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Title: Management implications for invertebrate assemblages in the Midwest American agricultural landscape
Issue Date: 2017-02-02

Chapter 1

General Introduction

Introduction

Global population is growing ~ 1.1% per annum with projected populations reaching 9.6-12.3 billion by 2100 (Gerland *et al.*, 2014). Extreme poverty has declined globally by more than half falling from 1.9 billion in 1990 to 836 million in 2000 (Millennium Assessment, 2005). Increased agricultural production of food, fuel and fiber will be necessary to meet the needs of the growing population. Agricultural production of food, fuel and fiber involves socio-economic issues as well as environmental issues (Garnett, 2014). Controversy swirls around the use of food to produce fuel (Davis *et al.*, 2012), increase of production through genetic modification of seeds (Wisniewski *et al.*, 2002), pesticide use (Fernandez-Cornejo *et al.*, 1998), high fructose corn syrup food additives (Rippe and Angelopoulos, 2013), dietary preferences (Hansen and Gale, 2014), malnourishment and its counterpart obesity (Horvath *et al.*, 2014), and food access and security (Godfray and Garnett, 2016).

Apart from these issues around agriculture itself, goals to increase agricultural production are also often in competition with other societal goals. Water used to increase food production leads to reduced availability for other purposes, including human consumption (Haddeland *et al.*, 2014) and clearing forested land for use in growing agricultural products decreases biodiversity and carbon sequestration (Carlson *et al.*, 2012). The difficult and critically important challenge is to balance the multiple needs of society in the most sustainable way possible.

This thesis concentrates on how common vegetation management impacts invertebrate biodiversity, as a critical resource, in an area which is used for intensive agricultural production. Biodiversity conservation is the basis for preserving existing ecosystems and ecosystem services (Griggs *et al.*, 2013). The Earth Summit held in Rio de Janeiro, the United Nations developed a treaty called the Convention on Biological Diversity (CDB) that sets goals for biodiversity conservation on both national and global scales. The goals of the CDB are the conservation of biological diversity, sustainable use of components of biological diversity, and the fair and equitable sharing of the benefits arising out of the utilization of genetic resources (Bell, 1992). In 2010 the international community set targets for biodiversity to be achieved within a decade. A mid-term analysis of progress toward these targets has shown little progress and some deterioration (Tittensor *et al.*, 2014).

This shows that increasing the efforts to conserve biodiversity is needed, including in agricultural areas. The aim of this thesis is to look at vegetation management within the agricultural landscape and determine which practices are most beneficial to invertebrate assemblages and the associated food web. The debate on how best to achieve biodiversity goals in relation to the need to increase agricultural production concentrates presently around two contrasting strategies, "land sparing" and "land sharing" (Phalan *et al.*, 2011; Grau *et al.*, 2013). My study is based on the question whether biodiversity conservation measures from the sharing strategy, which is traditionally applied in the countries like the Netherlands and United Kingdom in Europe, would be applicable in the Midwest of the USA, which follows traditionally a sparing strategy. The strategies find their theoretical bases in different interpretations of the ecological theories that describe the relationship between local populations or communities and the surrounding landscapes. Before I will discuss the research of this thesis, I will explore these ecological theories.

Ecological theory

Early ecological studies operated on the premise that species were distributed wherever there was appropriate habitat. At first, the abiotic characteristics of the habitat were thought to be the most important for the presence of species, later the biotic characteristics, i.e., species interactions, were added to the abiotic characteristics. These abiotic and biotic factors were the base of early niche theory for individual species (Soberón, 2007). Assembly theory uses the idea of species sorting to explain how abiotic and biotic factors influence certain species out of all possible species in the species pool to form a community.

The theory of island biogeography challenged this viewpoint (MacArthur and Wilson, 1967). They proposed the idea that species go extinct on a regular basis, leaving the habitat vacant until the next colonization event, but that extinction rate is highest on small islands and that colonization rate is smallest on islands far from the mainland. The theory is now applied to any patch of habitat, e.g., a mountain top, lake, park or conservation area, that is surrounded by a completely different type of habitat. For practical biodiversity conservation, this theory is usually interpreted as that conservation areas should be as large as possible and well connected to other areas with the same habitat in order to keep or get a high species richness within the conservation area.

In an expansion of this theory, the concept of metapopulations views populations of a single species in balance between colonization and extinction within a group of habitat patches (Hanski, 1998). A metapopulation of a species will have a high survival probability when the group of habitat patches is large and well connected. This theoretical paradigm further embraces the concept of source-sink dynamics, which supports the idea that a species can and does occupy suitable habitat as well as maintain stable populations in a large enough set of habitat patches that combines suitable and unsuitable patches (Pulliam, 1988).

