

The effects of conventional and organic cropping systems on food and reproductive success of Song Sparrows in eastern Ontario farmland.

by

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Abstract

Farmland birds have undergone population declines and range contractions in both Europe and North America, as a consequence of agricultural intensification. Understanding how conventional agriculture affects birds within farmland will help us move towards the goal of maintaining both agricultural productivity and farmland bird populations. I investigate the effects of food supply and foraging habitat availability in eastern Ontario on Song Sparrows (*Melospiza melodia*), a common farmland bird. I found that i) organic soybean cropping systems support higher nestling food abundance than conventional soybean cropping systems, ii) of the invertebrates I tested, hoppers (Auchenorrhyncha) and caterpillars (Lepidoptera) are the most important in the diet of nestling Song Sparrows, iii) the portion of nestling diet that came from semi-natural habitats, especially hedgerows, was higher than expected, based on habitat availability, iv) food availability does not have a measureable effect on mass of Song Sparrow nestlings, or on the number of fledglings produced and vi) that the amount of semi-natural habitat close to the nest does not have a measurable effect on Song Sparrow nest survival. In addition, I have demonstrated a stable isotope approach for studying the diet and foraging habitat of farmland birds, when there is a contrast in carbon and nitrogen stable isotopes among food sources. My findings demonstrate the importance of hedgerows in providing foraging habitat for Song Sparrows nesting in farmland. In addition, my findings suggest that while conventional cropping systems do reduce food supply for birds in farmland compared to organic cropping systems, conventional

cropping systems in eastern Ontario are relatively benign for Song Sparrows,
and potentially other birds that nest and forage in hedgerows.

Preface

Co-authorship Statement

This thesis is formatted using the integrated thesis format, and each data chapter is written as an independent manuscript. Chapters 3 and 4 have been published, and Chapters 2 and 5 have been submitted. The two published manuscripts are co-authored with A. Baril, P. Mineau and L. Fahrig. Chapters 2 and 5 are co-authored with P. Mineau and L. Fahrig. Since each chapter is written as an independent manuscript, there is some repetition in introductions and discussions, however I have cross-referenced between chapters to reduce repetition as much as possible.

This thesis results in large part my own work. I proposed and developed the research questions, in cooperation with Pierre Mineau, Alain Baril and Lenore Fahrig. I worked with P. Mineau and A. Baril to design the stable isotope work and the field work for Chapters 3, 4 and 5. I designed the field work for Chapter 2 myself, with input from P. Mineau. I conducted or supervised all the field work myself, with the help of field assistants. I conducted or supervised the invertebrate identification together with the help of lab assistants. Preparation of materials for stable isotope analysis was carried out by Barry Joyce. Stable isotope analysis was carried out at the G. G. Stable Isotope Laboratory at University of Ottawa. I carried out all statistical analysis myself, and wrote the first draft of all materials included in the thesis.

I have permission from the publishers and my co-authors to include the two published manuscripts in this thesis. A specific chapter that has been published elsewhere must be cited using the journal citation information provided below. However, to reference my thesis as a whole or an unpublished chapter, I recommend using the following citation:

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Chapter One: General introduction

Agriculture covers more of the earth's surface than any other landuse (Ramankutty et al. 2008, Foley et al. 2011), with about 10% of the terrestrial land surface used for crops and a further 19% used for pasture in 2000 (Ramankutty et al. 2008). Over the last five decades, intensive management techniques including inputs of synthetic fertilizers and pesticides, high-yielding crop varieties and increasing mechanization and irrigation have vastly increased crop yield, leading to an increase in global crop yields of about 20% between 1985 and 2005 (Foley et al. 2011).

Due to the large area of land it occupies and the high levels of inputs and physical manipulation required to maintain crop yields, intensive agriculture has a strong impact on biodiversity within and adjacent to agricultural areas (Mineau and McLaughlin 1996, Freemark and Kirk 2001, Davidson et al. 2002). Impacts can occur directly (e.g. habitat loss, Cornulier et al. (2011), direct mortality from pesticides, Mineau et al. (2005)), or indirectly through changes in habitat quality (e.g. indirect effects of pesticides, Boatman et al. (2004)). Negative effects of agricultural intensification on the species richness and abundance of plants, invertebrates and birds in farmland have been demonstrated most clearly through long-term or large scale correlative studies (Fuller et al. 1995, Wilson et al. 1999, Fuller 2000, Donald et al. 2001, Siriwardena et al. 2001, Benton et al. 2002, Chamberlain 2002, reviewed by Robinson and Sutherland 2002, Geiger et al. 2010, Flohre et al. 2011, but see Wretenberg et al. 2006), and through direct comparisons between organic and conventional farming methods (Freemark and

Kirk 2001, Bengtsson et al. 2005, Fuller et al. 2005, reviewed by Hole et al. 2005, Roschewitz et al. 2005, Schmidt et al. 2005, Rundolf and Smith 2006, Feber et al. 2007, Diekötter et al. 2010, Ekroos et al. 2010, Gabriel et al. 2010, Geiger et al. 2010, Smith et al. 2010, Fischer et al. 2011, Jonason et al. 2011, Kirk et al. 2011, Winqvist et al. 2011, Batáry et al. 2012, reviewed by Winqvist et al. 2012). The negative effects of intensive agriculture on biodiversity are not simply due to practices such as pesticide and fertilizer use managed by individual farmers, but also to the simplification of modern farming landscapes through loss of semi-natural habitats and homogenization of crop types (Benton et al. 2003, Winqvist et al. 2012). However, despite the widespread negative impacts of intensive agriculture on biodiversity, farmland can support species and communities of conservation importance (Freemark et al. 2002, Andreasen and Stryhn 2008, Norment et al. 2010).

Birds have been a strong focus of ecological studies in farmland.

Farmland birds have shown population declines and range contractions over the last few decades in both North America and Europe (Knopf 1994, Fuller et al. 1995, Siriwardena et al. 1998, Peterjohn and Sauer 1999, Murphy 2003, Sauer et al. 2011). Population declines have been linked to aspects of agricultural intensification including increased inputs of pesticides and fertilizers, increased field size, loss of semi-natural habitats within farmland, the shift to row and field-crop monocultures, and denser, more uniform crop structure (Chamberlin et al. 2000, Fuller 2000, Murphy 2003, Newton 2004).

Research in North America has focused on grassland birds, which have shown more severe population declines than any other habitat guild over the last four decades (Knopf 1994, Peterjohn and Sauer 1999, Murphy 2003). The original declines were driven by habitat loss as farmland replaced natural grassland. As natural grassland was lost, semi-natural grasslands (i.e. hay, pasture, rangeland) became important habitat for many species, but continuing loss and fragmentation of semi-natural grasslands, as well as more intensive management of remaining areas have driven further declines (Herkert 1994, Vickery et al. 1999, Murphy 2003, McCracken 2005). Population declines have occurred in both short- and long-distance migrants (Murphy 2003, McCracken 2005), so changes on the breeding grounds are thought to have driven population declines. For the majority of species, population declines are related to changes in landscape, with loss of rangeland most commonly related to population declines (Murphy 2003).

Comparatively little is known about farmland birds in other agricultural habitats in North America, such as crop fields and field margins. Indeed a candidate list of North American farmland birds has only recently been developed (Kreuzberg 2011). Nevertheless, a large community of birds uses these habitats for foraging and breeding across the continent (Best et al. 1990, Bryan and Best 1991, Boutin et al. 1999, Best 2001, Freemark and Kirk 2001, Jobin et al. 2001, Beecher et al. 2002, Cederbaum et al. 2004, Jones et al. 2004). Kreuzberg (2011) defined a guild of farmland birds for Ontario, and demonstrated that members of this guild are affected by farmland landscape structure and

management practises. Ontario breeding bird atlas results show the majority of these species (30 out of 45) showed population declines between the first atlas in 1981-1985 and the second atlas in 2001-2005 (Kreuzberg 2011).

Because agricultural intensification is a multi-faceted process, isolating mechanisms behind population declines is not straightforward, and multiple mechanisms may interact to cause the decline of a single species (Fuller 2000, Newton 2004). For example, population declines of Yellowhammers (*Emberiza citrinella*) in the UK are likely to have resulted from a combination of hedgerow loss which has reduced the amount of available breeding habitat, reduced food supply in the breeding season leading to reduced productivity, and reduced food supply during the winter leading to reduced survival (Fuller 2000, Boatman et al. 2004, Cornulier et al. 2011). Nevertheless, three factors have emerged as leading mechanisms in farmland bird declines; the direct effects of pesticides, habitat loss, and reductions in food supply (Fuller 2000, Newton 2004).

Direct mortality from pesticides can take a substantial toll on bird populations. For example, carbofuran is estimated to have killed between 17 and 91 million songbirds annually in the US cornbelt in the 1980s (Mineau 2005). Regional population declines of some species of farmland birds in the Canadian prairies have been linked with indices of pesticide use (Mineau et al. 2005). However, the use of pesticides that are toxic to birds has steadily declined, as older chemicals are replaced by less-toxic compounds (Mineau and Whiteside 2006), and the indirect effects of pesticides on food supply are now considered more important than direct mortality (Fuller 2000, Newton 2004).

Habitat loss within farmland occurs when changing management techniques eliminate a specific land use (e.g. a switch to winter sown crops eliminates stubble fields used for foraging by some bird species in the winter), or when land that was not previously under production is converted to crops (e.g. hedgerow removal to expand field size). Widespread population declines in grassland birds in North America were originally triggered by conversion of native prairies to agricultural land uses (Peterjohn and Sauer 1999), but on-going declines are associated with loss of rangeland within agriculture (Murphy 2003). Loss of hedgerow habitat has reduced foraging and breeding habitat for farmland birds in the UK (Hinsley and Bellamy 2000). However, hedgerow losses peaked in the mid-1960s, whereas farmland bird population declines began in the mid-1970s, so current farmland bird declines are thought to be more closely linked to reductions in habitat quality such as reductions in food supply than hedgerow loss (Fuller et al. 1995, Gillings and Fuller 1998). In Canada, hedgerows and woody riparian strips provide important breeding habitat for farmland birds as well as supporting populations of plants, mammals, amphibians and reptiles (Jobin et al. 2001, Maisonneuve and Rioux 2001, Jobin et al. 2004), but loss of hedgerow habitat is difficult to study due to lack of historical data (Boutin et al. 2001).

Reductions in food supply can result from the effects of agricultural intensification, particularly the effects of pesticides, on the plants and invertebrates that birds rely on for food. By design, insecticides and herbicides reduce the abundance of invertebrates and plants respectively. Herbicides can

have an additional negative effect on invertebrate abundance by removing the plants on which they rely for shelter and food (reviewed by Potts 1986). Campbell and Cooke (1997) suggested that most invertebrate groups in farmland in the UK have either declined or remained stable, while plant species abundance and species richness has also generally declined, except for a few pest species. Benton et al. (2002), linked long-term declines in the abundance of aerial invertebrates to farmland bird declines in the UK. Long-term data on farmland invertebrate and plant populations or abundance do not seem to exist in North America.

The clearest example of a farmland bird population decline as a result of a reduction in food supply is that of the Gray Partridge (*Perdix perdix*) in the UK. Pesticide use reduces the abundance of small invertebrates required for growth and survival of chicks (Rands 1985, Potts 1986, Southwood and Cross 2002). Since population size in Gray Partridge is strongly dependent on chick mortality, pesticide use has been directly linked to population declines (Blank et al. 1967, Southwood and Cross 1969, Potts 1986). Such a clear example of the effect of reduced food supply on population declines is rare, but the effects of reduced food supply on reproductive success of several farmland birds have been demonstrated. For example, in Yellowhammers and Corn Buntings (*Miliaria calandra*), nestling growth and condition is related to the abundance of small invertebrates in habitats close to the nest (Brickle et al. 2000, Morris et al. 2005, Hart et al. 2006). Insecticide use in crop fields adjacent to nests reduces the abundance of these invertebrates, resulting in poorer nestling condition, higher

rates of partial brood loss (Yellowhammers, Morris et al. 2005, Hart et al. 2006) and reduced nest success (Corn Buntings, Brickle et al. 2000). Overwinter food supply has also been found to be critical for seed-eating birds in the UK.

Reductions in weed abundance and diversity, and loss of winter stubble fields which harbour high density and diversity of weeds reduce over-winter survival, which has been linked to population declines in a number of species of farmland birds in the UK.

Although abundance of invertebrate food has been suggested as a mechanism for increased abundance and species richness of birds on organic farmland in North America (Freemark and Kirk 2001, Beecher et al. 2002), and for reductions in the reproductive success of Tree Swallows (*Tachycineta bicolor*) in landscapes with high proportions of intensive agriculture (Ghilain and Bélisle 2008), very little research has been conducted on diet, foraging habitat or food supply of farmland birds in North America. Farmland breeding birds are known to use crop fields, hay fields and boundary features, such as hedgerows, for foraging (Best et al. 1990, Boutin et al. 1999; Puckett et al. 2009). Boutin et al. (1999) surveyed birds in corn, soybean, apple orchards and vineyards in southern Ontario and found that of 14 species regularly observed within field centers and in edges adjacent to crop fields, most species were observed in edges more than expected based on habitat availability. The only detailed North American study of foraging habitat use within crop fields that I have found was carried out by Rodenhouse and Best (1994). They showed that Vesper Sparrows (*Pooecetes gramineus*) nesting in corn and soybean fields forage most often

within 1 m of weedy or unplanted areas, and prefer fields with crop residue over bare fields, presumably due to increased food abundance. However, the detailed autecological studies that have demonstrated the importance of food supply to farmland birds in Europe have not been replicated in North America.

It is not known whether reduction in food supply has played a role in population declines of farmland birds in North America. Although many aspects of agricultural intensification have followed the same trends in Europe and North America (e.g. loss of semi-natural habitats, increasing field sizes, reductions in crop diversity), there are major differences in crop types and insecticide regimens between the two continents. Just as mechanisms for farmland bird declines differ between regions within Europe (Wrettenberg et al. 2006), so the role of food supply in driving bird declines may differ between Canada and the UK.

Current guidelines for promoting biodiversity, including birds, within Canadian farmland are centered around preserving or increasing native habitat in farmland (e.g. NAESI, Agriculture and Agri-Food Canada 2008). However, this requires farmland to be taken out of production and may even be detrimental to birds that are closely associated with crop fields, such as Horned Larks (*Eremophila alpestris*) and Vesper Sparrows. Understanding how conventional agriculture affects birds within farmland may allow us to adjust management guidelines to maintain birds and other wildlife within farmland, without taking land out of production or reducing crop yields.

The objective of this thesis was to examine how food supply and foraging habitat availability affect farmland birds during the breeding season in eastern Ontario. To do this, I will:

- 1) Investigate the effects of conventional agriculture on invertebrate food abundance for birds breeding in farmland in eastern Ontario;
- 2) Investigate the diet and foraging habitats of the Song Sparrow (*Melospiza melodia melodia*), a common farmland bird in North America and
- 3) Investigate links between food abundance, foraging habitat availability and productivity of Song Sparrows nesting in farmland.

Study species

Song Sparrows were selected as the study species, because they are widespread and common in farmland across North America (Best et al. 1995, Jobin et al. 1996, Boutin et al. 1999, Best 2001, Freemark and Kirk 2001, Jobin et al. 2001, Henningsen and Best 2005). Song Sparrows are usually considered habitat generalists, using a wide range of forest, shrubby and riparian habitats (Arcese et al. 2002). They typically prefer moist or wet habitats to dry areas. In classifying habitat associations of farmland birds, Kreuzberg (2011) categorized Song Sparrows as 'farmland generalists', meaning they use a variety of habitats within farmland, such as row crop, pasture and farmstead. Song Sparrows were found to be the most abundant bird in field margins (hedgerows, windbreaks and herbaceous field margins) in southern Quebec, Canada (Jobin et al. 2001). Their

use of field edges for foraging, together with their relatively high abundance in these areas, led Boutin et al. (1999) to rate Song Sparrows as high risk for pesticide exposure. However, Song Sparrows nesting in field margins adjacent to corn fields in southern Ontario did not show effects of granular insecticide use on either adult survival or nest survival, although there was a trend for lower clutch size, and fewer fledglings from nests adjacent to treated fields (Knapton and Mineau 1995). Song Sparrow populations in the study region (lower Great Lakes/St. Lawrence plain) have shown a long-term pattern of decline (1966-2009: -0.6 % change/year, 95% CI: -0.9% to -0.4%, n=167, Sauer et al. 2011).

Song Sparrows forage primarily on the ground, feeding primarily on invertebrates during the breeding season, together with seeds and fruit, and a mixture of seeds, fruits and invertebrates during the rest of the year (reviewed by Arcese et al. 2002). They nest in shrubby to open habitats such as hedgerows and other field boundaries, hayfields and fallow or abandoned fields. Nests are typically well concealed by surrounding vegetation and are most often built on or close to the ground, although they can be built as high as 4 m in shrubs or bushes (reviewed by Arcese et al. 2002). Song Sparrow nests are frequently parasitized by Brown-headed Cowbirds (*Molothrus ater*), which can lead to nest abandonment (Smith et al. 2003), and reductions in the number of Song Sparrows fledged per nest (Smith and Arcese 1994).

Song Sparrows have been used extensively to study songbird ecology, beginning with the landmark studies of Nice (1937, 1943) on Song Sparrow life history and behaviour. Most recently, long-term manipulative studies on food

supplemented Song Sparrows have shown the importance of food supply to reproductive output. For example, food supplemented Song Sparrows lay more eggs over the course of a breeding season (Zanette et al. 2006A), and produce heavier recruits (i.e. birds that defended breeding territories the year after hatching, Zanette et al. 2009) than Song Sparrows without additional food. However, the mean biomass of eggs produced by food supplemented Song Sparrows is less than that of un-fed Song Sparrows (Zanette et al. 2009). In addition, male Song Sparrows with food-supplemented parents have smaller song repertoires than their counterparts whose parents did not receive extra food, suggesting a trade-off between the number of young produced and their quality (Zanette et al. 2009). Zanette et al. (2006B) found that food supply and predation pressure interact in their effects on both partial nest losses and total nest survival, such that nests with supplemental food and low predation pressure produced significantly more fledglings than those with supplemental food but high predation pressure, or no supplemental food but low predation pressure. Annual reproductive success in Song Sparrows is a function of the total number of eggs laid over the breeding season, partial losses of eggs or chicks from nests and nest survival (Zanette et al. 2006B). Since total number of eggs laid is food restricted (Zanette et al. 2006A), annual reproductive success in Song Sparrows is affected by both food restriction and the combined effects of food restriction and predation pressure on partial nest losses and nest survival (Zanette et al. 2006B).

Study area

This study was carried out in eastern Ontario, Canada, which is part of the Great Lakes/St Lawrence Lowlands Bird Conservation Region (BCR13, Sauer et al. 2003). Within Canada, cover of agriculture is relatively low (approximately 7%, Ramknutty et al. 2008), but is geographically concentrated into relatively small areas where farmland dominates the landscape. Approximately 25% of eastern Ontario is farmed; the three dominant crops in the region are hay (approx. 275,000 ha, 37% of farmed land), corn (*Zea mays*, approx. 111,000 ha, 15%) and soybean (*Glycine max*, approx. 84,000 ha, 12%, OMAFRA 2011). The rest of the region is dominated by forest, wetland and urban areas. Due to the large areas of agricultural grassland (hay, pasture and fallow fields) throughout BCR13, this region supports nationally and globally important grassland bird populations (e.g. 20% of global breeding population of Bobolinks (*Dolichonyx oryzivorus*), significant portion of Upland Sandpiper (*Bartramia longicauda*) breeding population, Hartley 2007).

Previous studies on cropland and farmland bird communities in BCR13 have focused on community composition (Boutin et al. 1999, Kirk et al. 2001), habitat use within and adjacent to crop fields (Boutin et al. 1999), and the importance of organic farming. In addition, in southern Quebec, the reproductive success of Tree Swallows has been shown to decline as the proportion of intensive agriculture increases in the surrounding landscape (Ghilain and Bélisle 2008). The importance of hedgerows and shelterbelts for supporting diversity and abundance of birds in farmland has been emphasized by Jobin et al. (2001),

Freemark and Kirk (2001) and Kirk et al. (2011). Finally, organic agriculture in this region has been shown to support higher richness and abundance of birds during the breeding season (Freemark and Kirk 2001, Kirk et al. 2011).

Hedgerows play an important role throughout this thesis, by providing breeding habitat for Song Sparrows. Following Boutin et al. (2001), I define hedgerows as "linear structures bordering fields, comprising woody and herbaceous flora and including adjacent field margins". Hedgerows used in the study typically have a herbaceous understory and vary from a few trees and shrubs with large gaps between them, to fully treed (Fig. 1.1). Trees and shrubs in hedgerows used in the study were primarily deciduous. Hedgerows in the study area typically receive very little management, other than removal of overhanging branches. There are few data available on the extent of hedgerows in eastern Canada, but Boutin et al. (2001) showed that hedgerow density has declined in southern Quebec, particularly in landscapes dominated by intensive agriculture. They suggest that hedgerow density in Ontario has also declined, due to increases in mean field size.

Chapter Summaries

In Chapter 2, I test the hypothesis that the biomass of invertebrates suitable for Song Sparrow nestlings is greater on organic soybean fields and adjacent habitats than on conventional fields. In addition, I test whether local variables help explain variations in invertebrate biomass.



Figure 1.1: Examples of two hedgerows included in my thesis.

In Chapters 3 and 4, I focus on the foraging habitats and diets of Song Sparrows nesting in farmland. Studies of diet and foraging habitat of farmland birds have relied on foraging observations and fecal analysis. Foraging observations are time-consuming, and difficult to analyze because birds may visit numerous habitats on a single foraging trip, so that determining which habitats are actually used for foraging can be difficult. Fecal analysis is biased towards hard-bodied invertebrates that pass through the digestive system. To study the diet and foraging habitat of Song Sparrows in farmland, I develop an approach based on stable isotope analysis. In Chapter 3, I demonstrate that stable isotope methodology can be used to distinguish between invertebrates captured in hedgerows and hayfields from those captured in crop fields. In Chapter 4, I use this knowledge of stable isotope patterns of farmland invertebrates to investigate the foraging habitats and diet of nestling Song Sparrows, by comparing stable isotope values of nestling claw tissue with stable isotope values of different invertebrate taxa and different foraging habitats.

In Chapter 5, I compare the abundance of preferred nestling food types close to the nest with Song Sparrow nestling mass and number of fledglings. I also test the effect of availability of local foraging habitat on nest survival.

Together, my thesis advances our knowledge of the importance of food supply to birds breeding in farmland in North America, as well as developing a new approach for studying diet and foraging habitat of farmland birds, that will have applications in investigating pesticide exposure and provision of ecosystem services by farmland birds.

Chapter Two

Higher chick food biomass in organic than conventional
soybean fields in eastern Ontario, Canada.

This chapter forms the basis of a manuscript submitted to *Agriculture
Ecosystems and Environment*.

Girard, J., Mineau, P. and Fahrig, L. Higher chick food biomass in organic
than conventional soybean fields in eastern Ontario, Canada. *Agriculture
Ecosystems and Environment*.

Abstract

Intensive farming methods have been linked to population declines of farmland and grassland birds. One possible mechanism for these declines is a reduction in the biomass of small invertebrates, necessary for growth and survival of the young of insectivorous birds (nestling food). Organic farming, with its avoidance of synthetic pesticides and fertilizers, and longer more diverse crop rotations, may support higher nestling food biomass than conventional farming. I used pitfall traps to test this prediction in nine organic and nine conventional soybean fields in eastern Ontario, while controlling for landscape level crop cover. I also tested whether weed cover, tillage (measured as cover of crop residue) or hedgerow plant species richness helped explain patterns of nestling food biomass. I found that overall nestling food biomass was significantly greater on organic fields than conventional fields in both June (43 % greater in organic than conventional) and July (35 %). This difference was marginally significant in crop fields, and not significant in field edges or hedgerows, when habitats were examined separately. Weed cover and hedgerow plant species richness did not significantly predict nestling food biomass, while cover of in-field crop residue positively predicted nestling food biomass, but only in hedgerows. The greater nestling food biomass in organic fields suggests that organic management has the potential to support higher levels of avian breeding success than conventional management.

Introduction

Intensive farming practices, such as use of pesticides and artificial fertilizers, simplification of crop rotations, and reductions in semi-natural habitat in farmed areas have been linked to declines in abundance and richness of many taxa, especially birds (Jobin et al., 1996, Donald et al., 2001, Benton et al., 2002, Chamberlain, 2002, Geiger et al., 2010). One possible mechanism for reduced abundance and richness of birds in intensively managed agriculture is inadequate biomass of small invertebrates (Campbell and Cooke, 1997, Wilson et al., 1999, Chamberlain, 2002, Freemark and Kirk, 2001, Beecher et al., 2002, Piha et al., 2007). Small invertebrates (hereafter nestling food) are required for survival and growth of the young of many bird species, and farmland birds breeding at nest sites with lower nestling food abundance, or higher insecticide applications, can show reduced nestling condition or mass (Brickle et al., 2000, Boatman et al., 2004, Brickle and Peach, 2004, Morris et al., 2005, Hart et al., 2006, Dunn et al., 2010), reduced nestling or chick survival (Hill, 1985, Potts, 1986, Boatman et al., 2004, Britschgi et al., 2006), reduced fledging success (Boatman et al., 2004, Rodríguez et al., 2006), and reduced nest survival (Hill, 1985, Brickle et al., 2000, but see Powell, 1984, Bradbury et al., 2003, Zalik and Strong 2008). Reduced food abundance has been linked to population declines in Grey Partridges (*Perdix perdix*, Potts, 1986), and is suspected to be a factor in declines in a number of other species (Campbell and Cooke, 1997, Boatman et al., 2004).

Many studies have suggested that organic farming methods, which do not use synthetic pesticides or fertilizers and tend to use more varied crop rotations than conventional farming methods, might support higher abundances of invertebrates (reviewed by Kromp, 1999, Bengston et al., 2005, reviewed by Hole et al., 2005, Garratt et al., 2011). However results differ widely among and within taxa (e.g. Moreby et al., 1994, Bengston et al., 2005, Purtauf et al., 2005, Schmidt et al., 2005, Kragten et al., 2011) and with landscape context (Rundlöf and Smith, 2006, Diekötter et al., 2010). Since nestling food includes a range of invertebrate taxa (Wilson et al., 1999, Holland et al., 2006), assessing potential changes in nestling food from studies on different taxa is not straightforward. Also, most previous studies have focused on abundance of invertebrates, rather than biomass. Invertebrate abundance and biomass are not interchangeable, and can show different patterns in response to environmental variables (Saint-Germaine et al., 2007). Previous studies on nestling food in organic farmland have found mixed results; Moreby et al., (1994) found no difference in abundance of nestling food items in organic and conventional wheat fields in England. Reddersen (1997), working in Denmark, found that densities of bird food items were significantly greater in the midfield of organic cereal fields than in conventional fields, but not in field margins. Finally, Kragten et al. (2011) found that earthworms and aerial invertebrates were more abundant on organic farms in the Netherlands across a range of crop types, but that abundance of ground invertebrates was not significantly different between organic and conventional farms.

Possible mechanisms for differences in invertebrate biomass or abundance between organic and conventional management systems include differences in pesticide use, tillage and in plant species richness. Insecticides, by design, have a direct negative effect on invertebrates, whereas herbicides have an indirect effect on invertebrates, by reducing weed cover and diversity (Boatman et al., 2004). Weeds affect invertebrates by providing food, altering microclimate and providing shelter from predators (reviewed by Norris and Kogan, 2005, Diehl et al., 2012). However, relationships between weed cover and invertebrate abundance are not straightforward. For example, Buckelew et al. (2000) found both positive and negative relationships between weed cover and different pest species in soybean. Activity density of ground beetles (Carabidae) is typically found to decrease with decreasing weed cover (reviewed by Diehl et al., 2012), but the response of ground beetles to weeds can vary by species (Pavuk et al., 1997).

Organic farmers typically use high rates of tillage to control weeds, whereas tillage practises on conventional farms vary widely. Tillage not only kills invertebrates directly, it also alters soil structure and stratification of organic matter in the soil, and reduces the amount of crop residue left on the surface, affecting microclimate and food availability for invertebrates (House and Parlee, 1985, Stinner and House, 1990, Chan, 2001, Thorbek and Bilde, 2004). Tillage typically decreases the abundance or activity density of soil macro arthropods such as ground beetles and spiders (Araneae, House and Parlee, 1985, Stinner and House, 1990, Holland and Reynolds, 2003, Thorbek and Bilde, 2004,

Baraibar et al., 2009, Errouissi et al., 2011). However, some studies show either no effect of tillage (Carcamo et al., 1995, Baguette and Hance, 1997, Huusela-Veistola, 1998), or a positive effect of tillage (Carcamo 1995, Whalen et al., 2007), and results often vary by taxon (Tonhasca, 1993, Baguette and Hance, 1997, Shearin et al., 2007, Thorbeck and Bilde, 2004). Results for pests and invertebrates living in foliage are even more variable (Stinner et al., 1988, Stinner and House, 1990, Tonhasca, 1993, Costamagn and Landis, 2006, Whalen et al., 2007, Hatten et al., 2010). The negative effect of increased tillage on at least some invertebrate taxa might explain some of the variability in invertebrate response to organic field management.

