Effects of structural marsh management and winter burning on plant and bird communities during summer in the Gulf Coast Chenier Plain

Steven W. Gabrey, Alan D. Afton, and Barry C. Wilson

Abstract  Coastal wetlands in the Gulf Coast Chenier Plain commonly are managed intensively by fall or winter burning and constructing impoundments to improve habitat for wintering waterfowl, reduce wetland loss, and create emergent wetlands. However, little information is available on effects of these management practices on plant or bird communities during summer. We conducted experimental burns in 4 types of impounded and unimpounded marshes on Rockefeller State Wildlife Refuge in southwestern Louisiana. We recorded vegetation characteristics and species composition and relative abundance of plants and birds during April-June 1996-1998. We found that vegetation characteristics in burned marshes did not differ from those of unburned marshes by the first summer post-burn and that winter burning did not affect bird species richness or species composition. Birds/survey for all species combined and for sparrows (primarily seaside sparrows Ammodramus maritimus) did not differ between burned and unburned marshes during the first or third summers post-burn, but were 2 times greater in burned than in unburned marshes during the second summer post-burn. Our results indicate that winter burning for waterfowl is compatible with management for other marsh birds, provided that measures ensuring sufficient winter cover for passerines are included in management plans. Number of icterids/survey was greatest and sparrows/survey was least in intermediate impounded marshes compared to other marsh-management types. Sparrows generally were most abundant in brackish and saline unimpounded marshes, indicating that continued loss of unimpounded marsh habitat could impact coastal sparrow populations.

Key words  bird community, burning, coastal marshes, Gulf Coast Chenier Plain, habitat, impoundment, Louisiana, passerines, plant community, structural marsh management

The Chenier Plain of the Gulf of Mexico encompasses 1,295 km² of coastal marsh from Vermilion Bay, Louisiana, to East Bay, Texas (Gosselink 1979). Throughout the Gulf Coast, quantity and quality of coastal marshes have declined because of natural (e.g., land subsidence, sea level rise, storms) and human-induced factors (e.g., commercial and residential development, altered hydrology; Boesch et al. 1983; Mitsch and Gosselink 1993). Private and public landowners manage these wetlands intensively, primarily through controlled burning (usually during fall or winter) and structural marsh management (i.e., constructing impoundments, weirs, and other water-control structures) to 1) improve...
quality of existing marshes as habitat for waterfowl, furbearers, and other wildlife; 2) reduce wetland erosion; and 3) create emergent wetlands (Cowan et al. 1988, Chabreck et al. 1989, Nyman and Chabreck 1995).

Current marsh management practices are controversial because of uncertainty of results, unanticipated effects on nontarget plants or animals, and shifts in priorities from winter waterfowl and furbearer management to maintaining or creating marsh and restoring natural hydrologic processes (Cowan et al. 1988, Nyman and Chabreck 1995, United States Environmental Protection Agency 1998). Information regarding effects of current management practices is necessary to address these concerns; however, few such studies have been published. In this paper, we investigated effects of winter burning and structural marsh management on vegetation structure and plant and bird communities during summer (April-June) in the Gulf Coast Chenier Plain in southwestern Louisiana.

Plant and bird communities are affected by fire and may continue to change for some time afterward (Bendell 1974, Whelan 1995). Because of the relationship between bird abundance and vegetation structure (Cody 1968, Delisle and Savidge 1997), recovery of bird communities to pre-burn conditions should closely track post-burn succession in the plant community (Huff et al. 1984, Whelan 1995). Post-burn recovery of coastal marsh plant communities that are dominated by a few graminoid species is rapid (1-2 years post-burn, Mendelssohn et al. 1995, Gabrey et al. 1999). Accordingly, post-burn recovery of winter bird communities also is rapid (Gabrey et al. 1999). Thus, we predicted that summer plant and bird community differences would be greatest during the first summer post-burn but negligible during the second and third summers post-burn.

Impoundment construction alters marsh hydrology and salinity, which in turn influences interspersion of open water and emergent vegetation, plant species composition, and plant community diversity (Chabreck and Junkin 1989). We showed previously that vegetation characters (visual obstruction and percentage cover) and bird communities (species richness and number of birds/survey) differed between impounded and unimpounded marshes during winter (Gabrey et al. 1999). Therefore, we expected that summer vegetation characteristics would differ between impounded and unimpounded marshes and that such differences would result in differences in bird communities.

