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Land cover dynamics influence distribution of breeding birds in the Great Plains, USA

R Scholtz [a,*,1], JA Polo [a,1], SD Fuhlendorf [a,1], GD Duckworth [b]

[a] Department of Natural Resource Ecology and Management, Oklahoma State University, Stillwater, OK 74078, USA
[b] Centre for Statistics in Ecology, Environment and Conservation, Department of Statistics, University of Cape Town, Private Bag X3, Rondebosch 7701, South Africa

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ABSTRACT

Grasslands are one of the most endangered ecosystems globally. Large tracts of grassland in the Great Plains, USA have been converted to cropland since the early 1900s, and woodlands are rapidly encroaching into many grasslands of the region due to changes in land management practices. Changes in the arrangement and proportion of different land cover types can affect biodiversity. We used bird survey data to identify the effect of land cover change on breeding bird ranges within the Great Plains over a 10-year period. Each species was categorized into one of the following habitat guilds: grassland, shrubland, woodland, wetland and generalist. We calculated the proportion of each land cover (including cropland, grassland, woodland, developed, barren, water bodies and wetlands) within a 1.6 km radius of each bird survey starting point. Within an occupancy modeling framework, we estimated colonization and extinction rates for each species and averaged them to the guild level. We also quantified changes in land cover from grassland to other cover types. Results show that grasslands were mostly converted to cropland and woodland, which were accompanied by positive extinction rates for certain grassland species. Extinction rates at the guild level were unrelated to increases in any land cover type, and observed land cover changes largely favored the shrubland guild. Overall, habitat-guild responses show little influence of woodland or cropland expansion at regional scales, although certain species ranges are predicted to decrease with increases in woodland. Future research should consider a finer scaled approach focusing on species-level responses when short-term land cover changes are considered.

1. Introduction

Grasslands provide key services, especially in terms of food production, and key habitats for a range of biodiversity (Davis et al., 2016; Fuhlendorf et al., 2006; Howland et al., 2016) yet are currently one of the most threatened ecosystems globally (Fargione et al., 2009; Samson and Knopf, 1994; Vickery et al., 1995). Since the late 18th century, major land cover changes, such as grassland conversion to cropland and, elsewhere, woody plant encroachment, have occurred across large portions of the Great Plains in North America. These changes have contributed to habitat reduction for many species and subsequent biodiversity loss (McLeman et al., 2014; Samson et al., 2004; WWF, 2016).

Despite widespread and ongoing grassland loss (Clark et al., 2002; Raz-Yaseef et al., 2015; WWF, 2016), the Great Plains region still maintains some of the largest areas of natural grasslands, about 37%, in North America (Han et al., 2012; Homer et al., 2015). Natural processes such as fire, which gives rise to heterogeneity in the arrangement and structure of vegetation, are essential to maximizing biodiversity and maintaining integrity of grassland ecosystems (Davis et al., 2016; Fuhlendorf et al., 2006; Vickery et al., 1995). Achieving a balance between meeting requirements for human survival, e.g. increases in crop-production to meet food and export demands, and maintaining natural grasslands and their associated ecosystem services are generally governed by two main approaches: land sparing vs. land sharing (Fischer et al., 2014; Phalan et al., 2011). Both approaches have limitations and benefits, yet it is becoming increasingly challenging to generalize its implementation (Fischer et al., 2011).

Woodland encroachment is another major driver of land cover change within the Great Plains region that arises from anthropogenic alteration of fire regimes (Berg et al., 2015; Engle et al., 2008; Heisler et al., 2003; Twidwell et al., 2013). Woodland encroachment profoundly effects ecological processes ranging from hydrologic cycles (Zou et al., 2015) to fire dynamics (Fuhlendorf et al., 2006; Twidwell et al., 2013; Weir and Scasta, 2014). However, in contrast to cropland expansion, woodland encroachment in grasslands encompasses a more complex association with fire regimes. Specifically, changes in fuel type (e.g.

* Corresponding author.
E-mail address: r.scholtz@okstate.edu (R. Scholtz).
[1] Postal Address: 008D Ag Hall, Department of Natural Resource Ecology and Management. Oklahoma State University, Stillwater, OK 74078, USA.

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increases in coarse, woody biomass) influences fire severity and in turn fire regimes may be further influenced within this feedback loop (Keeley, 2009). As a consequence, ecological thresholds that change former grasslands into woodlands may be overcome by increasing fire activity (Briske et al., 2005; Fuhlendorf et al., 1996; Ratajczak et al., 2016).

Grassland birds throughout North America have been declining in recent decades (Sauer et al., 2014) and the primary cause of these declines is loss of habitat (Knopf and Samson, 1994; Samson and Knopf, 1994). Several arguments have been proposed for the mechanisms for grassland habitat loss such as fragmentation (Hobbs et al., 2008) and rangeland mismanagement (Briske et al., 2003) enabling debate for alternative management approaches (Bestelmeyer and Briske, 2012). Nevertheless, various drivers, such as changes in the fire regime and grazing, influence plant communities (Fuhlendorf and Engle, 2001). This leads to a reduction in diversity in grassland birds (Chapman et al., 2004a). Similarly, a reduction in grassland bird diversity has been associated with woodland expansion (Chapman et al., 2004b; Coppedge et al., 2001).

