Importance Of Agricultural Systems As Multifunctional Landscapes

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IMPORTANT OF AGRICULTURAL SYSTEMS AS MULTIFUNCTIONAL LANDSCAPES

A Dissertation Presented

by

Flavio Sutti

to

The Faculty of the Graduate College

of

The University of Vermont

In Partial Fulfillment of the Requirements
for the Degree of Doctor of Philosophy
Specializing in Natural Resources

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ABSTRACT

Agricultural landscapes provide our society with many benefits. While food production is the primary role of these landscapes, sociocultural and ecological benefits are also provided. However, the full scope of benefits that we obtain from agricultural landscapes are not always taken into account, and with the intensification of agricultural activities, more complex multifunctional landscapes are converted into simpler and less-functional landscapes. I used a heterogeneous agricultural landscape, the Champlain Valley of Vermont, as a case study to examine the interactions between landscape structure and the provision of landscape functions and services.

I analyzed sociocultural and production functions indices obtained via standardized landowner surveys, and ecological function indices collected in the field for 51 plots. Plots were clustered into landscape composition categories (forest, mixed and agriculture), and configuration categories (simple and complex).

I identified a tradeoff between the production and ecological function in agricultural landscapes. When a rural landscape was managed for intensive agricultural production, ecological benefits decreased. Landscapes with diversified land use/land cover and heterogeneously distributed elements returned the greatest number of benefits. Agricultural areas that comprise between 30 and 45% of the landscape can prevent the loss of ecological benefits while retaining high production.

I evaluated the importance of treed habitats in agricultural landscapes in maintaining biodiversity. I related landscape metrics to ecological function indices obtained from fine-grained land use/land cover maps. Metrics obtained from fine-grained maps more accurately predicted the abundance of edge tolerant birds than those obtained from coarse grained maps.

I also explored the importance of small treed landscape elements for common breeding birds and evaluated the convenience of monitoring nests comparing temperature loggers to direct observations. More heterogeneous landscapes, rich in small treed elements, supported a greater density of nests. Nests located on small treed elements in agricultural landscapes were as successful as nests located in large landscape elements.

These analyses deepen our knowledge about the relationship between landscape structure and function, facilitating the evaluation of the functionality of heterogeneous agricultural landscapes.
CITATIONS

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CHAPTER 1: LITERATURE REVIEW

1.1. Introduction

Agricultural landscapes are important for the production of goods vital to human life, but more recently are also being recognized as important for other services (Robertson and Swinton 2005). Biodiversity conservation, habitat for wildlife, biological control of pests, nutrient cycling, water quality control, pollination, aesthetic values, recreation, and preservation of rural and cultural traditions (Arriaza et al. 2004, Fleischer and Tchetchik 2005, Swinton et al. 2006), are some examples of additional values produced by agricultural landscapes. However, only the production function and a few other functions (i.e., regulation) can easily provide direct economic benefits to landowners (de Groot et al. 2002). Other services, even if known, are often discounted because of the difficulty in attributing a market valuation (Robertson and Swinton 2005, de Groot 2006). Perhaps most importantly, the production function of agricultural systems, especially those that are intensively managed, can conflict with ecological and sociocultural functions offered by these systems, diminishing the number and quality of the other services that they offer, including reduced productivity, soil erosion, decreased water quality, and decreased biodiversity (Forman 1995). Understanding how the diverse functions of agricultural systems are interrelated has the potential to offer insights for balancing the production function with the ecological and sociocultural functions of these systems.
Traditionally, researchers have studied ecosystem services and functions in natural ecosystems. Recently however, this approach has been broadened to include managed landscapes such as urban and agricultural systems, and the importance of these systems in providing services beneficial to both natural and human communities (Bills & Gross 2005; Groenfeldt 2006). Heterogeneous agricultural landscapes, where managed agricultural land is interspersed with natural or semi-natural habitats (small patches of woods, hedgerows, isolated trees), may offer a combination of services: production of food and fiber, but also sociocultural (spiritual, recreational, and aesthetic), and ecological (biodiversity, water conservation, carbon sequestration) functions that require more study (Kareiva et al. 2007).

The influence of landscape structure (or the composition and configuration of ecosystems that constitute a landscape), on plant and animal wild populations has been studied extensively (Donovan and Strong 2003). However, the relationship between landscape structure and economic and sociocultural functionality is largely unknown. For example, the directionality of drivers in the landscape-organism relationship can have a profound effect on both human and ecosystem health. Humans, like other living organisms, can be influenced in their settlement or land use patterns by the composition and configuration of the landscape. In some cases, the opposite might be true, as humans work to shape the landscape to fit their needs and aesthetic values. The functions provided by the landscape are the results of the interactions between the historical, cultural and technological human interventions and the ecological processes that are at play in the environment (Baudry et al. 2000). These functions provide ecosystem services.
that can have a positive or negative effect on the quality of life and on environmental integrity (Dale and Polasky 2007)

Ecosystem studies tend to analyze only one functional dimension and rarely integrate production and sociocultural functions with ecological services (Carpenter et al. 2006). Today, with GIS and the maturing field of landscape ecology, it is possible to explore the influence of landscape structure on production, ecological and sociocultural functions simultaneously, as combinations of factors contributing to multifunctionality (Strong et al. 2005). In this study I will explore the structure of a heterogeneous agricultural landscape, the Champlain Valley of Vermont (CV), and analyze its correlation to production, sociocultural and ecological functions through remote sensing, ecological field data, and landowner interviews. Below, I review some of the literature that has contributed to our understanding of landscape multifunctionality.

1.2. Agricultural landscape functions

Systems dominated by intensive agriculture frequently support diminished biodiversity and impoverished ecosystem services (McLaughlin and Mineau 1995). However, when intensively managed habitats are interspersed with more natural habitats and less intensively managed or set aside habitats, biodiversity and healthy ecosystems can be maintained (Benton et al. 2003). “Health” is not only measured in the number of species of wild animals and plants that the area can support, but also through services such as air filtration, micro-climate regulation, noise reduction, water control, soil retention, recreation, and aesthetic functions that enrich human quality of life.
Our means of measuring health has been to assess ecosystems across a range of functions. In this way, agricultural systems are defined as multifunctional when, in addition to the primary role of food production, one or more functions are added. In particular these added functions refer to environmental and sociocultural benefits (Rossing et al. 2007). Especially in Europe and Asia, multifunctionality of agricultural systems has been studied extensively, but this concept has received less attention in the U.S. (Groenfeldt 2006). Because of the delay in the U.S. to include “nonfood” services as a management objective for agricultural landscapes (Bills and Gross 2005), and in recognizing the importance of multifunctionality of these landscapes in the sustainability of rural areas (Groenfeldt 2006), there is still a great need for research. Identifying, understanding and valuing the many ecosystem services of multifunctional agroecosystems will require a joint effort from many fields (ecology, economy, sociology), new policies and public education (Robertson and Swinton 2005).

Addressing multifunctionality in agricultural landscapes needs to be pursued by assessing all functions jointly because of possible interactions between them. In fact, the capability of a landscape to provide goods and services is not the mere sum of all functions, but it is also influenced by trade-offs, synergies and non-linear relationships that may exist between functions (Willemen et al. 2010). In addition, production, economic, sociocultural, and ecological functions together influence the landowner's decisions in managing their property and thus affect the landscape in which the property is embedded (Baudry et al. 2000).
Agricultural systems are not only providers of ecosystem services (food, recreation, aesthetic), but they also take advantage of the services provided by the landscape in which each patch is embedded. The “natural” features (hedgerows, forest patches, wetlands) in an agricultural landscape can provide, for example, habitat for pollinators and contribute to the attenuation of negative effects associated with agricultural production such as the release of pesticides and fertilizers (Swinton et al. 2007).

To understand the influence of landscape structure on ecosystem functions, it is necessary to have some sort of scale by which functionality can be measured. Because the number of performance indicators of ecosystem function are essentially infinite, below, I present my rationale for the inclusions of ecosystem function indicators for each of the three categories: ecological (environmental), sociocultural (social), and production (economic) (Rossing et al. 2007).

1.3. Ecological function

Population declines and loss of biodiversity have been connected with habitat fragmentation, degradation, and loss (Fahrig 2003). However, most of the studies conducted on this relationship have focused on single systems (e.g., forested habitat or grasslands), often at the individual patch level (Fahrig 2003). The relationship between biodiversity, ecosystem services and agricultural landscapes has also been studied thus far mostly at the patch/habitat scale (Swift et al. 2004). However, in agricultural systems in Europe, biodiversity was connected with landscape-scale variables (Dormann et al. 2007).
2007), and the promotion of land use heterogeneity at the landscape or farm scale has been hypothesized to offer the best chance of maintaining ecosystem services and biodiversity (Swift et al. 2004).

The main objective of my study is to understand the relationship between landscape composition and configuration and a series of selected biodiversity metrics. Because there are many groups of organisms that perform and provide ecosystem functions and services (Swift et al. 2004), I chose groups that are sensitive to the changes in landscape structure and are fairly simple to measure and analyze. I limited my research to two taxa, birds and woody plants, and to the analysis of the health of riparian habitats (Sweeney et al. 2004).

The effect of landscape structure on forest bird communities has already been largely investigated (Robinson et al. 1995, Rodewald 2003). Similarly, the influence of landscape structure on grassland birds has been studied (Shustack et al. 2010). In both systems, decreasing patch size, increased patch isolation and intensification of management has resulted in a decrease in bird species richness. I will focus on heterogeneous landscapes to determine how variation in landscape composition and configuration affects bird species richness and abundance. Another metric that I will quantify is the reproductive success of birds, which will be obtained by monitoring their nests. Landscape configuration influences the effect of predation and parasitism on nest success (Robinson et al. 1995). Thus, reproductive success will provide an indication of the long term viability of the bird species/groups in the studied landscapes (Donovan et
Bird communities will be studied by collecting data using point count surveys, transect surveys, and nest monitoring.

The effect of fragmentation on plant communities has been studied (Dzwonko and Loster 1988, Drinnan 2005), but the scale at which fragmentation influences plant species richness has not yet been widely analyzed (Stiles and Scheiner 2010). I will collect data on woody plant abundance, richness, and biomass to investigate relationships of these metrics to landscape structure.

With land conversion to agriculture and agricultural intensification, natural and forested riparian habitats have been reduced in size and have become more isolated (Deschenes et al. 2003). The loss of riparian buffers not only negatively influences the aesthetic value of a landscape (Anbumozhi et al. 2005), but more importantly is connected to increased erosion and sedimentation, diminished ability to capture nutrients and contaminants before entering the water system, and decreased quality of wildlife habitat (Strong et al. 2005). Because water systems are intimately connected with the surrounding landscape and offer important functions that influence both human and natural communities, I will focus my attention on riparian habitats in the selection of research sites. In addition, for the selection of my study areas, I have the advantage of using a detailed spatial database of digitized habitats for riparian buffers that was recently produced by the spatial analysis lab at the University of Vermont. I am using the Rapid Geomorphic Assessment (RGA) score as a metric of stream channel integrity because it has been correlated to biological stream quality indicators and can be used to evaluate the ecological integrity of freshwater lentic bodies (Sullivan et al. 2004). The RGA score
is a quick method to survey a reach of a stream/river and collect information on the channel adjustment subsequent to changes induced by natural causes or human activities to the floodplain and channel condition. Each stream/river reach receives a score for its quality that can be used to verify relationships with the landscape structure of the riparian area (VTDEC 2001).

1.3.1. Landscape structure and bird communities

The effect of landscape structure on changes in bird communities has been investigated for forest (Robinson et al. 1995; Rodewald 2003) and grassland species (Perlut 2007, Shustack et al. 2010). In both systems, decreased patch size, increased isolation, and intensification of management has resulted in decreased bird species richness. However, heterogeneous landscapes that include both forested and agricultural patches have not been thoroughly studied, or if quantified, the effects of the relationship on biodiversity and landscape pattern were inconclusive (Wretenberg et al. 2010, Fahrig et al. 2011).

In agricultural landscapes, wooded habitats such as hedgerows, shrub patches and small clumps of trees provide birds with nesting, roosting, cover, and foraging areas (Hinsley & Bellamy 2000). The spatial arrangement of these wooded habitats in the landscape, their management, size and structure influence their value for birds. Hedgerow size (height, width and volume) and abundance of trees has been positively correlated with bird species richness and abundance as well as the presence of vegetative cover in and around the base of the hedgerow (Hinsley & Bellamy 2000; Batáry et al.
2010). In contrast, some authors (Major et al. 1999, Estrada et al. 2002), provide support for the idea of hedgerows as ecological traps. However, the effects of hedgerow type, structure, tree/shrub density, and species composition have not been well studied (Zuria et al. 2007). For example, there are only a few studies on nest predation in hedgerows and wooded linear strips; most nest predation studies are conducted with artificial nests and do not include data on nest success in different type of hedgerows in relation to natural habitats (Zuria et al. 2007). Wooded hedgerows in agricultural landscapes also provide socio-economic and ecological functions. Hedgerows can be important in maintaining species that rely on woodland and scrub when the landscape is converted to annual crops. They are also important in providing connectivity between remnant patches of natural vegetation, defining property lines, reducing erosion, wind speeds and water loss, and providing firewood and wild edibles (Zuria et al. 2007).

1.3.2. Capturing fine-grained landscape structure

An understanding of the relationship between bird diversity and landscape structure in agricultural systems at a fine-grained spatial scale is critical to developing wildlife-friendly management activities in agricultural landscapes. However, even if hedgerows and small natural and semi-natural (planted trees, small clump of trees etc.) landscape elements are recognized as important in rural systems, until recently the capacity to identify and record their presence on maps was limited by the high costs of digitizing (both in terms of money and time) and the lack of high resolution imagery. These deficiencies hindered the possibility of conducting comprehensive studies on the
influence of such elements on the diversity and breeding success of birds at a fine spatial scale. Now, the spatial capability of high resolution satellite images and advances in processing software allow for a more accurate mapping of these wooded elements (Vannier and Hubert-Moy 2010). However, in heterogeneous agricultural landscapes, large patches are interspersed with small units of potential habitat for birds (hedgerows, individual trees, clumps of trees, shrub), whose size can be <1 pixel. Even if the size of these objects is >1 pixel, automated classification procedures that use only spectral content (amplitude and number of bands associated with each pixel) may result in omissions because of the heterogeneity of the pixels surrounding the object. Development of object-oriented classification methods are able to identify small and large objects using both spectral and spatial (context and texture) information, overcoming the issues associated with pixel heterogeneity (Mathieu et al. 2007; Vannier & Hubert-Moy 2010; Zhang & Zhu 2011). In addition, the use of object-oriented versus pixel based classification increases the accuracy of the resulting land cover map (to > 90%), while preventing the artificial creation of small, non-existent objects (Vandersande et al. 2003; Perea et al. 2010).

The Lake Champlain watershed, and in particular the Champlain Valley of Vermont (CV), is a heterogeneous agricultural landscape. For this area a coarse land cover classification is already available (O'Neil-Dunne 2001). The availability of these data and of recently acquired detailed orthophotographs (Farm Service Agency (FSA) 2010), make this area a perfect candidate for the generation of fine scale land use/land
cover classification to better understand the effects of landscape structure on plant diversity, bird richness and breeding success.

1.4. Sociocultural and production functions

The demand for services from agricultural areas has increased with the increased demand for space and recreation alongside the ever-increasing need for goods production (Willemen et al. 2010). In rural areas, the impact of intensive agricultural activities influences both the environment and socio-economic conditions. Studying the impact of agricultural production on other functions provided by rural agricultural landscapes should be a priority for the sustainable development of these areas (Wiggering et al. 2003). However, the analysis of both sociocultural and production functions and their connection to landscape structure has not been thoroughly investigated (Strong et al. 2005).

To identify specific social, cultural and production functions connected with the landscape of the Champlain Valley of Vermont, I will administer surveys to landowners included in each study plot. Sociocultural and production values will be obtained from these surveys by combining responses into categories (i.e., production value from the agricultural portion of the property, production value from the forested portion of the property, recreation value of property, conservation value of property). Additionally, the survey answers will be used for a more qualitative understanding of the relationship between function and landscape structure.
1.5. Literature cited


CHAPTER 2: ENHANCING MULTIFUNCTIONALITY IN AGRICULTURAL LANDSCAPES

2.1. Introduction

Cultivated systems occupy over 25% of the Earth’s terrestrial surface (Millennium Ecosystem Assessment 2005). Through increased yields resulting from greater inputs of fertilizer, water, pesticides, and new crop strains, agricultural production continues to provide greater per capita rates of caloric production (FAO 2012). However, more efficient production systems combined with a greater land area under production have led to significant losses of natural habitat and alterations in the cycles of nitrogen, phosphorous, carbon, and water. These changes have led to stresses on the functioning of natural systems leading to degradation and unsustainable use of the planet’s resources.

However, a myopic focus on the adverse effects of agricultural commodity production ignores the non-commodity functions provided by farms. Agricultural land can provide ecological and sociocultural functions which are ignored by focusing on the production function. This concept of agricultural multifunctionality – that agricultural systems can provide an array of outputs and benefits, or “functions”, in addition to plant or animal products for food, fuel and fiber – has been gaining increasing attention in agricultural, environmental science, and policy circles (Renting et al. 2009). These functions can be linked to ecosystem services, categorized into ‘provisioning services’ (e.g. water and food); ‘regulating services’ (e.g. climate and flood regulation); ‘cultural services’ (e.g. aesthetic or recreational benefits); and ‘supporting services’, (e.g. nutrient
cycling) (Millennium Ecosystem Assessment 2005). We reclassified the services provided by agricultural systems under three functional axes: 1) production function, including the provisioning services; 2) ecological function, comprised of both regulating and supporting services; 3) sociocultural function, including cultural services.

Research surrounding the application of multifunctional agriculture has predominantly occurred in the European Union, with much slower adoption in the U.S. (Boody et al. 2005). However, recent trends in U.S. agricultural policy show a shift towards recognizing the importance of maintaining environmental and human health in rural areas, in addition to commodity production. This opens the possibility for exploring the potential of the multifunctionality concept as a framework to guide agricultural policy and management in U.S. rural landscapes. Despite this promise, there has been limited research into the application of multifunctionality assessment and its integration into management or policy frameworks.

Quantifying multifunctionality is difficult as a result of variation in the scale at which assessments are conducted as well as the nearly limitless array of functions or services that can be assessed. As such, there has been significant debate regarding what functions are provided, to whom are they provided, and how can functionality be improved. In this study, we applied a landscape ecology framework as a means to quantify the effects of landscape structure to more efficiently understand how agroecosystems enhance ecological, production, and sociocultural functionality. Landscape structure describes how landscape elements are spatially related and is defined by two aspects: 1) landscape composition or the proportion of element types in the
landscape, and 2) landscape configuration or the spatial arrangements of the landscape elements (Leitao et al. 2006).

Although there is overwhelming support for the influence of landscape composition and configuration on within-patch dynamics (e.g., patterns of species distribution and abundance), the application of landscape ecology to ecosystem services has been less well-studied, particularly in the context of agriculture. As we face continued degradation of ecosystem functionality, it will become necessary to consider where and how to prioritize the conservation and restoration of ecosystem services. As complete inventories of ecosystem services will be prohibitively expensive, landscape metrics may provide an efficient means to locate leverage points for management activities to maintain ecosystem functions. We used remote sensing, ecological field data, and landowner interviews to obtain landscape metrics and function indicators. We obtained landscape metrics from land use/land cover (LULC) maps, ecological function indices (bird and plant richness, stream channel integrity) by objectively measuring variables in the field, production function indices (cropping systems, alternative production functions of forest elements), and sociocultural function indices (aesthetic, recreation, ethical) by interpreting the answers from surveys administered to landowners. Because there are many groups of organisms and indices that can be tied to the provision of ecological functions and services (Swift et al. 2004), we limited our research to indices obtained from two taxa, birds and woody plants, and by measuring the stream channel integrity. These indices are sensitive to the changes in landscape structure, are fairly simple to measure and analyze, and are tied with the provision of multiple services (Sweeney et al.
Similarly, we limited the number of questions in the landowner survey to maintain interviewee interest and engagement for a reasonable amount of time. These multifunctionality indices were by no mean exclusive, but provided a framework for assessing production/non-market tradeoffs by looking at a few specific functional indices.

Heterogeneous agricultural landscapes, where managed agricultural land is interspersed with natural or semi-natural elements (small patches of woods, hedgerows, isolated trees), may offer a combination of services: production of food and fiber, but also sociocultural (spiritual, recreational, and aesthetic), and ecological (biodiversity, water conservation, carbon sequestration) functions that require more study (Kareiva et al. 2007). We undertook an empirical study of the effects of landscape composition and configuration on the functionality of agroecosystems across a heterogeneous region in the northeastern U.S. The objectives of this study were to: 1) determine the effect of landscape composition and configuration on the functionality of agricultural landscapes across the production, ecological, and sociocultural functional axes, and 2) explore the tradeoffs between functions. We predicted that uniform homogeneous landscapes would provide less functionality than complex heterogeneous landscapes. Also, uniform homogenous landscapes should have high functionality along only one of the axes, while complex heterogeneous landscapes should have high functionality along multiple axes.
2.2. Methods

The Lake Champlain watershed is a 2,132,600 ha region in eastern North America that surrounds Lake Champlain and is divided between the states of Vermont and New York and the Canadian province Quebec. The Lake Champlain basin extends into the Green Mountains to the west, the Adirondack Mountains to the east, and drains northward into the Saint Lawrence River Valley. The focus area for this study (Figure 1, red outline) was located in the Vermont portion of the basin and concentrated in the Champlain Valley of Vermont (CV), with a few study plots located in the western Green Mountains. The central part of the CV has the greatest population density and includes the largest city in Vermont: Burlington (population 42,417, 2010 census year). The northern and southern portions of the CV are made up of agricultural areas embedded in a matrix of forested land. Deciduous and mixed coniferous-deciduous forest patches are scattered within the agricultural patches in the western portion of the CV and are the dominant aspect of the eastern part of the landscape and in the Green Mountains. An extensive road and river network is present throughout the CV.

