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CRITICAL LITERATURE REVIEW:

IMPACT OF WIND ENERGY AND RELATED HUMAN ACTIVITIES ON GRASSLAND AND SHRUB-STEPPE BIRDS

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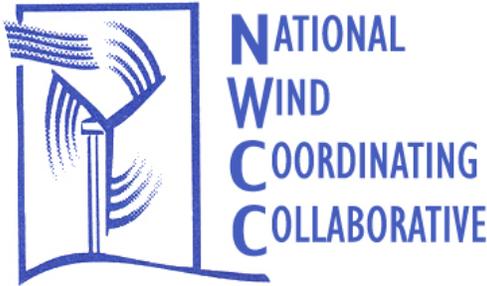


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A note about citations: As a courtesy to the reader, we have provided URLs whenever a cited paper is available online (other than papers published in scientific journals, which are often online, but on access-controlled websites). However, as these are materials that are published elsewhere, and the URLs are provided merely as a courtesy, we did not include “date accessed” because these are not “internet resources.” They are published resources that happen to be available on the internet. In contrast, where we cited material that is published only online, we included the “date accessed.”

Citations follow the formatting guidelines for the Journal of Wildlife Management. Following those guidelines, the citations give the official common names of bird species in lower case, regardless of the capitalization used by the authors in the original papers as published and regardless of the ornithological practice of capitalizing the official common names.

EXECUTIVE SUMMARY

Background and purpose of project

The Grassland and Shrub-Steppe Species Collaborative (GS3C), a subcommittee of the Wildlife Working Group of the National Wind Coordinating Collaborative (NWCC), commissioned this critical review of literature. This review pertains to the impacts of wind energy on grassland and shrub-steppe bird species. Its purpose was to examine the actual and potential impacts of wind energy facilities on grassland and shrub-steppe avian species. The impacts included mortality, avoidance, reductions in nesting success and adult survival, and behavioral changes.

Commercial wind energy began in the United States in the early 1980s and did not grow appreciably until 1999. Thus, there is relatively little literature, and most comprises site-specific pre-construction wildlife evaluations with post-construction assessments of actual impacts. Studies of sublethal impacts (behavioral responses such as avoidance) are even rarer. The GS3C therefore requested that other anthropogenic activities that are components of or share some common features with wind farms be included, in an attempt to understand potential impacts in the absence of a substantial body of literature. These other activities are roads, urbanization, tall structures (including telecommunications towers and electric transmission lines and stanchions), and oil and gas extraction facilities.

We conducted a comprehensive literature search that included “gray” literature – a wide range of papers, articles, summaries and transcripts of talks, and other materials that did not appear in the peer-reviewed literature – and research from around the world (provided that the paper was available in English). All papers were screened through quality and relevance filters. We considered studies that pertained to grassland or shrub-steppe species or habitats to be relevant. The goal of the quality screening was to focus on well-designed research with adequate sample sizes and sound statistical or qualitative analyses. The selection process was based on the premise that papers of greatest interest would be those from which inferences could be drawn. This aspect of the review was particularly critical because many key questions about human activities have not been studied, or have been studied inadequately. Further, despite some commonalities, the human activities covered by the review were so diverse that only studies that investigated and assigned causation were deemed useful for understanding which components of these activities are of concern in wind energy development. For example, if a study showed that traffic on heavily traveled roads leads to settlement of particulate matter on plants, which destroys food sources or nesting materials or sites, the low level of traffic on roads leading to wind farms is not likely of concern, at least insofar as this particular effect is concerned. In contrast, a paper that shows an effect of weather – such as low cloud cover – on the rate of collisions with obstructions such as telecommunications towers – has implications for any obstruction. Weaknesses in the design or analysis of the papers included in this review are identified and discussed. Some papers have significant weaknesses, but are the best (or only) studies that examined a particular aspect of the activity or behavior of interest.

Wind energy

Mortality caused by wind energy development has been more frequently studied than have other impacts, and the studies are site specific. We found no landscape-level or regional studies

of mortality, although combined fatality data from multiple independent studies in three regions of the country in which the study sites include some grassland or grassland-like habitat suggest that the lowest rate of mortality per megawatt of energy generated was 2.7 birds and the highest was 3.5 birds per megawatt. The lack of broader studies may be a function of the fact that studies of wind energy impacts tend to be initiated at the behest of regulatory agencies that respond to public concern or opposition, or by wind energy developers who seek to answer more general concerns about the impacts of wind energy. Studies have followed the development of the industry; thus, studies of Midwestern sites, which often include grassland sites, are fairly common. However, site variation made generalizing from the results difficult. Mortality appeared to be more strongly associated with migration than with local use of the area, although 30% of fatalities were resident or breeding birds. Passerines comprised 82% of the mortalities across nine sites. Few studies reported on the relative abundance of species at a site, and little inference could be drawn from the data.

From the few papers that reported behavioral impacts, interesting and useful – but somewhat contradictory – observations emerged. Birds avoided areas with turbines, and although most flew below rotor height, those flying within the span of the rotors adjusted their flight patterns to avoid the spinning blades. Another study determined that configuration of the wind farm affected avoidance behavior, as did distance to other anthropogenic features. Birds stayed farther from strings than from clusters; the overall loss of foraging habitat caused by avoidance was three times greater than the loss of foraging area. Site characteristics such as wind speed and terrain affected the birds' ability to avoid turbines, particularly if they entered the turbine area from below the turbines. In this particular study, there was no evidence that birds actively avoided the turbines.

Population-level impacts were difficult to assess, because there were so few papers, and they were not representative of all landscape and habitat types. The results of papers reviewed here were inconsistent. Two showed a reduction in breeding populations (passerines and raptors), whereas two others found no effect. However, the designs were such that the ability to detect a true difference was weak. One study showed a marked difference in species abundance at control plots, as well as increased species richness, but little difference in community composition. Of particular interest is that avian density did not increase when turbines were turned off. In a second study, no raptors nested in a wind plant area, compared to 5.94 nests per 100 km² in the surrounding area, which comprised similar habitat. However, the wind plant area would have otherwise been expected to support only two nests, so the finding might not have been significant. The results of a third study were mixed in that density of nonbreeding birds differed significantly from year to year and between the wind farms and the control sites, whereas the density of breeding birds was not significantly different between the sites. At an Oklahoma wind facility, nine of 22 breeding bird species (but only three grassland bird species) showed significant differences in density as distance from the turbines increased. Nine grassland bird species showed no difference in density as a function of distance. However, sample size was low and statistical power (ability to detect a difference) was probably inadequate.

Roads

Roads are key components of all the human activities included in this review. The papers discussed in this section assessed the edge effect of roads through grasslands, the quality of roadside habitat, and disturbance resulting from traffic. We found few studies of road-related mortality that focused specifically on grassland or shrub-steppe bird species. Behavioral impacts associated with edge effects were mixed, and none of the studies isolated the mechanism or mechanisms, other than noise and predation, responsible for observed effects. Loggerhead Shrikes, which are predatory, preferred nesting territories with road fence lines, probably as vantage points to spot prey, but nesting success was significantly lower than that at sites away from the fence lines. No statistical relationship between roads and nest success was found for Mallard, Gadwall, or Blue-winged Teal. Along two-lane roads bordered by agricultural land on one side and grassland on the other, Bobolink density increased as distance from edge increased, though forest edge had a greater impact than road or agricultural edge. Another study of Bobolinks found a very strong roadside edge effect up to 50 m from the road edge, although daily survival rates did not vary with distance from edge. In a recent study of the response of Lesser Prairie-Chickens to various human activities, unimproved roads had no effect on distance to nests, but only buildings had a greater impact than improved roads; nest distance averaged 859 yards from the road. For Red-winged Blackbirds, density and productivity were significantly lower – even to the extent that they constituted population sinks – along roadsides. In that study, predation was the only causal factor that explained this result. Abundance for five of eight grassland bird species was significantly lower along roads with drainage ditches and fences and planted with a non-native species than along trails (a trail is defined as a road with a single pair of wheel ruts). However, in agricultural areas where roadside vegetation was the only remaining natural grassland habitat, 10 grassland species used the habitat. Even if nesting density along roads was high, a predation rate of 52%, combined with mowing and cowbird parasitism, resulted in productivity that was no higher than that of the nearby agricultural fields. A study of managed and unmanaged roadside verges along roads with varying traffic intensity found no consistent correlations between nest density (or nest success) and road type or traffic volume. In fact, nest densities were highest along the busiest road. However, an experimental study that used artificial nests showed that road type and habitat adjacent to roadside habitat had no effect on predation rate, whereas roadside habitat and nest position (nests on backslopes) in that habitat were significant factors. Fences resulted in much higher predation rates.

Noise has been studied as one cause of the observed avoidance of roads, but the effect seems to vary with species. Studies of urban roads with significant traffic volume (5,000 to 50,000 vehicles per day) have been a major focus in the Netherlands. At lower traffic volumes, noise accounted for a reduction of more than 10% in density of seven species within 100 m of roads; higher traffic volumes resulted in a reduction of 40% in density of all 12 species studied within 100 m of roads. Yet another study that examined the effect of traffic volume found that low traffic volumes (3,000 to 8,000 vehicles per day) had no effect on the abundance of grassland bird distributions, although density within that range varied with patch size. At moderate volumes (8,000 to 15,000 vehicles per day), roads reduced breeding bird density within 400 m but had otherwise no effect on the distribution of grassland birds. At heavier traffic levels

(15,000 to 30,000 vehicles per day), roads affected presence and breeding density to a distance of 700 m. At the heaviest traffic volumes (30,000 or more vehicles per day), road effects extended to 1,200 m.

In Denmark, the avoidance of roads by Pink-footed Geese reduced available foraging habitat by 21% near large roads and 10% near small roads. Observations of a variety of human disturbances in the landscape led to the conclusion that wind turbines and other disturbances should be clustered or overlapped to minimize overall impact on habitat availability. In contrast, Barnacle Geese were unaffected by roads. No significant difference was found in grazing intensity regardless of the distance from the road.

Avian mortality from automobile collisions may be a function of the suitability of adjacent habitat for prey species. Barn Owls and Long-eared Owls suffered the greatest mortality near grain fields that were ideal habitat for voles and on roads that were at the same elevation as the surrounding habitat.

Urbanization

Urbanization entails a broad loss and degradation of habitat. Few studies of the impacts of urbanization on grassland and shrub-steppe species were found; the eight reviewed here document changes in bird species composition, richness, and abundance in relation to density of urban features. As wind farms are generally sited in open areas, the findings pertinent to urbanization may have limited relevance.

The impact of the growth of Tulsa, Oklahoma, from 1967 to 1991 was studied by contrasting two sites – a “low-density” rural area and a “high-density” rural area. Some species declined in both landscapes, others in only one landscape, and others did not change in either landscape. Protected grassland in an urbanizing area (Boulder, Colorado) maintained populations of 22 of 29 grassland bird species for nearly a century; however, four of these species declined significantly in Boulder County; the other seven disappeared entirely from the Boulder area. One of two raptor species in the same area declined significantly, but not until the latter half of the 40-year period of human growth, which suggests a possible threshold effect. The other species increased significantly and were found more commonly in the more densely developed areas. That same Boulder Open Space served as the setting for a study about the impacts of urbanization on grassland songbirds. Declines in songbird populations were determined to be significantly affected by the quality of the habitat; severe declines occurred when urban cover types in the 40-ha landscape area constituted more than 5% of the land cover. A similar result was seen in Sweden, where species richness declined in proportion to the extent of urban elements in the landscape.

Urbanization seemed to influence the life history traits of two populations of Lesser Prairie-Chickens. Higher female survival and lower reproductive effort (few re-nesting attempts within a single season) were observed in a population residing in an area with large open parcels and few fences, roads, or power lines. The other population suffered reduced female survival and higher reproductive effort, which resulted in population instability.

A reduction in vertebrate abundance in suburban areas (versus protected areas) was thought to cause partial brood loss in a suburban Florida Scrub Jay population because the last-hatched chicks starved. However, the results of this study suggest that starvation might result not from inadequate vertebrate abundance, but from the relative paucity of nest helpers, a prominent feature of Florida Scrub Jay behavior that involves younger, nonreproductive birds helping older birds to raise broods.

Tall structures

Budget and time constraints prevented us from reviewing the more than 1,000 papers in this category. However, earlier reviews of much of this literature led us to believe that most would not meet the quality criteria we had established. Much of it was purely observational and inadequately reported important variables. As the description of habitat was often lacking and many papers primarily reported species identification and counts of dead birds, identifying papers on grassland and shrub-steppe species would be extremely difficult. Therefore, a more general discussion is provided here to identify variables that might provide insight into causal mechanisms.

Mortality estimates for tall structures generally span orders of magnitude for any particular kind of structure. These estimates are based on biased observations, as per-structure estimates are often based on mass mortality events or nonrandom monitoring of structures that are suspected of causing mortality. If little or no mortality is observed, monitoring stops and the observations are not reported. There have been no randomized, landscape-level monitoring efforts. These per-structure estimates are then multiplied by the number of individual structures of a particular type, even though those estimates are probably not typical of the mortality rates for structures in that category. In addition, many studies result from regulatory inquiry into site-specific applications and employ the Before-After/Control-Impact (BACI) methodology, which, in the case of wind energy studies to date, comprised assessments of species composition and abundance before and after construction at construction and reference sites, but have not assessed mortality or changes in mortality rates.

Positive associations have been found between the incidence of avian collisions with telecommunications towers and structure height, lighting, and weather conditions (primarily fog, cold fronts, and storms). Numerous observers have reported nonlinear flight (e.g., circling behavior) around towers with lights, but the specific attributes of light – color, lighting type (steady versus flash, incandescent versus strobe) – have not been studied experimentally until very recently, and these data have not yet been published. Earlier reports about the impacts of light color and type are conflicting. Extinguishing the lights in buildings has been documented to reduce mortality, but as lights associated with tall structures tend to be aviation warning lights, this is not an option.

Oil and gas extraction

Elements of this activity overlap other categories discussed in this review, including roads and power lines. The analysis here focused on avoidance and other behavioral responses to oil and gas wells, though referenced studies focused on various elements of these features, including

noise, physical motion, associated structures such as pipelines, roads (as a type of habitat fragmentation and with regard to traffic), and habitat change that promotes the establishment of new pathogens into the environment. Avian population changes resulted primarily from avoidance rather than mortality; the loss of usable habitat resulted in population reductions.

Most studies we examined involved shrub-steppe species; some discuss the impacts on grassland passerines. Assessments of avoidance are more common than are studies that identify specific components associated with that avoidance. Retaining ponds that hold water pumped from coal beds that are in the process of extracting methane are ideal breeding grounds for the mosquito species that transmit West Nile Virus. The potential impact on sage-grouse and other species found in these areas is being evaluated. The water in these ponds may also be contaminated with petroleum and heavy metals, but the impact on birds has not been studied.

Some species do not avoid oil and gas wells. Prairie Falcons seemed unaffected, regardless of the density of wells, although they nested at some distance and reacted to blasting noise by flushing from nests or sitting upright.

Strong declines in sage-grouse populations in Alberta and Colorado have been amply documented, but the association with oil and gas development is weak because that development began at least two decades before population counts began. Sage-grouse will locate in or near oil fields if suitable habitat is available and if they can avoid paved roads and oil structures. However, power lines associated with oil and gas fields may lead to increased raptor predation and, in turn, lower population growth rates. Leks at a greater distance from compressor stations had more birds than did leks nearer to the stations, but whether this was due to the noise emanating from the compressor, roads, traffic, or habitat loss is not clear. Increased mortality of males at leks and a reduction in female population growth have also been attributed to wells. The impact of proximity to wells was significant – the negative impact was observed to a distance of 4.7 km for producing wells and 6.2 km for drilling rigs.

Two major studies of the impacts of oil and gas wells on sage-grouse were published in the past two years. The first comprehensive study examined relationships between breeding success and adult life history traits with numerous habitat conditions, including vegetation structure and composition, wetness (a measure of soil or surface moisture content derived from remotely sensed data), distance to wells, and other human activities such as roads. Chick and adult male mortality increased closer to wells and with increased well density. Male attendance at lek sites decreased closer to wells and with increased traffic volume, and in areas with high densities of wells. Females chose nest sites nearer to wells than would be expected by random distribution, but avoided areas with higher well density. Broods were also closer to wells than would be expected in a random distribution, but chick mortality increased closer to wells and with increased well density. In contrast, female sage-grouse that visited leks disturbed by wells established nest sites farther from those leks than did females that visited undisturbed lek sites.

Results with Lesser Prairie-Chickens were mixed: all nests at two sites were at a greater distance from wells than would be predicted by random distribution. At one site, the difference

in distances for all nests was statistically significant. However, at the other site, the difference in distances was not statistically significant for any of the nests.

A model of the relationship between distance to wells from sage-grouse nests, as well as 12 vegetation variables, predicted nest success 74.6% of the time. Another model combined two separate models for probability of use and habitat-associated risk for Greater Sage-Grouse. It was based on five vegetation characteristics and one parameter that combined all human activities in the area, which is called “edge,” though it does not actually measure distance to habitat edge. The model identified source habitat and attractive sink habitat (which has suitable vegetation characteristics but is also high risk) with 65% success in predicting nest location and 71% success in predicting brood location.

Traffic volume along roads associated with oil and gas wells affected sagebrush obligates such as Brewer’s Sparrow, Sage Thrasher, and Sage Sparrow. Declines of 60% were observed within 100 m of roads with higher traffic volumes (697 vehicles per day) compared to roads with lower volume (fewer than 344 vehicles per day). However, there was no observed difference in abundance within 100 m of pipelines and from 100 m to 200 m of pipelines.

Emergent questions and research needs

Though the body of literature is large, there are few well-designed studies with adequate sample sizes for all activities included in this review. However, the studies discussed here suggest hypotheses for further testing. Careful designs are needed to assess causation with regard to mortality and behavioral responses. Three design flaws were common to most of the studies reviewed here: (1) small sample size; (2) inappropriate sampling scale (temporal, spatial, or both); and (3) poorly described or controlled reference sites. Most studies lacked replicates. Some failed to adequately describe the habitat type and other habitat characteristics; few quantified those variables. Baseline studies of pre-impact conditions are almost universally lacking, although BACI studies are recommended for wind energy facilities by the NWCC Wildlife Workgroup, and are sometimes undertaken.

Also needed are multi-site, landscape-level studies that use stratified random samples to distinguish between the various attributes of activities that are thought to have negative impacts on avian behavior and survival. These large-scale studies will also result in much better mortality estimates, which will be useful for determining the needed level of mitigation and other conservation responses.

INTRODUCTION

Concern about the impact of wind energy facilities on wildlife has been a subject of discussion for more than a dozen years, and has increased with the growth of wind energy construction and operation. According to the American Wind Energy Association (AWEA), commercial-scale wind energy in the United States grew slowly after the first commercial wind facility came online in the 1980s. Commercial wind energy took 14 years to grow from 1,000 megawatts (MW) to 2,000 MW of generating capacity. Since 1999, however, “installed capacity” has grown fivefold to 11,803 MW in 2006 (AWEA 2006*a*). In 2006, more than 2,400 MW of wind capacity was added in the U.S. (AWEA 2007*b*), and there is significant wind power development in 30 states (AWEA 2006*b*). AWEA anticipates that 3,000 MW of wind power will be added in 2007 (AWEA 2007*a*).

As the number of wind facilities and the construction rate of new facilities have increased, so has the extent of the concern and attendant discussion. The National Wind Coordinating Collaborative (formerly Committee) (NWCC) (<http://www.nationalwind.org/>) was formed in 1994 to “identify issues that affect the use of wind power, establish dialogue among key stakeholders, and catalyze appropriate activities to support the development of environmentally, economically, and politically sustainable commercial markets for wind power.” The NWCC’s members include representatives from electric utilities and support organizations, state legislatures, state utility commissions, consumer advocacy offices, wind equipment suppliers and developers, green power marketers, environmental organizations, agriculture and economic development organizations, and state and federal agencies. When established, the NWCC formed an Avian Subcommittee in response to concerns about potential impacts of wind power on birds. In its early years, 1994 to 1998, the Subcommittee hosted three research meetings (proceedings available online at <http://www.nationalwind.org/events/past.htm>) and issued a document entitled “Studying Wind Energy/Bird Interactions: A Guidance Document” (NWCC, 1999). Between 2000 and 2002, the national focus on avian impacts decreased and the Avian Subcommittee became less active. However, interest rose again at the end of 2002; several scientific and conservation organizations gauged the level of concern among their members and expressed interest in becoming more active in the NWCC. Among these active participants were the Defenders of Wildlife, National Audubon Society, American Bird Conservancy, and the Nature Conservancy. Together with state and federal agencies, wind industry representatives, and other stakeholders, the NWCC Avian Subcommittee members agreed to a robust agenda for the group with an increased activity level and took on a new name, Wildlife Workgroup, to reflect concerns about potential impacts of wind power on bats and other wildlife. In recent years, the Wildlife Workgroup has hosted numerous workshops and national research meetings (November 2004 and 2006). It has formed a core group and several subgroups, producing additional publications and other information sources. For more information about the NWCC Wildlife Workgroup, please see the NWCC website: <http://www.nationalwind.org>.

Although the NWCC is the primary national forum for discussing wind energy issues, local interest is also generated when commercial wind energy facilities are proposed. Some examples of discussions about wildlife concerns with wind development are found in the

Horicon Marsh project in Wisconsin (Burquist 2005), the Cape Wind project (Cape Cod Times), and other projects from West Virginia to New York State. Congress directed the Government Accountability Office to study interim guidance issued by the U.S. Fish and Wildlife Service (USFWS), “to determine if such guidance provides adequate protection of migratory birds on Federal and non-federal lands in several mid-Atlantic states, particularly our State of West Virginia” (Rahall and Mollohan 2004). The report was issued in September 2005 (GAO 2005). In August 2007, the National Research Council, National Academy of Sciences, published another congressionally mandated report, “Environmental Impacts of Wind-Energy Projects” (NAS 2007).

Additionally, several major ad hoc meetings or conferences have been held, including a three-day conference titled, “Towards Wildlife Friendly Wind Power: A Focus on the Great Lakes (27-29 July 2006, Toledo Ohio) (USFWS 2006), a three-day meeting in Colorado (Wind Power and Wildlife, 23-25 January 2006), a meeting co-sponsored by AWEA and Audubon California (10-11 January 2006), and a meeting in Pennsylvania, focusing on wind energy in the Central Appalachians (Wildlife and Wind Energy, Kutztown University, 2 December 2006) (Kutztown 2006).

In 2003, the USFWS published interim siting guidance for avoiding and minimizing the impact of wind energy on wildlife (USFWS 2003). In April of 2007, the Secretary of Interior announced in the Federal Register the intent to form a Federal Advisory Committee to review the interim siting guidance. A number of conservation organizations have issued similar guidelines (NWCC 2007).

These myriad discussions and studies weigh the positive environmental attributes of wind energy – which is considered clean and renewable because it does not produce greenhouse gases or other airborne pollutants, nor does it deplete its energy source – against negative attributes such as potential wildlife impacts. The appeal of wind energy is obvious to those concerned about climate change, pollution, and depletion of traditional energy sources such as fossil fuel (and the attendant reliance on foreign energy suppliers, which in turn entails national security issues). Developers and proponents alike realize that there are also environmental considerations, such as the impacts on birds and bats. Their concern drives efforts to assess and reduce those impacts. On a more practical level, the local and state regulators who make licensing and siting decisions for wind facilities are beginning to delve into the questions about potential impacts to wildlife, because local opposition to particular projects puts the issue squarely before these regulators. Against this backdrop, in response to the growing interest in the development of wind power in grasslands and concerns about potential impacts of wind development on grasslands species, the NWCC Wildlife Workgroup formed a Grassland and Shrub Steppe Species Collaborative (GS3C) in 2005 to identify the impacts, if any, wind energy has on grassland and shrub steppe avian species.

