Management and Conservation

Effect of Wind Energy Development on Breeding Duck Densities in the Prairie Pothole Region

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ABSTRACT Industrial wind energy production is a relatively new phenomenon in the Prairie Pothole Region and given the predicted future development, it has the potential to affect large land areas. The effects of wind energy development on breeding duck pair use of wetlands in proximity to wind turbines were unknown. During springs 2008–2010, we conducted surveys of breeding duck pairs for 5 species of dabbling ducks in 2 wind energy production sites (wind) and 2 paired reference sites (reference) without wind energy development located in the Missouri Coteau of North Dakota and South Dakota, USA. We conducted 10,338 wetland visits and observed 15,760 breeding duck pairs. Estimated densities of duck pairs on wetlands in wind sites were lower for 26 of 30 site, species, and year combinations and of these 16 had 95% credible intervals that did not overlap zero and resulted in a 4–56% reduction in breeding pairs. The negative median displacement observed in this study (21%) may influence the prioritization of grassland and wetland resources for conservation when existing decision support tools based on breeding-pair density are used. However, for the 2 wind study sites, priority was not reduced. We were unable to directly assess the potential for cumulative impacts and recommend long-term, large-scale waterfowl studies to reduce the uncertainty related to effects of broad-scale wind energy development on both abundance and demographic rates of breeding duck populations. In addition, continued dialogue between waterfowl conservation groups and wind energy developers is necessary to develop conservation strategies to mitigate potential negative effects of wind energy development on duck populations. © Published 2012. This article is a U.S. Government work and is in the public domain in the USA.

Millions of glaciated wetlands and expansive grasslands make the Prairie Pothole Region (PPR) the primary breeding area for North America's upland nesting ducks (Batt et al. 1989). Wetland and grassland loss in the PPR due to settlement and agriculture has been extensive (Dahl 1990, Mac et al. 1998),

and conversion to agriculture continues to reduce available habitat for breeding waterfowl and other wetland- and grassland-dependent birds (Oslund et al. 2010, Claassen et al. 2011). During recent years, anthropogenic impacts in the PPR have expanded to include energy development (e.g., wind, oil, natural gas; see Copeland et al. 2011: table 2.1). From 2002 to 2011, industrial wind energy production has increased 1,158% (i.e., 769–9,670 MW), 205% during the past 5 years (United States Department of Energy [USDOE] 2011). Impacts from wind energy development including direct mortality from strikes and avoidance of wind towers and associated infrastructure have been widely documented for many avian species, including raptors, passerines, upland gamebirds, shorebirds, and waterfowl, as well as bats (Drewitt and Langston 2006; Arnett et al. 2007, 2008; Kuvlesky et al. 2007).

Wetland habitats in the PPR annually attract and support >50% of the breeding waterfowl population in North America (Bellrose 1980). The productivity and subsequent use of prairie wetlands by breeding ducks in the PPR are critical for the maintenance of continental duck populations (Batt et al. 1989, van der Valk 1989). Because of the potential for extensive wind energy development (USDOE 2008, 2011, Kiesecker et al. 2011), understanding the potential effect of wind power development on the use of wetland habitat by breeding duck pairs in the region is critical.

The potential impacts of wind energy development on breeding ducks are similar to other wildlife reviewed in Kuvlesky et al. (2007). Breeding pairs may abandon otherwise suitable wetland habitat, display behavioral avoidance thereby reducing densities of pairs using wetlands near wind turbines, and experience mortality from collision with turbines and associated infrastructure. Additionally, indirect effects on breeding ducks potentially include avoidance of associated grassland by nesting females, increased predation, or reduced reproduction. Wind towers and supporting infrastructure generally do not directly affect the wetlands that provide habitat for breeding ducks. However, ducks are sensitive to many forms of disturbance (Dahlgren and Korschgen 1992, Madsen 1995, Larsen and Madsen 2000). Avoidance related to the presence of towers, movement of blades (e.g., shadow flicker), blade noise (Habib et al. 2007), infrastructure development including roads and transmission lines (Forman and Alexander 1998, Ingelfinger and Anderson 2004, Reijnen and Foppen 2006), and maintenance activities have been documented for other avian species and may similarly affect breeding pairs and reduce the use of wetlands within and adjacent to wind farms.

The presence of wind energy development in high density wetland and breeding pair habitat in the PPR is relatively recent, and previous studies of the effects of land-based wind development on waterfowl (Anatidae) have focused primarily on collision mortality (Winkelman 1990, Johnson et al. 2000, Gue 2012) and the effect of wind farms on foraging behavior of wintering and migrating waterfowl (Winkelman 1990, Larsen and Madsen 2000, Drewitt and Langston 2006, Kuvlesky et al. 2007, Stewart et al. 2007). Wind development appears to cause displacement of wintering or migrating Anseriformes, and bird abundance may decrease over time (Stewart et al. 2007). However, habituation has been reported for foraging pink-footed geese (Anser brachyrhynchos) during winter (Madsen and Boertmann 2008). Displacement of duck pairs due to wind development could affect population dynamics similar to habitat loss (Drewitt and Langston 2006, Kuvlesky et al. 2007). However, little information exists on how land-based wind development affects the settling patterns, distribution, and density of duck pairs during the breeding season.
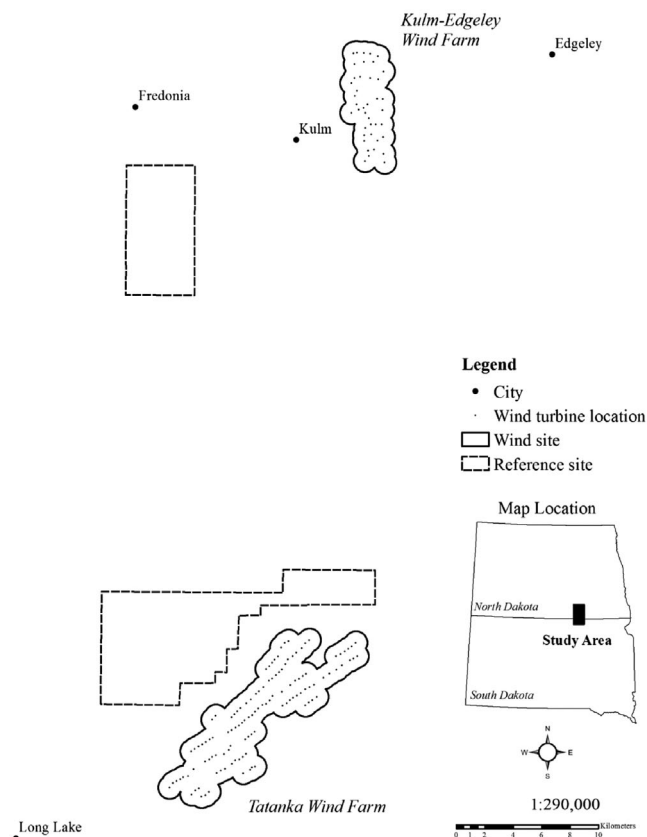
The number and distribution of breeding duck pairs in the PPR is related to annual wetland and upland conditions (Johnson et al. 1992; Austin 2002; Reynolds et al. 2006, 2007; U.S. Fish and Wildlife Service [USFWS] 2012). Wetland conditions in the PPR vary both spatially and temporally (Niemuth et al. 2010) and during dry years in the PPR, waterfowl are displaced to lesser quality habitats farther north (USFWS 2012) where productivity is generally reduced (Bellrose 1980). The long-term sustainability of breeding duck populations is dependent on availability and use of productive wetlands in the PPR that provide local breeding pair habitat when they are wet (Johnson and Grier 1988). Avoidance of wetlands near wind energy development by breeding ducks on otherwise suitable wetland habitat may result in displacement to lesser quality habitats similar to the effect of displacement during dry years. Given the relatively large development footprint (i.e., unit area/GW) for energy produced from wind relative to other energy sources such as coal (e.g., 7.4 times; wind = 72.1 km^2/TW-hr/yr, coal = 9.7 km^2/TW-hr/yr; McDonald et al. 2009) and the projected growth of the industry (USDOE 2008), a relatively large land area and subsequently a large number of wetlands and associated duck pairs in the PPR can potentially be affected.