Organic farms typically support higher species richness of plants than conventional farms (reviewed by Bengtsson et al., 2005, Roschewitz et al., 2005, Ekroos et al., 2010, Jonason et al., 2011, Batáry et al., 2012, Nascibene et al., 2012, but see Weibull et al., 2003). Higher species richness of plants has been linked to increased invertebrate abundance (Thomas and Marshal, 1999, Asteraki et al., 2004, Hyvonen and Huusela-Veistol, 2011) and size (Woodcock et al., 2005). However, Perner et al. (2005) showed that plant richness was not correlated with invertebrate abundance when additional factors such as soil, plant species composition and productivity were accounted for.

The aim of this study was to test a) whether nestling food biomass is higher in organic than conventional soybean fields and b) whether local variables related to farm type (weed cover, tillage (measured as cover of crop residue) and

hedgerow plant species richness) help explain variation in nestling food biomass in soybean fields. I examined these relationships for total nestling food biomass and for biomass in crop fields, field edges and hedgerows separately, because use of these foraging habitats differs among bird species (Boutin et al., 1999, Puckett et al., 2009), and because Reddersen (1997) showed that densities of bird food items between organic and conventional fields were stronger within fields than at field edges. I predicted that nestling food biomass would be higher in organic fields than in conventional fields, and that weed cover, cover of crop residue and species richness of hedgerow plants would all be positively related to nestling food biomass.

Methods

Study area

The study was carried out in eastern Ontario, to the northwest and southeast of Ottawa (Fig. 2.1). This area is part of the St Lawrence lowlands ecodistrict (Ecological Stratification Working Group, 1995). Mean annual temperature is approximately 5 °C and mean summer temperature approximately 16.5 °C (Ecological Stratification Working Group, 1995). Mean length of growing season is 205-230 days (Crins et al., 2009). Approximately 25 % of eastern Ontario is farmed; the three dominant crops in the region are hay (approx. 275,000 ha, 37 % of farmed land), corn (*Zea mays*, approx. 111,000 ha, 15 %)

and soybean (*Glycine max*, approx. 84,000 ha, 12 %, Ontario Ministry of Agriculture Food and Rural Affairs, 2011).

Study sites

I selected soybean as the focal crop, because it is widely grown by both organic and conventional farms in the study region. To find organic farms I used local directories of organic growers and contacted local sellers of organic seed. I found nine organic farms growing soybeans spread over an area of approximately 100 km x 25 km between Fitzroy Harbour (45°28'12.12"N, 76°12'48.76"W) and Iroquois, Ontario (44°50'58.74"N, 75°18'58.90"W, Fig. 2.1). One soybean field and adjacent hedgerow were selected for sampling on each farm. To control for changes in physiographic features and species assemblages over the study area, I found a conventional farm close to each organic farm, and selected a soybean field and adjacent hedgerow for sampling. Wherever possible I selected the conventional hedgerow to have similar structure to the organic hedgerow (e.g. presence of ditch, similar cover of shrubs and trees). Paired fields were between 2.8 and 6.4 km apart, and all fields were at least 2 km apart. I therefore had a total of nine organic and nine conventional soybean fields.

Model species

To aid in the study design, Song Sparrows (*Melospiza melodia*) were used as a model farmland bird species. Song Sparrows are common and abundant in farm fields (Best et al., 1995, Jobin et al., 1996, Boutin et al., 1999, Best, 2001,

Freemark and Kirk, 2001, Jobin et al., 2001, Henningsen and Best, 2005). They forage most often on the ground, and during the breeding season feed primarily on small invertebrates (Arcese et al., 2002).

Invertebrate trapping

I used pitfall traps to sample invertebrates active on the soil surface, and therefore accessible to ground-foraging birds. Pitfall traps consisted of plastic cups, placed in holes in the ground, so that the top of the cup was level with the ground surface. Traps were one-third filled with water, with a drop of soap added to break the water tension, and were in place for three days. The short length of time that the traps were in place meant that I did not need to use lids to prevent either flooding or by-catch.

Trapping was carried out twice in each field, once in late May-early June (31 May-8 June 2009, hereafter June), and once in July (14-30 July 2009). These time periods were chosen to approximate peak nestling-feeding times of Song Sparrows (J. Girard, unpublished data). Each pair of fields was sampled on the same days. Within each field, two 60 m sampling transects were set up, with the centers of the two transects at least 95 m apart along the same hedgerow. To select transect locations, I mapped locations of singing Song Sparrows on two visits in May, and randomly selected a consistent song location for the center of each transect. Each transect consisted of 12 pitfall traps, four each in the soybean field, in the field edge and in the hedgerow (Fig. 2.2).

When the traps were collected, the contents of each trap were bagged separately and returned to the lab. Samples were rinsed through a 1 mm sieve and stored in denatured ethanol until the sample could be counted. Invertebrates in each sample were identified to order or family (Appendix A), counted, and measured in 2 mm intervals. To calculate biomass of invertebrates important in Song Sparrow nestling diet (hereafter nestling food biomass), I first excluded invertebrates not important in nestling diet because they are noxious or difficult for nestlings to handle (Hymenoptera, Sternorrhyncha, Acari, Gastropoda, Oligochaeta) and invertebrates from aquatic habitats (Dytiscidae). I then used equations created from invertebrate captures in pitfall traps from the same habitat and study region (Chapter 5, Appendix R), to convert counts of invertebrates in each taxon size class to dry nestling food biomass (mg) for each trap.

Not all traps were recovered from the field sites, due to occasional destruction by animals, flooding, or farm machinery. The number of traps included in the analysis is shown in Appendix B.

Farm management

Farmers were contacted between 28 November 2009 and 26 February 2010, to assess how field management differed between organic and conventional farms. A standardized questionnaire was used (Appendix C) and interviews were conducted either in person or over the telephone. Seven conventional farmers and eight organic farmers provided information about farm management

(Appendix D). Planting dates in organic soybean fields had a smaller range than in conventional fields, and started and ended later (organic range: 18 May - 7 June, conventional range: 9 May – 4 June). Fertilizer was applied at only one conventional site and two organic sites. The number of spring tillage passes in organic fields was higher than the number of passes on conventional fields (organic: mean 2.9 ± 1.8 SD, conventional: 1.0 ± 0.6). All the conventional soybean fields were sprayed with herbicide at least once, but only one field was sprayed with insecticide, after invertebrate trapping was carried out. Four conventional fields were planted with treated seeds; two were treated with fungicides, and two with fungicides and insecticides. Two organic fields had inoculate seed treatments added to promote nitrogen fixing. Of the six conventional fields for which I had information, the previous year's crop was corn in five fields, and soybean in one field. Of the eight organic fields for which I had information, the previous year's crop was much more varied; two fields were grown in wheat the previous year, two in corn, one each in rye, hay and soybean, and one was planted in wheat and then seeded in clover after harvest. All the conventional fields used only two crops in the standard crop rotation (corn and soybean) whereas all the organic fields used at least three crops (mean 3.8 ± 0.8).

Cover of weeds and crop residue

At each sampling time, I recorded percent cover of weeds and crop residue in four 1 m² quadrats, within each invertebrate trapping transect. Two quadrats were placed in the field, and two at the field edge (Fig. 2.2).

Hedgerow plants

Between 12-14 August 2009, I surveyed hedgerow vegetation in each field to assess plant species richness and diversity. Plants were surveyed along the 60 m long invertebrate trapping transects (Fig. 2.2). I identified all plants in the grass/forb layer between the edge of the crop and the center of the hedgerow, and estimated total cover of each species along the whole transect in 3 categories; <10%, 10-50%, >50%. I calculated species richness and Shannon-Weiner diversity index in each transect. These were highly correlated ($t = 6.95$, $df = 14$, $P < 0.01$, $r = 0.88$), so I only used species richness for analyses.

Landscape data

Landscape context can have a strong effect on species richness and abundance of invertebrates and other taxa on farmland (reviewed by Winquist et al., 2012). In addition, management type can interact with landscape context such that the effects of organic agriculture tend to be smaller or disappear in 'complex' landscapes with less agricultural cover and more cover types (Roschewitz et al., 2005, Rundolf and Smith 2006, Holzschuh et al. 2007, Geiger et al., 2010, Smith et al., 2010, Batáry et al., 2012). I compared landscape metrics (below) between organic and conventional fields to look for possible

differences, and included crop cover in the landscapes in models predicting nestling food biomass, to control for differences in landscape complexity.

I obtained land use data from a 1 km radius around each field from aerial photographs taken in 2008 (DRAPE 2010). Using ArcMap™ v 10.0 (ESRI 2011), I digitized the following land uses: forest, hay fields, hedgerows, other (includes successional or abandoned land and other unmanaged habitats), pasture, crop fields, riparian/wetland, unknown and verge (roadside vegetation).

To compare the pattern of land uses surrounding the organic and conventional sites, I calculated percent area of crop fields, percent area of open semi-natural habitats (includes hay fields, hedgerows, other, pasture, riparian/wetland and verge), mean field area (calculated across crop fields, hayfields and pasture) and the Shannon-Weiner diversity index of all land uses within a 500 m radius of each site, approximating average farm size in the study area. None of the metrics was significantly different between organic and conventional sites (paired t-test, Table 2.1).

To include crop cover in models predicting nestling food biomass, I needed to know at what scale crop cover is important. The scale at which crop cover affects individual taxa is very variable; for example, the scale of effect for spiders tends to be quite large (e.g. 1750m Drapeala et al., 2011, up to 3km Schmidt et al., 2008), whereas for ground beetles the scale of effect can be much smaller (e.g. 250 m Maisonhaute et al., 2010). To my knowledge, no one has looked at the scale at which landscape affects nestling food biomass. To select a

scale at which percent crop cover should be included in the nestling food biomass models, I calculated crop cover at multiple radii (100 m to 1000 m at 100 m intervals) and created models predicting nestling food biomass at each scale. For all habitats combined, and for crop fields, models including crop cover at 1000m had the lowest score for sample-corrected Akaike's Information Criterion (AIC_c; Hurvitch and Tsai 1989), while for edge the most important scale was 500 m and for hedgerows 400 m (Appendix E).

Statistical analysis

Effect of management type on nestling food biomass

To account for the hierarchical study design, I used linear mixed effects models to investigate the effect of management type (organic vs. conventional) on nestling food biomass per trap (Pineiro and Bates 2000). This allowed me to include the variability between pitfall traps within the same sampling field in the model. Nestling food biomass was log transformed to improve normality. Fixed effects were management type, month and crop cover. Month was included as a fixed effect, to allow me to examine whether patterns in nestling food biomass changed with time. I first included an interaction between crop cover and management type, since previous studies have shown that the effect of management type can vary with crop cover (see above). However, this interaction was never significant, so only the additive effect of crop cover on nestling food biomass is reported in the results. To account for the hierarchical study design, the random effects of habitat, nested in month, nested in site

needed to be included in the model. Possible additional random effects were pair and transect. To test whether these additional random effects needed to be included in the model, I first created a global model including all predictor variables, and used AIC_c to select the most parsimonious combination of random effects (Zurr et al., 2009). AIC_c attempts to find the best expected Kullback-Leibler model for the current sample size, therefore incorporating a bias-variance trade off (Burnham and Anderson 2004). AIC_c is also appropriate for tapering effect sizes, which are nearly always a factor in model selection (Burnham and Anderson 2004). Since there are 18 sites in the study, 18 was used as the correction factor for AIC_c (Mazerolle, 2012). Neither improved the global model, and were therefore excluded from the analysis. I first analyzed nestling food biomass across all habitats (total nestling food biomass). I then analyzed each habitat (field, edge and hedgerow) separately, to see whether the effect of management on nestling food biomass differed among habitats.

Effect of local variables on nestling food biomass

The three local variables (weed cover, cover of crop residue and hedgerow plant species richness) were averaged across transects at each site for analysis. Weed cover and cover of crop residue data were arcsine transformed to improve normality. I first tested the effect of management type and month and their interaction on each variable, using an ANOVA. If there was no effect of management type and month, I then proceeded to use an ANCOVA, with a random effect of site, to test for the effect of the local variable on nestling

food biomass, while controlling for management type and month. If there was an effect of management type, I tested the effect of the local variable on nestling food biomass within month, using a linear mixed model with site as the random variable. If there was an effect of both management type and month, I tested the effect of the local variable on nestling food biomass within each management type by month combination, using linear regression, as appropriate. All analyses were carried out in R (v 2.14.1, R Development Core Team 2011). Mixed effects models were run using the nlme package (Pinheiro et al. 2009). Results are reported as mean \pm 1 SD.

Results

Invertebrate captures

Ground beetles dominated pitfall trap captures, comprising on average 30 % of dry biomass/trap (0.09 mg \pm 0.18 mg), followed by millipedes (Diplopoda , 20%, 0.03 mg \pm 0.07) and true spiders (15%, 0.02 mg \pm 0.02).

Management type

Both management type and month significantly affected total nestling food biomass; as predicted, the overall biomass of nestling food invertebrates was higher in organic fields than in conventional fields in both June and July (Fig. 2.3, Table 2.2, see Appendix F for graph of raw data). For an average site, the mixed

model predicted that nestling food biomass per trap was 43 % higher in organic fields than conventional fields in June, and 35 % higher in organic fields than conventional fields in July. Nestling food biomass was also significantly greater in July than in June (organic fields, 57% increase between June and July, conventional fields 70 %). The proportion of the surrounding landscape in crop fields did not significantly affect total nestling food biomass (Table 2.2, see Appendix Z for raw data for these analyses).

When the three habitats were examined separately, nestling food biomass in crop fields and field edges showed the same pattern of greater nestling food biomass in organic fields than conventional fields, and greater nestling food biomass in June than in July (Fig. 2.3), but the effects of management in crop fields were only marginally significant, and in crop fields were non-significant (Table 2.2). In hedgerows, the pattern was also similar to overall nestling food biomass, but neither management type, nor month, significantly predicted nestling food biomass; however, there was a marginally significant positive effect of crop cover in a 400 m radius (Table 2.2).

Weed cover

When I modelled weed cover based on management type and month, there was a significant interaction (management type: $F_{1,16}=6.42$, $P=0.02$, month: $F_{1,16}=48.23$, $P<0.001$, type*month, $F_{1,16}=12.92$, $P=0.002$), so I tested for the effect of management type on weed cover separately in each month. There was no significant difference between weed cover in organic and conventional fields

in June, but in July, weed cover was significantly higher in organic fields than in conventional fields (June organic: 1.53 ± 1.36 , June conventional: 3.63 ± 5.48 , $F_{1,16}=0.48$, $P=0.50$, July organic: 36.12 ± 18.74 , July conventional: 12.03 ± 12.11 , $F_{1,16}=11.86$, $P=0.003$).

In crop fields and in hedgerows, weed cover did not significantly predict nestling food biomass in either organic or conventional fields in June or July (Table 2.3). In field edges, weed cover did negatively predict nestling food biomass in organic fields in June, but did not significantly predict nestling food biomass in organic fields in July or in conventional fields in either month (Table 2.3, Fig. 2.4). The negative relationship in June was driven by one site, without which the relationship was non-significant (regression coefficient \pm SE including outlier: -4.64 ± 0.91 , excluding outlier: -2.94 ± 1.92).

Cover of crop residue

There was no significant effect of management type on cover of crop residue in either month, but residue cover was greater in June than in July (June organic: 25.49 ± 26.41 , July organic: 19.95 ± 28.39 , June conventional: 35.11 ± 27.86 , July conventional: 27.68 ± 24.34 , management type: $F_{1,16}=0.74$, $P=0.40$, month: $F_{1,16}=14.86$, $P=0.001$, type*month: $F_{1,16}=1.34$, $P=0.26$).

In crop fields and in field edges, cover of crop residue did not significantly predict nestling food biomass in either month (Table 2.4). However, for nestling

food biomass in hedgerows, cover of crop residue in the field was a significant positive predictor in July (Table 2.4, Fig. 2.5).

Hedgerow plant species richness

Mean hedgerow plant species richness/site ranged from 15 to 36.5 species (25.5 ± 6.0). Hedgerow species richness was not significantly different between organic and conventional hedgerows (organic: 27.17 ± 5.66 , conventional 23.83 ± 6.26 , $F_{1,16}=1.40$, $P=0.25$), and did not significantly predict nestling food biomass within soybean fields, at field edges or in adjacent hedgerows (Table 2.5).

Discussion

This is the first study I know of in North America to compare nestling food biomass in organic and conventional agriculture. As I predicted, nestling food biomass was higher in organic soybean than in conventional soybean fields despite the low level of insecticide use in the conventional fields. This was significant for all habitats combined, and marginally significant within crop fields. Contrary to my predictions, weed cover and hedgerow plant species richness did not help explain the difference in nestling food biomass between organic and conventional fields, and I found only weak evidence that cover of crop residue predicted nestling food biomass in hedgerows only.

I found that the difference in nestling food biomass between organic and conventional soybean fields was strongest within the crop field. This is presumably where the difference in management between organic and conventional fields is greatest. Reddersen (1997) also found that the differences in bird food densities between organic and conventional fields were greater in field centres than at field edges. This suggests that if nestling food biomass is driving bird abundance or richness in farmland, the birds that benefit most from organic farming will be those that are most dependent on crop fields for foraging. For example Piha et al., (2007), comparing the effects of landscape structure and organic farming on farmland bird assemblages, found that the bird species that showed increases in population densities in organic fields (Lapwing, *Vanellus vanellus*, and Skylark, *Alauda arvensis*) were those most strictly associated with agricultural fields.

Previous studies have shown that low nestling food biomass or abundance in intensively managed fields can reduce breeding productivity of farmland birds (Hill, 1985, Potts 1986, Brickle et al., 2000, Boatman et al., 2004, Brickle and Peach, 2004, Morris et al., 2005, Britschgi et al., 2006, Hart et al., 2006, Rodriguez et al., 2006, Dunn et al. 2010). These studies typically report food availability as abundance of invertebrates, weighted by foraging habitat availability (Brickle et al., 2000, Boatman et al., 2004) or as a food availability index (Brickle and Peach, 2004), so my data are not directly comparable. The one study that reports raw pitfall data (Britschgi et al., 2006) was carried out in grassland, so again the data are not directly comparable with my study.

However, that study showed that nestling food biomass measured by pitfall traps was about 3 to 7 times greater in the low management-intensity treatment, compared to the high management-intensity treatment, a much greater difference than reported here. A previous study comparing farmland bird productivity on organic and conventional farms found no difference in nestling body condition and growth (Bradbury et al. 2003) or clutch size and nest survival (Bradbury et al. 2000). It is not possible to say if the differences I found in nestling food biomass between organic and conventional fields would influence reproductive success of birds without further study.

I am not able to say exactly what caused the higher nestling food biomass in organic than conventional fields. Insecticide use does not seem likely as the sole explanation, because only two fields were exposed to insecticide use during the study (Appendix D, note that I have no data to evaluate whether insecticide use was more frequent or more wide-spread prior to the current study). My results also suggest that weed cover and hedgerow plant species richness were not important in explaining the difference in nestling food biomass between organic and conventional fields. However, it is possible that while each of the effects of insecticides, herbicides, and hedgerow plant species richness are small, they may accumulate to explain the difference in nestling food biomass between organic and conventional fields. Another possible explanation is the more diverse crop rotation in organic fields compared to conventional fields. Diverse crop rotations, together with absence of herbicide, often increase biomass of soil organisms (bacteria, fungi and micro and macro invertebrates,

Alteri 1999), which can support larger populations of surface-active invertebrates captured in pitfall traps (e.g. Salamon et al. 2011). Use of manure in place of synthetic fertilizers will enhance this relationship (Alteri 1999), but I do not have data on long-term fertilizer use in my study fields. Diverse crop rotations will also act to increase crop diversity in the landscape, which can further increase invertebrate abundance (e.g. Thorbeck and Topping 2005).

I found little evidence that weed cover was driving the difference in nestling food biomass between organic and conventional fields, despite the wide variability in weed cover among my sites (Fig. 2.4). While many previous studies have shown positive effects of weed cover on invertebrate abundance, the effects of weed cover do vary among invertebrate taxa (Pavuk et al. 1997, Buckelew et al. 2000, Hough-Goldstein et al., 2004, Powell et al., 1985), so differing responses among taxa may result in a lack of overall effect of weed cover on nestling food biomass. I also found no evidence of an effect of cover of crop residue, a measure of tillage, on nestling food biomass in crop fields or field edges. This might also be due to different responses to tillage among invertebrate taxa (Hatten et al., 2007, Tonhasca 1993, Baguette and Hance 1997, Shearin et al., 2007, Thorbeck and Bilde, 2004). It is also possible that the effects of tillage on invertebrates may be so short-term that potential effects are not reflected in my measurements of nestling food biomass. However, tillage passes in many fields occurred in the two weeks prior to the June invertebrate sampling, so if there was a negative effect of tillage, I should have been able to detect it. In this observational study, I cannot rule out the possibility that opposite

effects of weed cover and tillage on invertebrate abundance cancelled each other out. Weed cover has generally been shown to have positive effects on non-pest invertebrate abundance (reviewed in Marshall et al., 2003), whereas tillage has often been shown to have negative effects on ground invertebrate abundance (House and Parlee, 1985, Stinner and House, 1990, Holland and Reynolds, 2003, Thorbek and Bilde, 2004, Baraibar et al., 2009, Errouissi et al., 2011). Both weed cover and rates of tillage were higher in organic fields, so it is possible that the opposite effects cancelled each other out, so that I cannot clearly detect the effects of either variable on nestling food biomass.

I did find evidence of a positive effect of crop residue (i.e. a negative effect of tillage) on biomass of nestling food in hedgerows (Fig. 2.5). It is difficult to say why tillage should have a negative effect on nestling food biomass in hedgerows, but not in crop fields or field edges. The result could be spurious. Alternatively, undisturbed habitats, such as hedgerows can act as a source of colonizers after disturbance by agricultural management (Gravesen and Toft, 1987, Thomas et al., 1990, Topping and Sunderland, 1994, Alvarez et al., 2000), so it is possible that hedgerows adjacent to fields with high tillage have lower nestling food biomass due to invertebrates leaving the hedgerow to recolonize the adjacent fields.

I did not find an effect of hedgerow plant species richness on nestling food biomass. This is in contrast to previous studies which have often found that invertebrate abundance is correlated with plant species richness in farmland

(Thomas and Marshal, 1999, Asteraki et al., 2004, Hyvonen and Huusela-Veistol, 2011). This difference might be because I measured biomass of ground invertebrates rather than biomass of invertebrates living in the vegetation, which might be more affected by plant species richness (Thomas and Marshal 1999). In addition, I did not measure species richness of weeds within the crop fields, which may be more important than species richness of hedgerow plants, for invertebrates living in the crop field.

In summary, I did find a positive effect of organic management on nestling food biomass. This may be associated with the more diverse crop rotations typical of organic agriculture. The higher nestling food biomass found in organic fields suggests that organic management has the potential to support increased bird productivity compared to conventional management.

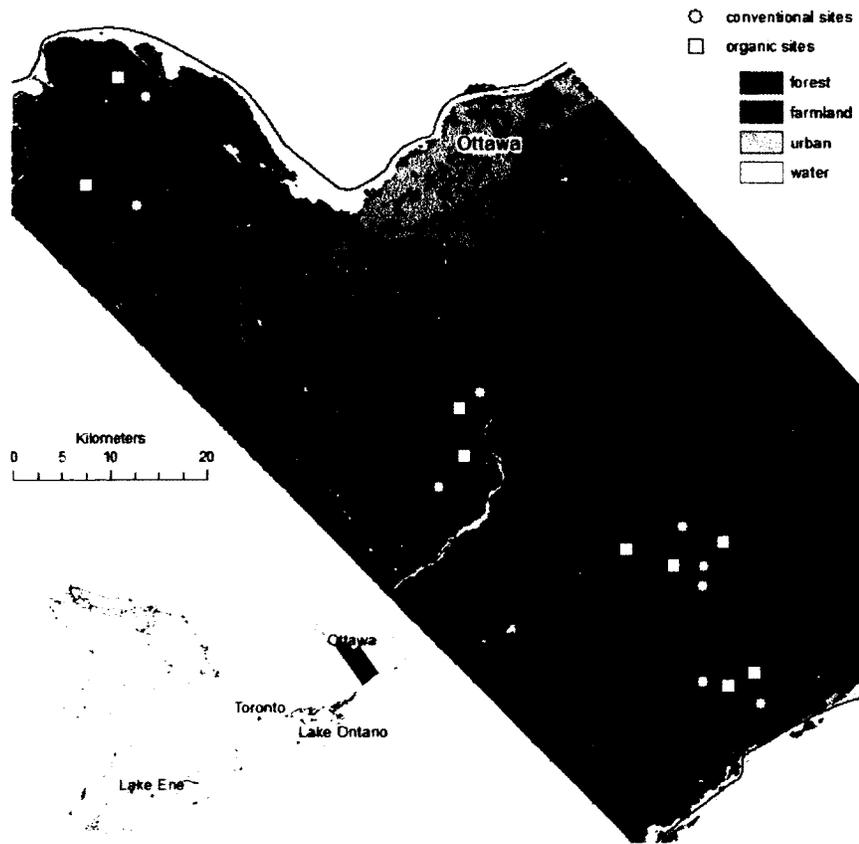


Figure 2.1 – Map of organic (squares) and conventional (circles) study sites in eastern Ontario.

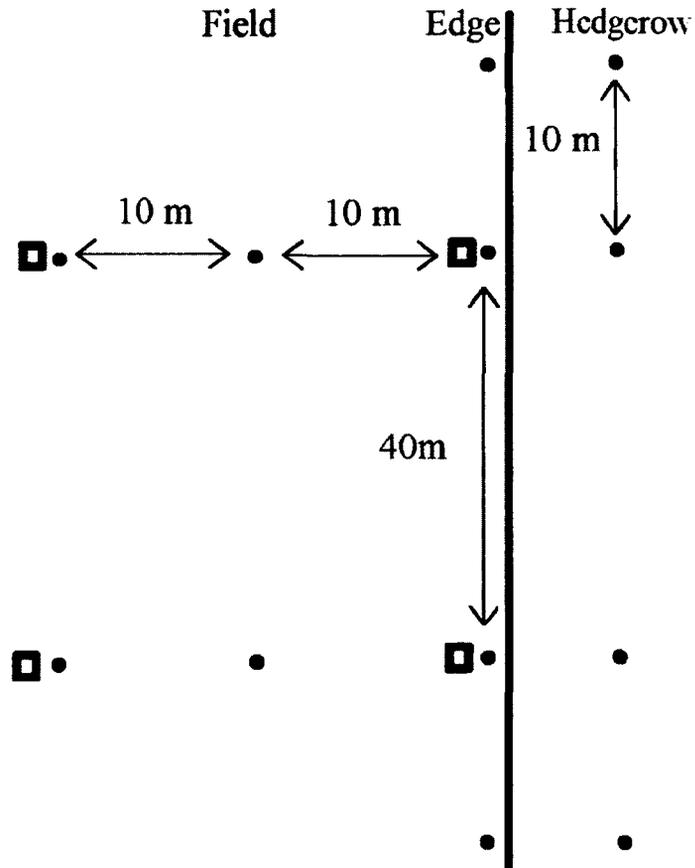


Figure 2.2: Layout of pitfall traps used to sample ground invertebrates. Dots represent position of 12 pitfall traps, four each in the soybean field, four in the edge (at the borderline between the hedgerow and the field) and four in the hedgerow. Squares represent the locations of 1 m² quadrats used to measure cover of weeds and crop residue.