Metacommunity theory expands the idea of metapopulations to a set of locally interacting species within a set of connected habitat patches and relate these to environment and regional processes (Logue *et al.*, 2011). Hubbell (2001), on the other hand, presumes in his neutral theory that both extinction and colonization may be largely stochastic processes so that the patterns of species distribution and abundance in sets of habitat patches that can actually largely be explained without taking the ecological traits of species or habitat characteristics into consideration, in other words without considering niches.

Of course, it was realized that habitat patches were seldom surrounded by 'empty' areas like seas. The study of the relationship between a local community and its surrounding landscape has led to a large body of literature and added further mechanisms, such as mass effects, spill over, dilution, etc., to ecological theory that may or may not be applicable in a certain specific situation. Tscharntke *et al.* (2012) has summarized these mechanisms into six hypotheses on the effect of surrounding landscape on local communities and populations and two hypotheses on the efficiency of biodiversity conservation measures in agricultural landscapes.

Vellend (2010) has tried to bring all theory discussed above together into a simple scheme of four key processes that determine a community at a certain location at a certain time: selection, drift, speciation and dispersal. Selection is the process that defines the relative abundance of species in a community based on abiotic and biotic factors. Drift is the stochastic process of species abundance fluctuations that can lead to local extinction. Speciation is the development of new species in an area and is probably mainly relevant at high levels of spatial and temporal scale. And dispersal is the process of colonization of species. In selection, the assembly theory and in drift, speciation and dispersal the neutral theory can be recognized.

Land sharing and sparing in agricultural areas. As said before, at the moment two main strategies for conserving biodiversity in relationship to agriculture are under discussion: land sharing vs land sparing. These strategies are discussed in view of the amount of biodiversity that can be conserved at the level of nations, regions or even at the global level. Traditional EU biodiversity conservation in agricultural areas follows a sharing strategy, while that in the USA follows a sparing strategy. Because this thesis partly focusses on the application of sharing techniques in the USA, it is worthwhile to explore both strategies a little further.

The discussion of balancing agriculture with nature has its roots in ecological theory. The problem of harmonizing agriculture and nature has long been a topic of research (i.e. Waggoner, 1995; Huguet, 1978). Green (2005) coined the terms 'land sharing' and 'land sparing' to express these opposing philosophies. Land sparing divides the land into homogeneous areas with distinctly different goals (Ausubel *et al.*, 2013). Agricultural land optimizes yield through the use of fertilizers, pesticides, and irrigation in order to use as small areas for agriculture as possible. Biological reserves target conservation of specific ecosystems and biodiversity. This strategy is associated with islands of nature being separate and protected from human disturbance (Higgs, 1981). Island theory is thought to be applicable here and the main goal of conservation is to establish large protected areas, connected by corridors if possible.

Land sharing integrates wildlife friendly farming techniques into a heterogeneous landscape (Fischer *et al.*, 2008; Pywell *et al.*, 2012). Proponents of this strategy emphasize interactions between farmed and unfarmed habitats. Larger farming areas may be required since farming is less intense resulting in lower agricultural production than would be possible if agriculture were optimized (Green *et al.*, 2005). Compensation may be given for the loss of yield that accompanies wildlife friendly farming techniques (Wilson and Hart, 2000). The eight hypotheses of Tscharntke *et al.* (2012) are applicable to this strategy.

Land sharing and land sparing strategies have been presented as dichotomous choices (Green *et al.*, 2005). Green (2005) presented yield density models to predict extinction risk of both individual and multiple avian species. These models make several assumptions regarding population density in non-farmed areas, impacts of chemical applications, water usage, and impacts of habitat fragmentation and dispersal corridors. This must take into account that reducing waste, increasing efficiency, and altering the food delivery system may have unintended consequences, as suggested by the Jevons paradox. The Jevons paradox, developed in 1866, was coined to explain the phenomenon of an action to conserve natural resources that allows more of the resource to be used and subsequently may harm that resource in the end (Polimeni, 2012). The land sparing model also assumes that as yields increase land will become freed for nature restoration. Freeing land from agricultural production with land sparing also provides land for other uses such as urbanization.

Law and Wilson (2015) provide analysis of land sharing and sparing philosophies and resulting policy decisions within an ecological context. Programs such as conservation set-asides, organic farming and environmental certification have a different impact in pristine environments than in areas degraded by agricultural practices (Law and Wilson, 2015; Cormont *et al.*, 2016). Law and Wilson (2015) examine two land sharing and three land sparing strategies under both pristine and agricultural baseline conditions. Their model shows that the initial proportion of the landscape devoted to agriculture as the most important parameter in predicting biodiversity changes.