The GS3C has contracted with the Ornithological Council for a critical review of the scientific literature about the impacts of wind on grassland and shrub steppe species. Specific goals of the project are to document:

- The most critical research elements needed to determine impacts of wind power projects on avian populations.
- The vulnerability and sensitivity of various avian species and populations, including the most sensitive avian species, most vulnerable avian populations, and the most critical avian habitat areas, to impacts of wind power projects.
- Limitations of past research to predict impacts of wind power projects on avian populations of grassland and shrub steppe habitats.

As some aspects of wind energy production are shared with other forms of energy production and other human activities, the scope of the literature review also includes the impacts of roads, oil and gas development, urbanization, transmission lines and substations, and telecommunication towers. The project also called for research gaps to be identified and a bibliography to be compiled. The inclusion of papers about other forms of energy production is not intended to provide a basis to compare the relative merits of different kinds of energy. Similarly, the discussion of other forms of anthropogenic impacts on wildlife is not intended to support a premise that the impacts of wind are more benign than the impacts of these other activities.

A critical literature review comprises a critical evaluation of a body of literature. The critical evaluation starts with a determination of the studies that are sufficiently relevant and informative to merit inclusion. As described more fully in the Methods section, that process entailed an assessment of the subject matter and the quality of the work. The intent was to devote attention to papers that could be useful for assessing the actual and potential impacts of wind energy on grassland and shrub steppe birds, or that shed light on specific aspects of wind energy that affect these birds, rather than to provide an abbreviated assessment of every paper that touches on the subjects covered by this review. The second stage of the critical evaluation pertains to the discussion of included papers. This discussion summarizes the papers, assesses the study design, and identifies problems that may limit the applicability of the results. For instance, avian studies may involve a methodology for counting birds. Grassland and shrub steppe birds often have somewhat cryptic coloration. Unless they are moving, vocalizing, or perched in a conspicuous place, they can be overlooked. Environmental factors such as noise or vegetation can also affect detection rates, as can observer training and skill. Therefore, the results of a study that relies on counts but does not correct for detection probability and observer bias will be less reliable than the results of a study that has considered these factors. This aspect of the analysis would also consider inconsistent outcomes of studies that examine the same phenomenon. When possible, the findings will be synthesized, though this is more than a summary or list. Rather, the synthesis identifies findings for which there is substantial evidence. For most of the topics considered in this review, synthesis was not possible. Few studies asked the same questions, few were conducted in sufficiently comparable conditions, and, in some cases – notably gas and oil production – there were too few studies to draw general conclusions.

The principal investigator for this project is Sarah Mabey, Ph.D. Dr. Mabey was assisted by Ellen Paul, Executive Director of the Ornithological Council. Steve Sheffield, Marty Piorkowski, Colleen Heise, and Curtis Smalling assisted with literature searches.

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METHODS

This critical literature review involved three stages. First, we collected all published or publicly available scientific documents written in English or with English abstracts that related to grassland or shrub-steppe birds and five types of human activities or structures: wind energy, roads, urbanization, tall structures (including transmission lines and communication towers), and gas and oil production. In this first step, we used relaxed standards to assess relevance; we cataloged and collected all documents that mentioned a relationship between human activity categories and grassland or shrub-steppe birds either in their abstract, introduction, or discussion. (See Appendix A for the complete bibliography of this first stage in the literature review.) We did not consider standard environmental impact assessments that provide a catalog of species in an area and an assessment of possible or probable impacts to be scientific documents.

Next, we subjected the documents to a narrower relevance filter. Each document was assessed for original or reviewed research and data to determine whether it provided empirical evidence for a relationship between grassland and shrub-steppe birds and any of the human activity categories. Papers that presented original research that was designed specifically to document relationships were accepted as “highly relevant” for the critical review. If a study was not designed specifically to address relationships between grassland and shrub-steppe birds and any of the human activity categories, but reported empirical evidence for such a relationship, we classified the paper as “relevant.” Papers that only mentioned findings from other papers or reported possible, untested relationships were classified as “not relevant” and were not included in the critical review. An exception was made for subject areas for which very little empirical work has been published. If a paper represented the only available source of information on a given human activity category, it was classified as “sole source” and included in the critical review regardless of relevance.

Finally, we assessed documents that passed the relevance filter (classified as highly relevant, relevant, or sole-source) for scientific merit and contribution to our understanding of relationships between grassland and shrub-steppe birds and the human activities under consideration. We evaluated scientific merit based on empirical data and study design and execution. Papers achieved the minimal level of scientific merit if they presented any empirical data. All papers containing empirical data were ranked into three levels of scientific merit from lowest to highest: high-quality description (HQD), empirical – weak inference (EW), and empirical – strong inference (ES). High-quality descriptive papers ranked lowest for scientific merit, as they cannot establish cause and effect. We classified papers as HQD if they provided detailed, documented observation but had no design or analytic basis for determining causality in relationships or for assessing statistical confidence in the findings. (Detailed descriptions are valuable for generating hypotheses to be tested through rigorous research.) We did not include papers that reported only mortality (e.g., tower kills) that did not also provide detailed contextual information (e.g., tower location, height, lighting characteristics). Studies designed to address questions about relationships between grassland and shrub-steppe birds and one or more of the human activities were classified as having higher scientific value. These papers

were further divided into those that allowed for weak inference (e.g., descriptive/comparative studies with statistically valid design) or strong inference (e.g., experimental or controlled studies). Reports of causal relationships between human activities and impacts on bird behavior or demographic parameters (e.g., reproductive success) carry the most weight if they are supported by research that is designed to provide strong inference (James and McCulloch 1995). Consequently, we consider any finding that is not supported by experimental or controlled studies to be suggestive rather than conclusive. These findings, again, have high value for establishing hypotheses and providing an initial description of possible relationships between human activities and grassland and shrub-steppe birds.

We then identified three groups of papers for each human activity category:

- The first group includes only relevant papers of the highest scientific merit (ranking of ES). We present a detailed summary of each paper in this group and rely on the findings from these studies as the central body of established, empirical information about relationships between human activities and grassland and shrub-steppe birds.
- The second group includes relevant papers ranked as EW or HDQ. We present a general summary of the main findings of these papers and outline the hypotheses and predictions suggested by their findings.
- The third group includes sole-source papers. We present a general summary of the main points of these papers individually and outline the hypotheses and predictions they suggest.

LITERATURE CITED - METHODS

James, F. C., and C. E. McCulloch. 1995. The strength of inferences about causes of trends in populations. Pages 40-51 *in* T. E. Martin and D. M. Finch, eds. *Ecology and Management of Neotropical Migratory Birds*. Oxford University Press, New York.

ASSESSMENT OF HUMAN ACTIVITIES

Wind Energy

Overview of relevant literature

Tall structures in open habitats may be particularly disruptive to avian behavior because they are novel elements in the environments of bird species that are not habituated to overhead presences. Noise and visual disturbances from turbine operations placed in shrub steppe or grassland habitats may disrupt breeding or other behaviors. Because there is so little shelter, these disruptions may be more difficult to avoid and may affect a larger area than if the turbine operations were in other habitats. Additional disturbances may develop from the maintenance and operation of wind energy facilities, including road construction and use, habitat alteration, fire suppression, and other management practices that may kill birds or disrupt their normal behaviors.

The most common studies about the impact of wind facilities on birds in grassland and shrub-steppe habitats document mortality at specific facilities. This is not unexpected; most studies are commissioned by wind energy companies to determine potential and actual mortality to satisfy regulatory concerns. Thus far, regulators seem to be concerned primarily, if not exclusively, with mortality. A much smaller set of studies document behavioral responses (e.g., changes in flight behavior) or effects on breeding bird density or distribution. A recent report by Young et al. (2006) documents breeding birds, nests, and reproductive success of Mountain Plovers (*Charadrius montanus*) at the Foote Creek Rim Wind Plant in Wyoming, but we found no similar papers for other grassland and shrub-steppe birds. Thus, examination of the literature assessing behavioral responses to human activities that share components of wind energy (such as roads, tall structures, and other visual disturbances) is useful. We present our review of relevant literature in three sections: habitat alteration and behavioral studies, mortality studies, and population impacts.

We have omitted original studies that document mortality at the Altamont Pass Wind Resource Area (APWRA), because it is generally considered an outlier. The designs of most of its turbines are outdated, and its topography and location lack relevance. However, the APWRA is the best-studied WRA sited in a grassland and shrub-steppe ecosystem. We refer readers to Curry and Kerlinger (1998) and Smallwood and Neher (2004), who discuss the causes and mitigation of mortality at the APWRA.

Impact summary

HABITAT AND BEHAVIORAL IMPACTS

We identified four papers that present the results of quality scientific studies about the behavioral responses of grassland and shrub-steppe birds to wind turbines (Osborn et al. 1998, Larsen and Madsen 2000, Barrios and Rodríguez 2004, Young et al. 2006). Two additional papers were assessed as weakly relevant and are not reviewed here because they do not contain detailed descriptions of habitat that would support their classification as relevant and

support their claims to controlled design (de Lucas et al. 2004, 2005). In a study of the diurnal abundance and flight behavior of birds near turbine strings, Osborn et al. (1998) found significantly fewer birds and significantly fewer species in the vicinity of turbine strings than at control sites and noted that birds adjusted their flight behavior to avoid the turbines. Larsen and Madsen (2000) reported that wintering Pink-footed Geese (*Anser brachyrhynchus*) avoided foraging within 100 to 200 m of wind turbines. Barrios and Rodríguez (2004) presented data suggesting that seasonal, locational, and species-specific behaviors affect the probability of turbine-related mortality for grassland and shrub-steppe birds. Young et al. (2006) documented the use by breeding Mountain Plovers of the Foote Creek Rim Wind Plant in Carbon County, Wyoming, from pre-construction baseline studies in 1994 through several construction phases beginning in 1996 and during operation of the turbines thereafter, culminating in 2006. The study reported and mapped the use of the area by individuals, but not in relation to the turbine strings. The population decreased during construction and slowly increased during the operations phase, but at the end of the study, had not yet reached pre-construction levels. However, the study documented an increase in the number of Mountain Plover nests during the operations phase of the project, and nest location seemed to be unaffected by distance to the turbine strings.

Osborn et al. (1998) conducted a study of diurnal avian flight behavior and abundance from 1994 to 1995 at the Buffalo Ridge Wind Resource Area (BRWRA) in Minnesota after the first phase of the facility's development. The purpose of their study was to monitor and describe seasonal movements, activity patterns, and the relative abundance of birds in relation to wind turbines. At the time of the study, the BRWRA consisted of three units, one of which contained 73 operating wind turbines (Kenetech KVS-33 model; approximate height of turbine 37 m with blades of 33 m in diameter) arranged in 10 strings. Habitat at the BRWRA was a mosaic of corn, soybean, and Conservation Reserve Program (CRP) fields. The researchers conducted activity surveys at two observation sites in each of the three units (1 "treatment" unit with turbines and 2 "reference" units without turbines), at three time intervals – two hours beginning at sunrise, two hours between 1100 and 1300, and two hours before sundown. Surveys consisted of a sequence of eight, 10-min observations during which observers recorded (1) the number and taxonomic identity of all birds seen in flight; (2) the direction of flight and bearing of the individual relative to the observer; (3) the estimated flight height above the ground; (4) the estimated flight distance from the observation point; and (6) the distance to the nearest turbine. Surveys were conducted every two weeks during spring and fall and once per month during the summer and winter after preliminary summer surveys indicated low levels of activity during those seasons. Abundance and activity estimates were not adjusted for observer biases or detection probabilities.

Osborn et al. (1998) reported 70 species and more than 37,000 individuals using the BRWRA. The most common species were Red-winged Blackbird (*Agelaius phoeniceus*), Mallard (*Anser platyrhynchos*), Common Grackle (*Quiscalus quiscula*), and Barn Swallow (*Hirundo rustica*). Red-winged Blackbirds were nearly three times more common than the next most common species. Only 2% to 4% of all birds counted were raptors. The average number of birds did not differ between units in 1994. In 1995, however, the average number of individuals was significantly

lower in the treatment unit that had operating turbines than in the two reference units without turbines. In both years, the average number of species per count was significantly lower in the treatment unit. Seasonal and diurnal differences in activity were noted. In 1994, more birds were counted in the September-October timeframe than in the July-August timeframe. In 1995, fall and summer counts were intermediate between the significantly low counts for November to March and the significantly high counts for April to June. More birds were counted during the morning than at midday.

Fewer birds flew near sites with turbine strings than at reference sites. Birds flying through turbine strings adjusted their flight patterns when turbine blades rotated. The majority of birds (70% to 75%) flew below rotor blade height (21 m), whereas 16% to 17.5% of birds flew at height ranges that put them at risk for collision with rotors (21 to 51 m). Most (74.8% to 80%) also flew at least 31 m away from turbines and only 5% to 14.1% flew within 16 m of turbines. Raptors and waterfowl were most likely to fly at heights and distances that put them at risk for collision, whereas passerines generally flew below rotor blade height. Specifically, American Kestrel (*F. sparverius*) had the highest recorded rates of flying within the risk zone defined by flight height and distance from turbines.

The authors concluded that turbines present little risk to birds that actively use the BRWRA because most fly at heights that would not bring them in contact with rotor blades. Additionally, most birds, particularly passerines, flew a suitable distance from turbines and therefore did not risk collision. Raptors and waterfowl, however, might be at higher risk for collision at this site.

Larsen and Madsen (2000) examined the foraging behavior of wintering Pink-footed Geese in relation to wind turbines and other anthropogenic elements in the landscape to determine avoidance distances and develop a spatial model of available versus utilized habitat. They conducted their research in northwest Denmark in a landscape that comprises primarily cultivated grassland, agricultural fields (winter cereal and stubble), and pig farms. The area is intersected by roads, hedgerows, and two power lines. It includes 61 wind turbines (200 to 600 kW, height 25 to 50 m) that are distributed across the landscape as solitary turbines or “wind farms” that comprise strings or clusters. To quantify avoidance distance, the authors surveyed the density of goose droppings along transects perpendicular to physical elements of interest (e.g., wind turbines). They defined the avoidance distance as the point at which dropping density reached 50% of the maximum recorded on a given transect. They measured wind farm avoidance distance for one large cluster that comprised three rows each of 10 turbines and one row of five turbines (35, 600-kW turbines) and one string (five, 225-kW turbines). They further assessed dropping density along two transects placed between turbines within the large cluster to determine whether geese used the space in the wind farm. All transects were placed in homogenous pastures. The authors created a geographic information system (GIS) data layer for the study area that included regional distribution of fields, settlements, roads, power lines, wind turbines, windbreaks (hedgerows), and forests. These spatial data were used to calculate available foraging habitat and the utilized portion of that habitat based on avoidance distances for each physical landscape element.

The results of this study indicated that avoidance distance depended on wind farm structure. The string farm had an avoidance distance of 75 to 125 m and the cluster farm had an avoidance distance of 175 to 200 m. Transects in the cluster farm had no goose droppings, indicating that geese avoided the interior of the cluster. Larsen and Madsen (2000) estimated that in individual fields, turbines decreased available foraging habitat by 8.5%. On a regional landscape level, wind energy development contributed to 4.1% of the loss of total pasture area and an additional 13% of available foraging habitat above losses caused by other anthropogenic elements.

The authors concluded that positioning of wind turbines relative to extant or planned physical/anthropogenic landscape features is critical to the overall impact of wind energy development on the availability of suitable habitat. For example, in their study area, the large cluster of wind turbines was located in a field that had otherwise little disturbance from roads, windbreaks, power lines, or settlements. Locating wind turbines such that their avoidance zones overlap avoidance zones associated with other anthropogenic elements may reduce the overall impact on habitat. However, the authors also cautioned that synergistic avoidance effects may arise from co-occurrence of multiple anthropogenic elements in the landscape.

Barrios and Rodríguez (2004) studied bird behavior, habitat use, and mortality in relation to the two oldest wind energy facilities in the Strait of Gibraltar in southern Spain. The E3 and PESUR wind facilities are located on hills and ridges within about 3 km of the coastline in a mosaic landscape that comprises scrubland, forest, and pasture. The E3 facility has 66 turbines (15 to 180 kW; 35 to 40 m tall; all tubular towers) and the PESUR facility has 190 turbines (100 to 150 kW; 27 to 45 m tall with “wind wall” configuration of alternating high/low towers; 155 lattice towers). The researchers conducted mortality surveys twice per week for one year at 15 randomly chosen sub-strings of tubular and lattice towers. Special surveys for Griffon Vulture (*Gyps fulvus*) carcasses were conducted once per week along the whole length of both farms. Data from mortality surveys were carefully adjusted for detectability (*as in original*) rates and disappearance. Behavioral observations were made at both wind facilities to quantify passes of all soaring birds within 250 m of turbines. Observers recorded turbine location, turbine status (rotating/not rotating), weather (wind speed, direction, and visibility), height and direction of bird flight before and during pass, and minimum distance between birds and rotor blades. Barrios and Rodríguez (2004) used these behavioral data to examine three possible explanations for “risk situations” (risk index = # passes within 5 m: total # passes) when soaring birds passed with 5 m of a rotating blade: (1) type of flight (straight, circling, or slope-associated); (2) height at which birds began pass (height 250 m from turbines); and (3) wind speed.

The results of mortality surveys and adjusted mortality estimates indicate that Griffon Vultures and Common Kestrels (*F. tinnunculus*) are the most vulnerable species in this area; their respective average mortality rates are 0.12 and 0.14 deaths per turbine per year. Deaths for these two species were clustered in time and space. All 30 Griffon Vulture deaths were recorded in winter, and most between December and February. Moreover, 15% of the turbines were responsible for 57% of vulture mortality. Turbine type (lattice versus tubular) was

excluded as a cause of differential mortality. The eight kestrel deaths were clustered in summer; and most occurred between mid-July and mid-August. The ages of half the kestrels killed could be determined; all were juveniles. All kestrel mortality occurred at PESUR and was apparently distributed evenly throughout the facility. Turbine type was not an important factor in kestrel mortality.

Griffon Vultures were observed making 4,809 passes over turbines during the one-year study (33% of all passes). There was a clear seasonal difference in vulture activity, and greater total numbers and variances occurred during the winter. The risk index for vulture passes increased from a range of 0.15 to 0.34 in summer to a range of 0.04 to 0.76 in autumn and winter. There was strong agreement between risk index values and mortality for Griffon Vultures in space and time. Explanatory models of factors affecting risk situations for Griffon Vultures at PESUR found that wind speed was negatively related to risk and that risk was greatest for circling flight compared to straight or slope-associated flight. The flight height of birds entering the turbine zone (250 m) strongly influenced risk such that birds entering from above turbine height had significantly lower risk than those entering from below. An interaction between flight height and wind speed suggested that wind speed did not affect risk for birds entering the turbine zone from above turbine height, but higher wind speeds would reduce risk for birds entering the turbine zone at low heights. Falcons as a group (*F. tinnunculus* and *F. naumanni*), Griffon Vultures, and Short-toed Eagles (*Circaetus gallicus*) all flew closer to turbines than expected from a regular distribution. Nine additional species of soaring birds were recorded making passes over the E3 and PESUR wind facility with low risk (total 7,881 passes and 56 risk situations). Of particular note were the data for White Stork (*Ciconia ciconia*) indicating 4,768 passes and zero risk situations. The next most common soaring bird in the area was the Black Kite (*Milvus migrans*), which made 2,567 passes with a risk index of only 0.018. Soaring behavior and proximity to turbines was a sound predictor of relative collision and mortality rates. The authors suggested that the higher mortality at PESUR may have been related to the sloping terrain and weak updrafts associated with the hilly rather than mountainous ridges and the locations of the turbines relative to areas of updraft. They suggested that Griffon Vulture mortality was mediated by flight behavior that in turn depended on site-specific wind and relief. They further concluded that flying birds did not actively avoid turbines, as had been reported for raptors at BRWRA (Osborn et al. 1998) and sea ducks at an offshore wind facility (Guillemette and Larsen 2002). Barrios and Rodríguez (2004) found no evidence to suggest that lattice towers were more dangerous than tubular towers or that poor visibility was associated with higher risk.

The primary purpose of the surveys conducted by WEST, Inc. (Young et al. 2006) was to document the use by Mountain Plovers of the Foote Creek Rim (FCR) Wind Plant in Carbon County, Wyoming. This ground-nesting species has been in decline for many years. Baseline surveys began in 1994, when the project was first proposed, and continued through construction and operation of the project, except in 1996, when the project owner went into bankruptcy and terminated all studies. The new owner resumed the surveys in 1997 and continued through 2006. From 1997 to 2000, several units totaling 133 turbines of several models were built on the southern end of the site (FCR Phase I). Twenty-nine transects were

established throughout the site, 300 feet apart and perpendicular to the turbine strings. The transects ran across the entire mesa on which the site was located, so transect length varied. The population rose from 51 in 1994 to 60 in 1995. No data were collected in 1996. In 1997, when construction began, the population declined to 41; in 1998, the population dropped to 31, and reached a low of 18 in 1999. It fluctuated for several years, and was recorded as 41 in 2006. From 1995 on, these data were corrected for detection probability. Density was highest at the northern end of the site, where there is a white-tailed prairie dog (*Cynomys leucurus*) colony; there is a well-recognized tendency of Mountain Plovers to use prairie-dog towns as breeding sites. The authors did not describe the distribution of Mountain Plovers relative to the turbine strings. Beginning in 1997, each observation of Mountain Plovers was plotted on study area maps. The analysis of distribution is presented in north-south bands and not in east-west bands, so whether more birds were observed at a greater distance (farther east) from the turbines than near the turbine strings, which are at the western edge of the mesa, cannot be determined from this report. However, the surveys also counted and located Mountain Plover nests and reported locations relative to the turbines for the post-construction years. In 2001, the one nest found was 337 m from the nearest turbine, but the single nest found the following year was only 63 m from the nearest turbine. Five nests were found in 2003, ranging from 79 m to 195 m from the nearest turbine (mean = 118 m). The means (82, 73, and 71 m) and ranges (60 to 112 m, 40 to 135 m, and 35 to 96 m) continued to decline in each of the next three years for three nests each in 2004 and 2005 and seven nests in 2006. Although nest success was higher for the five nests in 2003 (80%) than for the three nests found in each of 2004 and 2005 (67%), it increased to 86% in 2006, when mean distance and range were at the lowest points. The authors make no mention of agriculture on the site, but suggest that grazing may have occurred during some years. Mountain Plover nests often fail because some agricultural practices destroy nests and eggs and kill hatchlings and fledglings. The apparent absence of crop land at the FCR wind energy site may account for the observed nest success.

The surveys focused on censusing and mapping locations of adults, young, and nests, and described the behavior of the adults and young at the wind facility. The adults move on foot or in low level flight. During courtship, they engage in aerial display by flying to an altitude of 5 to 15 m and floating slowly to the ground. The lowest point of the windswept area of the rotors of any of the three models installed at FCR was 17.4 m above the ground. Thus, there was little risk of collision during the breeding and nesting seasons. Migration flight takes place at higher altitude. Though the authors did not provide estimates for the altitude of migration flight, they suggested that the risk of collisions with wind turbines may be higher in May and August, when the birds migrate to and from the site. The chicks begin walking from the nest within a few hours of hatching and, while moving about on foot, will stop to sit motionless to avoid detection by predators. This behavior could lead to fatalities if the chicks cross or sit on roads when vehicles are present. Nonetheless, as noted in the discussion of mortality, they found no Mountain Plover carcasses during the entire study period.

