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# Whooping and sandhill cranes visit upland ponds proportional to migration phenology on the Texas coast

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## Abstract

Two crane species, whooping cranes (*Grus americana*) and sandhill cranes (*Antigone canadensis*), overwinter along the Texas Gulf Coast. Periodic, extreme drought conditions have prompted concerns that potential freshwater limitations could hinder conservation of cranes, especially endangered whooping cranes. In response, land managers constructed and maintained freshwater ponds in upland areas near saltmarshes on the wintering grounds. We monitored 30 of those constructed ponds using camera traps (1 Oct 2013–31 May 2014) to quantify crane visits. For each species, we modeled pond visits as a function of migration phenology and environmental variables at 2 scales. Pond-scale variables included distance to saltmarsh and monthly salinity, and broad-scale variables included bay salinity, drought index, and tide level. We found pond visits by both crane species followed migration phenology with the greatest pond use in January–February. Both crane species visited ponds more on the mainland than on Matagorda Island. Sandhill crane visits were fewer at ponds with higher salinities and those filled by well water. Cranes visited ponds during the diurnal period and tended to avoid visiting ponds during the first 10% of the day. Pond visits by whooping cranes were ≤0.15 times/pond/day and by sandhill cranes were ≤0.28 times/pond/day. Our results suggested crane visits to constructed ponds may not be as frequent as once assumed nor driven by tidal and salinity conditions in the bay. The greater number of crane visits to constructed ponds on the mainland compared to Matagorda Island may be related to shrub encroachment around natural freshwater swale wetlands on the mainland, which is not as prevalent of a problem on the island. With proper management, swales on the mainland may provide alternatives to constructed ponds for cranes to obtain freshwater and forage.

**KEYWORDS**

*Antigone canadensis*, Aransas National Wildlife Refuge, camera trapping, drought, freshwater, *Grus americana*, Gulf of Mexico, salinity, swale wetland, tide

One non-reintroduced, migratory flock of the endangered whooping crane (*Grus americana*) exists and overwinters along the Gulf of Mexico, primarily on and around the Guadalupe and Mission-Aransas estuaries at Aransas National Wildlife Refuge (NWR) in Texas, USA (Canadian Wildlife Service [CWS] and U.S. Fish and Wildlife Service [USFWS] 2007). Periodically, extreme drought conditions occur around these estuaries. Wildlife biologists have hypothesized that as salinity in the saltmarshes surrounding these estuaries increases during drought, whooping cranes will need to leave saltmarshes for upland areas to access freshwater (Chavez-Ramirez and Wehtje 2012). Specifically, biologists hypothesized that when salinity in the estuaries exceed 15 to 23 ppt, whooping cranes must seek freshwater in upland areas (CWS and USFWS 2007, Chavez-Ramirez and Wehtje 2012, Stanzel and Smith 2017, Kirkwood and Smith 2018). Aerial survey data also confirm that whooping cranes use upland areas more during drought conditions (Butler et al. 2014).

The hypotheses regarding increased salinity have prompted concern that drought conditions reduce freshwater availability to whooping cranes, which in turn could slow the species' recovery. In response, land managers constructed and maintained freshwater ponds in upland areas to provide whooping cranes with reliable sources of freshwater. Wells with solar pumps have been added to some ponds to increase their reliability (CWS and USFWS 2007, Stanzel and Smith 2017). However, it is unknown to what extent whooping cranes use or need upland freshwater sources.

Using camera traps, Ritenour et al. (2016) monitored whooping crane use of 7 constructed ponds over 3 winters (2012–2014) with drought conditions. Kirkwood and Smith (2018) used camera traps to monitor whooping crane use of 3 freshwater ponds for 2 months (Feb–Mar 2016) without drought conditions. Ritenour et al. (2016) found whooping crane use varied among ponds and across years. Also, Ritenour et al. (2016) noted that whooping crane use of constructed ponds might be associated with migration phenology because pond use appeared to peak after most of the birds had arrived on the wintering grounds. Ritenour et al. (2016) emphasize the importance of upland ponds to whooping cranes during drought because they observed use on ≈30% of the days in which the 7 ponds were monitored. Further, Kirkwood and Smith (2018) found evidence to suggest whooping cranes used ponds when salinity in saltmarshes was >23 ppt but stopped using the ponds when salinity in the saltmarshes dropped to ≈15 ppt.

Both studies focused on a few ponds already known to be used by whooping cranes, were not drawn from a random sample, sampled as few as 3 to 7 ponds, and limited sampling to a small area of the mainland (Ritenour et al. 2016, Kirkwood and Smith 2018). Limiting sampling to ponds known to be used by whooping cranes causes problems with inference to all upland ponds available to cranes (i.e., non-representativeness) because biased samples have a propensity for biasing results (Thompson et al. 1998, Thompson 2002). By only sampling ponds known to be used by whooping cranes, researchers could have systematically favored an outcome such that whooping cranes exploit upland ponds over other outcomes.

Sandhill cranes (*Antigone canadensis*), whose overwintering area is much more extensive than that of the whooping crane (Tacha et al. 1986), also visit constructed ponds. No studies have examined sandhill crane use of freshwater ponds. Two studies have examined whooping crane use of constructed freshwater ponds at Aransas NWR but were deficient in their sampling design (Ritenour et al. 2016, Kirkwood and Smith 2018). To address concerns with previous sampling designs, we monitored crane visits to freshwater ponds by randomly sampling 30 constructed ponds in the uplands on both the mainland and barrier island. We monitored ponds with camera traps during the dry winter of 2013–2014. Our objective was to describe patterns of pond visits by whooping and sandhill cranes. Specifically, we determined the timing of visits to ponds by modelling the visits as a function of environmental variables and migration phenology. We also examined crane daily activity patterns at constructed ponds to determine if some parts of the day received more visits than others. Our research provides managers with information useful for determining if resources spent on pond management is commensurate with the biological contributions that ponds may provide.