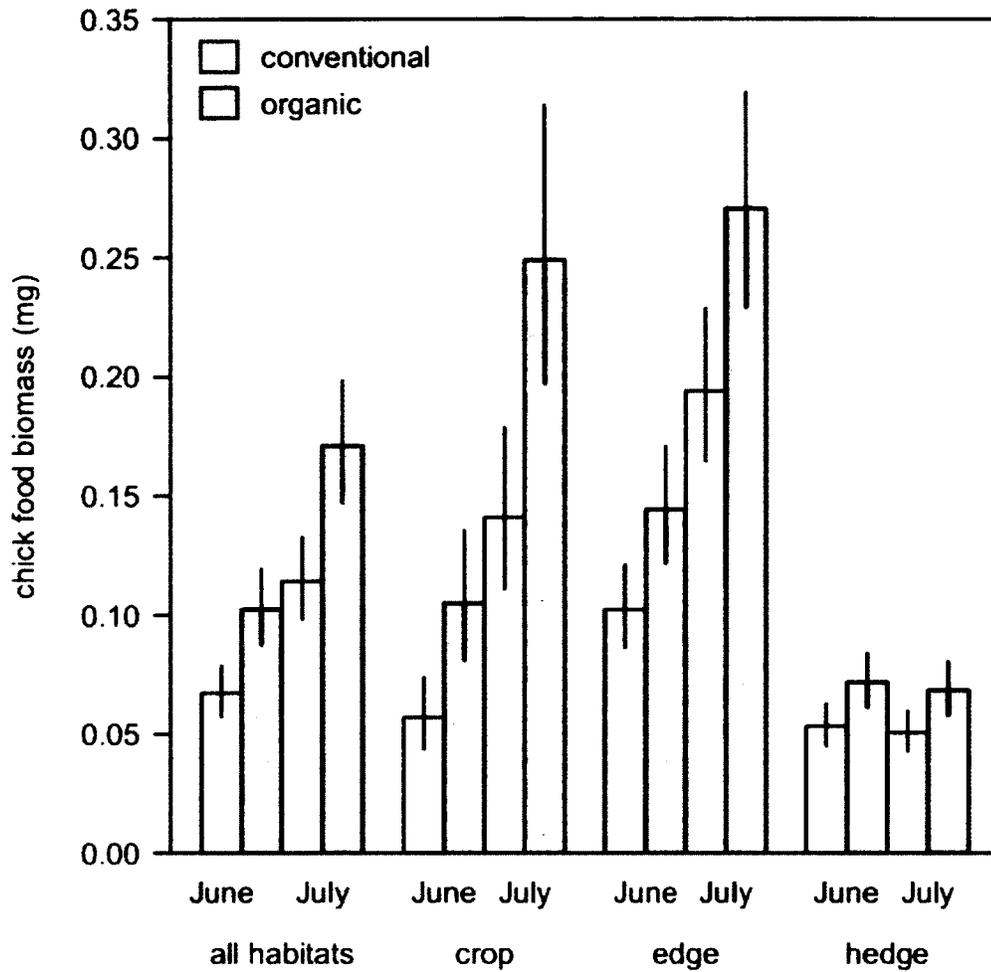
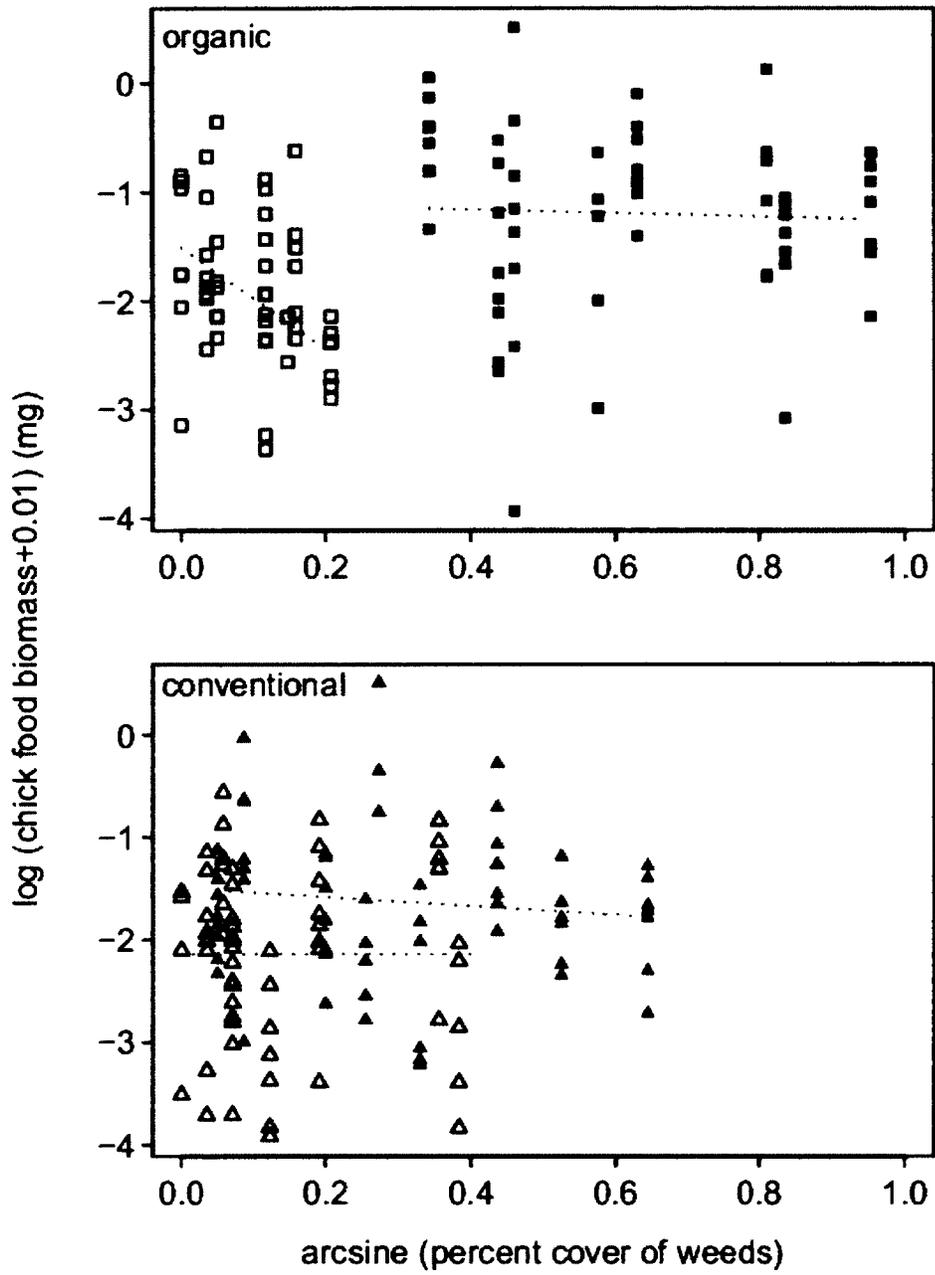


Figure 2.3: Effect of management type (conventional or organic) and month on chick food biomass in 18 soybean fields. Bars show estimated means \pm SE, conditional on random effects of habitat nested in month nested in site for all habitats and site nested in month for the other analyses. See appendix F for graph of raw data.

Figure 2.4: Relationship between mean arcsine transformed percent cover of weeds and chick food biomass in field edges of 18 organic and conventional soybean fields in June (open symbols) and July (filled symbols). Line shows predictions from a mixed effects model predicting chick food biomass from mean weed cover, with site as the random effect.



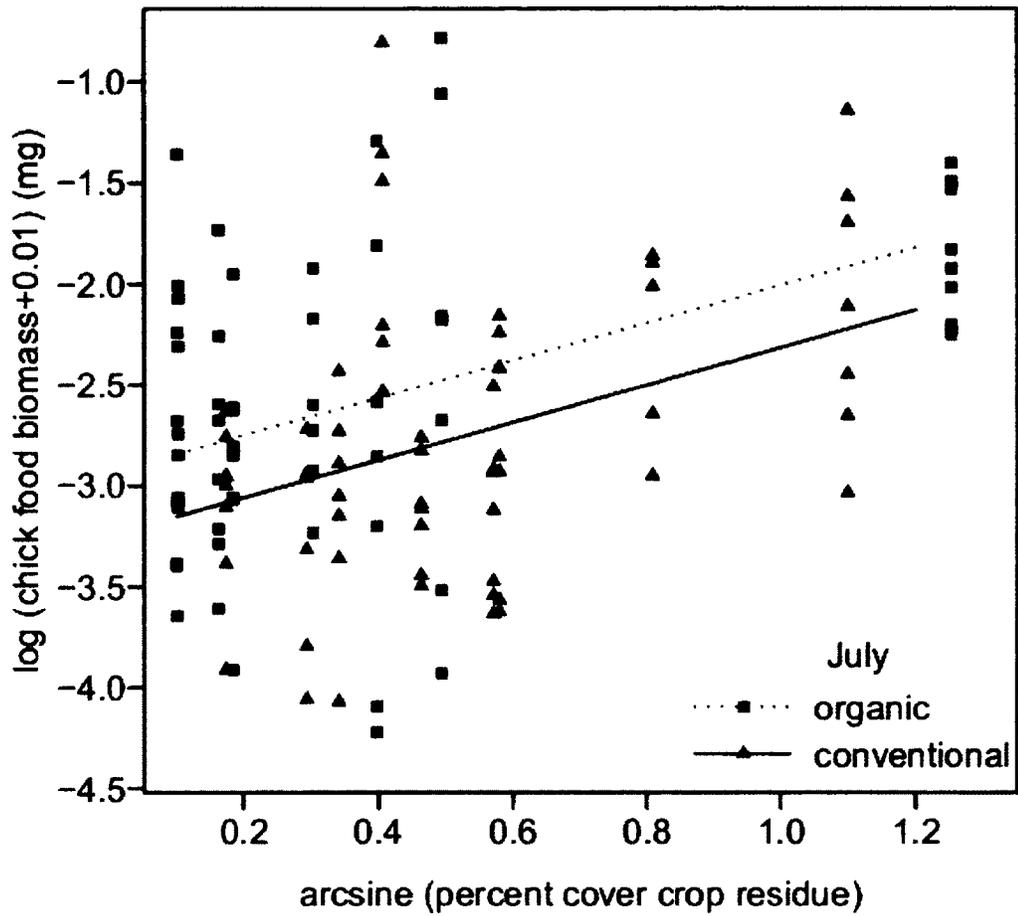


Figure 2.5: Relationship between mean percent cover of residue and chick food biomass in hedgerows adjacent to 18 organic and conventional soybean fields in July. Line shows predictions from a mixed effects model predicting chick food biomass from mean cover of crop residue and management type, with site as the random effect.

Table 2.1: Landscape metrics in a 500 m radius around 9 conventional and 9 organic soybean fields.

Metric	conventional			organic			t ^A	p
	mean (SD)	min	max	mean (SD)	min	max		
percent crop fields	0.57 (0.21)	0.27	0.93	0.57 (0.17)	0.22	0.74	0.03	0.98
percent seminatural								
habitat	0.32 (0.17)	0.05	0.63	0.32 (0.13)	0.17	0.51	0.11	0.91
mean field size (ha)	4.50 (2.34)	2.30	9.33	3.40 (1.09)	1.52	5.53	1.10	0.30
landuse diversity ^B	1.07 (0.43)	0.28	1.51	1.17 (0.35)	0.74	1.91	-0.62	0.55

^A Paired t-test

^B Shannon-Weiner diversity index of all land uses

Table 2.2: Results of mixed models predicting chick food biomass in soybean fields from farm management type (conventional or organic) and sampling month (June or July). Chick food biomass is measured in all habitats (total), or in different habitats associated with soybean fields (within the crop, at the field edge or in the adjacent hedgerow). In the total model, the random effects are habitat type, nested in month, nested in site. In the other models, the random effects are month, nested in site.

Habitat	Management type			Month			Crop cover		
	F	df	p	F	df	p	F	df	p
Total	5.11	1,15	0.04	12.17	1,17	0.003	2.45	1,15	0.14
Crop	4.55	1,15	0.05	10.03	1,16	0.006	2.00	1,15	0.18
Edge	2.54	1,15	0.13	13.00	1,16	0.002	1.35	1,15	0.26
Hedgerow	2.50	1,15	0.13	0.18	1,16	0.68	4.59	1,15	0.05

Table 2.3: Results of mixed models predicting chick food biomass from mean percent weed cover in soybean fields.

The random effect is site.

Habitat	Organic						Conventional					
	June			July			June			July		
	df	F	p	df	F	p	df	F	p	df	F	p
Crop	6	0.01	0.9	7	1.37	0.28	7	0.86	0.38	7	0.99	0.35
Edge	7	26.08	0.001	6	0.05	0.84	7	0.00	0.99	7	0.13	0.73
Hedgerow	7	0.56	0.48	6	0.36	0.57	7	0.40	0.56	7	1.35	0.28

Table 2.4: Results of mixed models predicting chick food biomass from mean percent cover of crop residue and mean species richness of hedgerow plants in soybean fields. The random effects are month nested in site.

Habitat	June						July					
	Management type			Crop residue			Management type			Crop residue		
	F	df	p	F	df	p	F	df	p	F	df	p
Crop	1.40	1,14	0.26	0.22	1,14	0.65	4.96	1,15	0.04	1.12	1,15	0.31
Edge	0.54	1,15	0.47	0.31	1,15	0.58	4.14	1,14	0.06	0.70	1,14	0.42
Hedgerow	3.75	1,15	0.07	1.51	1,15	0.24	3.37	1,14	0.09	10.74	1,14	0.006

Table 2.5: Results of mixed models predicting chick food biomass from mean percent cover of crop residue and mean species richness of hedgerow plants in soybean fields. The random effects are month nested in site.

Habitat	Management type			Month			Species richness		
	F	df	p	F	df	p	F	df	p
Crop	5.66	1,15	0.03	9.88	1,16	0.006	1.70	1,15	0.21
Edge	2.03	1,15	0.18	13.31	1,16	0.002	0.19	1,15	0.67
Hedgerow	2.16	1,15	0.16	0.18	1,16	0.68	2.09	1,15	0.17

Chapter Three

Carbon and nitrogen stable isotopes differ among invertebrates from field crops, forage crops, and non-cropped land uses.

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Girard, J., Baril, A., Mineau, P. and Fahrig, L. 2011. Carbon and nitrogen stable isotopes differ among invertebrates from field crops, forage crops, and non-cropped land uses. *Ecoscience* 18(2):98-109.

Abstract

Stable isotopes are an important tool for studying invertebrate food webs and movement of invertebrates in farmland. However, stable isotope values of farmland invertebrates have been reported for only a few crop types, and rarely for other land uses within farmland. I compared $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ of invertebrates captured in corn, soybean, hay and hedgerows in eastern Ontario, Canada. $\delta^{13}\text{C}$ was significantly lower in invertebrates captured in hay and hedgerows than in invertebrates captured in soybean and corn, but was not different between invertebrates captured in corn and soybean. This suggests invertebrates may be moving between crop fields during the growing season, using alternative food sources within crop fields, or retaining $\delta^{13}\text{C}$ values from the previous year. When all invertebrates were examined together, $\delta^{15}\text{N}$ was significantly higher in invertebrates captured in manured corn, than in those captured in soybeans, hedgerows or manured or unmanured hay, but was no difference between invertebrates captured in manured or unmanured corn. However, spiders from manured corn had significantly higher $\delta^{15}\text{N}$ than those from unmanured corn. Spiders had less variable $\delta^{15}\text{N}$ than other taxa in this study, because they occupy a single trophic level. This may make spiders more suitable for detecting changes in fertilization regimens. By demonstrating how invertebrate $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ vary with land use, this study will contribute to the understanding of agricultural food webs and of responses of invertebrates to land use change.

Introduction

Stable isotope studies can be used to trace carbon sources in farmland food webs by taking advantage of the different $^{13}\text{C}/^{12}\text{C}$ ratios of C4 and C3 plants. Due to their different photosynthetic pathways, C4 plants, such as corn (*Zea mays*), and warm-season grasses, are enriched in ^{13}C compared to C3 plants, which include most other plants, trees, and cool-season grasses (DeNiro & Epstein 1978). This gives C4 plants a higher $^{13}\text{C}/^{12}\text{C}$ ratio, measured as $\delta^{13}\text{C}$ (see methods). Values of $\delta^{13}\text{C}$ change little between trophic levels (DeNiro & Epstein 1978; McCutchan et al. 2003), so $\delta^{13}\text{C}$ can be used to distinguish between consumers using a C4 carbon source from those using a C3 carbon source (DeNiro & Epstein 1978; Ostrom, Colunga-Garcia & Gage 1997; Gould et al. 2002; Prasilka, Heinz & Winemiller 2004; Gratton & Forbes 2006; Vialatte et al. 2006).

$\delta^{13}\text{C}$ can also be used to track movement of invertebrates among farmland land uses. When an organism switches from feeding on C3 carbon sources to feeding on C4 carbon sources, it takes some time for the switch to be fully reflected in its $\delta^{13}\text{C}$ value (Ostrom, Colunga-Garcia & Gage 1997; Prasilka, Heinz & Winemiller 2004; Gratton & Forbes 2006; Albers, Schaefer & Scheu 2006; Vialatte et al. 2006). Therefore, an invertebrate with a $\delta^{13}\text{C}$ value distinctly different from the dominant local vegetation suggests a recent move from a different habitat (complete diet switch), whereas an intermediate value can be interpreted either as a less recent move (complete diet switch partially

assimilated) or as regular movements between habitats, resulting in a mixed diet (Gratton & Forbes 2006). Such a diet switch may be associated with a change in developmental stage, if larvae and adults forage on different carbon sources (e.g. Gould et al. 2002).

To accurately reconstruct food webs or track movement of invertebrates, it is necessary to have isotope values of all important food sources (Lubetkin & Simenstad 2004; Phillips, Newsome & Gregg 2005). However, most farmland studies have focused on only a few crop types. For example, I could find only one field study that reported isotope values for invertebrates in soybeans (*Glycine max*) (Haubert et al. 2009), and only one study that reported isotope values for farmland invertebrates in non-crop habitat (Latendresse 2004). Among other benefits, hedgerows and fencerows provide important habitat for a range of organisms, including invertebrates, birds and mammals in farmland (Baudry, Bunce & Burel 2000; Jobin, Choiniere & Belanger 2001; Jobin et al. 2004), and since they are dominated by C3 plants in temperate latitudes, it should be possible to distinguish between invertebrates from hedgerows and those from corn fields.

Nitrogen has been used in farmland stable isotope studies to investigate trophic levels of invertebrates (McNabb, Halaj & Wise 2001; Albers, Schaefer & Scheu 2006; Wise, Moldenhauer & Halaj 2006), and to track fertilizer type (Bateman, Kelly & Jickells 2005; Choi et al. 2006; Rogers 2008). All consumers become enriched in $^{15}\text{N}/^{14}\text{N}$ with respect to their diet (DeNiro & Epstein 1981), so

that herbivores have higher $^{15}\text{N}/^{14}\text{N}$ ratios (measured as $\delta^{15}\text{N}$, see methods) than the plants they consume, and in turn, predators have higher $\delta^{15}\text{N}$ than herbivores (McCutchan et al. 2003). This allows trophic levels of herbivores and predators to be compared using $\delta^{15}\text{N}$. Animal manure is also enriched in $\delta^{15}\text{N}$ with respect to the animal's diet (Steele & Daniel 1978), and fresh manure typically has values of $\delta^{15}\text{N}$ above 8‰ (Wassenaar 1995; Choi et al. 2002; Vitòria et al. 2004). Artificial nitrogen fertilizer however, has lower $\delta^{15}\text{N}$, typically between -2‰ and 2‰ (Wassenaar 1995; Vitòria et al. 2004). This difference is conserved in crops grown under different fertilizer regimens (Choi, Ro & Hobbie 2003; Choi et al. 2006), and has been used to differentiate between organic and conventional crops (Bateman, Kelly & Woolfe 2007; Rogers 2008). However, unfertilized plants typically show a $\delta^{15}\text{N}$ close to that of the local soil, which averages 5‰, whereas legumes, which actively fix nitrogen, have a $\delta^{15}\text{N}$ close to that of air at 0‰. Therefore, different $\delta^{15}\text{N}$ of leguminous crops, non-leguminous crops and unfertilized semi-natural vegetation could be used to assign invertebrates to different farmland land uses.

The objective of this study was to find out if I could use carbon and nitrogen stable isotopes to distinguish among invertebrates captured within crop fields (corn and soybean), hay fields and hedgerows. I made the following predictions, which should hold true both among and within invertebrate taxa: 1) $\delta^{13}\text{C}$ would be most enriched in invertebrates captured in corn, and lower in invertebrates captured in soybean, hedgerows and hay. 2) $\delta^{15}\text{N}$ would be enriched in invertebrates captured in fields treated with manure compared to

fields treated with artificial fertilizers, unfertilized fields, fields planted with leguminous crops or hedgerows. 3) Despite the variation in $\delta^{15}\text{N}$ values of plants under different fertilizer regimens, I should still be able to detect $\delta^{15}\text{N}$ enrichment in invertebrates at higher trophic levels. 4) Invertebrates captured in edges between hedgerows and crop fields will show intermediate $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values, due to movement across the boundary. The ability to identify invertebrates from these different land uses will help in tracking movement of invertebrates in farmland, and in better defining farmland food webs.

Materials and methods

The study was carried out between 2006 and 2008 in eastern Ontario, Canada, on farms located between the towns of Navan (45°25'15.68"N, 75°25'36.28"W) and Embrun (45°16'26.29"N, 75°16'30.04"W). This is an area dominated by arable and dairy farming. The most common crops are field corn, soybean and wheat (*Triticum* spp.), but hay fields are also common, especially on dairy farms. Tillage practises vary widely from no-till to conventional till. Fertilization practises also vary, with dairy farmers using manure on their fields more frequently than do farmers not involved in animal production. Thirteen sites were used in the study, where a site is a single continuous hedgerow and the adjoining fields. A single site can represent a hedgerow and fields owned by a single farmer, or more often, a hedgerow and fields owned by 2 or more farmers. Hedgerows were selected to avoid deep ditches or areas of permanent water.

This study was part of a larger project examining reproductive success of sparrows breeding in hedgerows, so invertebrate sampling was centered on active sparrow nests. Between 1 and 16 nests were found at each site. Invertebrates were sampled using pitfall traps, sweep netting, hand collection and a D-Vac (D-Vac Company, Ventura, CA, USA, model 122). Sampling methods differed between years. In 2006, pitfall traps consisted of a glass jar placed in a hole in the ground, so that the top of the jar was level with the ground surface. Traps were one-third filled with denatured ethanol, and were in place for 24 hours total. At each nest, 4 traps were placed in the field on one side of the nest, 4 were placed at the interface of the field and the hedgerow (edge traps), 2 were placed in the hedgerow, and 2 were placed in the adjoining field (Fig. 3.1). Edge traps were placed in bare soil, as close as possible to the edge of the hedgerow vegetation. Upon collection, traps were sealed and returned to the lab. The contents of each trap were passed through a 1 mm sieve and rinsed in water. Invertebrates were removed, and stored in a -20°C freezer. Sweep netting was used in hedgerows to collect additional invertebrates. One observer walked 20 paces along the hedgerow, passing the net through the vegetation at each step. The contents of the net were transferred to a jar of denatured alcohol, returned to the lab and processed as for pitfall traps. Finally, observers searched leaves and trunks of hedgerow bushes within 20 m of the nest for caterpillars and other invertebrates. These were transferred to a jar of denatured alcohol, returned to the lab and processed as for pitfall traps. In 2006, invertebrate sampling occurred between 26th May and 26th July.

In 2007 and 2008, pitfall traps consisted of plastic cups, placed in holes in the ground, so that the top of the cup was level with the ground surface. Traps were one-third filled with water, with a drop of soap added to break the water tension, and were in place for 3 days. Preliminary analysis showed there was no difference in the carbon or nitrogen values of invertebrates captured in the ethanol and those captured in water, so no distinction was made between trapping methods in the analysis. Trap layout was similar to 2006, but 8 more traps were used at each nest (Fig. 3.1). Upon collection, the contents of the trap was transferred to a Ziploc bag and returned to the lab. The contents of each trap was passed through a 1mm sieve and rinsed with water. Invertebrates were transferred to a vial of denatured ethanol until further processing. A D-Vac (Dietrick, Schlinger & van den Bosch 1959) was also used to collect invertebrates, in hedgerows, crop fields and hay fields. In the hedgerow and in soybean, a 340 mm diameter funnel was used on the D-Vac. A single sample was collected by placing the sampling funnel vertically over the vegetation and pressing it to the ground for 5 seconds in each of 3 places within a 2 m² area. Where corn was being sampled, a narrower 100 mm diameter funnel was used, and the collector passed the funnel directly over the plants from top to bottom, walking along the row as necessary until 20 seconds of sampling was completed. In each case, the contents of the net were then transferred to a jar of denatured alcohol. D-Vac sampling was carried out twice in the hedgerow, within 20m of the nest, and twice in each adjacent field. In 2007, invertebrate sampling was

carried out between May 30th and August 1st and in 2008 between May 24th and July 18th.

Five taxa were selected for analysis, due to their trophic level, their abundance in samples, and their expected importance in sparrow diet. These were true spiders (Araneae, n=171 samples analysed, carnivorous, abundant in pitfall traps, D-Vac and sweep net samples, important in nestling sparrow diets), ground beetles (Carabidae, n=167, various diets including herbivorous and carnivorous, abundant in pitfall traps), crickets (Ensifera, n=26, omnivorous, fairly abundant in pitfall traps), hoppers (Auchenorrhyncha, n=55, herbivorous, fairly abundant in pitfall traps, D-Vac and sweep net samples), and caterpillars (Lepidoptera larvae, n=35, herbivorous, very important in nestling sparrow diets). This coarse level of taxonomic resolution reflects selection of food items by larger insectivorous predators, like songbirds. The contents of each trap were sorted under a microscope, and individuals of the focal taxa were removed for isotope sampling. When individuals from one trap were not numerous or large enough for isotope sampling (i.e. the dried sample was less than the 1mg required for stable isotope analysis), individuals from different traps from the same field at the same nest were combined. This may result in individuals from different species being combined into a single sample. Once isolated, each sample was either freeze-dried (2006) or oven dried at 60^oC for at least 48 hours until dry (2007, 2008). Samples were then crushed into a fine powder, and approximately 1mg was weighed into a tin capsule.

To assess $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ of local plants, I collected plant samples near 41 sparrow nests. Plant samples were taken from the hedgerow and from the adjacent fields in late July or early August. Corn and soybean samples were collected by walking approximately 20 m into the field and collecting a leaf from the top of each of 2 plants, ≥ 20 m apart. Hay samples were collected by walking approximately 20 m into the hay, and taking a sample of each plant species within a 1 m radius. This was repeated at two sampling points, ≥ 20 m apart. Plant surveys were conducted in hedgerows to determine the dominant herbaceous plant types. A 100m transect was laid out parallel to the hedgerow, and centered on a sparrow nest. Plant species which were estimated to cover $>70\%$ of the hedgerow along the transect were sampled. Samples were collected by taking the top leaves of two individual plants ≥ 20 m apart. At each nest, samples of each plant species were combined to make one sample. Samples were shredded by hand, and approximately 1g of material was weighed into a container and oven dried at 60°C for ≥ 48 hours until dry. Samples were then crushed into a fine powder, and approximately 1mg was weighed into a tin capsule.

Samples and standards were loaded into an elemental analyser (Vario EL III), which was interfaced to an isotope ratio mass spectrometer (Conflo II, Delta XP Plus Advantage). Standards used were C-51 Nicotiamide (0.07, -22.95), C-52 mix of ammonium sulphate and sucrose (16.58, -11.94), C-54 caffeine (-16.61, -34.46), blind standard C-55: glutamic acid (-3.98, -28.53) for carbon, and air for nitrogen. The data are reported in delta notation, defined as $\delta = (R_{\text{sample}} -$

$(R_{\text{standard}}/R_{\text{standard}})^*1000$ where R is the ratio of the abundance of the heavy to the light isotope. All samples were analysed at the Hatch Laboratories at the University of Ottawa.

I used nested linear mixed effects models to analyse the data (Pineiro et al. 2009). Mixed effects models allowed me to account for the nested structure of the sampling design, where the location of each sample was within a specific field or section of hedgerow (Fig. 3.2), at a specific nest and within a specific site. If these random effects are not considered, the assumption of independence is violated, since in many cases multiple invertebrates of the same taxa were sampled within the same field, and at the same site.

Prediction 1, that $\delta^{13}\text{C}$ would be most enriched in invertebrates captured in corn, and lower in invertebrates captured in soybean, hedgerows and hay, was tested by comparing $\delta^{13}\text{C}$ among different trapping locations, both among and within taxa. $\delta^{13}\text{C}$ was frequently non-normal, in which case it was log or square root transformed to improve normality. Prediction 2, that $\delta^{15}\text{N}$ would be enriched in invertebrates captured in fields treated with manure compared to fields treated with artificial fertilizers or not fertilized, hedgerows, or fields planted with leguminous crops, was tested in two parts. First, I tested for differences in $\delta^{15}\text{N}$ within spiders and ground beetles captured in corn and hay fields known to have been manured since the end of the previous growing season (manured), and corn and hay fields known not to have been treated with manure since the end of the previous growing season (unmanured). Only corn and hay fields were

included in this analysis because soybean fields used in this study were not commonly manured. Ground beetles and spiders were the only taxa captured in high enough numbers to perform the analysis. Secondly I compared $\delta^{15}\text{N}$ among all different trapping locations, among taxa. Prediction 3, that despite the variation in $\delta^{15}\text{N}$ values of plants under different fertilizer regimens, $\delta^{15}\text{N}$ should still be enriched in invertebrates at higher trophic levels, was tested by comparing $\delta^{15}\text{N}$ among taxa, while controlling for trapping location. Prediction 4, that invertebrates captured in edges between hedgerows and crop fields will show intermediate $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values, due to movement across the boundary, was tested within spiders and ground beetles only, due to sample size restrictions. The analysis was repeated separately for corn, soybean and hay. In each case, I compared $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ for fields, edges and hedgerows bordered on at least one side by the relevant crop.

In all analyses, possible random effects were date, nest (nest at which sample was collected), field or hedgerow (where each field or section of hedgerow sampled was given a unique code, Fig. 3.2) and site. Site represents a single, continuous hedgerow and all the fields connected to it. Date was included to account for seasonal differences, for example $\delta^{13}\text{C}$ values of invertebrates might be expected to change as corn grows and increases in biomass. Nest was included to account both for temporal and spatial proximity of samples taken at each individual nest. Including field or hedgerow section as a random variable accounts for the spatial proximity of samples, as well as for variation in management of fields and hedgerows over time. Including site as a

random variable accounts for the spatial proximity of fields, and for shared management over time. Year was not included, as preliminary analysis showed no effect of year on isotope values. For each taxa, I first averaged data within taxa from trapping locations at the same nest (e.g. if 2 spiders were sampled from the same corn field at the same nest, the values were averaged for further analysis). Fewer data were available for hoppers and crickets, so when these taxa were analysed individually, data were averaged within each field or section of hedgerow. Due to the small sample size, caterpillars were only included in the analysis of prediction 1. The sample sizes for each taxon in each trapping location can be found in Appendix G. Degrees of freedom were calculated following Pinheiro & Bates (2000, p 91).