MORTALITY

As part of the standard post-development monitoring at wind energy facilities in the U.S. and other countries, a number of mortality studies have been conducted in grassland and shrub-steppe and agricultural grassland habitats. Given that mortality studies do not easily lend themselves to controlled or experimental science, most turbine-mortality projects are designed only to document mortality through direct observation (body counts); post-hoc corrections are made for observer bias and local scavenging rates. A few of these studies also attempt to quantify “risk” by measuring bird abundance and behavior (bird use) in relation to wind turbines and correlating these data with fatality data. Fatality risk studies may be the most useful for planning, although observational data and correlation analysis provide only weak inference about relationships and do not necessarily provide insight into causal mechanisms. Moreover, high levels of variation in habitat, topography, landscape, and geographic region between wind facilities often make it difficult to generalize findings from one study to another site.

Two thorough reviews of avian mortality associated with wind power have been prepared in the past five years (Erickson et al. 2001, 2002). Although we examined monitoring reports from almost every grassland and shrub-steppe wind energy facility (see Appendix A for full bibliography), we have chosen to focus on Erickson et al. (2002) rather than repeat the cataloging effort published in Erickson et al. (2001) and used for a meta-analysis of fatality data by Erickson et al. (2002). This meta-analysis provides a more efficient and direct assessment of relevant patterns of turbine kills. Additionally, we have examined more recent monitoring reports and peer-reviewed literature that examined avian mortality at wind facilities in grassland or shrub-steppe habitats and included information presented in these reports in conjunction with our discussion of Erickson et al. (2002).

We examined two high-quality studies published in peer-reviewed journals (Osborn et al. 2000, Johnson et al. 2002) and four relevant fatality monitoring reports (Erickson et al. 2003, Johnson et al. 2003, Young et al. 2003, Koford et al. 2004, Young et al. 2006), all published after Erickson et al. (2002). We include findings from these papers only if they involve species or problems that are uniquely associated with grassland and shrub-steppe birds. Studies that specifically addressed fatality risk as it relates to behavior are reviewed in the Behavioral and Habitat Impacts section. In addition, we review an analysis by Strickland and Johnson (*in press*) that was presented to the NWCC Research Subcommittee (San Antonio, Texas, in November 1999) that combines mortality data from a suite of studies conducted in grassland or grassland-surrogate sites.

Erickson et al. (2002) presented a meta-analysis of available avian mortality and risk (use) data from 26 studies conducted at 22 U.S. wind facilities, 19 of which were located in landscapes dominated by grassland, agricultural grassland, and/or shrub-steppe habitats. The purpose of the meta-analysis was to systematically examine the growing body of monitoring data collected at wind facilities for emergent patterns of seasonal use and mortality of specific bird groups (raptors, waterfowl/waterbirds, and passerines) and bats that might aid in planning and mitigation.

Erickson et al. (2002) divided wind energy sites based on local landscape characteristics that were defined as either agricultural or native. They then standardized fatality and use/presence data for specific bird groups and compared data across facilities within each landscape category. They also analyzed seasonal use data for large birds (raptors and waterfowl/waterbirds) to determine the minimum number of seasonal replicates necessary to assess the annual risk of mortality. (We do not include a discussion of this second analysis here as it is beyond the scope of the current critical review.) Unfortunately, their meta-analyses do not include comparisons of average avian use of the two landscape types or, importantly, of mortality data in relation to use data. Additionally, only three monitoring projects apparently included reference areas that could generate data that might be able to help answer the question of whether wind facility sites are used more or less than areas in the surrounding landscape.

Based on these authors' data and results, mortality rates at U.S. wind facilities average 2.19 bird fatalities per turbine per year (range: 0 to 4.45). Outside the California facilities (Altamont Pass, Montezuma Hills, San Geronio, and Tehachapi Pass), where studies focused on more than large birds, passerine species comprise most turbine fatalities (82% of the 225 fatalities). Representation of waterbirds, waterfowl, shorebirds, diurnal raptors, owls, and fowl-like birds in the total of non-California kills ranges from less than 1% to 4% by group. The authors state in the executive summary that "[m]any passerine species are represented in the fatality lists, and data do not suggest distinct patterns indicating a particular species or group of species (e.g., flycatchers) are more susceptible to collision" (Erickson et al. 2002, p. 4). However, data that these authors summarized for the FCR (Wyoming), Ponnequin (Colorado), and Stateline (Oregon/Washington) wind facilities would suggest that Horned Larks (*Eremophila alpestris*) may be particularly vulnerable to collision with wind turbines as they accounted for 30% to 60% of recorded fatalities at these three sites (32% of 87 passerine fatalities at FCR, 30% of 10 passerine fatalities at Stateline, and 63% of eight passerine fatalities at Ponnequin). Results of mortality surveys at the Nine Canyon Wind Power Project (Washington) indicated that Horned Larks comprised 47% of fatalities at this site as well (Erickson et al. 2003). Given the aerial display behavior of Horned Larks, the authors suggested that this pattern deserved further study. However, Horned Larks were also the most abundant species observed on point counts at these sites (Erickson et al. 2002, Johnson et al. 2000). The other group of passerines that appeared to be most vulnerable to turbine fatalities comprised nocturnal migrants.

There were no clear patterns of landscape characteristics or other factors (e.g., lighting, weather) that helped to explain this vulnerability. In a multi-year study of the BRWRA, Johnson et al. (2002) found that 71% of fatalities were birds migrating through the area. Most of these were probably killed during migratory flight, although some may have been killed during stopover periods of resting and refueling. The remaining fatalities were year-round or breeding season residents. Of these, there was no indication that any single species or group of birds was particularly vulnerable to turbine-related deaths.

The meta-analysis presented in Erickson et al. (2002) suggested that standard monitoring data have great value in tracking absolute impact, but are currently insufficient for clearly identifying risk and the causal mechanisms underlying risk.

To evaluate the impact of wind energy on grassland species, Strickland and Johnson (*in press*) combined data for grassland or grassland-surrogate (cropland) sites, or a mixture of the two, for three regions. At five Pacific Northwest sites, the average fatality for all birds per megawatt of energy produced was approximately 2.7. At two Rocky Mountain sites, the average was 2.3 birds per megawatt. The average was higher in the East (3.0 birds per megawatt), and highest in the upper Midwest (3.5 birds per megawatt). Calculating the average number of fatalities per megawatt allowed for a comparison between larger and smaller wind energy sites and between different sized turbines. However, these authors noted substantial variation in the study methods. In some studies, searches were as short as two days. Other studies used 28-day search intervals; the average search duration was 14 days. At some sites, all turbines were searched; at others, only a sample of turbines was searched. They did not specify the duration of the searches or whether the studies included fall and spring searches.

POPULATION IMPACTS

We reviewed four studies that examined the effects of wind turbines on breeding bird populations and distributions in grassland and shrub-steppe habitats (Leddy et al. 1999, de Lucas et al. 2005, O'Connell and Piorkowski 2006, and Usgaard et al. 1999). The studies by Leddy and colleagues (1999) and O'Connell and Piorkowski (2006) included carefully controlled field designs and represent the best examples of research that examines the relationship between the distribution and density of breeding birds and wind turbines within grassland and shrub-steppe habitats. The results of these studies are not consistent. Leddy et al. (1999) documented a significant negative effect of proximity to wind turbines on population densities of breeding grassland and shrub-steppe birds. Usgaard et al. (1999) documented a similar effect on breeding raptors. The other two studies reported little or no effect of turbines on breeding bird density. However, the statistical power associated with the data in both studies appears to be low, indicating that the neutral findings from these studies must be cautiously interpreted.

Leddy et al. (1999) studied passerines nesting in Conservation Reserve Program (CRP) grasslands adjacent to, near, and removed from wind turbines at the BRWRA. These authors tested the hypothesis that breeding bird density on CRP lands would not differ in relation to distance from wind turbines. They selected three of the Buffalo Ridge turbine strings (model KVS-33; approx. 55 m tall), each located in separate CRP fields seven to eight years of age and three turbine-free CRP fields of the same age as the control sites. They established six survey transects paralleling each turbine string such that one transect ran directly underneath the string, transects at 40 m and 80 m ran on both sides of the string, and a single transect was located 180 m from the string. A randomly placed transect of comparable length was established in each control field. The researchers employed standard transect survey techniques, but did not adjust their density estimates for biases from species-specific detection

probabilities or observers. This paper does not present data on vegetation characteristics of turbine and control sites that would allow assessment of control/treatment site similarity.

Leddy et al. (1999) reported comparable species richness and almost complete overlap in avian community composition for turbine and control CRP fields. They also report species richness by transect distance from turbines without statistical test; these data suggest a trend toward greater richness with increasing distance from turbines. Although the relative abundance of particular species within turbine and control fields differed markedly (no statistical test reported), Bobolinks (*Dolichonyx oryzivorus*) and Savannah Sparrows (*Passerculus sandwichensis*) were among the three most common species in turbine and control sites.

The authors reported no difference between density estimates generated from surveys conducted while turbines were operating versus those conducted while turbines were offline. They reported that breeding bird density was significantly lower in CRP grasslands with turbines than in those without turbines. They also found that densities were significantly lower along the transects 0 m and 40 m from turbines compared to those 80 m from turbines and significantly lower in the 80 m transects compared to those 180 m from turbines. Bird density between transects 180 m from turbines and transects in turbine-free CRP grasslands did not differ. There was a significant and strong positive linear relationship between bird density and distance from turbines. Mean densities of breeding birds 180 m from turbines and in control fields were at least four times higher than densities within 40 m of turbines.

These authors recognized that their data did not address the mechanisms that caused the differences in breeding bird density, but suggested that the noise and human activity associated with turbines may disturb nesting birds. They further suggested that wind facilities may reduce the area of available nesting habitat for grassland birds, resulting in greater fragmentation of grasslands. Their results provide solid evidence that breeding bird densities in CRP grasslands are negatively affected by wind turbines, and this study was conducted at a spatial scale that was appropriate for assessing the impacts of wind development on local breeding bird populations.

Usgaard et al. (1999) surveyed raptor use of the BRWRA in the summers of 1994 and 1995. Though they did not report tower design or height in this paper, in related papers they noted that the turbines were 37-m tubular, model KVS-33, with rotor diameter of 33 m (Osborn et al. 2000). They compared raptor nesting density on the 293 km² BWRA, including the 32 km² wind plant (with 73 turbines) and the surrounding 261 km² intended for additional turbines in the future. Only 3% of the BWRA was considered to be raptor breeding habitat, and the availability of suitable habitat within the wind plant area was similar to the availability of habitat in the surrounding area. Biweekly surveys were conducted to determine the species composition and relative abundance of raptor species; nest searches were conducted to determine the species composition and relative abundance of breeding raptors. No nests were found in the wind plant area. Nesting density was not significantly different between years, and the nest density for both years was 5.94/100 km². The authors did not discuss the distance from the wind plant area to the nest sites.

de Lucas et al. (2005) presented results from a three-year BACI study of breeding and nonbreeding birds at a wind energy development (turbine model Ecotècnia 28/225; approximately 42 m tall) in a mixed coastal shrub-steppe and maritime woodland in northwestern Spain. Transect surveys were conducted during the early summer in the year before construction, the year of construction, and the first year of operation. The study also included a reference, or control, site several kilometers from the wind development that was monitored in all years. The researchers measured species richness, relative abundance, and density of birds. No correction for sampling bias (e.g., observers, detection probabilities) was made. The methods were not reported in great detail, making an assessment of the quality of the research design and analysis difficult. Additionally, the paper did not report the quantitative measurements of vegetation characteristics at the wind facility and reference area, and whether the reference area was an appropriate control site is unclear. No attempt was made to identify or quantify mechanisms by which avian populations or communities might be affected by wind power development.

The results of this study include a few statistically significant findings. The relative abundance of breeding birds was lower at the wind facility than at the reference site in the year of construction. However, there was no difference in relative abundance at either the wind facility or reference site across years. Density estimates for breeding birds showed no differences across years or between sites, whereas density estimates for nonbreeding birds during the breeding season differed significantly by year and site. Given that the researchers did not adjust their density estimates to control for sampling biases, this difference in density for nonbreeding birds must be interpreted with caution.

de Lucas and colleagues (2005) found little evidence to suggest that wind facilities have an impact on avian communities. However, they cautioned that bird populations are inherently variable over time and that differences between impact and control sites may have been due to natural differences between the two areas. This study did not have the statistical power or a design that would have allowed for definitive assessment of the impact of wind power development on avian communities and populations.

The most recent study to examine the effects of wind power development on grassland and shrub-steppe bird populations was conducted in Oklahoma by O'Connell and Piorkowski (2006). They examined the indirect effects of wind development on breeding birds at the Oklahoma Wind Energy Center (OWEC). The site has 34 GE 1.5-MW turbines, each 64 m in height, with blades of 55.5 m in length (Oklahoma Municipal Power Authority 2006). Specifically, these authors asked whether breeding birds avoided suitable breeding habitat if they could see or hear operating turbine rotor blades and whether habitat type influenced avoidance patterns. The OWEC contains three primary habitat types: native mixed-grass prairie, cropland (wheat), and Eastern red cedar-dominated (*Juniperus virginianus*) habitats (riparian cedar woodland and mixed-grass rangeland with cedar). To address the question of avoidance, they measured breeding bird density in all three available habitat types at three distances from operating turbines (adjacent, intermediate [1 to 5 km away], and distant [5 to 10 km away]). They used a fixed-radius point count method (200 m radius) with points arranged along road transects. All

roads were secondary and unpaved. They sampled 103 points on 26 transects distributed across nine “treatments” (three distances times three habitat types). For logistical reasons, sampling was uneven across treatments. All counts were conducted by a single observer to control for observer bias, and density estimates were corrected for detection probability by means of post hoc application of count removal models for those species encountered at least 25 times. These authors sampled all transects twice during the breeding season and reported mean abundance and density values. Vegetation at all sites was quantitatively characterized to document the variation among sites within and across habitat types. Based on the vegetation data presented, habitat type was a well-controlled variable in this study.

O’Connell and Piorkowski (2006) recorded 66 avian species during point counts and an additional 14 species encountered outside official counts. Twenty-three were common enough to allow for application of density estimate correction. This group included many of the grassland species of interest, such as Lark Sparrow (*Chondestes grammacus*), Grasshopper Sparrow (*Ammodramus savannarum*), Dickcissel (*Spiza americana*), and Horned Lark. Their report lists species by distance and habitat treatments. Although the authors have not presented species richness values or statistically compared richness across treatments, their lists indicated that distant sites and prairie sites had the greatest species richness.

The results of the count removal models suggested that detection probabilities varied by species and distance category. However, there appears to be no consistent pattern to detection across distance categories. The authors analyzed corrected breeding density estimates for each species across distance categories. Nine of 22 species for which density estimates were corrected showed significant differences in density across distance categories within habitats without any consistent pattern. Of these, a few are considered to be grassland and shrub-steppe species. Significant results for grassland and shrub-steppe species suggest that:

- Killdeer (*Charadrius vociferous*) in crop habitat are most abundant at intermediate distances from turbines.
- Western Meadowlark (*Sturnella neglecta*) in crop and prairie habitats are most abundant at sites 5 to 10 km from turbines.
- Greater Roadrunners (*Geococcyx californianus*) in mixed and prairie habitats are most abundant at sites 5 to 10 km from turbines.

Other species such as Northern Bobwhite (*Colinus virginianus*), Horned Lark, Grasshopper Sparrow, Cassin’s Sparrow (*Aimophila cassinii*), Dickcissel, Eastern Meadowlark (*Sturnella magna*), Bewick’s Wren (*Thryomanes bewickii*), Painted Bunting (*Passerina ciris*), and Scissor-tailed Flycatcher (*Tyrannus forficatus*) showed no differences in breeding density in relation to proximity to wind turbines. An analysis of all breeding birds combined also indicated no significant difference in density relative to distance from turbines.

These authors concluded that most breeding grassland birds in the area of the OWEC have experienced no negative effects that would translate into reduction of breeding density. This study’s main findings were negative (lack of statistical significance), but sample sizes were low and statistical power to detect differences across the 3 × 3 treatment design was probably insufficient. Given that this report represents one of the best efforts at controlled study of the

population-level effects of wind turbines on birds, this shortcoming in sample size and statistical power should be noted and considered in the design of future research.

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Roads

Overview of relevant literature

Roads can affect bird populations directly through collision mortality and indirectly through a suite of factors related to habitat degradation and disturbance. Roads fragment habitat and create edges, and can further influence overall landscape integrity and habitat quality by creating barriers to movement, providing access corridors for predators and brood parasites, and changing microclimatic conditions for prey species (Robel et al. 2004). Traffic associated with roads can disrupt breeding and survival activities, including courtship, nesting, feeding young, and foraging. Management practices associated with roadways such as mowing, applying herbicides, planting non-native vegetation, establishing hedgerows or windbreaks, suppressing fires, and containing runoff may also affect birds that use roadsides and adjacent habitats. Moreover, in many landscapes roadside management practices lead to the establishment of man-made grasslands that may act as local or regional population “sinks” (i.e., habitats in which levels of reproductive success are consistently low and mortality exceeds fecundity). We identified some studies of the relationship between roads and grassland birds, but only one dealt with roads and shrub-steppe birds (Robel et al. 2004). However, Holloran (2005) examined the impact of roads on Greater Sage-Grouse (*Centrocercus urophasianus*) in the context of oil and gas production.

Roads as sources of habitat edges and the consequent “edge effects” (e.g., increased predation rates) have been very well studied in forested landscapes. However, we found few studies of roads as distinct sources of edge in grassland habitats. Roadside edges were often associated with some ecotone, hedgerow, or abrupt change in landcover (e.g., grassland to forest) making it difficult to identify the role roads per se played in creating the observed effects. We identified and reviewed four studies that specifically evaluated avian response to roads as edges in grassland habitats (Pasitschniak-Arts et al. 1998, Evely and Bollinger 2001, Fletcher and Koford 2003, Bollinger and Gavin 2004). A fifth study intended to examine edge effects of paved two-lane roads on Dickcissel and Henslow’s Sparrow (*Ammodramus henslowii*) nesting success in Missouri as part of a larger study of forest edge effects (Winter et al. 2000). Unfortunately, the study had insufficient sample sizes to include road edges in analyses and reported no findings that were relevant to this topic. At the request of the GS3C, we also reviewed a thesis that compares the edge effect of a lightly traveled two-track road (Helzer 1996).

We reviewed another group of eight studies that explored the closely related issue of the quality of roadside habitats or “verges.” Most grassland-related road studies have been conducted in agriculture-dominated landscapes where roadsides may be major sources of semi-native grassland habitats. Consequently, researchers have sought to quantify bird abundance, species richness, and reproductive success as measures of roadside habitat quality. The four most relevant papers in this group report research that compares roadside habitat quality to that of other habitats within the landscape (Vierling 1999, 2000; Sutter et al. 2000) or that compares the effects of roads to other anthropogenic features (Robel et al. 2004). An additional paper examining the effect of roads as landscape features on grassland birds includes data from a busy four-lane highway outside Boston, Massachusetts (Forman and Deblinger 2000). We do not include a review of this paper because the characteristics of this road (e.g., traffic volume

range 34,000 to 50,000 vehicles per day) and surrounding area (mixed urban, forest, and grassland) were not likely to be comparable to those associated with wind facilities in grassland or shrub-steppe habitats. We briefly reviewed the major findings of three additional studies in this category that evaluated quality as measured by avian species abundance and richness, predation rates, nesting success, etc. in relation to road and roadside characteristics (e.g., size, traffic rate, verge slope) without comparison to other habitats in the landscape (Warner 1992, Camp and Best 1994, Bergin et al. 1997). Studies of this nature are valuable to solving the problem of roadside management and to understanding grassland bird ecology in landscapes where roadsides provide the only available grassland habitat. Essentially, these studies ask: Are some roads better than others, and if so, how? We found two additional papers in the edge/roadside habitat category that we do not include here, primarily because they addressed the question of how surrounding landscape characteristics affect roadside habitat quality (Meunier et al. 1999, Bergin et al. 2000). We considered this question of low relevance to the purpose of our critical literature review. However, both papers concluded that landscape context influences the variability of roadside quality and avian response to roads, which would suggest that landscape level factors should be considered and controlled in comparative studies of road impacts. We have not included any papers based on non-comparative studies documenting predation rates along roads. We considered these studies to be of low relevance, as they provide evidence that predation occurs in roadside habitats without the comparative context necessary to evaluate whether such predation varies significantly (e.g., higher or lower, differences in species vulnerabilities, etc.) from the surrounding habitats.

Studies we reviewed that assessed disturbance related to roads in grassland habitats all relied on breeding bird or foraging bird density to measure “avoidance” and quantify disturbance in relation to road characteristics, generally traffic rates. Of the five papers on road-related disturbance that we reviewed here (Percival 1993; Reijnen et al. 1996, 1997; Larsen and Madsen 2000; Forman et al. 2002), only the studies by Reijnen et al. (1996, 1997) provided any direct information about the mechanism causing avoidance of roads. Traffic volume was generally used as a surrogate measure of noise. However, although noise does increase with traffic volume, other possible sources of disturbance (e.g., visual clutter volume, frequency of visual obstruction) correspondingly increase. This covariance can obscure the true source of disturbance. From a management perspective, being able to isolate these factors to develop successful mitigation plans would be valuable. We found no studies that quantified the relationship between grassland or shrub-steppe bird behavior and roads, traffic, or other road-associated factors.

We found no large body of work about road-related mortality in grasslands or shrub-steppe habitats and thus cannot identify particularly vulnerable species or groups. A single short paper from France documented some factors related to owl mortality on roads and is briefly reviewed here (Baudvin 1997). A paper that evaluated Burrowing Owl (*Speotyto cunicularia*) survival, including the incidence of mortality from collisions with vehicles, is included in the discussion (Clayton and Schmutz 1997).

Impact summary

HABITAT AND POPULATION IMPACTS – Roads as Edges

Esely and Bollinger (2001) investigated nest placement and reproductive success of Loggerhead Shrikes (*Lanius ludovicianus*) relative to the proximity and extent of a number of habitat and landscape variables, including roadways at microhabitat, territory, and landscape scales in northwestern Missouri. The study area was predominantly agricultural with row crops, pastures, and hayfields. Roads (gravel and dirt) traverse the region along the edges of 1 mi² land sections and primarily run north-south and east-west. Esely and Bollinger (2001) measured nesting success as any nest that fledged one or more young and estimated overall nest success based on Mayfield models and recorded a number of other measures of reproductive success (e.g., mean number of eggs per nest, hatching and fledging success). After nesting was completed, they measured vegetation characteristics surrounding nest trees and randomly selected trees within 25 shrike territories. Territory scale habitat measurements were collected within 200-m radius circular plots centered around nest trees and at 25 unoccupied sites of equal area. Habitat measurements included distance from nest to nearest roadways as well as area of nine landcover types (including roadway) and other variables. Nests were classified as “roadside” or “interior.” Nests located within 15 m of a road or along a road fence were considered “roadside” nests. All others were classified as “interior.” For landscape level analysis, shrike nests and unpaired random sites were used to determine cover type percentage within 300 m, 600 m, 900 m, and 1500 m of the nest or random, unoccupied site. They then applied correct and stringent (i.e., Bonferroni corrected) statistical tests to compare occupied and unoccupied sites and reproductive success relative to various habitat characteristics, including roads. The authors identified one important design flaw: they used roads to access habitat and find nests and therefore expected a bias in distance of nests to roads. They attempted to control for this bias by selecting unoccupied sites to match distances to roads of occupied sites and by conducting paired comparisons.