# STUDY AREA

Whooping and sandhill cranes overwinter on and around Aransas National Wildlife Refuge (NWR), located 30 km northeast of Rockport, Texas, USA, along the Gulf of Mexico (Figure 1). The wintering grounds for whooping cranes primarily occur in Aransas, Refugio, and Calhoun counties, Texas, along the Guadalupe and Mission-Aransas

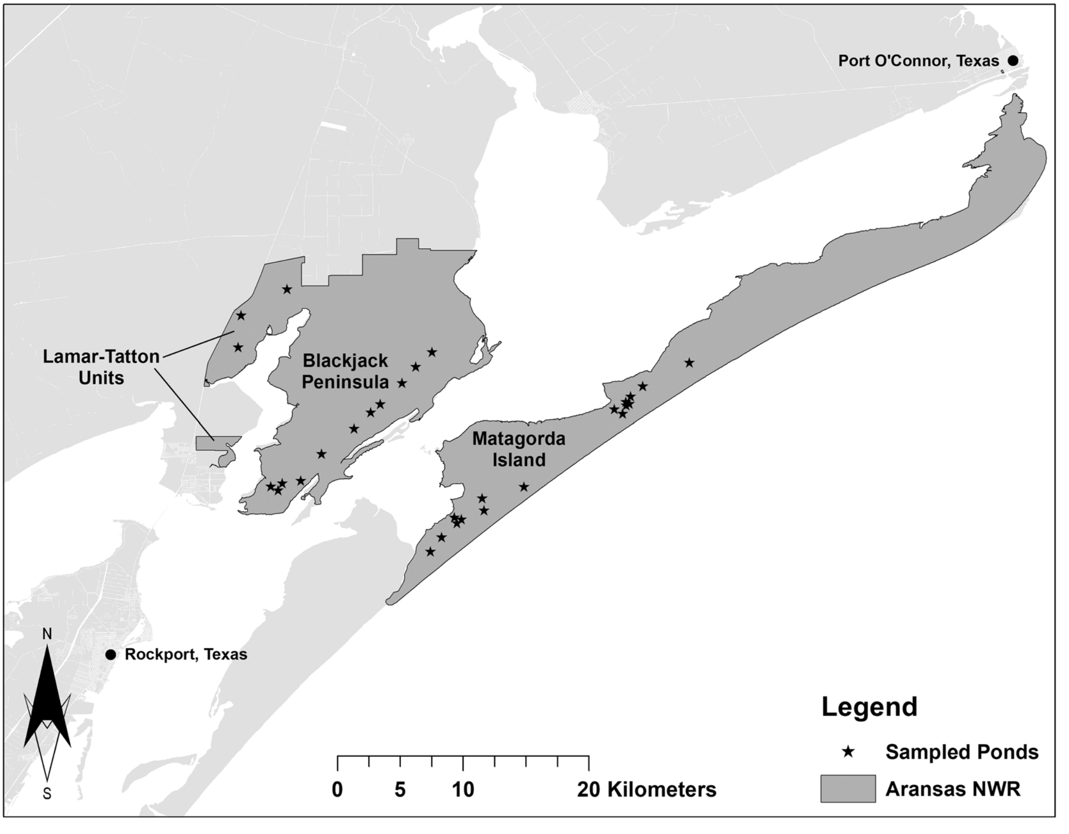


**FIGURE 1** Location of 30 constructed freshwater ponds on Aransas National Wildlife Refuge (NWR) located near Rockport, Texas, USA, along the Gulf of Mexico, where we monitored whooping and sandhill crane use during October 2013–May 2014.

estuaries. Aransas NWR is designated critical habitat for whooping cranes (CWS and USFWS 2007). Sandhill cranes overwinter along a greater extent of the Texas Gulf Coast than whooping cranes (Tacha et al. 1986).

Aransas NWR is located on the mainland (Blackjack Peninsula, Lamar-Tatton Units, and Myrtle Foster Whitmire Unit) and a barrier island (Matagorda Island). The refuge occurs in the southwestern portion of the Gulf Coastal Prairie of Texas. The landscape is flat with micro-topography of rolling shallow swale wetlands and ridges inland from coastal saltmarshes (Mowery and Bower 1978). Swale wetlands consist of depressions that lie between ridges of sand and shell deposition and range from temporarily flooded to semi-permanently flooded (USFWS 2010).

Whooping cranes primarily use saltmarsh habitats dominated by salt grass (*Distichlis spicata*) and cordgrasses (*Spartina patens* and *S. alterniflora*; Chavez-Ramirez et al. 1996, CWS and USFWS 2007). Occasionally, whooping cranes forage in upland areas composed of grasslands mixed with oak stands (*Quercus virginiana*) and other brush (Butler et al. 2014). Across the Gulf Coast, sandhill cranes primarily use agricultural areas for foraging (Hunt and Slack 1989). At Aransas NWR, sandhill cranes also use nonagricultural food items associated with saltmarshes and upland grasslands and brushlands (Hunt and Slack 1989).

