Final Report

THE ECOLOGICAL IMPLICATIONS OF MARSH MANAGEMENT TO WETLAND BIRDS

By

Owen N. Fitzsimmons, Bart M. Ballard, M. Todd Merendino, and Kevin M. Hartke

9 November 2009

INTRODUCTION

The Texas Gulf Coast has suffered widespread degradation and loss of its coastal wetland habitat, particularly fresh and intermediate marshes. Along the central Texas Coast this has been the result of conversion of wetland habitat to agriculture, rural and urban development, human recreation, and other socioeconomic factors (Moulton et al. 1997). The Texas Gulf Coast is extremely diverse in its wetland habitat, and it provides critical stopover and wintering habitat to a wide variety of bird species. It serves as the principal wintering site for waterfowl in the Central Flyway (Stutzenbaker and Weller 1989) and is also a key area for migratory wading birds (Mikuska et al. 1998) and shorebirds (Withers and Chapman 1993). The impacts of wetland loss to birds are well documented and have prompted the widespread use of marsh management techniques by private and public landowners in an attempt to mitigate these losses (Erwin et al. 1986, Tori et al. 2002, Kaminski et al. 2006). Marsh management typically employs the use of structural modifications, such as levee systems and water control structures, to impound an area in order to manipulate flood duration, frequency, and depth. This is often supplemented with a prescribed fire regime, livestock grazing, or mechanical manipulations in an attempt to improve vegetation and habitat for wetland birds, primarily waterfowl, and to reduce saltwater intrusion (Cowan et al. 1988, Stutzenbaker and Weller 1989).

Prior research has been conflicting on the effectiveness of impounded, managed marshes in providing quality waterbird habitat. Anderson and Smith (1999) and Kaminski et al. (2006) found that marsh management seemed to support greater waterbird richness and relative abundance and recommended its future use. Additionally, Weber and Haig (1996) compared diked, managed habitat to that of natural coastal mudflats for shorebirds in South Carolina and concluded that managed areas provide valuable supplemental habitat during winter and spring high tides and may even be preferred at low tides. Alternatively, Mitchell et al. (2006), who summarized existing literature regarding marsh management effects on target and non-target species, implied that, while impounded marshes seem to attract a variety of target species, some endemic salt marsh species might be negatively affected.

Another issue at the forefront of coastal wetland management is the effect of impoundments on fisheries. It is widely accepted that coastal estuarine ecosystems serve as nurseries for a variety of fishes and crustaceans, although this has rarely been clearly stated (Beck et al. 2001). The general hypothesis is that marine-transient species spend critical portions of their life cycles within marsh systems, whether for refuge or nursery, and then migrate to estuarine and marine systems. Thus, if migrations are impeded by levee systems and water control structures, disastrous effects will be seen (Herke 1995). Rogers et al. (1994) suggest that it is difficult to properly assess the effects of structural marsh management on fishes and crustaceans due to the wide variety of structural modifications and marsh management techniques; however, much of the literature available illustrates negative impacts on many transient species. Hoese and Konikoff (1995), however, suggest that most studies linking restricted migration with reduced productivity ". . . suffer from various constraints, such as a lack of replication, high degree of variability in both study sites and temporally, and interference by the observer." Additionally, Hoese and Konikoff's (1995) own research suggests that simple adjustments to marsh management schemes may easily maintain overall fishery production in these ecosystems. Thus, it remains unclear as to what extent impoundments affect fishes and crustaceans.

Despite any negative associations, marsh management continues to be a widely accepted practice along the Texas coast. More research is needed to evaluate the ecological role of managed areas compared to natural areas in coastal systems. Given the conspicuous nature of birds and their strong ties to vegetation characteristics and prey availability, they seem to be ideal indicators of habitat quality (Weller 1988, Gawlik 2002).

Purpose and Objectives

The purpose of this study was to assess the ecological role of coastal wetland habitat that is managed for wetland birds through marsh management compared to that of unmanaged, estuarine wetland systems.

Specific Objectives include:

I. Document seasonal use (densities) of waterbird species on managed and unmanaged areas.

II. Assess the functional role that managed and unmanaged units serve to waterbirds (e.g., loafing, foraging, brood-rearing, etc.).

III. Determine invertebrate biomass and potential energy for waterbird species on managed and unmanaged sites.

IV. Document wetland vegetation communities (e.g., species composition, percent occurrence, and screening cover) on managed and unmanaged sites.

METHODS

Study Area

This study was conducted on 2 Texas Parks and Wildlife Department (TPWD) owned and managed Wildlife Management Areas (WMA) located along the central coast of Texas; Justin Hurst WMA, and Mad Island WMA. Justin Hurst WMA, formerly named Peach Point WMA, is located in Brazoria County west of Freeport. The 4,831-ha area was acquired by TPWD during a period from 1985 - 1988. Mad Island WMA comprises 2,946 ha in Matagorda County and was acquired by TPWD from The Nature Conservancy of Texas in 1987. Both WMA's are comprised of managed, palustrine emergent wetlands, coastal prairie meadows, estuarine intertidal marshes, and unvegetated intertidal mudflats. Management schemes in the impounded areas include spring/summer drawdowns followed by mechanical treatments, prescribed fires, and/or livestock grazing, and flooding in the fall. Natural, estuarine wetlands may also be subject to periodic prescribed fires and livestock grazing. Both managed and natural areas are frequented by the public for waterfowl hunting during fall and winter.

Two managed wetlands at each WMA were chosen with adjacent, unmanaged wetlands serving as control sites, totaling 4 managed/unmanaged pairs. Managed and unmanaged wetlands in each pair were directly adjacent to each other to reduce natural variation since managed areas were, prior to construction, similar to the unmanaged areas. At Justin Hurst WMA, the managed wetlands comprised the Greenwing and Mottled Duck sites, and at Mad Island WMA, the Rattlesnake and North Savage sites. The managed wetlands at Mad Island WMA were constructed in August 2001, and the managed wetlands at Green-wing and Mottled Duck sites were constructed in July 1998 and July 1999, respectively. Within each managed and unmanaged wetland, we delineated a 400-m^2 (16 hectares) area to keep the sampling sites similar and to match the smallest size of a managed site.

Sampling Period

Sampling was conducted during 3, 45-day seasons during both 2007-08 and 2008-09. Fall (1 September-15 October) and spring (1 April-15 May) seasons corresponded to peak migratory periods based on historical waterbird surveys from state lands along the Central Texas Coast (Brent Ortego, Texas Parks and Wildlife Dept., unpublished data). Winter sampling occurred from 1 January-15 February and included the largely non-migratory period during mid-winter.

