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Source: The American Midland Naturalist, 173(1):122-132.

Published By: University of Notre Dame

DOI: <http://dx.doi.org/10.1674/0003-0031-173.1.122>

URL: <http://www.bioone.org/doi/full/10.1674/0003-0031-173.1.122>

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Effects of Grassland Biomass Harvest on Nesting Pheasants and Ducks

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ABSTRACT.—Grasslands enrolled in conservation programs provide important habitat for nesting game birds and waterfowl, but conservation grasslands have been targeted as a source of biomass for bioenergy and this could impact nesting birds. We studied the effects of biomass harvest on nest success and density using 109 blue-winged teal (*Anas discors*), mallard (*Anas platyrhynchos*), and ring-necked pheasant (*Phasianus colchicus*) nests found in southwestern Minnesota during 2009 (pretreatment) and 2010 (posttreatment). Grassland biomass was harvested in late autumn of 2009 with production-scale machinery. Harvest treatments included controls (0% biomass removal), partial harvest (50 or 75% biomass removal), and full harvest (100% biomass removal) from 8 ha plots. Nest success averaged 31% and was not influenced by biomass harvest. Daily survival rates were greater for nests located closer to wetlands. Estimated total nest density (0.42 nests ha⁻¹; corrected for survivorship) was similar across harvest treatments, but within-plot analysis revealed nest density was greater in unharvested refuge regions. Estimated nest density was positively correlated with vegetation height and the spatial extent of wetlands surrounding each plot. Harvesting relatively small-scale patches of conservation grasslands in late autumn does not appear to be detrimental to nesting ducks and pheasants the following spring, but managers should consider leaving unharvested refuges near wetlands when harvesting large continuous tracts.

INTRODUCTION

State and federal governments have instituted numerous programs to expand and manage native grasslands as wildlife habitat for grassland birds, including several ecologically and economically important game and nongame bird species (Herkert *et al.*, 1996). For example the Minnesota Department of Natural Resources manages restored grasslands in the Wildlife Management Area (WMA) program, which is publically accessible for hunting. WMAs cover more than 1.1 million acres of Minnesota and some require regular maintenance to sustain early-successional herbaceous plants. Minnesota agencies plan to expand WMA acreage by 64% by 2050 (Yunker, 2010), but increased land value due to rising crop prices (Rashford *et al.*, 2011) and increased management costs could hinder expansion goals. Land acquisition and management have been primarily funded by hunting

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license fees and state funds, but it is not known if these sources can support future habitat goals.

Biomass from conservation grasslands can be harvested and sold to bioenergy producers or other markets to potentially finance the expansion and maintenance of conservation grasslands (Fargione *et al.*, 2009). Biomass yields from WMAs in southwest Minnesota were about 3 Mg ha^{-1} (Jungers *et al.*, 2013), which could bring revenues for achieving expansion goals. Moreover, biomass harvest could be used as an alternative to more resource-intensive prescribed burning to maintain early-successional plant communities (Devries and Armstrong, 2011). If resulting habitat characteristics and wildlife benefits are similar for both management operations, biomass harvest could provide funds through sales of biomass and also conserve funds by reducing costs of prescribed burning.

Conservation grasslands, such as WMAs, provide productive breeding habitat for upland-nesting waterfowl and pheasants (Kantrud, 1994; Reynolds *et al.*, 2001). It is unclear how this habitat might be impacted by biomass harvest, and even though the effects of other land management activities on nest success and density have been well studied, results are inconsistent. For instance spring grazing and prescribed burning decreased the density of blue-winged teal (*Anas discors*) nests in North Dakota but did not influence nest success (Kruse and Bowen, 1996). Positive effects of biomass removal were evident when waterfowl nest success and density increased after mowing and burning of restored grasslands in Canadian prairies (Devries and Armstrong, 2011). The mechanisms underlying the varying effects of other biomass removal techniques on nest success and distribution are related to both local and landscape characteristics. Increases in nest success have been associated with nest-scale habitat variables such as vegetation height (Luttschwager *et al.*, 1994), field-scale variables such as legume cover (Arnold *et al.*, 2007), and landscape-scale variables such as surrounding grassland cover (Stephens *et al.*, 2005; Thompson *et al.*, 2012) and fragmentation (Horn *et al.*, 2005). Therefore, analysis at multiple spatial scales is important for understanding the effects of management activities on reproductive rates (Koper and Schmiegelow, 2006).