During wet years, high freshwater inflows and rainfall (Wozniak et al. 2012) result in drinking water for cranes at Aransas NWR being available in the saltmarsh. Freshwater is also available in the uplands in swale wetlands. In addition to these freshwater sources, ≥119 ponds have been constructed on Aransas NWR (59 on Blackjack Peninsula, 55 on Matagorda Island, and 5 on the Lamar-Tatton Units). These ponds were manmade dugouts or modified swales designed to hold rainwater runoff. Many of the ponds have been maintained for crane use (primarily mowing and brush control; CWS and USWFS 2007, USFWS 2010), though some were developed historically for cattle and provide resources to other wildlife as well. Some ponds were inaccessible to cranes due to brush encroachment.

During drought conditions, some ponds become ephemeral or too saline for wildlife use. Therefore, Aransas NWR established wells at some ponds to provide greater freshwater reliability. During winter 2013–2014, 5 ponds had a functional freshwater well and pump associated with them and were considered accessible to cranes (one on Blackjack Peninsula, one on Matagorda Island, and 3 on Lamar-Tatton Units).

Our study area had prolonged drought conditions beginning in February 2011 until March 2015, based on Palmer hydrological drought index from Texas Climatological Division 7 (National Oceanic and Atmospheric Administration [NOAA] 2021). Drought conditions during April 2011–February 2012 were considered severe to extreme. After that, drought conditions were considered moderate to severe through December 2014. In January 2014, salinity in Mesquite Bay was >29 ppt, but during wet years salinity in the bay during January can fall to approximately 18 ppt. Precipitation on our study area averaged approximately 20 cm/year and winter temperatures averaged approximately 13°C (U.S. Climate Normals for station USC00410305; https://www.ncei.noaa.gov/access/us-climate-normals/).

# METHODS

## Camera trapping

We deployed infrared motion-triggered cameras (Bushnell Trophy Cam, Overland Park, KS, USA) on selected ponds (one camera trap/pond). Each of the 5 ponds with functioning wells and no impediments to crane use received a camera trap: 3 on the Lamar-Tatton Units, one on Blackjack Peninsula, and one on Matagorda Island. From the ponds without wells, we selected 15 from Matagorda Island, 16 from Blackjack Peninsula, and one from the Lamar-Tatton Units to place camera traps. We randomly ordered the list of ponds on Matagorda Island and Blackjack Peninsula and then selected ponds in order from the list. If a pond was surrounded by woody brush and inaccessible to cranes, did not contain water, or salinity was >10 ppt (salinity was expected to increase into the winter), it was not selected and the next pond in the ordered list that met the criteria was selected. We selected ponds and established cameras in September 2013. Shortly after establishing camera traps, 7 of the selected ponds (6 on Blackjack Peninsula and 1 on the Lamar-Tatton Units) dried up completely. These 7 dry ponds were excluded from the analysis, which resulted in 30 monitored ponds. We monitored 10 constructed ponds without wells and 4 well-fed ponds on the mainland (i.e., Blackjack Peninsula and Lamar-Tatton Units). On Matagorda Island, we monitored 15 constructed ponds without wells and one well-fed pond.

Camera traps were attached to a metal t-post approximately 1.5 m above the ground and aimed toward areas of likely crane use (i.e., flat, low density vegetation occurring on a pond's edge). Camera traps were operated from September 2013 through May 2014, though exact start and end times varied by site. We excluded data from September as few cranes were on the wintering grounds at this time. Camera traps were set to infrared motion-trigger mode with a 10-sec delay among triggers (i.e., 10-sec delay in image collections). The camera's specifications indicate the passive infrared sensor is effective to an average of approximately 14 m.

## Crane pond use

Our study relies on visits to ponds by whooping cranes and sandhill cranes as a surrogate species for use of the pond for obtaining water or forage. We identified animals in pictures and sorted them by date, time, species, and count of individuals (Harris et al. 2010). We considered pictures of the same species taken within one hour of one another to be from the same visit (Harris et al. 2015). Kirkwood and Smith (2018) also found that whooping crane groups typically used pond areas for <30 min at a time (maximum observed was 51 min). We estimated the mean number of visits per day for each pond for sandhill cranes and whooping cranes (count of group visits per month divided by the number of days per month that the camera trap was active). We modeled counts of crane group visits per month (Oct–May) for each crane species. We used pond type (well-fed or not), monthly pond salinity, distance to saltmarsh, bay salinity, tide levels, rainfall, time of year, and migration phenology as predictor variables (measured on a monthly-basis).

## Predictor variables

We visited each camera trap once per month to conduct maintenance and download data. At each visit, we measured salinity of the pond (YSI a Xylem Brand, YSI model 85, Yellow Springs, OH, USA). If pond salinity data were not available for a month, we used the mean of adjacent months. We used a saltmarsh layer from the Texas Ecological Systems Classification Project (Ludeke et al. 2012, Elliott et al. 2014). We measured distance (m) from each pond to the nearest saltmarsh in a geographic information system and standardized this variable ([observation–mean]/SD) for modeling purposes (i.e., model convergence); salinity was not standardized. We divided the wintering grounds into 2 study areas, mainland (Blackjack Peninsula and Lamar-Tatton Units) and barrier island (area around West Marsh of Matagorda Island).