Proposals have been made to reconcile these diametrically opposing strategies to address the challenges related to food security and access, ecosystem services and land scarcity (Fischer *et al.*, 2014; Grau *et al.*, 2013). Fischer et al. (2008) propose strategies for biodiversity conservation which draws on the strengths of both land sparing and land sharing philosophies. Phalan *et al.* (2011) suggests a more sophisticated land sparing philosophy that utilizes indigenous reserves, habitat banking, local knowledge and avoidance of agrochemicals and mechanization. Hayashi (2011) suggests the importance of management intensity as an important consideration when examining these opposing strategies. Application of various proposals requires consideration of social, political and technical issues (Phalan *et al.*, 2011). While specific proposals addressing biodiversity in Ghana are not directly applicable to Illinois, the concepts can be applied globally, e.g. decreasing management intensity through reduced tillage.

I have elected to focus on those management practices that probably do not affect agricultural yield or management within agricultural fields. Yield was not measured in any of our studies. I was mostly interested in enhancing invertebrate biodiversity in the existing landscape, not influencing global or regional policy (Grau *et al.*, 2013). I believe it is important to acknowledge that high yielding agricultural areas (USDA, 2016) be maintained and valued for what they are as well as enhanced where they are lacking.

Agriculture

Overview of agricultural history in Europe and the USA. As said before, traditional EU biodiversity conservation in agricultural areas follows a sharing strategy, while that in the USA follows a sparing strategy. This is undoubtedly related to the history of agriculture in both regions. European and United States (US) agriculture is interconnected since the early 20th century. World War I disrupted European agriculture and the US responded by increasing production. After the war ended, demand dropped and crop prices fell dramatically in the USA (Sumner, 2007). American farmers struggled to make payments on the land and machinery they had purchased to meet production that was no longer needed. The agricultural system was already struggling when the depression hit in 1929 (Sumner, 2007). Prices fell further to a third of what they had been a decade earlier. In a reaction, the first farm bill was launched to raise commodity prices by paying farmers to limit production (Bowers *et al.*,1984).

In 1932 the Soil Conservation Service was formed to promote conservation practices and allow the land to recover after the severe erosion known as "the dust bowl years". Practices included tree and grass plantings to anchor the soil, terraces and contour plowing, and crop rotation which included allowing the land to remain fallow during the rotation cycle (Bowers *et al.*, 1984). Crop prices again rose during World War II creating again an overproduction bubble that burst with the end of the war (Sumner, 2007). Since that time there have been opposing pressures on famers with the need from the agri-industry encouraging hedge-row to hedge-row planting and the societal need to conserve the land for future production. New technologies such as the combine harvester improved farming efficiencies (Dimitri, 2005). Current practices focus on reducing soil erosion and inputs to waterways (Reimer and Prokopy, 2014). Enhancing habitat for pollinators is a relatively new practice with the specific goals of reducing loss of honey bee colonies, providing habitat for Monarch butterflies, and the general goal of creating and restoring habitat (Obama, 2014).

As a result of the privations of WWII, food security in Europe became a high priority. In 1957, a group of 6 countries signed the treaty of Rome which was the precursor to the European Union (EU). In 1962 the Common Agricultural Policy (CAP) went into effect which had the result of assuring food security for Europe (European Commission 2012). Agri-Environment Schemes (AES) began in the EU in 1985 with the goal of compensating farmers for loss of yield and subsequent income loss that resulted from less intensive agricultural practices (Kleijn *et al.*, 2006). More developed countries in the north and west are the drivers of most AES which are now the main tool for biodiversity conservation in agricultural areas (Kleijn *et al.*, 2006). Currently all EU members are required to participate but may develop their own AES practices (Kleijn *et al.*, 2006). In addition to conserving high-value natural areas, AES provide subsidies for the protection of traditional farming as current farming practices replace traditional farming methods that are no longer economically practical (Kleijn *et al.*, 2006). Countries in the EU have the additional mission to preserve the cultural landscape and heritage (Antrop, 2005).

Programs in the EU and US have many goals in common. Both have subsidies for reducing water pollution and soil erosion. Both programs offer subsidies to stimulate organic farming and to address the decline in pollinators. But also, large differences exist. Government agriculture programs in the US remain mostly focused on conserving the soil and reducing input into the waterways, while EU programs now also offer subsidies to address species loss in the agricultural landscape. In the US, the protection of threatened species and habitats are often under the aegis of other agencies (Endangered Species Board 2011; Innes *et al.*, 1998) and private organizations (Dobson *et al.*, 1997; Kareiva *et al.*, 2014).