Methods

Study area

We chose Rockefeller State Wildlife Refuge (Rockefeller SWR) as a representative area of the Chenier Plain of southwestern Louisiana and southeastern Texas (Figure 1). This 30,700-ha refuge, managed by the Louisiana Department of Wildlife and Fisheries, consists of 17 impoundments from 200 to >4,000 ha (Wicker et al. 1983) and approximately 11,700 ha of tidally influenced unimpounded marshes. Most impoundments were constructed during the late 1950s and are separated by a network of canals. Management burns are conducted
Brackish unimpounded marsh on 8 February 1996, about one month after burning. Height of Spartina patens regrowth is approximately 2.5 cm and total vegetation percentage cover is less than 10%. Herbivore enclosure at left is 1.5 m x 1.5 m x 1.5 m on a 3-year rotation, with approximately one-third of the refuge area burned during a single fall or winter (October-February). Lightning-ignited fires also occur on Rockefeller SWR, usually from June to August (0–6 fires/year during 1993–1998; T. J. Hess, Louisiana Department of Wildlife and Fisheries, unpublished data).

Marsh types on Rockefeller SWR ranged from a band of saline marsh along the Gulf Coast to a band of brackish marsh farther inland and to intermediate marsh still farther inland (Chabreck 1970, Chabreck and Linscombe 1988). Saline marsh (salinity >10 ppt) was dominated by Spartina alterniflora, S. patens, and Distichlis spicata. Brackish marsh (5–10 ppt) was characterized by S. patens, D. spicata, and Scirpus spp. Intermediate marsh (1–5 ppt) was dominated by Spartina patens (Chabreck 1970, Chabreck 1972, Chabreck and Linscombe 1988). Impounded marshes in our study were intermediate or brackish; unimpounded marshes, exposed to Gulf tides, were brackish or saline. Intermediate unimpounded and saline impounded marshes were not present on Rockefeller SWR.

Using vegetation-type and fire-history maps of Rockefeller SWR, we selected 14 study marshes (Figure 1) that 1) were ≥100 ha of emergent vegetation with little open water, 2) had a firebreak (bayou or canal), 3) had a homogeneous marsh type and fire history, 4) were accessible, and 5) were absent of other ongoing research projects or physical structures that could be damaged by fire. Three impounded marshes were brackish and 5 were intermediate. Three unimpounded marshes were brackish and 3 were saline. We conducted experimental burns on 9–11 and 13 December 1995 and 9 January 1996. Unintentional fires in March and September 1996, caused by neighboring landowners or other research crews, burned 3 impounded (2 intermediate, 1 brackish) marshes; consequently, we did not collect data from these sites. A lightning fire in July 1997 burned all of one unburned impounded marsh; we included this marsh in all analyses because this fire was a natural occurrence.

We used a split-plot experimental design (Sokal and Rohlf 1995), in which a firebreak bisected each ≥100-ha study marsh (the whole-plot) into 2 split-plots of ≥50 ha. Using a gridded United States Geological Survey topographic map to randomly select locations, we placed a 100-m x 100-m quadrat in each split-plot and randomly assigned burn treatment to one side of each firebreak. We located quadrats ≥50 m from a levee, bayou, or other habitat edge when possible. Distance between paired quadrats ranged from 0.5 to 3.5 km and was less than 1 km for all but 2 pairs. We collected vegetation data at 40 points spaced at 10-m intervals around the perimeter of each quadrat. We conducted bird surveys in a 0.4-ha (100-m x 40-m) subplot of each quadrat. We marked the 0.4-ha bird survey subplots with conduit pipe at 20-m intervals along the longer centerline.

Vegetation characteristics

We collected vegetation data annually (1996–1998) at each quadrat during the last week of May through the first week of June. At each of 40 points in a quadrat, we estimated visual obstruction by recording the lowest visible line of a 3-m pole marked at 0.1-m intervals from a distance of 4 m...
We determined percentage vegetation cover (total vegetation percentage cover and individual species percentage cover) at each point by laying a 1-m pole marked at 0.1-m intervals on the ground and determining percentage of the pole covered (Chabreck et al. 1985). Cover scores were 7 (76–100%), 6 (51–75%), 5 (26–50%), 4 (6–25%), 3 (1–5%), 2 (few stems), 1 (single stem), and 0 (absent, Mueller-Dambois and Ellenberg 1974). To calculate mean values for these categorical data, we converted cover classes to discrete responses using the midpoint of the class (i.e., Class 7 = 87.5%, Class 6 = 62.5, Class 5 = 37.5, Class 4 = 15, Class 3 = 2.5, and Classes 1 and 2 = 0.5 [Agresti 1996, Pahl et al. 1997]). We categorized all rooted dead vegetation as a single species because of difficulties in identifying dead material. Points located in a pond or on unvegetated mud were given visual obstruction and cover scores of 0. We considered a plant species abundant if its overall percentage cover score exceeded 10%.