In Vermont, the topography of the CV is dominated by low to moderate elevations closer to Lake Champlain; elevations increase from west to east toward the Green Mountains. The lowest elevations (30 m above sea level) are found along the shore of the Lake and the highest, 100-200 m, at the base of the Green Mountains. The land use/land cover of the CV is 26% agriculture, 50% forest, 9% urban, 13% lakes and rivers and 2% wetlands (O’Neil-Dunne 2005). Seven major rivers (Missisquoi, Lamoille, Winooski, LaPlatte, Lewis, Little Otter Creek, Otter Creek), drain into Lake Champlain.
Our research was focused on landscapes adjacent to the major riparian corridors in the Lake Champlain watershed. We worked in these riparian landscapes for two reasons. First, given the importance of riparian habitat structure and composition to the integrity of aquatic ecosystems, these landscapes are likely to have a disproportionate influence on the environmental quality of the region (De Palma et al. 1979). Second, we had access to a preexisting spatial database of mapped habitats from the Vermont LULC Mapping Project (O’Neil-Dunne 2005).

For the seven major rivers and their primary tributaries, LULC classes were hand-digitized within an 800 m buffer (1600 m wide) on each side of the river using a minimum mapping unit of 0.1 ha. In total, approximately 135,000 ha of riparian buffer corridors were digitized and classified into 14 categories, roughly corresponding with those of the Anderson et al. (1976) classification scheme. These data were substantially more accurate than those from the 1992 National Land Cover Database or data from satellite imagery obtained in 2002, with substantial improvements estimates of the spatial extent of wetlands, agricultural land, shrub, and urban (O’Neil-Dunne 2005). We collapsed the 14 categories classification into a 6-category classification (agriculture, shrub, forest, urban, water, and wetland), and clipped the digitized buffer to the plots level.

We identified a set of 100 random circular plots centered on the rivers of the CV with the number of plots per river proportional to the river length. Plot centers were at least 1000 m apart from each other (to avoid overlaps), and had a 500 m radius (78.5 ha). We assessed the randomly selected locations to assure that differences in landscape composition and configuration were equally represented, and we visited all plot’s
landowners to ask for permission to work on their property. We selected our final plots’ list from the plots in which we had permission to work while assuring that we had a stratified sample able to represent different composition and configuration landscape categories (Figure 2).

Of the many metrics that measure landscape heterogeneity (Leitao et al. 2006), we identified two easily measurable and interpretable metrics that describe the two components of landscape heterogeneity: compositional and configurational heterogeneity (Fahrig et al. 2011). We used the proportion in the plot of each LULC category as the landscape composition metric and the number of patches in all LULC categories in the plot as our configuration metric. To obtain a landscape composition metric that described the proportion in the plot of each LULC category with a single value, we used principal component analysis (PCA). We used the proportion of the plot occupied by each of the 6 LULC categories as input data for the PCA. We extracted the principal components from the covariance matrix but retained only the first principal component (PCA1) as this axis explained over 60% of variation. PCA1 represented a gradient between forest- and agriculture-dominated plots. PCA1 can be used as an index of evenness: when PCA1 values are close to 0 the composition of the plot is even with no dominant LULC type, when PCA1 values are negative the composition of the plot is dominated by forest, and when the PCA1 values are positive the composition of the plot is dominated by agriculture. Within the 500m radius plots we used the FRAGSTATS v3.4 (McGarigal et al. 2012) fragmentation analysis module to derive the number of
patches value. Number of patches is a simple and easily explainable metric and can be interpreted in terms of landscape configuration and fragmentation (Leitao et al. 2006).

We selected plots to represent two sets of three treatments, one set for landscape composition and one set for landscape configuration. We grouped plots into 3 categories on the basis of their value for PCA1 (composition: agricultural, forest and mixed plot), and number of patches (configuration: complex, simple and mid complexity). We classified plots compositionally on the basis of PCA1 as 1) agricultural plots, in which agriculture is the predominant LULC (between 33 and 82 percent), 2) forest plots, in which agriculture is embedded in a forest matrix (contained between 0 and 21 percent agriculture), and 3) mixed plots, in which the amount of forest and agriculture LULC is similar. We classified plots configurationally on the basis of number of patches as 1) simple complexity plots with less than 23 patches, 2) medium complexity plots with 25 to 37 patches, and 3) complex plots with 40 patches or more. To attribute plots to the different composition and configuration categories, we used Jenks Optimization Method (JOM). JOM is a method used in choropleth mapping to classify plot values, into a set number of categories. Categories were identified such that the difference between the plot value and the plot’s category average was minimized while maximizing among-category average distances (Jenks & Caspall 1971).

For the ecological function, we collected field data on three subfunctions: bird and plant species richness to measure terrestrial ecological functioning, and river channel integrity to measure aquatic ecological functioning. Within each plots, we sampled birds at 4 point count locations. From the center point of the circular plot, points were positioned
250 m in the four cardinal directions. At each bird point, we conducted a standard 10 minute point count survey (Hutto et al. 1986), recording, in intervals of 3 minutes and 20 seconds, all birds heard and seen. From the center point of the circular plot, we also conducted two bird count transects along the river, following the river's edge in opposite directions on both sides of the river. Along each transect, we recorded all birds detected (sight and sound) on the river water and 50 m inland from the river. Bird detection was conducted for 10 minutes, during which the researcher walked using a standardized pace (20m/min) along the river. We used the data from the bird point counts and transects to calculate bird species richness. The bird richness was standardized using rarefaction techniques and calculated using software EcoSim (Gotelli & Entsminger 2010). Point counts and transects were both conducted once per plot.

We quantified plant species richness on 50 m transects starting at the bird points locations and moving in the direction that offered the maximum concentration of woody plants. If points were located in forested habitat, transects were oriented in the north direction. Each woody species ≥3 m in height (tree and shrubs) was tallied if present within 1.5 m on either side of the transect centerline. Only species presence was recorded.

We collected information on the stream channel integrity at the plot center, implementing the guidelines established by the Vermont Rapid Geomorphic Assessment (RGA) and Assessment Field Notes (VTDEC 2001). The RGA score is a metric used to summarize information on stream channel integrity, channel condition, and the channel adjustment subsequent to changes induced by natural causes or human activities to the floodplain (VTDEC 2001). We used the RGA score because it has been correlated to
biological stream quality indicators and it is an efficient tool for evaluating the ecological integrity of rivers (Sullivan et al. 2004).

To assess the production and sociocultural functions, we conducted semi-structured interviews with one landowner whose land was included within the plot (Table 1). The questions were designed to assess functionality of the landscape across a variety of functions and subfunctions. The set of questions in the survey (see supporting information for a full list of questions) addressed demographics (information on the landowner and his/her family, size and time of ownership of the property, type of land cover), livelihoods (information on income from farming, marketable products from farming, clientele, tourism business, products from ancillary activities such as hunting, fishing, firewood, timber, wild edibles), recreation (information on time spent outdoors, public access and trail maintenance), visual quality (information on the importance of aesthetics of Vermont, agricultural landscape and their property, attachment to the landscape aspect), and conservation practices (information on the enrollment in governmental or non-governmental conservation programs, protection of riparian buffers). A series of practice interviews were performed prior to the starting of the project to test the questions and address potential imprecisions or misunderstandings. Interviews were conducted over the phone by two trained interviewers and lasted on average 0.5 hours.

From the survey results we identified dichotomous questions (yes/no answers), that described aspects of either the production or sociocultural functions. We grouped production-related answers into two subfunctions: agricultural “goods” production, and
“non-agricultural” production. We divided the sociocultural function into four subfunctions: wildlife (landowner interviewed allows hunting, fishing or manage its land for wildlife), recreation (landowner spends time recreating outdoors, maintain trails, or allow public access to its property), visual (landowner improves its land for visual quality), and conservation (landowner has enrolled their land in conservation programs or manages their land for riparian areas and native vegetation conservation). We converted the yes/no answers into 1/0 and averaged these values across functions and subfunctions. For the ecological function we identified 3 subfunctions corresponding to the 3 types of data collected: rarefied bird species richness, plant species richness, and stream channel integrity.

Functions and subfunctions differed in the number of questions used to obtain their summary scores, the type of data used, and/or the scale of data used. Thus, function and subfunction values were standardized using a linear scalar transformation so that different numerical scales could be compared on a scale from 0 to 1 (Malczewski 1999). The formula used to standardize values in each plot was:

$$x' = \frac{x - x_{\text{min}}}{x_{\text{max}} - x_{\text{min}}}$$

where $x'$ is the new standardized sub/function value, $x$ is the original sub/function value, $x_{\text{max}}$ is the maximum value of $x$ and $x_{\text{min}}$ is the minimum value of $x$.

We tested for differences in functionality among landscape composition and configuration classes with a non-parametric test (Kruskall-Wallis). We used a non-
parametric test because in our count data (from the interviews) we had a small sample for each subfunction. When we found significant differences among categories, we performed a non-parametric multiple comparison test (Steel-Dwass all pairs), that is comparable to the parametric Tukey-Kramer test. We used regression analysis to compare functions and landscape structure metrics to obtain tradeoff graphs. Statistical analyses were performed using JMP Pro (JMP(R) 2013) and program R (R Core Team 2014).

2.3. Results

We obtained permission to work on 51 plots. The plots were relatively evenly distributed across the three composition (agricultural, mixed, and forest) and configuration categories (simple, intermediate, and complex; Table 2). The exception was agriculturally-dominated landscapes with complex configuration for which we had no plots. Because of the relatively small sample size, we did not analyze the effects of composition by configuration interactions, making this gap in the data less important.

Landscape Composition

Our study plots were characterized by agricultural land cover varying between 0 and 82%. Forest plots included the lowest average proportion of agriculture (11%), mixed plots had an average agriculture cover of 28%, and agricultural plots had the greatest average agricultural LULC (56%). Landscape composition had a significant effect on two of the three functions (Figure 3). The greatest differences were between
agricultural and forest landscapes; mixed landscapes typically showed intermediate values across all functions (Figure 4). Across the three functions, we found significant differences in the ecological (Kruskal-Wallis test $X^2_2 = 10.9, P = 0.004$) and production (Kruskal-Wallis test $X^2_2 = 10.0, P = 0.01$) functions. Forest landscapes showed significantly greater ecological functionality than agricultural landscapes (Steel-Dwass multiple comparison, $Z = 3.1, P < 0.005$), whereas agricultural landscapes showed significantly greater scores on the production function than forest landscapes (Steel-Dwass multiple comparison, $Z = 2.8, P < 0.02$). There was no effect of landscape composition on the sociocultural function (Kruskal-Wallis test $X^2_2 = 0.7, P = 0.70$).

Likewise, we did not find an effect of landscape composition on any of the four sociocultural subfunctions (wildlife, recreation, visual, or conservation; Kruskal-Wallis test, all $X^2_2 < 2.6$, all $P > 0.28$; Figure 4). Within the production function, production from agricultural sources varied by landscape type (Kruskal-Wallis test $X^2_2 = 8.4, P = 0.01$) with agricultural landscapes having greater functionality than forest landscapes (Steel-Dwass multiple comparison, $Z = 2.6, P < 0.03$); for production from non-agricultural sources (Kruskal-Wallis test $X^2_2 = 7.8, P = 0.02$), mixed landscapes showed greater functionality than forest landscapes (Steel-Dwass multiple comparison, $Z = 2.5$, both $P < 0.04$). The increase in the functionality of mixed landscapes between the agricultural and non-agricultural subfunctions is notable in Figure 4. Within the ecological function we found a significant effect of landscape composition on bird and tree species richness subfunctions (Kruskal-Wallis test, both $X^2_2 > 14.4$, both $P < 0.001$) with forest and mixed landscapes showing significantly greater ecological functionality.
than agricultural landscapes (Steel-Dwass multiple comparison, all $Z > 2.7$, all $P < 0.015$). Landscape composition had no effect on the stream channel integrity subfunction (Kruskal-Wallis test $X^2_2 = 1.6$, $P = 0.44$).

**Landscape configuration**

We used number of patches as the metric of landscape configuration. The plots contained between 10 and 59 patches. Simple plots averaged 17 patches per plot, intermediate plots had an average number of patches of 31, and complex plots had an average of 46 patches.

Landscape configuration had a significant effect on only the ecological function (Figure 5). This variation was not consistent across configuration categories as the moderately complex landscapes were more similar to complex landscapes for some functional dimensions and more similar to simple landscapes for others, especially within subfunctions (Figure 5).

We found a significant effect of landscape configuration on the ecological function (Kruskal-Wallis test $X^2_2 = 10.3$, $P = 0.006$; Figure 5) with moderately complex landscapes showing greater ecological functionality than simple landscapes ($Z = 3.2$, $P = 0.004$). However, landscape configuration did not have an effect on functionality on either the production (Kruskal-Wallis test, both $X^2_2 = 1.6$, $P = 0.53$) or sociocultural functions (Kruskal-Wallis test $X^2_2 = 2.4$, $P = 0.29$). Within the ecological function, moderate and complex landscapes showed greater functionality scores than simple
landscapes (both $Z > 2.9$, both $P < 0.009$) for both bird (Kruskal-Wallis test $X^2_2 = 9.0$, $P = 0.01$) and plant species richness subfunctions (Kruskal-Wallis test $X^2_2 = 10.3$, $P = 0.006$). Landscape configuration had no effect on the stream channel integrity subfunction (Kruskal-Wallis test $X^2_2 = 1.6$, $P = 0.44$). Within the production function, landscape configuration had only a marginal effect on agricultural production (Kruskal-Wallis test $X^2_2 = 5.7$, $P = 0.06$) and no effect on non-agricultural production (Kruskal-Wallis test $X^2_2 = 0.9$, $P = 0.64$). Within the sociocultural function, the greatest effects of landscape configuration were on the visual quality (Kruskal-Wallis test $X^2_2 = 6.3$, $P = 0.04$) and recreation subfunctions (Kruskal-Wallis test $X^2_2 = 6.9$, $P = 0.03$). On the recreational subfunction, complex landscapes showed significantly greater functionality than moderately complex landscapes ($Z = 2.4$, $P = 0.04$) and on the visual subfunction, complex landscapes showed significantly greater functionality than simple landscapes ($Z = 2.5$, $P = 0.03$). There was no effect of landscape configuration on wildlife or conservation subfunctions (Figure 6).

**Tradeoffs**

We found significant relationships between percent agriculture in the landscape and the ecological (negative, non-linear; $F_{2,48} = 10.1$, $P < 0.001$; Figure 7) and production (positive, linear; $F_{1,49} = 8.9$, $P = 0.004$) functionality (Figure 7). Ecological functionality remained relatively constant until agricultural land made up ~45% of the landscape at which point ecological functionality declined fairly rapidly. This suggests that there are tradeoffs between the ecological and production functions as a result of changes in
landscape composition. By contrast, production functionality increased linearly with proportion of agriculture in the landscape. There was no relationship between percent agriculture in the landscape and sociocultural functionality. Only ecological functionality showed a significant (positive) relationship with number of patches (Figure 8). Although not significant, production functionality tended to decrease and sociocultural functionality tended to increase with increasing number of patches.

2.4. Discussion

Landscape functionality varied with both composition and configuration metrics. In general, we found a greater effect of landscape composition on production and ecological functions and a greater effect of landscape configuration on the sociocultural function.

The mixed composition category behaved like a “hybrid” land cover type between agriculture and forest dominated landscapes. For mixed landscapes nearly all subfunctions values were intermediate between forest and agriculture landscapes. The exception was in the production function: mixed landscapes scores in the “agricultural production” subfunction were as low as in forest landscapes, and mixed landscape scores in the “other production” subfunction were as high as the scores for agricultural landscapes. Interestingly, species richness of birds and plants for mixed landscapes were very close to forest landscape scores although the average tree cover in forest plots (57%) was substantially greater than in mixed plots (37%).
Forest- and agricultural-dominated landscapes are not only different visually, but they also differ in terms of multifunctionality. Forest landscapes had greater scores within the ecological function and lower scores in the production functions, while agricultural landscapes scores showed the opposite pattern. These results were supported by the nonlinear tradeoff between production and ecological functions with change in landscape composition. As agricultural land cover became more dominant, goods production increased and ecological functionality decreased. A small amount of agricultural coverage in forest landscapes increases ecological functioning by providing habitat diversity. Only when agricultural coverage exceeded 30% of the landscape did ecological function start to decline in favor of agricultural production.

Complex landscapes, which are characterized by a large number of patches, scored fairly high in many subfunctions. Particularly, for 3 out of 4 social subfunctions complex plots had the highest multifunctionality scores. These landscapes seem to offer greater prospects for recreational and aesthetic appreciation opportunities possibly because a varied use of the landscape is facilitated by the presence of heterogeneously distributed land covers. Also, people prefer heterogeneous landscapes where they can visually perceive wild areas, wooded buffers, variety of land covers, and water features (Arriaza et al. 2004; Morse et al. 2014; Klein et al. 2015).

The scale at which our analysis was performed (1000 m² resolution) did not have a significant influence on the composition metrics. However, the number of patches in our landscape was highly influenced by the resolution of the maps used. Many of the small landscape elements that characterize a landscape as heterogeneous (hedgerows,
small clump of trees, single trees, etc.) are obscured at coarse spatial resolutions, such that the complexity of agricultural plots are likely more sensitive than forest and mixed plots to the scale of the LULC maps utilized. The heterogeneity of traditional cropped land can potentially be captured at coarse scales if the fields are large and the sensor used to capture the imagery is able to detect differences in vegetation. However, if the resolution of the LULC used is not sufficiently fine to identify single trees, hedgerows, or small shrubby areas that enhance the complexity of intensively managed agricultural landscapes could result in the perception that heterogeneity is diminished.

Because mixed plots have a high level of multifunctionality with high scores in most subfunctions, we recommend maintaining the amount of agriculture in the landscape between 30 and 45%. Mixed landscapes performed poorly in terms of agricultural production: a slight increase in agricultural cover in the landscape (up to 45%) should not compromise the performance of mixed landscapes in maintaining multifunctionality, while allowing an increased agricultural production. Considering landscape configuration, a high number of patches corresponded to a high level of multifunctionality. However, we cannot recommend a specific number of patches necessary to maintain a high level of multifunctionality, but simply suggest maintaining landscape heterogeneity at a variety of scales. At the scale we conducted our analysis, agricultural production was negatively correlated with landscape complexity. Increasing fine scale landscape heterogeneity by increasing hedgerows, small treed habitats, vegetating riparian buffers, and setting aside marginal lands should increase the
ecological (and possibly the sociocultural) functioning without limiting agricultural production.

Multifunctionality can be understood and evaluated on the basis of a wide variety of functions that are influenced by specific biophysical, sociocultural and policy contexts. Because we could have sampled an infinite array of functions across different geographical, cultural, and political regions, we limited our research in spatial extent (Vermont’s Champlain Valley), time frame (one season snapshot), and plot size (78.5 ha). Further, we analyzed standard production, sociocultural, and ecological functions and limited the number of interviewed landowner to one per plot. Having just one interview per plot may have limited our ability to identify significant differences between landscape categories. However, more than 60% of the people interviewed reporting the amount of land owned, were the largest landowner if not the only landowner living in the plot. We found the greatest differences between landscape composition categories in the ecological data that were collected across the entire plot (birds and trees). Fewer significant differences were found in the analysis of the interview data and on the single RGA score per plot. Another possible reason for the greater sensitivity in the ecological data could be ascribed to the continuous nature of these data that provided greater variability.

Certainly one of the reasons behind our incomplete understanding of landscape multifunctionality stems from the challenges in aligning studies that integrate multiple data types and sources. Here, data obtained both from continuous in-field and interview-based data led to variability across response variables. The number of question asked for
each category were not standardized and the typology of questions could have included fewer binary and more multiple-choice questions to enhance the sensitivity of the analysis that we performed. All of the plots that we surveyed were, also, embedded in a region that although heterogeneous within plots, is fairly uniform region-wide. This broad uniformity might have driven some of the plots to be more functionally similar to each other. Even though we found differences based on composition and configuration at the scale of the plot, drivers of landscape functionality might be working at a broader scale.

As shown by our results, the ability to predict the multifunctionality of a landscape depends on landscape structure. We considered each one of our plots as individual landscapes, or spatially heterogeneous areas composed of a mosaic of ecosystems, habitat patches, and elements (Turner et al. 2001). These landscapes are embedded in the Champlain Valley agricultural region that is characterized by 26% agriculture and 50% forest covers. This region is also highly fragmented with landscape elements distributed heterogeneously. We believe that recommendations for enhanced multifunctionality on the basis of landscape structure are valid for landscapes embedded in rural regions similar to the CV. Many portions of the Eastern United States, for example, have similar amounts of land in farms (between 10 and 50% of the entire land) to the values recorded in the CV (USDA & NASS 2012). On the other hand, our suggestions for the enhancement of multifunctionality on the basis of landscape structure might not be as effective for landscapes in regions (i.e. Midwest USA) where intensive agriculture is predominant (> 70%) throughout, and forest remnants are rare.
With any level of agriculture within a landscape, a certain degree of multifunctionality is maintained (Harden et al. 2013). The most simplistic explanation for the existence of multifunctionality, even in intensively cultivated landscapes, is the variety of landowners present in the landscape and the uses they have for different portions of the landscape. A landscape dominated by intensive agriculture might include the farmer homestead, a flower garden, one or more small subdivisions used for housing, a hunting stand etc. The different uses of these fine scale landscape elements contributes to the increased multifunctionality of the landscape. However, above the 45% threshold of agriculture in the landscape, increasing the size and number of agricultural patches in the landscape will result in a progressive decline in functionality, particularly in the ecological and sociocultural functions. This tradeoff between functions in a landscape can be used to assess the proper balance of composition and configuration of land use/land cover patches, thereby enhancing multifunctionality.