Vegetation Community

Vegetation communities were surveyed once per season. Logistic constraints limited us from surveying the North Savage location in fall 2007. Four to 5 transects, totaling 1600 m, were placed equidistant and parallel through each wetland, perpendicular to the change in water-depth gradient. Vegetation measurements were made within a 1-m^2 floating quadrat constructed of small diameter PVC pipe (Tanner and Drummond 1985). The quadrat was placed every 30 m along transects, totaling 56 sampling points in each managed and unmanaged wetland. Percent cover was visually estimated for all plant species, bare ground, and open water located within the quadrat. A 3-m modified Robel pole, constructed of 3.81-cm diameter PVC pipe and marked every 10 cm with red tape, was placed in the center of each quadrat and viewed from 4 m to the north and 1 m above ground or water surface (Robel 1970). The highest point obstructed 100% by vegetation was recorded to the nearest quarter decimeter to measure screening cover, and the tallest vegetation in contact with the pole was recorded for vegetation height. Water depth at each quadrat was also recorded with the Robel pole to the nearest quarter decimeter.

Aquatic Invertebrates

Aquatic invertebrate samples were collected once each season at 5 evenly spaced points along a 500-m transect that ran perpendicular to the change in water-depth gradient through each site. Samples were collected using a standard D-frame dip net. The net was bumped along the substrate and pulled up through the water column in 1-m strips in the 3 cardinal directions least disturbed by the observer. This allowed for the collection of benthic, water column, and water surface dwelling invertebrates. Samples were placed in 3.79-L sealed containers and preserved in 70% ethanol solution until they were transported to the Texas A&M University-Kingsville laboratory. In the laboratory, the invertebrate samples were sorted to order, dried in an oven at 60^{o} C until constant mass was reached, then weighed to the nearest 0.0001 g to determine biomass. True metabolizable energy (TME) values for common waterfowl food items compiled by DiBona (2007) were used to estimate available energy based on invertebrate biomass. The taxa present in our samples but not listed by DiBona (2007) were grouped with similar listed taxa based on size and morphology.

Water salinity and dissolved oxygen levels were measured with an YSI Model 85 Dissolved Oxygen, Conductivity, Salinity and Temperature System at the beginning and end of each invertebrate sampling transect.

Avian Community

Avian surveys to determine abundance and species richness took place in accordance to methodology outlined by Buckland et al. (1993). We deemed this to be a more effective survey method than stationary counts for detecting a wider variety of species due to the secretive nature of many waterbird species and the dense vegetation at some of the sites. In each wetland, trained observers walked the length of 2, 400-m line transects that ran equidistant to each other and perpendicular to the change in gradient to thoroughly cover any variation in habitat. Group size and distance from transect line were recorded for each bird species observed, except for those deemed by the observer to be passing by the area in flight. Aerial foragers were recorded only if they were observed actively feeding or resting in the survey area. Surveys took place between 0.5 - 3.5 hours after sunrise and 3.5 - 0.5 hours before sunset, with no surveys conducted in winds > 25 km/hr or during rain or fog due to likely reductions in detection rate. Up to 4 surveys were conducted per season, with no surveys taking place on days that public hunting occurred.

We grouped bird species into guilds based on primary foraging habitat. This allowed us to explore relationships between specific bird groups and physiognomic habitat characteristics. Foraging habitat guilds consisted of basin substrate, mudflat, water column, water surface, above water, transition zone, and terrestrial.

Partners In Flight (PIF) is an organization originally created to assess conservation needs of non-game neotropical migrants and has expanded its directive to include all non-game landbirds. They developed a system to rank bird species by their conservation priority based on seven parameters that take into account global and local threats, population status, and habitat availability. The rankings range from 1 to 5, 1 being lowest priority and 5 being the highest priority (Carter et al. 2000). For our research needs, we have referenced the PIF conservation priority database for all landbird species, the U.S. Shorebird Conservation Plan for all shorebird species, and the Southeast United States Waterbird Conservation Plan for all other waterbirds (Brown et al. 2000, Hunter et al. 2006). We compiled a list of all species detected on bird surveys that have a Threats to Nonbreeding (TN) conservation priority score of 4 or 5 (Carter et al. 2000). This provides a different approach to assessing habitat quality besides traditional uses of overall density or species richness that may mask the contribution of species of conservation concern to the avian community being studied.

In order to assess how birds were using the managed and unmanaged wetlands being studied, we recorded 10-minute activity budgets of randomly encountered birds. For ease of data collection and interpretation, we chose *a priori* to focus on wading bird species because of their conspicuousness and the ability to assess their foraging success. For each bird recorded, we estimated foraging rate (the number of foraging strikes/minute) and foraging success (the proportion of these strikes that were successful). Video samples were recorded up to 20 minutes per individual.

Statistical Analysis

We calculated Shannon's (Shannon-Wiener) diversity index ($H' = \sum(p_i)(\ln p_i)$) and Jaccard's similarity index ($C_j = j/a+b-j$) to help explain relationships in vegetation and bird communities between managed and unmanaged wetlands (Begon et al. 1990).

To account for site differences in detection probabilities of birds, we averaged screening cover for each year/season/site combination and observed natural gaps in average screening cover that provided evidence for dividing the year/season/sites into 3 classes. We then investigated histograms of all detection distances within each class and delineated natural cut points based on marked drops in frequency of observations across detection distances. As a result, maximum detection distances were 30 m, 60 m, and 120 m for our 3 classes. Encounters detected beyond these distances were not considered for density estimation within their respective class. Based on the nature of our design (treatment and paired control), we used paired t-tests to compare differences in measured parameters (i.e., bird densities, invertebrate biomass, etc.) between managed and paired, unmanaged wetlands. We also used ANOVA (PROC GLM; SAS Institute, Inc. 2003) to investigate effects of seasons and years and any interactions when appropriate. We did not test for differences for any indices that we calculated and present absolute differences. We considered any differences significant if $P < 0.05$.

