Looking Back to Look Ahead

Providing Temporal Context for the Spare or Share Debate Using Land Use Change and Bird Populations

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0. Abstract

The “spare or share” debate attempts to identify a solution that best mitigates biodiversity loss while simultaneously providing food for a growing population. In order to investigate how temporal changes conflict with the assumptions of this framework, we conducted a study that identified the changes in land use and bird populations in Rice County. As land use changes are deeply interconnected with social and political shifts, we conducted a historical analysis of the area to further understand the agents driving these changes. We tracked changes in land use by using manual classification of aerial photographs and satellite imagery from 1964 to present. Through this analysis, we observed increasing urban land, increasing forest cover, and decreasing land in agriculture in our study region. Visual analysis of the differences between these time periods revealed trends of increasing size of fields and reduction in crop diversity. Our historic research revealed societal trends such as the loss of small family dairy operations and the industrialization of agriculture as drivers of some of these changes. By comparing bird species with varying habitat preferences, we observed correlations between their population sizes and land use change in this area. The most significant correlation was the decline of grassland species that occurred alongside increasing agricultural industrialization and the loss of small family dairies. Overall, our study reveals how human land management decisions shape the assemblages of species that can live in these altered landscapes.

1. Introduction

The founding concern of the "spare or share" debate is the loss in biodiversity associated with the conversion of agricultural lands to feed a rapidly growing population (Green et al. 2005). "Land sparing" and "land sharing" are two opposing strategies proposed by this framework that balance this need for both conservation and production (Fischer et. al 2008). The "land sparing" strategy advocates for intensely farmed agricultural land, leaving more land to be set aside for the purpose of species conservation (Ibid). "Land sharing," also known as wildlife-friendly farming, is the opposite strategy where less land is set aside for conservation in tandem with less intensive and biodiversity-friendly agriculture practices on more land (Ibid). Many of the scholars engaged in the debate fear that high-biodiversity natural lands, such as the Amazon rainforest, will be converted to agriculture (Luskin et al. 2018). Through this debate, they aim to find a solution that will protect areas with high biodiversity while also maintaining a functional global agricultural system (Kremen and Merenlender 2018; Williams et al. 2018). However, there is little consideration within the "spare or share" framework for land that has been in non-static agricultural production for centuries. This gap in the debate is due to the fact that the framework views landscapes through a single snapshot in time, therefore ignoring their larger social and temporal contexts (Fischer et al. 2014).

North America, for example, experienced a major turning point in its history with the beginning of European settlement over four centuries ago. Prior to their arrival, much of the land was shaped by the lifestyles of indigenous peoples and their interactions with natural processes (Cole and Taylor 1995). In the Midwest, their practices of burning coupled with climatic conditions helped maintain wide expanses of grassland (Anderson 1983). Landscapes across the Americas changed dramatically as Europeans forced these inhabitants off the land. The fires that once maintained vast grasslands were quelled to make the landscape more hospitable to permanent agricultural fields and urban centers. Yet this new landscape was not stagnant; the fields changed from wheat to dairy and then to corn and grew ever larger. Corners of the land not in agriculture were overtaken by forest regrowth, newly unchecked by fire. These are merely examples of the many changes that have occurred in areas similar to the Midwest, but they continue to shape the present landscape. In the same way that current agricultural practices are contingent on past developments, present ecological relationships and communities are also contingent on these historical landscape changes (Lunt and Spooner 2005; Siebert and Belsky 2014). Therefore, understanding the history and development of these altered landscapes, so-called novel ecosystems, is vital in making locally relevant conservation decisions (Hobbs, Higgs, and Harris 2009; Mattison and Norris 2005).

In our study, we investigated how past patterns of human activities in the agricultural Midwest have shaped current land use practices and modern ecological assemblages. By incorporating history, ecology, and land use into this study, we revealed the significant, yet sometimes surprising, relationships between facets of human society and our local environment's non-human inhabitants (e.g. the relationship between milk prices and grassland bird species).

Through understanding the interconnectedness of human and ecological systems, we can make more informed decisions as to how different policies will affect people and the environment. Our findings contrast with the notion that conservation efforts should aim to restore an arbitrary biodiversity baseline and instead point to management that works within the current trends we see in our study region. We investigated the following questions in the context of Rice County:

- How do changes in land use patterns shape novel population trends in different species and reorder ecological assemblages? How do these results affect the assumptions of the "spare or share" debate?
- How does the history of land development and agricultural intensification correlate with population changes of different bird species? How might this be important for bird-oriented conservation and land management decisions moving forward?

Based on our results, we will argue that 1) political, economic, and technological changes drive the choices made by land managers, 2) landscapes and species populations shift in response to land use decisions, and 3) the biodiversity benefits predicted by the "spare or share" framework are not realized for all species without intentional human management. In Rice County, we see a changing assemblage of species following land use changes, such as growing field sizes. Due to agricultural intensification in the region, marginal land is being taken out of agriculture in a seemingly unintentional sparing scheme, however there doesn't seem to be any pointed management of this spared land for fostering biodiversity. Alongside these changes we noticed declines in grassland bird species populations likely as a result of the decrease in pastured land in Rice County. In contrast, forest edge species seem to be thriving in the new matrix of urban land and the aforementioned forest patches.

2. Literature Review

The "land sparing" versus "land sharing" debate originated in the 1990s but was brought back to the forefront of academic discussion when Green et al. published their seminal article in *Science* (Fischer et al. 2014). Fischer et al. (2014) traced the origin of the debate to work by Pimentel et al. (1992), who linked agricultural land use, population increases, and declining biodiversity. Pimentel et al. (1992) also proclaimed that agricultural systems and other systems of human management, such as forestry, are important for maintaining high biodiversity. Fischer et al. (2014) point out that around the same time, ecologists began to question how increasing agricultural intensity might affect ecological systems (see Kendall & Pimentel 1994 and Goklany 1998). However, these ideas gained far more attention twenty years later when Green et al. (2005) outlined the opposing models of "sparing" and "sharing," which provided the debate with new momentum. Within Green's proposed framework, sparing and sharing are two opposing strategies for maximizing agricultural commodity production and biodiversity. The sparing model argues for intensely efficient and productive agriculture, which allows for more land to be in conservation. Proponents of the other strategy, sharing, advocate that low-intensity and less productive agriculture should be practiced, but on more land overall. This strategy is also called "wildlife-friendly" agriculture, since supposedly less intensive agriculture can be simultaneously used as habitat by more species.

Since Green et al. (2005)'s initial proposal of the "spare or share" framework, the scholarly debate surrounding the topic has splintered into several directions. In a recent review, Phalan et al. (2018) found that sparing, overall, is the better strategy for preserving biodiversity as species that rely on niches that are not available in agricultural contexts cannot benefit from the share approach. Additionally, as Crespin and Simonetti (2019) point out, land sharing never considers human-wildlife conflict (e.g. between families with small children and wolves). Consequently, they argue that the sparing approach is the only viable option in maintaining species that cause conflict with humans when they inhabit the same landscape. Despite the arguments for the sparing approach, Grass et al. (2019) remind us that there are significant benefits, called environmental services, that natural lands provide for humans. Luskin et al. point out that sparing is the better approach for maximizing biodiversity, but sharing is a better approach if environmental services are being prioritized. Grass et al. (2019) argue there is a need for sharing to maintain environmental services, such as pollination, within areas where higher intensity agriculture is practiced. This sharing matrix is especially significant as it can help maintain natural corridors and allow wildlife to move between patches of spared lands (Kremen and Merenlender 2018).

Though a majority of studies have found sparing to be better for biodiversity and sharing for preserving environmental services, these findings cannot be used generally, as they have only been studied in specific landscapes. One issue researchers have noted is that findings of studies applying the "spare or share" framework to specific locations are often dependent on the context of that study (Egan and Mortensen 2012; Grass et al. 2019). When studies do include variables specific to their study area, such as the proportion of the landscape in agriculture, these factors often have significant effects on whether sparing or sharing is preferred (Law and Wilson 2015). Phalan et al. (2019) argues this lack of consideration for the context of study regions is a barrier to drawing conclusions regarding the efficacy of the "spare or share" model. Essentially, studies and reviews are finding that the context of regions are important considerations, yet context is not an explicit consideration within the framework originally set out by Green et al. (2005).

In an article published in *Conservation Letters*, Fischer et al. (2014) argue that as the model moves forward, explicit considerations of historical contexts and baselines are essential in improving the framework. Because historical factors determine current biodiversity levels, it is vital that studies investigate how present species assemblages came about in the first place (Newbold et al. 2015, McNeely 1994). Though the model is not designed to help understand political, social, and economic processes, these various processes influence agricultural activity and conservation decisions that are inherently intertwined with the framework. Hence, using the model alongside studies that engage these factors is crucial in order to understand how sparing or sharing might benefit a specific landscape or ecosystem (Phalan 2018).

Based on the issues within the "spare or share" framework pointed out by Fischer et al. (2014), we are interested in exploring what historical land use trends can tell us about an area's present-day species assemblages. Land use changes are not devoid of human action, yet the "spare or share" model does not account for these factors. We believe this lack of understanding for the human dimensions that shape land use choices is a significant obstruction in developing the context Grass et al. (2019) identified as being so crucial for the "spare or share" model. As Mattison and Norris (2005) point out, understanding these social interactions is essential for ecologists hoping to protect biodiversity. In our research, we drew from the methods of several fields to help us understand the role of historical events in shaping both land use change and species assemblages. In doing so, our research can help illustrate what the "spare or share" model is missing and how considerations of time and the effects of human choices have strong bearing on the conservation decisions available for a particular area.

2.1 Historical ecology

Historical ecology provides the framework necessary to relate the factor of time to ecological changes. In this framework, landscapes are recognized as complex, non-equilibrium systems that are interconnected with human activities, such as advances in agriculture and resulting changes in habitat (Szabó 2015). It acknowledges that historical changes can shape species assemblages by having irreversible effects on dynamics of previous ecosystems. These novel ecosystems, such as abandoned fields, are therefore a reflection of the influences from previous policies, agricultural trends, economies, and the process of industrialization (Ihse 1995), but can also be shaped by new species assemblages resulting from climate change and the introduction of non-native species (Hobbs, Higgs, and Harris 2009). Ecosystems that develop in these lands are driven by past land use patterns which have altered nutrient levels, reduced in the seeds of native plants, and introduced non-native species (Standish, Cramer, and Yates 2009).

Previous studies on bird species have found that their populations are significantly affected by historical land use changes, such as loss of forested land and conversion to agricultural fields. Some bird species benefit from historic, less intensive agricultural systems and their populations face declines following technological innovations that reduce their habitat (Chamberlain et al. 2000; Donald, Green, and Heath 2001; Wolff et al. 2001). For other species, the matrixes of habitats and the various resources (e.g. nesting sites, herbaceous vegetation capable of supporting insect populations, etc.) provided by less intensive agricultural systems are vital to supporting their populations (Brambilla et al. 2010). Although the correlation between agricultural intensification and declines in bird species populations has been established, it should be noted that the degradation of the winter habitats of migrating birds can also affect their populations (Wretenberg et al. 2006). Additionally, bird species richness[1] has been identified to be higher in parcels of land that have retained the same land use over multiple decades than areas of land that had experienced land use changes (Culbert et al. 2017). As a consequence, the specific habitat needs of bird species and how they are affected by land use changes is a vital consideration in understanding bird population trends.

[1] Species richness is defined as the number of different species present in a specified area.

2.2 Ecological Effects of Habitat Loss and Fragmentation

The proliferation of agriculture has led to overall habitat loss and fragmentation, both of which are significant factors in biodiversity loss (Fahrig 1997). The smaller the fragment, the fewer resources available, so there tends to be a negative relationship between fragment size and species diversity (Metzger et al. 2009). Some bird species, such as the Bobolink (*Dolichonyx oryzivorus*), strongly prefer fragments with large areas (Herkert 1994). The temporal factor further complicates this relationship, as current species populations do not always correlate to the present-day habitat availability. Species can be slow to adjust to changing habitat conditions, which delays the establishment of new equilibrium in population size (Bommarco et al. 2014). Certain species experience a temporal delay between landscape changes and population responses. This phenomenon, known as "extinction debt," is defined as the future extinction of species due to historical events (Chaudron et al. 2018; Tilman et al. 1994). The slow decline of vital ecological processes following habitat loss can cause bird species, such as the Brown Treecreeper (*Climacteris picumnus*), to experience extinction debt (Ford et al. 2009). Therefore, population losses from previous conversion to agriculture can continue for decades after the original loss of habitat.