Landowners making decisions on how to modify their property are the main agents of landscape change (Lovell et al. 2010). This is particularly true in the USA and in the Northeast where most of the land is privately owned. However the landowners’ efficacy in affecting the functionality of a landscape is also dependent on: 1) their willingness to collaborate with other land-holders present in the landscape and 2) by the zoning policies in place in the region. High levels of trust and cooperation between landowners, low socio-economic heterogeneity, and flexible action-oriented policies that allow collaboration between farmers can enhance landscape multifunctionality (Harden et al. 2013). Wise landscape planning by landowners supported by policies that consider
multifunctionality as one of the top priorities will greatly increase the number of landscape services provided and benefit human and natural communities.
2.5. Tables

Table 1 Metrics (interview or field based), used to quantify functions and sub-functions

<table>
<thead>
<tr>
<th>Function</th>
<th>Source</th>
<th>Sub-function</th>
<th>Metrics</th>
</tr>
</thead>
<tbody>
<tr>
<td>Ecological</td>
<td>Field</td>
<td>River channel</td>
<td>Stream channel integrity</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Birds</td>
<td>Bird species richness</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Plants</td>
<td>Plant species richness</td>
</tr>
<tr>
<td>Production</td>
<td>Interview</td>
<td>Agricultural</td>
<td>Which marketable products does your farm produce? 1) dairy, 2) meat, 3) veggies, 4) grain/hay, and 5) other farm animal products.</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Non-agricultural</td>
<td>1) Do you have a tourism business? Which products or functions are provided by your property? 2) timber/lumber, 3) firewood, 4) maple syrup, 5) wild edibles/medicinal, 6) crafts &amp; ornamentals.</td>
</tr>
<tr>
<td>Social</td>
<td>Interview</td>
<td>Wildlife</td>
<td>Which products or functions are provided by your property? 1) hunting birds, 2) hunting large game, 3) fishing. 4) Do you manage your property in any way to encourage wildlife?</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Recreation</td>
<td>1) Do you spend any time doing outdoor recreation on your property? 2) Do you maintain trails on your property for recreation? 3) Do you allow public access to your property?</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Visual</td>
<td>1) Do you manage your property’s landscape to improve visual quality or aesthetics? 2) Do you keep any treed or forested areas for their contribution to visual quality?</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Conservation</td>
<td>1) Do you have land enrolled in any governmental conservation programs? 2) Do you have land in any other non-governmental arrangements? 3) Do you voluntarily provide a buffer along streams/rivers? 4) Do you allow grazing in the riparian habitat? 5) Do you maintain/retain any areas of the property specifically to conserve native trees?</td>
</tr>
</tbody>
</table>
Table 2 Number of plots in each of the composition and configuration categories.

<table>
<thead>
<tr>
<th>Landscape composition</th>
<th>Simple</th>
<th>Intermediate</th>
<th>Complex</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td>Agriculture(^1)</td>
<td>8</td>
<td>5</td>
<td>0</td>
<td>13</td>
</tr>
<tr>
<td>Mixed</td>
<td>4</td>
<td>11</td>
<td>3</td>
<td>18</td>
</tr>
<tr>
<td>Forest(^2)</td>
<td>5</td>
<td>9</td>
<td>6</td>
<td>20</td>
</tr>
<tr>
<td>Total</td>
<td>17</td>
<td>25</td>
<td>9</td>
<td>51</td>
</tr>
</tbody>
</table>

\(^1\)Agricultural plots contained between 33 and 82 percent agriculture.
\(^2\)Forest plots contained between 0 and 21 percent agriculture.
\(^3\)Simple, intermediate, and complex plots contained between 10 and 23 patches, 25 and 37 patches, and 40 and 59 patches, respectively.

2.6. Figures

Figure 1: Lake Champlain Basin and area of interest for this study (outlined in red). (Image modified from http://www.lcbp.org/)
Figure 2: Plot locations along the seven main river of Northwestern Vermont (USA)
Figure 3 Effect of three types of landscape composition (agricultural (A), mixed (M), and forest (F)) on functionality across three functional metrics: production, ecological, and social. Functionality values increase along each axis starting at 0 at the center of the triangle and having the maximum value at each vertex. * indicates significant overall effects of landscape composition with significant pairwise differences shown below each functional metric. Data were collected in the Champlain Valley of Vermont, 2009.
Figure 4 Effect of three types of landscape composition (agricultural (A), mixed (M), and forest (F)) on functionality across nine subfunctional metrics: agricultural and non-agricultural production, bird and tree species richness and river channel integrity, and the social components of wildlife, recreation, visual, and conservation. Functionality values increase along each axis starting at 0 at the center of the polygon and having the maximum value at each vertex. * indicates significant overall effects of landscape composition with significant pair-wise differences shown below each subfunctional metric. Data were collected in the Champlain Valley of Vermont, 2009.
Figure 5 Effect of three types of landscape configuration (complex (C), intermediate (I), and simple (S)) on functionality across three functional metrics: production, ecological, and social. Functionality values increase along each axis starting at 0 at the center of the triangle and having the maximum value at each vertex. * indicates significant overall effects of landscape composition with significant pair-wise differences shown below each functional metric. Data were collected in the Champlain Valley of Vermont, 2009.
Figure 6 Effect of three types of landscape configuration (complex (C), intermediate (I), and simple (S)) on functionality across nine subfunctional metrics: agricultural and non-agricultural production, bird and tree species richness and river channel integrity, and the social components of wildlife, recreation, visual, and conservation. Functionality values increase along each axis starting at 0 at the center of the polygon and having the maximum value at each vertex. * indicates significant overall effects of landscape composition with significant pair-wise differences shown below each subfunctional metric. Data were collected in the Champlain Valley of Vermont, 2009.
Figure 7 Tradeoff between ecological and production functions as a function of percent agriculture in the landscape. Data were collected from the Champlain Valley, Vermont in 2009.
Figure 8 Tradeoff between functions as a function of number of patches in the landscape. Data were collected from the Champlain Valley, Vermont in 2009.
INTRODUCTION/INVITATION

You are being invited to take part in a research study which involves completing a survey. The broad purpose of this three year study is to assess the various benefits and functions of the Vermont agricultural landscape and to eventually recommend ways to enhance conservation without compromising economic and cultural factors. This should take approximately 20 minutes. There are no risks to you and your responses will remain anonymous. While there are no direct benefits to you by participating, the information you provide will enhance our understanding of the values of this landscape. If you do not feel comfortable answering any of the questions, please feel free to decline.

Please indicate on the map the portion of land you own within the sampling area >>>>

LANDOWNER DEMOGRAPHICS

1. Name: _______________________________________________________________
2. Address: _____________________________________________________________
3. Phone: ____________________________  Email: ____________________________
4. Do you own the property of this residence (yes/no): ____________ (property with the study area)
   If no, what is your relationship: _______________________________________
5. Years living/working on the property: ______  Years living in Vermont: _______
6. Size of the homestead property (acres):_________ Estimate % acreage in the following:
   asture_____  Hay_____ Corn ____  Forest _____  Wetland ____  Other ______________
   ______________________________________________________________________

7. Number of family members living in the household: ________ Number contributing income: ________

LIVELIHOODS

8. Do you farm any of the property (yes/no): _________ Estimate % income from farming __________

If YES, answer the following questions. If NO, skip to question 9.
Acreage in farmland: ________ Types of marketable products (circle all that apply):

Veggies/Fruit  Meat  Dairy  Forage Crops  Grain  Silage  Syrup  Other

Top 3 Products (based on income): ________________________________

Animals (type/number): ____________________________________________

Markets for selling agricultural products:   CSA %___   Coop %___
   Wholesale %___   Farmers’ Market %___   VT stores%___
   VT restaurants%___   Farmstands%___   U-Pick%___
   Other________

9. Do you lease any part of the property (yes/no) ____  Area (a) _____

Function ____________

10. Do you have a tourism business (ex: B&B) (yes/no): _______

Type: __________________________________________________________

11. Please indicate which products or functions are provided by your property and estimate annual value:

<table>
<thead>
<tr>
<th>Product/Function</th>
<th>Applicable?</th>
<th>Rate importance (0=none to 5=high)</th>
<th>Income contribution (estimate $)</th>
<th>Annual value (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Timber/lumber</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Firewood</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Maple Sugaring</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Wild edibles/medicinal</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Crafts &amp; Ornaments</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Hunting birds (turkey)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Hunting large game (deer)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fishing</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Comments:______________________________________________________________
________________________________________________________________________

49
RECREATION

12. Estimate hours/week you spend on outdoor recreation (on or off property): ________

What % on your property? _____  What % on neighboring properties? ____________

13. Do you maintain trails on your property for recreation (yes/no): ______  Type:

_____________________________________________________

14. Do you allow public access to your property (yes/no):____________

Type:_________________________

15. Do you manage the landscape in any way to encourage wildlife (yes/no): ______

Describe: ______________________________________________________________

16. List your most frequent outdoor recreational activities and the type of landscape in which they occur:

<table>
<thead>
<tr>
<th>Activity</th>
<th>Landscape (forest, open pasture, golf course, etc)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1.</td>
<td>____________________</td>
</tr>
<tr>
<td>2.</td>
<td>____________________</td>
</tr>
<tr>
<td>3.</td>
<td>____________________</td>
</tr>
</tbody>
</table>

VISUAL QUALITY

17. Do you change your property’s landscape to improve visual quality or aesthetics (yes/no):____

Describe: _____________________________________________________________

________________________________________________________________________

18. Do you keep any treed or forested areas for their contribution to visual quality (yes/no): _____

Describe: _____________________________________________________________

________________________________________________________________________
19. Rate the visual quality of Vermont overall (1=low – 5=high): _____

Describe what you like the most about the Vermont landscape.

________________________________________________________________________
________________________________________________________________________
________________________________________________________________________

Describe what you don’t like about the Vermont landscape.

________________________________________________________________________
________________________________________________________________________
________________________________________________________________________

20. Rate the visual quality of the highlighted part of your property (1=low, 5=high): ____

21. In terms of visual quality (aesthetics), what would be your preferred mixture of landscapes in Vermont? (ex: agricultural, forested, residential)

________________________________________________________________________
________________________________________________________________________
________________________________________________________________________

22. Rate your level of attachment to the Vermont landscape in general (1=low – 5=high): ______

23. Rate your level of attachment to the landscape of this property (1=low – 5=high): ______

24. Indicate the level of importance of cultural functions provided by the landscape (1=low – 5=high):

Visual Quality _____  Recreation ______  Historical legacy _____  Spirituality ______
CONSERVATION

25. Do you currently have land enrolled in any governmental conservation programs (yes/no): _______

26. Do you currently have land in any other non-governmental arrangements (i.e. VLT) (yes/no): ________

If YES to above, fill out chart below. If NO, skip to question 27.

<table>
<thead>
<tr>
<th>Conservation Program</th>
<th>Acres enrolled</th>
<th>Years enrolled</th>
<th>Benefit from enrolling</th>
</tr>
</thead>
<tbody>
<tr>
<td>Wildlife Habitat Incentives Program (WHIP)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Environmental Quality Incentive Program (EQIP)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Conservation Reserve Enhancement Program (CREP)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Farm and Ranch Lands Protection Program (FRPP)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Other</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

27. If not, have you had land enrolled in programs in the past (yes/no): _______

28. Do you voluntarily provide a buffer along streams/rivers (yes/no/NA): _______

    Width (estimate average): _______

29. Do you allow grazing in the riparian buffer zone (yes/no/NA): ______

30. Do you maintain/retain any areas of the property specifically to conserve native trees (yes/no): ______

    Describe: __________________________________________________________________________
    __________________________________________________________________________
OPEN QUESTIONS

31. What cultural values do you associate with your property, for example historical/family, social, educational or spiritual values
________________________________________
________________________________________
________________________________________
________________________________________

32. Please describe how this property and broader landscape has changed over time (since you have been familiar with it and state time span you are referring to):
________________________________________
________________________________________
________________________________________
________________________________________

33. Please describe how you would like this property and broader landscape to look in thirty years:
________________________________________
________________________________________
________________________________________
________________________________________

34. What force do you think will have the most effects on landscape change in the next thirty years for this property and the surrounding landscape:
________________________________________
________________________________________
________________________________________
________________________________________

35. Do you see the possibility and/or potential of working with your neighbors to manage ecological, cultural and economic factors of your shared landscape? And specifically: managing the riparian zone?
________________________________________
________________________________________
________________________________________
________________________________________

36. IF FARMER: Do you have any thoughts on ways to simultaneously support conservation and production goals in your property and in the broader landscape?
________________________________________
________________________________________
________________________________________
________________________________________

CLOSING STATEMENT:

Thank you for your participation. If you have any questions about this research, please feel free to contact us: Ernesto Mendez, emendez@uvm.edu.
2.7. Literature cited


Harden, N. M., L. L. Ashwood, W. L. Bland, and M. M. Bell. 2013. For the public good: Weaving a multifunctional landscape in the Corn Belt. Agriculture and Human Values 30:525–537.


CHAPTER 3: ARE EFFECTS OF LANDSCAPE STRUCTURE ON BIRD ABUNDANCE SENSITIVE TO SPATIAL GRAIN?

3.1. Introduction

In agricultural landscapes, wooded elements such as single trees, shrubs, small clump of trees, and hedgerows can provide the avian community with nesting, roosting, cover, and foraging areas (Hinsley & Bellamy 2000). The spatial arrangement of these wooded elements in the landscape influence their value for birds. For example, vegetative cover as measured by individual trees and hedgerows has been positively correlated with bird species richness and abundance (Hinsley & Bellamy 2000; Batáry et al. 2010).

An understanding of the influence of landscape structure on birds in agricultural systems at a fine-grained spatial scale is critical to developing wildlife-friendly management actions. However, even if small wooded landscape elements are recognized as important in rural systems, the capacity to identify and record their presence in geographic information systems has been limited by the high digitizing costs (both in terms of money and time) and the lack of widely accessible high resolution imagery. Until the mid-2000s, these short-comings hindered the possibility of conducting comprehensive studies on the influence of such elements on the diversity, ecology, and breeding success of birds at a fine spatial scale (Wulder et al. 2004). Now, the spatial capability of high resolution satellite images and advances in processing software allows for more accurate mapping of these wooded elements (Vannier & Hubert-Moy 2014).
Relationships between landscape patterns and ecological processes so far has been mostly addressed at coarse resolution (Mayer & Cameron 2003; Gottschalk et al. 2011). With the recent availability of high resolution imagery, fine resolution land use and land cover (LULC) maps can be produced, and the interaction between landscape structure and wildlife can be studied at fine spatial scales. When creating fine resolution LULC maps, an important choice is the selection of the minimum mapping unit (MMU). The minimum mapping unit (or grain) of a LULC dataset describes the size of the smallest feature that is retained in the generated map and a large minimum mapping unit ignores small land elements by incorporating them into the surrounding larger features (Saura 2002).

Spatial resolution (grain or pixel size, and map extent), has been analyzed in relation to bird abundance and occupancy patterns (Thompson & McGarigal 2002; Seoane et al. 2004; Gottschalk et al. 2011; Morelli et al. 2014). As an alternative to fine grain maps, on-site survey metrics have been used in determining the relationship between bird abundance and fine landscape structure (Betts et al. 2006). Most of these studies, which typically used grain sizes greater than 30m, did not find a significant improvement in the predictive power of models with finer resolution. Only with the most recent studies that use fine-grained data models does there seem to be support for the hypothesis that with smaller spatial sampling units we can obtain more accurate results when modeling bird-habitat relationships (Gottschalk et al. 2011).

Many songbirds have territory sizes with radii generally < 100 m (Poole 2005) and thus may perceive their habitat at a finer grain than most LULC maps currently
available. With coarse LULC maps the heterogeneity of the landscape is simplified and small landscape elements, potentially important to songbirds, are under-represented by being absorbed into larger LULC categories. We used 72 plots to test whether LULC maps that include small MMUs can predict bird abundance more accurately than LULC maps with large MMUs. We hypothesized that for some bird species abundance is related to small landscape elements that only LULC maps with small MMUs can appropriately capture.

3.2. Methods

The Champlain Valley of Vermont (CV) is an agricultural region bounded by the Green Mountains to the west and Lake Champlain to the east. This area supports a mix of agriculture and forest with residential areas scattered throughout. The central part of the CV has a greater population density and includes the largest city in Vermont: Burlington.

As part of a larger study of agricultural multifunctionality and landscape ecology (Sutti and Strong unpublished data), we randomly identified 60 circular (800 m radius) plots along the 7 main rivers (Missisquoi, Lamoille, Winooski, LaPlatte, Lewis, Little Otter Creek, Otter Creek) of the CV. We grouped the plots on the basis of landscape structure and distance from Burlington. We asked permission from landowners to work on their land, and selected as study plots the first 18 locations for which we obtained permission.
Within each plot, we sampled bird communities at 4 point count locations. Points were positioned 250 m in the four cardinal directions from the center of the plot. At each point, we conducted a standard 10 min point count (Hutto et al. 1986), recording all birds heard and seen. Point counts were repeated twice in the same season by different observers. We conducted point counts from 0510 to 0850 in fair weather with a maximum of 15 days between counts at a point. We used the data from the bird point counts to estimate bird occupancy and bird abundance for a subset of species at the 100 m distance.

To be able to adequately interpret the correlation between bird abundance and landscape covariates, we selected only species that were recorded in more than 30 of the 72 locations surveyed (n = 106 species of birds detected in total). The species included in the analysis were: American robin (AMRO, Turdus migratorius), black-capped chickadee (BCCH, Poecile atricapillus), common yellowthroat (COYE, Geothlypis trichas), ovenbird (OVEN, Seiurus aurocapilla), red-eyed vireo (REVI, Vireo olivaceus), red-winged blackbird (RWBL, Agelaius phoeniceus), song sparrow (SOSP, Melospiza melodia), veery (VEER, Catharus fuscescens), and yellow warbler (YWAR, Dendroica petechia). We used N-mixture models (Poisson and negative binomial) to estimate abundance of this subset of bird species at each point count location (Royle 2004). Because of the period of sampling (breeding birds have established territories) and sampling protocol (distance from observer) we can assume that the population monitored in each location was closed.

For the creation of fine scale maps we used National Agriculture Imagery Program - NAIP 2008 imagery. The imagery has a 1 meter resolution and includes red,
green, blue and near infrared bands. We used ERDAS Imagine® mosaic pro (Hexagon Geospatial, Norcross, Georgia, USA) to manipulate the orthoimagery around the center of each plot to obtain eighteen 100 ha imagery subset squares. We used the imagery and a coarse scale thematic layer generated by the spatial analysis lab at the University of Vermont (O’Neil-Dunne 2005) in eCognition® (Trimble, Westminster, CO, USA) to obtain a fine scale land use/land cover (LULC) with 1 meter spatial resolution and 5 m² minimum mapping unit (MMU).

We identified 10 LULC classes: forest (patches of forest that maintain some core area using a 50 m buffer), small forest (forest patches greater than 1000 m² without core area), trees (treed patches less than 1000 m²), hedgerow (linear treed patches less than 50 m wide and at least 4 times longer than wide), shrub, wetland (ground wet for most of the field season with hydrophilic vegetation), agriculture (hay fields, pastures, crops, and gardens), water, urban (paved or compacted surfaces, buildings), and lawn (mowed areas, cemeteries, and golf courses). We summed the area of hedgerow, small forests, and tree classes to obtain the small treed landscape elements total cover. We also summed hedgerow, small forest, trees and shrubs, to obtain the area covered by small vegetated elements in the plots.

We modified the LULC with MMU of 5 m² to generate increasingly coarser layers. To assure a constant classification scheme across the different resolution LULC layers, we selected patches with areas less than 25 m² and aggregated (using ArcGIS tool Eliminate) these small polygons with the adjacent polygon (> 25 m²) with which it shared the longest border. We used the same procedure to generate a 100 m² base map (using
the 25 MMU layer as an input), a 1000 m² (from the 100 m² base map), and a 4000 m² base map (from the 1000 m² base map). The 1000 MMU should provide similar results to the coarse thematic layer generated by the University of Vermont with a MMU of 0.1 ha. The 4000 MMU LULC is similar to the NLCD classification which has 30 by 30 m resolution and ~4000 m² minimum mapping unit (Fry et al. 2011).

We used patch analyst (Rempel et al. 2012) to calculate values for a list of landscape composition and configuration metrics for each plot and each MMU map. Because many of the configuration metrics provided by patch analyst are correlated, and because the following metrics show predictable changes and simple scaling relationships (Wu et al. 2002), we reduced the number of configuration metrics used for our analysis to include area weighted mean shape index (AWMSI), area weighted mean patch fractal dimension (AWMPFD), edge density (ED), mean patch size (MPS), and number of patches (NumP). A shape index provides a measure of geometric complexity of a patch resulting in values starting at 1 for simple circular patches, and increasing with increasing patch shape irregularity. Fractal dimension indices are measures of shape complexity, with values closer to 1 for shapes with simple perimeter and values closer to 2 for more complex shapes. Edge density measures the amount of edge relative to the landscape area (Leitao et al. 2006; Rempel et al. 2012). Composition metrics are the area of each class expressed in hectares.

We estimated abundance and detection probabilities for bird species using each landscape metric as single covariate in Poisson or negative binomial models for each plot at all MMUs (Figure 9). We compared the null model, in which abundance and detection
probabilities are not influenced by covariates, with each covariate model using Akaike’s Information Criterion (AIC). We defined the top model (= the most parsimonious model) as the model with the lowest AIC that converged and performed better than the null model. We also reported all the models within 2 AIC units from the most parsimonious model because all these models fit the data similarly (Burnham & Anderson 1998).