Months were assigned their ordinal number (i.e., Oct = 1 and May = 8). These ordinal values were used as a quadratic descriptor of temporal use of ponds by cranes. Whooping cranes have been surveyed with repeated aerial surveys distributed across the October–May period of each winter beginning in 1950 through 2010 (Stehn and Taylor 2008, Butler et al. 2016). During surveys, data were recorded on paper maps and later digitized and archived (Taylor et al. 2015). Data were summarized as the number of whooping cranes counted on each survey date and the maximum number counted each winter. Using the aerial survey data, we developed a model of the proportion of the whooping crane population available during each month (quadratic of month) that describes migration phenology by year (M. J. Butler, U.S. Fish and Wildlife Service, unpublished results). We used program R and the glmer function (lme4 package, Bates et al. 2015, R Core Team 2020) to fit a logistic mixed-effects model with survey year as a random effect. We used that model to predict the proportion of the population available during each month of winter 2013–2014 (i.e., migration phenology; Oct = 0.148, Nov = 0.624, Dec = 0.871, Jan = 0.920, Feb = 0.894, Mar = 0.724, Apr = 0.257, May = 0.019).

We used the Palmer hydrological drought index (PHDI) from Texas Climatological Division 7 (NOAA 2021) as an indicator of long-term moisture supply, and because drought influences the use of upland habitats by whooping cranes (Butler et al. 2014). We obtained bay salinity measurements (ppt; National Estuarine Research Reserve System [NERRS] 2019) from the Mesquite Bay water quality station (MARMBWQ), which was most central to our study area (28.1384, −96.8285). From the NERRS data, we also obtained local rainfall (mm) from the Copano East meteorological station (MARCEMET) and tide level (m) measurements from the MARMBWQ station, averaging them by month (NERRS 2019).

## Modeling of crane visits

We analyzed crane group visits to ponds at 2 scales. First, we considered the local pond-scale, where monthly crane visits per pond for each species were modeled using generalized linear mixed-effects models (Zuur et al. 2009). We used Program R, the lme4 package, and the glmer function to analyze the data (Bates et al. 2015, R Core Team 2020). We assumed a Poisson distribution, treated each pond as a random effect to account for dependencies among subsamples at each pond, and used the number of days the camera operated during a month as a linear offset (Zuur et al. 2009). We examined relationships between predictor variables using Pearson's correlation coefficient or a $t$-test prior to modeling to screen variables and avoid issues with multicollinearity (Zar 1999). We considered time of year, migration phenology, pond type, pond salinity, distance to saltmarsh, and study area in this model set. We examined all combinations of models except distance to saltmarsh, study area, and pond type were not included as predictors in the same models and time of year and migration phenology were not included in the same models due to collinearity. We used Akaike's Information Criterion adjusted for small sample size ($AIC_c$) for model selection (Burnham and Anderson 2002). We considered models with $\Delta AIC_c \leq 2$ as competitive.

We also analyzed crane group visits to ponds at a broader scale because weather and bay conditions were not measured at the pond scale. We counted the number of crane group visits by species across all ponds for each month and study area. We modeled crane use with generalized linear models in program R (glm; Zuur et al. 2009, R Core Team 2020). We assumed a Poisson distribution and used the total number of days the cameras operated during a month at a study area as a linear offset. We considered time of year, migration phenology, drought, mean rainfall, mean bay salinity, mean tide level, and study area. Because of data correlations, we examined single variable models for the model set except study area with time of year or migration phenology. We used $AIC_c$ for model selection and considered models with $\Delta AIC_c \leq 2$ as competitive (Burnham and Anderson 2002).

## Daily activity patterns

We examined daily activity patterns for each crane species at ponds. Because day length varied from 10.35 to 13.57 hours over the period of study (October 2013–May 2014), we determined the proportion of the day in which a visit occurred. First, we calculated sunrise and sunset times, and day length for each date cranes visited a pond using the suncalc package in Program R (Thieurmel and Elmarhraoui 2019, R Core Team 2020). Then, for each crane visit, we calculated the amount of time since sunrise that a visit began. Lastly, we calculated the proportion of the day in which a visit occurred as the time since sunrise divided by day length. Since day length varies throughout the year (≈10.3 hr on 21 Dec 2013, ≈13.2 hr on 30 Apr 2014), we split the daylight period into 10 equal portions plus a period of equal length after sunset (depending on the day of the year, periods ranged from ≈62 to 79 min). We then counted the number of visits by each crane species during each portion of the day. To determine if diel activity was consistent across the day, we used a $\chi^2$ test (Zar 1999).

# RESULTS

We analyzed pictures from camera traps at 30 constructed ponds (5 well-fed ponds and 25 ponds without wells; Figure 1). Camera traps at each pond operated for 202.1 days on average between October 1 and May 31 (range = 142–241 days; SD = 32.4 days; Table 1). Ponds were on average 351 $m^2$ in size (range = 37 to 1,112 $m^2$, SD = 264 $m^2$; based on 2014 National Agriculture Imagery Program (NAIP) images, U.S. Department of Agriculture [USDA] 2021) and 313 m from saltmarsh (range = 0 to 856 m, SD = 220 m). The monthly salinity in the ponds ranged from 0.1 to 22 ppt ($\bar{x}$ = 2.7, SD = 2.95). We observed 587 pictures of whooping cranes and 1,443 pictures of sandhill cranes resulting in 194 visits to ponds by whooping crane groups and 295 visits to ponds by sandhill crane groups (Table 2).