The effects of historical land use choices on habitat size and fragmentation should be significant factors in land management decisions (Foster et al. 2003). For example, a study in Italy observed historical land use patterns to identify the best habitat types and traditional agricultural practices for the birds in question (Falcucci, Maiorano, and Boitani 2007). Their recommendation to revert to traditional agricultural systems in order to protect species was a novel conservation strategy, given that agriculture is dependent on human cooperation. This is a new way of thinking about conservation that is starting to gain ground in academic conservation biology (Higgs et al. 2014). As landscapes and species habitats have been shaped by humans for as long as humans have altered landscapes, our choices and their fates are intertwined.

2.3 Land Use Change

Various factors that influence land use change, including increasing mobility of capital and technological advances, are changing with increasing speed and result in unprecedented challenges (Ostrom et al. 1999). Many of these changes drive agricultural intensification, which is defined as increased inputs (e.g. inorganic pesticides) to agricultural systems resulting in increasing outputs. The increased outputs from agricultural intensification have been essential in providing food for a growing population, yet there is concern for their ecological impacts (Tilman 1999). Forest fragmentation, habitat loss, beneficial insect loss, and declines in bird populations have been linked to agricultural intensification (Bélanger and Grenier 2002; Benton et al. 2002; Tscharntke et al. 2005). Therefore, there is growing concern, as is reflected in the "spare or share" movement, that the continued growth of high-intensity agriculture will lead to decreases in biodiversity and other ecosystem services. Intensification can be triggered by the three interacting pathways of land scarcity, market opportunities, and interventions (Lambin et al. 2001). Generally, agriculture has intensified in the United States as a result of the adoption of increasingly agricultural technologies and production systems (Olmstead and Rhode 1993).

3. Methodologies

In our study we engaged three lines of inquiry to understand the relation between shifting land use practices and the response of the region's ecological species assemblages. To track how the availability of different habitat types has changed over time, we performed a geographic information system (GIS) analysis of aerial and satellite photography beginning in 1964. For our analysis of evolving ecological assemblages, we looked at the changes in populations of several bird species between 1964 to present. We choose to focus on bird species as they are charismatic, and many are decreasing in population in connection to agricultural changes (Canterbury et al. 2000; Culbert et al. 2017; Rosenberg et al. 2019). Such declines are expected to continue as agriculture continues to intensify (Green et al. 2005). In addition, we conducted historical research of Rice County as land use change is a symptom of changing relationships in human society. Together, these analyses allowed for a deeper understanding of how human decisions and actions shape our landscapes and the species that live there.

3.1 Study Site

Our study region, the northeastern portion of Rice County (Figure 1), is bounded by the county lines to the north and east, Interstate 35 to the west, and by Nerstrand Boulevard to the south. Despite focusing on a very particular region, studying this area can reveal larger trends, as Rice County shares a similar history with the rest of the agricultural Midwest. By uncovering the drivers of agricultural and ecological change in this region in greater depth, our study can increase our understanding of the larger landscape changes that have occurred throughout the Midwest.

Much of the land in this area is devoted to agriculture. In 2017, 85 percent of the farmland in Rice County was cropland and only 3 percent was pasture ("Census of Agriculture - 2017 Census Publications" n.d.). Like the rest of the Midwest, most of the cropland in Rice County is in two-year rotations of corn and soy. Despite the predominance of these cash crops, the South-Central Region of Minnesota has a longstanding tradition of small, family-owned dairy farms. There is a very particular demographic of farmers in this region. A majority of farmers are older white men working on their family farms (Ibid). Whereas the ratio of male to female farmers is relatively even, there are disproportionately high percentages of white farmers, older farmers and family farms. Around 20% of the farmers are considered to be "new and beginning," which points to the large number of older farmers and the difficulties younger farmers face in entering the market (Ibid).

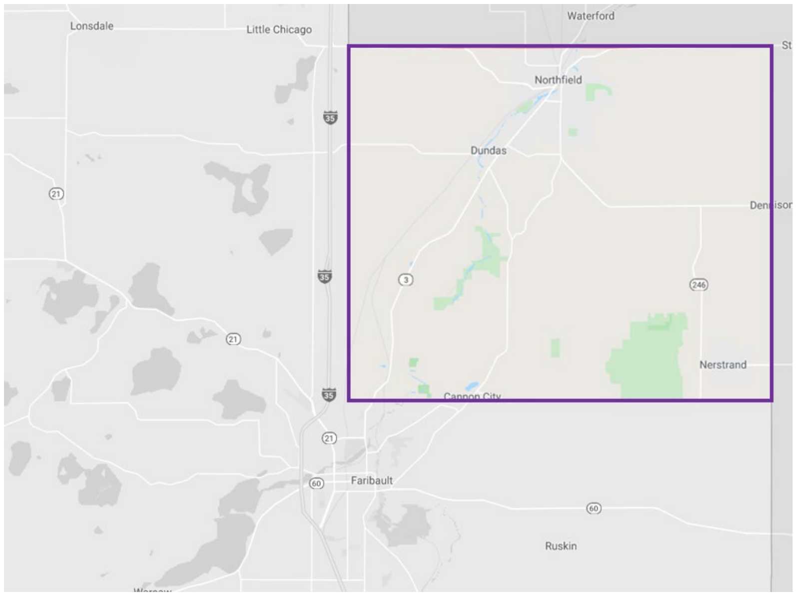

Figure 1: Our study site is highlighted by the purple box.

3.2 Land Use Change Analysis

One line of inquiry involved tracing the changes in the landscape over time. For this part of the project we used GIS to analyze how the amount of area in different types of land use, such as agricultural fields, developed lands, forests, and wetlands have adjusted over time. A more in-depth explanation of our methods can be found in the appendix.

As the technologies available for tracking changes in the landscape have changed drastically over our study period, we used several different sources. Due to the varying qualities and formats of these data, they needed to be processed in different ways. For the earliest time periods (1964 and 1970), photographs from aerial flyovers of the area were the highest quality source of information. We obtained these aerial photographs from the Minnesota Historical Aerial Photographs Online ("MHAPO" n.d.) database and the John R. Borchert Map Library at the University of Minnesota. These photographs came in the form of scanned copies of individual pictures that required stitching together. In this process, we smoothed out distortions and color differences between the individual images. For this data we only know the year the photos were taken, however judging from the tree cover and appearance of the fields these photos appear to have been taken in the summer or late spring. From 1991 to present, we used satellite imagery from Google Earth, specifically from April 1991, September 2003, March 2005, September 2009, August 2012, April 2015 and May 2017. To make these images usable in our analysis, we converted them into digital formats usable by GIS through georeferencing. For this process, we used landmarks such as road intersections and park boundaries to accurately pin down the images to their correct locations.

Once these images were georeferenced, we began to identify different habitat areas. In order to quantify changes in land use across our maps, we performed a random sample manual classification of each image. Classification is a technique where regions of a map are tagged as a particular category, or class. Manual classification means that each pixel being classified is examined by a person who codes that pixel as the category that best matches that region. For each image, we only classified a small sample of the pixels since classifying the entire image would have been too time consuming. We initially had intended to use supervised classification, a method in which AI is trained to identify the land use types across the entire map, however this method turned out to be too challenging for classifying black and white photography.

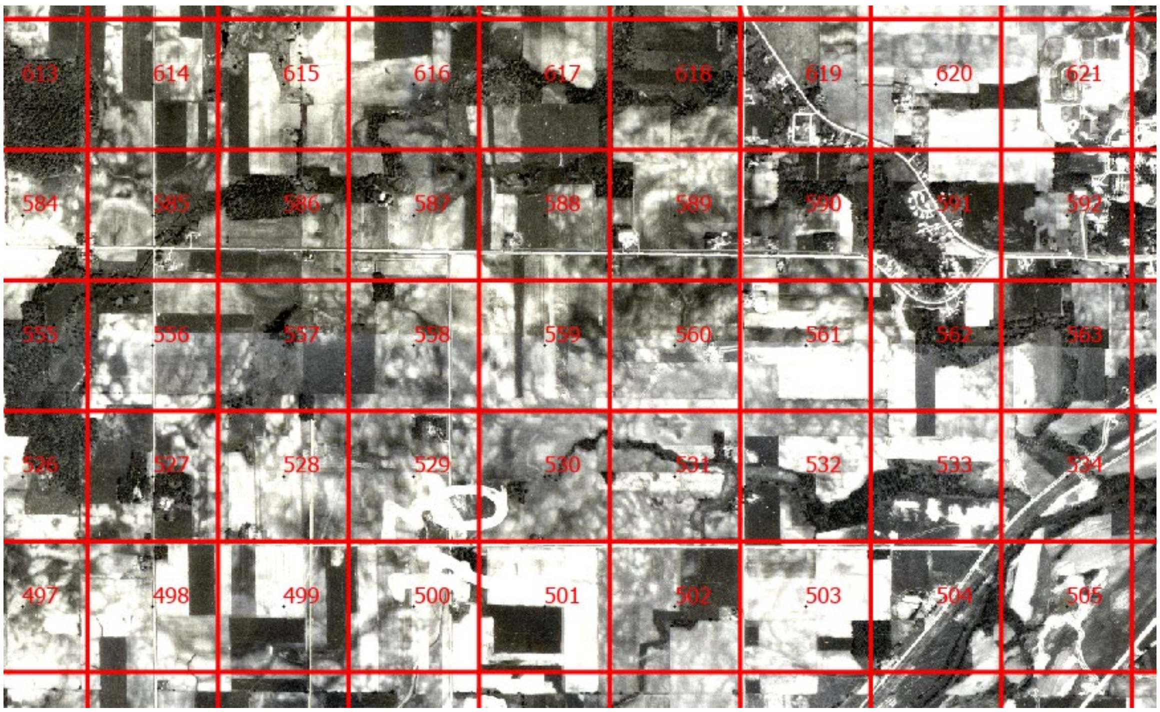

Figure 2 - Example of our grid system for manual classification over the 1970 image.

We classified 400 samples: 100 samples each from the maps dated 1964, 1970, 1991, and 2017. First, we divided the images into numbered grids, with each square in the grid measuring 1000 meters by 1000 meters (Figure 2). We then randomly generated one hundred numbers for each map and sampled the squares that corresponded to those random numbers. For each random square, we would assess which type of land use took up at least 50 percent of the square’s area and then coded the sample as this land use type. Our classes were: “forest”, “cropland,” “cropland with forest,” “wetland,” and “developed land.” We chose these land use types because they were easily identifiable from aerial imagery and corresponded to bird habitats used by similar studies (Rosenberg et al. 2019). The only exception to the 50% rule was the class “cropland with forest,” which was used when the random square had at least 10% tree cover. We created this category to account for smaller sections of forest that would not necessarily fulfill the 50% rule, but would likely provide sufficient habitat for certain bird species. After all of our samples were classified, we calculated the proportions of each land use type for every year. This enabled us to estimate the changes in land use proportions between the years 1964 and 2017.

In addition to this quantitative analysis, we also noted any qualitative changes visible from these images. These minute changes in the landscape, such as the increasing sizes of agricultural fields and a local trend of contour planting, would otherwise go unnoticed if we were to only quantify the change in land use. As we analyzed these images, we sorted these observations into different categories based on the type of change we were seeing. Additionally, we drew upon our research of the region's agricultural history to understand the changes we were seeing and provide context for these broader landscape changes.

