyed Vireo	WEVI	<i>Vireo griseus</i>	R	Y	0.016	0.148	0.103	0.280	0.422
		Red-eyed Vireo	REVI	<i>Vireo olivaceus</i>	M	Y	0.000	0.030	0.279	0.152	0.098
		Blue-headed Vireo	BHVI	<i>Vireo solitarius</i>	RW	N	0.000	0.000	0.005	0.000	0.000
		American Crow	AMCR	<i>Corvus brachyrhynchos</i>	R	Y	0.008	0.123	0.483	0.104	0.078
	Corvidae	Fish Crow	FICR	<i>Corvus ossifragus</i>	R	Y	NA	NA	NA	NA	NA
		Blue Jay	BLJA	<i>Cyanocitta cristata</i>	R	U	0.000	0.009	0.000	0.000	0.000
	Hirundinidae	Barn Swallow	BARS	<i>Hirundo rustica</i>	M	Y	NA	NA	NA	NA	NA
		Purple Martin	PUMA	<i>Progne subis</i>	M	N	NA	NA	NA	NA	NA
Tree Swallow		TRSW	<i>Tachycineta bicolor</i>	M	N	NA	NA	NA	NA	NA	
Paridae	Tufted Titmouse	TUTI	<i>Baeolophus bicolor</i>	RM	Y	0.134	0.533	0.618	0.409	0.649	
	Carolina Chickadee	CACH	<i>Poecile carolinensis</i>	R	Y	0.285	1.231	1.665	1.297	1.091	
		Winter Wren	MAWR	<i>Cistothorus palustris</i>	W	N	NA	NA	NA	NA	NA

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Order	Family	Common Name	Species Code	Scientific Name	Resident Status ^a	Breeding Status ^b	Overall relative abundance				
							Degraded	Relict	Throughput	BLH	Mixed
Passeriformes	Troglodytidae	Sedge Wren	SEWR	<i>Cistothorus platensis</i>	W	N	NA	NA	NA	NA	NA
		Carolina Wren	CARW	<i>Thryothorus ludovicianus</i>	R	Y	0.264	1.022	0.646	0.686	0.716
	Regulidae	Ruby-crowned Kinglet	RCKI	<i>Regulus calendula</i>	W	N	NA	NA	NA	NA	NA
	Sylviidae	Blue-gray Gnatcatcher	BGGN	<i>Poliptila caerulea</i>	RW	Y	0.033	0.307	0.302	0.834	0.876
	Turdidae	Veery	VEER	<i>Catharus fuscescens</i>	M	N	NA	NA	NA	NA	NA
		Wood Thrush	WOTH	<i>Hylocichla mustelina</i>	M	N	NA	NA	NA	NA	NA
		Eastern Bluebird	EABL	<i>Sialia sialis</i>	R	Y	0.000	0.014	0.000	0.000	0.000
	Mimidae	Gray Catbird	GRCA	<i>Dumetella carolinensis</i>	M	N	NA	NA	NA	NA	NA
		Brown Thrasher	BRTH	<i>Toxostoma rufum</i>	RM	N	NA	NA	NA	NA	NA
	Bombycillidae	Cedar Waxwing	CEDW	<i>Bombycilla cedrorum</i>	W	N	NA	NA	NA	NA	NA
	Parulidae	Yellow-rumped Warbler	YRWA	<i>Dendroica coronata</i>	M	Y	NA	NA	NA	NA	NA
		Yellow-throated Warbler	YTWA	<i>Dendroica dominica</i>	M	Y	0.398	0.655	0.999	0.033	0.471