At the present time, biodiversity conservation in agricultural areas in the US is typically a sparing strategy, while that in western Europe is a sharing strategy (Batáry *et al.*, 2011; Boitani and Sutherland, 2015). In the Midwest United States, management is mostly focused on patches of habitat with buffer areas of protection. In contrast, practices in Europe are often focused on providing habitat for certain species or taxonomic groups: birds (Kleijn and Sutherland, 2003; Kragten *et al.*, 2008), mammals (Boatman, 1999), and insects (Desender and Turin, 1989; Noordijk *et al.*, 2009). Practices in the EU often focus on the edges, verges and hedgerows of agricultural fields and roadways creating a matrix of available habitat within the agricultural system.

Government incentive programs in the US generally have not been established to benefit invertebrates, but have probably enhanced invertebrate populations as a side-effect of practices with other end-goals. A recent addition of practices to enhance pollinators has uncertain funding (USDA FY 2015 Budget Summary and Annual Performance Plan, U.S. Department of Agriculture). Habitat restoration projects may require special effort to host specific native vegetation for highly specialized invertebrates or translocation of insect species where no local populations currently exist (Fischer and Lindenmayer, 2000).

Differences in conservation strategies in the EU and the US are somewhat understandable because of the differing lengths of time the landscape has been farmed. Farming began in the middle-east and moved gradually through Europe. First People (aboriginal people) farmed in close proximity to settlements in North America (Fritz, 1990). Indigenous farming practices were rudimentary and in relatively small areas of the landscape (Fritz, 1990). In Europe, much of the land has been intensively farmed since Roman times and in the US since European settlement, fewer than 200 years ago. Farmers came to the US mostly from Europe and brought European practices with them (Hewes and Jung, 1981). However, the European practices came not from a united Europe but from individual countries with vastly different practices. In the US, these practices were adapted to a landscape that differed in the amount of available space and access to support from developed communities. Farms in the Midwest were often begun with homesteads provided by the government for little or no cost and were isolated from population centers (Bell, 2012).

As the Midwest becomes more and more like Western Europe with few large areas left unaffected by agriculture, agri-systems may benefit from techniques developed for a more managed landscape. There is some support for moving to a sharing approach in the US (Rosenzweig, 2003). Adoption of a mix of land sharing and land sparing approaches may increase the odds of success (Fischer 2014; Grau *et al.*, 2013). This mix includes habitat restoration, expansion of remnant vegetation patches and restoration of both vegetative structure and function of waterways and riparian zones (Tscharntke *et al.*, 2005). Vegetation corridors are generally recognized as a means of linking isolated patches (Cook, 2002). Making optimal use of linking corridors through agricultural programs is an underutilized resource in the United States.

Understanding the published research. A survey of the literature provides much information about invertebrate assemblages in varied habitats across the globe. However, translating the results and conclusions of these studies requires some caution. Terminology, vocabulary and native language present opportunities for misunderstanding. Countries in the EU each have their own native language and generally have little difficulty with "scientific" English as the language for peer reviewed papers. But even English presents difficulties. Instructions to authors generally indicate that either "British" English or "American" English may be used, but not mixed within the same paper. Relevant to this dissertation, is a discussion of agricultural terminology which often differs in the US and EU. Terms are so well understood that they are not always defined. But they may have different definitions in the US and EU. For example, "hedgerow" creates different mental pictures in the Netherlands, Great Britain, France, and in the Midwest, United States. The misunderstanding of terminology may result in a lack of understanding of study results. A glossary of easily misunderstood terms is provided for the purposes of this dissertation.

Farming itself is different in the US and EU. The EU has a wide diversity of crop types including beets (*Beta vulgaris*), spelt (*Triticum spelta*), barley *(Hordeum vulgare*) and rye (*Secale cereale*). In the Midwest US, most crops are corn (*Zea mays*) and soybeans (*Glycine max*) with occasional winter wheat (*T. aestivum*). There is more emphasis on natural or organic crops in the EU while most crops in the Midwest are genetically modified to withstand chemical input of herbicides and pesticides (Wier and Calverley, 2002; Gaskell *et al*., 1999). Reduced tillage was promoted as a means of reducing soil erosion, soil compaction and chemical run-off, but is used to varying degrees in the US. Much of the landscape in the EU is kept open by farming and farm animals. In the US, management tools for grasslands include grazing and mowing (Jonas *et al*., 2002; Knapp *et al*., 1999). Additionally, fire is a frequently used tool for vegetation management in the US. Grazing by indigenous mammals is preferred over domestic livestock to preserve the prairie landscape (Knapp *et al*., 1999). Europe uses commercial production of food as an agricultural practice in the conservation of areas of high biodiversity (Boitani and Sutherland, 2015). Similarly, some forests in the US are managed by the department of agriculture with timber production a primary goal.