We observed either crane species at 24 of the 30 ponds monitored. Whooping crane groups visited 13 ponds and sandhill crane groups visited 23 ponds; 12 ponds were visited by both crane species. Most visits by crane groups occurred from December through February (Table 2). We observed ≤2.3 whooping crane and ≤5.2 sandhill crane visits per day across all 30 ponds in a month; in most months ≤0.3 visits per day occurred by either species. Whooping crane visits occurred on average 0.15 times/pond/day on the mainland in January and 0.03 times/pond/day on Matagorda Island in February. Sandhill crane visits occurred on average 0.28 times/pond/day on the mainland in January and 0.12 times/pond/day on Matagorda Island in February (Table 2). Most use by whooping cranes was observed on one pond on the Blackjack Peninsula in December that was visited, on average, once per day by whooping cranes. Most use for sandhill cranes was observed on one pond on the Blackjack Peninsula in January that was visited, on average, approximately 2.2 times per day.

**TABLE 1** Summary of camera trapping effort (trap days) on constructed freshwater ponds at Aransas National Wildlife Refuge, Texas, USA, October 2013–May 2014.

| | | | Trap days | |
|---|---|---|---|---|
| **Study area** | **Month** | **No. ponds** | **Mean** | **SD** |
| Mainland | Oct | 12 | 22.8 | 10.4 |
| | Nov | 12 | 26.7 | 7.6 |
| | Dec | 12 | 31.0 | 0.0 |
| | Jan | 14 | 29.2 | 5.4 |
| | Feb | 14 | 28.0 | 0.0 |
| | Mar | 14 | 30.3 | 2.7 |
| | Apr | 13 | 28.5 | 5.5 |
| | May | 12 | 24.8 | 7.5 |
| Matagorda | Oct | 13 | 31.0 | 0.0 |
| | Nov | 16 | 28.3 | 4.3 |
| | Dec | 16 | 31.0 | 0.0 |
| | Jan | 16 | 29.5 | 6.0 |
| | Feb | 16 | 26.4 | 6.5 |
| | Mar | 16 | 26.0 | 10.4 |
| | Apr | 12 | 27.0 | 6.9 |
| | May | 9 | 24.2 | 9.4 |

**TABLE 2** Summary of whooping crane and sandhill crane camera trapping on constructed freshwater ponds at Aransas National Wildlife Refuge, Texas, USA, during October 2013–May 2014.

| | | Whooping cranes | | | | Sandhill cranes | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | | | Visits/pond/day | | | | Visits/pond/day | |
| Study area | Month | Pictures | Visits | Mean | SD | Pictures | Visits | Mean | SD |
| Mainland | Oct | 0 | 0 | 0.000 | 0.000 | 0 | 0 | 0.000 | 0.000 |
| | Nov | 10 | 8 | 0.022 | 0.067 | 2 | 1 | 0.003 | 0.010 |
| | Dec | 191 | 51 | 0.137 | 0.277 | 306 | 61 | 0.164 | 0.360 |
| | Jan | 168 | 64 | 0.147 | 0.231 | 572 | 113 | 0.277 | 0.614 |
| | Feb | 116 | 38 | 0.097 | 0.168 | 77 | 23 | 0.059 | 0.151 |
| | Mar | 20 | 8 | 0.021 | 0.041 | 0 | 0 | 0.000 | 0.000 |
| | Apr | 4 | 3 | 0.008 | 0.028 | 0 | 0 | 0.000 | 0.000 |
| | May | 3 | 2 | 0.006 | 0.021 | 0 | 0 | 0.000 | 0.000 |
| Matagorda | Oct | 0 | 0 | 0.000 | 0.000 | 0 | 0 | 0.000 | 0.000 |
| | Nov | 2 | 1 | 0.002 | 0.008 | 0 | 0 | 0.000 | 0.000 |
| | Dec | 0 | 0 | 0.000 | 0.000 | 9 | 7 | 0.014 | 0.020 |
| | Jan | 14 | 5 | 0.010 | 0.028 | 242 | 41 | 0.083 | 0.141 |
| | Feb | 48 | 12 | 0.027 | 0.058 | 180 | 40 | 0.118 | 0.194 |
| | Mar | 0 | 0 | 0.000 | 0.000 | 55 | 9 | 0.018 | 0.037 |
| | Apr | 11 | 2 | 0.006 | 0.019 | 0 | 0 | 0.000 | 0.000 |
| | May | 0 | 0 | 0.000 | 0.000 | 0 | 0 | 0.000 | 0.000 |

## Pond-scale models

For whooping crane pond visits, 3 models were competitive (Table S1, available online in Supporting Information). The best model suggested time of year and the study area were important predictors of whooping crane pond visits ($AIC_c$ = 328.7, model weight [$w$] = 0.319). The best fitting model suggested more whooping crane group visits occurred in January than other months (Time, $\beta = 2.957$, SE = 0.359, $P < 0.001$; Time$^2$, $\beta = -0.361$, SE = 0.042, $P < 0.001$), which coincides with the most whooping cranes on the wintering grounds. The best model also suggested whooping cranes visit ponds more on the mainland than on Matagorda Island ($\beta = -2.819$, SE = 1.163, $P = 0.015$). The second best model ($AIC_c$ = 329.6, $w = 0.198$) included an additional variable, pond salinity, and it indicated that more visits occurred at ponds with reduced salinity ($\beta = -0.083$, SE = 0.078, $P = 0.283$). Because $P > 0.15$ for the pond salinity coefficient, the salinity relationship might be spurious (Arnold 2010) or too weak to detect with this study's sample size. The third model ($AIC_c$ = 330.0, $w = 0.165$) was a variant of the first, where time of year was replaced with migration phenology. In the third best model, more whooping crane group visits occurred at ponds when a higher proportion of the population was on the wintering grounds ($\beta = 5.090$, SE = 0.632, $P < 0.001$). Like the first model, it also suggested whooping crane groups visit ponds more on the mainland than on Matagorda Island ($\beta = -2.816$, SE = 1.162, $P = 0.015$).