We checked goodness-of-fit of the top model using a parametric bootstrap procedure. Fit was determined by comparing the sum-of-squared errors (SSE) for the observed data to the SSE values obtained from the bootstrapped datasets (Kéry et al. 2005). If the fit of the top Poisson model was inadequate, we used negative binomial models and tested their fit as Poisson models are a special case of negative binomial models (Royle 2004). We estimated the bird abundance and detection probabilities for each 100 m plot at all MMUs using R cran package unmarked (Fiske & Chandler 2011).

3.3. Results

3.3.1. Effect of MMU on landscape configuration

With the increase of MMU, mean patch size significantly increased (Kruskal-Wallis test, $X^2_4 = 18.55, P=0.001$), and number of patches significantly decreased (Kruskal-Wallis test, $X^2_4 = 15.61, P=0.003$). Of the five landscape configuration metrics used in the mixture models, area weighted mean patch fractal dimension showed the least variation with changes in MMU. All the other metrics showed slight increasing or decreasing trends across MMU values (Figure 10), and could prove useful in identifying abundance changes for certain bird species.
Small landscape elements numbers decreased in abundance with the increase of MMU size, while larger patches (forest and agriculture in particular) maintained a constant number of patches regardless of MMU (Figure 11).

### 3.3.2. Effect of MMU on landscape composition

Increase in MMU had a significant negative effect on composition classes for trees (Kruskal-Wallis test, $X^2_{4}=112.46, P<0.0001$), small treed ($X^2_{4}=30.28, P<0.0001$), and small vegetated ($X^2_{4}=12.59, P=0.014$) habitats. Despite the effect of MMU on number of patches, the area in each LULC class did not vary greatly, with the exception of the trees category (Table 3). Land cover types that occupied a large portion of the landscape tended to increase in area with changing MMU by absorbing small land cover types. Due in part to the aggregation algorithm used, linear landscape features (water and roads), tended to increase their area by absorbing adjacent smaller land cover types even if they occupied a small portion of the landscape. Proportionally, shrub and trees were the land cover classes that showed the greatest change in area covered with variation in MMU (Table 3).

### 3.3.3. Bird abundance analysis

For two species (BCCH, VEER), mixture models did not provide interpretable results due to poor fit and large standard errors. In general, mixture models using configuration covariates had stronger support than composition models (Table 4).
Configuration

The Top models for AMRO, COYE, and OVEN included covariates at a variety of MMUs. For SOSP and YWAR, all the models that performed better than the null model and were within 2 AIC scores from the top model included a covariate with lower MMUs values.

For YWAR, the top model was mean patch size (MPS) at 25 MMU, closely followed by MPS at 5 MMU. All other models that performed better than the null model were more than 2 AIC scores from the top model. YWAR abundance decreased with increasing mean patch size, indicating a preference for small patches (Figure 12). For SOSP, number of patches and area weighted mean patch fractal dimension at low MMU were the best performing models. SOSP abundance increased with increasing number of patches and increased with patch perimeter complexity. The RWBL abundance model did not fit using either a Poisson or negative binomial distribution. For REVI no models performed better than the null model.

Composition

The top models for most species included either or both the agriculture and forest covariate at all MMUs (Table 5). Models with these two covariates resulted in almost identical AIC scores (within 2 AIC units) across MMU values within species.

We expected bird abundances of interior forest, forest, and open habitat species to vary as a function of the forest or agriculture covariate (Figure 13), thus we focused our attention on bird species that were commonly observed nesting in or associated with
small landscape elements (AMRO, YWAR, SOSP). Only in the case of AMRO did models with a small landscape element covariate (trees at all MMU below 100) perform better than any other covariate. The abundance of robins increased with tree cover in the landscape. No other covariate models performed better than the null model.

3.4. Discussion

The effect of changing grain size on species/habitat relationships has been shown to be weak or nonexistent (Gottschalk et al. 2011). One reason for the lack of sensitivity of these models might be ascribed to the coarse grain size used in most studies (Gottschalk et al. 2011). In the past decade we have seen a progressive increase in availability of very high resolution digital imagery. These newly available digital data have allowed researchers to improve map resolution and further assess the effect of grain size at scales that more closely match the scale at which species relate to their habitat. Our study joins the limited research conducted thus far at fine resolution and supports, at least for some species of edge tolerant birds, the hypothesis that bird abundance in heterogeneous agricultural landscapes is predicted more accurately by covariates obtained from fine grain maps.

Changes in MMU were more evident for covariates that were influenced by the presence of small landscape elements. Mean patch size values increased with increasing MMU because of the disappearance of small landscape elements absorbed by the neighboring larger elements. Also, the number of patches in each landscape decreased with increasing MMU. Similarly, the composition covariates with more significant
changes in values with changing MMU were those including the trees covariates. Using coarse LULC maps that do not capture small landscape elements actively used as settlement cues might hinder the ability to discern meaningful conclusions about species distribution patterns (Wiens 1989).

Analyzing the landscape configuration covariates, for the two species for which we had satisfactory model fit, the top models included only covariates at low MMUs. Two of these species, YWAR and SOSP, prefer heterogeneous landscapes and associate most often with small landscape elements (Poole 2005). For YWAR, the two models including mean patch size (MPS) at 5 and 25 MMU clearly provided greater predictive power than the third best model (over 2.5 AIC values lower). MPS thus seems to be an important metric in determining the abundance of YWAR. The number of patches was included in the top model for SOSP. However for this species, many covariates resulted in low AIC scores within 3-5 units from the top model. The precision of the top model including number of patches was fairly low with wide confidence intervals around estimated SOSP abundances in landscapes with many patches.

Most of the composition landscape metrics we included in the models did not show significant differences in their average values with changing MMU. Only the trees covariate and the other two covariates that included tree cover (small treed and small vegetated) varied significantly with variation in MMU. Because of the small size of the trees class (defined as treed patches less than 1000 m²), we expected covariates including trees to have greater support for more species. Despite the significant variation observed, the trees covariate was important only for AMRO in explaining abundance.
Edge species associate more closely with small wooded elements in agricultural landscapes (Sykes & Hannon 2001; Poole 2005; Puckett et al. 2009). Our models support the hypothesis that heterogeneous agricultural landscapes with high concentrations of small wooded elements are favorable to edge species (YWAR, SOSP, and AMRO). Thus, maps able to capture the landscape heterogeneity at fine scales might be necessary to study these species/habitat relationships.

Agriculture, forest, and area weighted mean patch fractal dimension covariates did not exhibit substantial variation across MMU values. However, these covariates were included in the highest ranking models for many species. If bird species abundances are affected by agriculture and forest composition, researchers could use coarse maps obtaining results similar to the ones obtained with more accurate fine scale maps.

For some of the species included in this research, the effects of landscape structure on bird abundances appeared to be insensitive to spatial grain both when configurational and compositional covariate are considered. For other species, mixture models, even with strong goodness-of-fit, lacked precision which resulted in large standard errors. A possible solution for this problem might be to increase the number of visits to each site. We were able to visit our point count locations only twice per season. Having a greater number of visits per site should increase the precision of the models.

Because we used an aggregation algorithm that merged selected polygons below the MMU threshold with neighboring polygons sharing the longest border, long and narrowly shaped elements become artificially enlarged at coarse MMUs. Small vegetated areas that should have been aggregated to agricultural or forested patches, were
inappropriately merged with roads and water features by the software. We manually corrected the most evident aggregation errors. However, a similar analysis to the one proposed here might provide more accurate results if researchers compared increasingly coarse LULC obtained from different digitization processes, or used alternative aggregation algorithms.

Object-oriented classification has a great potential for the generation of fine resolution classification maps. Because object-oriented classification software can accept different formats of high precision data inputs (imagery, already generated classifications, LiDAR-based elevation models etc.), and because computing power is quickly improving, we can expect fine resolution LULC maps to become more readily available in the future. However, until that time, we recommend the use of the finest map resolution available. Despite the fact that our research showed that coarse resolution maps could adequately predict abundances of some bird species, scale and grain size affect other important avian ecology aspects. Researchers should continue to evaluate the importance of small wooded elements since many birds use these elements in agricultural landscapes as cues in habitat selection.
Table 3: Average percent area in the entire plot for each land use/land cover (LULC) class at different minimum mapping unit (MMU). Data were collected at 72 locations in the Champlain Valley of Vermont (USA) in 2010 and 2011.

<table>
<thead>
<tr>
<th>LULC</th>
<th>5MMU</th>
<th>25MMU</th>
<th>100MMU</th>
<th>1000MMU</th>
<th>4000MMU</th>
</tr>
</thead>
<tbody>
<tr>
<td>agriculture</td>
<td>28.1</td>
<td>28.1</td>
<td>28.2</td>
<td>28.4</td>
<td>28.6</td>
</tr>
<tr>
<td>forest</td>
<td>49.6</td>
<td>49.6</td>
<td>49.6</td>
<td>50.0</td>
<td>51.0</td>
</tr>
<tr>
<td>hedgerow</td>
<td>1.3</td>
<td>1.3</td>
<td>1.3</td>
<td>1.3</td>
<td>0.9</td>
</tr>
<tr>
<td>lawn</td>
<td>2.1</td>
<td>2.1</td>
<td>2.2</td>
<td>2.2</td>
<td>1.7</td>
</tr>
<tr>
<td>shrub</td>
<td>4.3</td>
<td>4.3</td>
<td>4.3</td>
<td>4.2</td>
<td>3.8</td>
</tr>
<tr>
<td>small forest</td>
<td>2.2</td>
<td>2.2</td>
<td>2.2</td>
<td>2.2</td>
<td>1.9</td>
</tr>
<tr>
<td>trees</td>
<td>1.1</td>
<td>1.0</td>
<td>0.8</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>urban</td>
<td>2.2</td>
<td>2.2</td>
<td>2.2</td>
<td>2.3</td>
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<td>water</td>
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<td>4.5</td>
<td>4.5</td>
<td>4.5</td>
<td>4.7</td>
</tr>
<tr>
<td>wetland</td>
<td>4.6</td>
<td>4.6</td>
<td>4.7</td>
<td>4.8</td>
<td>4.7</td>
</tr>
<tr>
<td>Small treed</td>
<td>4.6</td>
<td>4.6</td>
<td>4.3</td>
<td>3.5</td>
<td>2.8</td>
</tr>
<tr>
<td>=hedgerow+trees+small forest</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>small vegetated</td>
<td>8.9</td>
<td>8.8</td>
<td>8.6</td>
<td>7.7</td>
<td>6.6</td>
</tr>
</tbody>
</table>
Table 4: Top configuration covariate’s abundance models. We used the detection probability null model (p) and evaluated the influence of each single covariate on bird abundance (Lam), using either a Poisson (Poi) or negative binomial (NB) model. We listed for each species of bird models that performed better than the null model and were within 2 AIC from the top model. Models are ranked by AIC score (lowest AIC at the top). Data were collected at 72 locations in the Champlain Valley of Vermont (USA) in 2010 and 2011.

<table>
<thead>
<tr>
<th>Species</th>
<th>Top model</th>
<th>AIC top</th>
<th>AIC null</th>
<th>Other models within 2 AIC from top model</th>
<th>Distribution</th>
</tr>
</thead>
<tbody>
<tr>
<td>AMRO</td>
<td>p(.)Lam(AWMPFD4000)</td>
<td>259.5</td>
<td>267.64</td>
<td>p(.)Lam(AWMPFD1000), p(.)Lam(AWMPFD5), p(.)Lam(AWMPFD100), p(.)Lam(AWMPFD25), p(.)Lam(AWMSI4000), p(.)Lam(AWMSI1000)</td>
<td>Poi</td>
</tr>
<tr>
<td>COYE</td>
<td>p(.)Lam(AWMPFD1000)</td>
<td>338.82</td>
<td>341.32</td>
<td>p(.)Lam(AWMPFD4000), p(.)Lam(AWMSI1000), p(.)Lam(AWMSI4000), p(.)Lam(MPS5), p(.)Lam(MPS25), p(.)Lam(AWMPFD5), p(.)Lam(AWMPFD25), p(.)Lam(AWMPFD100), p(.)Lam(AWMSI100), p(.)Lam(MPS4000), p(.)Lam(MPS100), p(.)Lam(AWMSI5), p(.)Lam(AWMSI25), p(.)Lam(MPS1000)</td>
<td>Poi</td>
</tr>
<tr>
<td>OVEN</td>
<td>p(.)Lam(AWMSI5)</td>
<td>313.12</td>
<td>333.76</td>
<td>none</td>
<td>NB</td>
</tr>
<tr>
<td>REVI</td>
<td>p(.)Lam(.)</td>
<td>314.05</td>
<td>314.05</td>
<td>none</td>
<td>NB</td>
</tr>
<tr>
<td>SOSP</td>
<td>p(.)Lam(NumP5)</td>
<td>363.53</td>
<td>380.6</td>
<td>p(.)Lam(NumP25), p(.)Lam(AWMPFD5), p(.)Lam(AWMPFD25)</td>
<td>NB</td>
</tr>
<tr>
<td>YWAR</td>
<td>p(.)Lam(MPS25)</td>
<td>324.35</td>
<td>339.35</td>
<td>p(.)Lam(MPS5)</td>
<td>NB</td>
</tr>
</tbody>
</table>
Table 5: Top composition covariate's abundance models. We used the detection probability null model (p) and evaluated the influence of each single covariate on bird abundance (Lam), using either a Poisson (Poi) or negative binomial (NB) model. We listed for each species of bird models that performed better than the null model and were within 2 AIC from the top model. Models are ranked by AIC score (lowest AIC at the top). Data were collected at 72 locations in the Champlain Valley of Vermont (USA) in 2010 and 2011.

<table>
<thead>
<tr>
<th>Species</th>
<th>Top model</th>
<th>AIC top</th>
<th>AIC null</th>
<th>Other models within 2 AIC from top model</th>
<th>Distribution</th>
</tr>
</thead>
<tbody>
<tr>
<td>AMRO</td>
<td>p(.)Lam(trees5)</td>
<td>264.38</td>
<td>267.64</td>
<td>p(.)Lam(trees25), p(.)Lam(trees100)</td>
<td>Poi</td>
</tr>
<tr>
<td>COYE</td>
<td>p(.)Lam(ag5)</td>
<td>336.52</td>
<td>341.32</td>
<td>p(.)Lam(ag25), p(.)Lam(ag100), p(.)Lam(ag1000), p(.)Lam(ag4000), p(.)Lam(forest5), p(.)Lam(forest25), p(.)Lam(forest100), p(.)Lam(forest1000), p(.)Lam(forest4000)</td>
<td>Poi</td>
</tr>
<tr>
<td>OVEN</td>
<td>p(.)Lam(ag4000)</td>
<td>287.39</td>
<td>333.76</td>
<td>p(.)Lam(ag1000), p(.)Lam(ag100), p(.)Lam(ag25), p(.)Lam(ag5)</td>
<td>Poi</td>
</tr>
<tr>
<td>REVI</td>
<td>p(.)Lam(ag100)</td>
<td>290.02</td>
<td>317.97</td>
<td>p(.)Lam(ag1000), p(.)Lam(ag25), p(.)Lam(ag5), p(.)Lam(ag4000)</td>
<td>Poi</td>
</tr>
<tr>
<td>RWBL</td>
<td>p(.)Lam(ag4000)</td>
<td>468.13</td>
<td>515.22</td>
<td>p(.)Lam(ag1000), p(.)Lam(ag100), p(.)Lam(ag25), p(.)Lam(ag5)</td>
<td>NB</td>
</tr>
<tr>
<td>SOSP</td>
<td>p(.)Lam(forest5)</td>
<td>361.04</td>
<td>380.6</td>
<td>p(.)Lam(forest25), p(.)Lam(forest100), p(.)Lam(forest1000), p(.)Lam(forest4000), p(.)Lam(ag4000)</td>
<td>NB</td>
</tr>
<tr>
<td>YWAR</td>
<td>p(.)Lam(ag4000)</td>
<td>322.75</td>
<td>330.34</td>
<td>p(.)Lam(ag1000), p(.)Lam(ag100), p(.)Lam(ag25), p(.)Lam(ag5), p(.)Lam(forest1000), p(.)Lam(forest100), p(.)Lam(forest25), p(.)Lam(forest5)</td>
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3.6. Figures

<table>
<thead>
<tr>
<th>Imagery</th>
<th>5MMU</th>
<th>25MMU</th>
<th>100MMU</th>
<th>1000MMU</th>
<th>4000MMU</th>
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</table>

<table>
<thead>
<tr>
<th>Species</th>
<th>Models for all species and each MMU</th>
</tr>
</thead>
<tbody>
<tr>
<td>AMRO</td>
<td>Composition</td>
</tr>
<tr>
<td>BCCH</td>
<td>Area of each of the 10 LULC class, area of small treed elements, and area of small vegetated elements.</td>
</tr>
<tr>
<td>COYE</td>
<td>Configuration</td>
</tr>
<tr>
<td>OVEN</td>
<td>Area weighted mean shape index, area weighted mean patch fractal dimension, edge density, mean patch size, and number of patches</td>
</tr>
<tr>
<td>REVI</td>
<td></td>
</tr>
<tr>
<td>RWBL</td>
<td></td>
</tr>
<tr>
<td>SOSP</td>
<td></td>
</tr>
<tr>
<td>VEER</td>
<td></td>
</tr>
<tr>
<td>YWAR</td>
<td></td>
</tr>
</tbody>
</table>

Figure 9: N-mixture model set. For all species we estimated abundance and detection probabilities at each minimum mapping unit. We kept detection probability constant and used each composition and configuration metric as a single covariate for bird abundance. For each species the model set included: 12 LULC area covariates*5 MMU maps = 60 configuration models, 5 covariates*5 MMU maps = 25 composition models. Data were collected at 72 locations in the Champlain Valley of Vermont (USA) in 2010 and 2011.
Figure 10: Average values for configuration metrics at different minimum mapping units (MMU). Data were collected at 72 locations in the Champlain Valley of Vermont (USA) in 2010 and 2011. AWMSI = area weighted mean shape index; AWMPFD = area weighted mean patch fractal dimension; ED = edge density; MPS = mean patch size; NumP/10 = number of patches divided per 10. * = significant changes correlated with increasing MMU.
Figure 11: Average number of patches by LULC in all plots with changing minimum mapping units (MMU). Data were collected at 72 locations in the Champlain Valley of Vermont (USA) in 2010 and 2011. * = significant changes correlated with increasing MMU.
Figure 12: Bird abundance trends obtained from models including the top configuration covariate. Bird abundances and landscape metrics were collected at 72 locations in the Champlain Valley of Vermont (USA) in 2010 and 2011.
Figure 13: Bird abundance trends obtained from models including the top composition covariate. Bird abundances and landscape metrics were collected at 72 locations in the Champlain Valley of Vermont (USA) in 2010 and 2011.
3.7. Literature cited


CHAPTER 4: IMPORTANCE OF SMALL LANDSCAPE ELEMENTS FOR
BREEDING BIRDS IN AGRICULTURAL LANDSCAPES

4.1. Introduction

Intensification of farming practices during the past decades has corresponded with a decline in biodiversity in agricultural landscapes (Green et al. 2005). The intensification of farming practices has also corresponded to a shift in the structure and functionality of agricultural landscapes; agroecosystems have become simpler and more functionally homogeneous than in the past (Brown & Schulte 2011; Flohre et al. 2011). Agricultural landscapes constitute a large component of the habitat available for many terrestrial species, occupying around 40% of the Earth’s terrestrial surface (Foley et al. 2005), and have an important role in providing resources such as food and shelter for many species (Fahrig et al. 2011). Because of their predominance, agricultural landscapes are key areas in which to focus management and conservation strategies for biodiversity preservation.

With the maturation of the discipline of landscape ecology and the availability of high resolution aerial imagery, it is now possible to investigate more thoroughly the influence of landscape structure on plant and animal wild populations (Donovan & Strong 2003). The effect of landscape structure on changes in bird communities has been investigated for forest (Robinson et al. 1995; Rodewald 2003) and grassland species (Shustack et al. 2010). In both systems, decreased patch size, increased isolation, and intensification of management has resulted in decreased bird species richness. However,
heterogeneous landscapes that include forested and agricultural patches in a heterogeneous matrix have not been thoroughly studied and the results have been equivocal (Wretenberg et al. 2010; Fahrig et al. 2011).

In agricultural landscapes, small wooded habitats such as hedgerows, shrub patches and small clumps of trees provide birds with cover for nesting, roosting, and foraging (Hinsley & Bellamy 2000). The spatial arrangement of these wooded habitats in the landscape, their management, size and structure influence their value to birds. For example, the abundance of trees within the hedgerow, hedgerow size, and the presence of vegetative cover in and around the base of the hedgerow has been positively correlated with bird species richness and abundance (Hinsley & Bellamy 2000; Batáry et al. 2010). By contrast, other authors have shown that hedgerows can function as ecological traps (Major et al. 1999; Estrada et al. 2002).

Reproductive success has the advantage of providing an indication of the long term viability of the population studied (Donovan et al. 1995). The effect of small wooded landscape elements on birds’ reproductive success have not been well studied. For example, there are only a few studies on nest predation in hedgerows and wooded linear strips; most nest predation studies are conducted with artificial nests (Major et al. 1999; Estrada et al. 2002), and do not include data on nest success in different type of hedgerows in relation to natural habitats (Zuria et al. 2007).

The influence of landscape scale factors on nest success has been studied extensively (Saab 1999; Thompson & Mcgarigal 2002; Knutson et al. 2004; Chalfoun & Martin 2007; Gulka 2014). However, when we move from a fine scale (more detailed,
elements are represented as relatively large) to a broad scale (less detailed, elements are represented as relatively small) map we generally loose precision due to increasing grain size (i.e., the smallest spatial resolution of the data) and increasing map extent (i.e., the size of the area portrayed in the map). With coarse grain size and large extent, habitats of interest will be “absorbed” into more predominant land use and land cover (LULC) classes. Only recently LULC maps with fine grain resolution (i.e., small elements are captured and distinguished) at broad scale and large extent are becoming more easily obtainable due to the spatial capability of high resolution satellite images and advances in remote sensing processing software and in object-oriented classification (Vannier & Hubert-Moy 2014).