		Magnolia Warbler	MAWA	<i>Dendroica magnolia</i>	M	N	NA	NA	NA	NA	NA
		Chestnut-sided Warbler	CSWA	<i>Dendroica pensylvanica</i>	M	N	NA	NA	NA	NA	NA
		Yellow Warbler	YEWA	<i>Dendroica petechia</i>	M	N	NA	NA	NA	NA	NA
		Black-throated Green Warbler	BTNW	<i>Dendroica virens</i>	M	N	NA	NA	NA	NA	NA
		Common Yellowthroat	COYE	<i>Geothlypis trichas</i>	RW	U	0.000	0.005	0.012	0.017	0.038
		Yellow-breasted Chat	YBCH	<i>Icteria virens</i>	M	U	NA	NA	NA	NA	NA
		Black-and-white Warbler	BAWW	<i>Mniotilta varia</i>	M	N	NA	NA	NA	NA	NA
		Northern Parula	NOPA	<i>Parula americana</i>	M	Y	0.512	2.515	2.813	1.780	3.082
		Prothonotary Warbler	PROW	<i>Protonotaria citrea</i>	M	Y	0.362	3.404	2.676	1.651	2.305
		Ovenbird	OVEN	<i>Seiurus aurocapilla</i>	M	N	NA	NA	NA	NA	NA
		Louisiana Waterthrush	LOWT	<i>Seiurus motacilla</i>	M	N	NA	NA	NA	NA	NA
		Northern Waterthrush	NOWA	<i>Seiurus noveboracensis</i>	M	N	NA	NA	NA	NA	NA
		Orange-crowned Warbler	OCWA	<i>Vermivora celata</i>	MW	N	NA	NA	NA	NA	NA
		Tennessee Warbler	TEWA	<i>Vermivora peregrina</i>	W	N	NA	NA	NA	NA	NA
		Blue-winged Warbler	BWWA	<i>Vermivora pinus</i>	M	N	NA	NA	NA	NA	NA
		Hooded Warbler	HOWA	<i>Wilsonia citrina</i>	M	N	NA	NA	NA	NA	NA
	Thraupidae	Scarlet Tanager	SCTA	<i>Piranga olivacea</i>	M	N	NA	NA	NA	NA	NA
		Summer Tanager	SUTA	<i>Piranga rubra</i>	M	N	NA	NA	NA	NA	NA
	Emberizidae	Swamp Sparrow	SWSP	<i>Melospiza georgiana</i>	MW	N	NA	NA	NA	NA	NA
		White-throated Sparrow	WTSP	<i>Zonotrichia albicollis</i>	RM	Y	NA	NA	NA	NA	NA
	Cardinalidae	Northern Cardinal	NOCA	<i>Cardinalis cardinalis</i>	R	Y	0.484	0.522	0.508	1.074	1.431
		Blue Grosbeak	BLGR	<i>Passerina caerulea</i>	M	N	NA	NA	NA	NA	NA
		Painted Bunting	PABU	<i>Passerina ciris</i>	M	U	NA	NA	NA	NA	NA
		Indigo Bunting	INBU	<i>Passerina cyanea</i>	M	U	0.008	0.213	0.197	0.117	0.431
	Icteridae	Rose-breasted Grosbeak	RBGR	<i>Pheucticus ludovicianus</i>	M	N	NA	NA	NA	NA	NA
		Red-winged Blackbird	RWBL	<i>Agelaius phoeniceus</i>	R	Y	6.500	0.259	0.272	0.052	0.173
		Bobolink	BOBO	<i>Dolichonyx oryzivorus</i>	M	N	NA	NA	NA	NA	NA
		Indigo Oriole	BAOR	<i>Icterus galbula</i>	M	N	NA	NA	NA	NA	NA