The first step in model building was to test for significance of random effects (Zuur et al. 2009). For each analysis I built models with all combinations of nested random effects (no random effects (i.e. linear model), nest, field, site, field nested in site, and nest nested in site, with and without date). I then tested for the most parsimonious model using AIC_c (Hurvich and Tsai 1989). AIC_c attempts to find the best expected Kullback-Leibler model for the current sample size, therefore incorporating a bias-variance trade off (Burnham and Anderson 2004). AIC_c is also appropriate for tapering effect sizes, which are nearly always a factor in model selection (Burnham and Anderson 2004). Model validity was checked using residual analysis. If visual inspection of residual graphs showed that assumptions were not being met, I either altered the variance structure, or considered the next best model, as appropriate. Altering the variance structure,

by including a variable to account for heterogeneity in variances among groups, allows me to relax the assumption of homogeneity of variances, by allowing the variance to change with levels of the dependent or random variables (Pineiro & Bates 2000). Once the most parsimonious model was selected, I tested for significance of the model and levels of fixed variables.

All analyses were carried out in R (R Development Core Team 2010), and results are presented as mean \pm standard error unless otherwise stated.

Results

Prediction 1: Carbon values

I predicted that $\delta^{13}\text{C}$ would be most enriched in invertebrates captured in corn, and lower in invertebrates captured in soybean, hedgerows and hay. $\delta^{13}\text{C}$ varied widely among and within invertebrate taxa (Fig. 3.3). However, there were significant differences in $\delta^{13}\text{C}$ among trapping locations, once taxa were taken into account (Table 3.1). As predicted, $\delta^{13}\text{C}$ was lower in invertebrates captured in hay and in hedgerows than those captured in corn (Fig. 3.4). However, contrary to my prediction, $\delta^{13}\text{C}$ values of invertebrates captured in corn were not significantly different from those captured in soybean. Invertebrates captured in corn had lower $\delta^{13}\text{C}$ than corn plants, while invertebrates captured in soybean, hedgerow and hayfields had higher $\delta^{13}\text{C}$ than dominant plants in these habitats (Table 3.2, raw data for these analyses can be found in Appendix AA).

Within taxa, spiders and ground beetles captured in corn and soybean had significantly higher $\delta^{13}\text{C}$ than those captured in hedgerows and hay (Table 3.3, Fig. 3.5a, 3.6a). Values of $\delta^{13}\text{C}$ of crickets captured in corn, soybean and hedgerows were very variable, and there was no significant difference among land uses (Fig. 3.7). Hoppers captured in hedgerows had significantly higher $\delta^{13}\text{C}$ than those captured in hay, but there was no significant difference among hoppers captured in corn and in other trapping locations (Fig. 3.7). $\delta^{13}\text{C}$ values of hoppers captured in corn were variable, but much lower than that of the corn plants (Table 3.2). The two lowest values of $\delta^{13}\text{C}$ of hoppers from corn came from two corn fields on the only organic farm included in this study. These fields were extremely weedy (personal observation), which might dilute the $\delta^{13}\text{C}$ of the corn plants. If these two values are excluded, $\delta^{13}\text{C}$ for hoppers captured in corn is significantly greater than that of hoppers captured in hedgerows or hayfields ($\delta^{13}\text{C} -24.05 \text{‰} \pm 2.19$, $F_{2,24}=9.24$, $p=0.001$).

Prediction 2: Nitrogen, fertilizer regimen and plant type.

I predicted that $\delta^{15}\text{N}$ would be enriched in invertebrates captured in fields treated with manure compared to fields treated with artificial fertilizers, unfertilized fields, fields planted with leguminous crops, or hedgerows. In agreement with my prediction, mean $\delta^{15}\text{N}$ of spiders from manured corn was higher than for spiders from unmanured corn (Table 3.4, manured corn: $\delta^{15}\text{N} = 11.01 \pm 0.23$, unmanured corn $\delta^{15}\text{N} = 9.65 \pm 0.52$). However, ground beetles did not show a significant difference in $\delta^{15}\text{N}$ values between manured and

unmanured corn fields (Table 3.5, manured corn: $\delta^{15}\text{N} = 8.80 \pm 0.30$, unmanured corn: $\delta^{15}\text{N} = 8.06 \pm 0.35$). In contrast with my prediction, neither spiders nor ground beetles showed significant differences in mean $\delta^{15}\text{N}$ between manured and unmanured hay fields (spiders in manured hay: $\delta^{15}\text{N} = 8.77 \pm 0.58$, unmanured hay: $\delta^{15}\text{N} = 8.62 \pm 0.77$; ground beetles in manured hay: $\delta^{15}\text{N} = 6.85 \pm 0.70$, unmanured hay: $\delta^{15}\text{N} = 6.36 \pm 0.31$).

When all invertebrates were analyzed together, the results partially agreed with my prediction, in that invertebrates captured in manured corn had significantly higher $\delta^{15}\text{N}$ than invertebrates captured in soybeans, hedgerows or manured or unmanured hay (taxon: $F_{3,209}=239.11$, $p<0.0001$, trapping location: $F_{5,209}=7.56$, $p<0.0001$, Fig. 3.8). However, there was no significant difference in $\delta^{15}\text{N}$ between invertebrates captured in manured and unmanured corn, or between invertebrates captured in manured and unmanured hay (Fig. 3.8).

Prediction 3: Nitrogen and trophic level

There were significant differences in $\delta^{15}\text{N}$ among taxa, once trapping locations were taken into account (Table 3.1). As predicted, the herbivorous hoppers and caterpillars had the lowest $\delta^{15}\text{N}$, while the spiders had the highest $\delta^{15}\text{N}$ (Fig. 3.9). This suggests that once capture location was controlled for, the different fertilizer regimens and plant types did not obscure the basic trophic level increase in $\delta^{15}\text{N}$.

Prediction 4: Values of $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ in edges

I predicted that $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values of invertebrates captured in edges would be intermediate between values of invertebrates captured in the adjacent fields and hedgerows. Both spiders and ground beetles in corn and in soybean showed a tendency for this to be true for carbon, but it was only significant for spiders in corn (Table 3.4, Fig. 3.5a). For spiders in soybean, $\delta^{13}\text{C}$ in the hedgerow was significantly lower than in the edge or the soybean, but there was no significant difference between edge and soybean. For ground beetles, there was a significant difference in $\delta^{13}\text{C}$ between corn and hedgerow, but not between corn and edge or between edge and hedgerow (Table 3.5, Fig. 3.6a).

For nitrogen, the pattern was not as clear; spiders in corn showed a significant difference in values among hedgerow, edge and field, but in soybean, field and edge values were not significantly different (Table 3.4, Fig. 3.5b). There was no significant difference in nitrogen values in ground beetles between hedgerow, edge and field for either corn or soybean (Table 3.5, Fig. 3.6b).

Carbon and nitrogen values were not significantly different between hay, hay-edge and hedgerows adjacent to hay for either taxon (Tables 3.4 and 3.5, Figs 3.5 and 3.6).

Discussion

This study is the first to report $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values of invertebrates captured in hedgerows, and one of the first to report $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values of

invertebrates captured in soybean fields. I showed that invertebrates captured in hay fields and hedgerows have significantly lower $\delta^{13}\text{C}$ than invertebrates captured in soybean and corn fields. $\delta^{15}\text{N}$ was important in distinguishing different taxa, due to the different trophic levels they occupy, as well as differentiating invertebrates captured in soybean fields, hayfields and hedgerows from manured corn fields. $\delta^{15}\text{N}$ was higher in spiders from manured corn fields than spiders from unmanured corn fields, but there was no difference for spiders in manured and unmanured hay fields, or for ground beetles from either field type.

Although I only collected invertebrates at 10 m and 20 m into crop fields, both $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values of spiders and ground beetles captured in corn were comparable with values obtained by Latendresse (2004), who sampled invertebrates ≤ 250 m from a corn-forest edge. In that study, field values of $\delta^{13}\text{C}$ stabilised within 5 m of the edge, so my values from invertebrates collected 10 m or 20 m into the field probably reflect $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ of field invertebrates accurately.

This study was carried out in the context of a larger study of insectivorous songbirds, and the level of taxonomy used here is fairly coarse, reflective of the level of selection of most insectivorous vertebrates in farmland. Because invertebrates were grouped at the level of orders or suborders, different species were combined to make of samples of each taxa in each cover type. Therefore, I cannot say whether the invertebrate community differs among cover types in this

study. Combining different species probably increases the variability of $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ in my samples. This is especially important for ground beetles, since foraging behaviour differs widely among species and between life-stage within the same species (reviewed in Goulet 2003). For example, Birkhofer et al (2010) show that different species of ground beetles, spiders and rove beetles can each vary in $\delta^{15}\text{N}$ by approximately 2‰ within wheat fields. The additional variation introduced by combining species at each taxonomic level makes my analysis conservative, since differences among cover types appear despite pooling of species. Future studies at finer taxonomic resolution may well clarify some issues raised in this study.

Carbon values

As I predicted, carbon values were lower in invertebrates captured in hedgerows and hay fields than those captured in corn. However, there was no significant difference in $\delta^{13}\text{C}$ between invertebrates captured in corn and soybean fields. I predicted that invertebrates from corn would have the highest $\delta^{13}\text{C}$ due to the high $\delta^{13}\text{C}$ value of corn. However, my results show that none of the invertebrates captured in corn and soybean fields fully reflected the $\delta^{13}\text{C}$ values of the dominant crops in the local area (Table 3.2, Figs 3.4-3.7). Invertebrates captured in soybean, hedgerows and hay fields tended to have higher $\delta^{13}\text{C}$ than the dominant plants in the respective trapping locations. In contrast, invertebrates captured in corn had much lower $\delta^{13}\text{C}$ than the corn plants. Isotope values tend to increase with trophic level, but these changes are

typically small for carbon (e.g. 0.5‰ per trophic level, McCutchan et al. 2003). This might explain the relatively small differences between plant values and invertebrate values found in hedgerows and hay fields, but cannot explain the large differences found in soybeans and in corn. The largest difference between plants and invertebrates was found for hoppers in corn (Table 3.2, Fig. 3.7), despite hoppers being herbivores. This strongly suggests the hoppers I captured in corn were not primarily feeding on corn plants, or had recently switched from another food source and had not yet incorporated the relatively high $\delta^{13}\text{C}$ value of corn into their tissues. In partial support of this, the two lowest $\delta^{13}\text{C}$ values of hoppers in corn came from two corn fields on an organic site. The weed prevalence in the organic fields would have provided hoppers feeding in corn fields abundant C3 carbon sources, which may have produced the low $\delta^{13}\text{C}$ values.

There are at least three reasons why invertebrates captured in crop fields did not fully reflect the $\delta^{13}\text{C}$ value of the local dominant crop plants; movement among different land uses, use of alternative food sources or retention of previous years $\delta^{13}\text{C}$ by overwintering invertebrates. First, as suggested by Haubert et al. (2009), low $\delta^{13}\text{C}$ values of invertebrates from corn fields and high values of invertebrates from soybean fields might reflect movement of invertebrates among different agricultural land uses. Some species of ground beetles move between crop fields during the growing season, while other species remain associated with a single field, or with hedgerows (Frampton & Çilgi 1995; Mauremooto et al. 1995; Thomas et al. 2001). Spiders disperse from permanent

habitats into crop fields early in the growing season (reviewed by Marc, Canard & Ysnel 1999), but little information is available about movements between crop fields during the growing season. However, spiders in farmland can recolonize areas of local extinction within a few weeks (Wick & Freier 2000; Holland, Winder & Perry 2000), so movements between crop fields can occur. The different sowing and growing patterns of different crops compared to hedgerows and hay fields make it probable that herbivores also move between land uses during the growing season. For example, McNabb, Halaj & Wise (2001) found that herbivores feeding on C3 plants (cucumbers, *Cucumis sativa*, variety “Spacemaster” and squash, *Cucumis pepo*, variety “Yellow Crook Neck”) had higher than expected $\delta^{13}\text{C}$ values and suggested that herbivores might be getting some carbon from C4 grasses outside their experimental plots. Laboratory studies have shown that when invertebrates are switched between diets with contrasting $\delta^{13}\text{C}$ values, it can take between several days to several weeks for the new carbon source to be fully reflected in the $\delta^{13}\text{C}$ value of the organism (Ostrom, Colunga-Garcia & Gage 1997; Gratton & Forbes 2006), so movement between different land uses could explain the observed similarity in $\delta^{13}\text{C}$ values of invertebrates captured in corn and soybean fields.

Second, the availability of alternative food sources within the crop fields could affect the $\delta^{13}\text{C}$ values of generalist foragers. For herbivores, this would take the form of C3 weeds (e.g. alfalfa *Medicago sativa*, clover *Trifolium* spp, dandelions *Taraxacum* spp, timothy *Phleum pratense*) and C4 weeds (primarily foxtail grasses *Setaria* spp and sedges), both of which were found in the study

fields (personal observation). For higher trophic levels, alternative food sources include detritivores living in the soil. Both spiders and ground beetles are known to feed on detritivores in agroecosystems (Wise, Snyder & Tuntibunpakul 1999; Halaj & Wise 2002). Crop residues take time to break down in the soil, so detritivores may have $\delta^{13}\text{C}$ values that reflect the previous crop, not the current one. For example, Albers, Schaefer & Scheu (2006) found that 18 months after corn was planted in a field that had previously been planted only in C3 crops, small detritivores had incorporated < 40% carbon from corn. The delay in the appearance of a new carbon value in the detritivore food web, may explain why carbon values of ground beetles and spiders were actually higher in soybean fields than in corn fields (Fig. 3.5a, 3.6a).

One possible method for testing these two hypotheses is to analyse invertebrates in separate pieces, rather than as whole organisms. Different parts of an organism assimilate carbon isotopes at different speeds; for example, after a diet switch, the reproductive and fatty tissues of beetles reflect the new carbon source more quickly than do the hind wings (Gratton & Forbes 2006). Therefore an invertebrate that has recently moved between land uses will have different carbon values in different tissues, whereas an invertebrate that has a constant diet, even if it is composed of alternative food sources, will have similar carbon values in different tissues. Such further analysis was beyond the scope of this study.

A third explanation for the dissimilarity between invertebrate $\delta^{13}\text{C}$ values and the $\delta^{13}\text{C}$ values of the dominant plants in capture locations, is carryover of $\delta^{13}\text{C}$ values in overwintering invertebrates. Any invertebrate that overwinters, either as an adult, larva or egg, might reflect the $\delta^{13}\text{C}$ and/or $\delta^{15}\text{N}$ value of the habitat it (or its parent) used in the previous year. For example, the larval food source of adult moths can be determined using $\delta^{13}\text{C}$ (Gould et al. 2002). If after emerging in the spring, an invertebrate is captured in a land cover type with a different dominant plant cover than that used for foraging prior to overwintering, it would show a $\delta^{13}\text{C}$ different from that expected based on its capture location. Given the strong corn-soybean rotation in the study area, this may explain why I could not differentiate between invertebrates captured in corn and soybean using $\delta^{13}\text{C}$.

Invertebrates captured in hedgerows and hayfields showed $\delta^{13}\text{C}$ values close to that expected for invertebrates using C3 plants as the primary food source. This may reflect either that invertebrates in these less disturbed habitats move around less than those in crop fields (either within or between seasons), or the overall dominance of C3 plants in the area.

In my study, the main carbon sources provided by corn, and by various C3 plants, differed considerably in $\delta^{13}\text{C}$ (Table 3.2). However, $\delta^{13}\text{C}$ ranges of C3 and C4 plants overlap (or very nearly so, Rounick & Winterbourn 1986; O'Leary 1988), so these differences may not be apparent in all systems. This method of

using $\delta^{13}\text{C}$ to identify carbon source, and therefore habitat of invertebrates (or other consumers) will only work where large differences in carbon sources exist.

Nitrogen, fertilizer regime and plant type

I predicted that nitrogen values would be higher in manured fields, compared to all other land uses. $\delta^{15}\text{N}$ values were higher in invertebrates from manured corn fields than in invertebrates from soybean fields, hedgerows or hay fields, and for spiders in manured and unmanured corn fields. However, this was not true for ground beetles in corn fields or for spiders or ground beetles in hay fields. Powell et al. (2005) found that in the year of manure application, corn plants took up 14-16% of available ^{15}N from manure, and 4-8% in the year after application. This additional uptake in years when corn or hay is not treated with manure may make detecting changes in fertilization using $\delta^{15}\text{N}$ more difficult. Spiders were less variable in $\delta^{15}\text{N}$ than ground beetles (Figs 3.3, 3.5, 3.6), probably because spiders are predatory, whereas ground beetles vary in trophic level (Lovei and Sunderland 1996; Bennett and Hobson 2009). This may make changes in fertilization regimen easier to detect in spiders than in ground beetles.

Nitrogen and trophic level

As I predicted, $\delta^{15}\text{N}$ did reflect trophic levels of the different taxa sampled, with hoppers and caterpillars having the lowest $\delta^{15}\text{N}$, and spiders the highest $\delta^{15}\text{N}$, while the omnivorous crickets and ground beetles had intermediate values. Based on a trophic level $\delta^{15}\text{N}$ increase of 2.3‰ (McCutchan et al. 2003), spiders

were 3 trophic levels higher than hoppers. Other agricultural food web studies have compared trophic levels of spiders and detritivorous Collembola, and found that spiders are between 2 and 3 trophic levels above Collembola (McNabb, Halaj & Wise 2001; Wise, Moldenhauer & Halaj 2006). Ground beetles were less than 1 trophic level lower than spiders, suggesting they are largely predatory in this system.

Values of $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ in edges

My prediction that $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values of invertebrates captured in edges would be intermediate between values of invertebrates captured in the adjacent fields and hedgerows was only partially supported. Although the prediction was supported for $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values of spiders in corn, it was not supported for spiders in soybean, or for ground beetles in either corn or soybean.

Intermediate values of $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ in edges suggest a movement of invertebrates between hedgerow and field, but not necessarily between fields. If there was a direct movement of invertebrates between corn and soybean fields, one would expect that values of $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ for invertebrates in soybean edges would be higher (more similar to corn fields) than the values of invertebrates in soybean fields. This is shown only for $\delta^{15}\text{N}$ values of spiders (Fig. 3.5b). Therefore the edge values reported here do not strongly suggest direct movement of ground beetles and spiders between crop fields.

$\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ were not significantly different between hay, hay-edge and hedgerows adjacent to hay for either taxon. This is not surprising as both land uses were dominated by C3 plants, and there was no significant difference in $\delta^{15}\text{N}$ values between manured and unmanured hay fields.

I have shown that $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ can be used to distinguish between invertebrates captured in different agricultural land uses, even at a small spatial scale. In addition, $\delta^{15}\text{N}$ can distinguish between invertebrates at different trophic levels. The ability to distinguish between invertebrates from different agricultural land cover types will benefit research into food webs and land use change in farmland. However, further studies at finer taxonomic levels together with laboratory studies into discrimination factors and foraging preferences will be necessary to further investigate how changes in crop type are reflected in the isotope values of invertebrates.

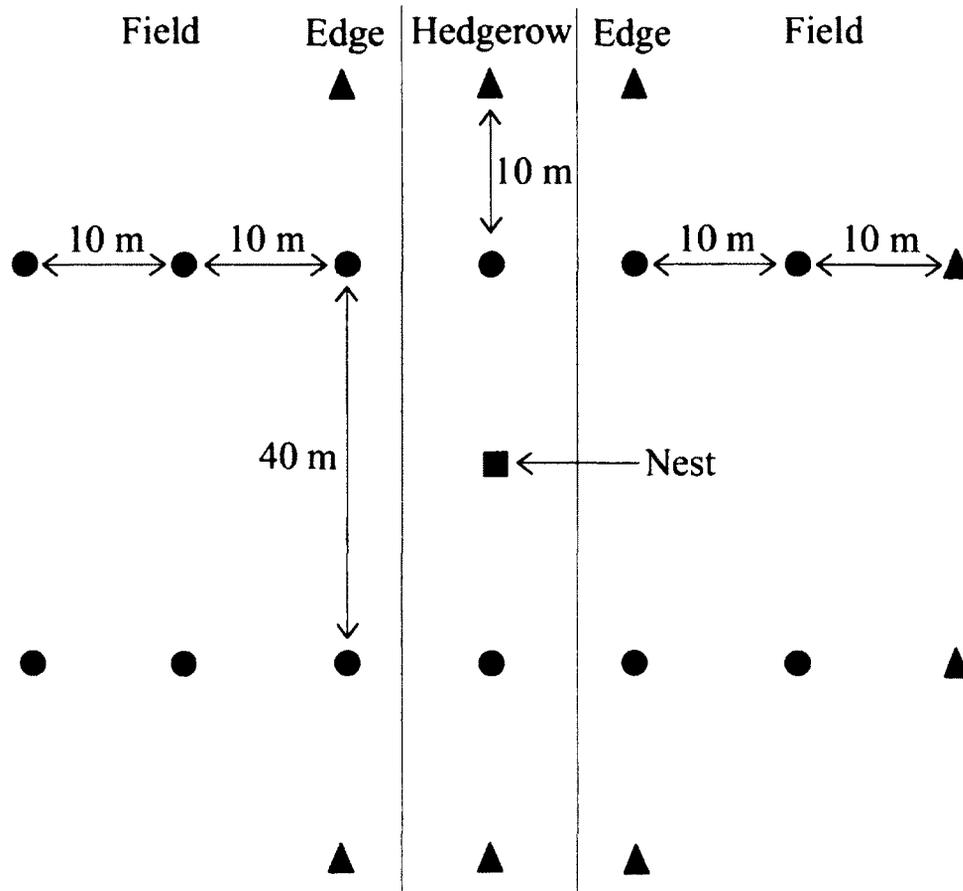


Figure 3.1: Pitfall trap layout, showing a single nest site with 4 hedgerow traps, 8 edge traps (at the borderline between hedgerow and field) and 8 traps in the adjacent fields. Dots represent pitfall trap locations used in all years (2006-2008), triangles represent pitfall trap locations used only in 2007 and 2008.

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Figure 3.2: Aerial photograph of site H, showing how the fields and hedgerow were divided up for analysis. Each site consisted of one hedgerow and all the adjoining fields. Hedgerows were divided into sections, so that each section adjoined only two fields. Reprinted with permission from © 2011 Ecoscience.

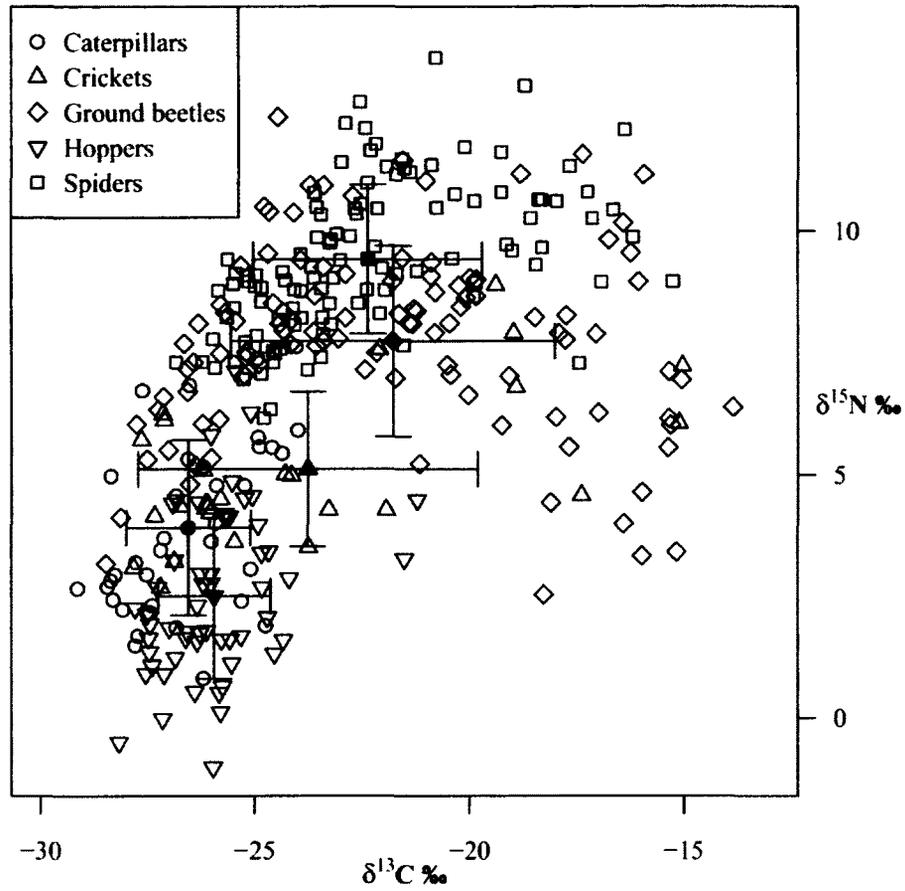


Figure 3.3: $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values for caterpillars (circles), crickets (triangles), ground beetles (diamonds), hoppers (downward triangles), and spiders (squares). Filled symbols show means \pm 1 SD. Reprinted with permission from © 2011 Ecoscience.

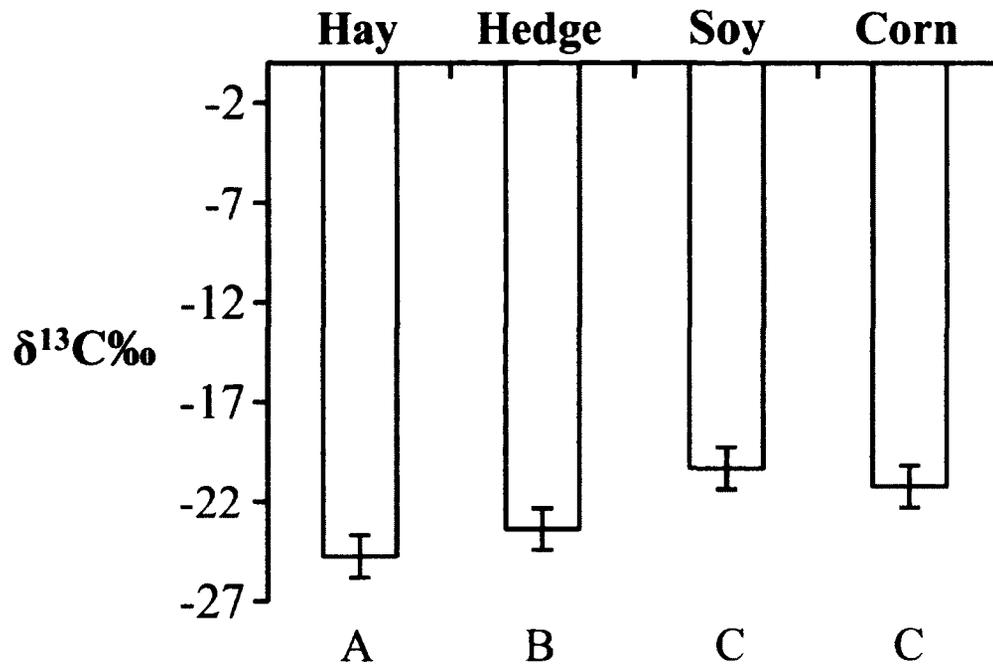


Figure 3.4: Mean values of $\delta^{13}\text{C}$ for all trap locations, once taxa is controlled (± 1 SE). Means with the same letter are not significantly different ($p < 0.05$).

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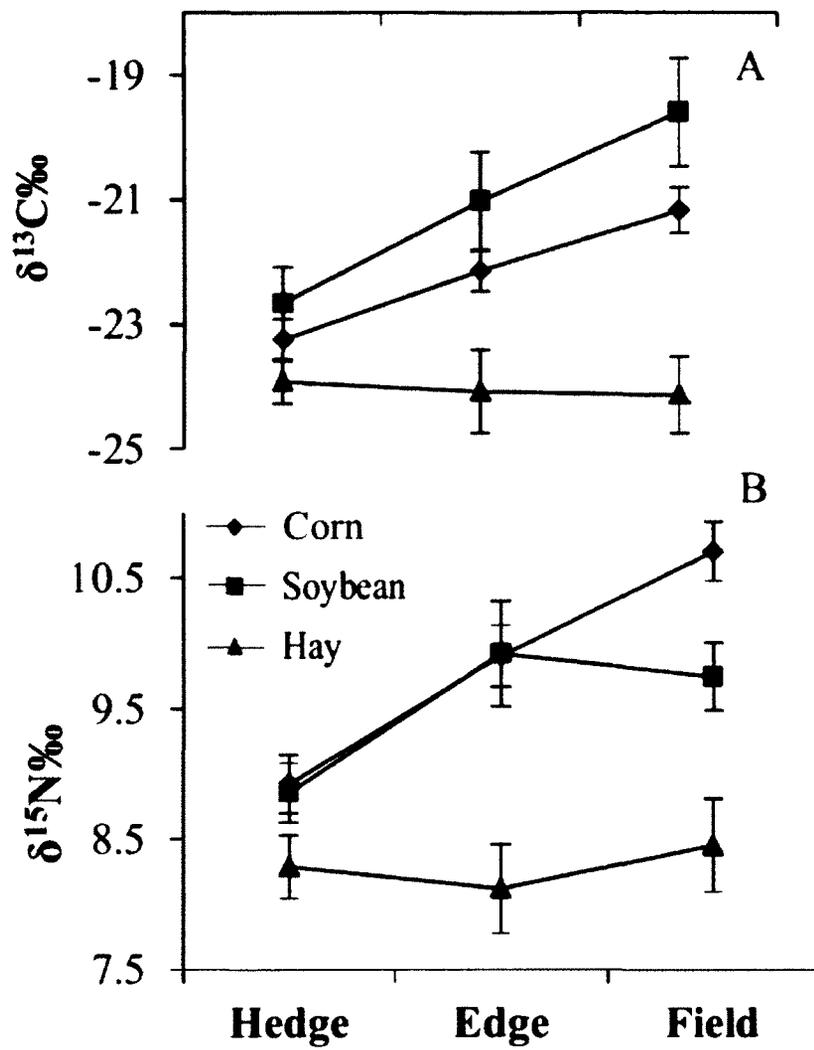


Figure 3.5: Mean a) $\delta^{13}\text{C}$ values (± 1 SE) and b) $\delta^{15}\text{N}$ values (± 1 SE) for spiders captured in different trap locations. Hedgerow values include only hedgerows adjacent to the specified crop. Reprinted with permission from © 2011 Ecoscience.