Two models for describing sandhill crane pond visits were competitive (Table S2, available online in Supporting Information). The best model suggested time of year (quadratic; Time, $\beta = 7.415$, SE = 0.645, $P < 0.001$; Time$^2$, $\beta = -0.891$, SE = 0.076, $P < 0.001$), salinity and pond type (well-fed or not) were important predictors of pond visits ($AIC_c$ = 393.8, $w = 0.363$). More sandhill crane group visits occurred at ponds with reduced salinity ($\beta = -0.185$, SE = 0.066, $P = 0.005$) and they visited well-fed ponds less than ponds without wells ($\beta = -1.490$, SE = 0.997, $P = 0.135$). The second best model was the same as the first except it did not include pond type ($AIC_c = 394.0$, $w = 0.338$).

## Broad-scale models

The broad-scale modeling for whooping crane pond visits suggested 2 models were competitive (Table S3, available online in Supporting Information). Like the pond-scale modeling, the best broad-scale models suggested whooping crane migration phenology and the study area were important predictors of pond visits ($AIC_c = 102.7$, $w = 0.543$). The best broad-scale model suggested more whooping crane group visits occurred when a higher proportion of the population was on the wintering grounds ($\beta = 5.225$, SE = 0.624, $P < 0.001$). Whooping cranes also tended to visit ponds more on the mainland than on Matagorda Island ($\beta = -2.316$, SE = 0.236, $P < 0.001$). The second best model suggested time of year and the study area were important predictors of whooping crane pond visits ($AIC_c = 103.0$, $w = 0.457$). The second best model suggested more whooping crane group visits occurred in January than other months (Time, $\beta = 3.007$, SE = 0.345, $P < 0.001$; Time$^2$, $\beta = -0.371$, SE = 0.041, $P < 0.001$). Also, the second best model suggested whooping cranes visit ponds more on the mainland than on Matagorda Island ($\beta = -2.322$, SE = 0.236, $P < 0.001$). None of the models containing tide level, precipitation, bay salinity, or drought condition were competitive for whooping crane visits.

For sandhill cranes, only one broad-scale model was competitive ($AIC_c = 133.1$, $w = 0.999$; Table S4, available online in Supporting Information). The best fitting model suggested time of year (quadratic; Time, $\beta = 6.734$,

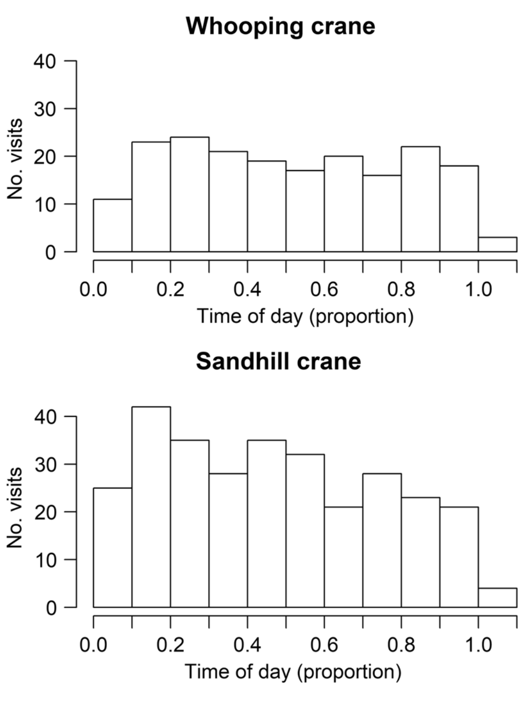


**FIGURE 2** Daily activity patterns for whooping and sandhill cranes at constructed freshwater ponds on Aransas National Wildlife Refuge, Texas, USA, October 2013–May 2014. Each day was divided into 10 equal portions plus a period of equal length after sunset (depending on the day of the year, periods ranged from ≈62 to 79 min since day length changes throughout the year). Time of day is represented as a proportion, where 0 = sunrise and 1 = sunset.

SE = 0.573, $P < 0.001$; Time$^2$, $\beta = -0.834$, SE = 0.070, $P < 0.001$) and study area were important variables. Just as in the pond-scale models, sandhill crane group visits at ponds occurred more on the mainland than on Matagorda Island ($\beta = -0.871$, SE = 0.124, $P < 0.001$). None of the models containing tide level, precipitation, bay salinity, or drought condition were competitive for sandhill crane visits.

## Daily activity patterns

We never observed whooping cranes at ponds prior to sunrise and in only 3 cases after sunset. We observed sandhill cranes twice at ponds prior to sunrise (<10 min before) and 4 times after sunset. During daylight hours, whooping cranes and sandhill cranes did not visit ponds proportionally among diel periods (whooping cranes, $\chi^2 = 20.897$, df = 10, $P = 0.022$; sandhill cranes, $\chi^2 = 37.422$, df = 10, $P < 0.001$; Figure 2). Both crane species tended to avoid visiting ponds during the first 10% of the day and avoided ponds after sunset.