**Bird surveys**

We surveyed birds 6 times (2 surveys/month) at each 0.4-ha bird survey subplot during each summer (April-June) of 1996-1998. To survey secretive species, we played tapes (30 sec of calls followed by a 30-sec listening period) of 5 species (Peterson field guide series, Houghton Mifflin Co., Boston, Mass.): king rail (Rallus elegans), clapper rail (R. longirostris), least bittern (Ixobrychus exilis), American bittern (Botaurus lentiginosus), and black rail (Laterallus jamaicensis, Gibbs and Melvin 1995). Tapes could be heard at 100 m by humans. We recorded all birds that responded from within the 0.4-ha bird survey subplot. After the 5-minute tape playback period, the observer slowly walked through the subplot 4 times, beginning 15 m north of the center line, then 5 m north, 5 m south, and 15 m south of the center line, and recorded species and numbers of birds seen or heard within the subplot boundaries. We used this method because many species, including rails and most passerines, often did not flush unless the observer approached within 5 m. We carefully noted location and flush direction of each bird to minimize repeated counting of individuals. Surveys averaged 16 minutes in duration (range = 10–23 min, SD = 2, n = 388). We defined a bird species as common if more than 15 individuals were recorded over the 3 summers.

We completed most (>95%) bird surveys by 1100 h, but weather conditions and logistic constraints required that some surveys be conducted later; all were completed by 1500 h. Surveys were not conducted when wind was >20 km/h or in fog or steady rain.

**Statistical analysis**

We defined plant and bird species richness as the total number of species recorded in a quadrat or subplot during a summer. For each quadrat, we averaged each remaining vegetation response variable (visual obstruction score, total vegetation percentage cover, and percentage cover for each plant species) over the 40 points to give 1 value/quadrat/summer for each variable. Similarly, we averaged number of birds/survey for all species combined and for 3 species groups (icterids, sparrows, and wrens) over the 6 surveys to give 1 value/subplot/summer for each variable. Icterids included red-winged blackbirds (Agelaius phoeniceus), boat-tailed grackles (Quiscalus major), eastern meadowlarks (Sturnella magna), and orchard orioles (Icterus spurius). Sparrows included seaside (Ammomimus maritimus), Nelson’s sharp-tailed (A. nelsoni), swamp (Melospiza georgiana), and unidentified. Wrens included marsh (Cistothorus palustris), sedge (C. platensis), and unidentified. Mean visual obstruction scores and number of birds/survey were log-transformed before analysis (log_{10}[Y+1], Sokal and Rohlf 1995) to meet assumptions for parametric procedures; means and confidence intervals reported here are back-transformed values.

Marsh type (brackish, intermediate, or saline) was confounded with management type (impounded or unimpounded) in our experiment because available impounded marshes were either intermediate or brackish, whereas available unimpounded marshes were either brackish or saline. To separate marsh type effects from management type effects, we defined the class variable marsh-management type (M-M TYPE) to include the 4 combinations of marsh type and management type (brackish impounded, intermediate impounded, brackish unimpounded, saline unimpounded). We used *a priori* contrasts to compare response variables among the marsh-management types.

Because our data consisted of multiple response variables taken from a single experimental unit (the quadrat), we first conducted multivariate analyses of variance (MANOVA, PROC GLM, SAS Institute 1990) for vegetation responses and bird responses...
separately. Our model included M-M TYPE as the whole-plot effect, QUADRAT (the 11 pairs of quadrats), burn treatment (BURN, burned or unburned) as the split-plot effect, and summer (SUMMER, 1996, 1997, or 1998) as a repeated measure. We tested the M-M TYPE main effect with the QUADRAT(M-M TYPE) mean square, SUMMER and SUMMER × M-M TYPE effects with the SUMMER × QUADRAT(M-M TYPE) mean square, and all effects containing BURN with the residual mean square (see Gabrey 1999 for details). Both overall MANOVA models were significant ($P<0.05$); consequently, we conducted univariate analyses of variance (ANOVA) for each response variable using models similar to those described for MANOVAs.

When a significant M-M TYPE main effect or interaction involving M-M TYPE was detected in an ANOVA, we used orthogonal contrasts to determine effects of marsh type (intermediate impounded vs. brackish impounded, brackish unimpounded vs. saline unimpounded) and management type (brackish impounded vs. brackish unimpounded). When a significant burn treatment or summer main effect, or any interaction not involving M-M TYPE, was detected in an ANOVA, we conducted pairwise comparisons using the PDIFF option in the LSMEANS statement (PROC GLM, SAS Institute 1990).