RESULTS

Vegetation Community

Average number of plant species recorded in managed wetlands ranged from 14.83-19.82 and was greater ($t = 2.09$, $P = 0.048$) than in unmanaged wetlands which ranged from 11.75-16.82 (Table 1). Plant species that dominated the unmanaged wetlands included Gulf cordgrass (*Spartina spartinae*), marshhay cordgrass (*Spartina patens*), and saltgrass (*Distichlis spicata*). Species that were most dominant in treatment sites included maidencane (*Panicum hemitomon*), saltgrass, broad-leaf signal grass (*Brachiaria platyphylla*), and stonewart (*Nitella* spp.).

Plant species diversity was 15-29% greater in managed than unmanaged wetlands during each year*season combination, except spring 2009 when it was 7% greater in unmanaged than managed wetlands (Table 2). Plant diversity was inversely correlated with water salinity at the site (r = -0.54, n = 46, P < 0.001). Water salinities in managed wetlands were lower than unmanaged wetlands during each season in 2007-08 (Table 3). Salinities remained below 2 ppt in managed wetlands and below 10 ppt in unmanaged wetlands during fall and winter 2007-08. The managed wetland at Greenwing generally had the lowest salinities throughout the study. In 2008-09, Hurricane Ike greatly influenced both managed and unmanaged wetlands by increasing water salinities well above those recorded the previous year (Table 3). The effects of Hurricane Ike were particularly noticeable at Mad Island WMA, as salinities were < 1 ppt in fall 2007 and 33–40 ppt in fall 2008 following Hurricane Ike. In winter 2009, the dominate vegetation class in both managed and unmanaged wetlands was dead litter, most likely the result of the higher water salinities from Hurricane Ike's storm surge. Dead litter comprised a large component of the vegetation community through spring 2009, but at levels lower than that recorded in winter. Following Hurricane Ike, water salinities were greater in managed than unmanaged wetlands on Mad Island WMA throughout 2008-09 sampling periods (Table 3).

Similarity indices for vegetation communities between managed and unmanaged wetlands were quite variable across the study, ranging from 10% similar at North Savage in Fall 2008 to 57% similar at Mottled Duck in spring 2009 (Table 4). Similarity indices fluctuated more widely at Mad Island WMA sites than at Justin Hurst WMA sites.

Aquatic Invertebrates

Invertebrate biomass varied greatly across years, season, and sites. Because of this large variation, the difference between managed and unmanaged wetlands only approached significance ($t = 1.99$, $P = 0.058$). We detected no 2 or 3-way interactions among year, season, and treatment ($P \geq 0.061$), and no season effect ($P = 0.780$). On average, invertebrate biomass during 2007-08 was more than 2-times ($F = 4.87$; $P = 0.028$) the amount recorded in 2008-09. Coleoptera, Diptera, and Hemiptera occurred most frequently across samples in both managed and unmanaged wetlands. In 2007-08, Gastropods comprised the greatest proportion of the samples during each season in managed and unmanaged wetlands (range: 29%–90%). In 2008-09, Gastropods comprised a larger component of the invertebrate community in unmanaged compared to managed wetlands. In contrast, Hemiptera and Ostracoda dominated the invertebrate biomass in managed wetlands in 2008-09. Gastropoda, Coleoptera, Hemiptera, Diptera, Ostracoda, Decapoda, and Odonata consistently comprised > 80% of the invertebrate biomass, except for spring 2009 when Trichoptera contributed 31% of the total biomass in managed wetlands.

Estimated TME values (kcal/total sample biomass) were consistently greater in samples collected from managed wetlands compared to those from unmanaged wetlands (Table 5). Differences ranged from 1.38 times greater in fall 2008 to 296 times greater in fall 2007. Total energy was directly related to biomass as average kcal/g was similar between managed and unmanaged wetlands. Amphipoda, Ostracoda, Brachyura, Cladocera, and Decapoda had the highest TME values.

Avian Community

Bird species richness was greater (t = 5.29, $P < 0.001$) in managed than in unmanaged wetlands across the study (Table 6). We detected a total of 115 bird species using managed wetlands and 91 using unmanaged wetlands over the entire study with 79 species common to both. We observed 36 species only using managed wetlands, whereas 11 species were only observed using unmanaged wetlands (Table 7). Analysis also indicated a season effect on species richness in year 2 ($F_{2,62}$= 13.41; $P < 0.001$), where species richness was less in fall 2008 than in winter or spring 2009 ($P \leq 0.003$). Species richness averaged across surveys was greater in managed than unmanaged wetlands by 91% in Greenwing, 34% in Mottled Duck, 34% in North Savage, and 61% in Rattlesnake. Managed wetlands supported more species from each foraging habitat guild except for above water and transition zone foragers (Tables 8 and 9). Managed wetlands also appeared to support more migrant species relative to unmanaged wetlands as species richness of foraging guilds exhibited the most differences between managed and unmanaged wetlands during fall and spring.

Bird densities were 46% greater (t = 2.44, P = 0.018) in managed than in unmanaged wetlands. Bird densities were generally greater during 2007-08 than in 2008-09, except during spring (Table 10). On average, bird densities in managed wetlands were greater than unmanaged wetlands each season except winter 2008-09 following Hurricane Ike. Among the sampled sites, we detected much higher densities in the managed wetlands at Greenwing (236%) and Rattlesnake (88%) compared to their paired unmanaged wetlands. Mottled Duck and North Savage sites had similar densities between managed and unmanaged wetlands, differing by only 6%.

Bird diversity tended to be greater in managed than unmanaged wetlands; however, at Mottled Duck the unmanaged wetland had higher bird diversity during 4 of 6 sampling periods (Table 6). Unmanaged wetlands had higher bird diversity than managed wetlands during one sampling period at each of the other sites. Bird diversity was lowest during fall 2008 at both sites on Mad Island WMA. Bird communities were relatively dissimilar throughout the study, ranging from 10%–57% similarity at any combination of year, season, and site (Table 6). Managed and unmanaged wetlands differed the most in fall of 2008 as Jaccard's Similarity Indices were collectively the lowest (ranging from 10%–32%). In spring 2009, similarity in bird communities between 3 of the 4 wetland pairs approached or exceeded 50% and collectively was the greatest in any season.

Of the 23 bird species detected with conservation priority scores of 4 or 5, 19 were detected in managed areas and 17 were detected in unmanaged areas. Twelve of the 23 species were detected in both managed and unmanaged wetlands. More mudflat and water-column feeding species were detected in managed than unmanaged areas, including Stilt Sandpiper (*Calidris mauri*), Wilson's Plover (*Charadrius wilsonia*), Least Grebe (*Tachybaptus dominicus*), and Least Tern (*Sterna antillarum*). However, nearly all Rallidae species were detected only in unmanaged areas, with Sora (*Porzana carolina*) being the only exception. Two passerine species with high conservation priority rankings were detected only in managed areas; Nelson's Sharp-tailed Sparrow (*Ammodramus nelsoni*) and Olive-sided Flycatcher (*Contopus cooperi*) (Table 11).