To address these issues, we characterized broad scale landscape heterogeneity of the Champlain Valley of Vermont by generating fine-grained LULC maps. For the purpose of our research we defined as small landscape elements every wooded element smaller than 1000 m$^2$, or if larger, those that did not maintain any core area when buffered at 50 m (Batáry & Baldi 2004). We defined as large landscape elements all forest elements larger than 1000m$^2$ which maintained core area when buffered at 50 m, and every agriculture element. Utilizing the newly created broad scale/fine-grained maps, we posed four questions that could explain the value of small landscape elements as nesting locations for many species of birds: (1) are small landscape elements used more frequently than large landscape elements as nesting locations?; (2) is daily nest survival rate different in small and large landscape elements?; (3) which fine and broad scale
landscape scale covariates influence daily survival rate?; and (4) are there interactions between landscape-level factors and life history traits such as nest type and height?

4.2. Methods

4.2.1. Study system

We studied 18 circular plots (201 ha) centered along the seven main rivers of the Vermont portion of the Lake Champlain watershed. This agricultural region supports a mix of agriculture and forest with residential areas scattered throughout. The central part of the study area has a greater population density and includes the largest city in Vermont: Burlington. To identify the study plots, we chose a random set of locations on the basis of landscape structure and distance from Burlington. We asked permission from landowners to work on their land, and selected as study plots the first 18 locations for which we obtained permission.

4.2.2. Nest searches

In each plot, we searched for and monitored nests of all avian species. We concentrated the search efforts on forests, riparian habitats, and edges. To find nests we used behavioral observations and systematic searches of suitable habitat. Following discovery, we monitored nests at 2-4 days intervals, for as long as the nest was active, determining its fate as either “failure” or “success.” Potential causes of failure were classified as predation, weather events, abandonment, and human disturbance. We
deemed a nest as successful if we were able to ascertain that at least one host nestling fledged.

4.2.3. Map base layer

We used remote sensing to create a fine-grained land use land cover map (LULC) with 1 meter spatial resolution and 5 m² minimum mapping unit (i.e. the area of the smallest landscape element identified is 5 m²). We identified 10 LULC classes: forest (patches of forest that maintain some core area using a 50 m buffer), small forest (forest patches greater than 1000 m² without core area), trees (treed patches less than 1000 m²), hedgerow (linear treed patches less than 50 m wide and at least 4 times longer than wide), shrub, wetland (ground wet for most of the field season with hydrophilic vegetation), agriculture (hay fields, pastures, crops, and gardens), urban (paved or compacted surfaces, buildings), lawn (mowed areas, cemeteries, and golf courses), and water. For our analyses, we considered forest and agriculture to be large landscape elements and small forest, shrub, trees, and hedgerows to be small landscape elements. We classified landscape elements into the large and small class on the basis of the average and median element area for LULC classes in all plots. Large landscape elements average area (> 5 ha) was tenfold greater than small landscape elements average area (< 0.5 ha). Large landscape elements median area (> 1 ha) was also greater than small landscape elements median area (< 0.15 ha).

We excluded from the nest success analysis the land use classes wetland, urban, lawn, and water. We excluded wetlands because during the classification we were unable
to identify individual trees and shrubs within this class making us unable to attribute this class, or portions of it, to the small or large landscape element categories. Also, we excluded the land use classes urban, water and lawn because $< 5$ nests were built in these landscape classes.

4.2.4. **Effort analysis and habitat use**

We calculated an unbiased estimate of the number of nests present per habitat and of the number of nests found per areas searched on the basis of nest searching events for which effort was considered. Because the area searched to find nests was not equal among plots and habitats, because nest searching was conducted by different technicians, and because nests of different species are more difficult to find than others, in 2011, we recorded the paths of the field technicians using handheld GPS units during a subset of nest searching events. We used these paths to quantify search effort to assess whether birds showed differential habitat use patterns across land use classes.

We recorded nest searching tracks across a variety of portions of the nesting season and times of the day. Because we obtained nest searching GPS tracks of various length that included different numbers of nests (0 to 6), we performed the effort analysis on a randomly chosen subset of 11 tracks per plot (99 total tracks out of 290 total searching tracks recorded). We buffered the linear GPS tracks (1.5 m on either side), and interpolated the searched surfaces with fine scale land use land cover maps using ArcGIS 10.1 (ESRI 2011). We did not merge the buffers obtained for each track to include in our analysis areas searched more than once. We used the interpolated results to calculate the
area (ha) searched per habitat type and we used the LULC layer to determine the habitat associated with each nest. We calculated the number of nests per ha of habitat searched and we also calculated the number of nests per ha in each of the land use classes.

4.2.5. Estimates of nest survival

We used package RMark (Laake 2013) for R software (R Core Team 2014) to estimate daily nest survival rates (DSR - Rotella et al. 2004). We included in the analysis 453 nests of 57 species of birds found in two field seasons. We excluded nests for which the fate was uncertain. We categorized the covariates used in the models into three groups (Table 6): 1) fine landscape scale habitat attributes, 2) broad landscape scale habitat attributes, and 3) avian functional groups (Knutson et al. 2004).

To better understand the factors that influenced nest success at the fine scale we generated a series of covariates using 100m radius (3.14 ha) buffers around each nest. We selected this radius because the average territory size for many passerine species is smaller than 3ha (Poole 2005). The covariates included in our analysis were: habitat (agriculture, forest, small forest, tree, hedgerow, and shrub), habitat group (small or large landscape element), element area (area in hectares of the element in which the nest was found), distance to edge (distance of the nest to the closest habitat different from the nest element), percent agriculture in the 100m buffer surrounding the nest (an index of landscape openness), Shannon-Wiener elements diversity index, and mean element size in the 100m buffer surrounding the nest. The latter two covariates, give us an indication
of the effect of compositional and configurational heterogeneity, respectively on daily survival rate (Fahrig et al. 2011).

We created broad landscape scale habitat covariates considering the entire plot (201 ha) as the landscape extent of interest. We separated the 201 ha plots into 3 categories on the basis of the amount of agriculture and forested elements in the landscape – mostly forest landscape, mostly agricultural landscape, and mixed landscape. Also we included a configuration covariate, where the 201 ha plots were divided into 3 categories on the basis of mean patch size. Mean patch size values were attributed to the simple, mid or complex category using Jenks optimization method (Jenks & Caspall 1971). We also included the covariates mean element size, number of elements, and Shannon-Wiener elements diversity index for the 201 hectares plots.

To capture both aspects of landscape structure (landscape composition and configuration) with one index, we calculated a landscape heterogeneity covariate for both fine and broad landscape scales. We performed a principal component analysis based on the correlation matrix including mean element size and Shannon-Wiener index. We retained only the first principal component which explained more than 80% of the variation for each landscape scale. We used the calculated principal component covariate to test if nests success was influenced by the overall heterogeneity of the landscape at both fine and broad landscape scale.

In the avian functional groups we included “nest height” and “nest type” (nest characteristics covariates) to assess the effects of species’ life history strategies. We included the nest height as a continuous covariate. We also grouped nests on the basis of
type: open cup canopy nesters, open cup shrub nesters, open cup ground nesters, cavity nesters, oven-shaped nests, reed nesters, and other (Ehrlich et al. 1988). In the functional group covariates we also included “habitat guild” in which species were grouped by habitat type on the basis of an occupancy model developed in Vermont (Schwenk & Donovan 2011). We also tested the effect of the covariates year, and time (non-constant DSR between each day of the nest monitoring season).

We used corrected Akaike’s Information criterion (AICc) to compare regression models including each covariate individually. We excluded from the analysis models with AICc scores lower than the null model in which DSR was maintained as constant without the influence of any covariate. For models that performed better than the null model, we compared DSRs and assessed significance by determining if the confidence intervals overlapped one another.

4.3. Results

4.3.1. Nest searches

We found and established the fate for 202 nests in 2010 and 251 nests in 2011 for a total of 453 nests of 57 species (Table 7). More than half of the nests were successful (252). Of the 201 failed nests, 113 were definitively determined to have failed as a result of depredation.

More nests were found in agricultural (188) and mixed (171) plots, than in forest plots (94). Nests were distributed more evenly across the gradient of configuration complexity (166, 154, and 133 in mid-, simple, and complex plots).
4.3.2. Effort analysis and habitat use

*Are small landscape elements used more frequently than large landscape elements as nesting locations?*

We covered more area searching in large landscape elements (agriculture and forest; 71% of the area searched) than in small landscape elements (hedgerow, shrub, small forest, and trees; 16% of area searched). However, we searched a greater proportion of small landscape elements.

After adjusting for nest searching effort, we found 2.3 nests/ha in small landscape elements, versus 0.9 nests/ha in large landscape elements. Excluding LULC classes where we did not find any nests (lawn, urban, and water), we found the fewest nests per area searched in agricultural habitats, and the greatest number of nests per area searched in trees and hedgerows (Table 8).

Agricultural (1.6 nest/ha searched) and mixed (1.3 nest/ha searched) plots, had a greater nest density than forest plots (0.7 nest/ha searched). Mid complexity and complex plots had a greater nest density (mid: 1.5 nest/ha searched, complex: 1.3 nest/ha searched) than simple plots (0.9 nest/ha searched). More heterogeneous landscapes (both compositionally and configurationally) had on average a greater (1.5 nest/ha) nest density than less heterogeneous landscapes (1.0 nest/ha).
4.3.3. Estimates of nest survival

*Is daily nest survival rate different in small and large landscape elements?*

DSR estimates for large and small elements (habitat group covariate) were not significantly different.

*Which fine and broad scale landscape scale covariates influence daily survival rate?*

Three models including fine landscape scale habitat covariates performed better than the null model. The models including broad scale habitat covariates did not perform better than the null model (Table 9). We did not find an effect of year or time between visits to nests. We analyzed the performance of additive and interaction models for most covariates that performed better or close to the null model AICc value. None of these more complex models performed better than the single covariates models.

The best performing models in the DSR analysis included covariates belonging to the functional groups. The model with the strongest support was the nest type model. However, nest height, habitat and element area models were more than 2 AIC scores greater than the null model indicating that these models have considerably less support (Burnham & Anderson 1998).
**Fine landscape scale group**

Nests were distributed in the following habitats: forest 142, shrub 111, trees 76, hedgerow 50, small forest 41, and agriculture 33. The majority of these nests were in small landscape elements (278).

The model including the covariate habitat (= patch or element in which the nest is located), was the best model in the fine scale landscape group models. Nests in trees had significantly greater DSR than nests in agriculture (Figure 14). Nests were more successful in forest, small forest, and trees habitat classes (confidence intervals for beta parameters did not include 0).

The area of the smallest landscape element in which we found a nest was 0.0015 ha, and the largest was 104 ha (median: 5.19 ha mean: 20.52 ha). The median distance from the edge was 4.5 m and the mean was 20.9 m (range = a few centimeters, to 619 meters). We found 365 nests within 20 m from an edge, 57 nests were more than 40 m away from an edge, and 31 nests were between 20 and 40 meters from an edge. The element area and distance to edge models however, performed just slightly better than the null model. Because the AICc value for these two models was within 2 units from the null model, we can assume that the inclusion of these covariates did not significantly improve the performance of the model.

**Broad landscape scale group**

Support for the models including the broad landscape scale covariates composition treatment (prevalence of agriculture or forest in the landscape), and
configuration treatment (small mean patch size for complex plots, medium mean patch size for medium complexity plots, and large mean patch size for simple plots), were not as well supported as the null model.

*Are there interactions between landscape-level factors and life history traits?*

Most nests (232) belonged to species associated with edge and open areas, 167 nests were associated with forest edge habitats, 32 nests were associated with interior forest, and 22 nests belonged to other guilds (Schwenk & Donovan 2011). We identified 7 nests types: open cup canopy nesters 146, open cup shrub nesters 138, open cup ground nesters 53, cavity nesters 50, oven shaped nests 12, reed nesters 27, other types of nests (saucer 9, pendant 9, platform 4, scrape 3, cup 1, cavity 1) 27 (Ehrlich et al. 1988). Daily survival rate was significantly greater for cavity nests compared to any other nest type. The model including habitat guild was less well supported than the null model. Ovenbirds and red-winged blackbirds (reeds nesters) had the lowest DSR, but the confidence interval for these two types of nests were so wide that we cannot consider their DSR significantly less than other nests types (Figure 15).

We found a significant effect of nest height on nest success. The height of the nests varied from 0 to 30 meters above ground, with a median of 1.7 m and a mean of 2.8 m. Nest height was the second best performing model. Despite the low model weight, nest height had an influence on nest survival. The daily survival rate was lowest for ground nesting species and increased with nest height (Figure 16).
We did not have enough data for individual species to investigate the effect of nesting habitat on their DSR except for American robin (AMRO). For AMRO only two variables performed better than the null model: percentage of agriculture at the fine landscape scale (2 AICc values greater than null model), and configurational treatment at the broad landscape scale (3 AICc values greater than null model). However, within each of these models, there was no significant difference in DSR. Daily survival rate for robin nests was lowest in complex habitats (0.939, SE = 0.021), simple plots had intermediate DSR value (0.956, SE = 0.012), and DSR was greatest in mid-complexity plots (0.989, SE = 0.008). Daily survival rate for robin nests increased with increasing proportion of agriculture in the 100 m plots. Even if models including these two covariates performed better than the null model, the configurational treatment confidence intervals overlapped among all categories, and the confidence intervals for the beta estimate of percentage of agriculture included zero.

We also analyzed the relationship between DSR and landscape covariates for the two most common nest types: shrub-open cup and canopy-open cup nests. All the other nest type groups did not have sufficiently large sample size for precise estimates. For canopy nesters, the model with the best support was treatment (broad landscape). Several other models (nest height, percentage of agriculture, land structure treatment, element area, and percent of forest, ordered by AICc values) performed better than the null model, but were within two AICc units from the null model. Canopy nests in agricultural plots had a significantly greater DSR (0.977, SE = 0.005) than canopy nests in mix plots (0.949, SE = 0.008). Canopy nests in forest plots had a DSR value of 0.968 (SE = 0.009).
For shrub nesters, the model with the best support was Shannon-Weiner diversity (fine landscape scale). Several other models (area of the element, configuration treatment, and agriculture percentage, ordered by AICc values) performed better than the null model, but were within two AICc units from the null model. Shrub nests had greater DSR with increasing diversity within the 100 m radius plot (Figure 17).

### 4.4. Discussion

Our research confirmed that (1) small landscape elements are used more often than large landscape elements as nesting location; (2) DSR does not differ between small and large landscape elements; (3) Daily survival rate is influenced partially by fine landscape covariates, only slightly by broad landscape covariates, and most strongly by nest characteristics covariates; and (4) nest success is influenced by life history traits.

Many species use small landscape elements for nesting. We found that the density of nests was greater in small landscape elements (trees, hedgerows, shrubs, and small forests) and near edges than in other landscape elements. Even if most of the nests found in the small elements belonged to edge tolerant species, their nests were at least as successful in small landscape elements as in large landscape elements.

Nest predation has been shown to be one of the main causes of reproductive failure for birds (Batáry & Baldi 2004; Zuria et al. 2007). Because many predators tend to hunt along ecotones, or the border between different habitat elements, small elements and edges of larger elements have been considered ecological traps for some avian species (Robertson et al. 2006; Woodward et al. 2014). Even though our research did not
focus on predation, our results showed that DSR was similar in large and small landscape elements. Further, distance to edge did not influence daily survival rates, supporting the hypothesis that small elements are not ecological traps. At least for the many edge-tolerant species that constituted the majority of our sample, small landscape elements did not negatively affect reproductive success in these habitats. Our study did not use artificial nests to test the effect of predation on nest success. However, the majority of the failed nests were ascribed with certainty to depredation (113 nest of the 201 failed nests); for many of the other failures, we did not have enough evidence to attribute the failure to a specific cause. A large portion of studies that have tested nest success across landscape attributes have used artificial nests and mostly focused on forest species (Major et al. 1999; Estrada et al. 2002; Peak et al. 2004; Zuria et al. 2007). Because the reliability of studies using artificial nests is questioned (Batáry & Baldi 2004), our results obtained with natural nests, provide a stronger support to the hypothesis that predation intensity is not greater along element edges.

Nest site information included the most informative covariates regarding DSR. The type of nest built by each species and nest height were the two covariates that belonged to the best performing models. Because we performed our analysis at the avian community level rather than focusing on single species, we were not able to address the influence that landscape characteristic might have on nest survival rates for a particular species. Separate analyses for American robin, shrub, and canopy nesters gave different results. Models with landscape covariates had stronger support than models with nest characteristics. For canopy nesters, openness of the landscape at a broad scale was
important. For shrub nesters the diversity of landscape elements at fine scale was the most influential covariate. Nests had greater daily survival rates when associated with more diverse local landscapes.

Several processes might be at play when studying the importance of small landscape elements. Small elements may complement other landscape elements in providing resources for certain species (Dunning et al. 1992). For example a species that relies on agricultural fields for foraging might use hedgerows as a nesting or perching location. Heterogeneous agricultural landscapes consist of a variety of land uses (e.g., hayfields, row crops, farmsteads) as well as natural and semi-natural elements (hedgerow, woodland, wetland, single trees, shrub areas etc.) that are interspersed across the landscape. Small natural landscape elements such as treed habitats in homogeneous agricultural landscapes could be of particular importance for nesting cover.

Fahrig et al. (2011) described two components of structural heterogeneity: variety of cover types (compositional heterogeneity) and spatial patterning (configurational heterogeneity). To test these effects, we included the Shannon-Weiner index (composition) and mean patch size (configuration) at both fine and broad scales. Only the model that included mean patch size at a broad scale performed better than the null model, implying that landscapes with a complex configuration (greater number of small patches) at the broad scale have a positive influence on the avian community reproductive success.

With the increasing availability of high resolution imagery and of software that can produce LULC maps similar to the ones digitized manually, more accurate analysis
of the influence of the landscape structure on organisms will become more commonplace.

However, computer-generated LULC maps obtained through remote sensing classification are still imperfect without time consuming checks performed by the user. Also, decisions about which LULC classes to include or exclude can influence the outcome of the analysis (St-louis et al. 2014). For example, we excluded wetlands from our analysis because during the classification it was impossible to identify individual trees and shrubs. The DSR was greatest in the wetland class, but we were not able to attribute this class to either small or large elements. In some cases a large portion of the plot was comprised of wetlands that supported trees and shrubs of different heights. The matrix in which the wooded elements are embedded determines the role of these elements for the species that rely on them. Using detailed fine grain maps that can discern small wooded elements is a cheaper and more efficient alternative to collecting field data on the extent and distribution of small landscape elements.

Most of the species that we monitored during our research were edge tolerant generalists. Trends for both Vermont and New England obtained from the North American breeding bird survey (Sauer et al. 2014), showed that several of these species are declining. Song sparrow, red-winged blackbird, eastern kingbird, Baltimore oriole, and chestnut-sided warbler are showing significant declines across Vermont and New England. Yellow warbler and American redstart also show significant declining trends in Vermont. Common yellowthroat show significant declining trends in New England. Only two of the species for which we monitored more than 5 nests showed significantly increasing trends, cedar waxwing and northern cardinal.
Because we found that small landscape elements were selected more often as nesting locations, and that the nests built in small landscape elements are as successful as the ones built in larger landscape elements, we recommend landowners maintain these small landscape elements. Simple management activities such as maintaining and planting hedgerows along field borders, planting trees, and promoting shrubby areas in unused fields will enhance the landscape heterogeneity and provide breeding habitat for declining wide variety of bird species.
### 4.5. Tables

Table 6. Models and covariates used to estimate nest survival. We categorized the covariates used in the models into three groups. Different covariates were used to address the specific questions presented at the end of the introduction.

<table>
<thead>
<tr>
<th>Covariate groups</th>
<th>Covariate and question</th>
<th>Definition</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fine landscape scale habitat attributes</td>
<td>Habitat group</td>
<td>2 Habitat in which the nest was found belong to a small landscape element (small forest, tree, hedgerow, and shrub) or a large landscape element (agriculture and forest)</td>
</tr>
<tr>
<td></td>
<td>Habitat</td>
<td>3 Habitat in which the nest was found (agriculture, forest, small forest, tree, hedgerow, and shrub)</td>
</tr>
<tr>
<td>(Covariates generated using 3.14 ha landscapes)</td>
<td>Element area</td>
<td>3 Area in hectares of the element in which the nest was found</td>
</tr>
<tr>
<td></td>
<td>Distance to edge</td>
<td>3 Distance of the nest to the closest habitat different from the nest element</td>
</tr>
<tr>
<td></td>
<td>Percent agriculture</td>
<td>3 Percent agriculture in the 100m buffer surrounding the nest (an index of landscape openness)</td>
</tr>
<tr>
<td></td>
<td>Shannon-Wiener elements diversity index</td>
<td>3 This diversity index provide an indication of the effect of compositional heterogeneity on DSR</td>
</tr>
<tr>
<td></td>
<td>Mean element size</td>
<td>3 This diversity index provide an indication of the effect of configurational heterogeneity on DSR</td>
</tr>
<tr>
<td></td>
<td>Landscape heterogeneity</td>
<td>3 Heterogeneity index obtained using Principal Component Analysis on the previous two covariates</td>
</tr>
<tr>
<td>Broad landscape scale habitat attributes</td>
<td>Landscape composition categories</td>
<td>3 Plots divided into 3 categories on the basis of landscape composition (amount of agriculture and forest in the landscape): mostly forested, mixed and mostly agricultural plots</td>
</tr>
<tr>
<td>(Covariates generated using 201 ha landscapes)</td>
<td>Landscape configuration categories</td>
<td>3 Plots divided into 3 categories on the basis of landscape configuration (mean element size in the landscape): simple (large mean element size), mid, and complex (small mean patch size)</td>
</tr>
<tr>
<td></td>
<td>Shannon-Wiener elements diversity index</td>
<td>3 This diversity index provide an indication of the effect of compositional heterogeneity on DSR</td>
</tr>
<tr>
<td></td>
<td>Mean element size</td>
<td>3 This diversity index provide an indication of the effect of configurational heterogeneity on DSR</td>
</tr>
<tr>
<td></td>
<td>Landscape heterogeneity</td>
<td>3 Heterogeneity index obtained using Principal Component Analysis on the previous two covariates</td>
</tr>
<tr>
<td>Avian functional groups</td>
<td>Nest height</td>
<td>4 Height at which the nest was found in meters</td>
</tr>
<tr>
<td></td>
<td>Nest type</td>
<td>4 Nests grouped on the basis of nest type: open cup canopy nesters, open cup shrub nesters, open cup ground nesters, cavity nesters, oven-shaped nests, reed nesters, and other (Ehrlich et al. 1998)</td>
</tr>
<tr>
<td></td>
<td>Habitat guild</td>
<td>4 Species were grouped by habitat type on the basis of an occupancy model developed in Vermont (Schwenk &amp; Donovan 2011)</td>
</tr>
</tbody>
</table>
Table 7. Species, habitat, nest type, and number of nests found during the study. Each species was attributed to one of the 4 habitat guilds identified by Schwenk & Donovan (2011), and to 1 of the 7 nest types. Nests were found in 2010 and 2011 in 18 plots distributed throughout the Champlain Valley of Vermont.