Esely and Bollinger (2001) found that shrike territories contained significantly more fence line than unoccupied sites, suggesting an attraction to roads and their associated structures, possibly for perching and hunting. However, they also found that roadside nests were significantly less productive than interior nests. Daily mortality rates during the incubation stage were significantly higher for roadside nests than for interior nests, although daily mortality rates during the nestling stage did not differ. They did not include roadside cover type in their landscape level results. Roads in this landscape were evenly distributed and road density did not vary; this may have been an uninteresting variable at this scale. The authors suggested that with an abundance of suitable habitat, the landscape level may not be of concern and they recommended directing management to a finer scale. At the finer territory scale, roads were the only factor associated with decreased reproductive success in this system. However, the authors noted that even with these areas of relatively low productivity, overall shrike nest success was comparable to values reported in other studies. They did not test or document the underlying causes of lower reproductive success in roadside shrike territories but they suggested that roadside corridors may have allowed for greater predation of roadside nests during the incubation period, whereas the probability of predation might equilibrate during the nesting stage when cues from nestlings and adults could make interior nests as

obvious as roadside nests. They also suggested an alternative suitability gradient between interior and roadside territories whereby higher quality individuals occupy interior territories and lower quality birds are found in the roadside territories.

Pasitschniak-Arts et al. (1998) investigated the effects of edges created by roads, habitat ecotones, and wetlands on the nesting success of three upland-nesting duck species – Mallard, Gadwall (*A. strepera*), and Blue-winged Teal (*A. discors*) – in a mixed-grass prairie and agricultural landscape in south-central Saskatchewan, Canada. Specifically, they were interested in testing the hypothesis that predation on nests of these three species was affected by proximity to edge. They also documented the distribution of duck nests relative to three edge types and compared the observed distribution to random distributions as a measure of avoidance/attraction. The researchers used standard methods to locate nests and revisited nests every seven to 14 days to document the fate of each nest. They recorded the number of eggs, incubation stage, clutch initiation date, exposure days, and fate (e.g., hatched, depredated). They defined successful nests as those that hatched at least one egg. They used the Mayfield method to calculate daily survival probabilities. They did not collect data on predator density or use cameras or any other method to identify predators. They classified destroyed nests as depredated by a mammal if the nest bowls were ripped, eggs smashed, and shells scattered; if nest bowls were intact and eggs were split or shells contained beak marks, they classified the nests as depredated by birds. They mapped nest locations and measured linear distances from nest to nearest wetland, road, and ecotone edges. They used logistic regression with exposure days as a covariate to determine the relationship between nest success and distance to edges. Random distributions for nests were generated and plotted with species-specific constraints (e.g., random or null Mallard nests sites were not located in open grassland because – according to the authors – Mallard nests are generally not found in such habitat). Statistical tests were appropriate to data and questions.

The authors found no relationship between nest success or nest destruction (predation) and distance to any edge type for Mallard, Gadwall, or Blue-winged Teal. Proximity to roads did not explain any variation in nest fate for any species. Based on the comparison of observed and random nest distributions, Mallards and Gadwalls nested closer to road edges than expected by chance, although the observed distribution of nests for both species relative to ecotone and wetland edges did not differ from random models. Blue-winged Teal nests were significantly closer to road and wetland edges than expected by chance but did not differ from random models for ecotones. No power analysis was presented to provide greater confidence in the negative results. However, sample sizes were not so small as to suggest obviously weak statistical power. The authors concluded that in this fragmented landscape, predation pressure was evenly distributed across habitat patches 64 ha or smaller, such that all distances to edges were ecologically equivalent in this system.

Fletcher and Koford (2003) examined Bobolink breeding distribution and territory size in relation to agricultural, road, and woodland edges in grassland habitats in northern Iowa. They used fixed-width transects parallel to edges to survey breeding Bobolinks and record distance of individuals to edges and standard territory mapping methods to estimate territory locations

and size relative to edges. They applied no density estimate corrections to transect survey counts to adjust for possible differences in detection probabilities associated with the three edge types, although they did compare survey density estimates with territory density estimates to determine consistency between the two techniques. These researchers used stratified random sampling of vegetation to characterize habitat within 150 m of edges and to determine whether edges were associated with habitat gradients. They used a mixed model framework to examine edge effects on Bobolink density, territory density, and vegetation variables. A critical constraint in the design of this study concerns the definition of road edge. For consistency, the researchers included only two-lane gravel roads with row crop agriculture across the road from the grassland habitat. As the authors recognize, this does not allow road effects to be isolated, but only enables an estimate of the relative impact of road edges compared to agricultural edges. Unfortunately, this constraint decreases the applicability of the findings beyond these particular comparisons and this particular system.

Fletcher and Koford (2003) found that Bobolink density was substantially lower near forest edges than near agriculture and road edges and that edge type was a significant explanatory variable for variation in density. However, they further found that density increased with distance from all edge types and that average territory size was similar for all edge types. Territory size increased closer to roads than at greater distances, as predicted, based on the hypothesis that territory size is inversely related to habitat quality. The authors found marginal evidence that Bobolink territory positioning was consistent with active avoidance of road edges, but cautioned that even this weak evidence must be interpreted with care because it falls within the range of measurement error. Habitat structure did not consistently change relative to distance from edge and could not explain observed Bobolink density gradient. The authors concluded that forest edges are most influential in determining Bobolink distributions.

Bollinger and Gavin (2004) also investigated how Bobolinks respond to edges in grasslands. The main purpose of their study was to better understand the effects of forest edge on Bobolink breeding populations in New York. However, they also asked whether Bobolink nest success was affected (reduced) near other types of edges. They included roads as an edge type, defining roads as paved or gravel two-lane county roads with mowed fescue borders that did not support nesting Bobolinks. They used appropriate field and analytical methods to find a very strong road edge effect on the density and distribution of nests, but not on reproductive success. Specifically, they found that Bobolink nest densities were significantly lower than expected 0 to 25 m and 26 to 50 m from road edges. Road edge effects beyond 50 m were site dependent. At one site nest densities were significantly higher than expected 50 to 100 m from roads, whereas at the other site nest densities were significantly lower than expected at this distance. However, estimates of daily survival rates (DSR) did not differ between edge nests (closer than 50 m from the road) and interior nests (further than 100 m from road) and were relatively high for both classes of nests. The authors suggest traffic and possible reduction of prey base near roads caused by pesticide use and run-off as the most likely explanations of lower nest densities close to roads.

In a study of the impacts of fragmentation on wetland meadow habitat, Helzer (1996) compared the effects of three kinds of edges (wood riparian strips, cornfields, and a lightly traveled two-lane track) by measuring the abundance of Bobolinks and Grasshopper Sparrows as a function of distance from these edges. Abundance was measured along 100-m wide transects that extended 200 m from wooded edges and 100 m from cornfield and road edges. Birds seen and heard were recorded, as was the distance of each bird to the edge. These censuses took place on 25 transects each for wooded edges and roads and 24 transects for cornfields. He found no significant differences in bird abundance as a function of distance from road edges for either Bobolinks or Grasshopper Sparrows. In fact, as many Bobolinks were found within the nearest 74 m to the road as between 75 and 150 m from the road, although only one bird was found within 24 m of the road. The author surmised that because the roads were quite narrow, birds did not perceive them as barriers. He speculated that the wide gravel roads common in the area might serve as barriers, but he did not measure the impact of these roads as edges.

Helzer also tested the impact of road edges as barriers to movement. Observers deliberately flushed birds to see if they flew across the road. He found no significant difference between the number of Bobolinks or Grasshopper Sparrows that flew across the road and the number that flew away from the road.

HABITAT AND POPULATION IMPACTS – Roads as Habitat or Landscape Features

In the most relevant published study we identified, Pitman et al. (2005) (first reported in Robel et al. 2004 and thus cited in a previous draft) presented the results of a well-designed multivariate study of the impacts of anthropogenic landscape features on the distribution of Lesser Prairie-Chickens (*Tympanuchus pallidicinctus*) in the limited remaining sand sagebrush habitat in three counties in southwestern Kansas. The researchers combined field data with remotely sensed landscape data to identify prairie-chicken nest locations and adult home ranges in relation to six anthropogenic features: oil and gas wellheads, buildings (houses, gas compressor stations, and one coal-fired electric generating station), transmission lines, improved roads (gravel or paved with up to 486 vehicles per day), unimproved roads (two-lane trails and ungraded service roads with fewer than three vehicles per day), and center-pivot fields. Lesser Prairie-Chickens were captured at two sites between 1997 and 2003, outfitted with radio transmitters, and released. Radio-tagged birds were located daily by triangulation. Radio-telemetry data were used to generate spatial data on 187 nest locations and home range locations for adult prairie-chickens. The authors used a 95% fixed kernel home range method that included only one location per nest or lek site to identify adult prairie-chicken home ranges. Sand sagebrush habitat that did not fall within home ranges was classified as non-use area. To compare vegetation structure relative to prairie-chicken distribution, they measured vegetation characteristics around 174 nests and paired random sites in the sand sagebrush prairie as well as around 42 random points in prairie-chicken use and non-use areas. They then used a Monte Carlo method to compare distances of nests and adult use areas from the six anthropogenic features with random distributions. For nest sites, the researchers selected a subsample of nests based on the 10% located closest to each feature, based on the assumption that these nests would be the most conservative group for measuring avoidance. Distances to

all six anthropogenic features for this subsample were compared with distances to random points in a Monte Carlo simulation and probability distributions were used to determine whether nests were significantly farther than expected by chance from a particular feature. For adult locations, the authors applied a similar method to the centroids of monthly home ranges to test whether adults were farther than expected by chance from roads (improved and unimproved combined), buildings, wellheads, and transmission lines.

Robel et al. (2004) found that Lesser Prairie-Chicken nests were significantly farther from all anthropogenic features – except unimproved roads – than would be expected by chance. Though the number of roads and extent of each type of road are not described, the roads included improved roads (graveled or paved) that carried up to 486 vehicles per day and unimproved roads (two-land pasture trails and ungraded service roads to wellheads) with fewer than three vehicles per day. The lower range of traffic volume on improved roads was not stated. Roads and buildings had the greatest effect; the average nest distance was 859 yd and 1,371 yd, respectively. Oil/gas wellheads had the slightest effect; the average nest distance was 93 yd and center-pivot fields and transmission lines had intermediate effects at 336 yd and 397 yd, respectively. Although the authors did not attempt to identify the underlying causes of avoidance, they suggested that all these features are sources of noise and movement that may lead to nest displacement. The authors cited several general observations of noise carrying from these features across large distances in the sand sagebrush landscape. Vegetation characteristics at nest sites differed for multiple variables from randomly selected sites, whereas only litter cover differed between adult use and non-use sites. Adult prairie-chickens had home ranges farther from wellheads (79 yd), buildings (659 yd), and transmission lines (693 yd) than predicted by chance. There was no significant effect on adult distribution of roads combined, and home-range centroids were located an average of 51 yd from roads. The authors suggested that had they been able to separate improved from unimproved roads for this analysis, they might have found a more significant road effect. They used these average adult avoidance distances to calculate the amount of sand sagebrush habitat made “less suitable” due to avoidance within this system. Improved roads caused the greatest impact, affecting more than 40% of the remaining sand sagebrush (89,297 acres) in 2001. Center-pivot fields had the second-highest impact at 53,694 acres. Wellheads affected 17,562 acres, transmission lines 16,803 acres, and buildings 15,774 acres. These authors concluded that Lesser Prairie-Chicken avoidance behavior, particularly for nesting females, creates “avoidance buffers” around anthropogenic features in the landscape that further fragment, isolate, and reduce available habitat. They suggested that avoidance buffers should be measured for individual species in various landscapes to assess the true impact of human disturbances.

Two papers (Vierling 1999, 2000) reported on related studies conducted in Boulder County, Colorado, that were designed to identify habitat-specific productivity and the characteristics of demographic source and sink habitats for Red-winged Blackbirds. In both studies, Vierling compared multiple variables related to blackbird productivity across two natural habitats (wetlands and tallgrass prairie) and two anthropogenic habitats (hayfields and roadside ditches). The primary purpose of the first study was to examine the relationship between habitat quality (productivity) and blackbird density. Vierling then used this relationship to

estimate regional productivity based on the availability of the four habitats and determine relative importance of each habitat for sustaining the regional blackbird population (Vierling 1999). This author used appropriate field and analytic methods to find that roadside habitats had the lowest Red-winged Blackbird densities and lowest productivity of the four habitats. She also found that these two characteristics were consistently correlated in roadside habitats. In contrast, tallgrass prairie habitats had high productivity but low population density. Tallgrass prairie and roadside ditch habitats were equally common in the landscape, but the author estimated that 81.3 young were produced over two years in tallgrass prairies region-wide compared with an estimated 5.1 young produced in roadside ditches. From this study, she concluded that roadside ditches may act as population sinks in this landscape.

Vierling (2000) tested this hypothesis in her second study. The primary purpose of this study was to identify source and sink habitats for Red-winged Blackbirds in the Boulder area and determine underlying factors that affect source or sink status of the four habitats. She again used appropriate field and analytic methods to find that roadside ditches acted as consistent demographic sinks in which productivity did not reach a level high enough to offset estimated mortality. Source populations were restricted to natural habitats (wetlands, tallgrass prairie), although wetlands that were also occupied by Yellow-headed Blackbirds were classified as sinks in two of three years. The author found no evidence that brood-parasitism from Brown-headed Cowbirds (*Molothrus ater*) contributed to source-sink status. Rather, predation rates showed the strongest relationship with status. Roadside ditches and wetlands, also occupied by Yellow-headed Blackbirds, had the highest predation rates and were strong sinks. Additionally, sink habitat patches contained significantly more buildings within a 200-m buffer area than did source habitat patches. Vierling concluded that the suburban/rural landscape in Boulder County acted as a regional population sink for Red-winged Blackbirds because of the low occurrence of source habitat patches within the area. She suggested that urbanization of the area with increasing roads and buildings has had a negative impact on the suitability of available habitat for Red-winged Blackbirds across the region.

Sutter et al. (2000) conducted a study that was intended to quantify differences in grassland bird abundances detected by roadside versus trailside point counts in grazed grassland habitats. The authors established bird survey routes along roads (defined by a drainage ditch planted with smooth brome and adjacent fence) and trails (defined as single pair of wheel ruts visually indistinguishable from surrounding habitat) in southern Saskatchewan, Canada. They conducted five-minute, 100-m fixed radius point counts at intervals of at least 800 m along routes during a single breeding season. They used only half of each point count area (one side of circle bisected by trail or road) to avoid complications from differences in grazing intensity on different sides of a given road within a single point. Species occurring in more than 15% of samples were included in the analysis. Abundances for five of eight species differed significantly between roadside and trailside samples. Baird's Sparrow (*Ammodramus bairdii*), Chestnut-collared Longspur (*Calcarius ornatus*), and Sprague's Pipit (*Anthus spragueii*) were significantly more common along trails. Savannah Sparrow and Vesper Sparrow (*Pooecetes gramineus*) were significantly more common along roads. No difference in abundance was found for Clay-colored Sparrow (*Spizella pallida*), Horned Lark, and Western Meadowlark.

The authors suggested that these differences were not related to differences in detection probabilities, although they did not measure this parameter. They speculated as to why these differences were observed, but the study included no tests of causal mechanisms. The primary conclusion was that surveys in grassland habitats should include routes within interior habitats to ensure that all species are captured by counts.

Three studies addressed grassland bird use of roadsides as breeding habitat without reference areas in row crop-dominated landscapes where roads represent most of the remaining semi-natural grassland habitat. Camp and Best (1994), working in Iowa, and Warner (1992), working in central Illinois, conducted observational studies of grassland bird nesting success and nest density alongside gravel roads that ran through agricultural fields planted to soybeans or corn. Camp and Best (1994) described patterns of nest density and daily nest survival rates for grassland birds in relation to roadside characteristics, including slope location (foreslope, backslope, ditch bottom), fences, and vegetation height and density. Field and analytical methods were appropriate for a descriptive study. No reference areas or controls were included in the sampling design. Ten grassland species were identified as nesting within the study area. The authors found that particular species preferred different vegetation characteristics for nesting (e.g., Red-winged Blackbird nest density was highest at sites with relatively tall vegetation) and that those preferences were consistent with reported habitat preferences. Daily nest survival rates for all species combined were 0.9428, and 52% of all nests were destroyed by predation. Based on the results of this and other studies, the authors recommended that roadside habitats be managed for native grasses with seeding and prescribed burning.

Warner (1992) investigated nest density and nest survival rates in managed and unmanaged roadside verges along roads with varying traffic volumes in central Illinois. Multiple uncontrolled variables associated with roads and surrounding landscape make this study difficult to interpret. Moreover, sampling at roadsides with differing characteristics was not always concurrent, which introduces the possibility of year differences that may confound observed results. This author did not find road type or traffic volume to be consistently significant factors associated with nest density or nest survival, although nest densities were highest along the busiest road. Given the multiplicity of covarying factors involved in this study, negative results may not have the statistical power to provide strong confidence. Width of roadside appeared to be important to species richness and nest density. Surrounding landscape composition and conditions were also found to be important predictors of nest density and survival.

Finally, Bergin et al. (1997) used artificial nests to experimentally examine the relationship between roadside characteristics and nest predation in Iowa. The study was based on a solid design with a relatively large sample size ($n = 136$ road transect replicates). Roadside characteristics measured and compared in this study included roadside habitat; road type; adjacent habitat type; position of nest within roadside; and roadside width, depth, and height differences between backslope and foreslope. They reported an overall average total nest predation rate of 23%. Most transects had zero or one nest lost to predation, but three had

80% predation and one had 100% predation. Road type and adjacent habitat had no effect on predation rates. Roadside habitat type and position of nest within the roadside were the two most significant factors related to nest predation. Specifically, wooded roadsides or herbaceous roadsides with fences had higher predation rates than herbaceous roadsides without fences. The backslope nests were more vulnerable to predation than nests in other locations within roadsides. These data suggest that roadside characteristics suitable for use by predators are associated with higher predation rates.

DISTURBANCE AND MORTALITY IMPACTS

Two papers by Reijnen et al. (1996, 1997) reported on the same study of road avoidance by birds in moist agricultural grasslands in The Netherlands. Both papers specifically examined the role of noise as a causal mechanism behind road avoidance. These authors presented the original study of bird abundance and noise level in relation to roads and traffic volume. Because they simply reviewed available data on road impacts, including a minor reanalysis of their own earlier data for comparison with data from other similar studies conducted in forested landscapes, we do not review this paper in detail here.

The 1996 study used a multi-visit territory mapping technique to record breeding territories for all birds within 15 transect study areas adjacent to roads. Roads were controlled for age and all study areas were located at least 250 m from possible confounding features such as houses, power lines, hedgerows, and wooded banks. Vegetation characteristics varied little between sites because agricultural activities were consistent. Study areas ranged in size from 70 to 297 ha. Each area was divided into three to 10 sampling strips of 20 to 25 ha arranged parallel to the road to allow breeding bird density to be compared relative to distance from road. Breeding bird density was based on the total number of territories contained in each strip and total territories across strips within each study area. The authors measured noise level at a sample of locations and used a mathematical model to estimate noise level based on publicly available traffic volume data. Noise levels ranged from 59 to 38 dB(A); the highest levels were found adjacent to roads. They measured no other possible road effects such as pollution and visual stimuli. They used a model that predicts the quality of agricultural grassland for specific bird species based on detailed field management data to estimate habitat quality. The authors reported that quality varied little across strips within study areas. Twelve species with at least 10 territories in at least four study areas were included in the analysis. The statistical analysis examined noise threshold models that estimate the noise level at which territory density markedly decreases and above which density continues to decrease.

Reijnen et al. (1996) found that disturbance distance varied by species, traffic volume, and related noise, increasing from 20 to 1,700 m for 5,000 vehicles per day to 75 to 3,530 m for 50,000 vehicles per day. Disturbance distances and territory density were used to calculate the potential percentage loss of population in three zones relative to roads: 0 to 100 m, 0 to 500 m, and 0 to 1500 m from roads. With relatively low traffic volumes, the authors estimated that noise would reduce densities for seven species, resulting in local population losses of more than 10% within 100 m of roads; at high traffic volumes all 12 species had an estimated loss of more

than 40% within 100 m of roads, and more than 10% loss in the 0- to 500-m zone. Five species were estimated to have reductions as great as 14% to 44% out to 1,500 m. These estimates of population loss are based on the untested assumption that habitat is absolutely limiting such that low densities in one area cannot be compensated by high densities in other areas. They concluded that although this study provides evidence that road noise is related to decreased breeding bird territory density, the mechanisms behind road impacts on breeding birds require further study.

Percival (1993) was interested in managing refuges for maximum goose density as a means of reducing Barnacle Goose (*Branta leucopsis*) visitation to agricultural fields and other human-sphere habitats. This experimental study was intended to determine whether reductions in human disturbance would increase goose densities. One aspect of the study included measuring goose foraging density by fecal counts at varying distances from a single lane road (20 to 50 vehicles per day) that ran across the main goose foraging area. Ten transects were established at 50-m intervals. Goose dropping density did not differ significantly across distances from the road. The authors concluded that the road had little effect on the birds' grazing site selection.

Madsen (1985) investigated the effects of roads and other landscape features on the use by Pink-footed Geese of one grassland field grazed by cattle and one farm. The farm comprised a number of fields planted in different grain crops over the course of the year. Fields were described as planted areas edged or divided by "windbreaks, plantations, banks, etc., which hinder a wide view, or by roads."

Madsen assessed the effects of distance and traffic volume. Density of goose droppings defined the rate different parts of the two sites were used. At the farm site, geese avoided roads carrying 20 to 50 vehicles per day. Use peaked at 500 to 600 m from the road. With traffic volume fewer than 10 cars per day, use was still depressed to a distance of 500 m, although birds were found within 50 m of the road and use rates were approximately twice as high as those near the busier roads. Madsen concluded that the disturbance distance, defined as the distance from the road where the geese feed without interruption, is likely to be approximately 500 m. At the grassland site, the disturbance distance from roads carrying 20 to 30 vehicles per day was 240 m from the road. Closer than 240 m from that road, use declined monotonically, reaching 50% at about 100 m from the road. When traffic volume dropped to one vehicle per day, usage at the roadside was about 40% and approached 100% at a distance of 160 m from the roadside. Even a road described by Madsen as a "lane" that carried less than one vehicle per day had a disturbance distance of approximately 100 m.

Larsen and Madsen's (2000) study of the Pink-footed Goose avoidance of wind turbines and other physical elements in a Danish farmland landscape is also reviewed in the wind section. The paper derived estimates for road avoidance distances as 50 m for small roads (hard or loose surface, narrower than 3 m) and 150 m for large roads (hard surfaces wider than 3 m). They used these road avoidance distances to estimate how much land area in the landscape is lost to goose grazing because of roads. Their model suggested that large roads decrease

available foraging habitat 21% and small roads lead to a decrease of 10%. The authors suggested that new disturbances such as wind energy facilities should be located within the avoidance shadow of landscape elements such as roads to reduce habitat loss and fragmentation.

Forman et al. (2002) reported the response of grassland bird populations to roads and the built environment (defined by the author as an area “containing buildings and associated yards, pets, and human activities”) in a rapidly urbanizing landscape near Boston, Massachusetts. This paper is outlined in the Urban section. These authors found that grassland bird presence and breeding density correlated with distance from road depending on road traffic volume. At low volumes (3,000 to 8,000 vehicles per day), roads had no effect on grassland bird presence or “regular breeding,” which the authors defined as evidence of breeding in three or more years. At moderate volumes (8,000 to 15,000 vehicles per day), roads reduced breeding bird density within 400 m but had no effect on the presence of grassland birds. At heavier traffic levels (15,000 to 30,000 vehicles per day) roads affected the presence and breeding density to a distance of 700 m; at the heaviest traffic volumes (at least 30,000 vehicles per day) road effects extended to 1200 m.