Glossary

Beetle bank: a strip of low semi-natural vegetation that runs in the middle of agricultural land that has been especially installed to promote predators in the field
Buffer strip: land maintained in permanent vegetation that helps to control air, soil, and water quality, along with other environmental problems, dealing primarily on land that is used in agriculture
Dike: a long wall or embankment to prevent flooding from the sea; a ditch or a watercourse
Drainage ditch: a ditch for removal of excess water
Field margin: a row of semi-natural vegetation alongside agricultural land
GM: crops with modified DNA to improve resistance to pests and diseases, environmental conditions, spoilage, chemical treatments or to improve the nutrient profile
Hay field: a field where grass, alfalfa, etc., are grown for making into hay
Headland: a row at ends of an agricultural field for turning equipment
Hedge-row: a row of bushes or small trees at the edge of an agricultural field
Lea: a tract of open ground, especially grassland; meadow
Meadow: a grassland, either in its natural state or used as pasture or for growing hay
Organic: a method of crop and livestock production that does not to use pesticides, fertilizers, genetically modified organisms, antibiotics and growth hormones. A GM crop can be organic in the US
Pasture: land covered with grass and other low plants suitable for grazing animals, especially cattle or sheep
Shrublands: naturally occurring or manmade plant community characterized by vegetation dominated by shrubs, often also including grasses, herbs, and geophytes.
Turn row: a row at the ends of an agricultural field for turning equipment
Verge: a row of semi-natural vegetation alongside a road, railway or agricultural land
Water way: broad, shallow channels designed to move surface water across farmland without causing soil erosion
Wood bank: a row of large trees at the edge of an agricultural field
Wood lot: a segment of a woodland or forest capable of small-scale production of forest products such as wood fuel, sap for maple syrup, as well as recreational uses like bird watching, bushwalking, and wildflower appreciation

Agricultural History of Illinois. The topography of the state was strongly influenced by the series of four Pleistocene glacial episodes. The last of the glaciers retreated about 13,000 years ago. As each of the great ice sheets advanced and retreated, it brought and left behind deposits of clay, sand, gravel, and boulders known as glacial till. Wind picked up the fine debris particles called loess and deposited it across the Illinois landscape. This material is rich in minerals, has a uniform consistency, and retains moisture (King, 1981). These qualities make excellent soil for growing crops.

Before the arrival of European settlers, what is now Illinois was covered with a mixture of grasslands and deciduous forest (King, 1981). Most of the central region was prairie, interspersed with wetlands and forested riparian areas. Fire (both natural and induced) was a major vegetation control factor, keeping the forests from encroaching on the grasslands. Indigenous people occupied the land from about 12,000 years ago. Cahokia (an area in southwestern Illinois) was the largest city in North America one thousand years before European contact (Fritz, 1990). Archeological sites document occupation in other areas of the state as well. There is evidence that fire was a tool often used by Native American people for a variety of reasons (Stewart, 1951; McClain and Elzinga, 1994).

Vegetative landscape was described by early explorers i.e. as Father Jacques Marquette and Louis Joliet in the 1670's (Shae, 1853). Illinois was controlled in succession by the French, British, and became a United States possession in 1778 (Wikipedia). In 1803, President Thomas Jefferson commissioned Meriwether Lewis and William Clark to document the plants, animals and geography of the region which included Illinois (Cutright, 2003). Illinois became a state in 1888. In the early part of the nineteenth century the Government Land Office (GLO) carried out the Public Land Surveys establishing the grid coordinate system for Township, Range, and Section lines. Vegetation composition was recorded in the surveys according to standard methodology (Hutchison, 1988). This has become the base line for understanding 200 years of change since European settlement. The Illinois invertebrate fauna has doubtless also been changed by human activity, although we lack the records to quantify the changes in invertebrate populations since the advent of the settlers. The vast expanses of land that have been given over to agriculture would seem to provide new kinds of ecosystems for certain insect species to exploit.

Settlers coming from Europe brought their farm practices to the "new world" (Hewes and Jung, 1981). Fences were used to exclude cattle from crops rather than enclose the crop fields. As settlers moved west, fencing large areas became less and less practical. Trees used for fencing were scarce in the prairie and used for construction of homes and barns when available. Digging ditches to mark field boundaries was labor intensive and the glacial loess did not have enough rocks to mark field edges. The development of barbed wire marked a turning point in Illinois agriculture (Hornbeck, 2010). Surveys were conducted regularly and sent to state agricultural departments to document type and amount of fencing (Hewes and Jung, 1981).