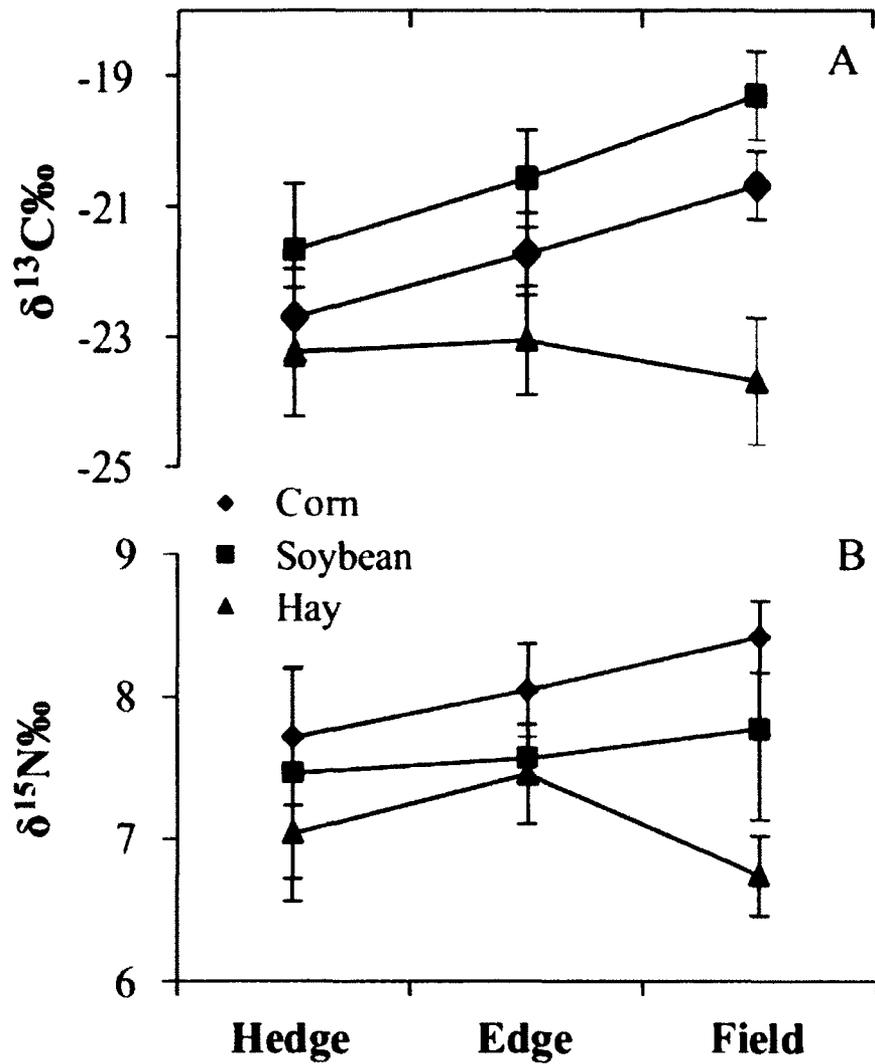


Figure 3.6: Mean a) $\delta^{13}\text{C}$ values (± 1 SE) and b) $\delta^{15}\text{N}$ values (± 1 SE) for ground beetles captured in different trap locations. Hedgerow values include only hedgerows adjacent to the specified crop. Reprinted with permission from © 2011 Ecoscience.

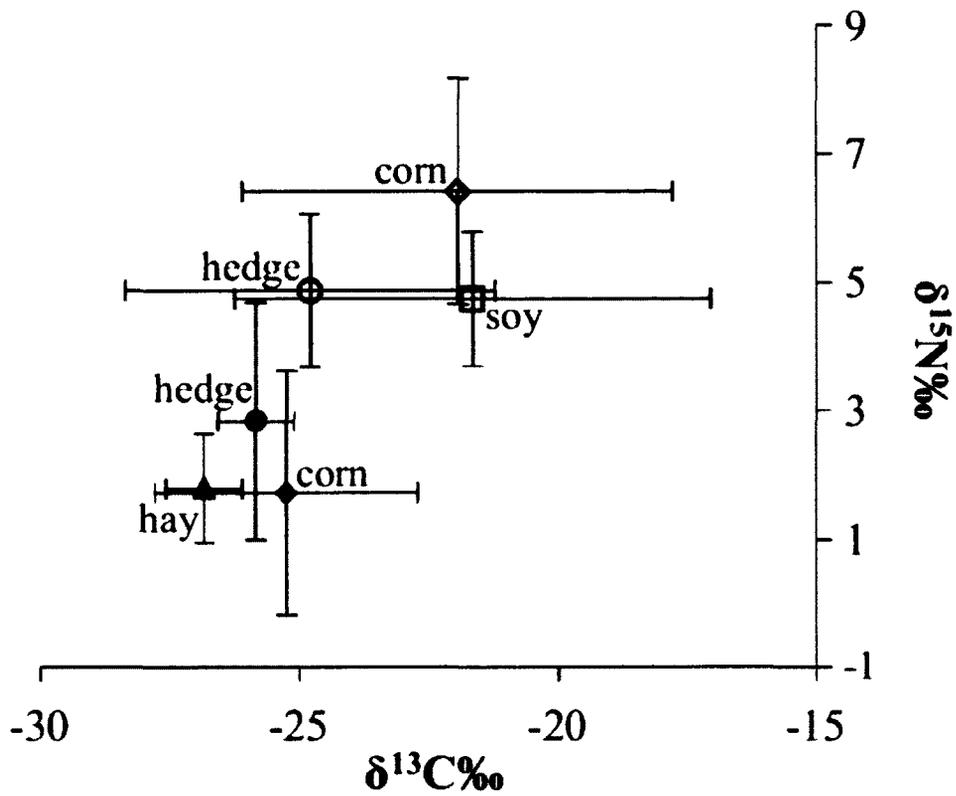


Figure 3.7: Mean $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values (± 1 SD) for crickets (open symbols) and hoppers (filled symbols) captured in corn (diamond), hay (triangle) and hedgerow (circle) and soybean (square). Reprinted with permission from © 2011 Ecoscience.

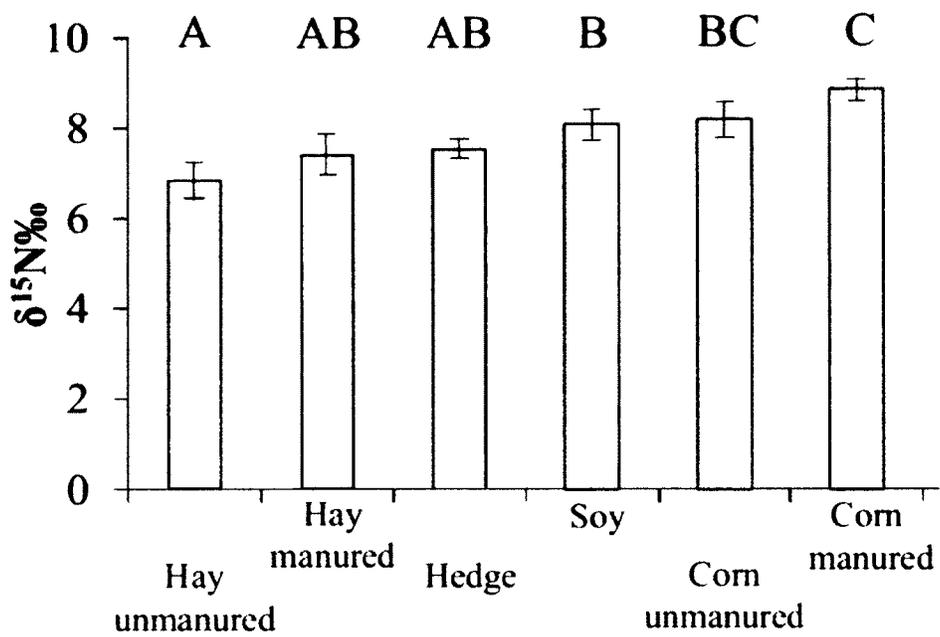


Figure 3.8: Mean $\delta^{15}\text{N}$ values for all trap locations, once taxon is controlled (± 1 SE). Means with the same letter are not significantly different ($p < 0.05$).

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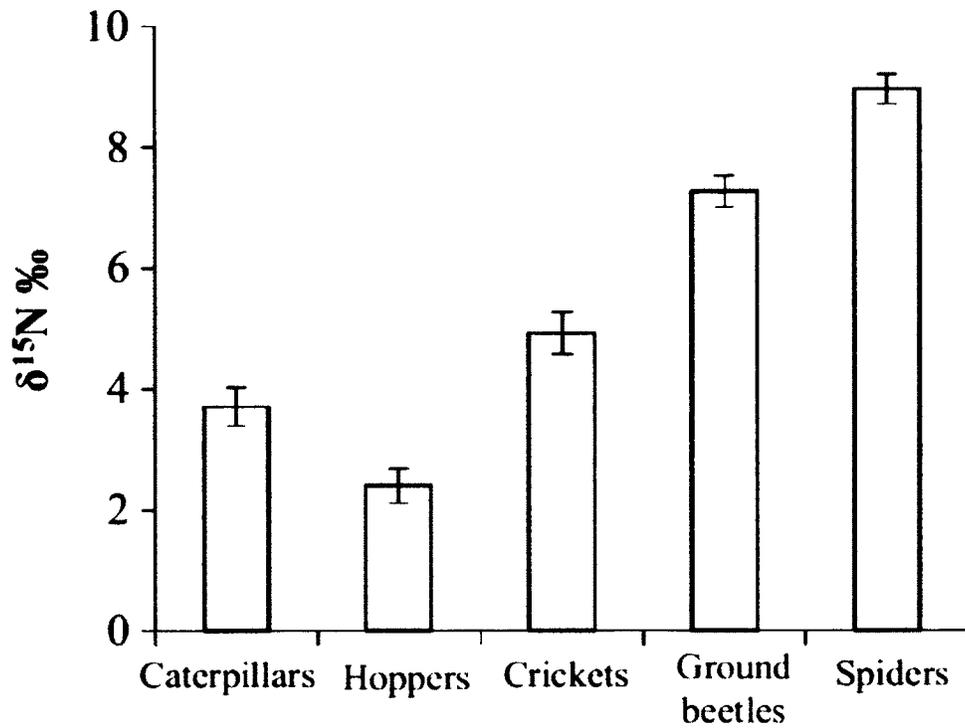


Figure 3.9: Mean $\delta^{15}\text{N}$ values for all taxa, once trap location is controlled (± 1 SE). Values were significantly different among all taxa ($p < 0.05$). Reprinted with permission from © 2011 Ecoscience.

Table 3.1: Results of mixed effects models showing differences in carbon and nitrogen stable isotopes among taxa (spiders, ground beetles, crickets, hoppers and caterpillars) captured in different trapping locations (corn, soybean, hedgerow, hay). Reprinted with permission from © 2011 Ecoscience.

Dependent variable	Random effects	Stratification variable ¹	Fixed effects	F for fixed effects	df for fixed effects	p for fixed effects
$\delta^{13}\text{C}$	date, field	species	taxa	54.33	271	<0.0001
			trapping location	26.71	271	<0.0001
$\delta^{15}\text{N}$	date, nest		taxa	222.58	267	<0.0001
			trapping location	14.14	267	<0.0001

¹ Models including a variance function allow for heterogeneity among levels of the stratification variable.

Table 3.2: Ranges of $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ of plants from different land covers adjacent to sparrow nests. Reprinted with permission from © 2011 Ecoscience.

Land cover	Min $\delta^{13}\text{C}$	Max $\delta^{13}\text{C}$	Min $\delta^{15}\text{N}$	Max $\delta^{15}\text{N}$	n	Number of species
corn unmanured	-13.18	-11.66	1.01	17.1	13	1
corn manured	-13.53	-11.67	1.82	19.88	21	1
hay unmanured	-30.57	-28.23	-0.86	2.54	11	6
hay c4 ¹ unmanured	-11.2	-10.82	7.8	10.25	2	2
hay manured	-29.75	-27.59	-1.31	13.31	15	6
hedge	-32.08	-25.62	-1.46	7.55	103	26
hedge c4 ¹	-12.39	-11.23	1.07	3.96	5	4
soy	-30.64	-27.75	-0.96	3.02	16	1

¹ C4 plants collected from within C3 dominated land cover types.

Table 3.3: Results of models testing differences in carbon stable isotopes among trapping locations for 4 different taxa of invertebrates. Reprinted with permission from © 2011 Ecoscience.

Taxa	Model Type ¹	Random Effects	Weights	Fixed Effects	F	df	p
Spiders	lmem	site, date		location (corn, soy, hay, hedge)	15.16	94	<0.0001
Ground beetles	lmem	field	location	location (corn, soy, hay, hedge)	5.74	56	0.002
Crickets	lmem	site		location (corn, soy, hedgerow)	3.42	10	0.07
Hoppers	gls		location	location (corn, hay, hedgerow)	4.20	26	0.03

¹ Model types were linear mixed effects models (lmem), and generalized least squares (gls).

Table 3.4: Results of models testing differences in carbon and nitrogen stable isotopes among trapping locations for spiders. Reprinted with permission from © 2011 Ecoscience.

Dependent variable	Model Type ¹	Random Effects	Weights	Fixed Effects	F	df	p
$\delta^{15}\text{N}$	lm			manured corn, unmanured corn	7.09	28	0.01
$\delta^{15}\text{N}$	lm			manured hay, unmanured hay	0.02	8	0.89
$\delta^{13}\text{C}$	lmem	nest	site	corn, corn edge, hedgerow	16.26	56	<0.0001
$\delta^{15}\text{N}$	lmem	site		corn, corn edge, hedgerow	14.59	88	<0.0001
$\delta^{13}\text{C}$	lmem	site		soy, soy edge, hedgerow	6.76	38	0.003
$\delta^{15}\text{N}$	lmem	field		soy, soy edge, hedgerow	4.23	28	0.02
$\delta^{13}\text{C}$	lm			hay, hay edge, hedgerow	0.28	45	0.75
$\delta^{15}\text{N}$	lm			hay, hay edge, hedgerow	0.27	45	0.77

¹ Model types were linear mixed effects models (lmem) and linear models (lm).

Table 3.5: Results of models testing differences in carbon and nitrogen stable isotopes among trapping locations for ground beetles. Reprinted with permission from © 2011 Ecoscience.

Dependent variable	Model type ¹	Random effects	Fixed effects	F	df	p
$\delta^{15}\text{N}$	lm		manured corn, unmanured corn	1.98	33	0.17
$\delta^{15}\text{N}$	lm		manured hay, unmanured hay	0.41	10	0.54
$\delta^{13}\text{C}$	lmem	site, date	corn, corn edge, hedgerow	6.04	86	0.004
$\delta^{15}\text{N}$	lmem	site, nest	corn, corn edge, hedgerow	1.89	25	0.17
$\delta^{13}\text{C}$	lm		soy, soy edge, hedgerow	1.96	39	0.15
$\delta^{15}\text{N}$	lmem	site, nest	soy, soy edge, hedgerow	0.25	21	0.78
$\delta^{13}\text{C}$	lmem	field	hay, hay edge, hedgerow	0.44	27	0.65
$\delta^{15}\text{N}$	lmem	site	hay, hay edge, hedgerow	0.98	39	0.38

¹ Model types were linear mixed effects models (lmem) and linear models (lm).

Chapter Four

Foraging habitat and diet of Song Sparrows (*Melospiza melodia*) nesting in farmland: A stable isotope approach.

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Abstract

Agricultural intensification has been linked to declines in farmland and grassland bird populations in Europe and in North America. One factor thought to be behind these declines is reduced invertebrate food abundance in the breeding season, leading to reduced reproductive success. However, little is known about foraging habitat or diet of farmland birds in North America. I used stable isotopes to study foraging habitat and diet of Song Sparrows (*Melospiza melodia*), a common hedgerow-breeding bird, by collecting claw clippings of Song Sparrow nestlings from farms in eastern Ontario, Canada. Mean values of $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ were compared between claws and invertebrate food sources from adjacent land cover types, using a Bayesian mixing model. The portion of nestling diet that came from semi-natural and forage habitats was higher than expected, based on the proportion of semi-natural and forage cover at the nest. This was particularly evident at nests with low semi-natural cover, where hedgerows were the only non-crop habitat. The most important food sources for Song Sparrow nestlings were hoppers (Auchenorrhyncha) and caterpillars (Lepidoptera). Management actions to increase invertebrate abundance in hedgerows may benefit Song Sparrows and other farmland birds with minimal impact on crop yield.

Introduction

Over the past half century, agricultural practices have intensified through the use of chemical pesticides and fertilizers, increased field sizes, farm specialization, destruction of edge and boundary features, and drainage of land among others (reviewed by Tschardt et al. 2005). This has increased agricultural productivity (Robinson and Sutherland 2002; Green et al. 2005), but has also led to declines in biodiversity on farmland (Benton et al. 2002; Robinson and Sutherland 2002; Benton et al. 2003). Consequently, farmland and grassland bird populations have declined in Europe and in North America (Fuller et al. 1995; Siriwardena et al. 1998; Donald et al. 2001; Murphy 2003; Sauer et al. 2011).

One of the factors thought to contribute to farmland bird declines is a reduction in invertebrate food abundance, which is important for feeding nestlings (Campbell and Cooke 1997; Newton 2004). Long-term use of pesticides and reduction in diversity of land uses within farmed areas (e.g. simplification of crop rotation, reduction in mixed farming, loss of semi-natural habitat within farmland, Wilson et al. 1999; Robinson and Sutherland 2002) has reduced invertebrate abundance in farmland (Aebischer 1990; Campbell and Cooke 1997; Wilson et al. 1999; Robinson and Sutherland 2002; Newton 2004). Farmland birds breeding at nest sites with lower nestling food abundance or higher insecticide applications can show reductions in nest success including reduced fledgling success and reduced nest survival (Brickle et al. 2000; Boatman et al. 2004;

Britschgi et al. 2006). Reduced food abundance has been linked to population declines in Grey Partridges (*Perdix perdix* L., 1758) (Potts 1986) and is suspected to be a factor in declines in a number of other species (Campbell and Cooke 1997; Boatman et al. 2004; Britschgi et al. 2006).

The detailed understanding of how farmland birds in Europe (particularly in the UK) are affected by reductions in invertebrate food abundance are underpinned by extensive knowledge of both the diet and the foraging habitat of these birds (e.g. Hill 1985; Stoate et al. 1998; Wilson et al. 1999; Moreby and Stoate 2001; Morris et al. 2001; Southwood and Cross 2002; Holland et al. 2006). For example, diet studies of Grey Partridge juveniles were critical in demonstrating the link between herbicide use and population declines (Green 1984; Potts 1986). Foraging habitat and diet of farmland birds have received much less attention in North America than in Europe. North American farmland breeding birds are known to use crop fields, hay fields and boundary features, such as hedgerows, for foraging (Best 1990; Boutin et al. 1999; Puckett et al. 2009). Boutin et al. (1999) surveyed birds in corn, soybean, apple orchards and vineyards in southern Ontario and found that of 14 species observed within field centers and in edges adjacent to crop fields, most species were observed in edges more than expected based on habitat availability. The only detailed North American study of foraging habitat use within crop fields I am aware of was carried out by Rodenhouse and Best (1994). They showed that Vesper Sparrows (*Pooecetes gramineus*) nesting in corn and soybean fields forage most often

within 1 m of weedy or unplanted areas, and prefer fields with crop residue over bare fields, presumably due to increased food abundance.

I studied foraging habitat and diet in Song Sparrows (*Melospiza melodia*) breeding in hedgerows and fencerows using stable isotope analysis. Unlike foraging observations, which provide information only on habitat use, or ligatures or fecal analysis, which provide a snapshot of diet items, and may be biased against certain food types, stable isotope analysis provides information on food sources that have been incorporated into an animal's tissues over time (Hobson and Clark 1992; Hobson and Bairlein 2003; Pearson et al. 2003; Bearhop et al. 2003). Stable isotope analysis is suited to studying foraging habitat and diet where habitats and food sources have different isotopic signatures (Gannes et al. 1997; Phillips and Gregg 2001). I have previously shown that carbon and nitrogen stable isotopes can be used to distinguish between invertebrates captured in crop fields and those captured in hedgerows and hay fields in my study area (Chapter 3). In this study I 1) test the null hypothesis that nestling food is obtained from crop fields and from semi-natural and forage habitats in proportion to local habitat availability and 2) determine which invertebrate taxa are most important in the diet of Song Sparrow nestlings in hedgerow nests. Stable isotope analysis has been infrequently used in passerine nestlings, so I also provide evidence to validate this methodology.

Materials and methods

Study sites

This study was carried out at 12 sites on dairy and arable farms in eastern Ontario, Canada, where each site consisted of a continuous hedgerow and the adjoining fields. Farms were located between the towns of Navan (45°25'15.68"N, 75°25'36.28"W) and Embrun (45°16'26.29"N, 75°16'30.04"W), south east of Ottawa. The dominant crop rotation is field corn (*Zea mays* L.) and soybean (*Glycine max* L.), but hay and wheat (genus *Triticum* L.) are also common. Hedgerows varied from grassy strips with occasional shrubs, to fully treed hedgerows. Hedgerows were selected to avoid deep ditches and areas of permanent water.

Study species

Song Sparrows were selected for this study, as they are widespread and abundant in farmland across North America (e.g. Best et al. 1995; Freemark and Kirk 2001; Jobin et al. 2001; Sykes and Hannon 2001). Their use of field edges for foraging, together with their relatively high abundance in these areas, led Boutin et al. (1999) to rate Song Sparrows as high risk for pesticide exposure. Song Sparrow populations in the study region (lower Great Lakes/St. Lawrence plain) have shown a long-term pattern of decline (1966-2009: -0.6 % change/year, 95% CI: -0.9% to -0.4%, n=167, Sauer et al. 2011).

During the breeding season Song Sparrows eat primarily invertebrates, along with some seeds and fruit (Arcese et al. 2002). Song Sparrows forage primarily on the ground, although I also observed them gleaning invertebrates from leaves and bark, and occasionally hawking for flying insects (Smith 1978, J. Girard, personal observation). Based on ligature samples of nestling food from three Song Sparrow nests, I found that nestling food is highly variable and includes hoppers (Auchenorrhyncha, 22 individuals, hoppers includes leafhoppers, plant hoppers, froghoppers, etc.), true spiders (Araneae, 8 individuals), caterpillars (Lepidoptera, 2 individuals), unidentified larvae, (2 individuals), beetles (Coleoptera, 2 individuals), bees or wasps (Hymenoptera, 1 individual), true bugs (Heteroptera, 1 individual) and flies (Diptera, 1 individual) (J. Girard, unpublished data). Due to the small number of samples obtained and the high taxonomic variability of the invertebrates in the samples, I did not use these data to create a prior distribution for the Bayesian mixing model described below.

Nest searching

Nest searching was carried out in 2006-2008 between mid-May and early August. Nests were found primarily by flushing females off the nest, as well as by observing adults and by chance. Once a nest was found, it was monitored every three to four days on average, until success or failure of the nesting attempt was established.

Nestling samples

To obtain $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values from nestlings, I collected claw samples from 81 nestlings in 28 nests in 2006 and 2007. Samples were collected by clipping approximately 2 mm from the middle claw of each foot, when the nestling was five or six days old. Claw samples were cleaned with methanol, cut into very small pieces and approximately 0.3-0.4 mg was weighed into a tin capsule for stable isotope analysis.

Animal care and sampling protocol was approved by the National Wildlife Research Centre and Ontario region Animal Care Committee, Environment Canada (project number 06X06P50).

Invertebrate sampling

Invertebrate samples were collected near 59 nests, including the 28 nests from which claws were collected. Sampling was carried out in the hedgerow and in adjoining fields, within a 35 m radius of each nest, which approximates the average territory size of Song Sparrows in these study sites (J. Girard, unpublished data). Samples were collected using pitfall traps, a D-Vac suction sampler (D-Vac Company, Ventura, CA, USA, model 122, Dietrick et al. 1959) and by hand (Fig. 4.1). See Chapter 3 for a full description of invertebrate sampling methods. Samples were always collected between hatching and fledging of the nestlings, and generally when the nestlings were between five and eight days of age. Note that many of the invertebrate samples analysed here were previously analysed in Chapter 3 in a different context. This does not include data on invertebrates captured in wheat fields, or data on sap beetles and

beetle larvae which were not presented previously. In addition, previously used data on invertebrates captured in field edges are not included here.

In the lab, invertebrates were sorted and identified to family (beetles) or to order or sub-order (all other invertebrates). Based on abundance in trapping samples, and presence in ligature samples (see above) I selected seven taxa for further analysis: caterpillars (Lepidoptera larvae, $n=39$ samples used in stable isotope analysis), crickets (Ensifera, $n=35$), adult ground beetles (Carabidae $n=129$), beetle larvae (Coleoptera larvae, $n=10$), hoppers (Auchenorrhyncha, $n=80$), sap beetles (Nitidulidae, $n=65$) and true spiders (Araneae, $n=158$). Invertebrates were either freeze-dried (2006) or oven dried at 60°C for ≥ 48 hours until dry (2007, 2008), then crushed into a fine powder, and approximately 1 mg was weighed into a tin capsule for isotope analysis.

Isotope analysis

Samples and standards were loaded into an elemental analyser (Vario EL III, Elementar Analysen systeme, Hanau, Germany), which was interfaced to an isotope ratio mass spectrometer (Conflo II, Delta XP Plus Advantage, ThermoFinnigan, Bremen, Germany). The data are reported in delta notation, defined as $\delta = ((R_{\text{sample}} - R_{\text{standard}}) / R_{\text{standard}}) * 1000$ where R is the ratio of the abundance of the heavy to the light isotope. $\delta^{15}\text{N}$ is reported as ‰ vs. AIR, and $\delta^{13}\text{C}$ is reported as ‰ vs. V-PDB and normalized to internal standards calibrated to international standards. Analytical precision (SD for repeated measurement of internal standard C-55 within runs) was better than 0.2 ‰ for invertebrate $\delta^{13}\text{C}$

and $\delta^{15}\text{N}$, and better than 0.45 ‰ for claw $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$. 91 out of 524 invertebrate samples were duplicated (mean difference between sample and duplicate \pm SD, $\delta^{13}\text{C}$ 0.24‰ \pm 0.23‰, $\delta^{15}\text{N}$ 0.22‰ \pm 0.21‰), and 11 of 86 claw samples were duplicated ($\delta^{13}\text{C}$ 0.11‰ \pm 0.08‰ $\delta^{15}\text{N}$ 0.23‰ \pm 0.21‰). All samples were analysed at the Hatch Laboratories at the University of Ottawa.

Land cover

I used 2008 aerial photographs of eastern Ontario to digitize landscape structure around each site (DRAPE 2010). Land uses within 100 m of each nest were added to the digitized map in two categories; crop cover (corn, wheat and soy) and semi-natural/forage cover (alfalfa, hay, pasture, hedgerow, riparian and verge, hereafter semi-natural cover), to reflect expected differences in stable isotope values of invertebrates. Maps were then used to quantify land uses at 35 m and 100 m radii from each nest. The 35 m radius reflects the mean territory radius of Song Sparrows nesting on these study sites (J. Girard, unpublished data). The larger scale accounts for long-distance foraging trips regularly made by Song Sparrows (J. Girard, personal observation). Crop cover and semi-natural cover were strongly correlated at each scale (100m: $\rho = -0.95$, $p < 0.001$, 35m: $\rho = -0.99$, $p < 0.001$), and crop cover was highly correlated between scales ($\rho = 0.85$, $p < 0.001$).

Statistical analyses

I used a Bayesian mixing model to estimate the contributions of the different invertebrate food sources to the diet of the nestlings (siar package in R, Parnell et al. 2010). This model incorporates input variability, and outputs a probability density function (PDF) for each food source, representing the probable contribution of each food source to the consumer tissue (Moore and Semmens 2008; Parnell et al. 2010). Since all the food sources in this study were invertebrates with similar C:N (Appendix H), I did not correct for differing C:N among food sources (Post et al. 2007).