Video monitoring to estimate foraging effort and success proved to be difficult due to heavy cover (particularly in unmanaged wetlands) and the discreet nature of the individual birds. Thus, our sample size was small (n = 15). Average strikes/minute was 35% greater for wading birds foraging in managed areas compared to those foraging in unmanaged areas. Average success rate of foraging strikes was similar for wading birds in both managed and unmanaged wetlands, averaging about 76% success.

DISCUSSION

Overall, managed sites supported greater plant diversity, higher aquatic invertebrate biomass, and more species and higher densities of birds than did unmanaged sites. It is well known that hydrology directly influences the type of biotic communities that are found in wetlands. Vegetation communities can respond dramatically to changes in hydroperiod or salinity (Mitsch and Gosselink 2000). Salinity was highly variable throughout the study and was inversely related to plant diversity. Following construction of impoundments, water circulation is often reduced, resulting in water quality issues such as extreme temperatures or salinities (Birkitt 1984, McGovern and Wenner 1990). Salinities recorded in managed wetlands at North Savage and Rattlesnake during 2008-09 were considerably higher than in the adjacent unmanaged wetlands and were 6 to 50 times higher than in 2007-08. The increased salinities appeared to have altered plant communities in managed wetlands to more closely match unmanaged sites, which were dominated by salt-tolerant species. Also, the increased vegetation community similarity was attributed to the large amount of dead litter that dominated both areas following Hurricane Ike. Hurricane Ike hit the Texas coast September 13, 2008 in the Galveston area; approximately 60 – 100 miles from the study areas. It was deemed a category 2 hurricane upon landfall, but the resulting category 4 storm surge reached up to 20 ft in some areas. Mad Island WMA is located further south than Justin Hurst WMA, or further from Hurricane Ike's landfall. However, it has slightly lower elevation and is closer to the Gulf of Mexico making it more susceptible to storm surge effects.

We found plant species diversity to be greater in impounded areas throughout the study period except spring 2009, when salinities in managed wetlands were high and water levels were at their lowest. Research has shown that wetland plant species richness is greatest in continually moist conditions, decreases in intermittently flooded conditions, and is lowest in perpetually flooded conditions (Baldwin et al. 2001). Hydrological control of an area allows managers to maximize the amount of time an area is subject to moist conditions by controlling drawdown speed and timing. This directly influences the plant community that responds to re-flooding in the fall (Merendino et al. 1990). In 2008-09, managers delayed drawdown into early spring in hopes that freshwater inflows would reduce salinity levels inside the wetlands. However, limited rainfall and high evapotranspiration rates threatened to further increase soil salinity, prompting managers to quickly drawdown the remaining, highly saline water from the wetlands. The high salinities in the managed wetlands at Mad Island WMA in 2008-09 had large influences on plant communities as plant species richness declined on average by 268% from the previous year. Further monitoring would be helpful in identifying plant community responses in impounded areas after major saltwater influx and could result in improved management techniques.

Aquatic invertebrate biomass varied greatly across the study period. This is not surprising given the observed fluctuations in water salinity and water levels in combination with the evolved life-history strategies that allow aquatic invertebrates to rapidly exploit seasonally available resources (Merritt and Cummins 1996). We noted that invertebrate biomass in managed wetlands was somewhat higher than in unmanaged wetlands. Wetland management practices have been shown to increase aquatic invertebrate biomass relative to unmanaged sites (De Szalay and Resh 1997, Anderson and Smith 2000, Davis and Bidwell 2008). Declines in invertebrate biomass within managed wetlands during 2008-09 were likely the result of increased salinities, reduced plant diversity, and decreased water levels. Research has shown a strong relationship between invertebrate biomass and abundance of submergent vegetation, with the greatest numbers of invertebrates found in areas where beds of submergent vegetation are interspersed with stands of emergent vegetation (Voigts 1976, Hornung and Foote 2006).

Aquatic invertebrates comprise a considerable portion of the diet of many wetland bird species and are particularly important to support nutrients and energy for annual cycle events (Krapu and Reinecke 1992, Skagen and Oman 1996). Total estimated energy from invertebrates was considerably greater in managed wetlands suggesting that these sites can support greater bird abundance than unmanaged wetlands. This is consistent with previous findings showing the diets of dabbling ducks foraging in freshwater sites to be of higher quality than those foraging in saltwater sites along the Texas Coast (Tietje and Teer 1996, Ballard et al. 2004). Longer hydroperiods in managed areas produce greater biomass and more diverse aquatic invertebrate taxa than areas not managed or with shorter flood duration (Anderson and Smith 2000). Impounded areas allow for the seasonal retention of water to increase invertebrate production, while also allowing for drawdowns to make invertebrate prey more available to foraging waterbirds during crucial periods of migration, increasing the likelihood that the area will be used by migrants (Epstein and Joyner 1988).

Overall bird species richness and density were greater in managed, impounded marsh than in natural areas. Greater bird densities coupled with greater invertebrate biomass and higher plant diversity, indicate that managed wetlands have a greater foraging capacity than unmanaged sites (Anderson and Smith 1999), assuming that resources are equally available in both managed and unmanaged sites. Kaminski et al. (2006) compared managed and unmanaged Wetland Reserve Program wetlands in central New York and came to a similar conclusion. They reported greater richness and relative abundance in managed areas and recommended further management to promote wetland bird use. However, they did not estimate food availability or any other habitat components in an attempt to explain bird use. Weber and Haig (1996) compared impounded, managed wetlands and natural areas and reported seasonally higher shorebird densities in managed areas, but found conflicting results when attempting to explain shorebird preferences with invertebrate prey densities. Similar research conducted on the Texas Gulf Coast compared shorebird abundance and invertebrate prey availability between created and natural sites and reported few significant differences, suggesting that managed areas could provide quality habitat compared to natural sites (Brusati et al. 2001). Our findings suggest that managed areas provide greater plant diversity, more available energy in prey items, and higher bird species richness and densities relative to unmanaged coastal marsh. We found similar numbers of species of conservation concern in unmanaged and managed wetlands elucidating the value of both wetland types.