3.3 Bird Population Data Analysis

Our next area of inquiry was investigating the trends in bird species. We investigated the trends in species using a couple of representative species of different ecological habitats (such as grassland, interior forest, and edge forest specialists) to capture how species with different breeding habitat preferences are changing in relation to land cover changes. Rosenberg et. al 2019 used similar delineations to study how the populations of bird species in different habitats are changing across North America. For each species, we first determined the changes in its population size.

There are several bird surveys in the region that we drew data from. Starting in 1968, data was available from the North American Breeding Bird Survey (BBS). The BBS began in response to concerns of the harmful effects of pesticides like DDT which were raised and popularized by Rachel Carson in her book *Silent Spring*. The highly standardized procedures of these yearly bird counts make them ideal for scientific studies. The yearly dates when the BBS takes place ensures that only breeding birds are captured in their counts. However, due to BBS observations occurring near roadways, they are biased in the types of birds they observe. As a result, these counts fail to capture interior forest species. Fortunately, we were also able to secure data taken inside Nerstrand State Park from 1990-2019, which allowed us to analyze the presence and prevalence of forest species in our area. Nerstrand appears to be the largest area of continuous forest in our area, excluding the narrower tracts of forest along the Cannon River. Consequently, it likely is not a perfect representation of the smaller forest fragments that we observe in the region, yet can still indicate whether different forest species are present in the area.

The Nerstrand State Park data comes from a Carleton class that recorded bird species in May almost every year since 1990. They approximately followed the BBS methodology; however, their route was shorter and went through the interior of Nerstrand Big Woods. One caveat of this data is that most of the students are not experienced birders and are only familiar with a small set of assigned bird species. Thus, it may not be as reliable as BBS data given that there are new students in the class every year.

Unfortunately, there are no BBS observation routes in Rice County. However, there are two stops located relatively nearby in Lakeville and Le Center. The Lakeville route has been surveyed 51 times since 1968 and Le Center has been surveyed 22 times from 1993 to 2016. Each of these surveys takes place from May 27 to July 7 in order to only record breeding birds and exclude any seasonal migrants. After analyzing bird species distribution maps from the

Minnesota Department of Natural Resources (DNR) for our study species, we have no reason to believe the populations of the birds we are interested in would vary significantly between our study area and these two routes. In our analysis, we assumed these observation routes could be used to represent the bird populations in the area.

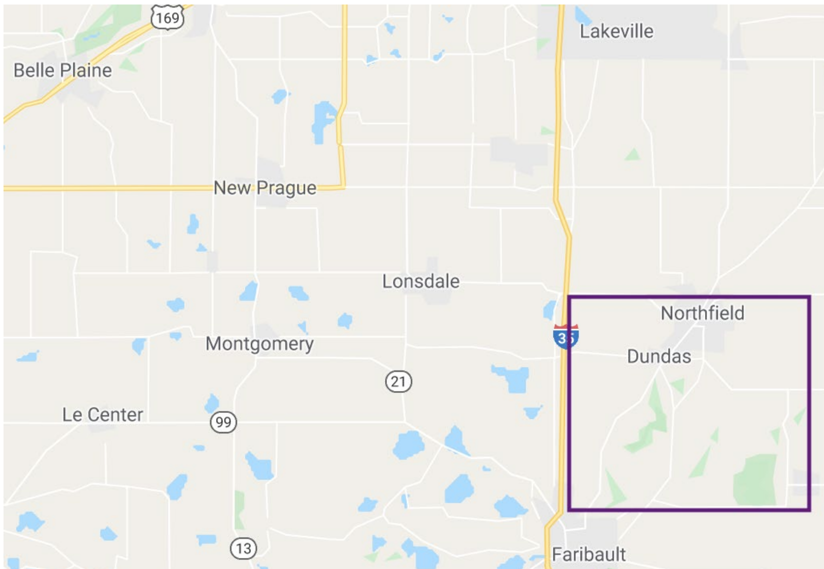

Figure 3: Locations of Lakeville and Le Center in relation to our study area.

3.4 Historical Analysis

Our last line of inquiry involved analyzing significant drivers in policy, farming practices, and economic factors that shaped agricultural land uses throughout our time frame. This analysis was not intended to be an exhaustive review of Rice County's agricultural history. Instead, we investigated a wide variety of sources to provide specific contexts for changes we observed in either bird populations or land use. To this end, we engaged a wide variety of primary sources including digital recordings of residents, newspapers such as *Northfield News* and *The Star Tribune*, agricultural data from the United States Department of Agriculture (USDA), and secondary sources stemming primarily from the work of historians and biologists. Our full list of sources is listed in detail in our bibliography. Together, this research helped establish connections between our land use maps, bird population data, and the broader story of human-environment relationships in Rice County.

4. Results

4.1 Land Use Analysis

The results of our analysis of the aerial photographs are discussed in several sections below. We begin by discussing the quantitative changes in land use determined by our manual classification. We then discuss these results in tandem with some qualitative changes we noted from comparing the maps. The trends we observed fall into roughly three categories: urban development, agricultural land use patterns, and distribution of forests and trees. For each of these categories, we also delve into the historical context of the changes we observed.

4.1.1 Manual Classification Results

From the results of our manual classification, we can begin to identify certain trends in land use between 1964 and 2017. Cropland with forest had the largest growth, jumping 14% over the course of our four maps. Forest and developed land also both grew a significant amount, increasing by 10% and 9% respectively. Cropland without any tree cover plummeted from 72% to 39%, which is the largest difference we saw across all types of land use. Lastly, wetlands remained relatively unchanged and stayed around 1% across all of the maps. Since these numbers are merely calculated from a small random sample from each map, they can only be used to approximate trends, not quantify changes. However, the trends we can see in our data line up quite closely with our historical research.

Land Use Category	1964 proportions	1970 proportions	1991 proportions	2017 proportions
Cropland	0.72	0.71	0.59	0.39
Cropland with forest	0.22	0.17	0.27	0.36
Developed land	0.02	0.06	0.09	0.11
Forest	0.03	0.04	0.04	0.13
Wetlands	0.01	0.02	0.01	0.01

Table 1: Summary of the results of our manual classification

4.1.2 Urban Development

One notable trend that occurred between 1964 to present is the extent of urban development in the area. Northfield has significantly increased in size during this time, as seen in Figure 4, below. Much of the development in Northfield appears to be residential, whereas both industrial and residential development in Faribault appears to have increased. Land that had previously been used for cultivating agriculture was converted into residential areas in order to facilitate this growth. According to our manual classification results, most of the urban development occurred from the 1970s onwards, with the fastest period of growth between 1970 and 1991. Beyond the expansion of city limits, there also seems to be a proliferation of pockets of neighborhoods and non-farm houses in rural areas (Figure 5). These types of neighborhoods are developing in both open country and forested areas. A smaller trend during this time is the disappearance of several roads in the country. Whereas some of the more traveled roads, such as

I-52, have become larger and expanded their lanes, several roads through the countryside have disappeared altogether across our time period.

Figure 4: Expansion of Northfield from 1964 (left) to 2017 (right)

Between 1940 and 1970, studies of agricultural southwestern counties in Minnesota reported a mass migration of primarily young adults from farms to larger towns and cities. This exodus left mainly poorer and older residents in the agricultural regions (Nass 2015, 148). In Minnesota, around fifty percent of the state's population lived in small towns or rural areas in 1940, but this number dropped by thirty-four percent over the next thirty years (Nass 2015, 149). This trend could be seen across the nation as well: in 1940, roughly forty-four percent of Americans lived on farms or in small towns and by 1970 that figure had dropped to twenty-seven percent. This phenomenon helps to explain the growing development we see occurring in Northfield across our time period, as young people moved away from their family farms and into larger towns.

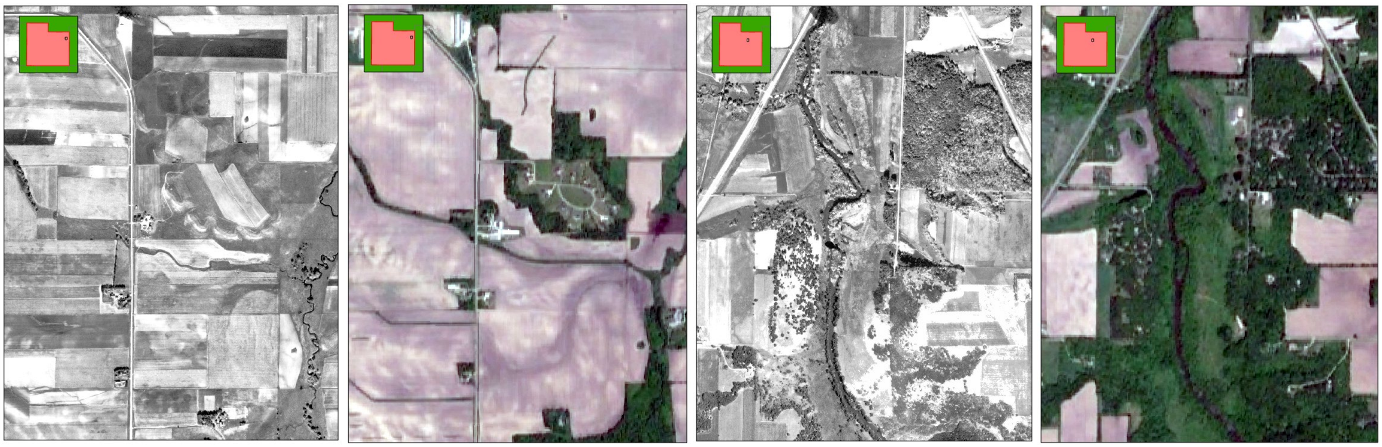

Figure 5: Image showing the development of neighborhoods in non-urban areas of the study region. From left to right, images one and two go together and images three and four go together. The leftmost image in each pair is from 1964 and the rightmost is from 2017.

4.1.3 Agricultural Innovations

Perhaps the most visible trends over this time period are the agricultural changes taking place. The increase in field size is exceptionally noticeable in our study area and can be seen with just a quick glance over the maps. In our map from 1964, the plots of land are highly subdivided, and many fields appear to be separated into strips of different types of crops. In contrast, the fields in 2017 appear to be larger and growing a single kind of crop, which suggest that there were a larger variety of crops being grown in the area in the 1960s. The shift from highly subdivided fields to larger continuous fields appears to correlate with a loss of trees and fence lines, which perhaps used to provide habitat for various bird species. In contrast, there are areas that appear to have previously been in cultivation, but have since transitioned to other types of land cover such as wetlands and forests. Another trend appears mostly in the southeast corner of our study area, where there is a significant amount of contour farming. Since there is not much contour planting in the rest of our study area, this seems to be a local trend. Closer examination of more recent images reveals that there might still be contour plowing occurring in this region, or perhaps terracing.

Figure 6: Increase in agricultural field size between 1964 (left) and 2017 (right)

Across our aerial maps, we can see that field size grew noticeably since 1964 and historical context supports this observation. Minnesotan farmers witnessed these increases in sizes of farms as they became highly mechanized and increasingly specialized (Nass 2015, 148). Especially after World War II, farming shifted from a labor to a capital-intensive industry (Nass 2015, 147). This change meant farms were no longer constrained by the amount of human labor a single family with livestock could put into the land. New developments shifted this burden from humans to machines and chemicals. For example, although farms had been moving towards tractor use since the late 1920s, by 1974, there were on average 2.6 tractors and no work horses on Minnesotan farms (Nass 2015, 141). According to the Minnesota Department of Transportation, from 1933 to 1970 the labor inputs on farms decreased by 70% while "chemical inputs increased by 1,800 percent and purchased feed, seed, and livestock rose 270%" (Granger and Kelly 2005, 125). Although technology allowed farms to grow, those who were not able to scale up were pushed out of the industry. Between 1950 and 1970, the number of farms in the Midwest decreased by half and then leveled off. These two decades saw the largest numbers of farms being consolidated or sold than any other point in history (Ganzel 2007). On a more local level, the number of Minnesotan farms between 1950 and 1974 dropped from 179,000 to 99,000 and average size increased from 184 to 280 acres (Nass 2015, 147).