^a R=resident, M=migrant, W=wintering

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							Degraded	Relict	Throughput	BLH	Mixed
Passeriformes	Icteridae	Orchard Oriole	OROR	<i>Icterus spurius</i>	M	N	NA	NA	NA	NA	NA
		Brown-headed Cowbird	BHCO	<i>Molothrus ater</i>	R	Y	0.008	0.059	0.032	0.109	0.209
		Boat-tailed Grackle	BTGR	<i>Quiscalus major</i>	R	Y	0.325	0.014	0.005	0.000	0.000
		Common Grackle	COGR	<i>Quiscalus quiscula</i>	R	Y	0.413	0.131	0.170	0.162	0.131
	Fringillidae	American Goldfinch	AMGO	<i>Carduelis tristis</i>	W	N	NA	NA	NA	NA	NA

^a R=resident, M=migrant, W=wintering.

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Appendix B. List of bird species detected, their resident and breeding status, and overall relative abundance of forest songbirds in various habitats in Maurepas Swamp in 2002-2006.

Appendix B. Point count sampling locations, habitats, and years in which points were sampled.

POINT	LAT/LONG	HABITAT	YEAR SAMPLED				
			2002	2003	2004	2005	2006
EIS-H1	N30 04.839 W90 37.553	Bottomland Hardwood					X
EIS-H2	N30 05.107 W90 37.602	Bottomland Hardwood					X
EIS-H3	N30 05.366 W90 37.678	Bottomland Hardwood					X
EIS-H4	N30 05.632 W90 37.756	Mixed Forest					X
EIS-H5	N30 05.890 W90 37.815	Mixed Forest					X
EIS-H6	N30 06.139 W90 37.963	Throughput					X
EIS-H7	N30 06.269 W90 38.206	Throughput					X
EIS-H8	N30 06.465 W90 38.418	Throughput					X
EIS-H9	N30 06.732 W90 38.424	Throughput					X
EIS-R1	N30 04.920 W90 32.771	Bottomland Hardwood					X
EIS-R2	N30 05.171 W90 32.777	Bottomland Hardwood					X
EIS-R3	N30 05.373 W90 32.781	Bottomland Hardwood					X
EIS-R4	N30 05.640 W90 32.786	Mixed Forest					X
EIS-R5	N30 05.915 W90 32.791	Mixed Forest					X
EIS-R6	N30 06.182 W90 32.794	Mixed Forest					X
MP-G1	N30 16.513 W90 23.392	Degraded			X		
MP-G2	N30 16.418 W90 23.278	Degraded			X		
MP-G3	N30 16.443 W90 23.125	Degraded			X		
MP-G4	N30 16.570 W90 23.077	Degraded			X		
MP-G5	N30 16.621 W90 23.218	Degraded			X		
MP-HC1	N30 07.748 W90 38.373	Throughput			X		
MP-HC2	N30 07.884 W90 38.352	Throughput			X		
MP-HC3	N30 08.023 W90 38.374	Throughput			X		
MP-HC4	N30 08.193 W90 38.375	Throughput			X		
MP-HC5	N30 08.342 W90 38.371	Throughput			X		
MP-HC6	N30 08.489 W90 38.386	Throughput			X		
MP-HC7	N30 08.631 W90 38.384	Throughput			X		
MP-HC8	N30 09.000 W90 38.380	Throughput			X		
MP-HC9	N30 09.175 W90 38.392	Throughput			X		
MP-JI2	N30 17.995 W90 24.677	Degraded			X		
MP-JI3	N30 17.904 W90 24.799	Degraded			X		
MP-JI5	N30 17.935 W90 25.119	Degraded			X		
MP-JI6	N30 18.061 W90 25.043	Degraded			X		
MP-JI7	N30 18.178 W90 24.954	Degraded			X		
MP-R1	N30 12.549 W90 26.101	Degraded			X		
MP-R2	N30 12.456 W90 25.991	Degraded			X		
MP-R3	N30 12.337 W90 26.074	Degraded			X		
MP-R4	N30 12.420 W90 26.197	Degraded			X		
MP-R5	N30 12.542 W90 26.268	Degraded			X		
MP-RR1	N30 09.634 W90 38.484	Throughput			X		
MP-RR2	N30 09.786 W90 38.485	Throughput			X		
MP-RR3	N30 09.946 W90 38.484	Throughput			X		
MP-RR4	N30 10.092 W90 38.487	Throughput			X		
MP-RR5	N30 10.243 W90 38.500	Throughput			X		
MP-RR6	N30 10.398 W90 38.495	Throughput			X		
MH-3	N30 10.651 W90 36.622	Relict Swamp			X	X	
MH-6	N30 10.696 W90 36.430	Relict Swamp			X	X	
MH-9	N30 10.685 W90 36.223	Relict Swamp			X	X	

Appendix B. List of bird species detected, their resident and breeding status, and overall relative abundance of forest songbirds in various habitats in Maurepas Swamp in 2002-2006.

Appendix B. Point count sampling locations, habitats, and years in which points were sampled.