A discrimination factor was used to account for changes in isotopic ratio between food source and consumer. Discrimination factors can vary with species, tissue, and diet, and using inaccurate discrimination factors can strongly affect the outcomes of mixing models (e.g. Cherel et al. 2005; Caut et al. 2009; Hussey et al. 2010). I could not find discrimination factors for claws of any bird species in the literature, so instead I used a discrimination factor for tail feathers of juvenile (23-26 day old) Song Sparrows (mean \pm 1 SD, $\delta^{13}\text{C}$: $0.2\text{‰} \pm 0.58$, $\delta^{15}\text{N}$: $2.74\text{‰} \pm 0.31$, Kempster et al. 2007). Tail feathers emerge at approximately 6 days in Song Sparrows (Jongsomjit et al. 2007), so the feathers incorporate material produced at the age at which claw samples were collected. Discrimination factors are not affected by growth in young Song Sparrows (Kempster et al. 2007), so I made no correction for any difference in growth rate between 6 day old chicks and the juveniles from which the discrimination factor was obtained.

To test my null hypothesis that nestling diet is obtained from local land cover types in proportion to their availability, I created a mixing model in which the food sources represented invertebrates captured in crop fields (corn, soybean and wheat) and those captured in hedgerows and hayfields (hereafter semi-natural habitats). Since not all invertebrate taxa were captured near every nest, and different numbers of nests were sampled in different fields, invertebrate $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ were first averaged by field or section of hedgerow and then by cover type. All sap beetles were included as a single food source, as they have relatively high values of $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$, which do not differ between crop fields and hedgerow and hayfields, probably due to their specialisation as crop pests (Nout and Bartelt 1998; Dowd 2000). The inputs to the mixing model were the mean claw $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ value from each nest, the mean and standard deviation of each food source, and the mean and standard deviation of the discrimination factor.

The mixing model was run for each nest individually, and then the PDF for crop sources and for semi-natural sources were summed to give a distribution of proportion of diet from crop sources and from semi-natural sources at each nest. A paired t-test was used to compare the median proportion of diet from crop sources and from semi-natural sources at each nest. A paired Wilcoxon test was used to compare the median proportion of the diet from semi-natural habitats with the proportion of semi-natural cover at each nest. Finally, I used a linear regression to test whether the median proportion of Song Sparrow nest diet from semi-natural habitats increased with the proportion of semi-natural land cover at

each nest, and used a t-test to test whether this relationship differed from the 1:1 relationship predicted if diet is obtained from semi-natural habitats in proportion to their availability close to the nest.

To determine which invertebrate taxa are most important in the diet of nestling Song Sparrows, I ran a second mixing model using the seven invertebrate groups as potential food sources. I then used the median value of the PDF for each food source at each nest, and averaged across nests to compare the importance of different food sources to nestling diet. A Kruskal-Wallis test followed by a Nemenyi-Damico-Wolfe-Dunn post-hoc test (Hollander and Wolfe 1999) was used to test for significant differences among food sources.

If food sources entering a mixing model vary in lipid content, a bias can be introduced, because lipids are more depleted in ^{13}C than other tissues (DeNiro and Epstein 1978). Using $\delta^{13}\text{C}$ values from food sources with high lipid levels could lead to erroneous results because claws have low lipid content. Lipids contain little or no nitrogen, so if lipid content is variable among food sources, I would expect a negative relationship between C:N and $\delta^{13}\text{C}$ across food sources. I used a Pearson's correlation to test for such a relationship.

Since I analysed whole invertebrates, bias could also be introduced into the mixing model if invertebrate exoskeleton is not assimilated by nestlings, and the exoskeleton has different $\delta^{13}\text{C}$ or $\delta^{15}\text{N}$ values from soft body parts of invertebrate food sources. The chitin in invertebrate exoskeletons is derived from excretory ammonia (Webb et al. 1998 and refs within), which generally has a

lower $\delta^{15}\text{N}$ than the diet. Chitin has a relatively high C:N ratio of about 6.9, so if the presence of chitin in my food sources is influencing $\delta^{15}\text{N}$, I would expect to see a negative relationship between C:N and $\delta^{15}\text{N}$. I tested for this relationship using a Pearson's correlation.

All tests were carried out in R (R Development Core Team 2010), and results are reported as mean \pm 1 SD.

Validation of methodology

Do $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values of nestling claws reflect diet?

Claws have rarely been used as a tissue to study diet in passerines (Bearhop et al. 2003; Gagnon and Hobson 2009; Robb et al. 2011). Claws are made of keratin, which is metabolically inactive, so stable isotope values of keratin do not change once it has grown (Caumette et al. 2007). However, keratin is continually deposited at the tip of the claw (Ethier et al. 2010), so the claw could integrate isotope values of diet over time. Rapid growth in nestlings should ensure that claws collected at five or six days represent isotopic values of nestling food. Nevertheless, it is possible for young chicks to reflect the isotopic value of maternal input from the egg rather than of the diet (Sears et al. 2009). A lab-based diet switch study was outside the scope of the current study. However, I was able to test the validity of using claws to represent nestling diet in two ways. First, I compared claw values with four unhatched eggs, collected from nests in this study. If claws were reflecting the isotope value of the egg, I would

expect them to be enriched in $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ compared to the eggs, due to standard fractionation processes. Eggs were homogenized, and freeze-dried, then lipids were extracted using 2:1 DCM:methanol. Both the lipid-extracted and the raw samples were then tested for $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values as described above.

Secondly, I compared the $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values of claws from Song Sparrows with those of Brown-headed Cowbirds (*Molothrus ater*) raised in the same nests. Brown-headed Cowbirds are obligate brood parasites, and once an egg is laid in a host nest, all parental care is provided by the host birds (Lowther 1993). If $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values of nestling claws reflect diet and not the egg, claws of Brown-headed Cowbirds and Song Sparrows from the same nests are predicted to have very similar values. Claws were collected from three Brown-headed Cowbirds at approximately 6-7 days of age, and treated as described above for Song Sparrow claws.

Validity of discrimination factor

Discrimination factors are often tissue specific. Since no discrimination factors are available for claw material, I used a discrimination factor generated for feathers (mean \pm 1 SD, $\delta^{13}\text{C}$: 0.2 ‰ \pm 0.58, $\delta^{15}\text{N}$: 2.74 ‰ \pm 0.31, Kempster et al. 2007). To test whether this discrimination factor is appropriate, I reran my mixing model based on predicted feather stable isotope values instead of claw values. Predicted values were generated by regressing mean claw values against mean feather values for four nests where both tissues were collected

from dead nestlings ($\delta^{13}\text{C}$, $F_{1,2}=104.2$, $p=0.009$, $R^2=0.98$; $\delta^{15}\text{N}$ $F_{1,2}=198.3$, $p=0.005$, $R^2=0.99$, (Appendix I, J).

Validity of food sources.

Mixing models work best when food sources have well separated isotope values (Gannes et al. 1997; Phillips and Greg 2001). Here I contrast invertebrates captured in semi-natural habitats, dominated by C3 plants, and crop fields. Corn is a C4 plant, and is therefore expected to have a high $\delta^{13}\text{C}$ value compared to C3 plants, which include most other plants trees and cool-season grasses (DeNiro and Epstein 1978). Feeding experiments have consistently shown that invertebrates raised on C4 food sources have higher $\delta^{13}\text{C}$ values than those raised on C3 sources (e.g. Tallamy and Pesek 1996; Ponsard et al. 2004; Gratton and Forbes 2006), so a clear difference in $\delta^{13}\text{C}$ is expected between invertebrates captured in the corn fields and the C3 dominated semi-natural habitats. For example, Latendresse (2004) showed significantly higher values of $\delta^{13}\text{C}$ of invertebrates captured in corn fields, compared to those captured in adjacent forests or at the field-forest boundary. To create the crop food source, I combined invertebrates captured in corn fields, soybean fields and wheat fields, even though soybean and wheat are C3 plants, and have much lower $\delta^{13}\text{C}$ values than corn. This is based on my previous work, where invertebrates captured from corn and soybean fields all have relatively high $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$, and are indistinguishable using these isotopes, probably due to the regular rotation of these crops (Chapter 3). Wheat fields in this study area

are also grown in rotation with corn, and invertebrates captured in wheat fields also had high $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values. Previous studies have not directly compared the stable isotope values of invertebrates from rotational farm fields. However, Haubert et al. (2009) showed the $\delta^{13}\text{C}$ of fatty acids from spiders captured in corn and soybean fields was not strongly affected by crop type. I tested for differences in $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ among taxa and cover types using a nested ANOVA where cover type was nested in taxon.

Results

Mean $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ were always higher for the food sources collected in crop habitats than those from semi-natural habitats, and were also significantly different among taxa (nested ANOVA, $\delta^{13}\text{C}$: taxon: $F_{3,149} = 19.7$, $p < 0.001$, land cover nested in taxon: $F_{4,149} = 18.0$, $p < 0.001$. $\delta^{15}\text{N}$, taxon: $F_{3,149} = 132.1$, $p < 0.001$, land cover nested in taxon: $F_{4,149} = 5.3$, $p < 0.001$, Fig. 4.2a, Appendix H). Sap beetles (all habitats combined) had the highest mean $\delta^{13}\text{C}$ of any of the food sources.

I found no relationship between mean C:N ratios of food sources and mean $\delta^{15}\text{N}$ ($\delta^{15}\text{N}$: $r = -0.3$, $df = 5$, $p = 0.5$, Appendix K). There was a strong but non-significant relationship between mean C:N ratios of food sources and mean $\delta^{13}\text{C}$, but this was entirely driven by the high $\delta^{13}\text{C}$ of sap beetles ($r = 0.58$, $df = 5$, $p = 0.2$, Appendix K). The lack of a relationship between C:N ratios and either

$\delta^{13}\text{C}$ or $\delta^{15}\text{N}$ suggests that neither differential lipid content of food sources, nor presence of exoskeleton in food samples was a factor in the results of my mixing models.

Foraging habitat

Mean claw isotope values for two of the 28 nests were outside of the mixing polygon, represented by the area encompassed by the food sources and their standard deviations (Fig. 4.2a, Appendix L). All but one of the mean predicted feather values were within the mixing polygon (Appendix Ma).

The median proportion of diet obtained from semi-natural habitats at each nest was significantly greater than the median proportion of diet obtained from cropped habitats (proportion of diet from semi-natural habitats mean \pm SD, 0.61 ± 0.07 , from cropped habitats: 0.39 ± 0.07 , paired t-test, $t = 8.0$, $df = 27$, $p < 0.0001$). However, credibility limits overlapped in all cases (Appendix N).

Contrary to my null hypothesis that nestling diet would come from crop fields and semi-natural habitats in proportion to local habitat availability, the mean proportion of diet from semi-natural habitats was significantly greater than the proportion of semi-natural land cover both within 100 m of the nest (mean proportion of diet from semi-natural habitats 0.61 ± 0.07 , mean proportion of semi-natural cover 0.29 ± 0.24 , $V = 9$, $p < 0.001$), and within 35 m of the nest (mean proportion of semi-natural cover 0.35 ± 0.24 , $V = 25$, $p < 0.001$). The median proportion of the diet coming from semi-natural habitats did increase

slightly with the proportion of semi-natural habitat within 100 m of the nest ($F_{1,26}=5.88$, $p=0.03$, $R^2=0.15$, Fig. 4.3a). The same pattern was observed at the 35 m scale, but was only marginally significant ($F_{1,26}=4.26$, $p=0.05$, $R^2=0.11$, Fig. 4.3b). At both scales, the slope of the relationship was significantly less than the 1:1 relationship I predicted (100m: $t = -16.6$, $df = 26$, $p < 0.001$, 35 m: $t = -16.9$, $df = 26$, $p < 0.001$).

Nestling diet

Hoppers had the highest average contribution to nestling diet, but the proportion of hoppers in the diet was not significantly higher than that of caterpillars (Figs 4.2b, 4.4). The taxa that made the lowest contributions to the diet were sap beetles, ground beetles and spiders. The mixing model based on predicted feather values had very similar results and the top four food groups were identical to the mixing model based on mean claw values (Appendix Mb, O).

Do $\delta^{13}C$ and $\delta^{15}N$ values of nestling claws reflect diet?

If stable isotopes of nestling claws were reflecting the egg, rather than the diet, I expected $\delta^{13}C$ and $\delta^{15}N$ of claws either higher than or close to egg values. Instead, claw values were depleted in $\delta^{13}C$ and $\delta^{15}N$ compared to eggs, in the direction of the invertebrate food sources (Fig. 4.5). In addition, $\delta^{13}C$ and $\delta^{15}N$ of Brown-headed Cowbirds and Song Sparrows from the same nest are similar,

especially compared with the differences between Song Sparrow nestling claws and eggs (Fig. 4.6).

Validity of discrimination factor

The mean feather $\delta^{13}\text{C}$ tended to be lower than mean claw $\delta^{13}\text{C}$, and mean feather $\delta^{15}\text{N}$ to be higher than mean claw $\delta^{15}\text{N}$ (Appendix I), but this did not result in a large change in the location of the nestling values in the mixing polygon, relative to the food sources (Appendix M, Fig. 4.2).

The results of the mixing model based on mean feather values were very similar to those based on mean claw values, showing that the proportion of diet from semi-natural habitats was significantly greater than the proportion of semi-natural land cover both within 100 m of the nest (mean proportion of diet from semi-natural habitats 0.62 ± 0.07 , mean proportion of semi-natural cover 0.29 ± 0.24 , $V = 9$, $p < 0.001$), and within 35 m of the nest (mean proportion of semi-natural cover 0.35 ± 0.24 , $V = 14$, $p < 0.001$, Appendix P).

Discussion

I used stable isotope analysis to show that the diet of nestling Song Sparrows is not obtained in proportion to local habitat availability. Instead, a higher proportion of diet comes from semi-natural habitats than expected. I also

found that hoppers and caterpillars make up the largest proportion of nestling diet.

I suggest two non-mutually exclusive hypotheses for why Song Sparrows obtain more food from semi-natural habitats: habitat choice and food availability. Although Song Sparrows use a wide range of habitats, they are most closely associated with shrubs adjacent to moist or wet ground (Arcese et al. 2002). Watts (1990) studied sparrow assemblages in winter and showed that Song Sparrows favoured weedy or woody cover over mown areas, and faced relatively high predation rates in open areas compared to Savannah Sparrows (*Passerculus sandwichensis*). Therefore, Song Sparrows might simply forage preferentially in the greater cover provided by hedgerows and, in spring at least, by hayfields, compared to more open cropland.

Alternatively, the disproportionate amount of nestling diet from semi-natural habitat could be due to higher invertebrate abundance in these habitats. Several studies from Europe have shown higher abundance of invertebrates in hedgerows than in crop fields (e.g. Thomas et al. 1991; Thomas and Marshall 1999; Varchola and Dunn 2001; Meek et al. 2002; Brickle and Peach 2004; Grübler et al. 2008), presumably due to a combination of lower disturbance, reduced pesticide inputs and greater plant biomass and diversity. Invertebrate abundance in hayfields varies depending on management (e.g. Vickery et al. 2001; Hoste-Danylow et al. 2010), but is expected to be higher than in crop fields early in the season, due to higher plant biomass. Data on invertebrate biomass

from fields included in this study were inconclusive, with relative invertebrate biomass measured using a D-Vac suction sampler being higher in hedgerow and hayfields than in crop habitats, but invertebrate biomass from pitfall traps not differing among land uses (Chapter 4). Given the propensity of Song Sparrows to use shrubby habitat and the likelihood of increased food abundance in semi-natural habitat compared to crop fields, it is probable that both these factors play a role in driving the pattern of disproportionate provision of nestling food by semi-natural habitats.

Hedgerow habitat in particular seems to be important to nesting Song Sparrows; at the nests with < 7% semi-natural cover (n=9, Fig. 4.3), hedgerows comprised $\geq 90\%$ of semi-natural habitat available and therefore provided the majority of nestling food. Hedgerow density has declined in eastern Canada, as hedgerows and fencerows are removed to increase field size (Boutin et al. 2001). Management actions to stop hedgerow removal, and to increase invertebrate abundance and diversity in hedgerows, such as sowing wildflower mixtures (Thomas and Marshall 1999; Meek et al. 2002) or actions to reduce drift of pesticides into hedgerows (reviewed by Maudsley 2000), could benefit Song Sparrows and potentially other farmland breeding birds.

The proportion of nestling diet coming from semi-natural habitats increased slightly with proportion of local semi-natural cover. This relationship was stronger at the 100 m scale than at the 35 m territory scale, suggesting Song Sparrows travel outside their defended territories to forage (although they may

remain in a home range). If Song Sparrows nesting in areas with lower semi-natural cover are forced to travel farther to obtain enough food, they will spend more time away from the nest, which has been shown to lead to reduced nest survival in Song Sparrows (Rastogi et al. 2006).

Hoppers and caterpillars were the most important food sources for Song Sparrow nestlings. Hoppers and caterpillars may provide a good food source for young nestlings due to their small size and soft bodies. In reviewing Song Sparrow diet, Arcese et al. (2002) mention hoppers and larvae as food items for adults, but although various kinds of larvae are mentioned in nestling diet, hoppers are not listed as an important food item. However, the importance of hoppers in the mixing model results agrees with the dominance of hoppers in the few ligature samples I obtained from nests in the study (see methods). It is possible that hoppers, which are pests of alfalfa and soybeans (Baute 2002), are more prevalent in the diet of Song Sparrow nestlings in farmland habitats than in other habitats.

I made two key assumptions in this analysis, first that nestling claw material reflected nestling diet and not egg material, and second that the discrimination factor I used accurately reflected the change in isotopic values between the food sources and the claw material. Two lines of evidence suggest that nestling claw material reflects diet and not the egg; first, claws had lower values of $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ than eggs, closer to the values of the majority of the food sources I analysed (Fig. 4.5). If the $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values of claws depended on

the egg, rather than nestling diet, I would have expected either no difference in $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ between egg and chick (Paszowski et al. 2004) or that $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ would be enriched in claws compared to eggs (Gladbach et al. 2007; Klaassen et al. 2004). Second, Song Sparrow claw $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values were similar to Brown-headed Cowbird nestlings raised in the same nests, i.e. nestlings with different mothers, but the same diet (Fig. 4.6). Since I cannot show how similar or different Brown-headed Cowbird and Song Sparrow eggs are, this does not provide strong evidence that nestling claws do not reflect the egg. However, the similarity in isotope values of nestlings of different species raised in the same nest does suggest that $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ in nestling claws reflects their diet as nestlings. While this is not conclusive evidence, I believe this shows that the influence of the egg on $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ of nestling claws is minimal, and does not strongly affect my results.

I also assumed that the discrimination factor I used, which was generated from the feathers of juvenile Song Sparrows, accurately reflects the change in stable isotope values between food sources and nestling claws. Both the tissue type and age class are known to affect discrimination factors (Cherel et al. 2005; Caut et al. 2009; Hussey et al. 2010). However, the similarity in results obtained when the model was run with either claws or feathers suggests the feather discrimination factor was adequate in this case.

The use of stable isotope analysis of nestling tissues to examine diet and foraging habitat is becoming more common in seabirds, which cover vast

distances on a single foraging trip (e.g. Ainley et al. 2003; Gladbach et al. 2007; Williams et al. 2007; Harding et al. 2008; Hedd et al. 2010; Browne et al. 2011). However, I know of only two examples of studies that have used this technique in landbirds. Koenig et al. (2008) used stable isotope analysis of blood samples from nestling and adult Acorn Woodpeckers (*Melanerpes formicivorus*) to compare nestling and adult diets, and Kosciuch et al. (2008) used stable isotope analysis of feathers from nestling Brown-headed Cowbirds to distinguish between cowbirds raised in grassland and shrubland nests. Using claws to sample nestlings is less invasive than taking a blood sample, or collecting feathers while nestlings are still growing, and shows potential for learning about nestling diet in situations where food sources differ in stable isotope values. However, more research is needed into discrimination factors for nestling claws.

Managing farmland for both agricultural productivity and to maintain bird populations requires understanding how birds use different cover types and resources. I have demonstrated how stable isotope analysis can be used to assess the foraging habitat and diet of farmland nesting birds, and have shown the importance of semi-natural habitats, and in particular hedgerows, to Song Sparrows nesting in farmland. Management actions to stop hedgerow destruction and to increase invertebrate abundance in hedgerows may benefit Song Sparrows and other farmland birds.

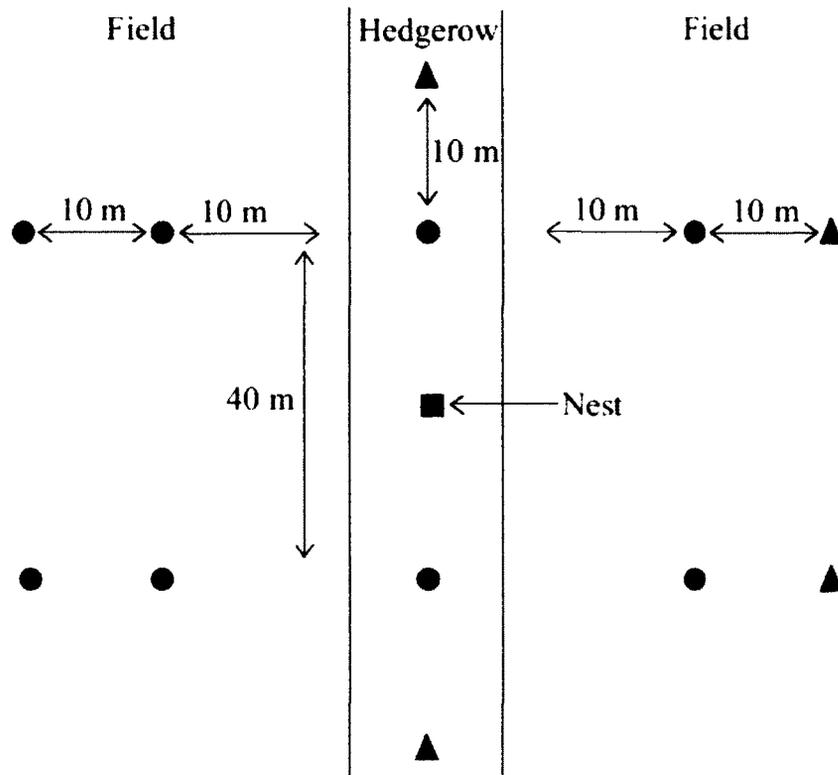
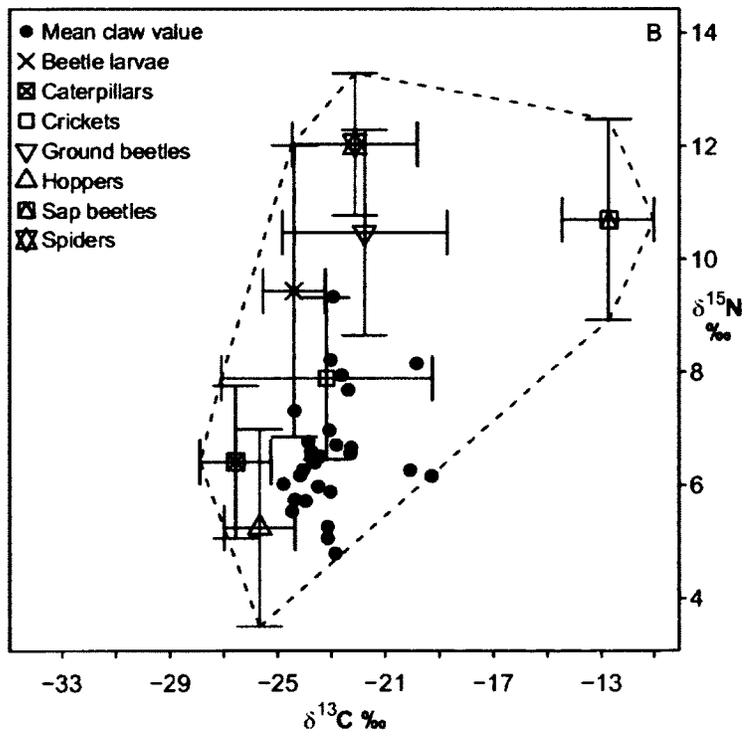
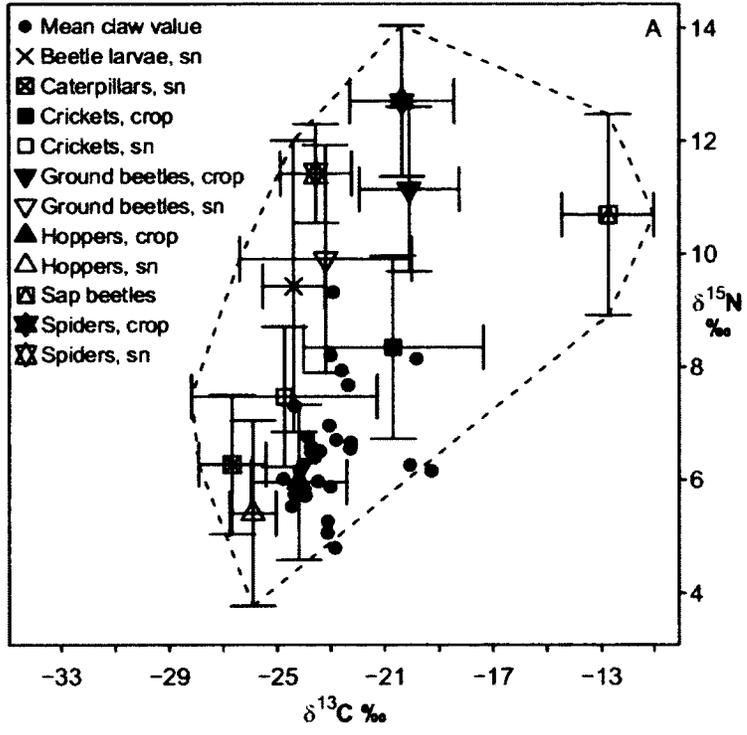


Fig. 4.1: Layout of pitfall traps used to sample invertebrates, showing a single nest site with four hedgerow traps and four traps in each adjacent field. Dots represent pitfall trap locations used in all years (2006-2008), and triangles represent pitfall trap locations used only in 2007 and 2008. D-Vac samples were collected twice in each field between the rows of traps, at least 10 m into the field and 10 m apart. Invertebrate samples were also collected by hand from shrubs and trees in the hedgerow. Adapted from and reprinted with permission of © 2011 Ecoscience

Fig. 4.2: Mean $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values of claws taken from Song Sparrow (*Melospiza melodia*) nestlings in 28 nests, together with a) $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values (mean \pm 1 SD) of invertebrate food sources collected from adjacent crop fields (solid symbols) and semi-natural and forage habitats (sn, open symbols), and b) $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values of invertebrate food sources averaged over all habitats. Food sources have discrimination factors added to them, to account for the concentration of heavier isotopes at higher trophic levels ($\delta^{13}\text{C}$: 0.2 ‰, $\delta^{15}\text{N}$: 2.74 ‰, Kempster et al. 2007). The dotted lines show the mixing polygon bounded by the food sources entered into the mixing model. Reprinted with permission from © 2012 Canadian Science Publishing or its licensors



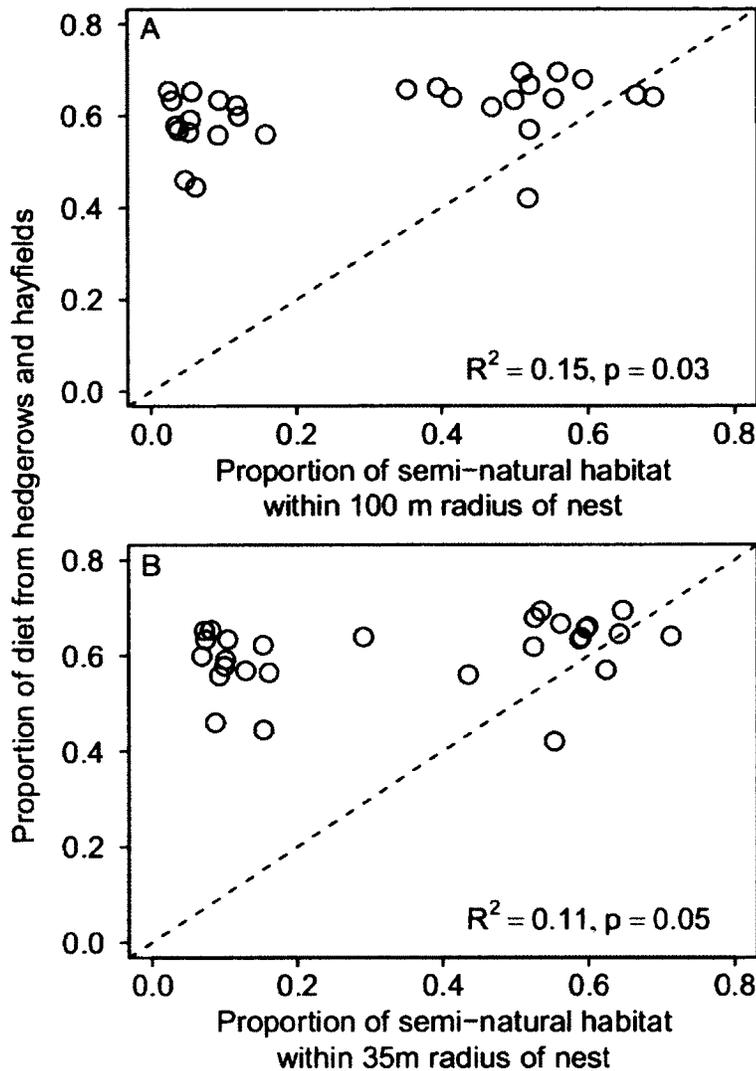


Fig. 4.3: Relationship between the proportion of semi-natural cover within a) 100 m and b) 35 m of Song Sparrow (*Melospiza melodia*) nests, and the median proportion of the diet of nestlings estimated to come from semi-natural habitats. The line shows the 1:1 relationship expected if nestling food is obtained from semi-natural habitat in proportion to local habitat availability. Reprinted with permission from © 2012 Canadian Science Publishing or its licensors