We assessed the potential effects of wind energy development and operation on the density of 5 common species of breeding ducks in the PPR of North Dakota and South Dakota: blue-winged teal (Anas discors), gadwall (A. strepera), mallard (A. platyrhynchos), northern pintail (A. acuta), and northern shoveler (A. clypeata). Our objective was to determine whether the expected density of breeding duck pairs differed between wetlands located within land-based wind energy production sites (hereafter wind sites) and wetlands located within paired sites of similar wetland and upland composition without wind development (hereafter reference sites). We predicted that if disturbance due to wind energy development caused avoidance of wetlands by breeding duck pairs, then expected density of breeding pairs would be lower on wind energy development sites. We interpreted differences in estimated breeding pair densities between paired wind energy development sites and reference sites in the context of the current Prairie Pothole Joint Venture (PPJV) waterfowl conservation strategy for the United States PPR (Ringelman 2005).

STUDY AREA

We selected operational wind energy and paired reference sites as a function of the geographic location, the local wetland community and its potential to attract breeding pairs (i.e., ≥40 pairs/km^2; Reynolds et al. 2006), and wetland conditions. In 2008, 11 wind farms were operational in the PPR of North and South Dakota, USA. Of those, only 3 were located in areas with the potential to attract relatively large numbers of breeding duck pairs for the 5 species in this study (Loesch et al. 2012, OpenEnergyInfo 2012). We identified 2 existing wind energy production sites in the Missouri Coteau physiographic region (Bluemle 1991) of south-central North Dakota, USA, and north-central South Dakota, USA (Fig. 1). Both wind sites contained wetland communities with the potential to attract an estimated 46 breeding duck pairs/km^2 (mean density = 8.5 pairs/km^2 for the PPR; Reynolds et al. 2006, Loesch et al. 2012). The Kulm-Edgeley (KE) wind energy development consisted of 41 towers in a cropland-dominated landscape (e.g., 83% of

Figure 1. Paired study sites with and without wind energy development surveyed for breeding waterfowl pairs in North Dakota and South Dakota, USA, 2008–2010.

ESRI ArcGIS 9.2 software (ArcGIS Version 9.2, Environmental Systems Research Institute, Redlands, CA) and United States Department of Agriculture National Aerial Imagery Program (NAIP) imagery (ca. 2007).

The potential zone of influence for breeding waterfowl from a wind turbine to a wetland during the breeding season is unknown. The limited research that has been conducted to measure displacement of birds in grassland landscapes has primarily targeted migratory grassland passerines, and has identified relatively short (e.g., 80–400 m) distances (Leddy et al. 1999, Johnson et al. 2000, Shaffer and Johnson 2008, Pearce-Higgins et al. 2009). Compared to grassland passerines, waterfowl have relatively large breeding territories and mallards use multiple wetlands within their home range (e.g., 10.36 km^2 generalized to a circle based on a 1,608 m radius; Cowardin et al. 1988). Because the objective of this study was to test the potential effects of wind energy development on breeding duck pair density and not to identify a potential zone of influence, we chose a buffer size with the objective to spatially position sample wetlands in proximity to 1 or many turbines where a potential effect of wind energy development would likely be measurable. Consequently, we used the generalized home range of a mallard hen and buffered each wind turbine by 804 m (i.e., half the radius of a circular mallard home range; Cowardin et al. 1988), to ensure overlap of breeding territories with nearby wind turbines. The wind sites contained different numbers of turbines and as a result the sites were not equally sized (KE wind site = 2,893 ha; TAT wind site = 6,875 ha; Fig. 1).

We derived wetland boundaries from digital USFWS National Wetlands Inventory (NWI) data. We post-processed NWI wetlands to a basin classification (Cowardin et al. 1995, Johnson and Higgins 1997) where we combined complex wetlands (i.e., multiple polygons describing a basin) into a single basin and then classified them to the most permanent water regime (Cowardin et al. 1979). Wetlands partially or completely within the buffer areas were considered treatment wetlands.

For each of the 2 wind sites, we employed a rule-based process to select paired sites to control for differences in wetland and landscape characteristics among sites. We first

uplands were cropland; Table 1) and was located 3.2 km east of Kulm, North Dakota, USA. The Tatanka (TAT) wind energy development, consisted of 120 towers in a perennial cover-dominated landscape (e.g., 92% of uplands were perennial cover; native grassland, idle planted tame grass, alfalfa hay; Table 1) and was located 9.7 km northeast of Long Lake, South Dakota, USA. The KE site began operation in 2003; approximately 50% of the TAT towers were operational by 28 April 2008 and all were operational by 21 May 2008. Turbine locations were on-screen digitized using

Table 1. Characteristics of wetland (i.e., number, area [ha], % of total wetland area) and upland (i.e., area [ha], % of total upland area) areas in development (wind) and paired reference sites in North Dakota and South Dakota, USA, where we surveyed wetlands for breeding duck pairs during spring 2008, 2009, and 2010. Sites included Kulm-Edgely (KE) and Tatanka (TAT) Wind Farms.