Our primary objective was to assess the effect of autumn biomass harvest on nesting biology of upland-nesting ducks and pheasants. We modeled densities and daily survival rates of duck and pheasant nests at two spatial scales, among fields with different harvest intensities and within fields that were subject to partial harvest, to identify responses to different harvest treatments. As a secondary objective, we tested the influence of habitat covariates on nest success and density as well as the potential effect of harvest on timing of nest initiation.

METHODS

STUDY SITE

We conducted our study on WMAs in Cottonwood, Jackson, and Nobles counties of Minnesota, U.S.A. (from 43.76° to 43.92°N , 95.15° to 95.63°W ; Fig. 1). In 2008 we delineated 28 plots within existing fields of restored grassland established >5 y before the project started. Each plot was approximately 8 ha and included a variety of warm- and cool-season grasses, legumes, and other forbs. Plots were selected to be dry enough to operate farm equipment during the autumn months.

Each plot was randomly assigned one of six harvesting treatments: (1) control at 0% harvest, (2) 100% full harvest, (3) 75% partial block harvest, (4) 75% partial strip harvest, (5) 50% partial block harvest, and (6) 50% partial strip harvest (Fig. 1). Partially harvested plots contained refuges of unharvested vegetation. In mid-Nov. of 2009, a contracted

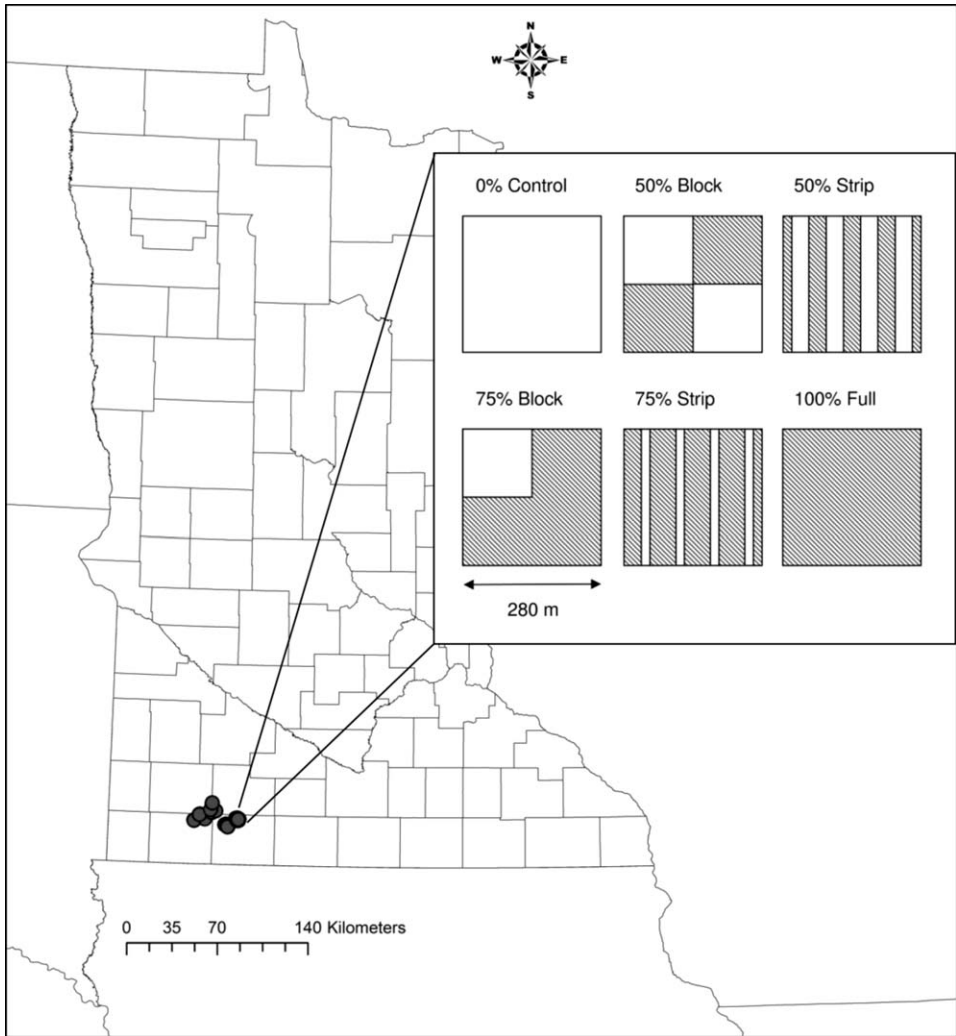


FIG. 1.—Distribution of 28 conservation grassland plots on Wildlife Management Areas in southwest Minnesota. Inset is a graphical depiction of the six biomass harvest treatments randomly assigned to each 8 ha plot (stippled regions indicate harvest). Actual harvested areas varied from the targeted proportion