4.1.4 Changes in Dairy Farming

One trend that we hoped to see in the older maps was the loss of dairy pastures. The expansion of dairies in Minnesota began in the late 19th century after a forty-year long obsession with wheat resulted in economic pains as larger operations further west drove prices down (Granger and Kelly 2005, 64). Incorporating livestock into farm systems was beneficial in two ways. For one, the profits from butter, cream, and meat diversified and supplemented farm income. Secondly, the companion crops in this mixed system, including alfalfa, oats, and barley, further diversified income so farmers were no longer reliant on one crop (Granger and Kelly 2005, 96). Rice County farms moved away from commercial wheat production and potatoes to include beef cattle, hogs, dairy, sheep, chickens, barley, oats, and flax in their rotations (Wanless 1985, 277-286). As the century progressed, the flat prairie regions in the Red River Valley increased in size as they transitioned to cash crops such as corn and soybeans. Unlike the Red River Valley, farms in southern Minnesota remained smaller and more diversified as the region's rolling hills were less suitable for row crops (Granger and Kelly, 111). Since the 1950s, Minnesotan farmers in most regions of the state have gradually specialized in either livestock or row-crop systems rather than previous mixed crop, livestock and dairy operations (Granger and Kelly 2005, 123-136). However, Rice County did not start to truly experience losses in dairy until the 1960s, when they began to mirror trends occurring across the state (Figure 7).

Due to the loss of small dairy operations, the pastures that were once maintained for these farms are also disappearing. This trend is often listed as a reason for the declines in many grassland bird populations, as the prairies and other prior habitats of these species have nearly disappeared (Brambilla et al. 2010; Jobin, DesGranges, and Boutin 1996; Stanton, Morrissey, and Clark 2018). Unfortunately, we were unable to distinguish between pasture and cropland because the older images were taken in black and white. However, we can draw on data from additional sources to supplement our aerial imagery. Data from the United States Department of Agriculture's (USDA) Economic Research Service indicates a downward trend in pastureland in Minnesota from 1945 to 1987, and then a leveling off until 2002 (Bigelow and Borchester 2017). The trend in Rice County follows this downward march with pastureland declining by 33% from 1995 to 2017 ("USDA/NASS QuickStats Ad-Hoc Query Tool" n.d.).

These losses in dairy farms can also help explain the increasing field size we noted in our region. Family dairy farms tend to be small, around 80 to 120 acres (Hart and Ziegler 2008, 141). With industrialization, dairy operations also needed to increase in size to remain viable. As a result, many of these small farms across the state followed the trend of "get big or get out." Rice County and its surrounding region are unusual in this process, as the transition to larger dairy operations took much longer here than in the rest of the state (Borchert et al. 1969, 47). Until approximately 1970, Minnesota and California shared equal portions of dairy sales. Unfortunately, due to increasing production scales in California, Minnesota's shares have been dropping. In Minnesota, the average milking herd increased from ten in 1950 to seventy-four in 2005. However, in California this increase was eightfold and jumped from an average of 16 cows in 1950 to 559 in 2002 (Hart and Ziegler 2008, 151). Increasing scales of production in California and elsewhere have pushed prices down by increasing the supply of milk in the market. Declining demand for milk, domestically and abroad, also contributes to low milk prices. Additionally, the market price for milk has been more volatile since the 1980s, due to changing federal policies regarding price supports (Morse 2001). These low prices have been slowly driving smaller dairies out of the market and the landscape. Filling their place are larger contiguous fields planted end to end with rows of corn and soy cash crops.

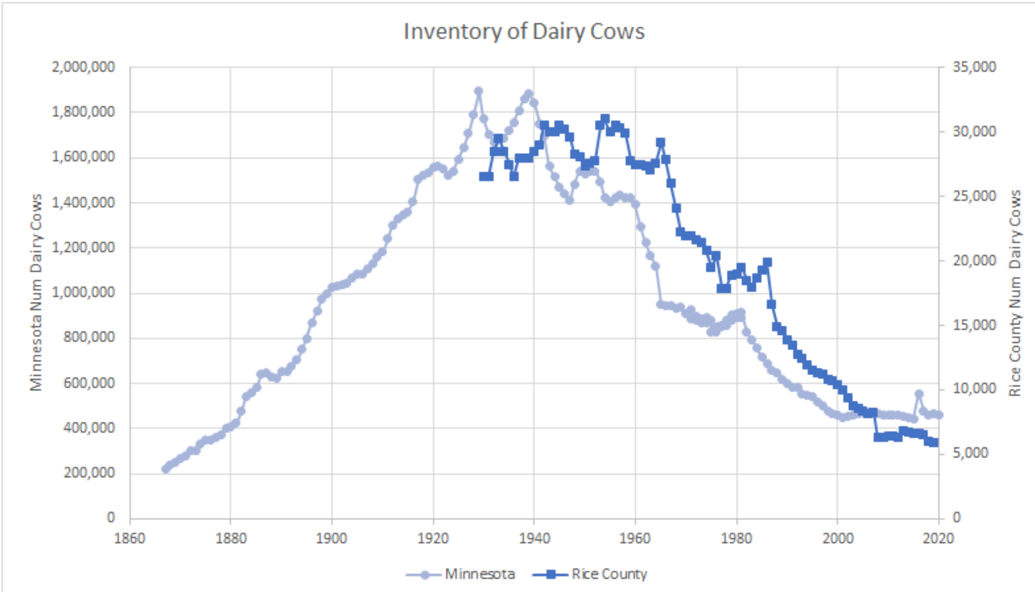

Figure 7 - Dairy operations in Rice County do not appear to have taken a hit until the 1960s, whereas declines statewide began around 1940 (Data from USDA Quick Stats)

In 2018, one tenth of dairy farms closed in Minnesota but production of milk rose by two percent. This points to the larger structural change that is taking place nationwide as smaller family dairy farms are closing down while larger, modernized operations dominate (Belz 2019). This is because it costs approximately $16.28 for a dairy farm with fewer than 200 cows to produce a hundredweight of milk (around 112 pounds), whereas dairy operations with over 1,000 cows can produce the same amount of milk for two dollars cheaper (Belz 2019). These larger farms are necessary for dairy farmers to remain viable in modern markets (Har and Ziegler 2008, 151-154), just as row-crop farmers have needed to increase the size of their fields. This trend is visible in Rice County as the number of larger farms (200 to 499 cows) increasing from 1997 to 2017 while smaller farms (with less than 200 cows) have decreased in number (Figure 8).

Despite efforts by various people to increase the number of larger dairy farms, these new larger operations would not bring back landscapes of yore. Modern industrial scale dairy farms have largely foregone pasturing their animals, and instead keep them in confined barns (MacDonald et al. 2007). The changes brought about by these larger dairies would have strikingly different effects upon the agricultural landscape than the previous smaller farms with pastures.

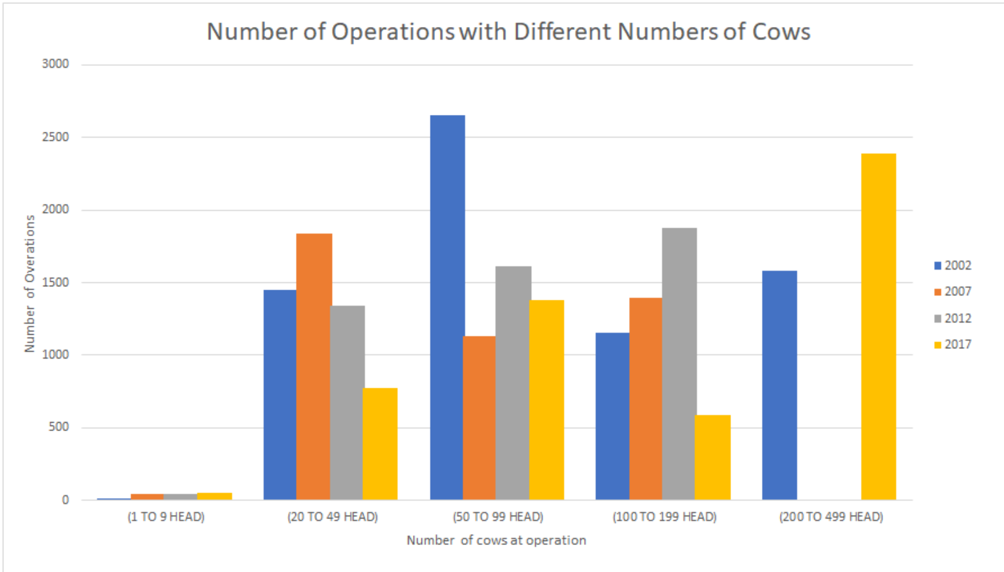

Figure 8 - Summary of the number of operations in the United States with amounts of dairy cows in their operations. Note there was no data from 2007 and 2012 about operations with 200-499 cows (Data from USDA QuickStats)

Yet some small dairy farms continue to exist. One of the main ways that smaller dairy operations are surviving is by producing class III milk, which is used to make cheese (Belz 2019). Prices for class III milk have risen by four dollars per hundredweight in the past year and are predicted to continue to rise (Belz 2019). However, this solution will not be enough to keep all of the current small dairy operations in business. The Family Dairy Farm Relief Act, a solution proposed by the National Farmers Organization, advocates for federal price supports based on costs of production. This act would give small dairies more money for each hundredweight and provide no funding for larger dairy operations. However, no such legislation has been officially written or introduced. The USDA predicts that there will only be 18,000 dairies left in the country by 2026, hinting that the move towards increased industrialization and greater efficiency in the dairy industry appears to be unavoidable, leaving smaller family farms in an increasingly vulnerable position (Belz 2019). Yet in this state there is a collective desire to maintain these traditional small farms. To this day, many older farmers in Rice County continue their dairy operations alongside their modern row-cropping enterprises (Wanless 1985, 277-286). There are hundreds of news articles decrying the fate of these small farmers throughout our study period and into the present. A public not yet tired of this story and farmers who uphold these practices despite not seeing returns indicates a long-term concern and interest in maintaining these vestiges of the Midwest's farming traditions.

4.1.5 Conservation and Forest Cover

The trends in abundance and spatial distribution of tree cover are highly varied across this region. It is immediately apparent, even without the quantitative analysis, that there is substantially more land covered with trees today than in 1964. There are many areas that were sparsely covered in trees in this first aerial photograph that are now filled in with forest approximately 60 years later (Figure 9). The majority of the increase in dense forest cover occurred between 1991 and 2017, as we saw a nine percent jump during this period in our manual classification results. The number of smaller forest patches alongside agricultural fields also increased dramatically, as our land use category of "cropland with forest" rose ten percent between 1970 and 1991, and then increased another nine percent between 1991 and 2017. Areas alongside railroads and rivers are now also more abundantly forested. There do not appear to be any areas where forest cover has significantly decreased in this time period, with the exception of urban areas for residential development.

Figure 9: Increase in forest cover between 1964 (left) and 2017 (right)

As evidenced by the photographs, one of the largest consistently wooded areas in this area has been Nerstrand Big Woods State Park. Big Woods, unlike the rest of Rice County's former 3,000 square mile forest, was able to survive settlement thanks to the land use decisions of settlers in the 1850s (Wilder 2013). Early settlers in the region divided the area that is now the state park into small woodlots ranging in size between two to thirty acres. These 170 families refused to sell the land for timber, as they saw these woodlots as a source for building supplies and firewood (Wilder 2013). In the 1930s, plans to clear-cut the entire area launched a series of protests from several organizations. After several failed attempts to have the state formally protect the area, the park was successfully established in 1945 (Sauve 2020, Wilder 2013).

Uncovering possible reasons for these increases in forest cover has proved difficult and might require the work of a future research project. Interviews with local farmers reveal that they are unaware of this change, so although this trend is obvious from our analysis of aerial photographs, it appears to be overlooked by researchers and locals alike. A study looking at land cover changes in Ohio observed similar trends in changing forest cover in this time period. They looked at the turnover of land parcels and found that parcels with low turnover rates had the largest increases in forest cover. In examining this trend more closely, they hypothesized this might be because the landowners with a longer history on the land were more likely to let areas previously in forest continue to grow, whereas newer landowners, more concerned with turning a profit, were more likely to convert forested land (Medley et al. 2008). Perhaps this has something to do with the trends we are viewing in our region, however there is a lack of well-developed analysis or information for these local trends.