Class	KE wind			KE reference			TAT wind			TAT reference		
	Number	Area	%	Number	Area	%	Number	Area	%	Number	Area	%
Wetland												
Temporary	272	41.4	9	283	41.7	7	362	29.9	3	462	97.3	8
Seasonal	372	167.2	37	240	347.3	55	917	253.5	29	815	419.9	36
Semi-permanent	37	239.5	53	37	242.9	38	322	581.7	67	231	636.5	55
Total	681	448.1		560	631.9		1,601	865.0		1,508	1,153.7	
Upland												
Perennial cover[a]		416.3	16		1,324.4	37		5,428.4	92		6,039.7	85
Cropland		2,120.5	83		2,232.8	63		455.3	8		1,064.1	15
Other		6.6	<1		13.4	<1		18.3	<1		11.4	<1
Total		2,543			3,570.6			5,902.1			7,115.2	

[a] Includes native grassland, undisturbed grassland, and alfalfa hay landcover classes.

considered physiographic region and proximity to wind sites when identifying potential reference sites. To reduce the potential for environmental variation, especially wetness (Niemuth et al. 2010), between wind and reference sites, we only considered sites <25 km from the nearest turbine and within the Missouri Coteau physiographic region. Additionally, we assumed that wetlands >2.5 km from the nearest turbine were beyond a potential zone of influence. Using the distance and physiographic region criteria, we identified 3 potential reference sites of similar size for each wind site based on upland land use (i.e., proportion of cropland and perennial cover) and wetland density. For the 6 potential sites, we compared the wetland number and area (ha) for each class (i.e., temporary, seasonal, semi-permanent) between each potential reference site and the respective wind site to select the most similar reference site (Table 1). The KE reference site was located 11.3 km west of the KE wind site and the TAT reference site was located 3.2 km northwest of the TAT wind site (Fig. 1).

We identified 5,146 wetland basins encompassing 3,410 ha from NWI data within the wind and reference sites and considered each wetland a potential sample basin. Only temporary, seasonal, and semi-permanent basins were present at the wind sites so we did not survey lake wetlands at reference sites. We did not survey basins that extended >402 m from the boundary of a site to eliminate linear wetlands that potentially extended long distances from the wind and reference sites.

METHODS

Surveys

We surveyed sample wetlands during spring 2008, 2009, and 2010 to count local breeding duck pairs. We used 2 survey periods (i.e., 28 April–18 May, early; and 21 May–7 June, late) to account for differences in settling patterns for the 5 species (Stewart and Kantrud 1973, Cowardin et al. 1995) and to reduce potential bias associated with differences in breeding chronology among species (Dzubin 1969, Higgins et al. 1992, Naugle et al. 2000). We divided the wind and reference sites into 3 crew areas to spatially distribute survey effort across the sites, and crews of 2 observers conducted surveys on each of the 3 crew areas daily. The detection probability of duck pairs was likely not equal among observers (Pagano and Arnold 2009) and we minimized potential confounding of detection, observer, and survey area by rotating observers among crew areas and partners daily. Additionally, our analytical approach was not to compare population estimates for wind and reference sites, which may require development of correction factors (Brasher et al. 2002, Pagano and Arnold 2009), but rather to compare expected rates of pair abundance. Consequently, we assumed non-detection of ducks to be equal among all sites.

We surveyed wetlands within each crew area in a 2.59-km grid pattern based on public land survey sections (PLSS). We used maps with NAIP imagery and wetland basin perimeters from NWI to assist orientation and navigation to survey wetlands. Permission, accessibility, wetness, numbers of wet-lands, size of wetlands, and numbers of birds affected the rate at which we surveyed PLSS. Surveys began at 0800 hours and continued until 1700 hours and were discontinued during steady rainfall or winds exceeding 48 km/hr. We surveyed most wetlands twice each year, once during each survey period. We visited all sample wetlands during the early survey period. We did not revisit wetlands that were dry during the early survey. Annual changes in access permission and wetland conditions due to precipitation resulted in some basins being surveyed during only 1 of the survey periods.

During the breeding season, waterfowl assemble into various social groupings that are influenced by sex ratios, breeding phenology, and daily activities (Dzubin 1969). We counted social groups of the 5 target species using established survey protocols (Hammond 1969, Higgins et al. 1992, Cowardin et al. 1995, Reynolds et al. 2006) and recorded observations for all sample wetlands that contained surface water regardless of whether birds were present or absent. We summarized field observations into 7 social groupings that we subsequently interpreted to determine the number of indicated breeding pairs for each species, basin, and survey period (Dzubin 1969, Cowardin et al. 1995). On average, the first count period (late April–early May) is regarded as an acceptable approximation of the breeding population for mallard and northern pintail (Cowardin et al. 1995, Reynolds et al. 2006). Consequently, we used observations during the early survey period to determine the number of indicated breeding pairs for mallard and northern pintail. Similarly, the second count period (late May–early June) is generally used to approximate the breeding population of blue-winged teal, gadwall, and northern shoveler (Cowardin et al. 1995, Reynolds et al. 2006) and we used observations during the late survey period to determine the number of indicated breeding pairs for these 3 species. We used indicated breeding pairs as the response variable in our models of estimated duck pairs.

We reduced disturbance during surveys by observing wetlands from 1 or more distant, strategic positions. We approached and surveyed portions of basins that were obscured by terrain or vegetation on foot. We noted birds leaving the wetland because of observer disturbance to minimize recounting on wetlands that we had not yet surveyed. We estimated the proportion of the wetland that was wet by visually comparing the surface water present in the basin relative to the wetland extent displayed on the field map. We recorded basins with no surface water as dry and not surveyed.

We used NAIP (ca. 2009) and on-screen photo-interpretation to develop a categorical variable describing the landcover of uplands (i.e., cropland, native grassland, idle planted tame grass, alfalfa hayland) adjacent to or surrounding all wetlands on the wind and reference sites. For wetlands touching multiple upland landcover classes, we assigned the class based on the largest wetland perimeter length. The exception was for idle planted tame grass, where we assigned the class if it touched any length of a wetland perimeter because of the limited presence of this class in

the landscape and its positive influence on pair settling densities (Reynolds et al. 2007).