harvester cut biomass with a self-propelled windrower to a minimum stubble height that prevented equipment damage (mean = 15 cm). Biomass was removed from the plot with a large round baler. One plot scheduled for harvest was not cut due to inclement weather and was treated as a control. Not all areas planned for harvest were cut due to woody encroachment, wet ground, and other obstructions. The actual harvested area within each plot was determined by measuring the perimeter of the cut areas with a global positioning system. Although the experimental design established categorical levels for harvest treatment, inconsistencies in harvest proportions required that we test the effect of harvest

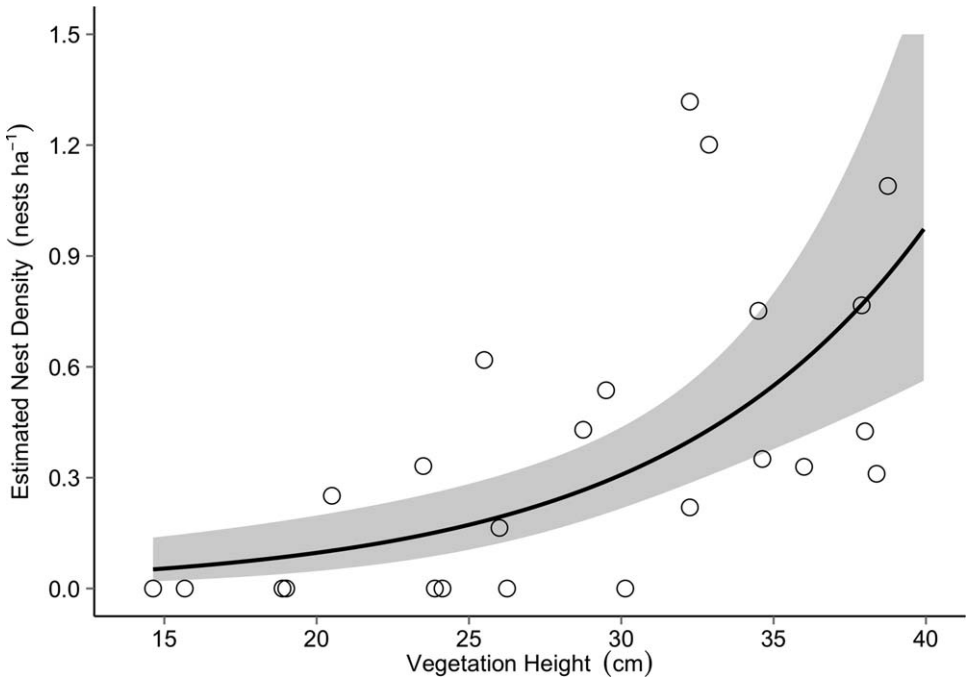


FIG. 2.—Model-based estimates of post-harvest nest density in relation to vegetation height (regression line; $\pm 95\%$ prediction intervals), with wetland cover and plot area held constant at their mean values of 17,654 m² and 7.9 ha, respectively. Data points are observed values from each nesting field, corrected for nests that failed prior to discovery

as a continuous variable. We also compared nest-level response variables within harvested and refuge regions; where refuge regions were unharvested areas within partially harvested plots and control plots. Harvested regions were the harvested areas within partially harvested plots and 100% fully harvested plots. The experiment was replicated in four blocks, each block further containing two replicates of the full harvest treatment, one of the control, and four partially harvested plots.