The implementation of buffer laws might also help to explain some of the forest growth we can see in our maps. Buffers, or riparian filter strips, are barriers made of grass, brush or trees that slow large rushes of water flowing over land during high precipitation events (Marcotty 2015). The usage of these strips helps to prevent soil from eroding into streams, lakes, and wetlands. Buffers also help mitigate any nitrogen or phosphorus runoff from fertilizer, a large source of water pollution, from entering bodies of water. Though buffer strips are intended to support soil conservation and improve water quality, they also are beneficial to wildlife. Wooded buffers support the movement of birds across landscape by acting as a habitat corridor (Machtans et al. 1996). Minnesota enacted its first buffer strip law in 1977, which required uncultivated one-rod buffer strips on either side of ditches (Meador 2015). Lakes and streams were required to have fifty-foot protective buffers, but these laws were rarely enforced and participation was low. A second buffer strip law was passed in 2015, which mandated fifty-foot buffers along lakes, rivers, and streams. This law was later amended to require sixteen-foot buffers along ditches ("Minnesota Buffer Law | MN Board of Water, Soil Resources" n.d.). As of July 2018, approximately ninety-eight percent of agricultural fields on the edges of water comply this law (Ibid). We can see evidence of the implementation of buffer strips across our maps, as land along the edges of rivers becomes increasingly heavily forested.

There are a variety of other explanations for this increase in forest cover as well. The emergence of these small forest patches might be related to increased participation in conservation programs, such as the USDA's conservation reserve program (CRP), which pays farmers to convert their fields into forest and grasslands ("Conservation Reserve Program" n.d.). The enrollment in this program peaked in Minnesota in 2007, but as demand for soybeans and corn has risen, over 1,200 square miles of conserved land has been put back into agricultural use (Stanley 2019). Enrollment in the CRP typically fluctuates with the prices of crops, and therefore CRP lands might not be the best explanation for the steady increase in forest patches. Rising participation in conservation easement programs may also explain this trend, since easements, once established, are kept out of agricultural production forever ("What Are Conservation Easements? | MN Board of Water, Soil Resources" n.d.). Land put into easements can be converted into a variety of land uses, such as wetlands, forest, or native grasslands ("What Are Conservation Easements? | MN Board of Water, Soil Resources" n.d.). A simpler explanation could also be that these patches are hilly areas that are too challenging to farm and have therefore been left unattended. Advancing agricultural technology, such as crop productivity heat maps, has made it easy for farmers to detect unproductive areas and set them aside.

Despite these increases in forest cover, it is possible that all this new growth might not be beneficial to native systems. Invasive species have the potential to alter forest regrowth and species composition through competition with native species (Flory and Clay 2010). Though we cannot track this invasion through our imagery, others have found spatial patterns in invasion dynamics. Forest sites that are near residential developments, or that have experienced high levels of disturbance (Moser et al. 2016) and fragmentation (Gavier-Pizarro 2010) are at higher risk than others for invasion. In Minnesota, there are several specific invading species of concern for forest sites. Common Buckthorn (*Rhamnus cathartica*) is an especially efficient invader which forms monoculture thickets in the understories of disturbed plots. Buckthorn affects ecosystem processes in several ways, yet it remains unclear the direction and magnitude of its effects on native species given a lack of causal evidence of improvements in removal studies or studies documenting significant effects on native species (Knight et al. 2007). Though Knight et al. 2007 did not find evidence linking Buckthorn to native species declines in their review, they believe that it could have the strongest effects on native plants through resource competition and allelopathy. Despite ardent beliefs of many land managers of the danger of Buckthorn on native species, the scientific community has yet to document native declines as a direct result of Buckthorn. These studies suggest we should not be as concerned with maintaining previous 'natural' ecosystems as we witness the development of novel ecosystems as a result of anthropogenic disturbance. These invaders are adept at colonizing disturbed and abandoned plots of lands left in the wake of human land use changes and might serve as habitat for new assemblages of species.

4.2 Bird Trend Analysis

For our analysis of bird populations, we focused on several bird species that have different land cover preferences for habitat. All of the species we selected are regular residents of this area.

4.2.1 Grassland Species

To analyze grassland species, we examined the population trends of Western Meadowlarks (*Sturnella neglecta*) and Killdeers (*Charadrius vociferus*). Both species prefer different types of open grassy areas. Western Meadowlarks are found in open grasslands, prairies, meadows, and some agricultural fields and avoid shrub and tree-covered land (Cornell Lab of Ornithology). These birds also inhabit grassed waterways, field edges, and occasionally no-till fields of wheat or sod-sown soy and corn (Dechant et al. 1999). Killdeer are shorebirds that prefer areas with shorter grassy vegetation including sandbars, mudflats, and grazed fields, but also in developed contexts including lawns, driveways, athletic fields, parking lots, airports, and golf courses (Cornell Lab of Ornithology). Due to these distinct habitat preferences, we would expect these species to react to landscape changes differently. The Western Meadowlark is highly dependent on undisturbed open grassland and is not as flexible in nesting in areas inhabited by people as Killdeer, but both of these species occupy pasture and agricultural fields.

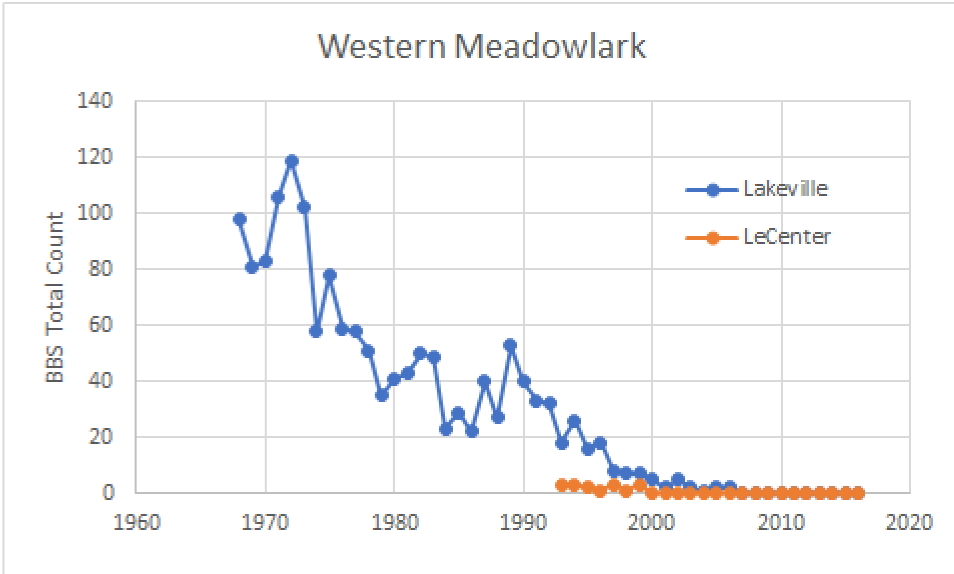

Figure 10 - BBS Counts for Western Meadowlark from Lakeville and LeCenter. This species was not observed in Nerstrand.

Our analysis found that both of these species are declining in our study region, as the BBS once recorded hundreds of Western Meadowlarks in the area (Figure 10). However, from 1996 onwards, less than ten of these birds have been observed in this yearly bird count. As previously discussed, the majority of our study site was once prairie, but it was quickly converted to agriculture beginning in the 1850s. The remaining habitat for these species was in the relatively undisturbed fields used for pasture and agricultural fields (Dechant et al. 1999). Of the 18 million acres of prairie first recorded by the initial land surveys of Minnesota from 1847-1908, less than one percent remained as of the 1980s (Horton 2010). The prairie that remains in the western part of the state continues to be converted to cropland or is being subsumed by woody vegetation due to lack of management (Lark et al. 2019). This is a trend we also witnessed in the increasing forest cover of our region.

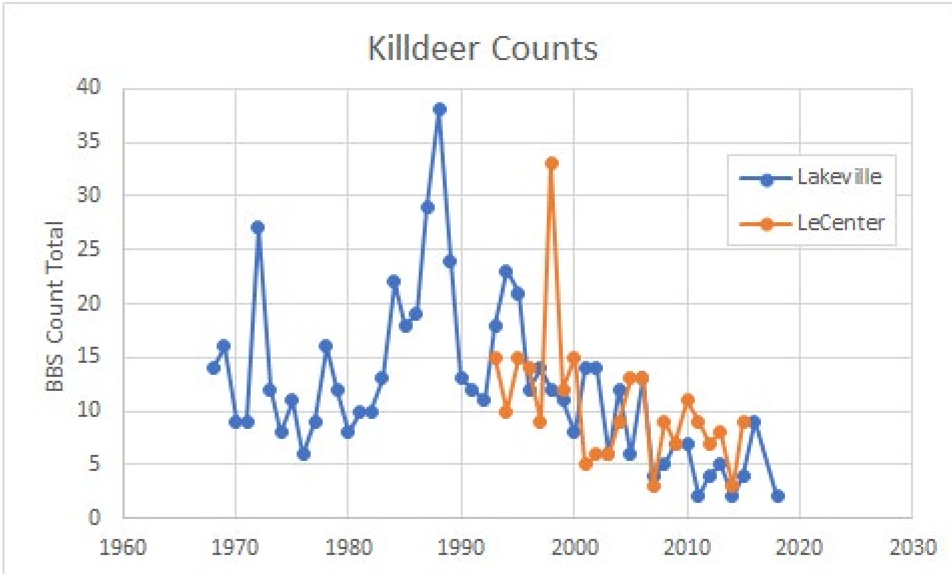

Figure 11 - BBS Counts for Killdeer from Lakeville and LeCenter. This species was not observed in Nerstrand,

Similarly, populations of Killdeer are also decreasing (Figure 11). From 1995 to 2019, there is a definitive decrease in the number of Killdeer recorded by the BBS surveys. Unlike the rest of North America where a consistent downward trend in their population resulted in an overall 46% decline from 1966 and 2014 (Cornell Lab of Ornithology), there appears to be a brief period of recovery in our region from 1980 to 1995. Despite the ability of these birds to reside in urbanized areas, we are seeing a decline in their numbers as residential development increases. One potential explanation of this trend is that Killdeer nest and feed in cultivated fields (Graber and Graber, 1963). These fields are often sprayed with herbicides and pesticides that can affect insectivorous bird species inhabiting these areas, which might explain the decline in these species and perhaps others that live near agricultural fields.

Both of these species are dependent upon grassland habitat, but the availability of alternative habitats (roadside ditches, field buffers, and the fields themselves) illustrate that loss of habitat is only one determinant in these species' decline. Instead, the increasing industrialization of agriculture seems to be highly pernicious for these grassland birds. This is especially true given that the alternative habitats of these species are situated either directly in cropland or adjacent to it - both of which are increasingly disturbed. There are two trends in agriculture that are highly influential to bird species populations: the widespread introduction and adoption of chemical pesticides and increasing disturbance of fields through mechanization.

4.2.2 Effects of Pesticides on Grassland Species

Throughout the 20th century, a changing mixture of pesticides had been widely adopted by American farmers due to changing government regulations and innovations by the chemical industry. Dichloro-diphenyl-trichloroethane (DDT), developed during the 1940s, was the first modern insecticide (US EPA 2014). Originally made to protect soldiers from malaria and other diseases carried by insects, it was later used in agriculture, home gardens, and livestock production (Ibid). Mounting evidence of its harmful effects on animals and the environment as well as the public concern raised by *Silent Spring* led the USDA to start regulating the pesticide beginning in the late 1950s (Ibid). In 1970, the Environmental Protection Agency (EPA) was formed and by 1972 the agency issued a cancellation order for DDT. This pesticide had adverse effects on bird populations, as it poisoned their nervous systems and altered their calcium metabolisms, resulting in eggshells that were too thin to support the incubating chicks (Ibid).