Data Analysis

The objective of our analysis was to compare estimates of expected wetland-level abundance of breeding pairs on the wind and reference sites among years. We used past analyses of breeding duck pairs in the United States PPR and their relationship to wetland and upland parameters to inform the selection of candidate covariates (Cowardin et al. 1988, 1995; Reynolds et al. 1996). Wetland-level covariates included wetland class (i.e., seasonal, semi-permanent, or temporary; Johnson and Higgins 1997), surface area of water in NWI basin (wet area), and square root (sqrt) of wet area to reflect the non-linear response to wetland area demonstrated by breeding ducks in the PPR (Cowardin et al. 1988, 1995; Reynolds et al. 2006). We used a categorical variable for upland landcover (i.e., perennial cover, cropland) adjacent to the wetland for the only upland covariate (Reynolds et al. 2007).

Generalized linear models with Poisson errors provided an appropriate statistical framework for the analysis (McCullagh and Nelder 1989, McDonald et al. 2000). Preliminary summaries of the breeding pair data showed, however, that all 5 species displayed indications of over-dispersion relative to standard Poisson assumptions (i.e., both excess zeros and infrequent large counts; Appendix A, available online at www.onlinelibrary.wiley.com; Zuur et al. 2007). We addressed these challenges, while maintain an approach consistent with past studies by conducting a 2-stage analysis. We began by selecting appropriate models and subsets of the covariates using a likelihood-based approach. Then we used a simulation-based Bayesian approach to estimate parameters of species-specific statistical models, site- and year-level contrasts between wind and reference sites, and lack-of-fit statistics. Our combined approach allowed us to take advantage of the strengths of both approaches (Royle and Dorazio 2008:74–75) to provide a thorough analysis of the data.

We analyzed indicated breeding pairs from counts for each of the 5 study species using separate models. Full Poisson regression models described expected breeding pairs as a log-linear function of site, year, wetland class, landcover, wet area, and sqrt (wet area). We used Akaike's Information Criterion (AIC) differences (Burnham and Anderson 2002) to compare full Poisson models with Zero-Inflated Poisson (ZIP) models. The ZIP models partially accounted for potential excess zeros due to 2 sources: 1) non-detections and 2) unoccupied, but suitable, wetlands. The ZIP models described the data as a mixture of the counts described by the log-linear model and a mass of excess zeros described by a logit-linear model (Zuur et al. 2007). We conducted a comparison of Poisson and ZIP models between the full Poisson model and ZIP model that included a single additional parameter describing the expected probability of a false zero. When AIC differences indicated the ZIP model was more appropriate (i.e., $\text{AIC}_{\text{Poisson}} - \text{AIC}_{\text{ZIP}} \geq 4$), we used ZIP models for all subsequent analysis. When ZIP models

were selected, the full logit-linear model for excess zeros included covariates describing the upland vegetation cover class associated with each wetland (cover class; Stewart and Kantrud 1973), the area of the NWI basin covered by water (wet area), and the square root of wet area.

We expected that the full models would likely be most appropriate for the study species, as they were parameterized with covariates that have been identified as useful predictors of pair abundance in the Four-Square-Mile Breeding Waterfowl Survey (FSMS) dataset, which has been collected by the USFWS National Wildlife Refuge System since 1987 (Cowardin et al. 1995; Reynolds et al. 2006, 2007). Nonetheless, we sought to efficiently use the information in our less-extensive dataset by ensuring that we had selected a parsimonious subset of the covariates for each species-specific model. We removed a single covariate, or group of covariates in the case of factor variables, from the full model, ran the resulting reduced model, and recorded its AIC value (Chambers 1992, Crawley 2007:327–329). We repeated this procedure for every covariate. This resulted in a vector of AIC values that described, for each covariate, or covariate group, the effect of its removal on the AIC value of the full model. Reduced models for each species contained the set of covariates in the full model or the subset of covariates that resulted in increases in AIC values greater than 2 units per estimated parameter when they were removed from the full model (Arnold 2010).

After selecting a model structure for each species, we estimated the posterior distributions of model parameters with Markov Chain Monte Carlo (MCMC) simulation (Link and Barker 2009) in the Bayesian analysis software WinBUGS 1.4.1 (Spiegelhalter et al., 2003). The structure of the Bayesian ZIP models differed from the maximum likelihood models in 2 ways. The 12 site and year combinations were hierarchically centered and parameterized as normally distributed displacements from a common intercept (Gelman et al. 2004, Congdon 2005), and extra-Poisson variation due to large wetland-level counts was accommodated by a normally distributed error term (Appendix B, available online at www.onlinelibrary.wiley.com).

We conducted all statistical analyses in the R environment (R Development Core Team 2011). We used the generalized linear models capability of base R and the contributed package pscl (Jackman 2008) to estimate likelihoods and AIC values for Poisson and ZIP models. When selecting models and subsets of the covariates, we considered AIC differences greater than 4 to provide good evidence in favor of the model with the smaller value (Burnham and Anderson 2002). To generate Bayesian estimates of model parameters, we used the contributed R2WinBugs (Sturtz et al. 2005) package to run MCMC simulations in WinBUGS via R. For each model, we ran 2 Markov chains for 500,000 iterations and discarded the first 100,000 iterations from each chain to minimize the influence of starting values and prior distributions. We used minimally informative prior distributions and random starting values for model parameters and random effects. We evaluated convergence to the posterior distribution by examining plots of sequential draws for

each parameter and also by the Gelman–Rubin statistic (Gelman et al. 2004). We estimated the number of uncorrelated samples generated by each Markov Chain by the Effective Sample Size (ESS; Kass et al. 1998, Streftaris and Worton 2008). We required at least 200 uncorrelated samples per chain for inference. We considered a model to have converged when its Gelman–Rubin statistic was <1.1 and the plots of sequential draws indicated that the chains had stabilized and were sampling from a similar space (Gelman et al. 2004). We tested for lack-of-fit of the model using a posterior predictive test (Gelman et al. 2004). Specifically, we compared the variance-mean ratio for the observed data to the variance-mean ratio of simulated data generated from the posterior draws of model parameters. We concluded that the model fit the data if the posterior proportion of simulated variance-mean ratios that exceeded the observed variance-mean ratio was greater than 0.01 and less than 0.99 (Congdon 2005). We then used the CODA (Plummer et al. 2009) package to summarize the posterior distributions of model parameters, convergence diagnostics, and derived quantities like lack-of-fit statistics and back-transformed estimates of abundance. Using the 800,000 posterior simulations from each model, modal values of categorical covariates, and median values of continuous covariates, we calculated species-, site-, and year-specific medians and 95% credible intervals of 1) the estimated posterior distribution of the log-scale model parameters, 2) the estimated posterior distribution of expected pair abundance on wetlands of median area, and 3) the estimated posterior distribution of the back-transformed contrast in expected pair abundance between wind and reference sites in each year. These quantities provided the basis for comparison of pair abundance between wind and reference sites.