DATA COLLECTION

We searched for nests from 20 May 2009 to 18 Jun. 2009 and from 20 May 2010 to 8 Jul. 2010 using the chain drag method (Klett *et al.*, 1986). We searched each plot twice per year at three-week intervals. Crews of three (two drivers, one spotter) pulled a 30 m chain between a pair of all-terrain vehicles to flush nesting females from nests. Upon flushing a female, we recorded the nest location, if one was found, with a global positioning system and a flag placed 3 m north of the nest. At discovery and each subsequent visit, we estimated nest age and initiation date by counting eggs (assuming females laid one egg per day) and estimating embryo development by candling (Weller, 1956). We estimated the hatch date for each nest by adding the clutch size to the expected 26 d incubation period. We revisited marked nests every 7 d until nests hatched, were abandoned, or were destroyed. We considered a nest successful if at least one egg successfully hatched. We took digital photographs of nest bowls and collected nest remains to assist in determining final nest fate.

TABLE 1.—Description of all tested predictor variables for nest survival and density models

Predictors	Code ¹	Description	Scale ²
DSR Model			
Year	Yr	Categorical: Indicates if the nest was found in 2009 or 2010	nest-level
Refuge	Refuge	Categorical: Indicates if the nest was in a harvested or refuge area within the plot	nest-level
Percent harvested	Harvest	Continuous: Indicates the % of each plot harvested in fall 2009	plot-level
Species	Spp	Categorical: Indicates which species initiated the nest	nest-level
Nest age	Age	Continuous: Julian day on which the nest was initiated relative to sampling start date	nest-level
Nearest wetland	Wet	Continuous: Distance (m) of the nest to the nearest wetland	nest-level
Nest Density Model			
Percent harvested	Harvest	Continuous: Indicates the % of each plot harvested in fall 2009	plot-level
Vegetation height	Veg	Continuous: Mean height (cm) of vegetation within plot	plot-level
Grassland	Grass	Continuous: Area (m ²) of grassland within 500 m radius of plot center	plot-level
Wetland	Wet	Continuous: Area (m ²) of wetland cover within 500 m radius of plot center	plot-level
Plot area	Area	Continuous: Area (ha) of each plot	plot-level

¹ Abbreviated code for each predictor variable used in Table 2

² Indicates if the parameters were measured at the scale of plot- or nest-level

We measured vegetation height at eight random locations in each plot between 27 May 2010 and 10 Jun. 2010. Vegetation height was determined by visually assessing the distance above ground in which 80% of biomass occurred (Stewart *et al.*, 2001). Values from the eight points were averaged to generate a mean vegetation height for each plot.

We quantified the amount of grassland and wetland in the surrounding landscape using ArcGIS (version 9.3.1, ESRI, Redlands, California) and GAP Land Cover data from the Minnesota Department of Natural Resources (United States Geological Survey, 2011). We calculated the amount of grassland and wetland areas that were within a 500 m radius from the plot center and outside of the plot boundary to be used as a plot-scale habitat covariate for examining variation in nest density. We also measured the distance from each nest to the nearest wetland edge using the same data layers.

NEST SURVIVAL ANALYSIS

We modeled daily survival rate (DSR) of nests with program MARK (White and Burnham, 1999) using procedures described by Dinsmore *et al.* (2002). The effect of biomass harvest on DSR was measured at two scales. The plot-scale predictor labeled “harvest” indicates the actual proportion of biomass removed from the plot during harvest (Table 1). For partially harvested plots (*i.e.*, those treatments with a refuge), we included a dummy variable denoted “refuge” to identify nests located in unharvested refuge areas. We tested for variation in DSR in relation to three categorical main effects: (1) refuge (cut or uncut), (2) year (2009 or 2010), and (3) species (mallard, blue-wing teal, or pheasant) along with one continuous main effect, percent harvested. We also tested the effect of two continuous covariates: nest

TABLE 2.—Akaike Information Criteria based on small sample size (AIC_c), differences in AIC_c between top ranked and null models (ΔAIC_c), Akaike weights (ω), and number of parameters (k) for candidate models estimating nest daily survival rate (DSR) and density in conservation grasslands