Field boundaries of agricultural fields recently cleared initially retain the character of the area that covered the land before clearing (Watts, 1975). Soil type and seed bank often remain in place for some time after clearing. Eventually the impacts of soil erosion and chemical amendments alter the character of the edge. In North America, fence rows may contain large canopy trees and dense shrubs and are generally left unmanaged (Fritz and Merriam, 1994). Seeds transported by the wind as well as those carried by birds and mammals repopulate the area with a new assemblage of plants (Poggio *et al.*, 2010). Fences are ideal perching points for birds and the edge provides a protective travel corridor for both small and large mammals (Gehring and Swihart, 2003). Many insects are dependent on specific plants for their existence. As the edges change in character, the invertebrates change as well.

One hundred years ago, farms were generally small and produced much of their own food (Shammas, 1982). Each farm would have chickens, a cow or two, maybe some pigs, and probably horses used as draft animals and for transportation. Livestock grazed in pastures; oats and grasses were planted to supplement the needs of the livestock. As time passed, agriculture has become more efficient (increasing yields) as well as utilizing more of the landscape (Table 1). While the area devoted to arable crops increased, other landcover types also changed. Tractors replaced draft animals; cows, chickens and pigs were moved to "confined feeding operations" (Burkholder *et al.*, 2007); wetlands were drained (Logan *et al.*, 1980); and city dwellers flocked to "McMansions" in the rural countryside (Nasar and Stamps, 2009). Forests increased and shrublands decreased (Table 2).

Farm-level intensification includes shortened crop rotation and increased chemical usage. On the landscape scale, fields have been consolidated and have increased in size resulting in a simplified landscape with the loss of non-crop field margins (Söderström and Pärt, 2000; Tscharntke *et al.*, 2005). At the level of the field, the use of chemicals, improved harvesting technologies, and increased tillage frequency has resulted in fewer invertebrates (Wilson and Hart, 2000). Crops are less diverse and planted in greater densities with uniform field margins (Table 1).

Table 1. Summary of land cover (in acres) number of cattle, human population and % change. Agricultural statistics are from the U.S. Department of Agriculture, National Agricultural Statistics Service: human population statistics are from the U.S. Census Bureau.

Central Illinois	1950's	2000's	%Change
Corn	1,473,000	2,512,000	+71
Soybeans	1,141,000	1,608,000	+41
Wheat	275,000	61,000	- 87
Oats	482,000	0	-100
Hay	338,000	46,000	-86
CRP	n/a	114,000	n/a
Forest	301,000	367,000	+22
Buildings	n/a	205,000	n/a
Cattle	556,000	132,000	-76
People	624,000	724,000	+16

Tillage practices vary greatly in the Midwest US as well as elsewhere in the world. Conventional tillage, reduced tillage, and conservation tillage are measures of crop residue left on the field surface. Impact on the soils is largely dependent on the type of machinery used, the frequency of equipment passage and chemical applications. The impact of tillage practices on invertebrates has been studied and is outside the scope of this dissertation (Stinner and House, 1990; Mirsky *et al.*, 2012; van der Laat *et al.*, 2015).

Table 2. Percent cover of crops, grasslands, forests, developed, shrublands and wetlands; number of land cover blocks, and average size of block (adapted from Taylor *et al.* 2009).

	% Cover		Number of Fields		Average Size (ha)	
Land Cover	1950's	2000's	1950's	2000's	1950's	2000's
Crop	62	59.6	313.3	165.9	7.4	13.8
Grassland	11.4	6.7	127.4	82	3.5	3.1
Forest	12.4	16.2	55.0	55.1	9.3	12.9
Buildings	4.6	8.5	50.0	42.4	3.3	7.0
Shrubland	2.2	0.5	22.3	10.1	3.4	2.0
Wetland	0.1	0.1	2.3	6.0	1.9	0.4

Vegetation Management. Vegetation management of both linear corridors and large blocks of land as wildlife habitat includes grazing, haying, mowing and prescribed fire. Often management techniques are selected for practical considerations such as controlling vegetation height as a traffic safety issue or fires being conducted under specified conditions of soil moisture and wind speeds with the proximate goal of averting succession or invasion of non-native species. The goal of much management in both the US and Europe, however, is to preserve or enhance biodiversity or a specific habitat type through management of the vegetation. Including biodiversity as well as other considerations requires an understanding of what management factors are less than optimal as well as those that can enhance ecological success.