Because of low water levels, we were able to conduct only 5 bird surveys in 2 subplot pairs in 1996 and 2 pairs in 1998. However, 5 surveys were sufficient to detect all bird species in the 18 subplots in which all 6 surveys were conducted in 1996; in 1998, 5 surveys were sufficient to record all species in 14 of 18 subplots in which 6 surveys were conducted. Therefore, we are confident that all species likely to be encountered were recorded in those subplots that were surveyed only 5 times; thus, all subplots were included for analysis.

**Results**

**Vegetation characteristics**

Of 28 plant species identified in our study marshes during the 3 summers, dead vegetation, *Spartina patens*, and *Distichlis spicata* were the most abundant, each with a mean percentage cover score $>10\%$ (Table 1).

We detected significant SUMMER × M-M TYPE ($F_{5,14}=8.60, P<0.01$) and BURN × M-M TYPE ($F_{3,29} = 3.00, P=0.05$) interactions in the analysis of plant species richness. In impounded marshes in 1996, plant species richness was greater in intermediate than in brackish marshes (Table 2). Species richness did not differ between marsh types within unimpounded marshes or between management types within brackish marshes in 1995 (Table 2). In 1997, plant species richness did not differ for any M-M TYPE contrast (Table 2). In unimpounded

<table>
<thead>
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<th>Year</th>
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<th>Unburned</th>
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<tbody>
<tr>
<td></td>
<td>BI</td>
<td>II</td>
</tr>
<tr>
<td>1996</td>
<td>Dead vegetation</td>
<td>51 (15)</td>
</tr>
<tr>
<td></td>
<td><em>Spartina patens</em></td>
<td>49 (33)</td>
</tr>
<tr>
<td></td>
<td><em>Distichlis spicata</em></td>
<td>21 (29)</td>
</tr>
<tr>
<td>1997</td>
<td>Dead vegetation</td>
<td>63 (2)</td>
</tr>
<tr>
<td></td>
<td><em>Spartina patens</em></td>
<td>45 (29)</td>
</tr>
<tr>
<td></td>
<td><em>Distichlis spicata</em></td>
<td>25 (35)</td>
</tr>
<tr>
<td>1998</td>
<td>Dead vegetation</td>
<td>83 (4)</td>
</tr>
<tr>
<td></td>
<td><em>Spartina patens</em></td>
<td>59 (27)</td>
</tr>
<tr>
<td></td>
<td><em>Distichlis spicata</em></td>
<td>18 (26)</td>
</tr>
</tbody>
</table>


b BI = brackish impounded marsh, II = intermediate impounded marsh, BU = brackish unimpounded marsh, SU = saline unimpounded marsh.
marshes in 1998, species richness was greater in saline than in brackish marshes. Species richness did not differ between marsh types within impounded marshes or between management types within brackish marshes in 1998 (Table 2). In intermediate impounded marshes, species richness was greater in burned than in unburned marshes; burning had no effect on species richness for the other 3 marsh-management types (Figure 2).

We detected a significant SUMMER effect for visual obstruction ($F_{2,20} = 5.00, P = 0.02$). Visual obstruction scores were less ($P < 0.05$) in 1996 (back-transformed mean = 6.3, 95% CI = 5.5-7.1) than in either 1997 (back-transformed mean = 7.5, 95% CI = 6.5-8.4) or 1998 (back-transformed mean = 8.2, 95% CI = 7.2-9.3). The mean visual obstruction score in 1997 did not differ from that in 1998 ($P > 0.05$).

We detected a significant SUMMER × M-M TYPE interaction in the analysis of total vegetation percentage cover ($F_{6,14} = 4.08, P = 0.01$). In impounded marshes, total vegetation cover was greater in brackish than in intermediate marshes during 1997 and 1998 but not in 1996 (Table 2). In unimpounded marshes, total vegetation cover was greater in saline than in brackish marshes in 1996 but did not differ between marsh types in 1997 or 1998 (Table 2). In brackish marshes, total vegetation cover was greater in impounded than in unimpounded marshes in 1996, but did not differ between management types in 1997 or 1998 (Table 2).

We detected a significant BURN × SUMMER interaction in the analysis of dead vegetation percentage cover ($F_{2,30} = 4.35, P = 0.02$). Dead vegetation cover in burned marshes was less in 1996 than in either 1997 or 1998, in unburned marshes dead vegetation cover did not differ among years (Figure 3). In 1996, dead vegetation cover was less in burned than in unburned marshes but did not differ between burn treatments in either 1997 or 1998 (Figure 3).