Model parameters ¹	AIC_c	ΔAIC_c	ω	k
DSR Model				
Wet	187.6	0.0	0.24	2
Age + Wet	188.2	0.6	0.19	3
Null	188.5	0.9	0.16	1
Age	189.6	2.0	0.09	2
Species	189.9	2.3	0.07	3
Age + Harvest + Wet	190.1	2.5	0.07	4
Yr	190.3	2.7	0.07	2
Harvest	190.3	2.7	0.06	2
Refuge	190.5	2.9	0.06	2
Age + Harvest	191.6	4.0	0.03	3
Yr + Harvest	192.3	4.7	0.03	3
Density Model				
Wet + Veg + Area	104.0	0.0	0.34	4
Harvest + Area + Veg + Grass + Wet	104.4	0.4	0.28	6
Harvest + Veg	105.0	1.0	0.2	3
Veg + Area	105.2	1.3	0.18	3
Null	126.9	23.0	0	1
Harvest + Wet	128.1	24.2	0	3
Harvest + Grass	129.7	25.7	0	3
Harvest + Area	130.3	26.3	0	3
Wet+ Area	130.3	26.3	0	3
Grass + Area	132.0	28.0	0	3

¹ See Table 1 for codes and description for each predictor

age (number of days between nest initiation and sampling start date) and proximity to wetlands (m from nearest wetland edge; Table 1). Ten models with various combinations of these variables comprised our *a priori* model set (Table 2). We ranked models based on Akaike's information criterion adjusted for small sample size (AIC_c) (Burnham and Anderson, 1998). We estimated overall nest success as DSR^{35} (Klett *et al.*, 1986).

NEST DENSITY ANALYSIS

We considered apparent nest density as the total number of nests found per plot. To account for nests that failed before discovery, we used a Horvitz-Thompson estimator of total nests initiated per plot based on model-estimated DSR and average nest age at discovery (Arnold *et al.*, 2007):

$$NEST = \frac{N_i}{DSR^{d_i}}$$

where N_i is apparent nest density in plot i , DSR is estimated daily survival rate for all species from the best-supported model, and d_i is the average nest age at time of discovery in plot i . We rounded $NEST$ (nest abundance corrected for survivorship) to the nearest integer and treated it as a measure of nest density (nests plot⁻¹).

We modeled estimated nest density per plot using negative binomial generalized linear regression from the MASS package in R (R Development Core Team, 2010). We developed

an *a priori* model set that included 10 models with biologically relevant combinations of main effects and habitat covariates (Tables 1, 2). “Percent harvest” was treated as a continuous main effect and continuous habitat covariates included average vegetation height within each plot, amount of surrounding grassland, and amount of surrounding wetland. Although plots were similar in size (mean 7.9 ha \pm 0.4 SD), we included plot area as an offset variable to control for subtle variation in plot size. These habitat covariates have been used to describe variation in nest density and survival in previous studies (Reynolds *et al.*, 2001; Stephens *et al.*, 2005; Arnold *et al.*, 2007; Kruse and Bowen, 1996). Models were compared and ranked based on AIC_c. Because the habitat covariates were only measured in 2010, we restricted this analysis to plots searched for nests in 2010 (n = 28 plots).

In partially harvested treatment plots, nests were found in both harvested and refuge regions. Because we generated nest density estimates at the plot scale, we could not use these estimates to examine density differences between refuge and harvested regions. To compare nest densities in refuge and harvested regions within plots, we used a chi-square test. We divided the total number of nests found by the total area searched in 2009 to calculate the expected number of nests ha⁻¹. We then multiplied this fraction by the total number of hectares searched in 2010 for both refuge and harvested regions to generate the number of nests we expected to find. All nests found in control plots were included with those analyzed in the refuge region group, and all nests found in the 100% harvest plots were included with those in the harvested region group. We compared observed and expected numbers of nests found in each region with a chi-square test with 1 d.f.

We used analysis of variance (ANOVA) to determine if nest initiation date varied by species and nest location (harvested or refuge region) using data from 2010. We determined significance for all tests at $\alpha \leq 0.05$.

RESULTS

We found 109 nests, including 62 blue-winged teal (*Anas discors*), 32 mallard (*Anas platyrhynchos*), and 15 ring-necked pheasant (*Phasianus colchicus*) from 28 plots (totaling 221 ha) during both years of the study. We determined nest fate for 80 nests, 46 in 2009 and 34 in 2010.

NEST SURVIVAL

Daily survival rate of nests did not vary by year, so we combined nests from both years for analysis. The best-supported model (Table 2) estimated DSR at 0.9653 ± 0.0054 SE, which translated to a nest success rate of 31%. The best-supported model indicated a negative relationship between DSR and distance to nearest wetland ($\beta = -9.8 \times 10^{-4} \pm 5.8 \times 10^{-4}$ SE). The next best model included the variable for distance to nearest wetland as well as nest age (Table 2). There was little support for an affect of harvest treatment on DSR. All models that included the percent harvest variable had Akaike weights <8%. Similarly, the model with the variable nest location ranked low in the candidate set ($\omega = 0.06$).