Baudvin (1997) looked at the mortality of Barn Owls (*Tyto alba*) and Long-eared Owls (*Asio otus*) in relation to roadside habitat characteristics and prey availability and identified three factors that explain owl mortality along roads in northeastern France. Adjacent landcover (biotops), road elevation, and roadside habitat suitability for voles all contributed to the likelihood of owl collisions. Specifically, most Barn and Long-eared Owls were killed on roads adjacent to cereal fields. Roads that were higher than or at the same level as the surrounding terrain accounted for most owl deaths. Lastly, owl mortality was highest in areas with high densities of voles. The authors concluded that road management should be sensitive to factors that might influence the probability of owl collisions.

Clayton and Schmutz (1997) assessed the survival of Burrowing Owls in prairie habitat in Alberta and Saskatchewan, Canada. They used radio-telemetry to track individuals, and determined that most mortality occurred during the post-fledging period and near the nest, primarily from mammalian predation. The Alberta site (“Hanna”) was studied in 1995 and 1996, and the Saskatchewan site (“Regina”) in 1996. For juvenile birds, vehicle mortality was observed only at Regina, where 17% of the juveniles were killed by vehicles. The authors did not describe traffic conditions or nest locations in relation to roads, so whether this difference resulted from a greater number of vehicles or from the proximity of the nest to the road is impossible to know. As mortality from raptor predation was five times as common at Hanna in 1995, and mammal predation at that site was 2.4 times the rate that was observed at Regina, predators may have simply taken the young birds before they could stray onto roads. In 1996, raptor predation was 60% greater, and mammalian predation was 2.7 times greater than at Regina. For adults, vehicle mortality was observed only at Regina, where it accounted for 18% of the adult male mortality, but none of the adult female mortality. Without a description of the roads or of traffic volume, interpreting these results is difficult. The authors briefly described conditions at Hanna as “dominated by ranching,” whereas the Regina area was 90%

under cultivation for cereal crops and was “extensively cultivated and highly fragmented.” The authors speculated that Regina probably supported lower densities of Buteo Hawks and Canids than did Hanna, but they did not survey for these predators or cite abundance estimates. Nonetheless, the mortality attributable to raptor and mammalian predation was considerably higher at this site, although of course other mammals and raptors live in the region.

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Urbanization

Overview of relevant literature

Urbanization includes – for purposes of this review – urban expansion, suburban sprawl, exurban development, and other development associated with local increases in residential human populations. It can affect grassland and shrub-steppe birds in a number of ways. Generally, human development fragments the landscape and decreases available habitat for any bird species that depend on native habitats. In addition, habitat surrounding human development may be subjected to a number of factors that degrade its quality and suitability over time. Invasive plant species, fire suppression, increased predator density, pesticide and herbicide use, disturbance from noise and human activity, and efforts to attract birds are all considered likely candidates responsible for changing bird abundance, species composition and richness, mortality rates, and reproductive success.

Most published studies that investigate the impacts of urbanization on birds involve forested habitats and forest-dependent species. We review here eight papers that examined urbanization in relation to grassland and shrub-steppe birds, including one that dealt with the Florida Scrub Jay (*Aphelocoma coerulescens*), which technically does not belong to this group but occupies a semi-open shrubby habitat and was therefore considered relevant because of the paucity of data on this subject. Most of these studies have documented species composition, richness, and abundance in relation to density of urban elements at the landscape level and dealt with areas in or around large cities (e.g., Boston, Boulder, Tulsa) (Boren et al. 1999, Haire et al. 2000, Soderstrom et al. 2001, Forman et al. 2002, Jones and Bock 2002, Schmidt and Bock 2004) and address questions of grassland bird changes over fairly long time periods and large spatial scales. These papers reported on well-designed studies with appropriate analyses, but their results and methods are not likely to be highly relevant to assessing impacts associated with wind energy facilities that tend to represent isolated moderate intensity development and activity in essentially rural landscapes. The most applicable of these papers took a narrower scope and reported on an investigation of responses of two buteo species in relation to urbanization and habitat parameters (Schmidt and Bock 2004). These six papers and their major findings are briefly summarized under Habitat Impacts. Three studies (Morrison et al. 2004, Shawkey et al. 2004, and Patten et al. 2005) examine how urbanization affects demographic parameters; Shawkey et al. (2004) presents data on causal factors mediating the observed effect.

A recent study by Morrison et al. (2004) on Rufous-crowned Sparrow (*Aimophila ruficeps*) survivorship in relation to habitat fragmentation in southern California is a model for methodology, but does not address or specifically quantify urban landscape elements. These authors present a good example of a substantial body of research related to effects of habitat fragmentation. Although fragmentation is often caused by urbanization or roads, we have not included general grassland and shrub-steppe habitat fragmentation papers that did not explicitly and quantitatively deal with parameters that are related directly to urbanization or roads. An additional paper with a promising title and abstract is not included because it involved no direct test of the relationship between grassland and shrub-steppe birds and urbanization (Bellar and Maccarone 2002). Bishop and Myers' (2005) paper examined the

relationship between avian functional guilds and landscape elements, including urbanization, across all of Pennsylvania, but is not included because the scale, methodology, and findings are not applicable to the issue of wind energy development. We found no papers that addressed urbanization and direct mortality for grassland or shrub-steppe birds.

Impact summary

HABITAT IMPACTS

Numerous analytic approaches have been used to assess the relationship between urbanization and grassland and shrub-steppe birds at a landscape level. Boren et al. (1999) examined changes in grassland bird communities around suburban Tulsa, Oklahoma, and neighboring rural communities over a 24-yr period (1967 to 1991) in relation to changes in vegetation cover and human population density. Specifically, they addressed two hypotheses: (1) avian community composition in two rural areas differs in relation to human population density in 1966 and further diverged as the high-density rural landscape became more urbanized; and (2) vegetation cover types available in the landscape differ in areas of high versus low human population density and in turn influence avian community composition.

To test these hypotheses, Boren and colleagues compared data from two Breeding Bird Survey (BBS) routes that differed in proximity to Tulsa and human population density, which allowed one area to be designated “high-density rural landscape” and the other “low-density rural landscape.” Human population density and vegetation differed between the sites at the beginning of BBS data collection in 1966. The authors then used correspondence analysis methods to track changes in population; landscape structure and vegetation types (e.g., developed area, roads, cropland, and native grassland,); and bird community composition at four- to five-year intervals (from 1967 to 1991). Initial differences in all variables confound the power of their analysis to identify relationships and explanatory factors.

Their relevant findings were that bird community composition along the two BBS routes diverged over time; Grasshopper Sparrow and Dickcissel increased in the low-density rural landscape, Yellow-breasted Chat (*Icteria virens*) declined in both landscapes, the Greater Prairie-Chicken (*Tympanuchus cupido*) declined only in the high-density landscape, and Eastern Meadowlark and Lark Sparrow showed little change in either landscape. Temporal changes in landscape structure did not explain any variation observed in the avian communities. They concluded that alterations in landscape structure are less important than changes in native plant communities in shaping avian community composition over time, although they have no original data to directly support that claim.

Jones and Bock (2002) set out to determine whether protecting grassland habitats in a heavily and rapidly urbanizing landscape also protected grassland bird species. To assess this question, they relied on historic bird checklists that provided presence and absence data and qualitatively indicated the relative abundance of species observed in the Boulder, Colorado, area early in the first half of the 20th century. They compared these with data collected in the 1980s and 1990s. They used transect and point count methods to collect later data, but reduced to relative

abundance measures for comparability. The authors were very explicit about the limitations of this methodology, but contended that if they observed extreme changes (e.g., from abundant to rare) they would be meaningful. With these qualitative data, they documented the complete loss of seven of 29 grassland species from the Boulder area and one additional species that declined from abundant to rare over nearly a century of land use change and urbanization. They concluded that the protected grassland open space in Boulder County had successfully protected most grassland birds in the area. They noted, however, that abundance scores for shortgrass prairie birds – especially Burrowing Owl (*Athene cunicularia*), Common Nighthawk (*Chordeiles minor*), Loggerhead Shrike, and Lark Bunting (*Calamospiza melanocorys*) – declined significantly over the century, whereas mixed grassland and tallgrass/hayfield species did not change in overall abundance.

A related study by Schmidt and Bock (2004) looked more specifically at habitat associations and population trends of Rough-legged Hawk (*Buteo lagopus*) and Red-tailed Hawk (*Buteo jamaicensis*) in the Boulder, Colorado, grassland open space preserves. They employed U.S. Census data to track human population growth in the Boulder area from 1960 to 2000 and Christmas Bird Count data to establish regional and local population trends for the two hawk species from 1970 to 2003. They conducted original winter counts (1999 to 2000 and 2001 to 2002), recording the locations of hawks and habitat and landscape variables for the hawks' locations and for concomitant, randomly selected control points for the same area. Their main findings show that Rough-legged Hawks declined significantly in the Boulder area during the 33-year sample period. During that time, human population density steadily increased, but Rough-legged Hawks did not show a decline until the early 1980s. The authors interpreted this result to mean that the hawks may have a threshold of human density to which they responded. In contrast, Red-tailed Hawks increased significantly during the same time period, also showing a marked response in the Boulder area in the early 1980s. Additionally, Rough-legged Hawks were found in locations that were characterized by significantly greater area of grassland and lower developed area and roadway values compared to Red-tailed Hawk perch sites and random sites. The authors stressed two points in their conclusion: (1) the responses of both hawk populations to human population increases were nonlinear; and (2) human development in grassland habitats increased the third dimension (vertical structure) compared with the natural state. Their data suggest that Rough-legged Hawks are negatively sensitive to vertical structure, whereas Red-tailed Hawks are positively influenced by this change. This hypothesis merits further direct testing.

Haire et al. (2000) also examined relationships between abundance of grassland nesting songbirds observed in the Open Space preserves surrounding Boulder, Colorado, and landscape and habitat parameters. The purpose of their study was to assess the relative role of habitat characteristics and landscape structure as limiting factors for grassland bird abundance and distribution. Their study was designed to separate the influence of encroaching urban development and the concomitant decline in grassland habitats across the landscape. They collected original point count data on breeding birds at 66 points systematically distributed throughout the Boulder Open Space so that points would represent the range of vegetation and landscape variation. Placement of plots was based on visual but not quantitative assessment of

vegetation and landscape elements. They used a variable-distance, 10-min point count method without correcting for observer bias or detection probabilities. The researchers made 10 counts at each point over three years. They did not stringently control counts with regard to seasonal timing, and did not pool data across three years to estimate an average abundance ($n = 10$) for each plot. They used satellite imagery to establish landscape characteristics and measured plant species composition at each count plot to identify plant communities and percent cover.

Data were analyzed by using regression quantiles to model relationships between birds and landscape and habitat variables. They recorded seven species of nesting grassland birds in the Boulder Open Space: Western Meadowlark, Vesper Sparrow, Grasshopper Sparrow, Bobolink, Lark Sparrow, Horned Lark, and Savannah Sparrow. They found that urbanization limited abundance of all seven grassland species in their study area; the Horned Lark and Savannah Sparrow appeared to be the most sensitive and Western Meadowlark the least. Abundance of most species dropped severely when the sum of all urban cover types was greater than 5% of the 40-ha landscape window.

At this spatial scale, a significant negative correlation suggested that urban cover types were replacing grassland cover types. Multivariate models of species abundance with urban and grassland cover type as explanatory covariables indicate that urbanization was responsible for changes in the abundance and distribution of Western Meadowlark, Horned Lark, and Grasshopper Sparrow, even after controlling for the grassland covariate. However, area of available grassland was found to have a more universal effect on species abundance and distribution in this system. Landscape parameters were found to be more important than habitat/vegetation factors. These authors concluded that grassland bird species are strongly influenced by landscape cover type distribution and extent, but caution that causal relationships must be pursued through species-specific studies of behavior and ecology.

The remaining two studies in this topic area are the least likely to provide insights for parallel situations with wind energy development. Soderstrom et al. (2001) investigated overall biodiversity and species richness for multiple grassland taxa in relation to urbanization and habitat factors in southern Sweden. Their primary purpose was to assess the value of land management practices for communities of grassland plants, insects, and birds and compare responses across taxonomic groups. Their main finding of interest here was that species richness for all taxa, including grassland birds, was negatively associated with increasing urban elements in the landscape. Unlike Haire et al. (2000), this study was not designed to separate the influences of covarying factors (increasing urbanization with decreasing grassland).

Finally, Forman et al. (2002) examined grassland bird distribution and the occurrence of breeding birds in relation to a range of road traffic volumes in a suburbanizing landscape outside Boston, Massachusetts. They used landscape data and bird presence data collected at 84 habitat patches within the landscape to model bird data as a response variable with distance to roads, area of open habitat within patch, percentage of landscape adjacent to patch classified as “built,” and distance to nearest open patch as independent variables in logistic regression models. This paper is reviewed in greater depth in the Roads section, but the

adjacent built area variable is relevant to the topic of urbanization and is dealt with here. The models generated with data from this system found no effect of adjacent built area on the presence of grassland birds, except for the Eastern Meadowlark presence, which was negatively correlated with this landscape variable.

POPULATION IMPACTS

Patten et al. (2005) took an evolutionary approach to identifying the reasons behind significant differences in reproductive effort observed in two populations of Lesser Prairie-Chicken located on the eastern (Oklahoma) and western (New Mexico) edges of the North American shortgrass prairie. The paper reported on the results of a five-year comparative study of nesting strategies and population trends in the New Mexico and Oklahoma populations. The Oklahoma site was characterized by smaller parcels crossed by more fences, roads, and power lines than the New Mexico site, which had much larger open areas. The authors did not describe other urban structures or landscape features. Preliminary analyses of these data suggested that Oklahoma prairie-chickens were five times more likely to renest in a single breeding season than New Mexico birds. Moreover, the population trend data indicated the New Mexico population may be more stable than that in Oklahoma. Drawing on five years of nest monitoring data, these authors asked four questions: (1) Does reproductive effort differ between populations? (2) Does survivorship differ between populations in a manner consistent with life history trade-offs that might be associated with differences in reproductive effort? (3) Is there a difference in human-induced habitat alterations at the two sites that might explain observed differences in life history traits? and (4) Can observed differences in life history account for apparent differences in population persistence?

Methods for collecting prairie-chicken demographic data involved radio-telemetry that allowed the researchers to find and monitor nests, estimate fecundity and survivorship, and determine causes of mortality for salvaged carcasses in both the New Mexico and Oklahoma populations. Patten et al. (2005) used the data in a standard population model to estimate persistence. Although the authors stated that one of their goals was to identify anthropogenic factors affecting habitat and, in turn, demographic parameters and life history traits of the two populations, they did not collect data that enabled them to directly address this problem. Their analysis of landscape history and anthropogenic landscape elements (e.g., roads, fences, and power lines) in relation to life history and demography is completely post hoc and provides only the weakest explanatory power. The data clearly indicated that Oklahoma females have significantly greater reproductive effort than New Mexico females. The data also show that female survivorship – but not male survivorship – differs between the two populations. Oklahoma females had a significantly lower probability of survival than New Mexico females. Causes of death varied between the sites as well. Human-caused deaths (vehicle, power line, and fence collisions) were more frequent in the Oklahoma population. Comparing a sample of eight, 10-km² plots in each study area, the researchers also documented a greater number of fences, roads, and power lines in the area occupied by the Oklahoma population. They concluded from this and other observational data that human influence on the landscape has changed the life history characteristics and evolutionary paths of the two Lesser Prairie-Chicken

populations. Although this combination of observations is suggestive, it is by no means conclusive. It requires direct testing that employs a strong hierarchical, comparative planned design before increased human impact on the landscape is established to affect the life history traits, demography, and population persistence of the Lesser Prairie-Chicken.

Shawkey et al. (2004) hypothesized that the high partial brood loss observed for Florida Scrub-Jays nesting in suburban scrub habitat results from starvation of the last-hatched nestlings due to differences in arthropod abundance in developed and natural landscapes. They tested this hypothesis with a comparative study of two scrub-jay populations in Florida. One was on the Archbold Biological Station, which is managed for scrub-jay habitat; the other was in fragmented patches of scrub in a matrix of suburban residential development. The researchers quantified arthropod abundance and conducted nest observations for nestling feeding rates and monitored nestling mass at both sites over two breeding seasons. Additionally, in the first year of the study, the researchers also monitored brood survival in the suburban population. They found greater and more consistent partial brood loss in the suburban scrub-jay population.

They further found that total arthropod abundance was significantly greater in the natural habitat than in the suburban habitat, that nestling mass was significantly lower in suburban scrub, and that suburban hatchlings from third or fourth eggs in a brood had lower mass relative to earlier hatched brood mates. However, nestling feeding frequency was influenced by year and differed between the sites in only one year. These authors concluded that high rates of partial brood loss in suburban habitats were due to starvation, but cautioned that starvation of suburban hatchlings may not be caused by lower arthropod abundance, given that suburban jays display feeding efforts that are comparable to their counterparts in natural habitats. They suggested that a number of factors, including the generally lower number of nest helpers in developed areas and the possibility that arthropod quality may be lower in suburban sites, may help explain hatchling starvation in suburban habitats. This study is significant for its scientifically strong attempt to determine a mechanism by which human residential development might affect bird populations.

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Tall structures and power lines

Overview of relevant literature

A fairly substantial body of literature pertaining to avian collisions with telecommunications towers and other tall structures has accumulated over the past 50 years, though reports date to 1888. Some of this material pertains to collisions with window glass in buildings. Structures associated with wind farms may have windows, but these buildings are unlikely to be tall structures and are not considered in this discussion. Windows in residences and other low structures cause mortality (Klem 1989, 1990), but the scope of the request for this review is limited to tall structures. Power lines are discussed with regard to collisions, but not with regard to the associated habitat change (clearing of land for power lines and stanchions) or as perches for avian predators. The use of power line transmission towers as nest sites is also discussed.

In a bibliography of avian mortality at man-made structures, Avery et al. (1980) found that the literature on this subject extends back as far as 1888. The 1,042 articles listed in this work (which revised an earlier compilation published in 1978) include 828 journal articles, 57 unpublished reports, 38 conference proceedings, 33 government and private publications, 33 books, 32 utility company reports, 10 theses, and 11 newspaper articles. Fifteen years later, the California Energy Commission published an annotated bibliography detailing 468 papers and other reports, ranging in publication date from 1876 to 1992 (Hebert et al. 1995). These include reports pertaining to power lines, wind turbines, towers, and other structures. A 2004 update is limited to utility structures (Hunting et al. 2004). Trapp (1998) published an annotated bibliography of bird kills at towers and other human-made structures in 1998, with 125 citations of papers and reports published between 1960 and 1998; 83 of these were published subsequent to the revised Avery et al. (1980) bibliography and 102 were not included in the Hebert et al. (1995) review (Trapp 1998). Kerlinger (2000) updated the Trapp (1998) effort in 2000, covering the period 1995 to 2000. The approximately 60 additional citations included a dozen or more citations for talks presented at a 1999 workshop at Cornell University on telecommunications and avian mortality, along with citations to the earlier bibliographies and secondary or “popular” literature. The National Wind Technology Center provides a database of literature about the effects of wind energy development and towers, power lines, and other obstacles on birds; this database comprises an aggregated searchable database of documents (NWTC, undated). The number of papers and the date range covered in this database are unknown; the most recent citation dates to 1995.

Avery et al. (1980) wrote, “The majority of the reports include the number of individuals and species killed, with some observations of weather conditions at the time of the incident, bird behavior near the structure, or comments on the attraction of birds to lights. Other reports use data derived from casualties to analyze migration routes and timing, examine the causes of death and injury, describe species morphometry and physiology, determine subspecific composition of migrant populations, and assess the impact of scavengers and predators on study reliability. Avian electrocution studies, which include both electric transmission structures and electric fences, report bird casualties, examine the circumstances causing death or injury, identify problem locations and lethal structural designs, and recommend design changes to reduce the harm to birds.” He also noted that “a long-term, coordinated, national or regional

effort to monitor and report bird losses from man-made structures is long overdue...Until a thorough, comprehensive survey is made, it will be impossible to fully assess the impact of man-made structures on bird populations.” Manville (2000) said much the same: “Unfortunately, most of the research that has been done regarding bird strikes with these structures only reviews carcass counts and species variability, not the presumed or suspected causes of bird collisions. Research into this arena is sorely lacking.” Kerlinger (2000) made a similar observation, “Our knowledge regarding tower kills is rudimentary, despite more than 50 years of history documenting the problem...Basically, we know that birds collide with tall towers and that on some occasions – particularly, but not necessarily always, during inclement weather – these towers kill large numbers of birds.”

As Avery et al. (1980), Manville (2000), and Kerlinger (2000) noted, most of the papers, reports, and talks reflected purely observational work, consisting of tower monitoring efforts that resulted in counts of birds and species found dead or injured at telecommunications towers. Many failed to report the character state of the variables that are believed to cause the collisions: tower height, structure, lighting color and type, and location. Similarly, search methodology was rarely described. This lack of data makes it difficult to draw inferences that might be applicable to other, similar structures or similar conditions. Many of these papers are not peer reviewed.

Finding and reviewing the hundreds of papers that are cumulatively listed in these bibliographies would have been impossible, given the scope of work for this review, and given that many are unpublished or are published in relatively obscure publications. Few describe the habitats or ecosystems in which the observations were made, and we cannot know whether the reports pertain to grassland or to shrub-steppe species. Even if these species were listed among the dead birds found beneath the obstacle, they could have been migrating through an area that was neither grassland nor shrub-steppe. Except for an unpublished study by Joelle Gehring (Michigan Natural Resources Inventory, unpublished data), little experimental work – or even well-designed observational work – has been conducted to determine causation. Without nationwide monitoring, we cannot even determine whether some geographic areas or physiographic features are more problematic than others. In sum, even a thorough review of most or all of these papers would have been unlikely to yield useful information for those wishing to apply this information to wind energy facilities to reduce avian mortality. Given that few would have qualified under the relevance and quality filters described in Methods, we instead draw primarily on recent syntheses that discuss aspects of mortality associated with tall structures that are likely pertinent to wind turbines and wind farms. Following this general discussion, we briefly describe the few papers that, as far as we could determine, involve grassland or shrub-steppe species.

Impact summary

MORTALITY AND POPULATION IMPACTS

A report prepared by Land Protection Partners (Longcore et al. 2005) for the American Bird Conservancy and other conservation organizations for use as a response to the report prepared by Avatar Environmental for a Federal Communications Commission Notice of Inquiry

summarizes the literature on key issues pertaining to telecommunications towers. With regard to mortality estimates and population-level impacts, the report focuses on neotropical migratory bird species. However, the report also assesses the effect of tower height, classifying the results of 22 studies into two mortality classes – more than 250 birds killed per year or fewer than 250 birds killed per year. Four of each class occurred at towers shorter than 1,000 feet, but 12 of the 250-plus occurred at towers taller than 1,000 feet, whereas only three of the fewer-than-250 occurred at these taller towers. They used a regression analysis on continuous data to determine that the probability of mortality in excess of 250 birds per year increased logistically with probability of .75 for towers of 1,000 feet. At 500 feet, the probability was about 12.5 ($r^2 = 0.27$, $P < 0.01$).

The authors noted that lighting is a confounding factor, because all structures taller than 199 feet must bear aviation warning lights. They repeated the analysis, excluding the one unlighted tower, and found that the relationship between height and mortality class was still significant ($r^2 = 0.18$, $P < 0.05$), but had a relatively low correlation.