We detected significant BURN × M-M TYPE ($F_{3,29} = 5.05, P < 0.01$) and SUM-
Figure 2. Plant species richness (top), Spartina patens percentage cover (middle), and Distichlis spicata percentage cover (bottom) recorded in burned or unburned marshes by marsh-management type for 3 summers (May-June 1996–1998) combined at Rockefeller State Wildlife Refuge, southwestern Louisiana. Experimental burns were conducted during December-January 1995–1996. Error bars represent upper 95% C.I. Asterisks (*) above paired bars indicate significant contrasts (P < 0.05).

Figure 3. Dead vegetation percentage cover in burned and unburned marshes during 3 summers (May-June 1996–1998) at Rockefeller State Wildlife Refuge, southwestern Louisiana. Experimental burns were conducted during December-January 1995–1996. Error bars represent upper 95% C.I. Common letters above bars indicate means did not differ (P > 0.05).

(F2, 30= 4.31, P= 0.02) in the analysis of bird species richness. Fewer bird species (P < 0.05) were recorded in 1996 (mean = 4.6 species, 95% C.I. = 3.7–5.6) than in either 1997 (mean = 6.3, 95% C.I. = 5.3–7.2) or 1998 (mean = 6.6, 95% C.I. = 5.6–7.5). Number of bird species did not differ (P > 0.05) between 1997 and 1998.

We detected a significant BURN × SUMMER interaction (F2, 30= 3.46, P= 0.04) for total birds/survey (all species combined). Total birds/survey in burned marshes was greater than that in unburned marshes in 1997 but did not differ between burn treatments in 1996 or 1998 (Figure 4). Total birds/survey in unburned marshes did not differ among summers, but in burned marshes total

birds/survey was greatest in 1997, intermediate in 1998, and least in 1996 (Figure 4). Total birds/survey differed among M-M TYPES (F3, 7= 9.06, P < 0.01). Within unimpounded marshes, total birds/survey was greater in brackish than in saline marsh (Figure 5). Within brackish marshes, total birds/survey was greater in unimpounded than in impounded marshes (Figure 5). Within impounded marshes, total birds/survey did not differ between marsh types (Figure 5).

We detected a significant SUMMER effect (F3, 20= 3.95, P= 0.04) in the analysis of icterids/survey; however, the LSMEANS test failed to detect any significant pairwise comparisons. Icterids/survey tended to be greater in 1997 (back-transformed mean = 2.3, 95% C.I. = 1.7–2.9) than in 1996 (back-transformed mean = 1.7, 95% C.I. = 1.3–2.4) or 1998 (back-transformed mean = 1.6, 95% C.I. = 1.2–2.1). Icterids/survey differed among M-M TYPES (F3, 7= 10.47, P < 0.01). Within impounded marshes, icterids/survey was greater in intermediate than in brackish marshes (Figure 5). Within unimpounded marshes, icterids/survey did not differ between brackish and saline marshes; within brackish marshes, icterids/survey did not differ between impounded and unimpounded marshes (Figure 5).

We detected a significant BURN × SUMMER interaction (F2, 30= 5.00, P= 0.01) in the analysis of sparrows/survey. Number of sparrows/survey was greater in burned than in unburned marshes in
### Table 3. Mean number of birds/survey (SD) of common bird species\(^a\) recorded in burned or unburned marshes during 3 summers (April–June 1996–1998) at Rockefeller State Wildlife Refuge, southwestern Louisiana.

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<th>Year</th>
<th>Species</th>
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<th>Unburned</th>
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<tr>
<td></td>
<td>Impounded</td>
<td>Unimpounded</td>
<td>Impounded</td>
</tr>
<tr>
<td></td>
<td>Brackish</td>
<td>Intermediate</td>
<td>Brackish</td>
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<tr>
<td>1996</td>
<td>Seaside sparrow</td>
<td>1.1 (0.6)</td>
<td>5.3 (1.0)</td>
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<tr>
<td></td>
<td>R.-w. blackbird</td>
<td>1.8 (0.5)</td>
<td>4.1 (1.6)</td>
</tr>
<tr>
<td></td>
<td>Boat-tailed grackle</td>
<td>0.1 (0.1)</td>
<td>0.5 (0.8)</td>
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<td></td>
<td>Marsh wren</td>
<td>0.1 (0.1)</td>
<td>0.1 (0.1)</td>
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<tr>
<td></td>
<td>Sedge wren</td>
<td>0.2 (0.0)</td>
<td>0.2 (0.0)</td>
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<tr>
<td></td>
<td>Least bittern</td>
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<td></td>
<td>Common yellowthroat(^b)</td>
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<td></td>
<td>Clapper rail</td>
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<td></td>
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<td></td>
<td>Sora(^c)</td>
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<td>1997</td>
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<td>R.-w. blackbird</td>
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<td></td>
<td>Sora</td>
<td>0.0 (0.0)</td>
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<tr>
<td>1998</td>
<td>Seaside sparrow</td>
<td>2.6 (2.0)</td>
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<td></td>
<td>Sora</td>
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\(^a\) In addition to common (>15 total individuals recorded over the 3 summers), other species observed included eastern meadowlark (Sturnella magna, \(n = 14\)), eastern kingbird (Tyrannus tyrannus, 13), orchard oriole (Icterus spurius, 9), swamp sparrow (Melospiza georgiana, 9), blue-winged teal (Anas discors, 8), tri-colored heron (Egretta tricolor, 6), tree swallow (Tachycineta bicolor, 6), red-winged blackbird (Agelaius phoeniceus, 5), black-crowned night-heron (Nycticorax nycticorax, 4), Virginia rail (Rallus limicola, 3), green heron (Butorides virescens, 3), black-necked stilt (Himantopus mexicanus, 3), Plegadis spp. (3), common night-hawk (Chordeiles minor, 2), Phalaropus spp. (2), semipalmated sandpiper (Calidris pusilla, 2), scissor-tailed flycatcher (Tyrannus forficatus, 1), osprey (Pandion haliaetus, 1), lesser yellowlegs (Tringa flavipes, 1), great blue heron (Ardea herodias, 1), Limmnocorax spp. (1), black tern (Chlidonias niger, 1), American bittern (Botaurus lentiginosus, 1).