For this study, we used data from the North American Breeding Bird survey (BBS, Sauer et al., 2014) as an indicator for biodiversity within a highly utilized landscape. Birds are useful indicators of ecosystem health since they are mobile and respond quickly to land cover changes (Fuller, 2000; Gregory et al., 2003; Gregory et al., 2009) especially in grasslands (Coppedge et al., 2001). The influence of land-use (Duflot et al., 2015) configuration on breeding bird communities at regional scales has received substantial research in recent years (Bled et al., 2013; Duflot et al., 2015; Flather and Sauer, 1996; Gutzwiller et al., 2015). Since the Great Plains is a rapidly changing landscape with a substantial amount of grassland lost to croplands (WWF, 2016) and woodland encroachment, assessing changes in bird communities offers an important starting point for understanding regional biodiversity responses to land cover change. To this end, our study has two objectives:

(i) Quantify grassland loss and land cover conversion dynamics between 2006 and 2015
(ii) Quantify the effect of land cover change on breeding birds range dynamics (colonization and extinction) between 2006 and 2015 using the North America Bird Breeding Survey data (BBS, Sauer et al., 2014)

2. Materials and methods

2.1. Study area

The study area consisted of the states Oklahoma, Kansas, Nebraska, South Dakota and North Dakota within the central United States of America (Fig. 1) based on land cover data availability. Based on plant phenology, the vegetation growing season in this study area is generally from May through October and the dormant season from November through April. Natural vegetation cover in the region is comprised mainly of grasslands with interspersed stands of woody vegetation. Shortgrass prairie is more common in the west and tallgrass prairie in the east (Lauenroth et al., 1999). The climate ranges substantially throughout the region with extremely cold winters (mostly in the north) and hot and humid summers (mostly in the south). The southeastern part of the region received the most rainfall (mean annual precipitation (MAP) ~1600 mm) with drier areas to the west and north (MAP ~200 mm). Mean annual temperature ranges from ~3 °C in the north to ~24 °C in the south (PRISM Climate Group, 2004).

2.2. Data acquisition and preparation

2.2.1. Land cover information

Land cover data were sourced from cropscape.org (Han et al., 2012) for a 10 year study period (2006–2015) to estimate land cover dynamics. This period was selected because complete land cover data for our entire study area was only available from 2006 onwards. CropScape is an online tool that hosts Cropland Data Layer (CDL). The CDL is a georeferenced raster with a crop-specific land cover data layer created annually for the continental United States using the Advanced Wide Field Sensor (AWiFS) and Landsat TM 5 and ETM+ 7 with extensive agricultural ground truth employed (USDA National Agricultural Statistics Service Cropland Data Layer, 2016). According to Reitsma et al. (2016) who used South Dakota as a representative area for testing the accuracy of CropScape products, the overall accuracy for remotely sensed data relative to ground-truthed data was 84%. Furthermore, cropland producer accuracy (% of ground collected sites that were correctly identified) ranged from 89% in the east to 43% in the west. Grassland producer accuracy ranged from 95% in the northwest to 39% in the southeast in both 2006 and 2012 (see Liu et al., 2004 for more information).

All land cover pixels taken from CropScape were aggregated into one of the following broader categories; barren land, tree crops (e.g. apples), non-tree crops (e.g. wheat), grassland, evergreen and deciduous woodland which included shrubs (later lumped as "woodland"), developed, permanent water and wetlands. Aggregation was conducted for each year and scaled up to 3 km^2 resolution. Land cover categories in the original data were determined by the Natural Resources Conservation Service. We simplified the categories prior to our analysis of the data. Thereafter, each pixel was classified by majority land cover type per year. We chose this method and scale as we were interested in broad patterns of land cover and we acknowledge that we may lose a substantial amount of information at this scale.

To obtain landscape-level land cover information at each bird survey starting point, we calculated the total proportion of each land cover type per year within a 1.6 km (1-mile), 5 km, 10 km and 50 km radius at each coordinate associated with the BBS starting point and compared land cover proportions at each scale. We acknowledge that this approach may not account for the entire 40 km route. However, a) GPS co-ordinates for every stop are not required (e.g. might not be recorded by each observer) and b) given that the area covered by each route is not equal (e.g. all routes are not straight lines), we chose this approach to avoid any land cover sampling bias (Gutzwiller et al., 2015). Initial results suggest no major differences in the total proportion of grassland, cropland and woodland land cover types per year at either scale, and we opted to include the 1.6 km (1-mile) land cover data for our yearly covariate information (Appendix A Figs. A4–A6). This radius was chosen, as most sections bordered by a road in North America are 1-mile blocks and bird sightings are not recorded beyond 400 m from the road according to BBS protocol.

2.2.2. Bird detection/non-detection data

Bird data were sourced from the North American Bird Breeding Survey (BBS) database for 2006–2015 (Pardieck et al., 2014; Sauer et al., 2014). The BBS is based on annual counts of birds along pre-defined routes conducted by volunteers (Sauer et al., 2014). Each 40 km route consisted of 50 stops located every 0.8 km where all birds data seen and heard within a 3 min period are recorded (Sauer et al., 2014). A total of 180 routes were surveyed in our study area during the study period. We used all stop data from each route to designate whether or not each species was detected in each year (i.e., a detection/non-detection approach that we analyzed under an occupancy modeling framework, as described in detail below). Across the entire study area and study period, 314 total species were observed. However, we only included species for analysis if they were observed at least once at ≥40% of the points. We chose this threshold because we sought to only include relatively common breeding species and to exclude migrating and overwintering birds and other individuals and species that occasionally appear on surveys but are under-sampled using road-based point count surveys (e.g., waterfowl, raptors) (O'Connor et al., 2000). This reduced our sample size of species to 83. Each species was placed into one of five habitat guilds based on categories in Ehrlich et al. (1988) and The Cornell Lab of Ornithology (2016),