We used point estimates of pair density for the median seasonal wetlands size (i.e., 0.2 ha) in grassland to assess the potential effect of wind energy development on breeding duck pair densities. We selected seasonal wetlands because they were the most numerous wetlands in our sample (58%) and because breeding duck pairs use seasonal wetlands at greater rates than other wetland classes (see Reynolds et al. 2006, 2007; Loesch et al. 2012); most pairs (54%) were observed on seasonal wetlands.

We evaluated the potential impact of wind energy development from both a statistical and biological perspective. We compared point estimates of density among sites and within years to either support or reject an effect. We assessed the potential biological impact of breeding pair avoidance of wind sites by calculating the proportional change in the estimated density of pairs between wetlands in wind and reference sites for each species and year. The percent change reflects the potential impact to breeding duck populations in the presence of wind energy development.

RESULTS

As a result of variable wetland conditions both within and among years, and annual changes in access to private land, we surveyed different numbers and area of wetland basins each year. Water levels in wetlands were low during 2008 and 35% of wetland basins visited during the early count contained water and generally were only partially full (e.g., seasonal regime, mean = 54% full, n = 684). Water levels increased in 2009 and 2010 and only 15% of 2,464 and 12% of 3,309 wetland basins, respectively, were dry during the early count. Basins containing water were also more full during 2009 (e.g., seasonal basin mean = 103% full, n = 1,089) and 2010 (e.g., seasonal basin mean = 93% full, n = 1,407). We conducted 5,339 wetland visits during the early count and 4,999 wetland visits during the late count. During the early count, we observed 5,287 indicated breeding pairs of mallard (3,456 [range = 146–552]) and northern pintail (1,831 [range = 51–310]), and 10,473 indicated breeding pairs of blue-winged teal (5,886 [range = 180–984]), gadwall (2,839 [range = 75–506]), and northern shoveler (1,748 [range = 55–318]) during the late count.

Model Selection and Estimation

Our ZIP models provided a substantially better fit than Poisson models for every species. Differences in AIC ($AIC_{poisson} - AIC_{zip}$) were 426 for blue-winged teal, 137 for gadwall, 218 for mallard, 384 for northern pintail, and 78 for northern shoveler. All of the covariates in the full model were retained for mallard, northern pintail, blue-winged teal, and northern shoveler. Wetland class was dropped for gadwall. Differences in AIC between the full model and the nearest reduced model were 11 for blue-winged teal, 3 for gadwall, 26 for mallard, 6 for northern pintail, and 29 for northern shoveler. The MCMC simulations converged for every species-specific model, indicating that the parameter estimates and credible intervals from these models provided a sound basis for inference. The maximum upper 95% credible interval of all R-hat values for any structural parameter was 1.01 for blue-winged teal, 1.01 for gadwall, 1.01 for mallard, 1.02 for northern pintail, and 1.04 for northern shoveler. The posterior predictive test indicated that the models fit the data for every species. The proportion of simulated variance-mean ratios that exceeded the observed variance-mean ratio was 0.52 for blue-winged teal, 0.75 for gadwall, 0.61 for mallard, 0.59 for northern pintail, and 0.72 for northern shoveler. Minimum effective sample sizes were 709 for blue-winged teal, 553 for gadwall, 307 for mallard, 346 for northern pintail, and 612 for northern shoveler.

Estimates

Differences in estimated breeding duck pair densities in a wind site and a reference site varied among site pairs (2), years (3), and species (5), and posterior median values of these 30 contrasts ranged from −0.281 to 0.130 (Table 2). Estimated patterns of contrasts for expected breeding duck pair density between wind and reference sites were similar for all species. Given median wet area and the mode of the categorical covariates, expected, basin-level densities of duck pairs for the 5 species was either statistically indistinguishable (14 of 30) between wind and reference sites or was lower (16 of 30) on wind sites than reference sites depending on site, year, and species (Fig. 2). Regardless of whether 95% credible intervals overlapped zero, density estimates were

Table 2. Log-scale estimated posterior medians and 95% of the estimated posterior distribution from the count portion of a zero-inflated, overdispersed Poisson model of indicated blue-winged teal (*Anas discors* [BWTE]), gadwall (*A. strepera* [GADW]), mallard (*A. platyrhynchos* [MALL]), northern pintail (*A. acuta* [NOPI]), and northern shoveler (*A. clypeata* [NSHO]) pairs on seasonal wetland basins for development (wind) and paired reference sites in North Dakota and South Dakota, USA. Sites are Kulm-Edgely (KE) and Tatanka (TAT) for years 2008 (08), 2009 (09), and 2010 (10).

Species	Site	Year	Reference			Wind		
			Median	2.5%	97.5%	Median	2.5%	97.5%
MALL	KE	08	0.47	0.21	0.73	0.15	−0.13	0.43
	KE	09	−0.49	−0.78	−0.22	−0.90	−1.17	−0.64
	KE	10	−0.42	−0.66	−0.20	−0.77	−1.04	−0.51
	TAT	08	0.29	0.02	0.56	0.41	0.17	0.65
	TAT	09	−0.38	−0.61	−0.14	−0.63	−0.89	−0.38
	TAT	10	−0.33	−0.55	−0.10	−0.47	−0.71	−0.22
BWTE	KE	08	−0.13	−0.25	−0.00	0.22	0.01	0.45
	KE	09	−0.46	−0.66	−0.27	−0.52	−0.74	−0.32
	KE	10	−0.13	−0.30	0.04	−0.58	−0.78	−0.39
	TAT	08	0.25	0.06	0.45	0.18	0.01	0.36
	TAT	09	−0.15	−0.32	0.02	−0.39	−0.58	−0.21
	TAT	10	0.03	−0.12	0.19	−0.19	−0.36	−0.02
NOPI	KE	08	−0.25	−0.61	0.12	−0.80	−1.24	−0.39
	KE	09	−0.80	−1.16	−0.45	−1.54	−1.93	−1.17
	KE	10	−0.72	−1.01	−0.42	−1.20	−1.56	−0.87
	TAT	08	−0.10	−0.46	0.27	0.16	−0.15	0.48
	TAT	09	−0.35	−0.63	−0.06	−0.76	−1.07	−0.44
	TAT	10	−0.15	−0.41	0.13	−0.38	−0.67	−0.07
GADW	KE	08	0.09	−0.17	0.37	−0.13	−0.43	0.18
	KE	09	−0.52	−0.77	−0.28	−0.91	−1.19	−0.64
	KE	10	−0.61	−0.83	−0.38	−1.42	−1.72	−1.14
	TAT	08	0.07	−0.18	0.34	0.17	−0.05	0.41
	TAT	09	−0.46	−0.69	−0.22	−0.55	−0.81	−0.29
	TAT	10	−0.69	−0.92	−0.46	−0.62	−0.86	−0.38
NSHO	KE	08	−0.35	−0.61	−0.08	−0.49	−0.79	−0.18
	KE	09	−0.91	−1.17	−0.67	−1.00	−1.29	−0.73
	KE	10	−0.78	−1.00	−0.57	−1.11	−1.39	−0.85
	TAT	08	−0.23	−0.49	0.00	−0.30	−0.52	−0.08
	TAT	09	−0.59	−0.80	−0.37	−0.99	−1.25	−0.74
	TAT	10	−0.36	−0.55	−0.16	−0.69	−0.90	−0.47