# DISCUSSION

The visitation of constructed ponds by 2 crane species on the Gulf Coast of Texas was proportional to their migration phenology. For example, the most visitation of ponds occurred during the periods in which the most cranes were on the landscape, which was January–February (Butler and Harrell 2018; M. J. Butler, U.S. Fish and Wildlife Service, unpublished results). However, this period also coincides with typical winter low tides (Kirkwood and Smith 2018). Low tides can result in loss of connection of saltmarshes to the bay and isolation of water in saltmarsh pools (Ragan and Wozniak 2019). As saltmarsh waters evaporate, salinity can increase to levels greater than the surrounding bay because saltmarshes are no longer connected to the bay during low tides (Kirkwood and Smith 2018). Once higher tides return in March, saltmarsh pool salinities are expected to freshen to salinity levels similar to the bay, and Kirkwood and Smith (2018) observed less use of upland ponds by whooping cranes after high tides in early-March. However, asserting that changes in pond use by cranes resulted from changes in saltmarsh salinity is difficult because northward migration is beginning during this period, with fewer cranes on the landscape as time progresses. Further, we observed no relationship between upland pond visitation and bay salinity or tide levels for whooping or sandhill cranes.

Salinity of upland ponds did affect visitation by sandhill cranes. As salinity increased in ponds, sandhill crane visits declined. The larger sample of sandhill cranes likely allowed detection of the relationship between pond salinity and visits. However, the relationship between pond salinity and visits for whooping cranes was not statistically significant and the effect was an order of magnitude less than for sandhill cranes, suggesting that whooping cranes are more tolerant of salinity than sandhill cranes. Researchers have speculated that whooping cranes can tolerate brackish water up to 15 ppt, but the threshold may be as high as 23 ppt (CWS and USFWS 2007, Chavez-Ramirez and Wehtje 2012, Stanzel and Smith 2017, Kirkwood and Smith 2018). Although salinity tolerances for sandhill cranes have been generally unstudied, Haley (1987) observed sandhill cranes drinking water with salinities up to 20 ppt. Thus, there may be little difference in the salinity tolerances of the 2 species.

We expected whooping cranes to use ponds closer to saltmarsh more than ponds at greater distances (Hunt 1987) because saltmarshes are their preferred habitat. For example, Hunt (1987) found ponds used by whooping cranes were closer to saltmarsh than those that were not used. However, none of the ponds monitored in our study were greater than 860 m from saltmarsh; the constructed pond used the most by whooping cranes was approximately 500 m from saltmarsh. Whooping cranes used upland areas more during drought conditions (Butler et al. 2014). Severe to extreme drought conditions during our study may explain why we did not observe distance from saltmarsh as an important determinant of pond visitation. Sandhill cranes, which are more generalized in their habitat use than whooping cranes, were not expected to be influenced by distance to saltmarsh. Similarly, Hunt (1987) observed distance to saltmarsh was comparable for used and unused ponds by sandhill cranes.

During the peak abundance period, whooping cranes visited mainland ponds >10 times more often than they did ponds on Matagorda Island, even though whooping crane abundance was similar between the 2 areas (Aransas National Wildlife Refuge, unpublished survey data), which had similar proportions of grassland swale wetlands (Elliott et al. 2014). Sandhill cranes visited ponds on the mainland >2 times more often than Matagorda Island ponds. The differences in pond visitation rates could be attributed to the quality of natural swale wetlands between the mainland and island ecosystems. On Blackjack Peninsula, up to 75% of swale wetlands had brush encroachment but few of those on Matagorda Island did so. Indeed, all swale wetlands on Matagorda Island were classified as grassland swale wetlands (Elliott et al. 2014). The brush encroachment of mainland swale wetlands is due to both native (i.e., live oak) and non-native (i.e., Chinese tallow [*Triadica sebifera*]) invasive species, altering vegetation structure preferred by cranes and likely reducing the hydroperiod of the natural wetlands (USFWS 2010). Thus, cranes may have frequented constructed ponds more on the mainland because many of the natural, freshwater swale habitat there remained inaccessible to them. Future study will be needed to determine if brush control on wetland swales improves their accessibility to use by cranes.

An alternative explanation for cranes using constructed ponds more on the mainland than Matagorda Island is the effect on salinity of water control structures, which surrounded much of the marsh on Matagorda Island. West Marsh, a major saltmarsh system on Matagorda Island, is under partial structural marsh management, which is the use of water-control structures and levees to manage water flow (Rogers et al. 1994). The mainland has no such water control structures or levees in the saltmarsh. Passive flow through the culverts from the bay would not have prevented saltwater flow into West Marsh but may have created an area of lower salinity in the marsh as compared to the bay due to slowed water exchange. If the structural marsh management of West Marsh yields lower salinities, cranes may have less need for alternative water sources on Matagorda Island.

Recent drought conditions have led some resource managers to augment upland freshwater sources with additional well-fed ponds (CWS and USFWS 2007, Stanzel and Smith 2017). However, installation of new wells at constructed ponds may not be as effective as other management actions like brush control on wetland swales. We observed that well-fed ponds did not attract more cranes than those without a well. Resource managers also mow the perimeter of some constructed ponds on Blackjack peninsula to control brush encroachment (CWS and USFWS 2007, USFWS 2010). Mowing is usually conducted during late summer to avoid the disturbance of wintering cranes. However, if mowing was not completed during the summer and had to be conducted during the fall, the fewest crane visits to ponds occurred early in the morning. Morning mowing may therefore minimize disturbances to cranes.