The unpublished data obtained by Gehring were also analyzed. This study was ongoing at the time the Longcore et al. (2005) report was published. Several towers 380 to 480 feet tall and taller than 1,000 feet were studied; the taller towers killed more than four times as many birds as did the lower towers. Confounding this result was the fact that several of the lower towers were unguied; these had virtually no mortality.

The extent to which these results apply to wind turbines is questionable, though, because the structures are so different. Shorter telecommunication towers are often monopoles or lattice work structures, whereas towers taller than 500 feet tend to be supported by guy wires. Some older wind turbines used lattice structures, but most commercial wind turbines today use tubular structures, partly to avoid providing potentially hazardous perch sites (NWCC 1997). So, for instance, the fact that wind turbines are not guied may be irrelevant; the span of spinning blades may or may not cause mortality in the same manner as static guy wires. Additionally, the mode of attracting (or failing to avoid) the hazard is poorly understood in both cases. Nonetheless, height is an avian mortality factor for any man-made structure. The altitude of flight under different circumstances (nocturnal migration versus diurnal foraging, for instance, or with reference to different physiographic features) may determine the probability that a bird will encounter an obstacle. For birds foraging during the day, at a low altitude, a low structure that is difficult to detect may pose as much of a threat as a 1,000-foot tower poses to nocturnal migrants.

Lower obstacles cannot be assumed to pose a lesser threat than are higher obstacles for all bird species. A leading cause of mortality for sage grouse, for instance, is fences. Though much of the peer-reviewed research on bird collisions with fences was conducted in Europe (Patten 2004, citing Baines and Summers 1997), 32.3% of 100 Lesser Prairie-Chicken carcasses recovered in Oklahoma and 13.3% of 98 carcasses recovered in New Mexico in studies by the Sutton Avian Research Center died in collisions with fences (Patten et al. 2005). To grouse and prairie-chickens, a fence is a tall structure.

Lighting is also thought to be a key causal factor in avian mortality at telecommunications towers. Gehring (Michigan Natural Features Inventory, unpublished data) reported the results of a comparison of 12 guyed towers in the 380 to 480 height range, three each with white strobes, red strobes, red incandescent blinking lights, and a mix of red strobes and steady red incandescents. In the spring of 2005, the towers with white strobes caused the deaths of three birds (mean = 1.0 per tower per 20 days) and the towers with red strobes killed 12 birds (mean = 4.0 birds per tower per 20 days). Towers with incandescent blinking lights killed eight birds (mean = 2.7 birds per tower per 20 days) and those with solid lights killed 37 birds (mean = 12.3 birds per tower per 20 days). For unguyed towers in the same height class, collisions with towers with white strobes killed three birds (mean = 1.0 birds per tower per 20 days). Four birds (mean = 1.3 birds per tower per 20 days) died after colliding with towers with red strobes, and another four (mean = 1.3 birds per tower per 20 days) died after colliding with unguyed towers with incandescent blinking lights.

In the fall of 2005, eight birds died after colliding with towers lit only by strobe lights (mean = 2.7 birds per tower per day), whereas 14 birds (mean = 4.7 birds per tower per 20 days) died after hitting towers with blinking incandescent lights. Collisions with towers with a combination of strobes and steady red lights killed 18 birds (mean = 6.0 birds per tower per 20 days). Collisions with nine unguyed towers in the same height class – three each with white strobes, red strobes, and red blinking incandescent lights – killed two (mean = 0.7 birds per tower per 20 days), one (mean = 0.3 birds per tower per 20 days), and two (mean = 0.7 birds per tower per 20 days), respectively. These results support the conclusions drawn by others from nonexperimental changes in lighting conditions, that lighting and lighting conditions play a role in avian collisions with telecommunications towers. In her unpublished thesis, J. Johnson (Swarthmore College) reported a study of a cluster of eight towers, each with steady red lights and flashing red lights. Circular flight was significantly greater at the tower site than at reference sites, and this phenomenon increased as visibility decreased.

The other significant issue covered by Longcore et al. (2005) was the effect of topography. The short discussion focused on ridgeline sites and the altitude of migration. Flight altitude, and variation in flight altitude that may be correlated with topography and/or weather, are of course key factors in determining collision risk in all locations.

Estimates of population-level impacts found in the literature usually cite Banks (1979), who evaluated the relative impacts of the various causes of annual bird mortality in the U.S. He determined that there were 1,010 television transmitting stations at the time. He estimated the average number of birds killed per tower from the three studies that he considered to be sufficiently methodical and long-term. In these three studies (Stoddard and Norris 1967, Taylor and Anderson 1973; and data from another, apparently unpublished study in North Dakota; full citations are given only in the complete search bibliography as these papers are not discussed in this review except as reported by Banks [1979], the average annual mortality ranged from 2,121 to 2,677). Banks used these studies to substantiate an estimate of an average of 2,500 bird deaths per tower per year, and calculated that annual mortality could reasonably be estimated at 1.25 million if only half the television transmitter towers then in place posed a

hazard to migrating birds; as he had no data on tower height or location, Banks made the calculation on the assumption that not all towers posed a risk.

More recent estimates of the extent of mortality simply multiply the Banks per-tower-per year estimate by the number of towers in the United States. A February 2005 analysis by the Federal Communications Commission's (FCC) Antenna Structure Registry (ASR) database (E. Paul, Ornithological Council, unpublished analysis) by height class showed that 53,069 towers were constructed, as follows:

1 to 199 feet = 31,169
200 to 299 feet = 24,845
300 to 399 feet = 17,950
400 to 499 feet = 6,855
500 to 599 feet = 1,613
600 to 699 feet = 449
700 to 799 feet = 219
800 to 899 feet = 189
900 to 999 feet = 177
1000 to 1099 feet = 324
1100 to 1199 feet = 106
1200 to 1299 feet = 65
1300 to 1399 feet = 42
1400 to 1499 feet = 41
1500 to 1599 feet = 63
1600 to 1699 feet = 27
1700 to 1799 feet = 28
1800 to 1899 feet = 11
1900 to 1999 feet = 47
2000 to 2099 feet = 18
None found taller than 2100 feet, searching up to 3,000 feet
TOTAL CONSTRUCTED TOWERS 200 FEET OR HIGHER: 53,069

The estimates that have appeared in the press and in talks given in conferences are usually based on estimates of 70,000 to 80,000 towers, without distinctions as to height class or tower type. A common misunderstanding is that every record in the ASR represents an extant tower. In fact, each record represents an application. A substantial number of applications are filed for towers that are never built, and the application record remains in the ASR database after the tower has been dismantled. The U.S. Fish and Wildlife Service estimated that 24% to 38% of tower owners failed to comply with registration requirements (USFWS 2000). This statement has been repeated by many authors, but never with substantiation. After a diligent search, we have found no evidence to support this allegation. Supposedly, a database compiled by a company called Tower Source lists towers that are not registered with the FCC, but none of those who have referred to this database have seen it or know the source of the data or the manner in which it was compiled. In September 2005, the FCC advised that, "We do know that in the past, a number of towers had failed to register, but we believe most towers required to

register are now registered.” (A. Goldschmidt, FCC Assistant Chief for National Environmental Policy Act Policy, personal communication to E. Paul).

The general belief is that towers shorter than 200 feet (which do not require aviation lights and are often monopoles or other unguied structures) pose no threat of mortality. However, mortality has been documented at guyed meteorological towers shorter than 125 feet. At the Foote Creek Rim wind facility, an average of eight bird carcasses were found per meteorological tower per year, over the three search years (1999, 2000, and 2001 to 2002); (Young et al. 2003). Using the figure of 53,059 towers above that height, and the Banks estimate of 2,500 deaths per tower per year, the number of bird deaths per year would be 132.7 million per year. However, we found no studies that assessed the relative impacts of tower height, location (including physiographic features such as mountain ridgelines and coasts), and altitude. This estimate assumes that the annual average mortality occurs at all towers, regardless of height, location, or altitude. The Communications Tower Working Group has for some years discussed the need for a nationwide study of towers of different heights and in different physiographic conditions to develop a more robust estimate of the overall annual mortality (Manville 2000). Additionally, some studies suggest that mortality increases under certain weather conditions – particularly nights with low cloud cover or limited visibility (Brewer and Ellis 1958, Avery et al. 1977, Johnson 2005). However, Avery et al. (1977) reported that 58% of losses in the spring took place on clear nights, though there was a difference among taxa. Warblers were killed mostly on overcast nights, but rails and finches were killed mostly on clear nights.

In a comparative literature review prepared for the NWCC, Erickson et al. (2005) assessed the relative mortality resulting from wind energy, tall buildings, high tension lines, and telecommunications towers. They noted a bias in seasonality of the studies (a focus on migration periods, and particularly fall migration) and that these studies are rarely random or representative (they are often conducted in response to a mortality event, or because an investigator suspects that a particular site or structure may be problematic). Extrapolating from these studies may lead to an inflated estimate. Erickson and co-workers’ meta-analysis (2001) derived an estimate from 17 studies ranging from one to 38 years. Of those two years or longer, mortality ranged from 82 birds per year to 3,199 birds in one year. The mortality estimate derived in this paper was not based on the current number of towers. Instead, Erickson cited a model developed by the Tall Timbers Research Station and Bill Evans of Cornell Laboratory of Ornithology (attributed to a pers. comm. by A. Manville, U.S. Fish and Wildlife Service, Washington, D.C. to W. Erickson, 2000) as support for an estimate by Manville (2000) that the mortality ranges from 4 million to 50 million per year, and the Manville assertion that the number could be an order of magnitude higher (attributed to a 2001 personal communication by A. Manville to W. Erickson).

BEHAVIORAL IMPACTS

Studies of the impacts of communications towers focus on mortality. To the extent that behavioral impacts have been studied, they have almost entirely focused on flight patterns near towers in response to various light colors and types. The literature about avoidance by breeding birds of communications towers, overhead structures, habitat fragmentation, or loss resulting

from the towers and associated structures (such as fences and maintenance and equipment sheds) or vegetation management around the base of the towers and under the guy wires, is scarce.

The Gehring study (J. Gehring, Michigan Natural Features Inventory, unpublished data) was the first to experimentally assess the impacts of different types of lights. Other observers have noted differences in flight behavior at towers with different kinds of lights (Cochran and Gabler 1958, Avery et al. 1976). All noted the circling behavior at towers lit by red incandescent lights, whether flashing or solid. The behavior ceased when the lights were extinguished. The LLP report reviewed a number of observational reports suggesting that solid lights are more attractive to birds than are intermittent lights, and that white lights are more attractive than are red lights. However, others note that red lights may disrupt magnetic orientation (Wiltschko et al. 1993) and in recent years, the U.S. Fish and Wildlife Service has suggested that color is less important than the type of lighting. The interim guidelines issued in 2000 state, “unless otherwise required by the FAA, only white (preferable) or red strobe lights should be used at night, and these should be the minimum number, minimum intensity, and minimum number of flashes per minute (longest duration between flashes) allowable by the FAA (USFWS 2000). The use of solid red or pulsating red warning lights at night should be avoided. Current research indicates that solid or pulsating (beacon) red lights attract night-migrating birds at a much higher rate than white strobe lights. Red strobe lights have not yet been studied.”

Yet others found that at telecommunications towers, a combination of flashing and solid red lights attracts birds. The authors stated that there is no conclusive evidence that the color of the lights affects bird attraction. Although Gehring (J. Gehring, Michigan Natural Features Inventory, unpublished data) had not yet conducted the experimental component of her study involving changing lighting systems, when the LPP report (Longcore et al. 2005) was written, the LPP authors suggested that strobes – either red or white – would prove less attractive than solid or blinking lights of either color. Kerlinger (2000) also reported that, “There have been no studies documenting the difference in risk of various lighting systems, although several researchers stated that white strobes are likely to be less risky than white or red blinking lights.” The LPP report (Longcore et al. 2005) stated that “all reports indicate that replacement of solid lights with white strobe lights (and no other lights) reduces bird kills.” The quoted text is attributed to a report by Evans Ogden (1996), which in turn, confirms that this has been the experience at lighthouses in Great Britain (Baldwin 1965), the Long Point lighthouse in Ontario (unpublished results reported by personal communication), and at a thermal generating station and two hydroelectric facilities (Broughton 1977, Chubbuck 1983). However, the Broughton (1977) report does not support the Evans Ogden (1996) statement.

The details of this report are recounted at some length to demonstrate the pitfalls of assuming that a secondary or tertiary citation is correct.

The Evans Ogden (1996) statement, which is the primary citation, is not incorrect, but the situation, as demonstrated below, is considerably less clear than the brief statement would suggest. The LLP authors (Longcore et al. 2005) probably did not see the original reports, as

they are unpublished, not available online, and they had only a very short time to compile their report (Longcore, Land Protection Partners, personal communication to E. Paul [2006]; it took Paul nearly five months to find and obtain these reports). A perusal of the document reveals that the period of time and lighting strategies covered during the Broughton (1977) report did not involve strobes. The summary of bird kills states that in 1970, when the Nanticoke power plant stacks were illuminated by floodlight, 300 birds were killed in the spring and 3,000 were killed in the fall. In 1971, a red filter was fitted to the lights on one stack and the other stack had a construction beacon. Kills fell to 16 in the spring and 803 in the fall. With the same lighting, 1,509 birds died in the fall; spring data were not available. The lights on two stacks were fitted with “mechanical flashers” and amber filters, which were used “independently on some days.” With this lighting system, 174 birds died in the spring and 2,768 birds died in the autumn. The following year, the amber filters and mechanical flashers were used at the same time. There were no reported spring kills, but fall kills soared to 9,118. The lighting system remained the same in 1975 and 1976. There were no data for spring kills in 1975; 564 birds died in the fall. In 1976, there was one recorded kill of 136 birds in the spring. The fall data were incomplete. In 1977, the stacks were floodlit and the spring total was 315. At the Lennox power plant, 5,300 birds were killed at one stack that had a construction beacon and a floodlight. For the duration of that fall and the following spring and fall, the floodlights were extinguished during migration and kills fell to 260 in the spring and 700 in the fall. In 1974, two stacks were lit only by construction beacons and the spring kill total was 92 but 1,188 in the fall. With no change in lighting, 297 birds died in the spring and 324 died in the fall of 1975, but in 1976, only 36 birds died in the spring and 327 died in the fall. In the spring of 1977, the power company tried to extinguish the floodlights when weather forecasts called for inclement weather. The spring kill total was 123 birds. In the fall, the effort was terminated midway (date not given) because the weather forecasts proved unreliable. By that time, 2,500 birds had died in collisions with the stacks. At Lennox, the stacks were lit with construction beacons and floodlights in 1972. In August and September, 5,300 birds were killed over 24 days. The mortality ceased after September 21 when the floodlights were extinguished.

The report does not state whether the construction beacons were still present in 1973, but 250 birds were killed in the spring and 700 in the fall, “as the floodlights were turned off for the spring and almost all of the fall migration periods.” The report does not state whether the mortality occurred on fall nights when the lights were on. In 1974, the stacks were lit only by construction beacons. The spring kill totaled 74 and 1,188 birds died that fall. The lights remained the same in 1975, when 297 birds were killed in spring and 324 in fall. In 1976, with the same lighting, 96 birds died in spring collisions and 327 birds died in fall collisions. As at Nanticoke, the floodlights were used in 1977, and extinguished on nights when inclement weather was expected. As the weather forecasts proved unreliable, the experiment was stopped. The fall kill exceeded 2,500 birds. The Chubbuck (1983) report is a brief recounting of the bird kill history at Lennox only. However, it goes on to say that white strobes were installed in 1982 to “substantially reduce impingement. Although no further studies were conducted to evaluate the effectiveness of strobes, security personnel did not report any further substantial impingement incidents.” However, there were many dozens of kills at both Nanticoke and Lennox before the installation that might be considered “not substantial”; without definition of

this term by the author, it is impossible to know if the kills that occurred in what was at most four seasons exceeded those low-mortality incidents before the lighting changed. Chubbuck (1983) wrote, “studies on strobe lighted stacks have also been conducted at Wesleyville and Thunder Bay and the effectiveness of this lighting verified.” We have been unable to obtain the reports of these studies.

Other tall structures

Included among the many types of tall structures at which avian mortality has been documented are smokestacks, multistory buildings, monuments, power lines, and power line stanchions. As was the case with regard to telecommunications towers, and given the constraints of this contract, we cannot evaluate this body of literature to determine which papers meet the relevance and quality criteria that were used for other sections of this review. The 1980 Avery et al. bibliography (1980) contains 1,042 citations. The annotated bibliography of the CEC (Hebert et al.1995) about avian collisions with an emphasis on power lines and electrocutions comprised 468 papers. Trapp (1998) added 83 studies published after the Avery compilation and included 102 studies that were not reported in the CEC bibliography. A significant number of the publications listed in these bibliographies comprise unpublished reports that were prepared in conjunction with site-specific applications to erect a structure, unpublished industry reports, unpublished theses or dissertations, or reports of single events. Few could be identified as involving grassland or shrub-steppe bird species. Therefore, a more general discussion with an emphasis on studies suggesting apparent causal mechanisms is provided. We also draw from the Erickson et al. (2001) paper and refer readers to that paper for mortality estimates, which are based on nonrandom observations of specific sites and are often associated with mass mortality events. Our purpose here is to look at common elements of these structures and those of wind turbines and wind farms, as these may be instructive for the development of wind energy structures and installations.

Weather seems to play a role in the incidence of collision with tall structures, in particular with regard to nocturnal migrants. Most Trumpeter Swan (*Cygnus buccinator*) collisions with power lines and fence wires occurred during winter fogs (Banko 1960). The occurrence of fog at wetlands, together with an abundance of waterfowl at these sites, seems to contribute to waterfowl mortality associated with power lines (Andersen-Harild and Block 1972). Stout and Cornwell (1976) found that 1,478 of more than 2 million (7.4%) fledged waterfowl died in collisions with power lines. Power line collisions accounted for 19% of 75 Trumpeter Swan deaths recorded from 1958 to 1973 (Weaver and St. Ores 1974).

Configuration and location of power lines also seem to affect the collision rate. In the Danish study (Andersen-Harild and Block 1972), most of the mortality occurred at one of the four study sites; this particular site had 12 wires at eight different levels. Others suggested that placing wires near trees, removing ground wires, and marking lines will reduce mortality (Beaulaurier 1981). These and other observations contributed to the development of the Avian Protection Plan by the U.S. Fish and Wildlife Service and the Edison Electric Institute (APLIC 2005).

For raptors, the problem with power lines and associated structures seems to result from the use of utility poles as perch sites. Birds are electrocuted when their wings touch the conductors and close the electrical circuit. The development of new designs intended to prevent birds from contacting the conductors by the Avian Power Line Interaction Committee (2005) seems to have reduced avian electrocutions and wire strikes. However, the negative impact of power lines may be offset by two apparent benefits. As noted in the discussion of roads, power lines and associated structures such as utility poles and transmission towers provide perches for hunting. Some also provide nesting opportunities. The overall impact on raptor populations is unclear. Four papers demonstrate that raptors use transmission towers as nest platforms, but the observed rate of nest success varies substantially.

Gilmer and Wiehe (1977) began to study the use of these towers by raptors in North Dakota after learning that raptors had been observed nesting on towers maintained by the U.S. Bureau of Reclamation. They compiled data from their own surveys of five power lines in mid-April and early and late June in 1976, data from three other power lines that were checked less frequently, and data from the Bureau of Reclamation. In all, 890 miles of lines were surveyed by air and by ground inspection. Although Red-tailed Hawks and Great Horned Owls (*Bubo virginianus*) were observed, 21 of the 29 nests were occupied by Ferruginous Hawks (*Buteo regalis*), and all were east of the Missouri River. The authors noted that the land west of the Missouri was marginal habitat for raptors, as it was intensively farmed. Nest density ranged from 2.0 per mile to 7.0 per mile. Nest productivity of the tower nests was compared to the productivity of nests found in trees, haystacks, and on the ground. From the four tree nests, all young fledged. Haystack nests ($n = 15$) fledged 87% of the young and 60% of the young fledged from the five ground nests. Tower nests were most numerous ($n = 21$) but fledged only 57% of the young, in part because three of these nests were blown from the towers by high winds.

The authors could not determine whether the towers would result in an increase in the overall population. They also could not determine whether raptors preferred towers as nest sites. Though they noted that the absence of suitable hunting territory apparently limited the use of towers in some areas, they did not assess how limitations of the prey base and collisions would serve as limiting factors, even if the towers provided additional, rather than alternative, nesting sites.

Gilmer and Stewart (1983) studied Ferruginous Hawks in North Dakota from 1977 to 1979. They observed a 51% increase in occupied nests in 1979. During the study period, tree nests were most common; 63.6% of the occupied nests were found in trees and 18% ($n = 32$ of 113) were found on power line towers. Success for tower nests for all years combined (86.7%) was higher than that for tree, ground, or haystack nests, but the authors posit that this result was due to the destruction by a storm of ground and haystack nests; the storm happened not to pass by the area where most tower nests were found. The authors suggested that the population increase might have resulted from increased survival and recruitment, but they also suspected that some birds immigrated into their study area. However, they also noted that although towers have been available since the mid-1950s, their use did not become common until 1976, perhaps because of a design change that provided a secure platform. They noted that these

towers allowed pairs to use habitats that previously lacked suitable nest sites. These authors also found a strong relationship between Ferruginous Hawk populations and grassland habitat, as few pairs nest where more than 50% of the land was under cultivation.

Gaines (1985) studied the nest site selection, habitat use, and breeding ecology of the Ferruginous Hawk in North Dakota. He found that the number of nests found in trees and on haystacks had not changed since the 1977-1979 period (no citation given by the author in text; a footnote to Table 6 suggests the data were reported in Gilmer and Stewart 1983). The number (16 of 79) and percentage (20%) of nests on towers had increased, however, compared to 1977-1979, when only six of 78 (7.7%) of nests were found on towers. The nests on towers had a success rate of 80%, compared to tree nests (58.5%), ground nests (57.1%), and haystack nests (50%). Gaines (1985) noted that the unattributed 1977-1979 data show a similar result in that tower nests fledged 86.7% of young, though tree nests were the least successful (65.3%). The overall success rate of 1.6 young per nest was lower than that observed in previous studies (Smith and Murphy 1973, Lokemoen and Duebbert 1976, Blair 1978, Gilmer and Stewart 1983; the first three of these are not discussed in this review, the citations are found only in the Complete Search Bibliography). However, the success rate for tower nests exceeded the success rate for tree, ground, and haystack nests. Like Gilmer and Wiehe (1977), Gaines (1985) could not detect a population impact. In fact, at the time of this study, researchers were unsure whether the population was recovering or depressed range-wide with a few stable local populations. Gaines (1985) attempted to assess habitat preference, but found that the land use within the study area made this analysis difficult. He used a binomial test for each of four habitat types – cropland, pasture, hayland, and other – to determine that there were no significant differences in nest location. He suggested that the increased rate of towers as a nesting substrate resulted from philopatry and that because these nests had a higher fledging rate, a larger proportion of young birds would likely nest on towers.

Steenhof et al. (1993) were able to attribute a population increase to the use of tower nests by four raptor species and by Common Ravens (*Corvus corax*) in southern Idaho and Oregon. They determined that the towers enhanced raptor nesting density and productivity. During construction of the 500-kV line and towers in 1980, artificial nesting platforms were installed on 37 of the 1,608 towers, chosen nonrandomly (total number of towers not stated; method of selecting towers for platforms not stated). By 1986, all nesting platforms had sticks inserted in the platforms, and on some of those platforms, the sticks were wired in place.