\(^b\) Geothlypis trichas.

\(^c\) Porzana carolina.

1997 (Figure 4). Sparrows/survey in unburned marshes did not differ among summers; in burned marshes, sparrows/survey in 1997 was 2 times greater than that in 1996 but did not differ from that in 1998 (Figure 4). Sparrows/survey differed among M-M TYPES (\(F_2 = 33.41, P<0.01\)). In impounded marshes, sparrows/survey was greater in brackish than in intermediate marshes (Figure 5). In unimpounded marshes, sparrows/survey was greater in brackish than in saline marshes (Figure 5).
Figure 4. Number of birds/survey for all species combined (top) and for sparrows (bottom) recorded in burned and unburned marshes during 3 summers (April-June 1996-1998) at Rockefeller State Wildlife Refuge, southwestern Louisiana. Experimental burns were conducted during December-January 1995-1996. Error bars represent upper 95% C.I. Common letters above bars indicate means did not differ (P > 0.05).

In brackish marshes, sparrows/survey was greater in unimpounded than in impounded marshes. Nearly all sparrows were observed in brackish or saline marsh: 5 of >1,200 seaside sparrows and 0 of 40 Nelson’s sharp-tailed sparrows were recorded in intermediate impoundments.

Wrens/survey did not differ among summers, burn treatments, marsh-management types, or any interactions (all Ps > 0.05). We recorded 0.4 wrens/survey (95% C.I. = 0.3-0.5).

Discussion

Vegetation characteristics

We found that winter burning increased plant species richness in intermediate impounded marshes but not in other marsh-management types. Thus, our results were consistent with those of 18 studies of marsh burning (reviewed in Mendelssohn et al. 1995) where plant species richness either increased or remained the same, but did not decrease, in response to burning. Individual plant species responses are influenced by season of burn, post-burn water levels, pre-burn plant communities, and soil type (Chabreck 1981, Mendelssohn et al. 1995, Whelan 1995).
One goal of winter burning is to remove *Spartina patens*, *Distichlis spicata*, and dead vegetation while promoting growth of preferred waterfowl and furbearer food plants, i.e., *Scirpus robustus* and *S. americanus* (Coley) (Mendelssohn et al. 1995, Nyman and Chabreck 1995). Contrary to previous studies (O’Neil 1949, Chabreck 1981), we found that, except for *Distichlis spicata* cover in brackish unimpounded marshes (Figure 2), burning did not decrease percentage cover of either *D. spicata* or *Spartina patens* and in some cases percentage cover of these 2 species increased following burning. Moreover, overall *Scirpus robustus* percentage cover was low in our study marshes and was relatively greater in unburned than in burned marshes (Gabrey 1999, Gabrey and Afton 2001); *S. americanus* was not recorded in any marsh until 1997 (Gabrey 1999), the second summer post-burn. Presence of the 2 *Scirpus* species in our study marshes indicates that propagules were present in the marsh substrate at the time of burning, but other environmental factors, such as season of burn, probably inhibited their growth. Our experimental burns were conducted in late December and early January, whereas burns conducted earlier in fall promoted *S. americanus* in greenhouse experiments (Chabreck 1981, but see Hess 1975). We conclude that the role of fire in establishing stands of *Scirpus robustus* and *S. americanus* is ambiguous. Identification of environmental factors that interact with burning and affect *Scirpus* productivity in the Chenier Plain is needed to enable managers to achieve their goals more efficiently.