Although DDT was banned decades ago, current pesticides continue to have harmful effects on birds. Organochlorine insecticides, which are chemically similar to DDT, as well as neonicotinoid pesticides, are both popular among farmers and have been proven to have adverse effects on biodiversity (Bittel 2019). Birds near farmland are especially prone to these negative effects, as these chemicals can both kill the birds directly, or kill off the insects that make up their food supplies (Cox). Studies have shown that grassland bird species populations have declined far more than can be explained by habitat loss, and the use of toxic insecticides provides a much more plausible explanation (Mineau 2013).

Fortunately, there have been efforts to help protect these grassland birds. The Minnesota DNR launched a campaign in 1965 to manage roadside habitats to create a “permanent, well-distributed wildlife habitat” across the state (Nelson, n.d.). They are urging managers of these structures to plant native prairie and manage them in ways that are beneficial to grassland species whose native prairie is all but gone.

4.2.3 Forest Edge Species

To study the effects of changing amounts of open forest and forest edge, we chose to look closely at the populations of the Black-Capped Chickadee (*Poecile atricapillus*) and the Northern Cardinal (*Cardinalis cardinalis*). The Northern Cardinal’s habitat includes overgrown fields, backyards, regrowing forest, ornamental landscaping, and forest edges (Cornell Lab of Ornithology). Luckily, this means they do very well with urban sprawl and can coexist happily with humans. They do not appear to be a conservation concern, as their population in the United States has increased between 1966 and 2014 (Figure 12). We can see this trend in our local data from Lakeville as well, as they were spotted in numbers between zero and five until 1990, but were seen around 15 times per count over the last two decades. This makes sense since we can see from our aerial photos that backyard habitat has increased with the expansion of Northfield. Additionally, the trend in forest regrowth we witnessed provides them with increasing habitat outside of developed areas as well.

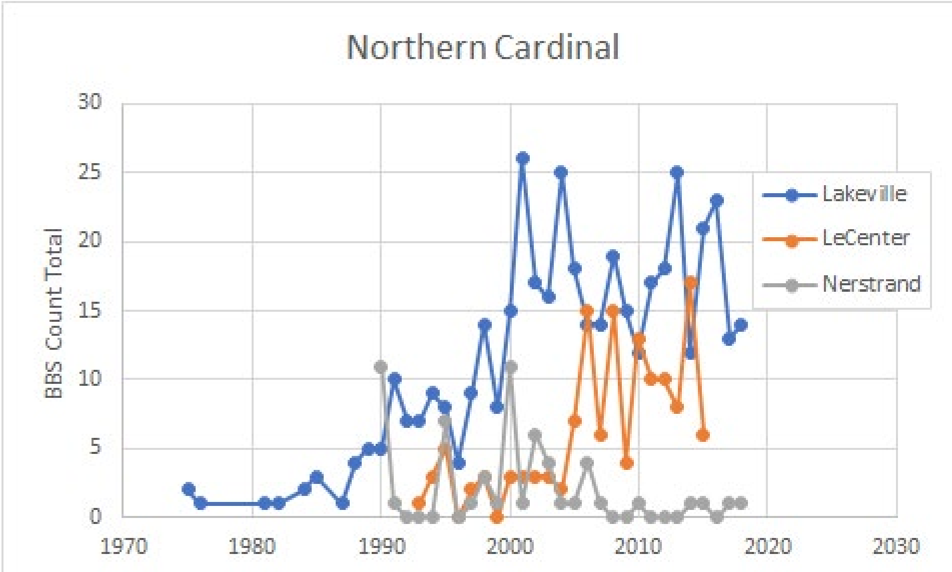

Figure 12 - Counts for Northern Cardinal from Lakeville, LeCenter, and Nerstrand

The habitat of the Black-Capped Chickadee is slightly different, as they prefer deciduous and mixed forest, parks, cottonwood groves and parkland (Ibid). These birds are also of low conservation priority, as the BBS has noted that their population has grown between 1966 and the present (Figure 13). In terms of their habitat being affected by agriculture, the clearing of forests for fields benefits them, as it creates a larger amount of forest edge. They also enjoy frequenting bird feeders, and therefore have seen a population growth due to the simultaneous expansion of towns, growing field size, and increasing coverage of patchy forests.

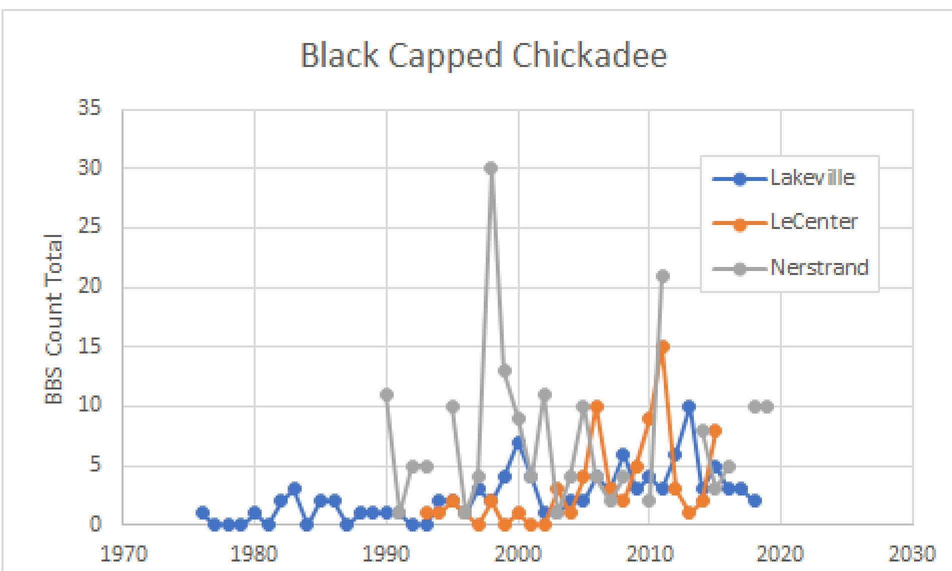

Figure 13 - Counts for Black-Capped Chickadee from Lakeville, LeCenter, and Nerstrand

4.2.4 Interior Forest Species

Interior forest species, on the other hand, have not been so well-suited to the land use changes in our area. For instance, Red-Headed Woodpeckers (*Melanerpes erythrocephalus*) have become a conservation concern over the past few decades. Their habitat consists of deciduous woodlands, groves of dead trees, parks, farmland, forest edges and roadsides (Ibid). Although they spend most of their time in forest interiors, they move over to forest edges while breeding. They prefer dead trees to build their nests and are somewhat nomadic, moving across locations between years. Between 1966 and 2014, the Red-Headed Woodpecker's population has declined by 2 percent every year, adding up to over a 70 percent drop over the last four decades ("BBS - USGS Patuxent Wildlife Research Center" n.d.). They are listed at 13 out of 20 on the Continental Concern Score and are present on the 2014 State of the Birds Watch List, meaning that they will become threatened or endangered without adequate conservation (Cornell Lab of Ornithology).

During the nineteenth century, Red-Headed Woodpeckers were so common that farmers would issue a bounty for them (Ibid). Their population was so great that over one hundred of these birds were shot from a single tree in 1840, as reported by the Audubon Society (Ibid). Their previous population size can be attributed to the larger amount of old-growth forest cover over the continent at the time. More recently, their population has been dropping due to urban sprawl. Urban landscaping management leads to removal of fallen dead trees and branches, which serve as the nesting grounds of this woodpecker (Ibid). The clearing of forests for agriculture as well as urban sprawl have reduced the Red-Headed Woodpecker population in our area, as they were seen in counts of around six in the early 1990s in Nerstrand but have only been seen once in more recent yearly counts. Our data from Lakeville and Le Center reflect a similar decline, as they were seen around eight times in counts from the late 1960s and 1970s, but since the 1990s they have only been seen once per count (Figure 14).

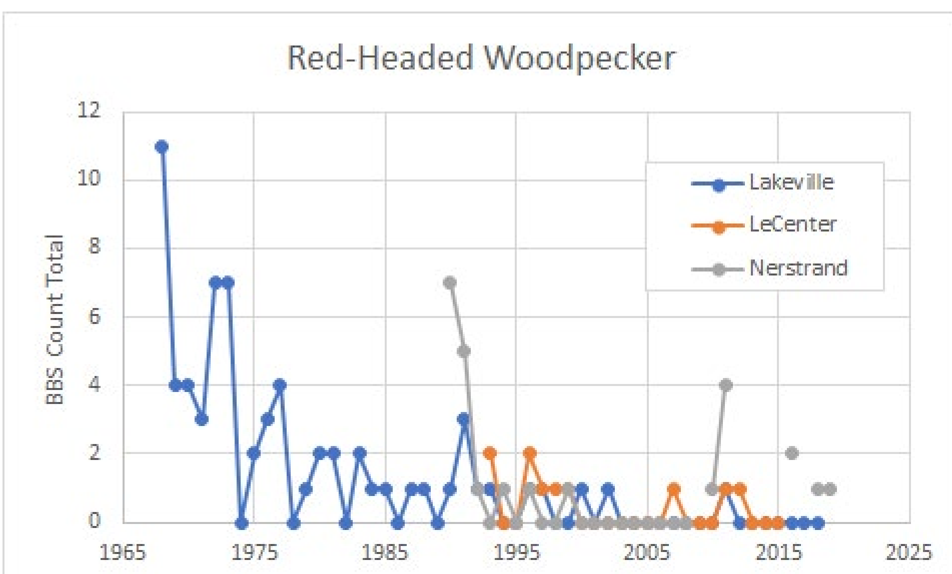

Figure 14 - Counts for Red-Headed Woodpecker from Lakeville, LeCenter, and Nerstrand

The other interior forest species we examined was the Acadian Flycatcher (*Empidonax virescens*). Unlike Red-Headed Woodpeckers, who require dead snags emblematic of old-growth forests but can be found in edge forest or field sites, the Acadian Flycatchers are highly sensitive to forest fragmentation (Ibid). According to the PIF population estimates, their populations have been relatively stable between 1966 and 2015 (Will et al., n.d.). As such, they are not considered a conservation concern (Ibid).

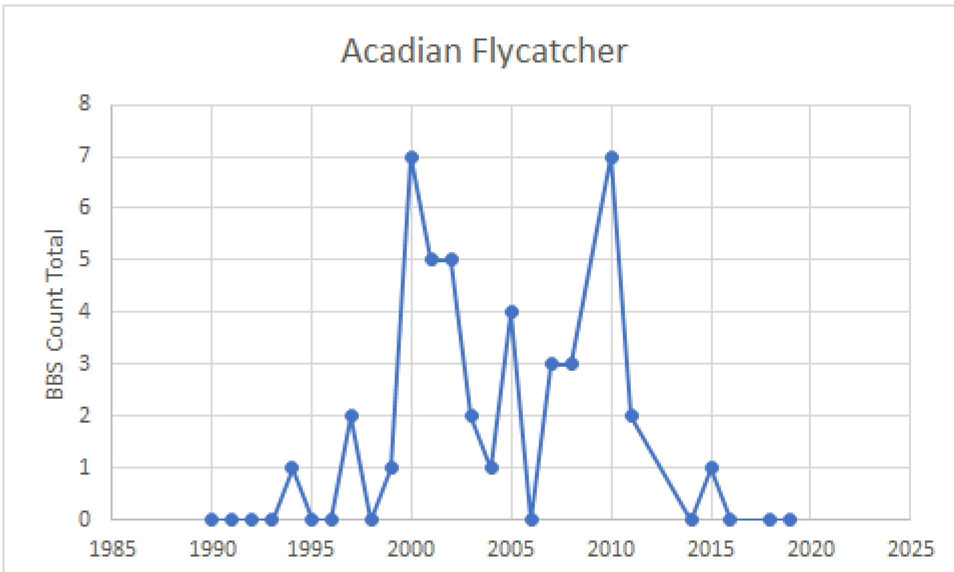

Figure 15 - Counts for Acadian Flycatcher from the Nerstrand data. They were not observed in Lakeville or LeCenter.