<table>
<thead>
<tr>
<th>Species name</th>
<th>Scientific name</th>
<th>habitat guild</th>
<th>nest type</th>
<th># of nests</th>
</tr>
</thead>
<tbody>
<tr>
<td>American Robin</td>
<td>Turdus migratorius</td>
<td>forest edge</td>
<td>open cup canopy</td>
<td>53</td>
</tr>
<tr>
<td>Cedar Waxwing</td>
<td>Bombycilla cedrorum</td>
<td>forest edge</td>
<td>open cup canopy</td>
<td>51</td>
</tr>
<tr>
<td>Yellow Warbler</td>
<td>Setophaga petechia</td>
<td>edge and open</td>
<td>open cup shrub</td>
<td>51</td>
</tr>
<tr>
<td>Gray Catbird</td>
<td>Dumetella carolinensis</td>
<td>edge and open</td>
<td>open cup shrub</td>
<td>40</td>
</tr>
<tr>
<td>Song Sparrow</td>
<td>Melospiza melodia</td>
<td>edge and open</td>
<td>open cup ground</td>
<td>31</td>
</tr>
<tr>
<td>Red-winged Blackbird</td>
<td>Agelaius phoeniceus</td>
<td>edge and open</td>
<td>reed</td>
<td>27</td>
</tr>
<tr>
<td>Eastern Kingbird</td>
<td>Tyrannus tyrannus</td>
<td>edge and open</td>
<td>open cup canopy</td>
<td>17</td>
</tr>
<tr>
<td>Black-capped Chickadee</td>
<td>Poecile atricapillus</td>
<td>forest edge</td>
<td>cavity</td>
<td>16</td>
</tr>
<tr>
<td>Veery</td>
<td>Catharus fusciscens</td>
<td>forest edge</td>
<td>open cup ground</td>
<td>13</td>
</tr>
<tr>
<td>American Goldfinch</td>
<td>Spinus tristis</td>
<td>edge and open</td>
<td>open cup shrub</td>
<td>12</td>
</tr>
<tr>
<td>Ovenbird</td>
<td>Seiurus aurocapilla</td>
<td>interior forest</td>
<td>oven shaped</td>
<td>12</td>
</tr>
<tr>
<td>Mourning Dove</td>
<td>Zenaida macroura</td>
<td>edge and open</td>
<td>other</td>
<td>9</td>
</tr>
<tr>
<td>Baltimore Oriole</td>
<td>Icterus galbula</td>
<td>edge and open</td>
<td>other</td>
<td>8</td>
</tr>
<tr>
<td>Chestnut-sided Warbler</td>
<td>Setophaga pensylvanica</td>
<td>forest edge</td>
<td>open cup shrub</td>
<td>8</td>
</tr>
<tr>
<td>Northern Cardinal</td>
<td>Cardinalis cardinalis</td>
<td>edge and open</td>
<td>open cup shrub</td>
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<tr>
<td>American Redstart</td>
<td>Setophaga ruticilla</td>
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<td>open cup canopy</td>
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<tr>
<td>Yellow-bellied Sapsucker</td>
<td>Sphyrapicus varius</td>
<td>interior forest</td>
<td>cavity</td>
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<tr>
<td>Common Yellowthroat</td>
<td>Geothlypis trichas</td>
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<tr>
<td>Downy Woodpecker</td>
<td>Picoides pubescens</td>
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<td>cavity</td>
<td>6</td>
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<tr>
<td>Tree Swallow</td>
<td>Tachycineta bicolor</td>
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<td>cavity</td>
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<tr>
<td>Black-throated Blue Warbler</td>
<td>Setophaga caeruleascens</td>
<td>interior forest</td>
<td>open cup shrub</td>
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<tr>
<td>House Wren</td>
<td>Troglodytes aedon</td>
<td>edge and open</td>
<td>cavity</td>
<td>4</td>
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<tr>
<td>Red-eyed Vireo</td>
<td>Vireo olivaceus</td>
<td>interior forest</td>
<td>open cup canopy</td>
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<tr>
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<td>Coccyzus erythropthalmus</td>
<td>other</td>
<td>other</td>
<td>3</td>
</tr>
<tr>
<td>Hairy Woodpecker</td>
<td>Picoides villosus</td>
<td>interior forest</td>
<td>cavity</td>
<td>3</td>
</tr>
<tr>
<td>Indigo Bunting</td>
<td>Passerina cyanea</td>
<td>edge and open</td>
<td>open cup shrub</td>
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</tr>
<tr>
<td>Willow Flycatcher</td>
<td>Empidonax traillii</td>
<td>edge and open</td>
<td>open cup shrub</td>
<td>3</td>
</tr>
<tr>
<td>Wild Turkey</td>
<td>Meleagris gallopavo</td>
<td>other</td>
<td>other</td>
<td>3</td>
</tr>
<tr>
<td>Wood Thrush</td>
<td>Hylcichla mustelina</td>
<td>forest edge</td>
<td>open cup canopy</td>
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</tr>
<tr>
<td>Blue-headed Vireo</td>
<td>Vireo solitarius</td>
<td>other</td>
<td>open cup canopy</td>
<td>2</td>
</tr>
<tr>
<td>Empidonax spp.</td>
<td></td>
<td>edge and open</td>
<td>open cup shrub</td>
<td>2</td>
</tr>
<tr>
<td>Eastern Wood-Pewee</td>
<td>Contopus virens</td>
<td>forest edge</td>
<td>open cup canopy</td>
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<tr>
<td>Hermit Thrush</td>
<td>Catharus guttatus</td>
<td>interior forest</td>
<td>open cup ground</td>
<td>2</td>
</tr>
<tr>
<td>House Sparrow</td>
<td>Passer domesticus</td>
<td>edge and open</td>
<td>cavity</td>
<td>2</td>
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100
<table>
<thead>
<tr>
<th>Species</th>
<th>Scientific Name</th>
<th>Habitat</th>
<th>Nest Type</th>
<th>Nest Site</th>
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</tr>
<tr>
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<td>open cup shrub</td>
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<tr>
<td>Black-and-white Warbler</td>
<td><em>Mniotilta varia</em></td>
<td>forest edge</td>
<td>open cup ground</td>
<td>1</td>
</tr>
<tr>
<td>Brown Creeper</td>
<td><em>Certhia americana</em></td>
<td>other</td>
<td>other</td>
<td>1</td>
</tr>
<tr>
<td>Brown Thrasher</td>
<td><em>Toxostoma rufum</em></td>
<td>other</td>
<td>other</td>
<td>1</td>
</tr>
<tr>
<td>Broad-winged Hawk</td>
<td><em>Buteo platypterus</em></td>
<td>other</td>
<td>other</td>
<td>1</td>
</tr>
<tr>
<td>Chipping Sparrow</td>
<td><em>Spizella passerina</em></td>
<td>edge and open</td>
<td>open cup canopy</td>
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<tr>
<td>Eastern Wood-Pewee</td>
<td><em>Contopus virens</em></td>
<td>forest edge</td>
<td>open cup canopy</td>
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</tr>
<tr>
<td>European Starling</td>
<td><em>Sturnus vulgaris</em></td>
<td>edge and open</td>
<td>cavity</td>
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</tr>
<tr>
<td>Great Crested Flycatcher</td>
<td><em>Myiarchus crinitus</em></td>
<td>other</td>
<td>cavity</td>
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</tr>
<tr>
<td>Louisiana Waterthrush</td>
<td><em>Parkesia motacilla</em></td>
<td>other</td>
<td>open cup ground</td>
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</tr>
<tr>
<td>Mourning Warbler</td>
<td><em>Geothlypis philadelphia</em></td>
<td>forest edge</td>
<td>open cup ground</td>
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</tr>
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<td>Orchard Oriole</td>
<td><em>Icterus spurius</em></td>
<td>other</td>
<td>other</td>
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<td>Rose-breasted Grosbeak</td>
<td><em>Pheucticus ludovicianus</em></td>
<td>forest edge</td>
<td>open cup canopy</td>
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</tr>
<tr>
<td>Red-breasted Nuthatch</td>
<td><em>Sitta canadensis</em></td>
<td>other</td>
<td>cavity</td>
<td>1</td>
</tr>
<tr>
<td>Ruby-throated Hummingbird</td>
<td><em>Archilochus colubris</em></td>
<td>forest edge</td>
<td>open cup canopy</td>
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</tr>
<tr>
<td>Savannah Sparrow</td>
<td><em>Passerculus sandwichensis</em></td>
<td>other</td>
<td>open cup ground</td>
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</tr>
<tr>
<td>Tufted Titmouse</td>
<td><em>Baeolophus bicolor</em></td>
<td>edge and open</td>
<td>cavity</td>
<td>1</td>
</tr>
<tr>
<td>Warbling Vireo</td>
<td><em>Vireo gilvus</em></td>
<td>other</td>
<td>open cup canopy</td>
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</tr>
<tr>
<td>White-breasted Nuthatch</td>
<td><em>Sitta carolinensis</em></td>
<td>forest edge</td>
<td>cavity</td>
<td>1</td>
</tr>
<tr>
<td>Winter Wren</td>
<td><em>Troglodytes hiemalis</em></td>
<td>other</td>
<td>cavity</td>
<td>1</td>
</tr>
<tr>
<td>White-throated Sparrow</td>
<td><em>Zonotrichia albicollis</em></td>
<td>other</td>
<td>open cup ground</td>
<td>1</td>
</tr>
<tr>
<td>Eastern Bluebird</td>
<td><em>Sialia sialis</em></td>
<td>other</td>
<td>cavity</td>
<td>1</td>
</tr>
</tbody>
</table>
Table 8. Effort analysis summary based on a subset of 11 tracks per plot summing to 99 tracks total. Land use/land cover categories are ranked by number of nests per ha searched. Nests were found and monitored in 2011 in 9 plots located in the Champlain Valley of Vermont (USA).

<table>
<thead>
<tr>
<th>LULC</th>
<th>Area searched (ha)</th>
<th>Total area (ha)</th>
<th>Percent searched of total area</th>
<th>number of nests</th>
<th>Nests per ha searched</th>
</tr>
</thead>
<tbody>
<tr>
<td>Trees</td>
<td>0.9</td>
<td>15.1</td>
<td>5.9%</td>
<td>4</td>
<td>4.5</td>
</tr>
<tr>
<td>Hedgerow</td>
<td>2.4</td>
<td>19.6</td>
<td>12.2%</td>
<td>7</td>
<td>2.9</td>
</tr>
<tr>
<td>Wetland</td>
<td>2.8</td>
<td>69.1</td>
<td>4.1%</td>
<td>6</td>
<td>2.1</td>
</tr>
<tr>
<td>Shrub</td>
<td>3.5</td>
<td>90.8</td>
<td>3.8%</td>
<td>6</td>
<td>1.7</td>
</tr>
<tr>
<td>Small forest</td>
<td>1.6</td>
<td>30.4</td>
<td>5.3%</td>
<td>2</td>
<td>1.2</td>
</tr>
<tr>
<td>Forest</td>
<td>25.8</td>
<td>895.1</td>
<td>2.9%</td>
<td>26</td>
<td>1.0</td>
</tr>
<tr>
<td>Agriculture</td>
<td>10.5</td>
<td>621.0</td>
<td>1.7%</td>
<td>7</td>
<td>0.7</td>
</tr>
<tr>
<td>Lawn</td>
<td>0.1</td>
<td>21.2</td>
<td>0.4%</td>
<td>0</td>
<td>0.0</td>
</tr>
<tr>
<td>Urban</td>
<td>2.8</td>
<td>28.8</td>
<td>9.7%</td>
<td>0</td>
<td>0.0</td>
</tr>
<tr>
<td>Water</td>
<td>0.8</td>
<td>49.8</td>
<td>1.6%</td>
<td>0</td>
<td>0.0</td>
</tr>
<tr>
<td>Total</td>
<td>51.2</td>
<td>1840.9</td>
<td>58</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Table 9. RMark analysis from 453 nests monitored in 2010 and 2011 in the Champlain Valley, VT. Results are ordered by AICc scores (lowest to highest). Only models performing better than the null model are included in this table.

<table>
<thead>
<tr>
<th>Group</th>
<th>Model</th>
<th>npar</th>
<th>AICc</th>
<th>Delta AICc</th>
<th>Weight</th>
</tr>
</thead>
<tbody>
<tr>
<td>Functional Groups</td>
<td>Nest type</td>
<td>7</td>
<td>1182.7</td>
<td>0</td>
<td>0.992</td>
</tr>
<tr>
<td>Functional Groups</td>
<td>Nest height</td>
<td>2</td>
<td>1192.3</td>
<td>9.61</td>
<td>0.008</td>
</tr>
<tr>
<td>Fine landscape scale</td>
<td>Habitat</td>
<td>6</td>
<td>1206.8</td>
<td>24.12</td>
<td>0</td>
</tr>
<tr>
<td>Fine landscape scale</td>
<td>Element area</td>
<td>2</td>
<td>1207.5</td>
<td>24.73</td>
<td>0</td>
</tr>
<tr>
<td>Fine landscape scale</td>
<td>Distance to edge</td>
<td>2</td>
<td>1208.4</td>
<td>25.7</td>
<td>0</td>
</tr>
<tr>
<td>Time</td>
<td>Time</td>
<td>2</td>
<td>1209.4</td>
<td>26.67</td>
<td>0</td>
</tr>
<tr>
<td>Broad landscape scale</td>
<td>Mean patch size</td>
<td>2</td>
<td>1209.4</td>
<td>26.7</td>
<td>0</td>
</tr>
<tr>
<td>Year</td>
<td>Year</td>
<td>2</td>
<td>1209.5</td>
<td>26.78</td>
<td>0</td>
</tr>
</tbody>
</table>
4.6. Figures

**DSR per habitat type**

![Bar chart showing daily survival rates (DSR) per habitat type.](chart1)

Figure 14: Daily survival rates by habitat type at fine landscape scale. Nests in trees had significantly higher DSR than nests in agriculture. All other habitats did not significantly differ from each other. Nests were monitored in 2010 and 2011 in 18 plots located in the Champlain Valley of Vermont (USA).

**DSR per nest type**

![Bar chart showing daily survival rates (DSR) per nest type.](chart2)

Figure 15: Daily survival rate by nest type. Daily survival rate was significantly greater for cavity nests compared to any other nest type. Nests were monitored in 2010 and 2011 in 18 plots located in the Champlain Valley of Vermont (USA).
Figure 16: Daily survival rate by nest height in meters. The daily survival rate was lowest for ground nesting species and increased with nest height. Nests were monitored in 2010 and 2011 in 18 plots located in the Champlain Valley of Vermont (USA).
Figure 17: Daily survival rate for shrub nesters increased with increasing landscape diversity (Shannon-Wiener elements diversity index) within plots. Nests were monitored in 2010 and 2011 in 18 plots located in the Champlain Valley of Vermont (USA).
4.7. Literature cited


ESRI. 2011. ArcGIS. Environmental Systems Research Institute, Redlands, CA.


Gulka, A. 2014. Effects of Habitat Fragmentation and Spatial Scale on Avian Nest Success in the Champlain Valley of Vermont.


CHAPTER 5: TEMPERATURE LOGGERS DECREASE COSTS OF 
DETERMINING BIRD NEST SURVIVAL

5.1. Abstract

We tested whether temperature loggers (iButtons®) placed inside bird nests could be used to estimate the date the nest hatched or failed and compared their efficiency and cost to observational data. Temperature loggers were more precise (approx. 1 hr error vs. 1–3-days error for researcher visit) in estimating fate date for 28 nests of 15 species of birds monitored in the Champlain Valley of Vermont (USA) between May and August 2010. Our nest sample represented a wide range of nest characteristics, and considerably expanded the range of species whose nests have been monitored successfully with temperature data loggers. The use of iButtons would have reduced the number of visits to the nest by 15%, decreasing disturbance, and allowing for more time spent searching for additional nests, or for employment of a smaller field crew.

5.2. Introduction

Nest survival has been used to obtain information on avian habitat quality (Robinson et al. 1995), community and population dynamics (Holmes et al. 1996), life-history strategies (Grant et al. 2005), predation pressure (Martin 1993, Robinson et al. 1995), reproductive rate (Rotella et al. 2004), and breeding biology (Dinsmore et al. 2002). Estimation of nest survival, or the probability that a nest will be successful (i.e., ≥1 nestling fledges), has gone through several revisions in the past few decades to rectify
errors in the estimation of this parameter (Dinsmore et al. 2002). The errors inherent in various estimation methods are related to uncertainties due to absence of the complete history of the nest. First, nests are generally not found when the building process starts and second, researchers are not sure of the exact date of final nest fate. Finding nests late in the nest cycle (when eggs or fledglings are already present) is a difficult problem to overcome, but a precise estimate of the date when the nest hatched or failed (fate date) can now be attained using temperature data loggers.

Temperature loggers that collect and record the temperature of the surrounding environment have been used recently to estimate nest fate date, assess nest predation, and quantify nest attendance patterns (Hartman and Oring 2006, Weidinger 2006, Schneider and McWilliams 2007). Temperature loggers were able to assess the difference in temperature between the nest and the environment and determine the time at which the temperatures equilibrated due to fledging or depredation.

Weidinger (2006) attempted to judge the efficiency of temperature loggers in comparison to video monitoring to determine fate date, time of fate, and predation on several monitored passerine species’ nests. His results supported the use of temperature loggers as efficient, inexpensive, and reliable tools to more precisely record the timing of nest fate and better distinguish between predation and success. Assessment of the efficacy of the Thermochron iButtons® temperature data loggers (Maxim, Sunnyvale, CA, http://www.maxim-ic.com/) for nest monitoring and nest attendance patterns have already been conducted; however, these studies only targeted 2 species (Hartman and Oring 2006, Schneider and McWilliams 2007). We used iButtons to monitor nests of 16
common bird species with diverse nesting structures. The goals of this study were 1) compare the estimated fate date in nests monitored with iButtons with the fate date in visited nests; 2) develop a quick and efficient way to deploy iButtons to minimize nest disturbance, maximize data retrieval, and avoid iButton loss or removal; 3) assess the suitability of temperature data loggers as substitutes for more frequent nest visits; and 4) determine the potential cost savings in using iButtons to monitor nests rather than field personnel.

5.3. Study area

The Champlain Valley of Vermont was an agricultural region bounded by the Green Mountains to the west and Lake Champlain to the east. This area supported a mix of agriculture and forest with residential areas scattered throughout. The central part of the Champlain Valley had a greater population density and included the largest city in Vermont: Burlington.

As part of a larger study of agricultural multi-functionality and landscape ecology, we randomly identified 60 circular (800-m-radius) plots along the 7 main rivers (Missisquoi, Lamoille, Winooski, LaPlatte, Lewis, Little Otter Creek, Otter Creek) of the Champlain Valley. We selected 9 of these plots on the basis of landscape composition and configuration and distance from Burlington to collect information on nest survival.
5.4. Methods

Starting in mid-May, we searched for nests within the 9 plots. We concentrated our search efforts in forests, riparian habitats, and edges. Because of previous work with grassland birds in this area, we did not search grasslands for nests (Perlut et al. 2006). To find nests, we used behavioral cues and systematic searches of grasses, shrubs, and trees (Martin and Geupel 1993). Following discovery, we monitored nests for as long as the nest was active, determining its fate as either ‘failure’ or ‘success.’ Potential causes of failure were classified as predation, weather events, human disturbance, and unknown. We determined a nest was successful if we were able to ascertain that ≥1 young fledged (or in the case of nests containing brown-headed cowbird [Molothrus ater] eggs or young, fledging by ≥1 host young). A nest was determined to be successful if any of the following were identified: flattened nest rim, feces in or close to the nest, feather sheaths in the nest, parents carrying food, fledglings near the nest, parents distressed and/or giving alarm calls.