Temporal differences in bird species richness and density are likely attributed to variation in migration chronology and seasonality in food availability. Migration is highly energy demanding, increasing the likelihood that areas with adequate food resources will be used as stopover or wintering habitat. Fluctuating water levels may make some prey items seasonally available, such as when draw downs are initiated in managed areas, or during tidal exchanges in unmanaged coastal marsh (Epstein and Joyner 1988, Gawlik 2002). The ability of invertebrates to rapidly exploit temporary resources may provide high concentrations of prey items in smaller, ephemeral pools, attracting large groups of feeding birds. Shorebird densities have been shown to correlate with the amount of exposed substrate (Darnell and Smith 2004). Consistently greater species richness and densities over time suggest that managed wetlandsmay provide more foraging opportunity for waterbirds throughout the migratory and over-wintering period.

We found similar numbers of species of conservation concern in both managed and unmanaged sites. More high-ranking species related to open water and mudflat habitat were detected in the managed wetlands whereas more secretive marsh species detected in unmanaged wetlands, illustrating the unique value of each habitat type for specific bird groups. In addition, the close proximity of the managed and unmanaged areas seemed to attract species that are not exclusive to one habitat type, as nearly half of the species of conservation concern we detected were found in both. This is consistent with our findings of overall bird species richness, where 79 of 126 species detected were found in managed and unmanaged wetlands. Given the dynamic nature of wetland resources, the location of managed areas near natural marsh is probably a driving factor influencing the diversity and abundance of bird species detected. Several studies have shown the value of wetland complexes in supporting greater species richness, and this should be taken into account when developing management plans for specific bird species or groups (Brown and Dinsmore 1986, Craig and Beal 1992, Weller 1994, Fairbairn and Dinsmore 2001). The value of managed wetlands is in adding biodiversity to the landscape. These areas increase habitat diversity and support resource needs for high-priority bird species.

Conclusion

Our findings suggest that proper management of impounded wetlands along the central coast of Texas can provide high quality habitat for wetland birds and increase bird diversity across the coastal wetland landscape. There are a number of factors that influence bird density and diversity and it is impossible to account for them all. We attempted to quantify some major components of wetland biotic communities, habitat and prey, in order to explain differences in seasonal bird use between the managed and unmanaged wetlands. We did not consider broad spatial effects such as site isolation or regional habitat availability (Brown and Dinsmore 1986). Nor did we consider breeding habitat, though that might have better qualified the differing habitats and better justified management efforts.

Future comparative studies should look into extended monitoring efforts to account for broader temporal changes in plant and bird communities, to better assess any patterns across years. Major events such as hurricanes can provide interesting pre and post event research opportunities, and future monitoring in these areas might provide clearer results on the post-event changes.

Enhanced plant diversity may yield greater bird diversity, and certainly both were greater in the managed areas. Dependent on specific objectives, managed, impounded wetlands on the Texas coast provide high quality habitat during crucial non-breeding periods to a large, diverse assemblage of birds, some of which are of high priority for conservation. The benefits of such habitat are justification for the establishment and continued management of impounded, freshwater marshes on the Texas coast to not only improve small-scale habitat, but improve landscape-scale diversity as well.

LITERATURE CITED

Alldredge, M. W., K. H. Pollock, T. R. Simons, and S. A. Shriner. 2007. Multiple-species analysis of point count data: a more parsimonious modeling framework. Journal of Applied Ecology 44:281-290.

Anderson, J. T., and L. M. Smith. 1999. Carrying capacity and diel use of managed playa wetlands by nonbreeding waterbirds. Wildlife Society Bulletin 27(2):281-291.

Anderson, J. T., and L. M. Smith. 2000. Invertebrate response to moist-soil management of playa wetlands. Ecological Applications 10(2): 550-558.

Baldwin, A. H., M. S. Egnotovich, E. Clarke. 2001. Hydrologic change and vegetation of tidal freshwater marshes: field, greenhouse, and seed-bank experiments. Wetlands 21(4): 519-531.

Ballard, B. M., J. E. Thompson, M. J. Petrie, M. Chekett, and D. G. Hewitt. 2004. Diet and nutrition of northern pintails wintering along the southern Texas coast. Journal of Wildlife Management 68: 371–382.

Batzer, D. P., and V. H. Resh. 1992. Macroinvertebrates of a California seasonal wetland and responses to experimental habitat manipulation. Wetlands 12: 1-7.

Beck, M. W., K. L. Heck, Jr., K. W. Able, D. L. Childers, D. B. Eggleston, B. M. Gillanders, B. Halpern, C. G. Hays, Kaho Hoshino, T. J. Minello, R. J. Orth, P. F. Sheridan, and M. P. Weinstein. 2001. The identification, conservation, and management of estuarine and marine nurseries for fish and invertebrates. Bioscience 51(8): 633-641.

Begon, M., J. L. Harper, and C. R. Townsend. 1990. Ecology: individuals, populations, and communities. 2nd Edition, Blackwell Scientific Publications.

Birkitt, B. F. 1984. Considerations for the functional restoration of impounded wetlands. in F. J. Webb, Jr. (Ed.), Proceedings of the 10th Annual Conference of Wetlands Restoration and Creation. Hillsborough Community College, Tampa, FL.

Brown, M., and J. J. Dinsmore. 1986. Implications of marsh size and isolation for marsh bird management. Journal of Wildlife Management 50(3): 392-397.

Brown, S., C. Hickey, B. Gill, L. Gorman, C. Gratto-Trevor, S. Haig, B. Harrington, C. Hunter, G. Morrison, G. Page, P. Sanzenbacher, S. Skagen, N. Warnock. 2000. National shorebird conservation assessment: shorebird conservation status, conservation units, population estimates, population targets, and species prioritization. Manomet Center for Conservation Sciences. http://www.Manomet.org/USSCP/files.htm

Brusati, E. D., P. J. DuBowy, and T. E. Lachery, Jr. 2001. Comparing ecological functions of natural and created wetlands for shorebirds in Texas. Waterbirds 24(3): 371-380.

Buckland, S. T., D. R. Anderson, K. P. Burnham, and J. L. Laake. 1993. Distance sampling: estimating abundance of biological populations. Chapman and Hall, London: New York.

Buckland, S. T., D. R. Anderson, K. P. Burnham, J. L. Laake, D. L. Borchers, and L. Thomas. 2001. Introduction to distance sampling: estimating abundance of biological populations. Oxford University Press Inc., New York.