Because these species are so reliant on interior forests, the BBS routes in our area rarely report hearing them. This in itself is not concerning as these surveys take place alongside road ditches and the both surveys in our region are relatively far from tracts of interior forest. However, the trend we see in our Nerstrand data is particularly interesting. In Nerstrand, their populations were low in the 1990s and the 2010s, but saw a jump during the 2000s. This might be because the Nerstrand data was collected by students, as opposed to the professional birders working the official BBS routes. Despite this, it is still surprising the flycatchers were not more common in this survey, as Nerstrand Big Woods is the largest tract of forest in our study region and their population was reported as being stable across the United States.

The reasons for their lack of presence might be more complicated than lack of habitat. Recent evidence indicates that the degree of urbanization in the surrounding landscape can have profound effects on the presence and success of the breeding patterns of Acadian Flycatchers (Bakermans and Rodewald 2006). Additionally, the species prefers open spaces for nesting (Wilson and Cooper 1998). Therefore, it is possible that Rice County's increasing rates of urbanization and potential changing density of understory vegetation dynamics due to the proliferation of invasive species might be affecting the presence of these species.

4.2.5 Conservation Case Study: Ring-Necked Pheasants

Although we were unable to analyze changes in wetlands and protected lands with our GIS data, the population changes of Ring-Necked Pheasants (*Phasianus colchicus*) serve as an illuminating conservation case study, since their population has fluctuated dramatically in direct relation to changing land uses and conservation attempts. The habitats of these birds include agricultural land and older fields with a preference for hedges, wetlands, woodland borders, and brush (Cornell Lab of Ornithology). Their preferred habitats switch depending on the season. For example, they prefer roosting in trees and shrubs during the warmer months and move to farm fields and forested wetlands in the fall (Ibid).

Around World War II, Ring-Necked Pheasants were abundant in the area, due to smaller field sizes, brushy fence lines, and more pastures (Anderson 2019). They were so numerous that, according to the Star Tribune, 1.8 million pheasants were killed by hunters in Minnesota in 1941. By 2016, these numbers had plummeted to 196,000 (Anderson 2019). According to the BBS, their population has dropped by one third since 1966. Most of this decline is caused by modern farm practices, namely the shift away from smaller family farms to larger monocultures, which resulted in a loss of grassland habitat (Cornell Lab of Ornithology). Other farming practices such as the draining of wetlands, mowing the sides of roads, and the use of pesticides and chemical fertilizers have also caused their numbers to plummet (Ibid). This trend is also visible in our study region. Our data from Nerstrand shows that they were seen as many as 18 times from counts in the mid 1990s, but now are hardly seen in counts at all. Lakeville numbers confirm this steep decline, as the number of pheasants recorded was around 60 times in bird surveys from the early 1970s and but have only been recorded 10 times in the past decade.

Whereas pheasants were introduced in the early 1900s for sport, the changing landscape has affected them in similar ways to native grassland birds. Unlike many native birds, there has been a concentrated effort through both public and private avenues to understand and combat their dwindling populations. Government cropland diversion programs beginning in the 1930s have always been beneficial to pheasants. Though these initial programs were intended to provide farmers relief from the Great Depression by limiting supply, it had the benefit of putting land into habitat beneficial for pheasants (Berner 1988, 45-63). The loss of these programs drives subsequent declines in their populations (Ibid). Their population decline in Minnesota within the last two decades has been tied to the decreasing number of acres set aside in the Conservation Reserve Program ("Conservation Reserve Program"). Unfortunately, participation in the CRP tends to sway with crop prices, leaving the pheasant population vulnerable to economic shifts. Unlike earlier government programs designed to directly benefit farmers that had unplanned direct benefits for birds, there are now organizations directly interested in advocating for these programs for the sake of wildlife alone. Beginning in 1982, the organization Pheasants Forever formed to revive the pheasant population. Among their various efforts, they advocate for grassland and wetland habitat conservation at the government level (which led to the creation of the CRP program for example), purchase and maintain critical wildlife habitat, fund research in grassland bird population, and continue to educate new generations (Knezevic 2009, "Pheasants Forever").

5. Discussion

Our results reveal the trends of growing agricultural industrialization and urbanization that has occurred in our section of Rice County since the 1960s. From our manual classification data and image observations, we show that while total land in agriculture is decreasing, the size of agricultural fields is increasing. This trend points to the widespread industrialization of agriculture and shows that its efficiency allows for more food to be produced on less land. With the shift away from smaller fields, our maps highlight the trend towards monoculture row crops and away from smaller farms with a variety of crops and dairy operations.

In terms of the "spare or share" framework, we see evidence that intentionally or not, land is being spared in our study area. This can be seen in the reduction of the amount of land used to grow crops and increasing area of forest regrowth. However, it is unknown whether the increasingly spared land is hospitable to a wide range of species or to what degree this growth is dominated by invasive species. Therefore, these recent trends in land use point to the rise of a new landscape, perhaps in line with the spare approach in terms of land use, but that benefits certain species over others which might not bode well for overall biodiversity.

Our analysis of these bird species reveals that some types of birds are doing better in this changing landscape than others. However, when drawing conclusions from our data, it is imperative to consider that while we do see significant trends and correlations, our study cannot establish causal relationships. Another limitation is that most of our studied species[2] are migratory, and therefore their populations may be affected by changes occurring in their winter habitats. Additionally, climate change is also likely affecting species ranges and our observed population changes.

Despite these limitations, our analysis shows patterns of decline for grassland and interior forest species, while the populations of forest edge species continue to grow. The dropping numbers of grassland species, such as Western Meadowlarks and Killdeer, seems correlated with the habitat loss due to fewer small dairy operations with pasture, increasing chemical use, and the shift towards larger fields and intensive row-cropping. Their slow declines might partially indicate an extinction debt following the loss of prairies that has only been exacerbated by the aforementioned changing land use practices. Unfortunately, these species will likely continue to decline, as pastures seem unlikely to return to Rice County due to competition from larger dairy operations. The fate of interior forest species, such as the Red-Headed Woodpecker, is less certain since they are not as well represented in BBS surveys. Whereas we observed increasing forest cover in our region, the interior forest species we examined did not appear to be benefiting from this change. The proliferation of forest growth we witnessed fell within our "cropland with forest" land use category, indicating that much of these increases are small fragments of forest. Our data is limited by the lack of BBS surveys conducted near these forested sites, yet larger

[2] Northern Cardinals and Black-Capped Chickadees are year-round residents

regional trends point to declines in these species, likely due to the conversion of old growth forests and unfragmented forest patches to agriculture. The significance of these slower processes, such as the process of older forests generating large stands of dead trees, might also indicate an extinction debt from prior forest loss.

In contrast, forest edge species seem to be adjusting well to this new habitat matrix. Locally and across America, both Northern Cardinals and Black-Capped Chickadees populations are thriving, despite the fact that Cardinals are a relatively new species in our area. The succession of abandoned lands into first-growth forests we observed might be beneficial for these species, as they are suited to smaller forest fragments and new growth.

However, as exemplified by the case of the Ring-Necked Pheasants, successful conservation measures can be initiated if there is sufficient public interest and will to organize. Hunting interest groups have implemented various policies aimed at increasing wetlands and grasslands to increase the numbers of these birds. These past policies, such as CRP programs, have historically benefitted this species. Yet these conservation-oriented policies and agricultural land use are often in competition with one another, leading to unstable habitat availability and corresponding population fluctuations. Therefore, conservationists seeking to restore bird populations must create policy initiatives that are independent of the rises and falls of the agricultural economy in order to ensure long-term habitat availability.

5.1 Directions for Future Research

There were several details we noticed that are beyond the scale of our project but might lead to compelling research in the future. From our aerial images, we noticed that the rural homesteads have changed significantly since the 1960s in terms of building structure and layout. A closer analysis of these changes might provide insight into how the lives of their inhabitants were changing over time. In terms of changing crop preferences, it was also difficult to identify types of crops, such as corn, soybeans, or grains from aerial photographs. Further research could be done to investigate decade by decade changes in the types of crops farmed across Rice County. This would help to flesh out the larger narrative of the trend towards monoculture that we witnessed across our time period.

Another narrative that continued to appear in our research was the disappearance of small family-run dairy operations over the last century. Historically, Rice County and the rest of this southern region of Minnesota has been reluctant to make the switch away from small dairy operations over to larger-scale dairy farms despite their profitability. This disappearance is ongoing - in 2018 the median income of Minnesota dairy farms dropped from $43,000 to less than $15,000 and one tenth of these farms ceased operations altogether (Belz 2019). However, family farms continue to find ways to survive. For example, current small farms are pivoting towards market niches such as organic milk and other specialty products such as cheese (Meersman 2017, Belz 2019). Ethnographic research could be done to uncover the story of dairy farms in Rice County, people’s perceptions of dairy farms, and how resistance strategies against consolidation have changed over time.

Continued research could also elucidate why and how tree cover is increasing across the county. Within the literature on land use change, we could not find many studies investigating the increase in forested fragments across agricultural landscapes. Therefore, a study in this area would contribute to a sparse field of research on these more recent changes. Interviewing the owners of these patches of uncultivated land could reveal whether they have an awareness of this larger trend and determine if there is any intentionality in the growth of these forests. Another avenue of this research could be more ecological in nature. Further spatial analysis of these areas could show whether these forest patches are close to rivers or occurring on hilly lands which makes them less suitable for farming. Spatial analysis could also be employed alongside ecological surveys of these growing forest fragments to understand the character of this new growth. A methodology like this could unveil the suitability of these fragments to different bird species in terms of their size, availability of habitat, and food sources.

Further studies related to forest cover also investigate the role of invasive species in their regrowth. It is possible that the increased forest cover may not necessarily be beneficial to native ecosystems if it is dominated by non-native species. Surveys of these patches would help identify the composition of these regrowing forests. Such research could also reveal whether these patches might provide adequate habitat for different bird species. In lieu of these biologic surveys, speaking with landscape managers (e.g. local DNR officers, state park managers, farmers, etc.) might help understand the complexities of these changes. Additionally, their opinions towards invasive species have also likely changed over time due to developments within conservation biology. An ethnographic study could investigate how the attitudes of land managers towards invasive species have changed and how this influences their conservation strategies. These studies would reveal how this increasingly abundant forest growth came about and what species it might benefit.

6. Conclusion

Over the course of the past eighty years, Rice County's agricultural landscape has shifted from small row crop fields and family dairy farms to larger fields of monocultures. This land use change was a result of farmers' responses to new technologies, such as pesticides and mechanization, and fluctuating markets. While these changes have increased the productivity of the land, it has also resulted in rising numbers of farmers being forced out of agriculture. The effects of these transformations have been felt by more than just people, as they also seem to correlate with declining grassland and interior forest bird species populations. Prior to settlement, our study region was half in prairie and half in continuous deciduous forest. This prior landscape was likely able to support far more interior forest species and grassland species than the patchy forests and limited amount of prairie present in the modern landscape. The results of our bird populations correspond to the notion that bird species in Rice County are responding to changes in habitat availability. Therefore, the historical relationship between societal changes and ecological assemblages is evident, which means that present levels of biodiversity cannot simply be viewed as a snapshot in time. Rather, we see that species populations are in flux, especially in an area such as Rice County, which has experienced pervasive landscape changes in recent history.

If we take the results of our study and fit them into the two land use regimes proposed by the “spare or share” debate, we can see that our region appears to align more with a spare approach. Agricultural productivity in the region has soared due to the increasing inputs used by modern farmers, such as larger machinery and widespread use of fertilizers. Associated with this increase in productivity are decreases in the number of farms and the amount of land in agriculture. It is crucial to notice that this switch from share to spare does not equally benefit or harm all bird species, which goes against the “spare or share” framework’s predicted overall benefits to biodiversity.