We found that visits to constructed ponds by whooping and sandhill crane groups during drought was low: <0.15 and <0.28 times/pond/day, respectively. Assuming an average group size of 2.2 birds (Aransas National Wildlife Refuge, unpublished data) and extrapolating to ≈100 useable constructed ponds, no more than 33 whooping cranes per day visited these ponds, or just 6.5% of the winter 2019–2020 population of 506 whooping cranes (Butler et al. 2020). By similar calculation for sandhill cranes, which had a similar group size (2.3 birds/visit based on images), no more than 65 sandhill cranes per day visited constructed ponds. Sandhill cranes have a much larger wintering population in coastal Texas than whooping cranes (Guthery and Lewis 1979).

Incomplete detection of crane visits could have biased our estimates of visitation rates low. A concurrent study (Ritenour et al. 2016) placed camera traps at 5 ponds we monitored. During November 2013 through April 2014, their camera traps detected whooping crane visits on 37% of days on average whereas ours detected visits on 10% of days (mean difference = 49.8 days, 95% CI = –13.0 to 112.6, paired $t = 2.202$, df = 4, $P = 0.0925$). Whooping crane visits occurred at 4 of the 5 ponds and those 4 ponds were less than 375 $m^2$ in area. We detected whooping cranes at every pond Ritenour et al. (2016) did plus one pond they did not. These differences in visitation could arise for several reasons (e.g., camera failure, placement of baited trap sites, differences in camera focal points, and the sensitivity of the passive infrared detector). For example, one of our cameras failed during November 2013, which

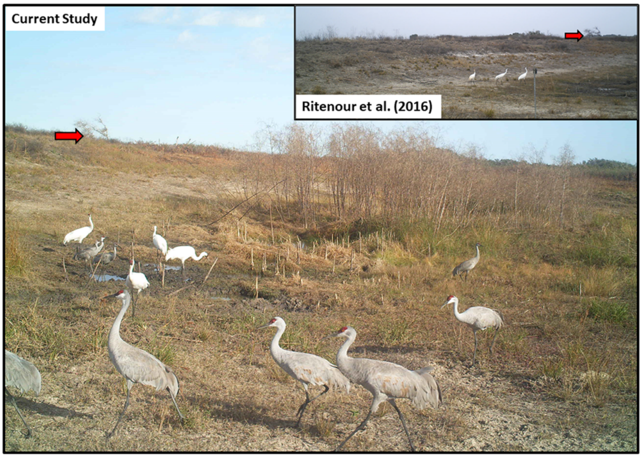


**FIGURE 3** Comparison of camera trap placement in current study and Ritenour et al. (2016) at a constructed freshwater pond on Aransas National Wildlife Refuge, Texas, USA, October 2013–May 2014. Red arrows indicate reference point common to both images.

eliminated 25 days of potential visitation. Further, our study coincided with another study on whooping cranes, which involved baiting areas with corn to attract birds for capture and tagging. Baiting associated with whooping crane capture occurred at 4 of the 5 ponds shared by the 2 studies. We do not know how baiting affected visitation rates or differences in detection between the 2 studies, but perhaps the Ritenour et al. (2016) camera traps were located closer to bait sites than ours. We know from our images that bait sites were placed on the opposite side of the pond from our cameras in 3 of the 4 cases (in the fourth case, we are unsure where the bait site was placed). Unfortunately, the Ritenour et al. (2016) images have been lost (E. Smith, International Crane Foundation, personal communication) so we cannot directly compare detections to determine if bait sites were in their view frame. However, we have one Ritenour et al. (2016) image that shows their camera was not focused on the pond but on areas adjacent to ponds (see Figure 3A in Bertram et al. 2015; Figure 3). Areas adjacent to ponds could be used for foraging or loafing instead of freshwater use.

Consistent missed detection over the winter period would not change our model results. Patterns of visitation indicating visits were proportional to migration phenology would remain. However, we might gain more power to detect avoidance of higher salinity ponds with greater sample size. Missed detection would result in biased estimates of visitation rates. However, even if we missed 75% of crane visits, pond visitation rates would still be low. For example, instead of 0.15 and 0.28 visits/pond/day during peak use, it would be 0.6 and 1.1 visits/pond/day for whooping cranes and sandhill cranes, respectively.

## MANAGEMENT IMPLICATIONS

The magnitude of resource use is not always the best indicator of the importance of that resource (Van Horne 1983), which makes it challenging to determine the importance of constructed freshwater ponds for whooping cranes with camera trap images. However, data such as ours can inform appropriate management actions. For example, visitation to constructed ponds on Matagorda Island was low compared to visits on the mainland. Brush has encroached onto 75% of the natural swale wetlands on the mainland. Brush encroachment could limit crane use of swale wetlands for freshwater resources, compelling their use of constructed ponds on the mainland. Restoration of swale wetlands through brush and invasive species removal could provide more freshwater resources without reliance on constructed ponds or installation of new wells (Stanzel and Smith 2017). Until such restoration is completed, however, continued management (mowing and brush control) of constructed ponds is prudent.

Restored swales would benefit cranes, waterfowl, and other coastal species by increasing available habitat and food resources, and providing freshwater. Such a strategy of restoration creates a more ecologically beneficial and fiscally responsible long-term investment. The use of restored swales and managed ponds by cranes could be monitored to compare crane use between them. Were cranes to use the restored swales as hypothesized, we anticipate constructed pond visits to decline and pond management phased out accordingly.

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## CONFLICTS OF INTEREST

The authors declare no conflicts of interest.

## DATA AVAILABILITY STATEMENT

Data used in this manuscript are available at https://ecos.fws.gov/ServCat/Reference/Profile/135204.

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## SUPPORTING INFORMATION

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