To monitor the temperature inside the nests, we used dime-sized (16-mm-diam, 6-mm-thickness, 2.9-g) temperature loggers (iButton®—models DS1921G-F5 and DS1922L), which record and store temperature data (range = −40–85° C) at intervals set by the researcher. iButtons placed in nests were paired with ‘control’ iButtons placed near the nest to record ambient temperature. Control iButtons were positioned on trees, generally within 20 m of the nest (not so close as to attract predators and not so far away as to record different microclimates), using flagging tape wrapped around the trunk. We placed iButtons in 34 nests, most of which were below 2 m in height. We refined the
iButton preparation and deployment technique, which consisted of 1) submerging half of the metal surface of the iButton in light blue or yellow ‘Plasti-Dip’ (Plasti Dip International, Blaine, MN); 2) preparing the buttons by attaching monofilament (fishing line) with glue at the plastic surface; 3) sending the monofilament line through the bottom or side of the nest structure with a sewing needle, taking care to avoid damaging the nest; 4) securing the monofilament to surrounding vegetation anchor points (branches or herbaceous stems); and 5) keeping the iButton flush to the inner nest surface among or just to the side of the eggs to discourage removal and keep the iButton in contact with the eggs. We were able to deploy iButtons in <3 minutes. To minimize predator discovery of nests as a result of researcher activity, we approached all nests using indirect routes and by changing the route taken to reach and leave the nest. We monitored nests with iButtons as frequently as nests without iButtons (every 2–3 days) to maintain equal levels of human activity. Once nests fledged or failed, we removed nest and control iButtons and downloaded the data to a computer. To estimate the time and date of fledging or nest failure, we compared temperature data from the control and nest iButtons to assess when the nest temperatures decreased to ambient levels (Fig. 1a and b). We designated ‘fate time’ as the time at which we recorded the last change in temperature direction (from increasing to decreasing), in the graph of the nest iButton temperature, before its temperature reached the control ambient temperature (= equilibrium temp; Fig. 1b). Fate date was defined as the day during which we recorded fate time. For the estimation of fate time and date, we assumed that young birds did not spend too much time on the nest
rim (thus technically to be considered still in the nest), but ‘jumped’ completely outside the nest when fledging.

We used a chi-square test to evaluate whether the proportion of successful nests differed ($\alpha = 0.05$) between nests with and without temperature loggers (Zar 1999). We measured effect size of nest success and iButton presence in the nest using the Cramer $\phi_c$ coefficient (Zar 1999). We estimated daily survival rates using Program Mark (White and Burnham 1999), and used 95% confidence intervals to evaluate differences between nest with and without temperature loggers. We did not include nests with uncertain fates in this analysis. We used evidence-based estimated fate dates for successful nests.

To calculate time and personnel saved by the use of iButtons, we estimated the reduction in the number of nest visits on the basis of incubation and nestling period data from The Birds of North America (Poole 2005). We assumed that nests with iButtons did not need to be visited as frequently as nests without iButtons, but needed to be visited around the expected times of hatching and fledging to precisely estimate these important life-stage dates. Hatching date cannot be precisely determined from temperature data, and visits around the predicted fledging date can help in determining nest age and in estimating survival rates for incubation and brood-rearing periods. Specifically, we assumed that nests should have been visited the day before the earliest recorded hatching date, as specified in The Birds of North America, and visits should have continued at 2–3-day (regular) intervals until hatching was observed. Then the next visit should have been scheduled for the earliest fledge date reported in the literature and followed at 2–3-day-interval checks until fledging or failure was confirmed. Using these estimates of
visits on monitored nests, we were able to compare actual visits (performed every 2–3 days) with the number of visits necessary to confirm hatching and fate dates. Finally, we evaluated expenses and potential savings with the use of iButtons in lieu of hiring extra field help.

5.5. Results

We found 270 nests during the spring and summer of 2010, of which 161 were accessible to assess nest fate. In 34 of these 161 nests we deployed iButtons. We placed iButtons in nests of 16 species: American goldfinch (*Carduelis tristis*), American robin (*Turdus migratorius*), black-billed cuckoo (*Coccyzus erythropthalmus*), black-throated blue warbler (*Setophaga caerulescens*), common yellowthroat (*Geothlypis trichas*), chestnut-sided warbler (*Setophaga pensylvanica*), gray catbird (*Dumetella carolinensis*), hermit thrush (*Catharus guttatus*), indigo bunting (*Passerina cyanea*), killdeer (*Charadrius vociferus*), ovenbird (*Seiurus aurocapilla*), red-winged blackbird (*Agelaius phoeniceus*), song sparrow (*Melospiza melodia*), veery (*Catharus fuscenscens*), alder flycatcher (*Empidonax alnorum*), and willow flycatcher (*Empidonax traillii*). We retrieved data from 28 iButtons in nests and from all control loggers.

Our sample size was inadequate to test for variation in iButton performance with respect to nest attributes. However, our sample included a wide range of variation in nest characteristics that could potentially affect the thermal environment of the nest. Fifteen nests were <0.5 m above the ground, with 8 of these on the ground. The remaining 19 were between 0.5 m and 2.2 m above the ground. Seventeen nests were constructed in
woody vegetation and 16 were constructed in or on top of herbaceous vegetation. Clutch sizes ranged from 3 to 6 eggs and one species (black-billed cuckoo) exhibits both male and female incubation. Three species (American robin, willow flycatcher, and American goldfinch; n = 5, 1 and 1, respectively) added species-specific insulation material to their nests. Ovenbird nests (n = 1) are domed. Additionally, black-billed cuckoo construct particularly flimsy nests with little fine nesting material on the platform. Thus, although we could not specifically test the efficacy of iButtons with respect to nest attributes, the sample incorporated substantial variability in factors that could affect the thermal environment and, hence, iButton performance.

There was no evidence that the use of iButtons affected the probability of nest success. Sixty-eight percent of nests with iButtons (19 of 28) were successful and 51.9% of those without iButtons (69 of 133) were successful ($\chi^2_1 = 1.8, P > 0.18$); effect size was small ($\phi_c = 0.11$). Daily survival rate estimates did not differ between nests with and without temperature loggers. Daily survival rate was 0.964 for all 161 nests (95% CI = 0.954–0.971), 0.981 for nests with iButtons (95% CI = 0.964–0.991), and 0.958 for nests without iButtons (95% CI = 0.947–0.967).

iButtons reliably collected and stored data in 28 cases, allowing a more precise determination of nest fate date and time. Six of the 34 nest iButtons were lost (prior to refinement of the deployment method), presumably because of displacement by adult birds, or removal by humans or predators.

We assessed fate date estimates using the average temperature differences between nest and control iButtons calculated the day before and the day after nest
termination. These temperature differences varied consistently across all 28 nests. The average temperature difference between nest and control iButtons estimated the day before the assumed fate date was $15.5^\circ \text{C} \pm 4.73^\circ \text{C}$ (temp in $^\circ \text{C} \pm \text{SD}$). The average temperature difference estimated the day after the assumed fate date was $1.33^\circ \text{C} \pm 1.32^\circ$. Also, the estimation of fate time seemed to be precise. We estimated the average temperature difference between nest and control iButtons 5 hours before and after the time at which equilibrium temperature was reached (generally within 1 hr of the estimated fate time). The temperature difference recorded 5 hours prior to equilibrium time was $11.2^\circ \text{C} \pm 4.36^\circ$, and the average temperature difference estimated 5 hours after equilibrium time was $1.16^\circ \text{C} \pm 0.97^\circ$.

Nests with iButtons were active for 497 days. We recorded temperatures with iButtons at these nests for 284 days. Across all iButtons days, on only 31 occasions did we record a higher ambient temperature than nest temperature. Greater ambient temperature occasions varied in length (15 min to 2 hr), and always coincided with times most likely to receive direct sun exposure. When iButtons were directly exposed to the sun, they recorded inflated temperatures, with peaks reaching an unrealistically high temperature of $48^\circ \text{C}$. If nest termination were to occur during a sunny day, and the nest iButton was exposed to the sun at that time, the temperature recorded by the nest iButton could be different from the control iButton. Because sun exposure varied during the day, we recorded only one nest in which the fate time estimate could have been influenced by sun exposure. Thus, fate date estimates were not affected by sun exposure.
In 20 cases, the fate dates obtained with the iButtons were 1–3 days different than the estimated fate dates from our visits failure (Table 10a). In the remaining 8 cases, the use of iButtons confirmed the date estimated by the visits. In addition, for all fate dates obtained with iButtons, we more precisely estimated the time of fate (Fig 18).

The number of visits to each active nest would have stayed the same (13 nests, particularly those that remained active for shorter periods after discovery and iButton deployment), or diminished by 1–3 visits (21 nests), if nests were monitored primarily through iButton presence (Table 10b). Overall, 15% of the visits to nests could have been avoided, allowing 1 out of 6 technicians to use his or her time to find more nests, thereby increasing sample sizes.

We estimated that 150–200 iButtons would be required to monitor all 161 nests, given the temporal distribution of active nests (nests monitored for a period of 80 days; min. no. of days of nest activity = 7; max. no. of days of nest activity = 34; mean and median no. of days of nest activity = 20), and the possible reuse of retrieved iButtons. Purchase of temperature loggers and an interface for downloading data costs between US$2,500 and US$3,400 depending on the number of loggers acquired (cost could increase up to US$6,000 if iButtons of greater precision and memory size were required). Temperature loggers such as iButtons can be used for a minimum of 5 field seasons depending on battery life, which is influenced by temperature of operation and interval between recordings. By contrast, the cost for hiring a field technician for one field season (10–15 weeks) was estimated between US$3,200 and US$5,000. Thus, the use of
approximately 200 low-precision iButtons for nest monitoring is roughly equal to the cost of one field technician (costs estimations were performed in the spring of 2011).

To evaluate savings provided by the use of iButtons, we devised 3 5-year (min. life span of iButton batteries) scenarios: in scenario 1, we employed 6 field technicians without using iButtons (yearly estimated cost: US$19,200); in scenario 2, we employed 5 field technicians coupled with the use of 200 low-precision iButtons (yearly estimated cost for first yr: US$19,250, yearly estimated cost for following 4 yr: US$16,000); in scenario 3, we employed 5 field technicians and the use of 200 high-precision iButtons (yearly estimated cost for first yr: US$22,800, yearly estimated cost for following 4 yr: US$16,000). The cost difference between scenarios 1 and 2 was minimal for the first year, but provided a US$3,200 yearly saving for the following 4 years (total saving of US$12,750 after 5 yr). Two years would be required to offset the cost difference between scenarios 1 and 3, but the use of the iButtons would save up to US$9,200 over the following 3 years.

5.6. Discussion

The use of iButtons proved to be a non-invasive, inexpensive, and effective way to monitor nests. We retrieved data from iButtons for 15 species (the iButton deployed in the only killdeer nest monitored was removed when the nest was depredated), expanding considerably the range of species whose nests have been monitored successfully with temperature data loggers. Both passerine and non-passerine nests can be monitored with minimal disturbance using iButtons. The exception is for species in
which deployment could compromise the nest structure (e.g., mourning dove \([Zenaida \, macroura]\)), and when nests are inaccessible (i.e., too high in trees or rock ledges). The deployment methodology can be used to successfully monitor arboreal and ground nests and can prevent iButton removal by adults. To avoid logger removal, a different strategy should be developed and used for cavity nesters and for species in which nest structures are absent or simple. Even if high nests could be reached with ladders or poles, the deployment of iButtons can still be impractical especially considering the disturbance necessary in reaching the nest. We did not observe any significant impact on nest success because of iButton deployment. Data loss was mainly due to improper logger deployment during the early part of the season as we were refining our methodology.

Nest survival and nest success estimates will be influenced by a more precise estimation of fate date and time. We did not have a large enough sample of nests with data from both personnel and iButtons to statistically compare differences in daily survival rate estimates and nest success. However, the more precise fate dates obtained with iButtons should provide a more accurate estimation of daily survival rates and nest success.

We estimated fate date using 3 methods: visual assessment (using the temp charts; Figure 18), mathematically (average temp differences 1 day before and after assumed fate date), and in a few occasions with field technician visits within 1 day of fate date. We found variation in precision among the 3 methods, but similar estimates of fate date. Thus, despite the lack of an alternative monitoring method (e.g., video surveillance), we
cannot find a more parsimonious explanation for the sudden change in temperature coincident with fate date besides attributing it to nest termination.

One issue related to the use of iButtons in lieu of field personnel for nest monitoring is the potential of data loss due to the loss of iButtons. Even if the deployment method described above prevented most iButton removal by adult birds, violent predation events could result in the loss of nest iButtons. If we had relied only on iButtons, the percentage of nests for which we would have not determined nest fate would have been 8-9%. Further refinement of deployment method (i.e., extra-strong glue) could prevent predator removal of iButtons and potential data loss.

During our research, cowbird parasitism was minimal (2 nests out of 161). In areas where cowbird parasitism is high, the use of iButtons will be insufficient to determine whether host or cowbird young have fledged. More frequent visits should be planned around the fledging date to document fledging by host young.

Other authors (Hartman and Oring 2006, Schneider and McWilliams 2007) used other types of temperature loggers (thermocouples paired with data loggers) to obtain bird incubation behavior and estimate fate date. However, iButtons have several advantages over thermocouples. Thermocouples are more costly, require more frequent checks to ensure proper functioning, and lack the small size, unobtrusiveness, and ease of deployment of the iButtons. On the other hand, thermocouples have more data storage capacity and record temperature more precisely.

The difference in costs between the low range of expenses for field technicians (US$3,200), and the highest estimated cost of acquiring 200 iButtons, software, and
connectors for downloading data (estimate cost: US$6,800) could be paid off in 1 or at most 2 years by employing one fewer field technician. If monetary issues are not a main concern, pairing the use of iButtons with the same number of personnel could increase the number of nests and sample sizes. Depending on the goal of the study, the number of field technician visits avoided using iButtons could increase if hatching date estimates are not deemed necessary. Fewer visits will result in increased time for nest searching.

In addition, iButtons guaranteed the retrieval of usable data that are more precise than those collected by direct observation. However, the fate time estimated with iButtons can thus far only be considered precise under the assumption that the last time of nest use closely follows predation or fledging events. Coupling the use of iButtons and video recording to monitor nests could provide a better understanding of the precision of iButtons in determining fate time (Weidinger 2006).

Temperature loggers could also be used to study temperature of incubation and incubation behavior. Nest temperature recorded over short time intervals (if using the iButton model that has larger data storage capacity), or continuously (assuming memory capacity will be further increased), could improve our knowledge of incubation temperatures for different species. For species that are known egg acceptors (Payne 1998), the iButton circuit, battery and storage unit could be encapsulated in egg-shaped casings that could be painted with species-specific color markings to assess differences between egg temperature and nest temperature (Hoover et al. 2004). Nest attendance patterns could also be estimated precisely using temperature loggers that continuously record temperature (Baldwin and Kendeigh 1927), especially by coupling direct
observations with iButtons set to short recording intervals (Cooper and Phillips 2002, Schneider and McWilliams 2007). Because the memory size of iButtons is still limited (8,192 bytes), continuous recording of temperature is still impractical.

iButtons can be effectively used to monitor nests and obtain information on nest failure date, time, and nest attendance patterns. We recommend the use of iButtons as inexpensive means for collecting nearly continuous data on nest fate and as a relatively low-disturbance solution for monitoring species that are sensitive to nest disturbance.

5.7. Acknowledgment

Funding for this research was provided by through the U.S. Department of Agriculture/National Institute of Food and Agriculture Managed Ecosystems Program (award no. 2009-35304-05349). We thank D. Parrish for loaning us iButtons at the start of the project and K. V. Briggs for sharing his knowledge on iButtons. Thanks also to the Fish and Wildlife Coop unit at the University of Vermont and to anonymous reviewers for comments and suggestions on how to improve this manuscript.
5.8. Tables

Table 10 (a) Differences in nest fate date between iButton and observer-based data for nests monitored in the Champlain Valley of Vermont (USA) between 23 May and 7 Aug 2010. With observer-based data, nest fate date was estimated as the mid-point between successive nest visits. For iButtons, we designated as fate time and date, the time and date of the last high point in the nest temperature chart before the nest and control iButton temperatures equilibrated. ‘NA’ refers to nests for which the iButton was lost prior to the ultimate nest-fate date. (b) Estimated decrease in the number of visits using iButton data. The minimum number of required visits was calculated assuming that nests should be visited the day before the earliest recorded hatching date and visits should continue at 2–3-day (regular) intervals until hatching. The next visit would be scheduled for the earliest fledge date reported in the literature and followed at 2–3-day intervals checks until fledging or failure.

(a) Fate date difference

<table>
<thead>
<tr>
<th>Fate date difference (days)</th>
<th>No. of nests</th>
<th>No. of species</th>
<th>Success nests</th>
<th>Fail nests</th>
<th>Species</th>
</tr>
</thead>
<tbody>
<tr>
<td>0</td>
<td>8</td>
<td>6</td>
<td>5</td>
<td>3</td>
<td>Alder flycatcher, American robin, gray catbird, hermit thrush, red-winged blackbird, song sparrow</td>
</tr>
<tr>
<td>1</td>
<td>7</td>
<td>6</td>
<td>6</td>
<td>1</td>
<td>American goldfinch, American robin, chestnut-sided warbler, red-winged blackbird, song sparrow, willow flycatcher</td>
</tr>
<tr>
<td>2</td>
<td>10</td>
<td>8</td>
<td>6</td>
<td>4</td>
<td>Black-throated blue warbler, common yellowthroat, chestnut-sided warbler, indigo bunting, ovenbird, red-winged blackbird, song sparrow, veery</td>
</tr>
<tr>
<td>3</td>
<td>3</td>
<td>3</td>
<td>2</td>
<td>1</td>
<td>Black-billed cuckoo, red-winged blackbird, song sparrow</td>
</tr>
<tr>
<td>NA</td>
<td>6</td>
<td>4</td>
<td>2</td>
<td>4</td>
<td>American robin, killdeer, red-winged blackbird, song sparrow</td>
</tr>
</tbody>
</table>
(b) Decrease in no. of visits

<table>
<thead>
<tr>
<th>Decrease in no. of visits</th>
<th>No. of nests</th>
<th>No. of species</th>
<th>Success nests</th>
<th>Fail nests</th>
<th>Species</th>
</tr>
</thead>
<tbody>
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<td>0</td>
<td>13</td>
<td>8</td>
<td>8</td>
<td>5</td>
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</tr>
<tr>
<td>1</td>
<td>16</td>
<td>9</td>
<td>9</td>
<td>7</td>
<td>American goldfinch, American robin, black-throated blue warbler, chestnut-sided warbler, hermit thrush, killdeer, ovenbird, red-winged blackbird, song sparrow</td>
</tr>
<tr>
<td>2</td>
<td>4</td>
<td>3</td>
<td>3</td>
<td>1</td>
<td>Alder flycatcher, American robin, red-winged blackbird</td>
</tr>
<tr>
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<td>1</td>
<td>1</td>
<td>1</td>
<td></td>
<td>Willow flycatcher</td>
</tr>
</tbody>
</table>
Figure 18: a) Temperatures recorded in 15-minute intervals by nest (Temp_N) and control iButtons (Temp_C) in a chestnut-sided warbler nest located in Milton, VT (USA). The nest (with 3 eggs) succeeded (>2 young fledged) on 27 June 2010 (fate date) around 0545 hours (fate time). (b) The temperature recorded by the 2 iButtons reached equilibrium temperature within 45 minutes of fledging. Variations in daytime temperatures after fledging are due to different sun exposure between the nest and control sites.
### 5.10. Supporting information

Table 11 Data used to calculate differences in estimated fate date between observers and iButtons and the comparison between the numbers of actual versus minimum required visits for determining nest fate. NA refers to nests for which the iButton was lost prior to the ultimate nest fate date. Data collected in the Champlain Valley of Vermont (USA) between May and August 2010.