POINT	LAT/LONG	HABITAT	YEAR SAMPLED				
			2002	2003	2004	2005	2006
MH-10	N30 10.616 W90 36.195	Relict Swamp			X	X	
MH-11	N30 10.503 W90 36.211	Relict Swamp			X	X	
MH-14	N30 10.643 W90 36.321	Relict Swamp				X	
MH-16	N30 10.631 W90 36.483	Relict Swamp			X	X	
MH-20	N30 10.474 W90 36.736	Relict Swamp			X	X	
MH-22	N30 10.628 W90 36.782	Relict Swamp			X		
MH-23	N30 10.887 W90 36.618	Relict Swamp			X	X	
MH-25	N30 10.776 W90 36.695	Relict Swamp				X	
MH-26	N30 10.561 W90 36.918	Relict Swamp			X	X	
MH-28	N30 10.621 W90 37.025	Relict Swamp				X	
MH-30	N30 10.703 W90 36.943	Relict Swamp			X		
MH-A1	N30 12.136 W90 36.238	Relict Swamp	X				
MH-A2	N30 11.830 W90 36.203	Relict Swamp	X				
MH-A3	N30 11.644 W90 36.035	Relict Swamp	X				
MH-A4	N30 11.557 W90 36.283	Relict Swamp	X				
MH-A5	N30 11.163 W90 36.599	Relict Swamp	X				
MH-A6	N30 10.872 W90 36.595	Relict Swamp	X				
MH-A7	N30 10.635 W90 36.787	Relict Swamp	X				
MH-A8	N30 10.498 W90 36.975	Relict Swamp	X				
MH-A9	N30 10.327 W90 37.077	Relict Swamp	X				
MH-A10	N30 10.334 W90 37.422	Relict Swamp	X				
MH-A11	N30 09.953 W90 37.562	Relict Swamp	X				
MH-A12	N30 09.632 W90 37.903	Relict Swamp	X				
MH-A13	N30 10.663 W90 36.541	Relict Swamp	X				
MH-A14	N30 10.712 W90 36.322	Relict Swamp	X				
MH-A15	N30 10.501 W90 36.240	Relict Swamp	X				
MH-A16	N30 10.293 W90 36.353	Relict Swamp	X				
MH-A17	N30 10.002 W90 36.475	Relict Swamp	X				
MH-A18	N30 09.844 W90 36.632	Relict Swamp	X				
MH-A19	N30 09.538 W90 36.650	Relict Swamp	X				
MH-A20	N30 09.254 W90 36.495	Relict Swamp	X				
MH-A21	N30 09.027 W90 36.417	Relict Swamp	X				
MH-A22	N30 08.782 W90 36.225	Relict Swamp	X				
MH-A23	N30 08.676 W90 35.854	Relict Swamp	X				
MH-A24	N30 08.778 W90 35.566	Relict Swamp	X				
SH-04A	N30 14.229 W90 39.055	Relict Swamp	X	X			
SH-04B	N30 14.088 W90 38.868	Relict Swamp	X	X			
SH-05A	N30 13.562 W90 38.452	Relict Swamp	X	X			
SH-05B	N30 13.380 W90 38.390	Relict Swamp	X	X			
SH-08A	N30 11.311 W90 36.522	Relict Swamp	X	X			
SH-08B	N30 11.501 W90 36.410	Relict Swamp	X	X			
SH-10A	N30 09.550 W90 38.324	Throughput	X	X			
SH-10B	N30 09.531 W90 38.492	Throughput	X	X			
SH-10C	N30 10.178 W90 37.617	Relict Swamp	X	X			
SH-11A	N30 07.985 W90 38.373	Throughput	X	X			
SH-11B	N30 08.158 W90 38.508	Throughput	X	X			
SH-11C	N30 08.442 W90 38.329	Throughput	X	X			
SH-12A	N30 05.891 W90 37.774	Throughput	X	X			

^a R=resident, M=migrant, W=wintering.

^b: Y=known breeding activity in Maurepas Swamp, N=no breeding activity, U=present in breeding season, but no breeding activity observed.

Appendix B. List of bird species detected, their resident and breeding status, and overall relative abundance of forest songbirds in various habitats in Maurepas Swamp in 2002-2006.

Appendix B. Point count sampling locations, habitats, and years in which points were sampled.

POINT	LAT/LONG	HABITAT	YEAR SAMPLED				
			2002	2003	2004	2005	2006
SH-12B	N30 06.130 W90 37.854	Throughput	X	X			
SH-14A	N30 07.021 W90 32.899	Relict Swamp	X	X			
SH-14B	N30 07.242 W90 32.708	Relict Swamp	X	X			
SH-15B	N30 07.803 W90 34.954	Relict Swamp	X	X			
SH-15C	N30 07.899 W90 34.757	Relict Swamp	X	X			
SH-16A	N30 07.962 W90 32.024	Relict Swamp	X	X			
SH-16B	N30 08.010 W90 32.242	Relict Swamp	X	X			
SH-17A	N30 09.548 W90 33.688	Degraded	X	X			
SH-17A2	N30 09.516 W90 33.201	Degraded	X	X			
SH-17B	N30 09.655 W90 33.492	Degraded	X	X			
SH-17B2	N30 09.727 W90 33.714	Degraded	X	X			
SH-18A	N30 09.921 W90 28.643	Degraded	X	X			
SH-18B	N30 09.703 W90 28.683	Degraded	X	X			
SH-19B	N30 12.693 W90 25.972	Degraded	X	X			
SH-19C	N30 12.568 W90 25.776	Degraded	X	X			
SH-20A	N30 18.077 W90 24.620	Degraded	X	X			
SH-20B	N30 17.938 W90 24.717	Degraded	X	X			
SH-20C	N30 18.085 W90 24.497	Degraded	X	X			
SH-21A	N30 11.835 W90 38.505	Throughput	X	X			
SH-21B	N30 11.176 W90 38.408	Throughput	X	X			
SH-22A	N30 16.474 W90 23.505	Degraded	X	X			
SH-22B	N30 16.638 W90 23.723	Degraded	X	X			

^a R=resident, M=migrant, W=wintering.

^b: Y=known breeding activity in Maurepas Swamp, N=no breeding activity, U=present in breeding season, but no breeding activity observed.