Burn treatment did not affect visual obstruction and total vegetation percentage cover but did reduce dead vegetation percentage cover during the first summer post-burn (1996). These results are consistent with a previous study (Gabrey et al. 1999) in which vegetation differences between burn treatments were most noticeable in the first 2 months immediately following burning (January-February), but negligible by the second winter post-burn. Under conditions of our study, vegetation structure in burned marshes recovered to pre-burn conditions relatively quickly (<12 months, Gabrey et al. 1999). Our findings support other studies where vegetation in marshes dominated by rhizomatous herbaceous species recovered to pre-burn conditions within 1 to 2 years following burning (see review in Mendelssohn et al. 1995).

Plant species richness generally is greater in impounded than in unimpounded marshes (Chabreck 1960) and increases with decreasing salinity (Odum 1988). In our study, plant species richness in impounded marshes in 1996 was markedly greater in intermediate than in brackish marshes, but did not differ between any marsh-management types in 1997 and 1998. A severe drought in April and May 1996 (National Oceanic Atmospheric Administration 1996) resulted in a natural drawdown in intermediate and brackish impoundments from April to June, allowing buried seeds of annuals and flood-intolerant species to germinate (van der Valk 1981, Baldwin et al. 1996). Plants present in intermediate impoundments included a number of seed-producing annuals that germinate under drawdown conditions (Gabrey 1999). Brackish impoundments, on the other hand, were dominated by rhizomatous species such as *Spartina patens* and *Distichlis spicata* and had few annual species. Species richness apparently was not affected by the 1996 drought in brackish or saline unimpounded marshes and was remarkably consistent among the 3 summers. Relatively few plant species can tolerate periodically flooded saline conditions; species tolerant to such conditions are predominantly clonal perennials that produce relatively small seed banks (Odum 1988). In addition, unimpounded marshes were exposed to tides and presumably were flooded with tidal salt water at a frequency and duration sufficient to prevent germination of any seeds present in the seed bank (Baldwin et al. 1996). Although southwestern Louisiana experienced another severe drought in 1998 (National Oceanic Atmospheric Administration 1998), impoundments did not dry until June and July, after we had collected our vegetation data.

Finally, vegetation structure showed only minor differences among the 4 marsh-management types. Total vegetation percentage cover varied but generally was less in intermediate than in brackish impound marshes. Small patches of open water or unvegetated mud were more numerous in our intermediate impound marshes than in other marsh-management types (S. W. Gabrey, personal observation). Consequently, a greater proportion of vegetation points in intermediate impound marshes received cover scores of 0, resulting in lesser total vegetation percentage cover scores.

**Bird surveys**

Changes in plant diversity and vegetation structure should result in different bird communities (Cody 1968), and bird communities should continue
to change with plant succession (Johnston and Odum 1956, Huff et al. 1984). Because post-burn plant succession in our study area progressed relatively quickly, we predicted that bird communities in burned marshes also would change quickly. Winter burning did not alter bird species richness or species composition; of the 10 most abundant bird species, only sedge wrens were absent from burned marshes during the first summer post-burn (although they did occupy these marshes in subsequent summers, Table 3). Species detected in burned habitat included most of the pre-burn species, with only a few species disappearing or moving in, suggesting that most species in periodically burned habitats are tolerant of a wide range of conditions (Bendell 1974).

Although bird species richness and composition apparently changed little in response to burning in our study, number of birds/survey in burned marshes increased dramatically over that in unburned marshes during the second summer post-burn. In other habitats, total density or relative total abundance following fire generally remains constant or increases (Bendell 1974, Lyon and Marzluff 1984, Fitzgerald and Tanner 1992), although decreases have been observed (Petersen and Best 1987, Pylypec 1997, Herkert 1994). In these studies, however, responses of individual species or groups of species were quite variable. We found that winter burning did not affect number of icterids/survey or wrens/survey but did affect number of sparrows/survey, of which >95% were seaside sparrows.