NEST DENSITY

We found an average of 1.9 nests plot⁻¹ \pm 0.04 SE, which translates to an apparent nest density of 0.25 nests ha⁻¹ \pm 0.01 SE. Estimated nest density corrected for survivorship averaged 0.42 nests ha⁻¹ \pm 0.01 SE across all treatments and years. The best-supported model for explaining variation in estimated nest density at the plot level included vegetation height, amount of surrounding wetland, and plot area (Table 2). Another competitive model included two additional variables: amount of surrounding grassland and percent

harvest. These two top models accounted for 62% of the Akaike weights (Table 2). Vegetation height, plot area, and the amount of wetland (m^2) within a 500 m radius of the plot center were positively associated with estimated nest density in both top models. The amount of surrounding grassland was negatively associated with estimated nest density and was not correlated with the amount of surrounding wetland (Pearson's correlation coefficient = 0.34). The proportion of area harvested was positively correlated with nest density, however the SE of this coefficient was more than four times the estimated coefficient ($\beta = 1.3 \times 10^{-3} \pm 5.4 \times 10^{-3}$ SE, $P = 0.83$). Models that included the percent harvest variable but not vegetation height were not well supported (Table 2).

In 2010 nest searches found 17 nests within 140 harvested ha for an apparent density of 0.12 nests ha^{-1} , versus 30 nests within 84 ha of refuge regions for an apparent density of 0.36 nests ha^{-1} ($\chi^2 = 16.2$; $\text{df} = 1$; $P < 0.001$). Averaged across species, nests were initiated 15 d earlier in refuge regions (mean = 15 May) compared to harvest regions (mean = 30 May; $F_{1,40} = 19.7$; $P < 0.001$). When we used this to adjust nest density for nests that failed before detection, it led to an increase in the estimated difference in density between harvested and refuge regions. Estimated nest density was 0.17 nests ha^{-1} in harvested regions versus 0.65 nests ha^{-1} in refuge regions. Nest initiation date was earlier for all species in the refuge regions, but also varied by species ($F_{2,40} = 5.0$; $P = 0.012$). Pheasants initiated nests about 19 d earlier than mallards (Tukey's honestly significant difference = 10 d), but initiation dates were similar for blue-winged teal (23 May) and mallards (28 May). The interaction between species and harvest treatment was not significant for initiation date ($F_{2,40} = 0.04$; $P = 0.95$).

DISCUSSION

Harvesting biomass from conservation grasslands in autumn did not decrease the number of nesting game birds, nor did it increase the risk of nest failure in 8 ha plots the following year. However, we observed fewer nests per hectare in harvested regions compared with refuge regions. Our results suggest that when ducks and pheasants have access to unharvested refuge regions for nesting, local nest densities will not decline due to biomass harvest, even though birds avoided nesting in recently harvested portions of WMAs.

Other studies have also found waterfowl preferentially select nest sites with some residual grass. Kruse and Bowen (1996) recorded species-specific declines in nest density in response to vegetation removal (burning and grazing) and associated these declines with differences in vegetation height among removal treatments. Likewise, Luttschwager *et al.* (1994) measured lower nest densities in hayed fields compared to idle fields after the earliest nest search the year after management, which they attributed to decreased vegetation height.

Other studies on the impacts of haying on waterfowl production observed a decline in nest success as a result of direct nest destruction by harvesting machinery, which can be mediated by delaying harvest until after waterfowl nesting occurs (McMaster *et al.*, 2005). Although the mechanical techniques for harvesting biomass for energy are similar to those for haying, the timing of biomass harvest is considerably later. As anticipated fall biomass harvest did not cause direct nest losses in our study. Delaying biomass harvest of perennial grasslands until after plant senescence also permits the translocation of nutrients from shoots to roots (Vogel *et al.*, 2002), therefore conserving resources for growth in following years and limiting emissions during combustion for energy (Ogden *et al.*, 2010).