Carter, M. F., W. C. Hunter, D. N. Pashley, and K. V. Rosenberg. 2000. Setting conservation priorities for landbirds in the United States: the Partners In Flight approach. The Auk 117(2): 541-548.

Cowan, J. H., Jr., R. E. Turner, and D. R. Cahoon. 1988. Marsh management plans in practice: do they work in coastal Louisiana, USA? Environmental Management 12(1): 37-53.

Craig, R. J., and K. G. Beal. 1992. The influene of habitat variables on marsh bird communities of the Connecticut River esturary. Wilson Bulletin 104(2): 295-311.

Darnell, T. M., and E. H. Smith. 2004. Avian use of natural and created salt marsh in Texas, USA. Waterbirds 27(3): 355-361.

DiBona, M. T. 2007. Seasonal food availability for wintering and migrating dabbling ducks and its implications for management at the Hackensack Meadowlands of New Jersey. Thesis, University of Delaware, Newark, Delaware, USA

Duffy, W. G., and D. J. LaBar. 1994. Aquatic invertebrate production in Southeastern USA wetlands during winter and spring. Wetlands 14: 88-97.

Epstein, M. B., and R. L. Joyner. 1988. Waterbird use of brackish wetlands managed for waterfowl. Proc. Annu. Conf. Southeast Assoc. Fish and Wildl. Agencies 42: 476-490.

Erwin, M. E., M. Coulter, and H. Cogswell. 1986. The use of natural vs. man-modified wetlands by shorebirds and waterbirds. Colonial Waterbirds 9(2): 137-138

Erwin, M. E. 2002. Integrated management of waterbirds: beyond the conventional. Waterbirds 25(Special Publication 2): 5-12.

Farmer, A. H., and A. H. Parent. 1997. Effects of the landscape on shorebird movements at spring migration stopovers. Condor 99:698-707.

Gawlik, D. E. 2002. The effects of prey availability on the numerical response of wading birds. Ecological Monographs 72(3): 329-346.

Herke, W. H. 1995. Natural fisheries, marsh management, and mariculture: complexity and conflict in Louisiana. Estuaries 18(1A): 10-17.

Hoese, H. D., Konikoff, M. 1995. Effects of marsh management on fisheries organisms: the compensatory adjustment hypothesis. Estuaries 18(1A): 180-197.

Hornung, J.P., and A. L. Foote. 2006. Aquatic invertebrate responses to fish presence and vegetation complexity in western boreal wetlands, with implications for waterbird productivity. Wetlands 26(1): 1-12.

Hunter, W. C., W. Golder, S. Melvin, J. Wheeler. 2006. Southeast Unites States regional waterbird conservation plan. http://www.waterbirdconservation.org/regional/

Kaminski, M. R., G. A. Baldassarre, and A. T. Pearse. 2006. Waterbird responses to hydrological management of wetlands reserve program habitats in New York. Wildlife Society Bulletin 34(4): 921-926.

Kushlan, J. A. 1986. The management of wetlands for aquatic birds. Colonial Waterbirds 9(2): 246-248.

McGovern, J. C., and C. A. Wenner. 1990. Seasonal recruitment of larval and juvenile fishes into impounded and non-impounded marshes. Wetlands 10: 203-221.

Merendino, M. T., L. H. Smith, H. R. Murkin, and R. L. Pederson. 1990. The response of prairie wetland vegetation to seasonality of drawdown. Wildlife Society Bulletin 18: 245-257.

Merritt, R. W., K.W. Cummins. 1996. Aquatic insects of North America., 3rd Ed. Kendall/Hunt Publishing Company, Iowa, USA.

Mikuska, T., J. A. Kushlan, and S. Hartley. 1998. Key areas for wintering North American herons. Colonial Waterbirds 21(2): 125-134.

Mitchell, L. R., S. Gabrey, P. P. Marra, and R. M. Erwin. 2006. Impacts of marsh management on coastal-marsh bird habitats. Studies in Avian Biology 32: 155-175.

Moulton, D. W., T. E. Dahl, and D. M. Dall. 1997. Texas coastal wetlands; status and trends, mid-1950's to early 1990's. U. S. Department of the Interior, Fish and Wildlife Service, Albuquerque, New Mexico.

Nur, N., S. L. Jones, and G. Geupel. 1999. A statis tical guide to data analysis of avian monitoring programs. USDI Fish and Wildlife Service, BTP- R6001-1999, Washington, DC.

Pennak, R.W. 1978. Fresh-water invertebrates of the United States., 2nd Ed. John Wiley and Sons, Inc., New York, USA.

Robel, R. J., J. N. Briggs, A. D. Dayton, and L. C. Hulbert. 1970. Relationships between visual obstruction measurements and weight of grassland vegetation. Journal of Range Management 23: 295-297.

Rogers, D. R., B. D. Rogers, and W. H. Herke. 1994. Structural marsh management effects on coastal fishes and crustaceans. Environmental Management 18(3): 351-369.

Stutzenbaker, C. D., M. W. Weller. 1989. The Texas Coast. p. 385-405. In L.M Smith, R. L. Pederson, and R. M. Kaminski (ed.) Habitat management for migrating and wintering waterfowl in North America. Texas Tech University Press, Lubbock, TX, USA.

Tanner, G. W., and M. E. Drummond. 1985. A floating quadrat. Journal of Range Management 38(3): 287.

Tietje, W. D., J. G. Teer. 1996. Winter feeding ecology of northern shovelers in freshwater and saline wetlands in south Texas. Journal of Widlife Management 60(4): 843-855.

Thomas, L., J. L. Laake, S. Strindberg, F. F. C. Marques, S. T. Buckland, D. L. Borchers, D. R. Anderson, K. P. Burnham, S. L. Hedley, J. H. Pollard, J. R. B. Bishop, and T. A. Marques. 2005. Distance 5.0. Release 2. Research Unit for Wildlife Population Assessment, University of St. Andrews, UK. <http://www.ruwpa.st-and.ac.uk/distance/>

Tori, G. M., S. McLeod, K. McKnight, T. Moorman, and F. A. Reid. 2002. Wetland conservation and Ducks Unlimited: real world approaches to multispecies management. Waterbirds 25(Special Publication 2): 115-121.

Voigts, D. K. 1976. Aquatic invertebrate abundance in relation to changing marsh vegetation. American Midland Naturalist 95(2): 313-322.