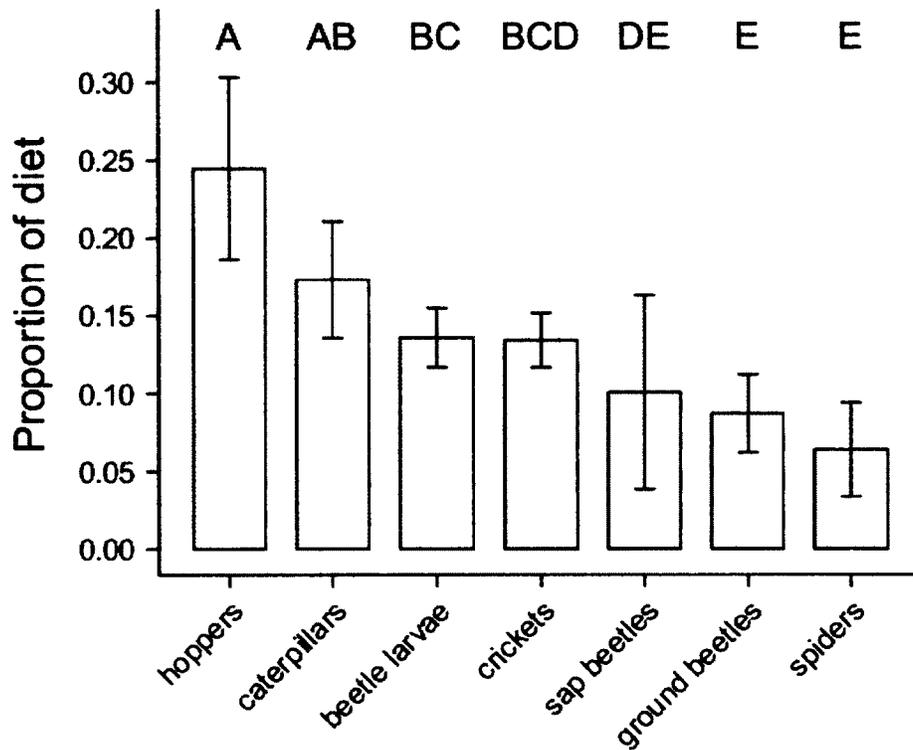


Fig. 4.4: Median estimated proportion of diet from invertebrate food sources for 28 Song Sparrow (*Melospiza melodia*) nests. Error bars show ± 1 SD. Letters indicate significant differences at $p < 0.05$ (Kruskal-Wallis $\chi^2_{7} = 143.62$, $p < 0.0001$). Reprinted with permission from © 2012 Canadian Science Publishing or its licensors

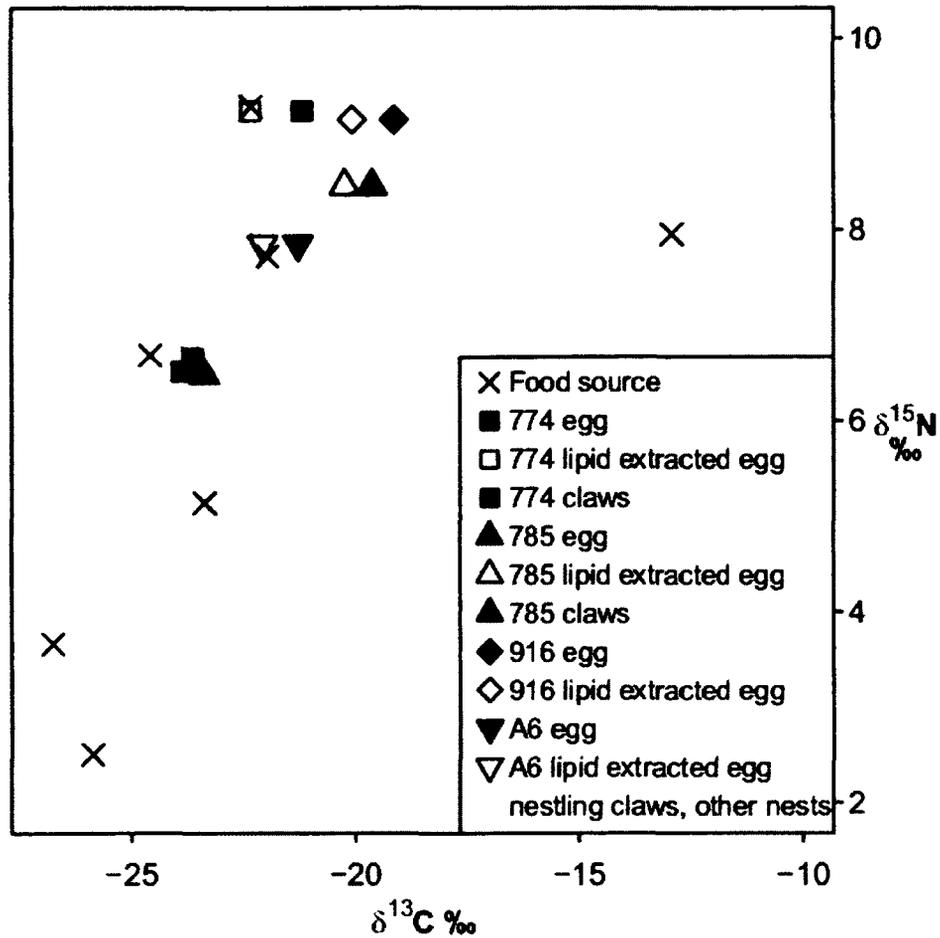


Fig. 4.5: $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values for Song Sparrow (*Melospiza melodia*) eggs and claws, together with $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values of invertebrate food sources. Values from four nests are highlighted; we analysed an egg, and two claws from nest 774, an egg and two claws from nest 785, and an egg only from each of nests 916 and A6. Grey circles show values of song sparrow claws from nests from which only claw material was analysed. No discrimination factor is applied. Reprinted with permission from © 2012 Canadian Science Publishing or its licensors

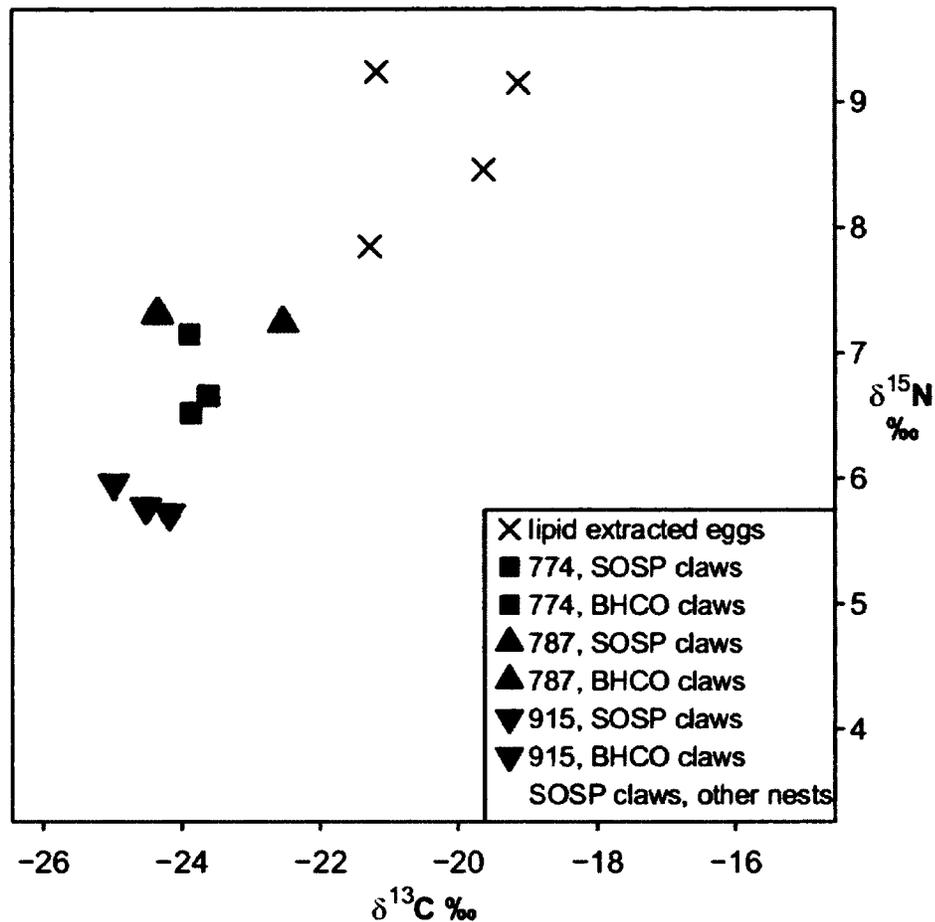


Fig. 4.6. $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values for eggs and claws of Song Sparrow (*Melospiza melodia*, SOSP) and Brown-headed Cowbird (*Molothrus ater*, BHCO) nestlings. Values from three nests are highlighted; we analysed, a cowbird claw and two sparrow claws from nest 774, a cowbird claw and two sparrow claws from nest 915, and a single cowbird claw and a single song sparrow claws from nest 787. Grey circles show values of song sparrow claws from nests from which only claw material was analysed. No discrimination factor is applied. Reprinted with permission from © 2012 Canadian Science Publishing or its licensors

Chapter Five

Effect of availability of food and foraging habitat on reproductive success of Song Sparrows (*Melospiza melodia*) nesting in farmland.

This chapter forms the basis of a manuscript submitted to Condor
J. Girard, P. Mineau and L. Fahrig. 2012. Effect of availability of food and foraging habitat on reproductive success of Song Sparrows (*Melospiza melodia*) nesting in farmland. Condor

Abstract

Low food availability can have negative consequences for avian reproductive success. Intensive agriculture has reduced the availability of invertebrate food in farmland, and low food availability has been associated with poor reproductive success of some European farmland birds. However, the importance of food availability to birds breeding in arable farmland in North America has rarely been addressed. I studied Song Sparrows (*Melospiza melodia*) breeding in hedgerows on 15 dairy and arable farms in eastern Ontario. In this study area, Song Sparrows obtain more nestling food from semi-natural habitats than from crop fields. I predicted that 1) nestling mass, number of fledglings and chick survival will be positively associated with local food availability, 2) nestling mass, number of fledglings and chick survival will be positively associated with semi-natural habitat and 3) nest survival will be positively associated with semi-natural habitat. I used pitfall traps and suction sampling to measure abundance of invertebrates important to nestlings in habitats close to Song Sparrow nests. Food availability and cover of semi-natural habitat affected neither nestling mass, number of fledglings nor nestling survival. I also found no support for an effect of semi-natural cover on nest survival. Adults may be compensating for low food availability through increased parental effort, masking or reducing the effect of low food abundance. Alternatively, the relatively low input of insecticides in the corn and soybean crops that dominate the study area, may mean that food availability is high enough to support successful reproduction in Song Sparrows in farmland in this region.

Introduction

Food availability has been shown to affect reproductive success of birds in both experimental and natural systems (reviewed by Martin 1987; Potts 1986, Boutin 1990, Rodenhouse and Holmes 1992, Zanette et al. 2006a). Low food abundance can result in reductions in nest survival (Duguay et al. 2000, Brickle et al. 2000, Richmond et al. 2011), nestling or chick growth (Potts 1986, Duguay et al. 2000, Hart et al. 2006, Brickle et al. 2000, Brickle and Peach 2004), nestling or chick survival (Potts 1986, Rodenhouse and Holmes 1992), number of fledged young (Rodenhouse and Holmes 1992, Siikamaki 1998, Boatman et al. 2004, Strong et al. 2004, Britschgi et al. 2006), fledgling size (Siikamaki 1998), and number of nest attempts (Rodenhouse and Holmes 1992, Nagy and Holmes 2005).

Reduced invertebrate food availability during the nesting season has been proposed as a mechanism behind the decline of some species of farmland birds (Campbell and Cooke 1997, Boatman et al. 2004, Newton 2004). Intensive agricultural practices are thought to reduce the availability of invertebrate food sources through pesticide use, simplification of crop rotation and removal or degradation of hedgerows and field margins (reviewed by Campbell and Cooke 1997, Wilson et al. 1999, Robinson and Sutherland 2002, Boatman et al. 2004, Newton 2004). In addition to directly reducing invertebrate food availability, intensive agriculture may also reduce availability of foraging habitat, through loss of field margins, reduction in the proportion of hay and pasture and intensification of grassland management (Brun and Smith 2003, Granbom and Smith 2006).

Compared to Europe, and especially to the UK, little is known about the potential effects of reduced food availability on reproductive success of farmland birds in North America. To begin to address this knowledge gap, I studied the effect of local food availability and cover of foraging habitat on nestling mass and nest survival of Song Sparrows (*Melospiza melodia*), nesting in farmland. Song Sparrows have been well studied in other habitats, and it has been shown that supplemental food can have a positive effect on many aspects of reproduction including clutch size, nestling weight and growth, number of independent young produced and nest survival (Arcese and Smith 1988, Searcy et al. 2004, Rastogi et al. 2006, Zanette et al. 2006a, Zanette et al. 2006b). In addition, I have shown that Song Sparrows nesting in hedgerows obtain a higher proportion of nestling food from hedgerows and hayfields than from crop fields (Chapter 4). I predicted that 1) nestling mass, number of fledglings and nestling survival will be higher in nests with higher local food availability, 2) nestling mass, number of fledglings and nestling survival will be higher in nests with higher cover of semi-natural habitat and 3) nest survival will be higher at nests with higher cover of semi-natural habitat.

Methods

Study area

The study was conducted at 15 sites in eastern Ontario, south east of Ottawa, between the towns of Navan (45°25'15.68"N, 75°25'36.28"W) and

Embrun (45°16'26.29"N, 75°16'30.04"W). Each site consisted of a hedgerow and the adjoining farm fields. The dominant crops in the study fields were corn (*Zea mays*), soybean (*Glycine max*) and hay, but other crops including wheat (*Triticum* spp.), alfalfa (*Medicago sativa*) and pasture were also present. Hedgerows were selected based on suitability for Song Sparrows (a mix of shrubs, trees and open herbaceous areas along the hedgerow) and willingness of farmers to cooperate. Some hedgerows contained ditches, but streams and areas of permanent water were avoided where possible.

Study species.

Song Sparrows are common and abundant in farmland in North America (Best et al. 1995, Jobin et al. 1996, Boutin et al. 1999, Best 2001, Freemark and Kirk 2001, Jobin et al. 2001, Henningsen and Best 2005). They nest in shrubby to open habitats such as hedgerows and other field boundaries, hayfields and fallow or abandoned fields. Nests are typically well concealed by surrounding vegetation and are most often built on or close to the ground, although they can be built up to 4 m high in shrubs or bushes (Arcese et al. 2002). Song Sparrow nests are frequently parasitized by Brown-headed Cowbirds (*Molothrus ater*), which can lead to nest abandonment (Smith et al. 2003), and reductions in the number of Song Sparrows fledged per nest (Smith and Arcese 1994). Song Sparrow populations in the study region (lower Great Lakes/St. Lawrence Plain) have shown a long-term pattern of decline (1966 - 2009: -0.6 % change/year, 95% CI: -0.9 to -0.4 , $n = 167$, Sauer et al. 2011).

Small invertebrates are the primary food of Song Sparrows during the breeding season (Arcese et al. 2002). I have previously shown that Song Sparrows breeding at my study sites preferentially feed their nestlings hoppers (Auchenorrhyncha, 'hoppers' includes leafhoppers, plant hoppers, froghoppers etc.) and caterpillars (Lepidoptera) and that Song Sparrows obtain proportionally more nestling food from hedgerows and hayfields than from crop fields (Chapter 4).

Nest searching and monitoring.

Nest searching was carried out in 2006 - 2008. Two observers walked on either side of the hedgerow, watching for females flushing off a nest. Once a female flushed, the vegetation was carefully searched until the nest was found. Nests were also found by observing adults, and by chance. During nest searching, observers wore gardening gloves and rubber boots to reduce scent left on the vegetation.

Once a nest was found, I tried to check it every 3 – 4 days, although days between nest checks ranged from 1 – 9 days (mean \pm SD, 3.84 \pm 1.78). During nest checks I recorded the contents of the nest, and the behaviour of the adults. I aged nestlings based on feather growth and development, using photographs in Jongsomjit et al. (2007), but I did not attempt to age eggs. A nesting attempt was recorded as failed when the nest was found empty before the nestlings could have reached 10 days of age, the average age of fledging in Song Sparrows (Arcese et al. 2002). A nest was recorded as successful when nestlings reached

10 days of age, or when fledging was witnessed. Information on nest status collected after the estimated fledge date of 10 days (e.g. fecal matter on the edge of the nest, sightings of fledglings) was not used to make decisions about nest status or outcome (Stanley 2004).

Measures of nestling mass.

Measurements of nestling mass were obtained from 34 Song Sparrow nests, eight of which were parasitized by Brown-headed Cowbirds. Most nestlings were measured at 5 or 6 days of age, except at five nests where nestlings were measured at four days. Nestling mass was recorded to the nearest 0.1 g.

Local habitat.

The mean defended territory size of Song Sparrows in these study sites is 35 m radius (J. Girard, unpublished data). However, I previously found a stronger relationship between the amount of food coming from semi-natural land uses, and semi-natural land cover within 100 m of the nest than within 35 m of the nest (Chapter 4), suggesting that birds forage outside their defended territories. Therefore, I quantified the land uses within 100 m of each nest using 2008 aerial photographs of eastern Ontario (DRAPE 2010). Land uses were classified as cropped (barley (*Hordeum vulgare*), canola (*Brassica napus*), corn, oats (*Avena sativa*) and peas, sorghum (*Sorghum* sp) and wheat), semi-natural

(includes alfalfa, hay, hedgerow, pasture, riparian, rough grass, verge) or non-foraging habitat (farmsteads, forest, river, road, track).

Food abundance.

To measure food abundance I trapped ground invertebrates and invertebrates in the vegetation in agricultural land adjacent to nests in 2007 and 2008. Invertebrates were collected only after nestlings were hatched, and usually when nestlings were 5 – 8 days old.

To measure abundance of ground invertebrates, I used pitfall traps. At each nest, four traps were placed in the hedgerow, four on each side of the hedgerow at the interface of the hedgerow and the field, and four in each adjacent field for a total of 20 traps per nest (Fig 4.1). Pitfall traps consisted of a plastic cup (85 mm diameter, 115 mm high), placed in a hole in the ground, so that the top of the cup was level with the soil surface. Traps were half filled with water, with a drop of dish soap added to break the water tension. After three days, traps were collected, and the contents of each trap were returned to the lab.

To measure the abundance of invertebrates in the vegetation, I used a D-Vac suction sampler (Dietrick et al. 1959). In all land uses except corn, I used a 340 mm diameter funnel. To collect a single sample, the D-Vac was raised vertically over the vegetation and pressed down to the ground for a total time of 5 sec, three times within 2 m². The structure of corn is so different to that of other

vegetation types that it was not possible to use the D-Vac in the same way. Instead, I used a smaller, 100 mm diameter funnel, which took invertebrates off individual corn plants more efficiently. The operator moved along the corn row suctioning plants up to a height of 1 m, for a total of 20 s. Once a sample was complete, the content of the net was placed into a glass jar filled with denatured ethanol, to kill and preserve the invertebrates. At each sampled nest, two samples were collected in the hedgerow, approximately 15 m either side of the nest, and two samples were collected in each adjacent field, approximately 15 m into the field, and ≥ 10 m apart.

In the lab, each invertebrate sample was identified to order, suborder or family (ground beetles only, Appendix Q), and all individuals were measured in 2 mm intervals. To generate equations to convert invertebrate length to biomass, I collected individuals from each taxon-size combination and dried them in an oven at 60°C for ≥ 48 hours, until completely dry. Samples were then weighed to the nearest 1 mg.

Statistical analysis

Nestling size and survival. Before calculating the mean nestling mass for each nest, I first standardized the mass of each nestling to the predicted mass at five days old (Zalik and Strong 2008). I first found the mean nestling weight at age 4, 5 and 6 days (nestlings from nests which experienced brood parasitism or

brood reduction were excluded from this calculation, since these factors could have unpredictable impacts on nestling size). For nestlings weighed on day 4 or day 6, I then took the difference between actual mass and average mass for that age, and applied it to the average mass at day 5. This method assumes that between the day of measurement and day 5, a) nestling growth rate is constant and b) deviations from predicted mass are constant (Zalik and Strong 2008). Since measurements were taken within one day of day 5, I believe these assumptions are reasonable. Once nestling mass was standardized to day 5, it was averaged at each nest to give mean nestling mass per nest.

I also calculated total biomass of nestlings in each nest, since nestlings in nests with fewer siblings might receive more parental care and therefore have larger mass or higher condition, despite requiring less total food (Brickle and Peach 2004). Where a nest contained only Song Sparrow nestlings, this was simply the total predicted five day mass. I did not weigh enough Brown-headed Cowbird nestlings to predict their mass at a standardized age, ($n = 7$), so when a nest contained Brown-headed Cowbird nestlings, I added the mean weight of all the Brown-headed Cowbirds weighed (19.28 g), for each cowbird nestling present.

Number of fledglings was calculated as the number of Song Sparrow nestlings fledged from nests that fledged at least one Song Sparrow. Nestling survival was calculated as the number of Song Sparrow fledglings divided by the number of Song Sparrow eggs laid.

Measures of food availability. I created two indices of nestling food availability. First, since Song Sparrows are generalist foragers, and have been recorded feeding a wide variety of foods to their nestlings (Arcese et al. 2002), the nestling food index included all taxa that were not noxious (e.g. wasps, bees), or difficult for young nestlings to consume in some way (e.g. earthworms). Taxa that occurred in invertebrate samples only by chance were also excluded (e.g. predaceous diving beetles, Appendix Q). The second food index included hoppers and caterpillars (Lepidoptera larvae), which are important food items for Song Sparrow nestlings in these study sites (Chapter 4). The hopper and caterpillar index was only created for the D-Vac data, as pitfall traps did not adequately sample these taxa. Invertebrates less than 2 mm long were excluded from indices of nestling food availability, partly because many such individuals were lost during sieving of samples, and partly because these small invertebrates are unlikely to be important in Song Sparrow nestling diet.

To create the food indices, I generated equations to predict invertebrate biomass by regressing log mean dry biomass against log body length for each taxon (Appendix R). Estimated nestling food biomass was then calculated from each invertebrate sample, and hopper and caterpillar biomass was calculated from D-Vac samples.

Pitfall trap captures reflect both the 'activity' (i.e. movement) of invertebrates across the ground, and the biomass of those invertebrates, hereafter activity biomass. Because pitfall traps are passive, it is impossible to

know what area each trap samples, so I use average activity biomass per trap as an index of the availability of prey to a Song Sparrow foraging on the ground. Relative activity biomass at each nest was estimated by totalling the average activity biomass per trap for each terrestrial habitat within 100m of the nest, and dividing by the number of habitats present. Pitfall traps in the hedgerow and field edges were not considered independent, because they were less than 10m apart, so biomass from edges and hedgerows were averaged together. At 10 nests, pitfall trap data were not available for some major habitats due to lack of permission from landowners, or destruction of traps. In these cases I used an average value for that habitat type from the other sites. In two of these cases, the missing habitat type was pasture. This data was missing because I was not permitted to sample in pasture due to the presence of animals. Instead, I used the value from an adjacent hayfield, as this was the habitat type most similar to pasture. I used a Kruskal-Wallis test, to test whether activity biomass differed among land use types.

Suction sampling has the advantage of estimating absolute invertebrate abundance (Southwood and Henderson 2000). However, the efficiency of suction sampling varies among taxonomic groups and between sites (Sanders and Entling 2011). These biases were shown to be lowest for Diptera and Auchenorrhyncha (Sanders and Entling 2011) which made up the bulk of the nestling-food invertebrates captured by the D-Vac (Auchenorrhyncha = 36 % of captures, Diptera = 30 % of captures, $n = 2540$ individuals captured), so I used

the biomass of invertebrates captured by the D-Vac as a measure of relative food abundance in the vegetation among nests.

I did not attempt to correct for the different suction sampling method used in corn compared to other land uses. The majority of a suction sample is collected in the first few seconds of the sample (Bell et al. 2002, Brooks et al. 2008), so the time difference between the two methods is considered insignificant. The use of the narrower funnel in the corn was specifically designed to collect insects from the corn plants, which would not have been collected with the wider funnel. I also note that the mean biomass of invertebrates collected from the corn was similar to that collected from other crop plants (Figs. 5.1b, c). I used a Kruskal Wallis test, followed by a Nemenyi-Damico-Wolfe-Dunn post-hoc test (Hollander and Wolfe 1999) to test whether invertebrate biomass differed among land uses.

I calculated mean food availability in the vegetation for each food index at each nest, by averaging the biomass from each foraging habitat sampled, then multiplying by the proportion of foraging habitat within 100 m of the nest comprised of that habitat type. Two foraging habitats were not sampled; roadside verges and riparian habitat. Verges occurred within 100 m of four nests included in the analysis, making an average of 8 % of habitat cover at those four nests. Since all verges were grassy, I used an average hay biomass for the verge area and included them in the food index. Riparian habitat never made up > 3 % of habitat cover, and was excluded from the analysis.

I previously showed that a higher proportion of Song Sparrow nestling diet comes from semi-natural land uses than from cropped land uses, per unit area, in this study area (Chapter 4). I accounted for this by calculating food availability in the cropped and semi-natural land uses at each nest, weighting them by the proportion of food expected to come from each land use type (cropped: 39 %, semi-natural 61%, Chapter 4), and summing to give adjusted food availability. This was done for both pitfall data and D-Vac data.

Prediction one: nestling mass, nestling survival and number of fledglings increase with local food availability. To test my prediction that nestling mass, number of fledglings and nestling survival would increase with local food availability, I used Spearman's rank correlations to test for correlations between my measures of reproductive success and my indices of food availability.

Prediction two: nestling mass, number of fledglings and nestling survival increase with availability of semi-natural land use.

To test my prediction that my measures of reproductive success would increase with availability of semi-natural habitat, I used Spearman's rank correlation to test for correlations between my measures of reproductive success and the proportion of semi-natural land use within a 100 m radius of each nest.

Prediction three: nest survival increases with availability of semi-natural land use. I used an information-theoretic approach to examine the effect of the availability of semi-natural cover on nest survival. To understand the effect of

semi-natural land use relative to other variables that might affect nest survival, I compared candidate models including semi-natural cover and temporal and biological factors known to affect nest survival. The variables included in the candidate models were:

(i) semi-natural cover; percent cover within 100m of the nest.

(ii) date and date squared; date can have a non-linear effect on nest survival (Grant et al. 2005, Davis et al. 2006).

(iii) nest stage; a categorical variable comparing survival in the egg stage (1) and in the nestling stage (0). Nest survival has been shown to vary non-linearly with nest age (Jehle et al. 2004, Grant et al. 2005, Kroll and Haufler 2009, Kerns et al. 2010), but I could not use a continuous age variable, because nests which were found with a complete clutch, but failed before hatching, could not be aged. Stage was assigned based on the contents of the nest at each visit.

(iv) brood parasitism; a categorical variable, 0 for non-parasitized nests, 1 for parasitized nests. Brood parasitism can affect nest survival in a number of ways including abandonment of parasitized nests and cowbird-induced nest failure (egg removal by cowbirds leading to abandonment of nest by hosts, or nestling destruction, reviewed by Smith et al. 2003, see also Zanette et al. 2007).

(v) year; a categorical variable comparing nest survival among the three years of the study.

I considered models that contained all possible additive combinations of these variables, as well as first order interactions between nest stage and brood parasitism, nest stage and date, and nest stage and semi-natural cover. I also included a null model, which included only the intercept and therefore modelled constant nest survival. This gave a total of 51 candidate models. All continuous variables were standardized using z scores.

Nest survival was modelled using the logistic exposure method (Shaffer 2004). This method uses a generalized linear model, with a binomial response (0 for failure, 1 for success) and a logit link function to model daily survival rate of nests. The logit link function accounts for the period of time each nest is under observation (Shaffer 2004). The sample size for this model is the sum of the total number of days all nests were known to survive and the number of intervals that ended in failure (Rotella et al. 2004). For nests with uncertain outcomes, the nest record was terminated at the last visit for which the status was known (Manolis et al. 2000). Nest survival was modeled using Proc Genmod in SAS (Shaffer and Thompson 2007).

I first checked for a random effect of nest or site, using Proc Nlmixed (Rotella et al. 2007). Neither random effect improved the global model, so I proceeded with the fixed effects model (Zurr et al. 2009).