lower on sites with wind development for 26 of the 30 combinations (i.e., mallard and blue-winged teal: 12 combinations, 11 negative [range −6% to −36%], 7 did not overlap zero; gadwall, northern pintail, northern shoveler: 18 combinations, 15 negative [range −5% to −56%], 9 did not overlap zero). The general pattern of results were similar for all species, consequently, we chose a representative early and late arriving species with the largest number of indicated breeding pairs, mallard and blue-winged teal, respectively, for detailed presentation of results.

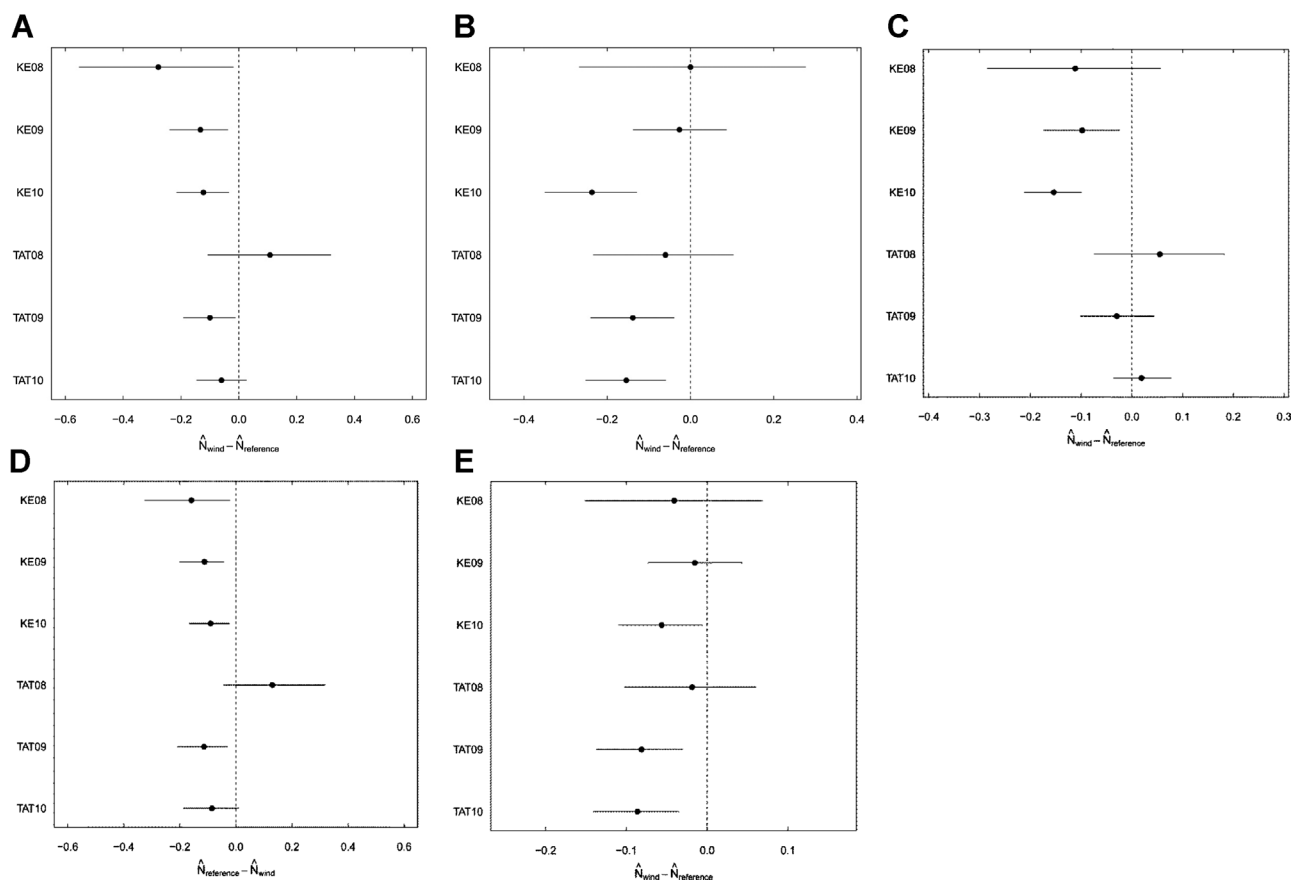## Mallard and Blue-Winged Teal

Mallard and blue-winged teal comprised 59% of the indicated breeding pair observations (i.e., 3,473 mallard; 5,928 blue-winged teal). Full models were retained for both mallard and blue-winged teal, and the point estimate of density was greatest in 2008 for both KE and TAT sites, but varied among years and sites (mallard: wind median = 0.42 [range = 0.30–1.03], reference median = 0.41 [range = 0.21–0.97]; blue-winged teal: wind median = 0.51 [range = 0.42–0.94], reference median = 0.66 [range = 0.47–0.96]). For mallard, estimated breeding pair densities on seasonal wetlands at wind sites were lower for 5 of the 6 site-year combinations (median = 0.11, range = −0.28 to 0.11) and error bars representing 95% of the posterior distribution of the estimate did not

overlap zero for 4 of the 6 site-year comparisons (Fig. 2A). Similarly, for blue-winged teal in 5 of the 6 site-year combinations, estimated pair densities were lower for seasonal wetlands on wind sites (median = −0.14, range = −0.24 to <0.01) and error bars representing 95% of the posterior distribution of the estimate did not overlap zero for 3 of the 6 site-year comparisons (Fig. 2B). Only 1 site-year combination for each of mallard and blue-winged teal suggested greater pair densities on wind sites, but in both cases 95% confidence intervals overlapped zero.