Weber, L. M., and S. M. Haig. 1996. Shorebird use of South Carolina managed and natural coastal wetlands. Journal of Wildlife Management 60(1): 73-82.

Weller, M. W. 1988. Issues and approaches in assessing cumulative impacts on waterbird habitat in wetlands. Environmental Management 12(5): 695-701.

Withers, K., and B. R. Chapman. 1993. Seasonal abundance and habitat use of shorebirds on an Oso Bay mudflat, Corpus Christi, Texas. Journal of Field Ornithology 64(3): 382-392.

Table 1. Plant species richness recorded from 4 managed and 4 adjacent unmanaged coastal marsh sites along the central Texas Coast during fall, winter, and spring 2007-08 and 2008-09.

			Species Richness	
			T	C
Year 1	Fall	Greenwing	21	26
		Mottled Duck	21	13
		North Savage	-	-
		Rattlesnake	16	11
	Winter	Greenwing	27	26
		Mottled Duck	13	12
		North Savage	27	16
		Rattlesnake	15	11
	Spring	Greenwing	22	30
		Mottled Duck	18	12
		North Savage	19	14
		Rattlesnake	19	14
		AVG	**19.82**	**16.82**
Year 2	Fall	Greenwing	28	14
		Mottled Duck	19	8
		North Savage	7	14
		Rattlesnake	7	12
	Winter	Greenwing	21	6
		Mottled Duck	15	8
		North Savage	5	10
		Rattlesnake	7	9
	Spring	Greenwing	31	25
		Mottled Duck	21	9
		North Savage	10	15
		Rattlesnake	7	11
		AVG	**14.83**	**11.75**
		TOTAVG	**17.22**	**14.17**

Table 2. Plant species diversity (*H'*) from 4 managed and 4 adjacent unmanaged coastal wetland sites along the central Texas coast during fall, winter, and spring 2007-08 and 2008-09.

		Managed	Unmanaged
Year	Season	*H'*	*H'*
1	Fall	2.41	2.10
	Winter	2.62	2.03
	Spring	2.44	1.97
2	Fall	2.31	1.93
	Winter	2.32	1.98
	Spring	1.54	1.65

Table 3. Average water salinities (ppt) recorded from points along a 400 m transect in 4 managed and 4 adjacent unmanaged coastal wetland sites along the central Texas Coast during fall, winter, and spring 2007-08 and 2008-09.

		Managed			Unamanged		
Year	Site	Fall	Winter	Spring	Fall	Winter	Spring
2007-08							
	GW	0.35	0.30	1.60	3.50	0.60	14.0
	MD	0.40	0.80	0.70	9.65	2.65	22.60
	NS	0.95	1.55	4.15	4.10	4.70	10.50
	RS	0.80	1.90	12.25	2.95	2.70	18.45
2008-09							
	GW	11.30	13.45	4.95	28.10	39.30	11.70
	MD	11.85	12.20	9.00	32.20	[a]	35.95
	NS	33.65	28.70	25.05	14.15	23.50	15.40
	RS	40.40	33.10	[a]	30.10	[a]	22.80

[a] salinity measurements not possible due to lack of measurable standing water.

Table 4. Jaccard's similarity index comparing vegetation community similarity at each managed and unmanaged wetland pair during fall, winter, and spring 2007-08 and 2008-09 along the central Texas coast.

	Area	Fall	Winter	Spring	
2007-08					Average
	Green-wing	0.18	0.26	0.27	**0.24**
	Mottled Duck	0.26	0.09	0.15	**0.17**
	North Savage	-	0.30	0.44	**0.37**
	Rattlesnake	0.29	0.37	0.44	**0.37**
	Average	0.24	0.26	0.33	
2008-09					
	Green-wing	0.20	0.17	0.24	**0.20**
	Mottled Duck	0.13	0.21	0.15	**0.16**
	North Savage	0.31	0.50	0.47	**0.43**
	Rattlesnake	0.46	0.46	0.39	**0.44**
	Average	**0.28**	**0.34**	**0.31**	
	Overall Average	**0.26**	**0.30**	**0.32**	

Table 5. Total invertebrate biomass (g dry mass) and total true metabolizable energy (TME; kcal) values from samples collected in managed and unmanaged coastal marsh along the central Texas Coast during the nonbreeding season, 2007-08 and 2008-09. TME values obtained from published sources and multiplied by total biomass of invertebrate samples.

		Managed		Unmanaged	
Year	Season	Biomass	TME	Biomass	TME
1	Fall	7.79	2.96	0.03	0.01
	Winter	9.59	3.65	2.16	0.82
	Spring	8.15	3.10	4.02	1.53
2	Fall	4.87	1.85	3.53	1.34
	Winter	0.57	0.22	0.01	< 0.01
	Spring	3.17	1.20	1.36	0.52

Table 6. Bird species richness, Shannon's (Shannon-Wiener) diversity index, and Jaccard's similarity index for bird communities in managed (M) and unmanaged (U) coastal marsh along the central Texas coast during fall, winter, and spring 2007-08 and 2008-09.

			Species Richness		Shannon's		Jaccard's
			M	U	M	U	
2007-08	Fall	GW	23	8	2.37	1.74	0.35
		MD	17	32	2.02	2.12	0.30
		NS	16	16	1.78	2.43	0.52
		RS	25	6	2.46	1.67	0.24
	Winter	GW	25	25	2.40	1.95	0.39
		MD	17	32	0.89	2.58	0.26
		NS	35	24	2.45	1.49	0.37
		RS	35	20	2.85	2.30	0.15
	Spring	GW	13	9	1.65	0.62	0.23
		MD	28	13	2.50	1.55	0.24
		NS	29	29	3.23	2.12	0.18
		RS	37	28	3.19	2.36	0.25
		AVG	25.0	20.17	2.32	1.91	0.29
2008-09	Fall	GW	25	16	2.01	2.21	0.32
		MD	28	17	2.18	2.78	0.29
		NS	23	11	1.73	1.52	0.10
		RS	21	17	1.48	2.50	0.19
	Winter	GW	34	27	2.72	1.98	0.27
		MD	26	13	2.49	2.28	0.35
		NS	26	33	2.70	2.32	0.44
		RS	39	22	2.21	1.29	0.39
	Spring	GW	23	12	2.24	1.99	0.25
		MD	27	17	2.47	2.58	0.57
		NS	31	26	3.17	2.58	0.46
		RS	30	29	2.88	2.48	0.48
		AVG	27.75	20.0	2.26	2.21	0.34
	TOTAVG		26.38	20.08	2.34	2.06	0.32

Table 7. Bird species found only in managed or only in unmanaged coastal marsh along the central Texas coast during fall, winter, and spring 2007-08 and 2008-09.