The two-year survey of use of the nesting platforms began in 1981 and continued through 1989. Aerial surveys of all towers along the line started in 1983. Survey frequency was twice as great in Idaho (four checks per nesting season) than in Oregon. A third area, lacking power lines and towers, was used as reference site; this area was also surveyed and data on nesting populations dating to the early 1970s were also used. This area was surveyed from the air and by ground during the course of other, unrelated research projects. Raptors and Common Ravens began using the towers within a year after construction. By 1989, the number of pairs using towers increased to 133; ravens were the most common species in each year. Each year,

nesting density for all species increased. Golden Eagles (*Aquila chrysaetos*) nested on the towers in all years; growing from a single pair to eight pairs during the study.

The most common raptor species was the Red-tailed Hawk, which first appeared in 1982 and increased to 33 pairs. A single pair of Ferruginous Hawks nested in 1981; by the end of the study, 11 pairs were found nesting on the towers. Similarly, a single pair of Common Ravens nested in 1981; 81 pairs were found in 1989, the last year of the study. Single pairs of Great Horned Owls were found in two of the last three years of the study. Steenhof et al. (1993) found that the towers provided new and alternative nesting substrate, as a pair shifted from natural substrate to towers, and new pairs that had not previously nested in the study area appeared and nested on the towers. Colonization began sooner and increased more rapidly on the Idaho stretch of the power line than on the westerly section, which was primarily in Oregon. In total, 274 of 1,608 available towers were used. Nest success ranged from 76% to 83% of breeding pairs over the course of the study. Success rates were higher for the towers than for natural substrates, but were not higher for towers with platforms than for those without or for other manmade substrates, except for Golden Eagles, which did not nest successfully on towers without platforms.

Steenhof et al. (1993) concluded that the lack of a nesting substrate had been a limiting factor that was removed when the towers were built. They noted that nesting densities of these species elsewhere in the area were as high as or higher than before the power line was erected. The towers in this region afforded a nesting substrate that was safe from range fires and mammalian predators. In addition, the tower nests, which were exposed to the wind, may have been cooler than canyon nests, and may even have been more shaded. Occupants of tower nests faced two threats – destruction of nests by high winds and individual birds becoming entangled in the tower stanchions (observed in three juvenile ravens).

These authors documented no ill effects of electromagnetic fields, but suggested that further study of this issue was needed. They did not discuss the issue of avian mortality from electrocutions or collisions with this particular power line. As most of the surveys were conducted by helicopter, and there were, at most, four surveys per year, such mortality would probably not have been observed during this study. Nonetheless, the population apparently increased despite any such mortality that occurred during the study period.

Lights are associated with mortality at tall buildings, smoke stacks, and monuments (Evans Ogden 1996). Some of these events are substantial. As Erickson et al. (2001) reported, as many as 4,700 dead birds were found in a three-day period at a Florida power plant (citing a study by Maehr et al. 1983). The Gehring study (J. Gehring, Michigan Natural Features Inventory, unpublished data) suggests that changing the color and type of lighting may reduce avian mortality. The Fatal Light Awareness Program has demonstrated that extinguishing building lights at night reduces mortality (Evans Ogden 2002). However, lighting at wind farms usually consists of aviation warning lights that cannot be extinguished. Thus, further research on the effect of light color and type is warranted.

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Oil and Gas Extraction

Overview of relevant literature

Oil and gas production facilities eliminate, fragment, or otherwise modify bird habitat in various ways: well drilling, facilities and road construction, pipelines, and electricity transmission lines can destroy habitat. Indirect effects include traffic, noise, disturbance of natural vegetation, increase of non-native invasive species, increased predation resulting from reduced shelter, increased perching sites for predators, and increased edge.

Most studies that examine the effects of oil and gas extraction focus on shrub-steppe species. The 41 species accounts comprising the U.S. Geological Survey's "Effects of Management Practices on Grassland Bird" series and the associated bibliographic database (Johnson et al. 2004), were examined. A number of papers were found; those that met the selection criteria described in the Methods section are included in this literature review. We also searched the Birds of North America species accounts at <http://bna.birds.cornell.edu/BNA/>, which include analyses of impacts of human activity and the Forest Service technical conservation assessments for grassland bird species. We found relatively few papers that assess the impact of oil and gas extraction on grassland species, compared to the number pertaining to shrub-steppe species, perhaps because of research interest in the impact of habitat loss to agriculture, which occurs on a massive scale, and urbanization. Together, these two activities dwarf the scale of the impact of oil and gas extraction in grasslands. There may also be a temporal explanation. Oil extraction from grassland areas began in the early part of the 20th century and peaked in the mid-1980s. In Texas, for instance, the number of wells has declined from 210,000 in 1985 to 151,000 today (Texas Railroad Commission 2005). Studies done during that time frame assessed habitat loss and fragmentation generally, but few focused on the relative contribution of specific causes of habitat loss and fragmentation. The relatively intense focus on the relationship between oil and gas activities and sage and shrub-steppe habitats seems to be a function of recent large-scale changes in these habitats.

Squires et al. (1993) found that radio-tracked Prairie Falcons (*F. mexicanus*) preferred grassland habitat and did not actively avoid oil wells. The intensity of the oil development had no apparent impact on habitat use. The birds foraged between the wells and perched on the power lines. However, the eyries were located on buttes that were distant from the wells. Van Horn (1993) found a similar pattern in Montana, but Prairie Falcons did react temporarily to blasting noise by flushing or sitting up in the nest (Holthuijzen et al. 1990). The birds ceased incubating and breeding in response to blasting, but production was not affected.

Some Forest Service technical conservation assessments mention the potential impacts of oil and gas development on grassland birds. The assessment for Burrowing Owl states that, "Practices such as extensive oil and gas development provide a new and unstudied potential for change whose effects are difficult to predict" (McDonald et al. 2004). The assessments for Grasshopper Sparrow and Cassin's Sparrow make statements to the effect that, "Additionally, the impacts of new construction for gas and oil exploration, wind-power development, and water-well drilling need to be investigated." (Slater 2004, Lynn 2006). Several papers cite Knick et al. (2003) for the supposition that increased predation will result from increases in the extent

of power lines available for use by corvids and raptors. However, the major papers, and those that met our selection criteria, focus on shrub-steppe species. Some studies described in the review of the major literature attempted to isolate the specific effects of different aspects of oil and gas production; others merely described the overall impact on bird populations. One major study investigated the impacts on specific components of reproduction and survival. Braun et al. (2002) concluded that, "The effects of oil and gas developments on sage-grouse and other sagebrush grassland avifauna are poorly understood because of the lack of replicated, well designed studies."

Two major studies have been conducted since then. The Holloran (2005) dissertation is discussed at length. The Naugle study, conducted for the Bureau of Land Management, is still in progress (D. Naugle, College of Forestry and Communication, University of Montana, personal communication 2006). Now in the third year of four years of data collection, Naugle found that sage-grouse have declined by an estimated 84% in areas of Montana and Wyoming, where coal bed methane gas is intensively extracted. Generally, though, the state of the research is such that impacts that are attributable specifically to some unique aspect of oil or gas activities as opposed to more general components, such as roads and power lines, are not understood. Holloran (2005) concluded that, "the specific components of a developing field that result in declines, as well as the root causes of lek abandonment, remain unidentified." This author suggested that displacement resulted from disturbance, but none of the studies identified in the search or discussed in this review empirically identified unique mechanisms of causation (such as noise, human presence, or increased soil salinity associated with coal bed methane extraction). Distance to wells and other structures associated with extraction, including roads and the response to increasing density of wells, have been studied (Aldridge 2005, Pitman et al. 2005). Most studies focus on sage-grouse, but passerine response has also been studied (Ingelfinger and Anderson 2004).

One unique aspect of coal bed methane extraction has recently been studied. The water pumped from coal bed seams is held in retaining ponds, where *Culex tarsalis* mosquitoes are thought to breed. This mosquito is the predominant carrier of West Nile Virus in Montana. Montana State University graduate student Melissa Doherty will present the findings in her thesis, which she is now writing (G. Johnson, College of Agriculture, Montana State University, personal communication 2006). Aldridge and Boyce (2004) have proposed a study to sample mosquitoes in relation to sage-grouse breeding habitat, and would include as a variable the distance to the nearest oil well. Based on a review of Aldridge's current research, as described on his undated website (<http://www.nrel.colostate.edu/%7Ealdridge/research.htm#OngoingResearch>), this research appears to be underway.

Seismic lines are often used in oil and gas exploration. A strip of vegetation 6 to 8 m wide is cleared of vegetation and topsoil. Truck-mounted equipment drills holes along the line for dynamite charges. The reflected sound waves are recorded and analyzed. We do not know how commonly this technique is used in North American grassland and shrub-steppe habitats today, as much of the exploration for current and planned extraction has already taken place. The

issue is mentioned here because one study examined the impact of seismic lines on biodiversity in the Patagonian steppe. The only bird included in the study was the Lesser Rhea (*Pterocnemia pennata*). Vegetation cover was lower in the seismic lines, which were also more susceptible to colonization by non-native species. However, impacts on the rhea specifically from the seismic lines could not be determined, because there were so many drilling pads and roads in the same places as the seismic lines (Fiori and Zalba 2003).

Water discharged from underground oil and gas stores may be contaminated with petroleum and various heavy metals. A study measured the concentration of these contaminants in Western Sandpipers (*Calidris mauri*) feeding on invertebrates near marshes in which contaminated water had been discharged (Rattner 1995). Though this study did not examine impacts on grassland or shrub-steppe bird species, the impacts of contaminated wastewater should be of concern. Concentrations of contaminants such as aromatic petroleum hydrocarbons in the stomach contents of birds at the discharge site were significantly higher than the concentrations found in the stomach contents of birds at control sites. Liver weights were significantly lower in the exposed birds, to an extent consistent with low-level exposure, but no other physiological changes were observed.

Habitat changes resulting from oil and gas development could benefit some species. For instance, the clearing of vegetation may provide suitable habitat for Mountain Plovers, which prefer short grass or even bare ground (Day 1994). This supposition was supported by observations (Parrish et al. 1993) of Mountain Plover habitat selection in the Powder River Basin.

Review of the major literature

In a review paper, Braun et al. (2002) summarized studies in Alberta, Colorado, and Wyoming, noting that the impact of these facilities on shrub-steppe species has been observed for three decades since production in sagebrush habitats began to increase dramatically. Over the decades covered by the review (approximately 1970 to present) various kinds of energy resources and extraction methods have been used (oil shale, oil, gas, coal, or coal-bed methane). During this same period, there have been drastic declines in sage-grouse populations.

The Alberta study included a 2000 thesis by Cameron Aldridge (Aldridge 2000) that examined reproduction and habitat use by sage-grouse at the northern edge of the species' range. Using radio-telemetry, Aldridge assessed adult survival, nest attempts, nesting success, and breeding success (clutch size, fledging success, and chick survival). He also assessed the vegetation characteristics of nest sites and brood rearing sites.

Aldridge (2000) wrote very little about the impact of oil and gas extraction. The one-page discussion simply reported that, "The removal of vegetation for well sites, access roads, pipelines, and associated facilities can reduce and fragment suitable habitat. Human and mechanical activities at well sites may disrupt breeding and nesting activities." Aldridge also reported one personal observation that sage-grouse did not return to a reclaimed site that formerly hosted a lek. He also reported that six strutting grounds in Alberta had been disturbed

by oil and gas activities, though four of the six were no longer active. Power lines and roads were the only specific mechanisms discussed; both provided increased access for predators and opportunities for collisions with power lines. However, there is no independent observation or even description of oil and gas activities in the areas of the leks or nests that were included in the study.

Braun et al. (2002) discussed the status of sage-grouse in Alberta in some detail, including correlations between population declines and periods of intensive oil and gas activities in the region. In the oil boom of the 1980s, for instance, the number of males displaying at lek sites fell from 524 to between 200 and 300. However, these data are from Aldridge (2000), who did not study the impacts of oil, gas, or any other human activity in the study area; the paper did not even report on the proximity of oil and gas activity. Apparently, a similar decline occurred in the 1990s, when an oil field was developed “in the heart of sage-grouse habitat,” again citing population data reported by Aldridge (2000). Braun (2002) also alluded to disturbances of lek sites that occurred within 200 m of wells or associated roads but these observations from the 1980s were not reported in Aldridge, and no other citation is given.

Braun (2002) noted that lek counts in Alberta began in 1968 and were conducted sporadically until the 1990s, making it difficult to ascribe with certainty the drastic sage-grouse population decline to the oil and gas development. However, he noted that 1,500 wells, with associated roads, trails, power lines, pipelines, compressor stations, and gas camps, have been drilled in the Alberta range of sage-grouse; the density of the 575 wells producing at the time of the review paper was approximately eight well sites/mi² of sage-grouse habitat. There is no discussion of other human activity that affected this region over this same time period.

The Colorado studies considered in the review paper are similar to the Alberta studies. As the review paper (Braun et al. 2002) notes, “No replicated, designed cause and effect studies have explored the impacts of oil and gas production on sage-grouse populations although Braun (1987, 1998; citations given only in Complete Search Bibliography, as these papers are not discussed in this review) generally discussed the apparent short-term impacts.” Sage-grouse populations in areas where oil shale or coal bed methane was likely to be extracted had already disappeared by 2000. Further, because oil and gas activity began two decades before sage-grouse counts were conducted in Colorado, which were sporadic and incomplete for the following three decades, no definitive conclusions can be drawn about the impact of oil and gas activity on sage-grouse in Colorado.

Apparently unpublished observational data (no citation was given) for a specific field where oil extraction began in the 1920s were discussed in the review paper. From 1973 to 2001, 11 leks were active within or immediately adjacent to the field; eight of these were on the periphery and three were in the main field. Only two were within sight of an active well or power line. The authors suggested that sage-grouse will locate in and near oil production facilities as suitable habitat is available and if there are sites where they can avoid physical structures and paved roads. However, few conclusions can be drawn without pre-development data.

Discussion of the impacts in Wyoming focuses on coal bed methane extraction, which has similar infrastructure but also requires that water be pumped and stored. Apparently unpublished data (as of the date of the review) have been collected over the four years since coal bed methane extraction intensified showed a “significantly lower growth rate” of sage-grouse on leks within 0.25 miles of an overhead power line; though these lines were not necessarily associated with methane extraction sites, there were 200 extraction sites (and associated power lines) within 0.25 miles of 30 known lek sites. No statistical analysis was provided, so it is not known if the term *significant* was used in the statistical sense. This observation is attributed to increased raptor predation. In addition, sage-grouse numbers were lower at leks within one mile of compressor stations than at more distant leks. Habitat loss, roads and traffic, and noise were assumed to be the causal mechanisms.

Aldridge (2005) returned to the issue of the impact of oil extraction on sage-grouse in his dissertation. The purpose of the research was to develop models to identify habitats where sage-grouse are abundant and have high fitness. The measures of abundance and fitness and landscape correlates were measured in a 1,100-km² core sage-grouse use area within a 4,000-km² area where they occur. The study area contained 16 of the 32 historically known lek sites, eight of which were active during the study. The area also experienced substantial gas and oil development.

Among the variables included in the sage-grouse habitat model were anthropogenic activities, including gas and oil activities. To develop this model, Aldridge (2005) tracked female grouse fitted with radio transmitters to find and monitor nests. The chicks were also fitted with transmitters. Aldridge used GIS to identify and locate habitat characteristics and human activities, including roads, trails, oil well sites, urban structures, and cropland. The analysis, which considered distance and density of these features, also included an assessment of roads and well sites that were visible from 250, 500, and 1,000 m. The author also used this analysis to test the findings by Schroeder et al. (1999) that most females will nest within 3.2 km of lek sites, and that a buffer of this size around leks will protect critical nest and brood habitat for sage-grouse in Alberta. The same analysis assessed guidelines issued by the Province of Alberta calling for protection of a 1-km buffer around lek sites, and limiting oil and gas activities within that buffer.

Holloran’s dissertation (2005) is the most recent and most extensive study to focus specifically on the impacts of oil and gas production on Greater Sage-Grouse in the upper Green River Basin of western Wyoming. The dissertation encompasses work previously published in Transactions of the 70th North American Wildlife Management and Natural Resource Conference and work that is to be published in the Condor (the journal of the Cooper Ornithological Society). Therefore, those papers are not reviewed here. Holloran investigated population change as a function of the level of oil and gas development. The intent of the study was to assess the impact of drilling rigs, producing wells, and main haul roads on several population and behavioral parameters:

- o Regional abundance

- o Change in number of displaying males over time
- o Lek tenacity, survival, and breeding season habitat selection among males relative to cumulative levels of gas development
- o Nesting and brood habitat selection by females
- o Female sage-grouse population growth.

The study investigated the influence of distance and density of the rigs, wells, and roads, as well as visibility and direction, and traffic levels on main haul roads.

The study area of 421 square miles encompassed numerous lek sites protected by 3.1 mile buffers. Impacts were characterized as heavy (more than 15 wells within 3.1 miles of the lek) or light (five to 15 wells within 3.1 miles of the lek). Sites with fewer than five wells within 3.1 miles of the lek were considered controls. Holloran conducted lek counts, traffic counts, and quantified the number and distance of the rigs, wells, and roads. Some grouse were radio tracked to determine habitat selection and to locate and monitor nests.

Pitman et al. (2005) examined nest site selection by Lesser Prairie-Chickens in relation to several anthropogenic activities, including well heads. On each of two study areas, the nests were farther from wellheads that would be predicted by a random distribution across otherwise suitable habitat.

Passerine response to natural gas extraction has also been studied (Ingelfinger and Anderson 2004). This study focused on roads associated with the Pinedale Anticline Project Area and the Jonah Field II projects in Wyoming, which is located in sagebrush steppe habitat. The study measured breeding bird distribution along one state highway and three dirt roads and one pipeline right-of-way. The term *breeding bird* does not imply that the birds were determined to be breeding in the areas surveyed. The investigators made no mention of the detailed observations of behaviors that confirm breeding. Rather, the term apparently signifies that birds present during the survey period (15 May to end of June) were considered to be species that breed in the region. The surveys measured bird density in 50-m fixed-radius counts. Densities were sampled out to 200 m from the dirt roads and out to 350 m from the paved road to concentrate sampling effort where the road effect was greatest. Nothing in the study design suggests that density estimates were corrected for observer bias. However, because the traffic volume varied substantially (444 to 697 vehicles per day on one road, depending on the time of day and day of week, versus 12 vehicles per day on the other), a linear regression was used to adjust for detectability. In addition, the analyses were limited to point counts located 50 m from the road and counts from roads with extremely low vehicle counts were not included in the analysis.

Aldridge and Boyce (*in press*) report further on their development of models to predict the relative probability of use and habitat-associated risk for Greater Sage-Grouse. The two were combined to identify source and sink habitat. The latter is characterized by attractive landscape features that pose increased risk of nest or brood failure. The features associated with failure are anthropogenic factors such as roads, cropland, and high density of gas and oil wells. The methods and data are discussed at much greater length in the Aldridge (2005) dissertation,

which is discussed here at length. The models linking habitat features to fitness were validated with the data collected for the Aldridge dissertation. The final model had six parameters: brightness value (a GIS measure of reflectivity that is used to assess land cover type and extent); mean % sagebrush cover within a 1-km² moving window and the square of this value; patchy sagebrush distribution; proportion of habitat within a 1-km² moving window that is patchy sagebrush; proportion of habitat within a 1-km² moving window that is badlands type habitats with juniper and needle-and-thread-blue grama; and proportion of habitat within a 1-km² moving window that is edge habitat.

This combined model can assess the habitat that is available for successful nesting and brood rearing. The results distinguish between habitat that has suitable vegetation and microhabitat characteristics (such as wetness) and habitat that will likely lead to nest or brood failure. The model did not expressly incorporate the proportion of anthropogenic activities, but the “edge” parameter combined roads, well sites, urban habitats, and cropland. While edge had no effect on nest success, the authors reported that females so strongly avoid edge that an edge effect could not be detected. Given the number of habitat features tested and the express linkage to various components of fitness and recruitment, this model would be useful at other sites to assess the availability of suitable habitat, net of what the authors call ecological trap habitats – those that are likely to be attractive to sage-grouse but where nests are likely to fail and/or chicks are likely to die – and from there to determine the probability of persistence of a local population should suitable vegetation decrease or risk factors increase.

When applied to the field data collected from the study area, 72% of the nests used to train the model and 65% of the nests used to validate it were found in areas classified as having good-to-high likelihood of sage-grouse nest occurrence. The brood occurrence model was equally successful. Only 20% of the selected brood habitat fell within the area predicted to be good-to-high quality habitat, but 77% of the training points and 71% of the validation points fell within these areas. The chick survival model was less successful, which was not surprising given that no variable was significant at the $\alpha = 0.05$ level. The predictive accuracy was low, but the classification accuracy was 70.7%. Unfortunately, the model could not be validated, as there was no independent sample against which to test it. The greater value of the models is not the prediction of source and sink habitat, but the identification of the underlying components, or mechanisms of the population responses, as the models are built on the risks associated with habitat features.

Impact summary

MORTALITY

Aldridge (2005) determined that sage-grouse chick failure increased in habitats with a greater number of visible gas and oil wells within 1 km. Aldridge modeled the relationship between chick survival and proximity to wells, predicting a 1.5 times increase in risk of chick failure for each additional well that is visible within 1 km of the brood location. He attributed the increased risk to roads and to predating raptors perched on power lines associated with wells, but did not directly test this hypothesis.

According to Holloran (2005), male survival probability declined significantly on heavily affected leks, which he defined as those affected by more than two gas-field related factors. He also concluded that the reduction in the female population growth that was associated with various attributes of gas extraction that resulted from reduced annual survival of adult and yearling females.

HABITAT AND BEHAVIORAL IMPACTS

Holloran (2005) found that male sage-grouse attendance at lek sites in relation to distance and density of rigs, wells, and roads, measured by the maximum number of males visiting a site in a given year, declined by 51% at heavily affected leks from 2003 to 2004. In contrast, attendance declined by only 3% at the control sites. Attendance at the three most heavily affected sites declined by 89%, and two of those sites were inactive in 2004. One male appeared at one site on one morning.

This author also determined that the distance at which drilling rigs appeared to have no influence on overall male lek attendance was greater than 6.2 km, whereas for producing wells, the distance was greater than 4.7 km and for main haul roads, the distance was 6.1 km. Leks within 1 km of drilling rigs had a significantly higher rate of decline than did control leks, but the rate of decline on leks 2.1 to 3.0 km away did not differ significantly from controls. Holloran did not detect differences in the rates of annual decline associated with the visual impacts of drilling rigs. However, compared to the controls, leks in all three sight categories – full, partial, and none – declined significantly. Compared to controls, leks southeast and northeast of the rigs had significantly greater rates of decline, and leks southwest and northwest of the rigs did not decline significantly from the controls.

For producing wells, the number of males on leks within 3 km of a well declined compared to controls. Well density also had an impact. When density exceeded four wells within 3 km, annual declines were significantly greater than declines on sites with fewer than four wells. If three or four quadrats (quarter circle wedges delineated by the four cardinal directions, centered on the lek and measured in distances of 1-km buffers) within 5 km of a lek site contained wells, male lek attendance declined significantly compared to controls.

Lek sites within 3 km of main haul roads experienced significant declines in numbers of males attending the leks, compared to lek sites that were farther than 6.1 km from such roads. The extent of the roadway (more than 5 km of road within 3 km of the lek) correlated with significant decline, but visibility and direction had no impact. Traffic volume resulted in significant declines of male attendance even at the lowest rate (one to 20 axle hits per day). Generally, the probabilities of adult male desertion (males captured from a particular lek and not documented on that lek after capture) were higher from leks affected by at least one gas-field related factor compared to unaffected leks.