Invertebrate management. Invertebrates make up the largest proportion of the biodiversity (MacArthur and Wilson, 1967). They play a major role in ecosystem services in agricultural systems as well as natural systems (Kellert, 1993). Attitudes cover a broad range from the need to protect commodity crops from pest depredation to focus on wider environmental costs of reducing complex ecosystems to simple ecosystems and removal of large parts of the food web or introduction of exotic species as pest control measures (Van Lenteren *et al.*, 2006). Current management in the agricultural landscape is focused on the fields. A more holistic approach would include the rest of the landscape. Refining common management practices within the agricultural landscape can help meet conservation goals within the broader ecological context. In this way, we offer land sharing techniques as an enhancement within an intensive agricultural system rather than a dichotomous choice.

As mentioned earlier, in the Midwest the crops are mostly genetically modified corn and soybeans. Application of herbicides, insecticides, and fertilizers and most drastic of all, fall harvesting virtually guarantees monospecific plant communities. Such simple communities offer a very limited number of ecological niches to be occupied. They are also drastically disrupted periodically by cultivation. Comparative studies of invertebrates inhabiting crop fields indicate that the most abundant species have high dispersal abilities (Young and Edwards, 1990). Since crop plants are removed at the end of the growing season, the invertebrates that repopulate the fields the following year are assumed to emigrate from adjacent habitats (woods, pastures, fencerows, etc.). Thus, crop fields are often inhabited chiefly by species that have superior dispersal powers and the ability to adapt to a highly artificial managed environment that is quite different from the natural ecosystems to which they have been fitted by a long history of natural selection (Hunter and Price, 1992).

Inventories of Illinois invertebrates are often restricted to 1) pest species such as soybean aphids (Tinsley *et al.*, 2012), emerald ash borer (Herms and McCullough, 2014), or gypsy moths (Manderino *et al.*, 2014); pollinators (Marlin and LaBerge, 2001), or rare butterflies (Panzer *et al.*, 1995). There are some long-term studies that monitor small areas and measure weather variables (Kendeigh, 1979; Marlin and LaBerge, 2001). In 1997, a statewide monitoring program (Critical Trends Assessment Project) was initiated to determine long term change in the biota of Illinois. As part of this effort collections of terrestrial insects are made. Unfortunately, due to funding constraints only a small portion are classified to species.

Research questions

As stated before, the aim of this thesis is to look at vegetation management within the agricultural landscape and determine which practices are most beneficial to invertebrate assemblages and the associated food web. I concentrate on invertebrates for several reasons. First of all, the richness of invertebrates is in itself a valuable aspect of biodiversity that has not yet received the due attention in the American agricultural landscapes. Secondly, invertebrates may play a crucial role in the ecological functioning of ecosystems that in the end may also be important for agriculture (Bengtsson, 1997; Weisser and Siemann, 2004.). Thirdly, invertebrates are an important source of food for birds that are highly appreciated inhabitants of the agricultural landscape. Measuring species richness and abundance is one means of quantifying ecosystem responses to conservation practices. Documentation of individual taxa can provide critical information about the impacts of conservation programs on ecosystem functioning. My study concentrates on the Midwest agricultural landscapes in central Illinois, being a typical USA landscape that might profit from a sharing strategy.

My general question is: How do different land sharing management practices in agricultural landscapes impact invertebrate assemblages and availability of food items during the avian breeding season? We used pitfall traps, sticky boards, and sweep netting to sample invertebrates under various management conditions.

More specifically, we focused on answering the following questions:

1) **How does mowing regime of agricultural roadsides impact invertebrate assemblages?**

2) **How does extreme earth-moving impact the invertebrate community in a newly created prairie restoration?**

3) **How does a mid-summer wildfire impact a grassland invertebrate community?**

4) **How do the invertebrate assemblages in agricultural fields and edges relate to local and landscape complexity?**

5) **How does the invertebrate population relate to food availability, particularly for birds during the breeding season?**

Research Design and Statistical Analysis

From the above summary of ecological theory it is clear that any study of local species assemblages should take the surrounding landscape into consideration. In this study, the effect of landscape on the local invertebrate assembly is the main focus of the two studies trying to answer the last two research questions. Here, the research design was the selection of ten study locations in different landscapes within three different regions (counties). The studies that tried to answer the first three research questions focused on the local effects of local management. For these studies, I made use of given situations: a road of which the edges were mowed according to different regimes, a prairie restoration project of which I was asked to assess the effects on the invertebrates and a wildfire that happened to occur in a restoration area to which I had access. None of these studies were properly designed to test the effect of an experimental treatment (Hurlbert, 1984). In spite of these limitations, I present these studies in this thesis because they give first-hand insight of potential effects of a sharing strategy. But these insights can only be preliminary and do not prove the applicability of a sharing strategy for the Midwest of the USA. For that, properly designed large-scale and long-term research is needed.