Our estimate of nest success (31%) was substantially greater than the 5–15% nest success observed in Canadian grasslands under delayed haying management (Emery *et al.*, 2005) and was also greater than the 13% nest success rate observed by Thompson *et al.* (2012) in

unharvested conservation grasslands about 200 km north of our sites. Our relatively high DSRs compared to other studies could be due to variation in predator communities across studies. Although we did not directly measure predator populations, we can infer predator habitat suitability based on land cover characteristics like grassland and wetland edges (Phillips *et al.*, 2003). In our study nests initiated further from wetlands had a lower DSR compared with those that were initiated closer to wetlands; whereas other studies observed an opposite trend as fields surrounded by a greater number of wetlands had lower DSRs (Stephens *et al.*, 2005; Thompson *et al.*, 2012). Some predators, such as red fox (*Vulpes vulpes*), prefer foraging along wetland edges in regions with low grassland cover (15–20%) compared to regions with high grassland cover (45–55%; Phillips *et al.*, 2003). Other studies did not identify a relationship between DSR and nest proximity to wetlands (Reynolds *et al.*, 2001; Arnold *et al.*, 2007). It is possible these study locations were more similar to our study location in terms of predator communities and landscape composition compared to those with contradictory findings (Stephens *et al.*, 2005; Thompson *et al.*, 2012).

Estimated nest density was relatively low (0.42 nests ha⁻¹) compared with those reported by Arnold *et al.* (2007; 1.5 nests ha⁻¹) and Devries and Armstrong (2011; 1.33 nests ha⁻¹), who recorded waterfowl nest densities in other areas of the prairie pothole region, where waterfowl densities are typically greater. Because we selected our research sites based on bioenergy potential rather than waterfowl productivity, it was not surprising we recorded lower nest densities. Modeling nest density as the number of nests per plot required measuring predictors at the plot scale, and the most important predictors were related to vegetation height in the plot and habitat surrounding the plot, with both vegetation height and the area of wetlands within 500 m of each plot center being positively correlated with nest density. Typically, mallard and blue-winged teal densities are greater in habitats with greater wetland densities (Johnson and Grier, 1988), and our study supports previous findings that nest density is positively correlated to the proximity of wetlands (Arnold *et al.*, 2007; Devries and Armstrong, 2011). Biomass harvesting equipment is vulnerable to damage and not efficient when operated near wetlands and on wet ground when used to harvest biomass in late autumn (Williams *et al.*, 2012). Therefore, until harvesting equipment is improved, harvesting operations will not likely occur on fields with greater relative densities of waterfowl nests.

Nest detection probability may not have been equal across treatments. The chain drag method we used to find nests required that females flushed in response to the chain. It is possible a female may have been more likely to flush from a nesting site with lower vegetation height since the lack of tall vegetation could have made it more difficult to avoid the disturbance. However, variation in nest detection probability across upland habitat conditions has not been tested. Moreover, if this bias were true, we would be underestimating the number of nests in the refuge area. Such a bias would not change our result that nest density appears greater in refuge regions compared to harvested regions but rather increase the magnitude of the difference.

MANAGEMENT IMPLICATIONS

Our data suggest autumn biomass harvest did not decrease the number of nesting ducks and pheasants nor was it detrimental to nest survival following one year of harvest management. Nest density was greater in refuge regions compared with harvested regions, which is evidence that the refuge regions are important for nesting waterfowl and pheasants when grasslands were managed for bioenergy. Female ducks and pheasants appeared to avoid nesting in harvested regions early in the spring, but this had no measureable effect on nest survival. Selecting perennial grassland sites for harvest that are further from wetlands –

which may increase bioenergy potential of the site – would alter habitat at sites less preferential for nesting waterfowl. Although more data are required to determine how much refuge is necessary to optimize the joint production of waterfowl and bioenergy, we recommend orienting refuges closer to wetlands to support nesting waterfowl. Similar studies are needed to record nest survivorship and density for two or more years following biomass harvest (Devries and Armstrong, 2011) and to expand the spatial scale beyond 8 ha plots.

Acknowledgments.—We are grateful to Kevin Johnson, Melissa DonCarlos, and all the technicians who conducted such diligent field work. We thank the Minnesota Department of Natural Resources, especially employees at the Talcot Lake Wildlife Management Area. We also thank Sarah Thompson for her consultation throughout the study and thoughtful comments on an earlier version of this manuscript, and two anonymous reviewers for helpful comments that improved the manuscript. This work was partially funded by the Minnesota Environment and Natural Resources Trust Fund as well as a Legacy Grant from the National Fish and Wildlife Foundation (2009-0028-000).

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