NESTING AND BROODING

Holloran (2005) found that nesting density was affected by well densities. Females nested farther from drilling rigs and gas wells in 2004 compared to available sites, and farther than they did from 2000 to 2003. Although adult females nested within previously selected nesting areas regardless of changes in local gas development levels, the nesting yearlings avoided road-related disturbances. Brooding females avoided producing wells during the early brood-rearing period.

Aldridge (2005) found that broods tended to be closer to well sites than expected by chance, but they avoided areas with a greater density of well sites within 1 km. He also found that sage-grouse broods tended to occur in areas with a greater density of trails. He did not describe trail characteristics, though the table (4-1) of GIS raster variables suggests that trails included in this analysis were unpaved or graveled truck trails. However, chick failure increased in habitats with more visible well sites within 1 km. When the model was applied to the landscape, areas with oil and gas activities fell into the extreme risk category for chick failure. The model predicted a 1.5 times increase in risk of chick failure for each well that was visible within 1 km. Aldridge concluded that this predicted mortality could be attributed to roads and predators perching on power lines and power poles associated with the wells.

Aldridge (2005) noted that no single human use within 1 km accounted for nest occurrence, but when all human uses were combined, there was strong avoidance by sage-grouse. He surmised that edge effect might account for this outcome, as edges are associated with higher predator density. Yet the analysis showed no effect of edge on nest success. Aldridge concluded that the failure to detect survival differences attributable to edge was a function of the strong avoidance of edge.

Lyon and Anderson (2003) found that females moved significantly farther from disturbed lek sites to their nest sites than did females at undisturbed sites. The difference in nest initiation rates was not significantly lower and hatching rates did not differ.

Pitman et al. (2005) found that Lesser Prairie-Chickens nested farther from wellheads than would be predicted by a random distribution, as measured by the proximity of the nearest 10% of nest sites. However, there were differences between their two study sites. At one site, the increased distance was statistically significant for all 11 nests. At the other site, the increased distance from the wellheads was not statistically significant for any of the eight nests. The authors acknowledge that in this study, the effect of wells is not clear. Although these are not reported in the paper, there may have been differences between the two areas that explain this clear difference in response. Some of the specific attributes considered by the authors include topography, noise level or type produced by the pump motors, and size of the pump jack. The area in which all cases of increased distance from wells was statistically significant was slightly more level, which may have made the wells more visible to the birds.

Like Aldridge and Boyce (2005), Pitman et al. (2005) used vegetation measurement and distance to anthropogenic features to develop a model to predict nest success. The highest

ranking model used five vegetation variables and a probability value of 0.26 (derived from observed nest success) to predict the probability of success only 58.1% of the time. The second model used 12 vegetation variables from a second set of data collected from a different area and successfully predicted nest success 74.6% of the time.

TRAFFIC VOLUME

Ingelfinger and Anderson (2004) found that the densities of sagebrush obligates – Brewer's Sparrow (*Spizella breweri*), Sage Sparrow (*Amphispiza belli*), and Sage Thrasher (*Oreoscoptes montanus*) – declined by as much as 60% within 100 m of the road with higher traffic volume (697 vehicles per day, depending on the season, time of day, and day of week) compared to roads with less traffic (344 vehicles per day). The 40% decline in density within 100 m of a road with an average volume of 12 vehicles per day was not statistically significant. When data were combined for all dirt roads surveyed in the second year of the study, statistically significant reductions in abundance were found for sagebrush obligates within 100 m of the roads. Along the pipeline, where there was no traffic, there was no observable difference in breeding bird abundance within 100 m and between 100 and 200 m. The investigators suggested that the noise associated with roads could not account for the latter result, and speculated that other road effects such as increased predators and non-native invasive plant species might account for the declines. They also speculated that the slight increase in Horned Larks along the disturbed areas might exclude Sage Sparrows in resource competition for the seeds found in the grassy areas that emerged after disturbance. The authors did not observe a decline in sagebrush obligates along the paved highway. They attributed this result to three factors: (1) the lower abundance of Horned Larks; (2) a fence that dissected the point count circles nearest the road and that served as an elevated perch, thus inflating detection probability; and (3) the relative absence of sagebrush cover compared to the other sites.

BUFFER ZONES

The models developed by Aldridge (2005), as applied to the available habitat, revealed that a 3.2-km buffer around each lek site would protect 54% of the critical nesting habitat and 62% of the brood-rearing critical habitat, where critical habitat is defined as primary or secondary as opposed to high-risk or sink. The 1-km buffer recommended by the Province of Alberta would protect only 9.9% of the critical brood-rearing habitat. In Aldridge's view, the use of lek-centered buffers was not adequate to protect habitat and he suggested his models should be applied to the landscape to identify and protect important primary and secondary nesting and brood-rearing habitats. Holloran (2005) posited that effect-distance from disturbance sources to leks during the breeding season could be conservatively estimated at 3 to 5 km. Breeding sites, or habitat suitable for breeding sites, would require a 5-km buffer.

POPULATION IMPACTS

Holloran (2005) assessed the impact on the regional population, finding an 8% decline of the male population growth rate. He considered the possibility that displaced male juveniles were dispersing beyond the study area, and thus were not counted in the recruitment rate, but based on typical dispersal distances, rejected this possibility. He also rejected the possibility that displaced juvenile males were breeding without a lek site. The failure of part of the

population to breed would lead to a gradual decline in that population, and eventually a local extirpation, which would result in a decline in the regional population.

In addition, the effect of natural gas development on female population growth was generally negative. Holloran found relatively consistent negative contributions to population growth from adult and yearling female annual survival, whereas productivity contributions were generally positive. He concluded that reduced population growth was attributable to decreased annual survival of adult and yearling females.

Aldridge (2000) reported that 67% of the 21 active leks found in Alberta and of the 31 active leks found in Saskatchewan in 1988 had been abandoned. The mean number of males per lek had declined by 80% in Alberta and 64% in Saskatchewan. However, he did not distinguish or attempt to distinguish among the numerous causes of habitat change in the region. Aldridge attributed the apparent stability in total number of males found in both provinces to increased survey effort.

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EMERGENT QUESTIONS AND RESEARCH NEEDS

Documenting and understanding the relationships between anthropogenic factors and grassland and shrub-steppe birds are complicated tasks. In reviewing published reports of research and monitoring dealing with the impact of anthropogenic factors on grassland and shrub-steppe birds, we have identified several projects with exemplary design, execution, and analysis (e.g., Leddy et al. 1999, Osborn et al. 2000, Robel et al. 2004). However, most “impact studies” we reviewed have multiple flaws that reduce the strength of their conclusions and make using them to compare or generalize their findings difficult. Rather, most of these studies can serve as valuable sources of hypotheses that will require further testing.

For every kind of human activity included in this paper, landscape-level studies are lacking. These multi-site, multivariate studies can be expensive, and so it is important to have coordinated efforts that include stratified random samples to ensure sample sizes and adequate power to detect statistically significant responses to a number of variables of interest. This is particularly important for variables such as tower lighting color or type, because observed differences may result instead from differences in topography or location. In addition, large-scale studies are needed to determine the overall impact of each of these activities. Current estimates of mortality are unreliable because the sample sizes are small relative to the number of a particular source. The extent of mortality associated with a given activity will help to adjust the extent to which mitigation and other conservation measures are needed, particularly if impacts on declining species are detected.

From a conservation and planning perspective, researchers need to know whether human structures and activities affect individuals by acting on reproductive success or survivorship and, ultimately, whether these effects are measurable at a population level. On paper, this appears to be a straightforward task of observing and recording demographic and population parameters. In reality, it can be a time-consuming and difficult endeavor. There is a growing recognition among ecologists that simple count data are rife with biases. As a result, there is also an encouraging trend toward employing empirically based corrective models to generate less biased estimates of demographic and population parameters. For example, the scavenging rate and observer proficiency studies now widely conducted in conjunction with wind power mortality studies (e.g., Young et al. 2003, Erickson et al 2004) provide valuable site-specific data for correcting fatality estimates. Recently, researchers have begun applying corrective models to abundance and density estimates based on point and transect counts. This should become the new standard for impact studies (O’Connell and Piorkowski 2006). Editors and reviewers for peer-reviewed journals are beginning to reject manuscripts that report only uncorrected counts (S. Mabey, personal observation)

There were three nearly universal design and execution problems: (1) small sample size; (2) inappropriate sampling scale or “grain”; and (3) inadequate descriptions of reference (control) and impact sites. All these problems often stem from logistical constraints, but they can seriously (and negatively) influence confidence in results. Small samples critically reduce statistical power and create skepticism regarding negative results. Statistical power is related to the probability of making a “Type II” error (mistakenly accepting a null hypothesis of no

impact). Researchers often argue that biologically significant effects can be overlooked when sample sizes are too small, because statistical tests with low power can detect only relatively large effects.

Sampling grain can refer to the temporal and spatial scales used to collect data. Ideally, sampling grain should mirror the frequency and extent of the parameters under study and should be able to capture the extremes and variability associated with the parameters. For example, carcass searches in mortality monitoring studies should be frequent enough to account for carcasses lost by scavenging, even if empirically derived post hoc estimate corrections are applied to the data. A recent meta-analysis of wind power fatality studies provides guidance about bird use/risk sampling frequency based on statistical correlations in count data across seasons (Erickson et al. 2002). Detecting patterns for species with highly variable spatial or temporal distributions (e.g., migrating birds or extreme habitat specialists) requires sampling at a higher frequency (finer scale) over a relatively large space or long time. An example of poor sampling grain imposed by logistical constraints can be found in an otherwise exemplary study by O'Connell and Piorkowski (2006). In their study of the impacts of wind power development on breeding bird abundance, they examined bird densities within three distance ranges from turbine (directly adjacent, 1 to 5 km, and 5 to 10 km). These distance ranges, coupled with the logistical and design constraints on the number of replicates within each distance range, undoubtedly diminished their ability to identify a threshold distance and mechanism (visual versus auditory disturbance) for the effects of turbines on breeding birds.

Most studies lack descriptions of “original conditions” as derived either from data and literature or from baseline studies. To adequately assess impacts, conditions before and after the activity in question need to be compared. Indeed, this is the premise behind the BACI design recommended as “preferred” in the Guidance Document for studying the impacts of wind energy on wildlife (NWCC 1999). Obtaining baseline data may not be within the control of the investigator. There may be no data about population size or other parameters for the species of interest in the area where development is to take place. There may be inadequate time to collect these data over several seasons and years; data vary from one year to the next, and any one year might represent an extreme of a normal range (or might even be outside a normal range), so a single season's or year's data may not represent conditions. Funding is always a limiting factor, particularly when the research funding is provided by the developer, who may seek to have studied only a very limited number of questions for a very short time. These questions are often demanded by regulatory bodies that have little or no scientific expertise.

Kennedy (1980) described in detail the kind of baseline information that is appropriate when studying the impact of energy development on birds. She noted that because raptors are high-level consumers in the ecosystem, impacts on raptors will have impacts throughout the ecosystem. We refer the reader to the discussion of the impact of power lines on raptors. Kennedy noted that baseline studies were generally limited to presence or absence, distribution, relative abundance, and general statements of species-specific habitat requirements and nesting success. Other data, which of course would not be site-specific, might

be obtained from literature. She recommended that baseline studies include species composition in the development area, each species' productivity, and the expected yearly change in population size and productivity. She also recommended that environmental and habitat conditions such as food availability be studied. Understanding regulating factors, such as feeding and nesting requirements and changes in the prey base, were also deemed necessary. Kennedy also stressed the need to sample over a biologically appropriate area rather than limiting the study to the development site. More generally, she recommended that baseline studies should answer the following questions:

- Which of the elements that control raptor populations may be affected by this development?
- How long would these impacts take to be manifested?
- Will the effects be short- or long-term?
- Will the effects be limited to the area of disturbance?
- Will mitigative measures and management plans alleviate the negative impacts?

A frequently encountered methodological flaw in papers examined for this review was lack of quantitative data describing the habitat at both the impact site (e.g., wind facility, road side) and the reference site used for comparison to establish "normal" or baseline patterns against which impact could be assessed. Although observational studies are greatly strengthened by comparative data from replicates found in treatment and control conditions (James and McCulloch 1995), access and resources can severely constrain where ecologists establish reference or control sites (Knick et al. 2003). Even when logistical factors limit the location and number of reference sites, relevant characteristics of experimental and control sites (e.g., vegetation, hydrology, topography) need to be quantified so that at a minimum post hoc analyses can help identify confounding factors that may have influenced observed patterns. Gill et al. (1996) outlined basic data needs for detecting effects of disturbance on a population. Bennett and Milne (2004) proposed a method for prescreening replicates at the beginning of long-term ecological monitoring studies.

In addition to documenting the effects of anthropogenic factors on grassland and shrub-steppe birds, there is a strong interest in uncovering the causal mechanisms behind individual and population-level responses that can help identify the best ways to mitigate and reduce impacts. Studies designed to investigate the causal mechanisms of human-induced disturbances and avoidances or changes in demographic parameters in grassland and shrub-steppe birds are the rare exception (e.g., Shawkey et al. 2004). Knick et al. (2003) suggest there is a need for carefully designed hierarchical studies that track multiple factors (e.g., weather, prey availability, disease vectors) in addition to human elements in the grassland and shrub-steppe dominated landscape. Research quantifying important fitness-related behaviors (e.g., foraging, parental care, courtship) could also contribute meaningfully to a greater understanding of how human activities and structures affect grassland and shrub-steppe birds (Gill et al. 1996, 2001).

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APPENDIX A: COMPLETE SEARCH BIBLIOGRAPHY

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APPENDIX B: GRASSLAND AND SHRUB-STEPPE BIRD SPECIES

To our knowledge, there are no authoritative or accepted lists of birds in habitat guilds. This may be because a species uses more than one habitat type, and because some species are obligates, whereas others are generalists that may or may not have a dominant or preferred habitat.

The lists of grassland and shrubland species provided below were generated by Dan Niven of the National Audubon Society, who developed a “bird × habitat” table for the Audubon Publication “State of the Birds” (Butcher 2004). Niven’s list of shrubland species includes birds of all types of shrubland. Therefore, we have modified it to include only those species identified in the Partners in Flight Continental Plan as “western shrub” species. They are so noted with an asterisk. However, the Continental Plan classifies species by conservation need (immediate, management, and long-term planning and responsibility), according to biome. It does not purport to be a comprehensive list of species associated with each habitat type. Therefore, some species associated with shrub-steppe habitat may not be identified from these lists.

Grassland bird species¹

Rock Ptarmigan (*Lagopus muta*)
White-tailed Ptarmigan (*Lagopus leucura*)
Greater Prairie-chicken (*Tympanuchus cupido*)
Northern Harrier (*Circus cyaneus*)
White-tailed Hawk (*Buteo albicaudatus*)
Ferruginous Hawk (*Buteo regalis*)
Rough-legged Hawk (*Buteo lagopus*)
Golden Eagle (*Aquila chrysaetos*)
American Kestrel (*Falco sparverius*)
Aplomado Falcon (*Falco femoralis*)
Prairie Falcon (*Falco mexicanus*)
American Golden-Plover (*Pluvialis dominica*)
Pacific Golden-Plover (*Pluvialis fulva*)
Mountain Plover (*Charadrius montanus*)
Upland Sandpiper (*Bartramia longicauda*)
Eskimo Curlew (*Numenius borealis*)
Long-billed Curlew (*Numenius americanus*)
Buff-breasted Sandpiper (*Tryngites subruficollis*)
Barn Owl (*Tyto alba*)
Snowy Owl (*Bubo scandiacus*)
Burrowing Owl (*Athene cunicularia*)
Short-eared Owl (*Asio flammeus*)
Sky Lark (*Alauda arvensis*)
Horned Lark (*Eremophila alpestris*)
Northern Wheatear (*Oenanthe oenanthe*)
Red-throated Pipit (*Anthus cervinus*)
American Pipit (*Anthus rubescens*)

Sprague's Pipit (*Anthus spragueii*)
Cassin's Sparrow (*Aimophila cassinii*)
Botteri's Sparrow (*Aimophila botterii*)
Vesper Sparrow (*Pooecetes gramineus*)
Lark Bunting (*Calamospiza melanocorys*)
Savannah Sparrow (*Passerculus sandwichensis*)
Grasshopper Sparrow (*Ammodramus savannarum*)
Baird's Sparrow (*Ammodramus bairdii*)
Henslow's Sparrow (*Ammodramus henslowii*)
McCown's Larkspur (*Calcarius mccownii*)
Lapland Longspur (*Calcarius lapponicus*)
Smith's Longspur (*Calcarius pictus*)
Chestnut-collared Longspur (*Calcarius ornatus*)
Snow Bunting (*Plectrophenax nivalis*)
McKay's Bunting (*Plectrophenax hyperboreus*)
Dickcissel (*Spiza americana*)
Bobolink (*Dolichonyx oryzivorus*)
Eastern Meadowlark (*Sturnella magna*)
Western Meadowlark (*Sturnella neglecta*)

¹ Generalists that may use grassland habitat are not included.

Shrub-steppe bird species¹

Greater Sage-Grouse (*Centrocercus urophasianus*)
Gunnison Sage-Grouse (*Centrocercus minimus*)
Willow Ptarmigan (*Lagopus lagopus*)
Sharp-tailed Grouse (*Tympanuchus phasianellus*)
Lesser Prairie-Chicken (*Tympanuchus pallidicinctus*)
Mountain Quail (*Oreortyx pictus*)
Scaled Quail (*Callipepla squamata*)
California Quail (*Callipepla californica*)
Gambel's Quail (*Callipepla gambelii*)
Northern Bobwhite (*Colinus virginianus*)
Harris' Hawk (*Parabuteo unicinctus*)
Crested Caracara (*Caracara cheriway*)
Bristle-thighed Curlew (*Numenius tahitiensis*)
Inca Dove (*Columbina inca*)
Common Ground-Dove (*Columbina passerina*)
Greater Roadrunner (*Geococcyx californianus*)
Smooth-billed Ani (*Crotophaga ani*)
Groove-billed Ani (*Crotophaga sulcirostris*)
Lesser Nighthawk (*Chordeiles acutipennis*)
Antillean Nighthawk (*Chordeiles gundlachii*)
Common Pauraque (*Nyctidromus albicollis*)

Common Poorwill (*Phalaenoptilus nuttalli*)
Buff-collared Nightjar (*Caprimulgus ridgwayi*)
Lucifer Hummingbird (*Calothorax lucifer*)
Costa's Hummingbird (*Calypte costae*)
Calliope Hummingbird (*Stellula calliope*)
Rufous Hummingbird (*Selasphorus rufus*)
Allen's Hummingbird (*Selasphorus sasin*)
Willow Flycatcher (*Empidonax traillii*)
Vermilion Flycatcher (*Pyrocephalus rubinus*)
Dusky Flycatcher (*Empidonax oberholseri*)
Ash-throated Flycatcher (*Myiarchus cinerascens*)
Scissor-tailed Flycatcher (*Tyrannus forficatus*)
Loggerhead Shrike (*Lanius ludovicianus*)
Northern Shrike (*Lanius excubitor*)
White-eyed Vireo (*Vireo griseus*)
Bell's Vireo (*Vireo bellii*)
Black-capped Vireo (*Vireo atricapilla*)
Gray Vireo (*Vireo vicinior*)
Florida Scrub-Jay (*Aphelocoma coerulescens*)
Island Scrub-Jay (*Aphelocoma insularis*)
Western Scrub Jay (*Aphelocoma californica*)
Tamaulipas Crow (*Corvus imparatus*)
Chihuahan Raven (*Corvus cryptoleucus*)
Verdin (*Auriparus flaviceps*)
Cactus Wren (*Campylorhynchus brunneicapillus*)
Bewick's Wren (*Thryomanes bewickii*)
Arctic Warbler (*Phylloscopus borealis*)
California Gnatcatcher (*Polioptila californica*)
Black-tailed Gnatcatcher (*Polioptila melanura*)
Black-capped Gnatcatcher (*Polioptila nigriceps*)
Bluethroat (*Luscinia svecica*)
Mountain Bluebird (*Sialia currucoides*)
Wrentit (*Chamaea fasciata*)
Gray Catbird (*Dumetella carolinensis*)
Northern Mockingbird (*Mimus polyglottos*)
Sage Thrasher (*Oreoscoptes montanus*)
Brown Thrasher (*Toxostoma rufum*)
Long-billed Thrasher (*Toxostoma longirostre*)
Bendire's Thrasher (*Toxostoma bendirei*)
Curve-billed Thrasher (*Toxostoma curvirostra*)
California Thrasher (*Toxostoma redivivum*)
Crissal Thrasher (*Toxostoma crissale*)
LeConte's Thrasher (*Toxostoma lecontei*)
Eastern Yellow Wagtail (*Motacilla tschutschensis*)

Blue-winged Warbler (*Vermivora pinus*)
Golden-winged Warbler (*Vermivora chrysoptera*)
Virginia's Warbler (*Vermivora virginiae*)
Lucy's Warbler (*Vermivora luciae*)
Yellow Warbler (*Dendroica petechia*)
Chestnut-sided Warbler (*Dendroica pensylvanica*)
Prairie Warbler (*Dendroica discolor*)
McGillivray's Warbler (*Oporornis tolmiei*)
Wilson's Warbler (*Wilsonia pusilla*)
Rufous-capped Warbler (*Basileuterus rufifrons*)
Yellow-breasted Chat (*Icteria virens*)
White-collared Seedeater (*Sporophila torqueola*)
Olive Sparrow (*Arremonops rufivirgatus*)
Green-tailed Towhee (*Pipilo chlorurus*)
Eastern Towhee (*Pipilo erythrophthalmus*)
Spotted Towhee (*Pipilo maculatus*)
Canyon Towhee (*Pipilo fuscus*)
California Towhee (*Pipilo crissalis*)
Albert's Towhee (*Pipilo aberti*)
Rufous-winged Sparrow (*Aimophila carpalis*)
Rufous-crowned Sparrow (*Aimophila ruficeps*)
Five-striped Sparrow (*Aimophila quinquestriata*)
American Tree Sparrow (*Spizella arborea*)
Clay-colored Sparrow (*Spizella pallida*)
Brewer's Sparrow (*Spizella breweri*)
Field Sparrow (*Spizella pusilla*)
Black-chinned Sparrow (*Spizella atrogularis*)
Black-throated Sparrow (*Amphispiza bilineata*)
Sage Sparrow (*Amphispiza belli*)
Fox Sparrow (*Passerella iliaca*)
Song Sparrow (*Melospiza melodia*)
Lincoln's Sparrow (*Melospiza lincolni*)
Harris' Sparrow (*Zonotrichia querula*)
White-crowned Sparrow (*Zonotrichia leucophrys*)
Golden-crowned Sparrow (*Zonotrichia atricapilla*)
Northern Cardinal (*Cardinalis cardinalis*)
Pyrrhuloxia (*Cardinalis sinuatus*)
Blue Grosbeak (*Passerina caerulea*)
Lazuli Bunting (*Passerina amoena*)
Varied Bunting (*Passerina versicolor*)
Painted Bunting (*Passerina ciris*)
Audubon's Oriole (*Icterus graduacauda*)
Scott's Oriole (*Icterus parisorum*)
Common Redpoll (*Carduelis flammea*)

Hoary Redpoll (*Carduelis hornemanni*)
Lawrence's Goldfinch (*Carduelis lawrencei*)

¹ Generalists that may use shrub-steppe habitat are not included.

Butcher, G. 2004. State of the Birds. Audubon October 2004 (special pull-out section; pages not numbered in sequence with magazine). Available online at <http://www.audubon.org/bird/stateofthebirds/>