<table>
<thead>
<tr>
<th>Species</th>
<th>Nest no.</th>
<th>Fate date</th>
<th>iButton fate date</th>
<th>Date of visits</th>
<th>Date difference</th>
<th>No. visits actual</th>
<th>No. of visit required</th>
<th>Difference (no. visits)</th>
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<td>24 Jul</td>
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<td>7</td>
<td>6</td>
<td>1</td>
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</tr>
<tr>
<td>American robin</td>
<td>226</td>
<td>25 Jul</td>
<td>24 Jul</td>
<td>1</td>
<td>8</td>
<td>6</td>
<td>2</td>
<td></td>
</tr>
<tr>
<td>American robin</td>
<td>10</td>
<td>NA</td>
<td>3 Jun</td>
<td>4</td>
<td>4</td>
<td>0</td>
<td></td>
<td></td>
</tr>
<tr>
<td>American robin</td>
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<td>30 Jul</td>
<td>0</td>
<td>7</td>
<td>6</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>American robin</td>
<td>266</td>
<td>NA</td>
<td>11 Jul</td>
<td>3</td>
<td>2</td>
<td>1</td>
<td></td>
<td></td>
</tr>
<tr>
<td>American robin</td>
<td>231</td>
<td>7 Aug</td>
<td>7 Aug</td>
<td>0</td>
<td>6</td>
<td>4</td>
<td>2</td>
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<tr>
<td>Black-billed cuckoo</td>
<td>236</td>
<td>3 Aug</td>
<td>31 Jul</td>
<td>3</td>
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<td>3</td>
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<td>90</td>
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<td>25 Jun</td>
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<td>6</td>
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<td>1</td>
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</tr>
<tr>
<td>Common yellowthroat</td>
<td>61</td>
<td>17 Jun</td>
<td>19 Jun</td>
<td>−2</td>
<td>4</td>
<td>4</td>
<td>0</td>
<td></td>
</tr>
<tr>
<td>Chestnut-sided warbler</td>
<td>195</td>
<td>12 Jul</td>
<td>10 Jul</td>
<td>2</td>
<td>6</td>
<td>5</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>Chestnut-sided warbler</td>
<td>100</td>
<td>11 Jul</td>
<td>10 Jul</td>
<td>1</td>
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<tr>
<td>Chestnut-sided warbler</td>
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<td>27 Jun</td>
<td>25 Jun</td>
<td>2</td>
<td>7</td>
<td>6</td>
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<td></td>
</tr>
<tr>
<td>Alder flycatcher</td>
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<td>18 Jul</td>
<td>18 Jul</td>
<td>0</td>
<td>8</td>
<td>6</td>
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<td>0</td>
<td>3</td>
<td>3</td>
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<td></td>
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<tr>
<td>Hermit thrush</td>
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<td>21 Jul</td>
<td>21 Jul</td>
<td>0</td>
<td>7</td>
<td>6</td>
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<tr>
<td>Indigo bunting</td>
<td>192</td>
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<td>4 Jul</td>
<td>2</td>
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<td>4</td>
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<tr>
<td>Killdeer</td>
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<td>NA</td>
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<td>4</td>
<td>3</td>
<td>3</td>
<td>1</td>
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<tr>
<td>Ovenbird</td>
<td>175</td>
<td>2 Jul</td>
<td>4 Jul</td>
<td>−2</td>
<td>5</td>
<td>4</td>
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<tr>
<td>Red-winged blackbird</td>
<td>206</td>
<td>3 Jul</td>
<td>3 Jul</td>
<td>0</td>
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<td>4</td>
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<td>15 Jun</td>
<td>5</td>
<td>5</td>
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<td></td>
<td></td>
</tr>
<tr>
<td>Red-winged blackbird</td>
<td>46</td>
<td>4 Jun</td>
<td>7 Jun</td>
<td>−3</td>
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<td>3</td>
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<tr>
<td>Red-winged blackbird</td>
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<td>NA</td>
<td>19 Jun</td>
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<td>4</td>
<td>1</td>
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<td></td>
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<tr>
<td>Red-winged blackbird</td>
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<td>15 Jun</td>
<td>16 Jun</td>
<td>1</td>
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<td></td>
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<td>Red-winged blackbird</td>
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<td>3 Jul</td>
<td>−2</td>
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<td>7</td>
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<td>6</td>
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<tr>
<td>Song sparrow</td>
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<td>11 Jun</td>
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<td>−1</td>
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<td>2</td>
<td>1</td>
<td></td>
</tr>
<tr>
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<td>2 Aug</td>
<td>4 Aug</td>
<td>−2</td>
<td>8</td>
<td>7</td>
<td>1</td>
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<td>-----</td>
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<td>----</td>
<td>---</td>
<td>---</td>
<td>---</td>
<td></td>
</tr>
<tr>
<td>Song sparrow</td>
<td>245</td>
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<td>17 Jul</td>
<td>1</td>
<td>5</td>
<td>4</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>Song sparrow</td>
<td>214</td>
<td>2 Jul</td>
<td>2 Jul</td>
<td>0</td>
<td>3</td>
<td>2</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>Song sparrow</td>
<td>4</td>
<td>14 Jun</td>
<td>11 Jun</td>
<td>3</td>
<td>6</td>
<td>6</td>
<td>0</td>
<td></td>
</tr>
<tr>
<td>Song sparrow</td>
<td>273</td>
<td>26 Jul</td>
<td>24 Jul</td>
<td>2</td>
<td>5</td>
<td>5</td>
<td>0</td>
<td></td>
</tr>
<tr>
<td>Song sparrow</td>
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<td>NA</td>
<td>30 May</td>
<td>2</td>
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<td>2</td>
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<td></td>
</tr>
<tr>
<td>Veery</td>
<td>83</td>
<td>11 Jun</td>
<td>9 Jun</td>
<td>2</td>
<td>2</td>
<td>2</td>
<td>0</td>
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<td>28 Jul</td>
<td>28 Jul</td>
<td>0</td>
<td>10</td>
<td>6</td>
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</tr>
</tbody>
</table>

5.11. Literature cited


CHAPTER 6: MULTIFUNCTIONAL LANDSCAPES AND THEIR CARBON SEQUESTRATION POTENTIAL

6.1. Introduction

Agricultural landscapes are important for the production of goods vital to human life, but are also being recognized as important for other services (Robertson & Swinton 2005). Biodiversity conservation, habitat for wildlife, biological control of pests, nutrient cycling, water quality control, pollination, aesthetic values, recreation, and preservation of rural and cultural traditions (Arriaza et al. 2004; Fleischer & Tchetchik 2005; Swinton et al. 2006), are some examples of additional values produced by agricultural landscapes.

Areas of intensive agriculture are characterized by diminished biodiversity and impoverished ecosystem services (McLaughlin & Mineau 1995). However, the creation of heterogeneous agricultural systems, where intensively managed habitats are interspersed with more natural habitats and less intensively managed or set aside habitats, is seen as a way to preserve biodiversity and maintain healthy ecosystems (Benton et al. 2003). These systems’ health are not only measured in number of species of animals and plants that they can support, but health also encompasses air filtering, micro-climate regulation, noise reduction, water control, soil retention, recreation and aesthetic functions that enrich human quality of life.

Agricultural systems are defined as multifunctional when, in addition to the primary role of food production, one or more functions are added. In particular these added functions refer to environmental and sociocultural benefits (Rossing et al. 2007).
One way to increase benefits or ecosystem services in an agricultural system is by managing it using agroforestry (USDA 2013). By integrating trees and shrubs into agricultural systems, agroforestry can enhance the benefits (wildlife habitat, pollination, pest control, soil enrichment, water quality control, erosion control, flood mitigation, biodiversity conservation, aesthetic values, recreation, carbon sequestration, etc.) that the agricultural system provides (Kremen 2005).

Carbon sequestration can be an important component in mitigating climate change. More trees and shrubs in a landscape will contribute to carbon sequestration over uninterrupted herbaceous monocultures or pastures (Sharrow & Ismail 2004; Jose 2009). Carbon sequestration, together with other benefits provided by a multifunctional landscape, are “free of cost” services. However, carbon sequestration benefits provided by multifunctional agroforestry systems could also provide economic returns to landowner if a market for carbon offsets is developed (Jose 2009). To quantify the potential contribution of agricultural landscapes in mitigating climate change, we analyzed the carbon sequestration potential of three types of these landscapes (mostly agricultural, mostly forest, or mixed agriculture and forest) in the Champlain valley of Vermont.

6.2. Methods

We collected information on woody vegetation in eighteen circular study plots (area 201 ha) selected along the main rivers of the Champlain Valley of Vermont (USA). Within each plot we sampled woody plants communities along four transects. Transect
starting points were 250 m in length in the four cardinal directions from the center of the plot and coincided with the bird points locations described in previous chapters. The orientation of the transects coincided with the direction that offered the maximum concentration of woody plants. We oriented transects in the north direction when the starting point was located in a forest. If the transect starting point was in an area without trees, we moved the point to the closest treed habitat. We did not include transects that were moved more than 100 m from their original starting point in the analysis. Within 1.5 m of each 50 m transect we recorded the presence of woody plants (trees) by species, plant height, and DBH (diameter at breast height).

We estimated the CO₂ tons equivalent stored in the trees on the basis of the DBH measured. First, we calculated the dry-weight biomass for individual trees using the following biomass equation:

\[
\text{biomass} = \exp(\beta_0 + \beta_1 \ln \text{DBH})
\]

where \(\beta_0\) and \(\beta_1\) depends on the group to which the trees’ species belong to (Jenkins et al. 2003).

Second, we converted the biomass to carbon equivalents by multiplying the biomass by 0.5, and third, we converted carbon equivalents in carbon dioxide equivalent by multiplying carbon by 3.67 (Ryan et al. 2010). We also calculated tree species diversity using the inverse Simpson index (Simpson 1949).

We divided the 18 study plots into 3 treatments: agricultural (agricultural land in a forest matrix), mix (~50:50 mix of forest and agricultural land), and forest (forest predominantly-agricultural landscapes). We tested for differences between treatments
with one way analysis of variance. For significantly different categories we performed post hoc comparison using Tukey HSD test. Statistical analyses were performed using JMP Pro (2013).

6.3. Results

We found no significant differences among treatments for DBH, height, number of trees, tree diversity, or carbon sequestered. However, a one way analysis of variance revealed a significant difference among treatments for the mean number of tree species found ($F_{2,15}=5.8$, $p=0.0135$). Post hoc comparisons indicated that the mean number of species for both forest ($M=10.33$, $SD=2.50$) and mixed ($M=9.83$, $SD=0.75$) plots was significantly greater than the mean number of species in agricultural plots ($M=6.33$, $SD=2.80$).

Tree communities in both forest and mixed plots most closely matched the hemlock-northern hardwood forest community (Thompson et al. 2000). In forest plots the most common species were (listed in order of abundance): eastern hemlock ($Tsuga canadensis$), red maple ($Acer rubrum$), yellow birch ($Betula alleghaniensis$), sugar maple ($Acer saccharum$), eastern white pine ($Pinus strobus$), and paper birch ($Betula papyrifera$). In mixed plots, the most common species found were: eastern hemlock, sugar maple, Northern white cedar ($Thuja occidentalis$), eastern white pine, and red maple.
The species composition of transects in agricultural plots more closely matched floodplain forest and the rarer valley clay-plain forest communities (Thompson et al. 2000). In these plots, the most dominant species were: green ash (*Fraxinus pennsylvanica*), silver maple (*Acer saccharinum*), American elm (*Ulmus Americana*), boxelder (*Acer negundo*), quacking aspen (*Populus tremuloides*), Eastern red cedar (*Juniperus virginiana*), and sugar maple.

Because we specifically selected plots to represent different forest/agriculture ratio cover, the surface covered by treed habitats varied significantly between treatments ($F_{2,15} = 85.5$, $P < 0.001$). Post hoc comparisons using the Tukey HSD test indicated that the mean area covered by trees per plot differed significantly between all treatments. The mean treed habitat in forest plots was 165 ha (SD=15.7), 108 ha in mixed plot (SD=15.7), and 47 ha in agricultural plots (SD=15.5).

Because CO$_2$ sequestration is measured as mass/unit area (here, 41.5 tons/ha), carbon storage also varies across treatments based on the variation in the proportion of treed habitat. Carbon storage and sequestration potential is greatest in forest plots (6,844 tons/plot), intermediate in mixed (4,506 tons/plot) plots, and lowest in agricultural plots (1,948 tons/plot).

A one way analysis of variance revealed a significant difference ($F_{2,15} = 17.65$, $P < 0.0001$) among treatments for the area of the plot covered by small treed elements (the sum of the area of the following land use land cover patches: forest patches greater than 1000 m$^2$ without core area, linear treed patches less than 50 m wide and at least 4 times longer than wide, and treed patches less than 1000 m$^2$). Post hoc comparisons indicated
that the mean area covered by small treed elements varied among all treatments with 12.6 ha (SD=2.63) in agricultural plots, 8.96 ha (SD=3.21) in mixed plots, and 3.10 ha (SD=2.54) in forest plots. In agricultural plots the small treed elements contributed for 30% of the area covered by trees. In mixed plots the area of small treed elements contributed 8.1 % to the tree cover of the plots and in forest plots the contribution from small treed elements made up only 1.8% of the total tree cover in the plots.

6.4. Discussion

The average carbon sequestration mass per unit area was consistent across treatments (41.5 tons/ha), thus study plots with greater percentage of forest or treed cover had a greater carbon storage and carbon sequestration potential. However, due to the limited space available for reforestation or afforestation, forest plots have a limited potential in increasing the area covered by trees and in sequestering additional CO₂ from the atmosphere (Ryan et al. 2010). Carbon offset schemes require that the project proposed to counterbalance the carbon emissions should provide an increase in carbon reduction capacity from the status quo (Goodward & Kelly 2010), making forest plots poor locations in which to increment even further carbon sequestration.

Frequently, carbon farming projects aim to enhance carbon sequestration of agricultural landscapes by promoting monoculture tree plantings. Despite being effective in offsetting carbon emissions, monocultures could have negative ecological impacts and could reduce landscape multifunctionality (Lindenmayer et al. 2012). Selecting native
and diverse type of planting for revegetation projects, instead of using quick growing
trees that have the sole purpose of offsetting carbon emissions, not only would further
contribute to carbon sequestration, but could enhance the local economy by providing
fruits, nuts, lumber, firewood, maple syrup, and many other byproducts (Scherr & Sthapit
2009).

Agroforestry has the potential to increase carbon sequestration and the
functionality of agricultural landscapes. Increasing the area that supports treed habitats
would contribute to the carbon sequestration potential of the landscape and increase
biodiversity (Scherr & Sthapit 2009; Lin et al. 2013). The types of landscapes that would
gain more from an agroforestry approach are agricultural and mixed landscapes where
tree cover can be increased without considerably encroaching on agricultural areas.
Using native species to increase the area covered by trees in intensively agricultural
landscapes, particularly if revegetation will take place on marginal, degraded or
underused areas, will increase the number of species and diversity of woody vegetation
present. Simply increasing the treed area cover in the plot would increase the carbon
sequestration and storage potential of the area. The addition of hedgerows and small
wooded lots in agricultural landscapes would also provide additional ecosystem services.
Woodlots contribute between 3 and 8% to landowner income by providing timber for
harvest (Egan 2007). A conversion from arable land to pastures or tree cover would
reduce tenfold the emission of N2O which has a far greater global warming potential than
CO2 (Falloon et al. 2004). Adding forested areas around rivers and streams will help in
retaining water thus mitigating surges connected with intense meteorological events
exacerbated by climate change. Adding hedgerows in simple landscapes will also enhance the fauna diversity. For example, bird richness and abundance are increased by hedgerows addition to the landscape as a result of greater food, nesting and sheltering areas availability (Batáry et al. 2010).

Growing more trees and increasing the cover of perennial vegetation in agricultural landscapes has the potential to increase the amount of carbon sequestered and stored in the vegetation and soil. However, when expanding tree cover in agricultural landscapes, consideration must be taken to prevent the disruption of food production and the other vital ecosystem services provided.

6.5. Literature cited


CHAPTER 7: ARE AGRICULTURAL LANDSCAPES AND VEHICLES SIMILAR?

When asked what I have been working on for the past several years, my short answer has been “I identified which agricultural landscapes can support a viable and vibrant rural economy while providing good quality habitat for wildlife.” If there seemed to be interest in the topic I would then introduce concepts like ecosystem services and multifunctionality. I used the word landscape several times in my discourse, but I originally did not think to define the term while using it. However, during a conversation with a friend, I was surprised to learn that for most of our talk she had imagined a landscape as somebody’s backyard and thought that my work was to increase the backyard quality for birds. Many backyards were actually included in the landscapes in which I worked, but there is more to it. Since this conversation took place, I started defining the word landscape as a wide spatially heterogeneous area composed by a mosaic of ecosystems, habitat patches, or elements.

While talking about my work I often compared a landscape to a vehicle to explain landscape multifunctionality. I asked my conversation partner to choose a vehicle and I associated the vehicle with different types of agricultural landscapes. I compared a semi-truck to an intensive agricultural landscape, a minivan to a mixed landscape, and a pickup truck to a highly forested agricultural landscape.

All of these vehicles help people with daily transportation needs and are designed to simplify our lives. Some vehicles are very good at only one task or function: the semi-
truck is very useful while moving, allowing us to store and transport all of our belongings in just one trip, but it is not the vehicle of choice when you want to go with your friends to a party downtown. Similarly an agricultural landscape is very good at producing goods, but we rarely associate many other benefits to this landscape. The minivan and pick-up truck are good for multiple functions. The minivan can transport simultaneously several items and people and its inner space is flexible and can be redesigned by changing the seats arrangement. Likewise a mixed rural landscape is able to maintain several functions at the same time. Agricultural goods production is still fairly high, but the many small wooded landscape elements provide countless other benefits and contribute to picturesque scenery. The pick-up truck might not be the most gas efficient vehicle to commute to work, but it is surely helpful in bringing home the new washing machine. Similarly a forested dominated agricultural landscape can still produce some goods, but also maintains high biodiversity due to the presence of large natural landscape elements. Different agricultural landscapes can provide benefits to people and be more or less multifunctional.

Agricultural landscapes produce goods and food. The more intensively cultivated landscapes provide humans with enormous amounts of grains, veggies, meat, and milk. The landscapes that maintain wooded and natural areas produce lumber, fuel, wood, wild edibles, medicinal herbs, mushrooms, diversified flowering plants that provide support to honey production, maple syrup, and material for crafts. Depending on their complexity, agricultural landscapes can be more or less helpful in: regulating the local climate and air quality, sequestering and storing carbon, producing soil, protecting from
soil erosion, reducing noise, retaining soil, filtering water, improving water quality, preventing nutrient runoff, controlling erosion of river banks, buffering wind, providing habitat for wildlife, regulating pests and vector borne disease, pollination, maintaining genetic diversity, and much more. In addition to the production and ecological functions listed above, agricultural landscapes provide innumerable sociocultural benefits. Agricultural landscapes offer many recreational opportunities such as hunting, fishing, wildlife viewing, hiking, playing, and biking. Tourism can enhance local economies. In many agricultural landscapes rural and cultural traditions are maintained because of the long lasting ties of the population to certain natural elements. Scenic agricultural landscapes inspire artists and are sought for meditation and relaxation. As impressive as this list of benefits is, it is far from complete. In any case, the many benefits provided by agricultural landscapes can immensely enrich human quality of life.

Landscapes, as any of our fictional vehicles, can break down. All of the machines we talked about are made of many parts: engine, steering wheel, tires, and seats are just some of the components of these machines. Similarly, landscapes are made of many parts: ecosystems, patches, small elements, rocks, soil, and living organisms are some of the landscape components. If you remove a small part from a vehicle, for example the car radio, your trip will not be as fun, but you will still be able to visit your friends. However, if you remove a vital component from the engine, for example the pistons, even if you do not know where they are and what they do, you will not be able to go anywhere until you replace the pistons, which will cost you a lot.
The same is true for landscapes. If you cut a tree from a forest, most people will not even realize that it was cut. If a bird was nesting in the tree, the removal of it will certainly effect the bird, but overall, the services provided by the forest will be preserved. Even when you clear a small portion of a forest to create space for your house, the landscape will most likely continue to provide the same kind of services (assuming the landscape is wide enough). If, however, you remove the entire forest cover from the landscape, than your landscape will certainly work differently. If your only use for a car is to sit in it in the same place listening to the radio, with a good battery and even without a working engine, you would be able to continue doing so for some time. Similarly, you would be able to find a place for your house in a deforested landscape. However, without forest cover the hill sides might be unstable and be prone to periodical mud slides, and your spirit would not be lifted by the beautiful song of the wood thrush that disappeared with the forest.

A vehicle can be easily fixed, its parts are fairly readily available, and a replacement car can be bought if the original one cannot be repaired. The same is not true for landscapes. The most simple replacement parts for a landscape might be available (trees can be regrown), but restoring a component of a landscape is not easy. It takes a long time, is very expensive, and the result might not be the same as the original. A replacement landscape might be found and visited somewhere else on the globe, but once a landscape is irrevocably damaged, it might never be replaced or it might take more than a lifetime to “grow” back to its original state.
Because rural areas in many regions are becoming the only available “natural” habitat for wildlife and are the most accessible “natural” sceneries for the majority of the world’s population, we should preserve and maintain agricultural landscapes not only for the goods they produce, but also for the ecological and sociocultural functions they provide. However, there is a tradeoff between the production and ecological function in agricultural landscapes. When a rural landscape is managed for intensive agriculture production we lose ecological benefits; to increase the ecological benefits we might have to give up some agricultural land. To have a highly functional agricultural landscape we should strive to maintain a diversified land use/land cover patchwork with landscape elements heterogeneously distributed. Particularly if the intense agricultural areas in a landscape are capped at 45% the landscape will continue in providing many of the ecological benefits that would be lost otherwise. Agricultural fields should also be interspersed and separated by small wooded elements that are very important in providing a safe place to nest for many species of birds and in increasing plant diversity. Of the landowners I talked to and interviewed, many expressed a deep connection to their land and loved most about their landscape the fact that it was a mix of agricultural fields and forests. Most intensive agricultural landscapes differ from this image, but their look and functionality could be improved without giving up much of their production capability. Hedgerows and small clump of trees are visually pleasing, greatly increase the services provided by the landscape, and can be easily grown in the least productive portion of the land. Adding hedgerows or allowing “nature” to take over road sides, river and streams corridors, along existing field borders and fences, and as connectors between
forests tracts, greatly contribute to the functionality of the landscape. Even better if the hedgerows are planned so that they are wider than 2 m, their structure is varied with tall trees, and shrub and grass vegetation is represented throughout. Hedgerow management should also be carefully planned to keep time and costs low for the owner while still maintaining high diversity of vegetation and organisms represented in it. To avoid impacting bird reproduction, for example, the small wooded areas should be managed in a rotation to always maintain some elements in their natural status. Also hedgerow maintenance should be timed to avoid flowering and wildlife breeding seasons, thus avoiding late spring/summer times for tree cutting and pruning. At the fine scale, judicious management that strives to maintain and improve natural field boundaries and small wooded elements, and thoughtful land cover design will promote multifunctionality. Forward thinking planning at the landscape scale that goes beyond property lines and involves neighboring landowners, local communities, and government has a greater success in maintaining high levels of landscape functionality.

Agricultural landscapes can be compared to vehicles. However, landscapes are complex vehicles. Even for the most modern and complex car you can find mechanics that intimately know all the components and functions of the vehicle and its parts. Landscapes are less well understood. There might be the need for several professionals to understand the more complex functioning of a landscape. Nevertheless, even calling upon all the landscape experts in the world, we still would not have a complete and full understanding of all the components and functions of the landscape. Both vehicles and landscapes are multifunctional. The most modern and luxurious car provides several
benefits: transportation, comfort, and potential change in the perception of the social status of the driver. Any landscape, even the ones that are highly impacted by human activities, provide a far longer list of benefits, many of which we are unaware off. These benefits impact many more people that a vehicle could. All the landowners inhabiting the landscape directly benefit from the services provided by it. At the same time everyone else who eats or uses the goods produced by the landscape, visits the landscape, sees pictures of the landscape, or lives downriver or upriver from the landscape benefits from the services provided by it. Landscapes “work” for the communities living in and surrounding it. Communities depend upon their landscapes, and should be good stewards by maintaining the highest landscape multifunctionality for themselves and future generations.
CHAPTER 8: COMPREHENSIVE LITERATURE CITED


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