Therefore, we see three takeaways from our study: 1) political, economic, and technological changes drive the choices made by land managers, 2) landscapes and species populations shift in response to these land use decisions, and 3) the biodiversity benefits predicted by the “spare or share” framework are not realized for all species without intentional human management. These changes have coalesced into novel landscapes and ecosystems and as such, those concerned with biodiversity must acknowledge the present-day landscape and species matrix. End to end row-cropping is unlikely to disappear and prairies will not rise to take their place. Instead, conservationists should pursue opportunities within this matrix such as decreasing pesticide use, decreasing field disturbance through conservation tillage, or working with landowners to turn marginal lands into areas suitable for species in decline. At the same time, the various societal realms that dictate these land management practices must be taken into account, otherwise strategies will only be short-term solutions, as evidenced by the fluctuation in CRP enrollment due to changing crop prices. Because social and temporal contexts play a key role in shaping current landscapes and species assemblages, the “spare or share” framework would benefit from this holistic perspective, so its future research is better situated in the complexities of managed landscapes.

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8. Appendix: Supplemental Methodologies and Materials

In this section our methods for processing our data are more thoroughly detailed so that future students can use them for similar projects.

8.1 Land Use Change Analysis

For non-digitized map materials, such as aerial photography in our case, the first step in this process is to scan the images for the area of interest using a high-resolution scanner. For best results, it is important that these images are scanned using similar lighting and each image is scanned individually. Although these are black and white images, we got our best results scanning them in color and with very high resolutions (around 800 dpi). The original aerial photographs all had black borders, which we removed using photo-editing software, such as Preview on Mac, or Photoshop on Windows. This made the stitching of the photographs easier in ArcGIS. The images also varied in their levels of contrast and exposure, which we adjusted using the same software in order to equalize the photos' color balances. Further color adjustments were made later in the ArcGIS Pro using the "Color Balance Mosaic Dataset" tool.

Once these images are scanned and available on a computer, they can be opened in a GIS, such as ESRI's ArcGIS Pro. Images like these can be opened by this system in raster format when they are in JPEG, TIF, or PNG format. However, these images will not be located in the proper location on the basemap initially. To make images appear in the right location in the GIS they need to be georeferenced. The best workflow we found for accomplishing this was to get the images in approximately the correct place using the "move," "scale" and "rotate" tools in the Georeference toolbar and then adding enough control points to ensure the photographs are lined up correctly. When selecting control points, we started by triangulating our first three points which helped to prevent distortion. Selecting right angles in the landscape, such as road intersections, are best for accurately placing control points. It's also good practice to zoom in very closely to the images as you select control points for the highest possible accuracy. After placing at least ten control points, the points with the highest residual values can be deleted in the control points table to reduce human error. Due to distortions in the photographs, we found it best to use at least a 2nd order transformation, which is a setting also located in the control point table. Sometimes a 3rd order transformation was necessary for the most distorted photographs. In our case, we did this for approximately 80 images across two data sets from 1964 and 1970.

Once all these images are georeferenced, they can be combined into one image. ArcGIS Pro has a tool called "Raster Mosaic Datasets." These datasets can contain multiple rasters at once and have several functions available to improve the quality of the resulting single raster. The first step is to ensure all the raster images have the same number of bands. If they do not, we found it best to separate the different colored images into separate raster mosaic datasets. Then the rasters must be imported into the raster mosaic dataset using the "Add Rasters To Mosaic Dataset" tool. We made sure that as these rasters were added into the dataset that it was building statistics for pyramids. We used the settings "maximum" for the mosaic operator input and “match” for the mosaic colormap mode input. Using the “match” setting and making sure the bands were the same allows us to use the functionality of the “color balance mosaic dataset” tool. This smooths out the color across the final image so it appears more cohesive. We preferred to use the “histogram” setting so that the final image had relatively balanced colors and was not excessively bright.

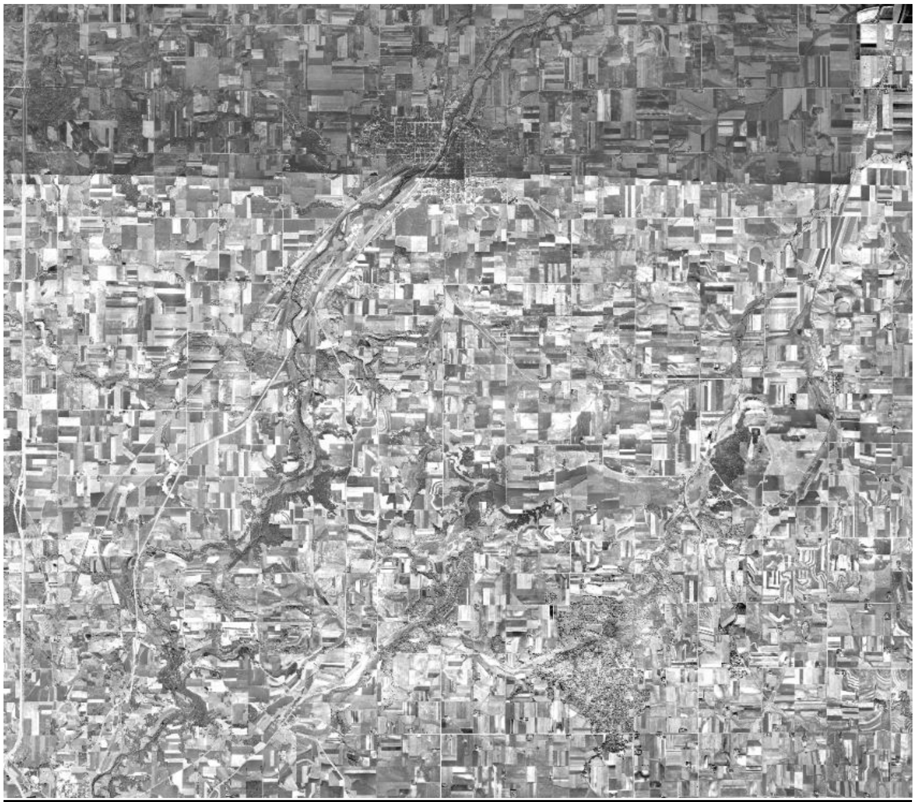
8.2 Manual Classification of Rasters

In our pursuit of classifying rasters we ended up doing a random sample manual classification of our images. For creating our grid, we used the “Fishnet” tool in ArcGIS to create squares of our chosen size. We settled on a square size of 1000m by 1000m as it was the size where it was still possible to make out features in our lowest resolution images. Our division of the study area resulted in 637 squares per raster. For our purposes, we decided to sample 100 squares (about 15% of each map). However, this can be modified according to the size of region studied and the desired precision of the statistics. The larger the sample size, the smaller the standard error will be which can provide smaller confidence intervals for estimating the true proportion of land use. We then randomly generated one hundred numbers between 1 and 637, and manually classified the squares in our generated fishnet whose ID corresponded to these numbers.

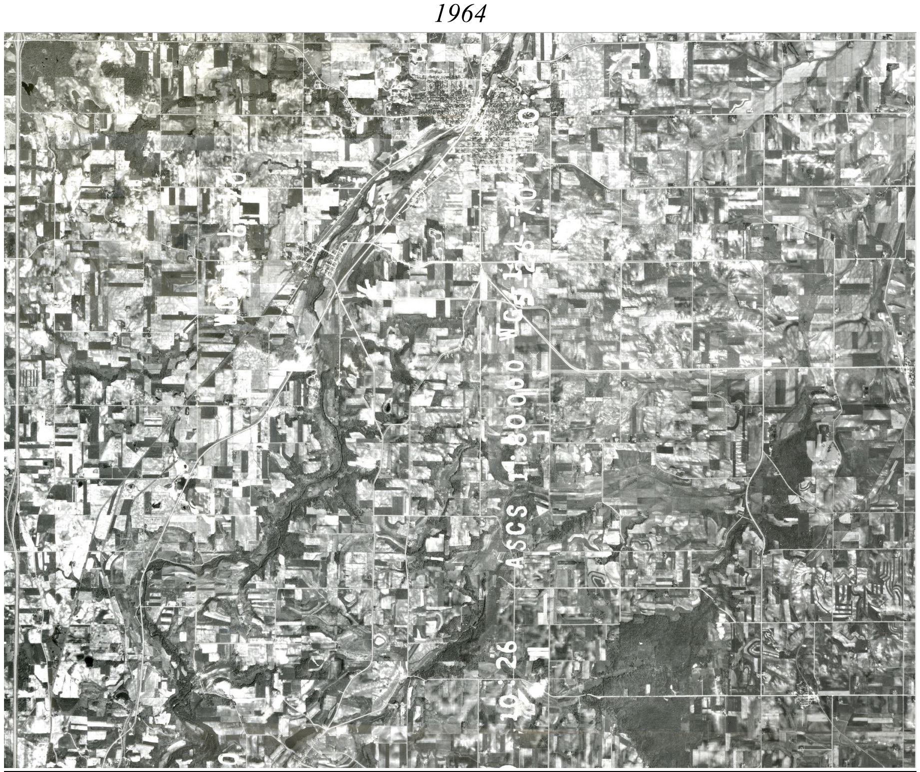
For this image classification, we settled on five classes: forest, cropland, cropland with at least 10% forest, wetland, and developed. We used the 50 percent rule for the forest, cropland, and developed categories, which means that we classified the square as whichever land use type covered at least half of the square’s area. This is a relatively standard rule of thumb in classification. However, we also wanted to account for smaller sections of forest that wouldn’t necessarily fulfill the 50% rule but would still provide sufficient edge forest habitat for certain bird species. This is why we created the “cropland with at least 10% forest” category. Originally, we were interested in including a pasture or grassland category. Unfortunately, it was impossible to distinguish pastures from cropland in the images, so we ended up removing it from our classes. This was our thought process in delineating our classes, but these classes can be adjusted in the future according to the detail of the images used and the intent of the study.

After classifying our samples, we then calculated the relevant statistics. Using Microsoft Excel, we calculated the proportion of each type of land use for all four years using. We then calculated standard error and 95% confidence intervals for each proportion.

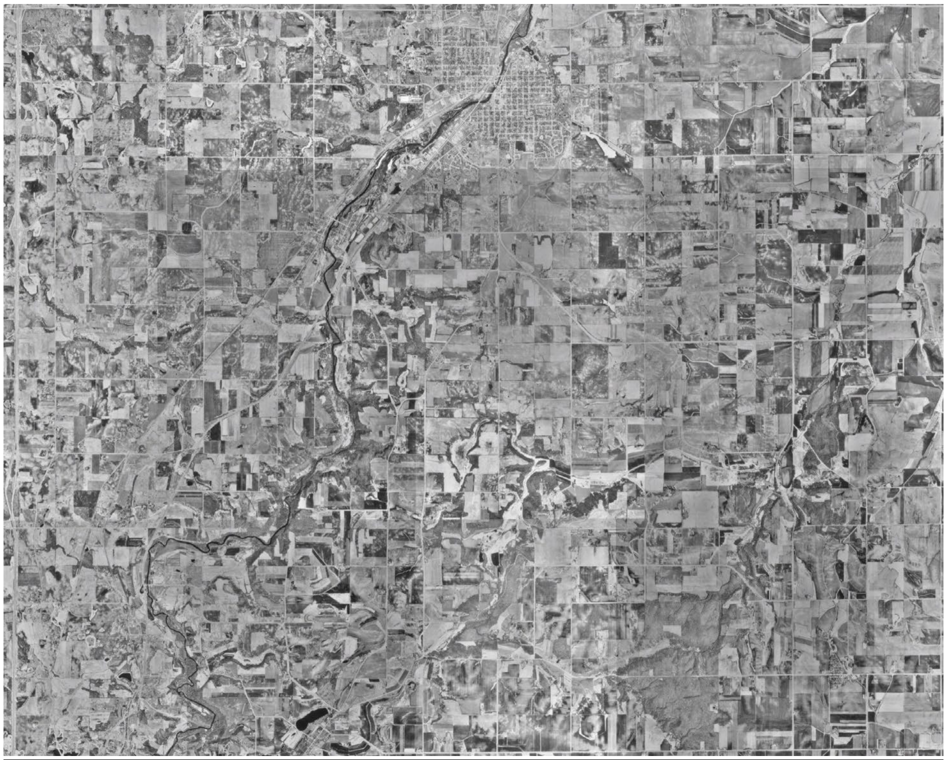
8.3 Final Maps



1964



1970



1991

2017