The estimated proportional change of mallard pair densities for wetlands in wind sites was negative in 5 of 6 site-year combinations (median = −10%, range = 13% [TAT 2008] to −34% [KE 2009]; Fig. 3A). The proportional change for blue-winged teal was also negative in 5 of 6 site-year combinations (Fig. 3B). The median estimate of proportional change for blue-winged teal densities between wind and reference sites was −18% (range 0% [KE 2009] to −36% [KE 2010]).

DISCUSSION

All 5 of our dabbling duck study species demonstrated a negative response to wind energy development and the reduced abundance we observed was consistent with behavioral avoidance. Avoidance of land-based wind energy development has been observed for numerous avian species during

Figure 2. Year-specific estimated differences between estimated posterior median abundance of mallard (*Anas platyrhynchos*; A), blue-winged teal (*A. discors*; B), gadwall (*A. strepera*; C), northern pintail (*A. acuta*; D), and northern shoveler (*A. clypeata*; E) on a seasonal wetland of median area (0.2 ha) embedded in perennial cover on a wind site and its corresponding reference site in North Dakota and South Dakota. Error bars represent 95% of the posterior distribution of the estimate. Site-year combinations are Kulm-Edgely (KE) and Tatanka (TAT) for 2008 (08), 2009 (09), and 2010 (10).

breeding (Leddy et al. 1999, Johnson et al. 2000, Walker et al. 2005, Shaffer and Johnson 2008, see Madders and Whitfield 2006), and does not imply complete abandonment of an area but rather the reduced use of a site (Schneider et al. 2003). This is consistent with our results, where breeding pairs continued to use wetland habitat at the wind sites but at reduced densities.

Our selection of paired wind and reference sites and analytical approach were designed to control for differences in site characteristics and annual variation in habitat conditions, and to use well-understood relationships between breeding duck pairs and wetlands (Cowardin et al. 1995; Reynolds et al. 2006, 2007). Despite the large amount of breeding pair data we collected, discerning if the presence of wind energy development was the ultimate cause of the lower estimated pair abundance on the wind versus reference sites is difficult. However, we did detect a directional effect of wind energy development sites over a 3-year period at the 2 sites that are representative of areas with greater estimated duck densities, and adds to the body of evidence suggesting a negative effect of wind energy development. Reduced wetland use in high density wetland areas with the potential to attract and support relatively greater densities of breeding duck pairs is of concern to waterfowl biologists and managers because when wet, these areas are vital to the sustainability of North

American duck populations. The somewhat limited temporal and geographic scope of our study and confounding between land use and duration of development prevents us from drawing strong conclusions about cumulative effects of wind energy development on breeding ducks (see Krausman 2011). Nonetheless, a 10–18% reduction in addition to other stressors is potentially substantial.

We observed larger negative displacement for most species and years in the KE wind site when compared to the TAT wind site. We found 2 notable differences in the wind sites that may have contributed to these results, the land use and age of development. The KE site was predominantly cropland and older than the grassland-dominated TAT site. The combination of multiple stressors, in this case agriculture and wind energy development, may have resulted in a greater impact to breeding ducks using wetlands in agricultural settings. Differences in estimated pair abundance between the cropland and grassland site suggest that greater habitat quality measured by the percent of grassland area and lack of cropping history in associated wetlands within a site may reduce avoidance of wind development when compared to agricultural landscapes. Breeding waterfowl may occupy wetlands at greater rates in grassland than cropland (Reynolds et al. 2007), nest success is generally greater in grasslands (Greenwood et al. 1995, Reynolds et al. 2001, Stephens et al.
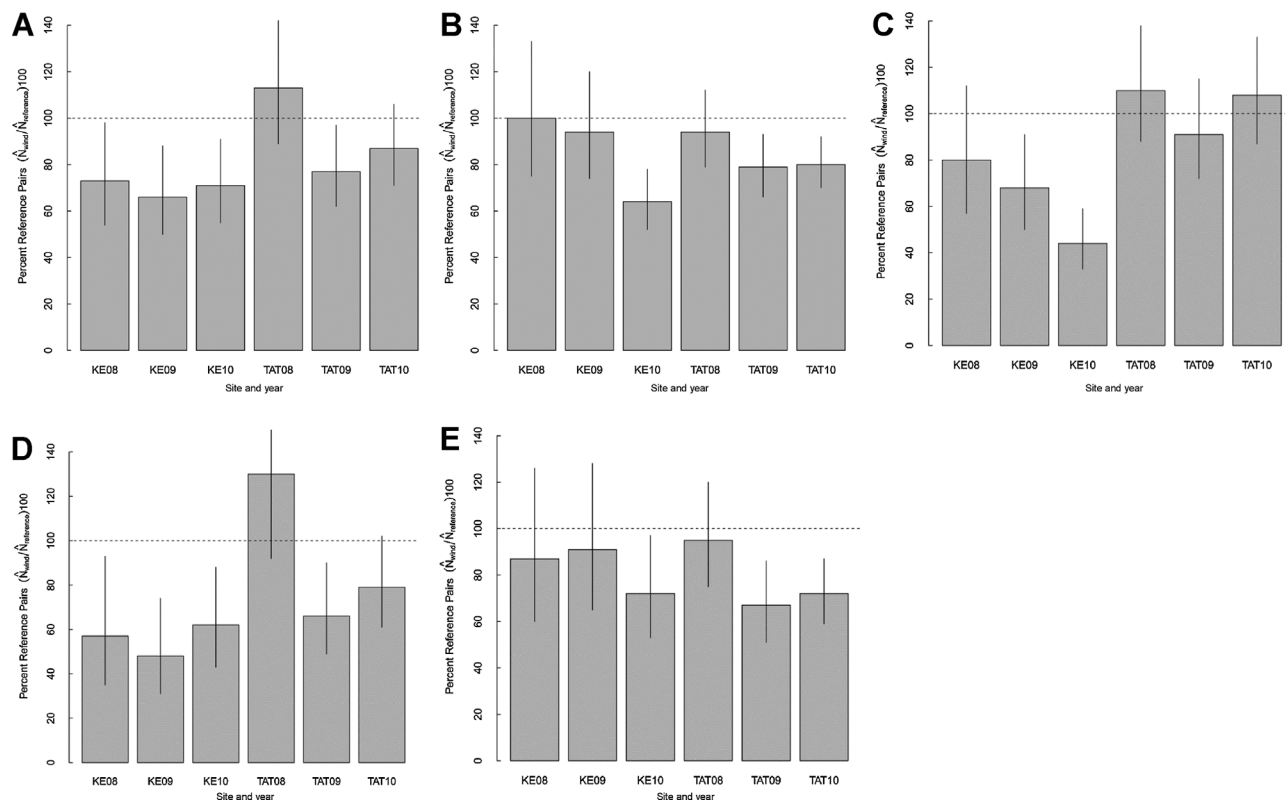
Figure 3. Year-specific estimated number of mallard (*Anas platyrhynchos*; A), blue-winged teal (*A. discors*; B), gadwall (*A. strepera*; C), northern pintail (*A. acuta*; D), and northern shoveler (*A. clypeata*; E) on a seasonal wetland of median area (0.2 ha) embedded in perennial cover on a wind site expressed as a percentage of pairs expected on the same wetland in the corresponding reference site in North Dakota and South Dakota. Error bars represent 95% of the posterior distribution of the estimate. Site-year combinations are Kulm-Edgely (KE) and Tatanka (TAT) for 2008 (08), 2009 (09), and 2010 (10).