Dead vegetation cover appears to be an important habitat component of sparrows inhabiting coastal marshes. Nesting activity and abundance of male Louisiana seaside sparrows (Ammodramus maritimus fisheri, Gabrey and Afton 2000) and abundance of Cape Sable seaside sparrows (A. m. mirabilis, Taylor 1983) were reduced markedly during the first breeding season post-burn, when dead vegetation cover was least. We found that number of sparrows/survey in burned marshes peaked in the second summer post-burn, coinciding with recovery of dead vegetation cover to pre-burn levels. Seaside sparrows feed primarily on invertebrates gleaned off the ground or low vegetation (Post and Greenlaw 1994); litter and dead vegetation may act as a substrate for invertebrate prey. In addition, sparrows use mostly dead vegetation for nest construction (S.W. Gabrey, personal observation) and burning probably reduces availability of nesting material. Although numbers of Cape Sable seaside sparrows in Florida increased in the second breeding season following burning, they began to decline after the fourth breeding season post-burn, presumably because continual accumulation of dead vegetation made the sites unsuitable (Taylor 1983). Thus, periodic but infrequent fires appear necessary to maintain suitable sparrow habitat. Long-term studies of sparrow populations in Chenier Plain marshes would provide useful information regarding optimum frequency of burning in managing coastal habitat.

Although we detected only minor differences in vegetation structure among marsh-management types, we believe that these differences contributed to differences in the associated bird communities. Number of icterids/survey was greatest in intermediate impounded marshes; the generally small total vegetation percentage cover in these marshes may have provided better communication within flocks. Also, vegetation height in these marshes was more variable than in brackish or saline marshes because of the presence of tall plant species (Typha spp. and Phragmites australis, see Table 1) as well as low-growing Spartina patens and Distichlis spicata. Red-winged blackbirds and boat-tailed grackles frequently nest in stands of Typha and Phragmites but rarely nest in S. patens-dominated stands (Gabrey, unpublished data). Typha and Phragmites were absent from most brackish or saline marshes. Number of sparrows/survey, on the other hand, was greater in brackish unimpounded marshes than in either brackish impounded or saline marshes, suggesting marsh-type (brackish) and management-type (unimpounded) preferences by this group. Vegetation in these marshes typically consists of S. patens and D. spicata in extensive stands of uniform height. These plant species form a dense canopy 1 m above the marsh surface that may provide suitable cover for ground-foraging sparrows.

The 1996 drought may have resulted in fewer bird species recorded in that summer than in summers of 1997 or 1998. Because many of the small ponds in or near our study marshes were dry, many bird species associated with open water, such as ducks, shorebirds, gallinules, and herons, were not observed.

Management implications

Our results indicate that winter burning in the Chenier Plain of Louisiana had little effect on physical structure or species composition of the marsh.
plant community by the following summer despite significant differences during the first winter post-burn (Gabrey et al. 1999). Consequently, bird community changes also were short-lived despite differences in bird abundance and species composition during the first winter post-burn (Gabrey et al. 1999). Periodic but infrequent fires that remove dense dead vegetation apparently benefit sparrow populations in coastal marshes. Consequently, managed burns for waterfowl or fur bearer management appear compatible with maintaining populations of other marsh birds, provided large contiguous marsh is not burned in any single winter and >2 years are allowed between burns (Nyman and Chabreck 1995, Gabrey et al. 1999, Gabrey and Afton 2000). Because many environmental factors probably influence post-burn plant and bird communities (e.g., season of burn, pre-burn plant community, water depth and salinity; Mendelsohn et al. 1995), we recommend that managers clearly define their objectives (i.e., “convert x ha of Spartina marsh to Scirpus marsh” rather than “burn x ha”) and burn only when conditions are appropriate to achieve these objectives.

We conducted our experimental burns during winter to mimic current marsh management practices; however, lightning fires in the Chenier Plain occur mostly from June–August. Season of burning may influence plant species responses (Chabreck 1981) and subsequent bird communities. Our study marshes were burned only at the start of the 3-year study; plant and bird community responses to more frequent fires in Chenier Plain marshes are unknown. Future comparative studies of managed burns of different frequencies and seasonality with natural lightning fires would help in understanding the role of fire in coastal marsh ecosystem function and management.

Other studies have shown that impoundment construction results in changes in marsh salinity, hydroperiod, and vegetation (Montague et al. 1987) that benefit some bird species, particularly those associated with open water or mudflats such as waterfowl, wading birds, and shorebirds (Chabreck et al. 1974, Weber and Haig 1996, Gordon et al. 1998). We found that seaside and Nelson’s sharp-tailed sparrows, although present in brackish impounded marshes, apparently preferred tidally influenced brackish unimpounded marshes, whereas icterids apparently preferred intermediate impounded marshes. Continued loss of unimpounded marshes could significantly impact coastal sparrow populations as well as numerous other species (Gosselink 1979). Consequently, preservation and augmentation of existing unimpounded marshes should be a conservation priority.

In addition, we recommend that managers create or maintain a mosaic of varying marsh types and management types within coastal marshes of the Chenier Plain. Such habitat diversity, including intermediate impounded marshes for waterfowl and brackish and saline unimpounded marshes for sparrows, would meet the varied requirements for the greatest variety of species.

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