Managed (*n* = 33)	Unmanaged (*n* = 11)
Baird's Sandpiper (*Calidris bairdii*	Black Rail (*Laterallus jamaicensis)*
Black-bellied Plover (*Pluvialis squatarola*	Black Skimmer (*Rynchops niger)*
Black-bellied whistling duck (*Dendrocygna autumnalis*	Brewer's Blackbird (*Euphagus cyanocephalus)*
Blue-gray Gnatcatcher (*Polioptila caerulea*	Brown Pelican (*Pelecanus occidentalis)*
Bonaparte's Gull (*Larus philadelphia*	Clapper Rail (*Rallus longirostris)*
Cinnamon Teal (*Anas cyanoptera*	Eastern Kingbird (*Tyrannus tyrannus)*
Common Moorhen (*Gallinula chloropus*	King Rail (*Rallus elegans)*
Common Nighthawk (*Chordeiles minor*	Lincoln's Sparrow (*Melospiza lincolnii)*
Fulvous Whistling Duck (*Dendrocygna bicolor*	Orchard Oriole (*Icterus spurious)*
Greater White-fronted Goose (*Anser albifrons*	Purple Martin (*Progne subis)*
Hooded Merganser (*Lophodytes cucullatus*	Sandwich Tern (*Sterna sandvicensis)*
Le Conte's Sparrow (*Ammodramus leconteii*	
Least Grebe (*Tachybaptus dominicus*	
Lesser Scaup (*Aythya affinis*	
Mallard (*Anas platyrhynchos*	
Neotropic Cormorant (*Phalacrocorax brasilianus*	
Northern Mockingbird (*Mimus polyglottos*	
Northern Rough-winged Swallow (*Stelgidopteryx serripennis)*	
Nelson's Sharp-tailed Sparrow (*Ammodramus nelsoni)*	
Olive-sided Flycatcher (*Contopus cooperi)*	
Pectoral Sandpiper (*Calidris melanotos)*	
Redhead (*Aythya Americana)*	
Ring-necked Duck (*Aythya collaris)*	
Semipalmated Plover (*Charadrius semipalmatus)*	
Snow Goose (*Chen caerulescens)*	
Snowy Plover (*Charadrius alexandrinus)*	
Solitary Sandpiper (*Tringa solitaria)*	
Song Sparrow (*Melospiza melodia)*	
Sprague's Pipit (*Anthus spragueii)*	
Stilt Sandpiper (*Calidris himantopus)*	
Wilson's Plover (*Charadrius wilsonia)*	
Yellow-crowned Night Heron (*Nyctanassa violacea)*	
Yellow-rumped Warbler (*Dendroica coronata)*	

Table 8. Number of bird species by foraging habitat guild that were observed in managed and unmanaged coastal marsh along the central Texas Coast during the non-breeding period in 2007-08 and 2008-09.

Guild	Managed	Unmanaged
Benthic	11	6
Mudflat	23	16
Water column	25	22
Water surface	14	9
Above water	6	6
Transition	6	9
Terrestrial	30	23

Table 9. Number of bird species by foraging habitat guild and season that were observed in managed and unmanaged coastal marsh along the central Texas Coast during the non-breeding period in 2007-08 and 2008-09.

Season	Guild	Managed	Unmanaged
Fall	Benthic	6	3
	Mudflat	12	3
	Water column	16	15
	Water surface	9	5
	Above water	6	6
	Transition	4	4
	Terrestrial	12	8
Winter	Benthic	10	5
	Mudflat	10	12
	Water column	17	15
	Water surface	9	8
	Above water	7	6
	Transition	5	5
	Terrestrial	18	14
Spring	Benthic	7	4
	Mudflat	18	10
	Water column	16	15
	Water surface	7	6
	Above water	5	6
	Transition	5	5
	Terrestrial	16	13

Table 10. Average densities of birds observed on managed and unmanaged coastal marsh along the central Texas Coast during fall, winter, and spring in 2007-08 and 2008-09.

		Managed	Unmanaged
2007-08			
	Fall	15.86	12.32
	Winter	20.10	18.34
	Spring	14.03	8.61
2008-09			
	Fall	8.21	2.18
	Winter	5.71	9.18
	Spring	16.55	6.77

Table 11. High conservation priority bird species (priority ranking of 4 or 5 from Partners in Flight) that occurred in managed (M) and unmanaged (U) coastal marsh along the central Texas Coast during fall, winter, and spring 2007-08 and 2008-09.

Guild	Species	M	U
Benthic	Roseate Spoonbill (*Ajaia ajaja*)	X	X
Mudflat	American Avocet (*Recurvirostra americana*)	X	X
	Long-billed Curlew (*Numenius americanus*)	X	X
	Short-billed Dowitcher (*Limnodromus griseus*)	X	X
	Stilt Sandpiper (*Calidris himantopus*)	X	
	Western Sandpiper (*Calidris mauri*)	X	X
	Wilson's Plover (*Charadrius wilsonia*)	X	
Water Column	American Bittern (*Botaurus lentiginosus*)	X	X
	Brown Pelican (*Pelecanus occidentalis*)		X
	Little Blue Heron (*Egretta caerulea*)	X	X
	Least Bittern (*Ixobrychus exilis*)	X	X
	Least Grebe (*Tachybaptus dominicus*)	X	
	Least Tern (*Sterna antillarum*)	X	
	Pied-billed Grebe (*Podilymbus podiceps*)	X	X
	Reddish Egret (*Egretta rufescens*)	X	X
Water Surface	Wilson's Phalarope (*Phalaropus tricolor*)	X	X
Above Water	Nelson's Sharp-tailed Sparrow (*Ammodramus nelsoni*)	X	
Transition zone	Black Rail (*Laterallus jamaicensis*)		X
	Clapper Rail (*Rallus longirostris*)		X
	King Rail (*Rallus elegans*)		X
	Sora (*Porzana carolina*)	X	X
Terrestrial	Dickcissel (*Spiza americana*)	X	X
	Olive-sided Flycatcher (*Contopus cooperi*)	X	
	Total	19	17