2005), and wetlands in grass landscapes have greater occupancy rates by duck broods (Walker 2011), suggesting an overall greater productivity potential for breeding ducks in grassland versus cropland landscapes. The ability of intact habitat to reduce impacts of energy development is supported in current literature. In Wyoming, sage-grouse (*Centrocercus urophasianus*) residing in a fragmented landscape showed a 3 times greater decline in active leks at conventional coal bed methane well densities (1 well per 32 ha) than those in the most contiguous expanses of Wyoming big sagebrush (*Artemisia tridentata*) in North America (Doherty et al. 2010). A similar relationship has been document for large mammals. In the Boreal forest, woodland caribou (*Rangifer tarandus caribou*) populations could sustain greater levels of industrial development and maintain an increasing population when they resided in large forest tracts that were not fragmented by wildfires (Sorensen et al. 2008).

Our ability to support the hypothesis that habitat quality mitigates impacts could be confounded by time-lags in detecting impacts, as well as the potential for ducks to habituate to wind energy development over time but at a cost to individual fitness (Bejder et al. 2009). The KE wind site was cropland-dominated and began operation in 2003, whereas the TAT wind site was grassland-dominated and began operation in 2008, and was 3 years old during the final field season. Many recent studies for a variety of species and ecosystems have shown time lags between dates of first

construction and full biological impacts. In Wyoming impacts to sage-grouse in some instances doubled 4 years post-development versus the initial year of development (Doherty et al. 2010) and lags varied from 2 to 10 years (Harju et al. 2010). In some instances, full biological impacts may not be apparent for decades. For example, 2 decades passed before impacts of forest logging resulted in woodland caribou population extirpation within 13 km of logging (Vors et al. 2007). In a review paper on the effects of wind farms to birds on 19 globally distributed wind farms using meta-analyses, time lags were important in detecting impacts for their meta-analyses with longer operating times of wind farms resulting in greater declines in abundance of Anseriformes (Stewart et al. 2007). Pink-footed geese foraging during spring appear to have habituated to the presence of wind turbines in Europe (Madsen and Boertmann 2008). We therefore cannot distinguish between these 2 competing hypotheses without additional study.

Wind resources are both abundant and wide-spread in the PPR in the United States (Heimiller and Haymes 2001, Kiesecker et al. 2011), and the development of an additional 37 GW of wind energy capacity in the PPR states is necessary to meet 20% of domestic energy needs by 2030 (USDOE 2008). The projected wind farm footprint in PPR states to support this target is approximately 39,601 km². Even if recommendations for siting energy development outside of intact landscapes suggested by

Kiesecker et al. (2011) are implemented by the wind industry, millions of wetlands occur in agricultural landscapes and our results indicate that wind energy development will likely reduce their use by breeding duck pairs.

Waterfowl conservation partners in the PPR use strategic habitat conservation (Reynolds et al. 1996, 2006; Ringelman 2005; USFWS 2006; Loesch et al. 2012) in an adaptive management framework to target protection, management, and restoration based on biological and landscape information, primarily in response to habitat loss from agricultural activities. From a habitat quality and conservation perspective, wind energy development should be considered as another stressor relative to the cumulative effects of anthropogenic impacts on limiting factors to breeding waterfowl populations.

The protection of remaining, high priority grassland and wetland resources in the United States PPR is the primary focus of waterfowl habitat conservation (Ringelman 2005, Niemuth et al. 2008, Loesch et al. 2012). Population goals and habitat objectives were established to maintain habitat for breeding pairs and the current productivity of the landscape (Ringelman 2005, Government Accounting Office 2007). Spatially explicit decision support tools (Reynolds et al. 1996, Niemuth et al. 2005, Stephens et al. 2008, Loesch et al. 2012) have been used effectively to target and prioritize resources for protection. New stressors such as energy development in the PPR that negatively affect the use of wetland resources have ramifications to breeding waterfowl populations (i.e., potential displacement to lower quality wetland habitat) and their conservation and management. Thus, population and habitat goals, and targeting criteria may need to be revisited if large-scale wind development occurs within continentally important waterfowl conservation areas like the PPR.

MANAGEMENT IMPLICATIONS

Balancing the development of wind energy and current conservation efforts to protect habitat for migratory birds is complex because most conservation and wind energy development in the region occur on private land (USFWS 2011). Given that breeding duck pairs do not completely avoid wetlands in and adjacent to wind energy developments and resource benefits remain, albeit at reduced levels, the grassland and wetland protection prioritization criteria used by conservation partners in the PPR (Ringelman 2005) could be adjusted to account for avoidance using various scenarios of acceptable impact. For example, the wind sites used in our study are in high priority conservation locations (Ringelman 2005, Loesch et al. 2012). After accounting for effects of duck displacement by wind development, their priority was not reduced for either site. Consequently, wind-development does not necessarily preclude these sites from consideration for protection. Additionally, using the measured negative impact of wind energy development and production on breeding duck pairs, opportunities to work with wind energy industry to mitigate the reduced value of wetlands in proximity to wind towers should be investigated. Continued partnership by the wind energy industry and wildlife conservation groups will be critical for continued research. Further, we suggest expanding our research both spatially and temporally to better address cumulative impacts, zone of influence, impacts on vital rates, potential habituation or tolerance, and/or lag effects of long-term exposure to wind energy development.

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