Independency of observations. A statistical problem that comes with the study of given situations is that the data collected cannot be regarded as independent observations. The way I dealt with this situation was to add "random effects" to most of the statistical models that were used for analysis. I often applied General Linear Mixed Models (GLMM's) with random effect variables for location, year and sample method, the obvious sources of dependency. This allowed us to resolve the non-independence by assuming a different 'baseline' value for each sample. We then modeled the effect of the other variables of interest, such as the 'treatment', on the difference of the observed value and the 'baseline' value. Thus, the non-independence of data was resolved statistically with the mixed model (Winter, 2011).

Likelihood Ratio Testing. A problem with applying GLMM's is that the classical tests for the significance of the differences between treatments or other categories of interest, such as t- or F-tests, can no longer be applied because the number of degrees of freedom have changed in an unknown way by correcting the residuals for the random effect variables. A solution for this problem is the comparison of the fit of the GLMM in which the variable of interest is included with that of the GLMM where the variable is left out: The Likelihood Ratio Test (LRT). When the GLMM's are fit using a maximum likelihood approach, the difference between the log(likelihood) of two models follows a chi-square distribution with the difference in number of degrees of freedom of the two models as one degree of freedom (Winter, 2011). In this way, the LRT is a means of attaining p-values of the effect of a variable of interest, a method that I will apply in a large number of cases.

Multiple working hypotheses. As an alternative for traditional evaluation of effects of variables thru p-values, I applied an approach based on multiple working hypotheses in a number of cases. The method of multiple working hypotheses was developed in the 19^{th} century by a geologist named Thomas Chamberlin (Rosen 2016, Chamberlin 1965). There has been a recent trend to resurrect this method as a means of addressing the complicated issues in ecological field work (Elliott and Brook 2007; Burnham and Anderson 2000; Rosen, 2016). Rather than using p-values for null-hypothesis testing of individual models, we used a model-selection technique introduced by Akaiki in the 1970's (Burnham and Anderson, 2011). Akaiki's information criterion (AIC) selection methods determine which set of models best explain the data collected (Elliott and Brook 2007; Burnham and Anderson, 2000). Burnham and Anderson (1998) suggest that models having ΔAICc (difference in AICc scores) within 1–2 of the best model have substantial support. Models within about 4–7 of the best model have considerably less support, while models with ΔAICc > 10 have essentially no support. This method of analysis is well matched to the field of ecology with the multitude of variables and degree of uncertainty in field work (Agresti and Kateri, 2011; Stephens *et al.*, 2005). An additional benefit is that some models which clearly do not fit the data can be eliminated and new models introduced during later research projects.

Outline of the thesis

This thesis is composed of this introduction (Chapter 1), 6 research chapters (Chapters 2-7) and a general discussion (Chapter 8).

In chapter 1, I introduce general background information, terminology, and statistical analysis.

In Chapter 2, we conducted a preliminary study to compare the influence of roadside management regime on biodiversity along a roadside with neighboring fields planted in no-till agriculture or land enrolled in a conservation set-aside program. Two of the management regimes are common in Illinois: mowing twice a year and regular mowing throughout the growing season, both leaving the clippings where they fall. The third regime was regular mowing and removing the clippings.

In Chapter 3, we looked at a relatively new restoration project with varied management including fire and regular mowing. We took the opportunity offered by a restoration project associated with a large-scale housing development in central Illinois to survey invertebrates in three phases of plant restoration that were part of a larger project. This cross-sectional study looked at invertebrate assemblages at two, four and five year's post-restoration.

In Chapters 4 and 5 we looked at the immediate and long-term results of an accidental wildfire that burned a hundred-hectare restoration of forest and grasslands. This was an unusual opportunity to study the effects of an unplanned fire that occurred mid-summer during a drought year. The study began 10 days post-fire and traced the re-establishment of invertebrates over a 3-year period. In Chapter 4 we looked at the immediate response of Lepidoptera to the flush of spring-like vegetation immediately following the fire. In Chapter 5 we looked at the long-term impact of the same wildfire on invertebrate assemblages in the burned and unburned areas of the same field for three growing seasons post fire.

In Chapter 6 we looked at the factors influencing invertebrate taxonomic richness and diversity in fields and edges at both local and landscape scales within the agricultural landscape. We sampled invertebrates in ten fields in each of three counties in central Illinois and measured local and landscape parameters that the literature has shown to influence invertebrate richness and diversity.

In Chapter 7 we looked at invertebrate availability for birds early in the breeding season as it relates to structural complexity at the local and landscape levels in three counties in central Illinois. We looked at linear agricultural areas as an opportunity to provide food for nestlings.

In Chapter 8, we discussed the results of the previous chapters and explored possible management recommendations and suggestions for further study as a result of our investigations.

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