I evaluated models using Akaike's Information Criterion, adjusted for small samples sizes (AIC_c , Hurvich and Tsai 1989). AIC_c attempts to find the best expected Kullback-Leibler model for the current sample size, therefore

incorporating a bias-variance trade off (Burnham and Anderson 2004). AIC_c is also appropriate for tapering effect sizes, which are nearly always a factor in model selection (Burnham and Anderson 2004). Models with $\Delta AIC_c < 2$ were considered to have the most support, and comprise the confidence set of models (Burnham and Anderson 2002). I used the Hosmer and Lemeshow (2000) goodness-of-fit to test the fit of the global model, and the model with the lowest AIC_c . The relative importance of predictor variables included in the confidence set was estimated by summing Akaike weights across all models in which each variable occurred. I calculated model averaged parameter estimates, and unconditional standard errors to assess parameter uncertainty. Parameter estimates were calculated across all candidate models for variables included in the confidence set of models. Model averaged estimates of daily survival rate and 95 % unconditional confidence limits were calculated for the two most important variables, while holding all other variables constant either at the mean value for continuous variables (date: June 2), or at a reference level for categorical variables (brood parasitism = 0, year = 2007).

All nest survival analyses, except the goodness-of-fit test were carried out in SAS 9.1; all other analyses were conducted in R v 2.12 (R Development Core Team 2011).

Results

Food availability.

Pitfall trapping was carried out at 29 nests for which nestling measurements were also available. Ground beetles (38 % of biomass captured), millipedes (Diplopoda, 19%) and true spiders (Araneae, 16 %) dominated the biomass of invertebrates in the nestling food index. Activity biomass was not significantly different among foraging habitats (Kruskal Wallis $X^2_5 = 6.7$, $P = 0.25$, Fig. 5.1a raw food availability data can be found in Appendix AC).

D-Vac sampling was carried out at 17 nests for which nestling measurements were also available. Hoppers (45 %) and flies (Diptera, 17 %) dominated the biomass of nestling food invertebrates captured with the D-Vac. Mean biomass of nestling food captured with the D-Vac was significantly higher in the hedgerows than the other foraging habitats (Kruskal Wallis $X^2_5 = 32.7$, $P < 0.001$, Fig. 5.1b). The mean biomass of hoppers and caterpillars was also significantly higher in the hedgerows than the other foraging habitats (Kruskal Wallis $X^2_5 = 31.5$, $P < 0.001$, Fig. 5.1c).

Effect of food availability on nestling mass.

Contrary to my prediction, there was no significant relationship between any index of food availability and either mean nestling mass, total nestling mass, nestling survival or number of Song Sparrow fledglings (Tables 5.1 and 5.2, Appendices S-X, nestling size raw data can be found in Appendix AB).

Effect of semi-natural cover on nestling mass.

Semi-natural cover within 100 m of Song Sparrow nests ranged from 1-93 % (Fig 5.2). Contrary to my prediction, there was no significant relationship between semi-natural cover within 100 m of Song Sparrow nests, and mean nestling mass, total nestling mass, nestling survival or number of Song Sparrow fledglings (Table 5.1, Fig. 5.2).

Effect of semi-natural cover on nest survival.

I found 118 active Song Sparrow nests, of which 116 had sufficient data for nest survival analysis. Evidence of brood parasitism was found at 14 nests (12%), and brood reduction occurred at 7 nests (6 %). Predation was the most common cause of nest failure ($n = 61$ nests), but nests also failed due to abandonment ($n = 4$) and disturbance or destruction by farm practices ($n = 4$). The effective sample size for logistic exposure analysis (the sum of the total number of days all nests were known to survive and the number of intervals that ended in failure) was 1131 nest days (raw data for nest survival analyses can be found in Appendix AD).

The nest survival model with the lowest AIC_c contained only the effect of nest stage (Table 5.3, Appendix Y). There are three additional models with $AIC_c < 2.0$, however these larger models all include nest stage and are not truly competitive with the top model, because the extra parameter does not reduce the log-likelihood (Appendix Y) and therefore does not improve the fit of the model (Burnham and Anderson 2002, Arnold 2010). Neither the global model, nor the nest stage model showed evidence of absence of lack of fit using the Hosmer

and Lemeshow goodness-of-fit test (global model: $X^2 = 10.4$, $P = 0.21$, nest stage model: $X^2 = 5.2$, $P = 0.53$). The variable with the highest estimated importance was nest stage, followed by semi-natural cover (Table 5.4). However, only nest stage had confidence intervals that did not include zero, suggesting that the effects of the other variables, including semi-natural cover are not interpretable.

The model averaged parameters showed that daily survival rate was higher in the egg stage than the nestling stage (Fig. 5.3). The model averaged effect of semi-natural cover on the daily survival rates was negligible during both the egg stage and the nestling stage, when other variables were held constant (Fig. 5.3).

Discussion

Contrary to my predictions, I did not find any evidence that food availability or cover of semi-natural habitat affects nestling mass, number of fledglings or nestling survival, or that cover of semi-natural habitat affects nest survival for Song Sparrows nesting in farmland. This suggests that either adult birds are able to compensate for low food availability, or that food availability in the study area is high enough to support Song Sparrow reproduction.

Adult birds can compensate for low food availability by travelling further to find food (Brickle et al. 2000, Brun and Smith 2003, Martin et al. 2000, Stauss et al. 2005, Low et al. 2010), increasing male parental care (Wittenberger 1982),

allocating food to the nestlings at the expense of adults (Cucco and Malacarne 1995) or by increasing the time spent foraging per day (reviewed in Zalik and Strong 2008). For example, Brun and Smith (2003), found a negative relationship between foraging distances of European Starlings (*Sturnus vulgaris*) and the proportion of pasture, a favoured foraging habitat. Other studies that found no evidence for a link between food or land use and nesting success of farmland birds also suggest that adults might be compensating for low food availability in some way (Bradbury et al. 2003, Zalik and Strong 2008, Gilroy et al. 2009). Zalik and Strong (2008) showed that despite a substantial reduction in food availability in hay fields after mowing, clutch size and nestling mass of Savannah Sparrows was similar between mown and unmown fields. They attribute this to increased parental effort, probably through an increase in the proportion of the day spent foraging for nestlings. Increased adult investment can affect future reproductive efforts, particularly by reducing adult survival (Nur 1988), future clutch size (Gustafsson and Sutherland 1988, Moller 1993) and increasing the length of time between nesting attempts (Moller 1993, Verboven et al. 2001, Mariette et al. 2011). I was not able to follow Song Sparrows through multiple nesting attempts, so I cannot rule out effects of low food availability on seasonal, or lifetime reproductive success.

A second explanation for the lack of relationship between food availability or semi-natural habitat and my measures of reproductive success, is that food availability was high enough to support successful reproduction of Song Sparrows in my study area. To my knowledge, none of the corn or soybean fields

from which I collected invertebrates was sprayed with insecticides during my study, and only one soybean field was sprayed with fungicide (personal observation, personal communication with farmers). However, seeds treated with pesticides are commonly used, which could potentially negatively affect activity density of ground invertebrates (Leslie et al. 2010, Bhatti et al. 2005). To the best of my knowledge, these practices are typical of farms in eastern Ontario. In contrast, insecticides were sprayed close to nests in previous European studies that showed negative effects of low food availability on songbirds breeding in arable farmland (Brickle et al. 2000, Boatman et al. 2004, Morris et al. 2005, Hart et al. 2006; however, Brickle and Peach 2004 do not mention whether insecticides were used close to nests). Herbicides, both pre- and post-emergent, are sprayed routinely on farms in this study. Although herbicide use can reduce abundance of food important for nestlings (Frampton and Dorne 2007) and juvenile birds (Rands 1985), the indirect effect of herbicides on nestling growth or survival is not as well established as the direct effect of insecticides (reviewed by Boatman et al. 2004). Therefore, I suggest either that current farming techniques in eastern Ontario do not impact invertebrate populations to the extent that Song Sparrow reproduction is affected, or that invertebrate populations are affected by a small enough margin that extra investment by adults is enough to compensate for reduced invertebrate abundance.

Food availability is difficult to measure because it requires knowledge of which prey are used by the bird as well as a measure of the abundance of those

prey in habitats used by the bird while foraging (Wolda 1990). I attempted to tailor my nestling food indices to Song Sparrows by removing taxa which are unlikely to be important to nestlings, either because they are difficult for nestlings to handle (e.g. worms, slugs) or because they were noxious in some way (e.g. wasps and bees). I also included a measure based only on hoppers and caterpillars, which I have shown to be important to nestling Song Sparrows in this study area (Chapter 4).

Song Sparrows forage primarily on the ground, and my pitfall traps measure the biomass of invertebrates moving around on the ground, where they could be encountered by a foraging bird. Pitfall traps do not give measures of absolute invertebrate abundance, because capture rate is biased by how active an invertebrate is (Greenslade 1964), which varies with species, sex, season, weather and potentially what other invertebrates have already been captured, among other variables (reviewed by Sunderland et al. 1995). Pitfall trap captures should be compared among habitats with caution, as changing vegetation types can alter the movement of invertebrates across the ground, and therefore affect capture rates (Greenslade 1964). However, I believe that the pitfall traps provide a good measure of invertebrates available to Song Sparrows foraging on the ground for two reasons. First, the changes between habitat types that affect pitfall captures will have the same effects on foraging sparrows; if pitfall captures are lower in dense vegetation because invertebrates do not move around as freely, sparrows foraging on the ground are also likely to encounter potential food items less often in dense vegetation. Secondly, the pitfall traps were set during a

period when Song Sparrows were foraging for nestlings, so factors that affect pitfall captures, such as seasonality and weather, will also be a factor for sparrows foraging for food.

While Song Sparrows forage primarily on the ground, they are generalists, and also forage in herbaceous vegetation as well as in shrubs and trees (Rosenberg et al. 1982, Arcese et al. 2002, personal observation). I used the D-Vac to sample invertebrates in the herbaceous vegetation in hedgerows and hayfields, and in the crop fields, where Song Sparrows were observed to forage. The D-Vac does efficiently capture hoppers, as well as Diptera, Hymenoptera, Curculionoidea, Heteroptera, and Araneida, and provides a reasonable measure of the abundance of these groups (Sanders and Entling 2011). I was only able to conduct D-Vac sampling near 17 nests where nestling measurements were available, due to poor weather conditions in 2008. However, the data show no indication of a positive trend between food abundance in the vegetation and nestling mass (Appendices U-X), and I do not think that a larger sample size is likely to have led to a different result.

Lack of variability in food availability among nests would compromise my ability to find an effect of food availability on reproductive success. However, the variability I recorded in food availability was comparable to that found in other studies that did find an effect of food availability on nestling mass or nest survival in an agricultural context. For example, Hart et al. (2006), used chick food items per sample, collected with a D-Vac to estimate chick food availability. This varied

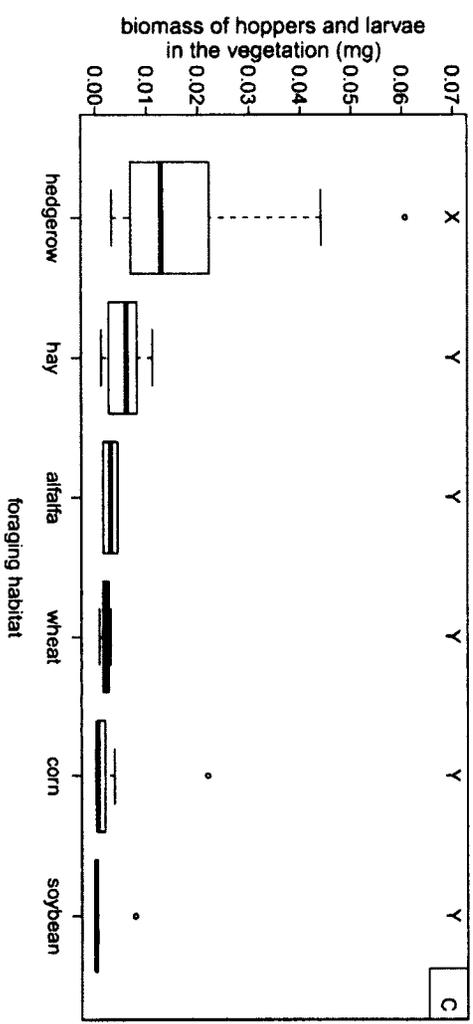
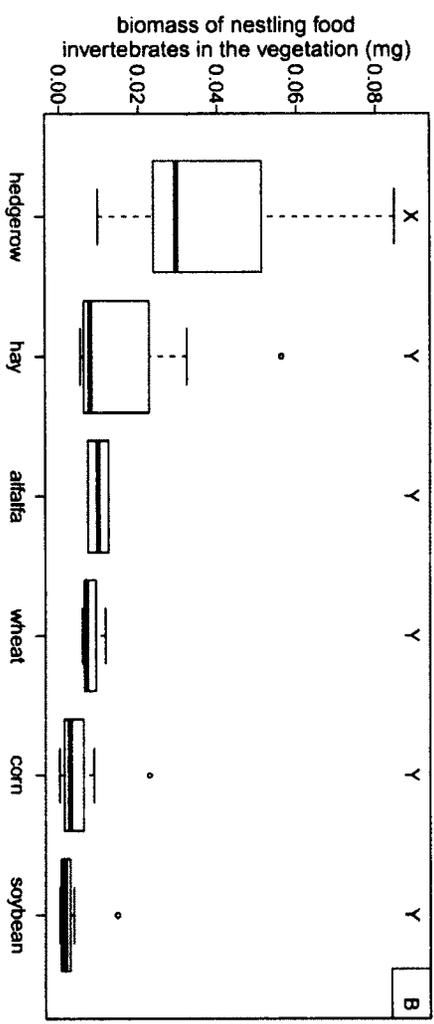
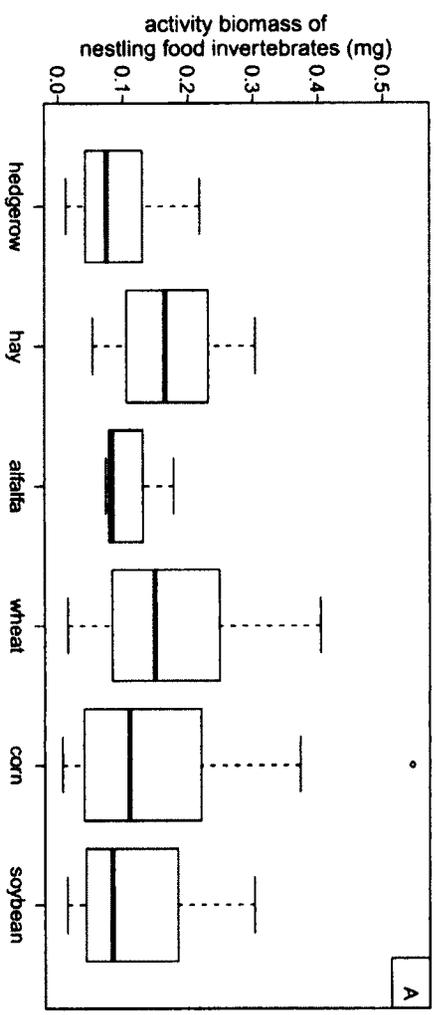
about 10-fold, between approximately 20 – 200 items (Fig 2 in Hart et al. 2006). Brickle and Peach (2004) used a food availability index of weighted density of invertebrates in Reed Bunting (*Emberiza schoeniclus*) foraging habitats, measured with a sweep net. Their measure varied by about 8-fold, between 2 and 16 (Fig 3 in Brickle and Peach 2004). My data show a variation of about 15 times in nestling food availability measured with a D-Vac (0.002-0.030 mg, Appendix U) and about 23 times in nestling food availability measured with pitfall traps (0.02-0.46 mg, Appendix S). Therefore, I believe that the variability in nestling food availability that I have measured was large enough to test whether food availability affects nestling mass in my study area.

I have previously shown that organic soybean fields in my study area support higher activity biomass of nestling food invertebrates than conventional soybean fields (Fig. 2.3, Appendix F). This measure is directly comparable to the activity biomass of nestling food invertebrates recorded here, as the same methods were used in both studies. The variability in activity biomass between the organic and conventional fields is similar in magnitude to the variability in activity biomass reported here (Appendix S). This suggests that, at least for Song Sparrows, the difference in invertebrate abundance between organic and conventional soybean fields in my study area is not large enough to produce a measureable difference in nestling mass or number of fledglings.

I found no evidence that food availability or semi-natural cover affects nestling mass, number of fledglings or nestling survival, or that semi-natural

cover affects nest survival in Song Sparrows nesting in farmland in eastern Ontario. It is possible that adults are compensating for low food availability with increased effort. However, I suggest that the relatively low input of insecticides in the corn and soybean that dominate the study area, means that farmland in eastern Ontario can provide enough food for successful reproduction in Song Sparrows, and potentially other generalist songbirds.

Fig. 5.1 Food availability among foraging habitats close to Song Sparrow nests, measured as a) activity biomass of nestling food invertebrates measured with pitfall traps (not significantly different among habitats), b) mean dry biomass of nestling food invertebrates in the vegetation, measured with a D-Vac, c) mean dry biomass of hoppers and caterpillars in the vegetation, measured with a D-Vac. Where there are significant differences among habitats, habitats that are significantly different are marked with different letters (Nemenyi-Damico-Wolfe-Dunn post-hoc test).



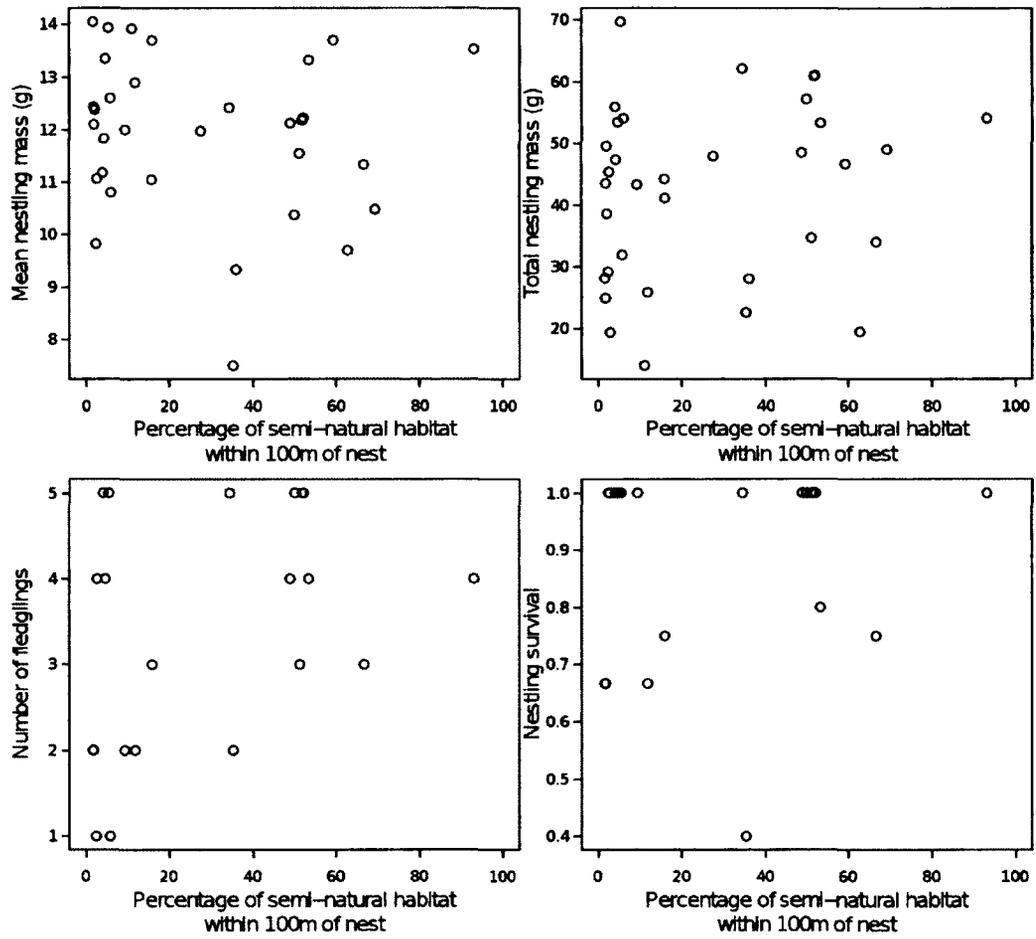


Fig. 5.2: Biplots showing percent cover of semi-natural vegetation within 100 m of Song Sparrow nests with measures of nestling size and survival.

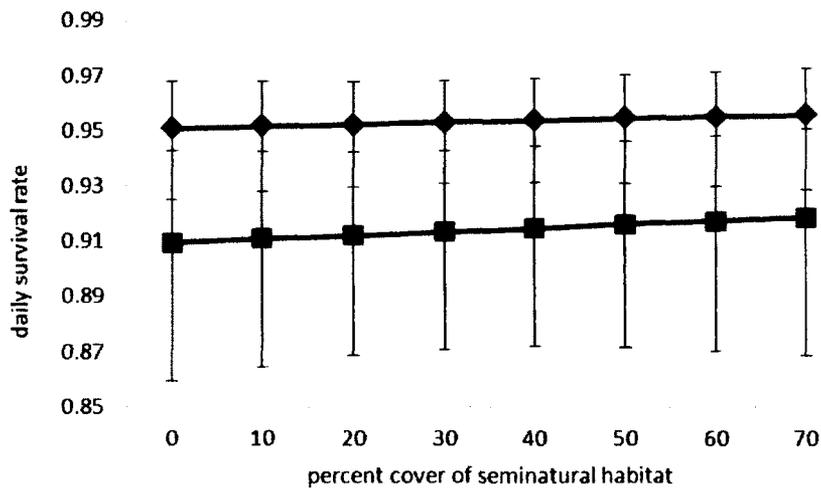


Fig. 5.3: Model averaged estimate of daily survival rate of Song Sparrow nests in the egg stage (diamonds) and the nestling stage (squares) with 95% confidence intervals. All other variables are held constant (continuous variables held at midpoint, brood parasitism = 0, year = 2007).

Table 5.1: Spearman's rank correlations of activity biomass of nestling food invertebrates and semi-natural vegetation cover with nestling size, number of Song Sparrow fledglings and Song Sparrow nestling survival.

	Activity biomass of nestling food invertebrates										
	Activity biomass of nestling food invertebrates			Activity biomass of invertebrates weighted by diet source ^a				Semi-natural cover within 100 m radius of nest			
	ρ^b	S ^c	P	ρ^b	S ^c	P	N ^d	ρ^b	S ^c	P	N ^e
Mean nestling mass	-0.03	3373	0.88	0.12	2882	0.55	27	-0.15	6291	0.40	34
Total nestling mass	-0.05	4274	0.79	-0.02	4130	0.93	29	0.22	5096	0.21	34
Number of fledglings	-0.09	1241	0.72	-0.07	1216	0.79	19	0.37	976	0.10	21
Nestling survival	0.01	1127	0.96	0.00	1140	1.00	19	0.13	1155	0.56	20

^a Food availability by diet source is weighted by the proportion of nestling food predicted to come from cropped (39%) and semi-natural habitats (61%), using a stable isotope mixing model of Song Sparrow nestling diet. ^b correlation

coefficient ^c test statistic ^d number of nests included in food availability analysis. Two nests which contained only Brown-headed Cowbird nestlings were excluded from the mean nestling mass analysis. Only nests which survived to fledge at least one Song Sparrow were included in analysis of number of fledglings and nestling survival. ^e number of nests included in landscape cover analysis. Only nests which survived to fledge at least one Song Sparrow were included in analysis of number of fledglings and nestling survival. One nest which fledged one young, but for which clutch size is unknown was included in the number of fledglings analysis, but not the nestling survival analysis.

Table 5.2: Relationship between food availability in the vegetation, measured with a D-Vac suction sampler, and Song Sparrow nestling size, number of Song Sparrow fledglings and Song Sparrow nestling survival. Food availability by diet source is weighted by the proportion of nestling food predicted to come from cropped and semi-natural habitats using a stable isotope mixing model of Song Sparrow nestling diet.

	Nestling food						Hopper and caterpillar						N ^d
	Nestling food			availability weighted			Hopper and			availability weighted			
	availability			by diet source ^a			caterpillar availability			by diet source ^a			
	ρ ^b	S ^c	P	ρ ^b	S ^c	P	ρ ^b	S ^c	P	ρ ^b	S ^c	P	
Mean nestling mass	0.13	710	0.62	0.09	739	0.72	-0.09	892	0.72	-0.17	951	0.53	17
Total nestling mass	-0.12	912	0.65	0.40	486	0.11	0.03	794	0.92	0.11	726	0.67	17
Number of fledglings	-0.07	236	0.84	0.39	134	0.23	0.05	210	0.89	0.10	197	0.76	14
Nestling survival	-0.32	290	0.34	0.44	124	0.18	0.07	205	0.84	0.26	162	0.43	14

^a Food availability by diet source is weighted by the proportion of nestling food predicted to come from cropped (39%) and semi-natural habitats (61%), using a stable isotope mixing model of Song Sparrow nestling diet. ^b correlation coefficient ^c

test statistic ^d Number of nests included in analysis. Only nests which survived to fledge at least one Song Sparrow were included in analysis of number of fledglings and nestling survival.

Table 5.3: AIC_c rankings of top models ($\Delta AIC_c < 4$) that predict daily nest survival of Song Sparrows, together with constant model and global model. For a full list of models with AIC_c values see Appendix Y.

Model	Number of parameters	AIC _c ^a	ΔAIC_c ^b	w_i ^c
Nest stage	2	351.202	0.000	0.225
Nest stage + semi-natural cover	3	352.320	1.118	0.129
Nest stage + date	3	352.976	1.774	0.093
Nest stage + brood parasitism	3	353.156	1.954	0.085
Nest stage + semi-natural cover + date	4	353.914	2.712	0.058
Nest stage + semi-natural cover + brood parasitism	4	354.326	3.123	0.047
Nest stage + date + date ²	4	354.774	3.572	0.038
Nest stage + year	4	354.821	3.618	0.037
Nest stage + date + brood parasitism	4	354.917	3.714	0.035
Nest stage * brood parasitism	4	355.133	3.930	0.032

Table 5.3 continued

Model	Number of parameters	AIC _c ^a	ΔAIC _c ^b	w _i ^c
Nest stage + semi-natural cover+ date + date ²	5	355.666	4.464	0.024
Constant	1	356.660	5.458	0.015
Global model	8	361.222	10.019	0.002

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^a Akaike's Information Criterion, adjusted for small samples sizes

^b Change in AIC_c relative to model with lowest AIC_c

^c Akaike weight (relative likelihood of each model /sum of relative likelihoods of all models)

Table 5.4: Model averaged parameter estimates, together with unconditional standard errors and confidence intervals, for all 51 candidate models of daily survival rate of Song Sparrow nests. Only variables from the candidate model set ($\Delta AIC_c < 2$) are included. The effective sample size was 1131 nest days at 116 nests.

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Variable	Estimated importance ^a	Estimated coefficient	SE	Upper confidence interval	Lower confidence interval
Intercept		2.369	0.284	2.925	1.812
Nest stage	0.947	0.720	0.255	1.220	0.219
Semi-natural cover	0.355	0.122	0.128	0.373	-0.129
Date	0.256	0.166	0.321	0.795	-0.462
Brood parasitism	0.292	0.076	0.396	0.853	-0.701

^a Summed Akaike weights across all models in which variable occurred.

Chapter 6: General discussion

Farmland in Canada supports a unique assemblage of birds. Few of these birds are rare or endangered. Nevertheless, in regions where agriculture is a dominant land use, it can play an important role in maintaining populations of common bird species. A better understanding of how birds use farmland, and how management practices and farmland structure affect birds, will help us move towards the goal of maintaining both agricultural productivity and farmland bird populations.

The objective of this thesis was to examine how food supply and foraging habitat availability affect farmland birds in eastern Ontario. I have shown that organic agriculture does provide more food for breeding birds than conventional agriculture, as previously hypothesized (Freemark and Kirk 2001, Beecher et al. 2002). I have also developed a new stable isotope-based approach for studying foraging habitat and diet of farmland birds, and used it to demonstrate the importance of hoppers and caterpillars to Song Sparrow nestling diets, as well as the importance of semi-natural habitat, especially hedgerows, to Song Sparrows foraging for nestlings. Finally, I have shown that neither food supply, nor cover of semi-natural habitat, has a measurable effect on Song Sparrow reproductive success in farmland. I believe my findings suggest that while conventional farming methods do reduce food supply for birds in farmland compared to organic methods, farming methods in eastern Ontario are relatively benign for Song Sparrows (and potentially other birds that nest and forage in hedgerows), because Song Sparrows can maintain nest survival and number of fledglings

produced, even when nesting in hedgerows completely surrounded by crop fields.

Food supply in organic agriculture

This is the first North American study to examine whether the abundance of invertebrates important to nestlings is higher in organic agriculture than in conventional agriculture, and the first study I know of to examine this question in soybean, a globally important crop. I showed that organic soybean fields and adjacent hedgerows supported higher chick food biomass than conventional soybean fields, and that this difference was most significant within the crop fields. The increased food abundance in organic fields provides a possible explanation for increased richness and abundance of birds found in organic fields in Ontario (Freemark and Kirk 2001, Kirk et al. 2011) and elsewhere in North America (Beecher et al. 2002). This may be especially important for birds that forage primarily within the crop fields, where differences in food abundance were strongest.

However, I also found that differences in food availability comparable with the differences between organic and conventional soybean fields did not affect the mass of Song Sparrow nestlings or the number of fledglings produced from Song Sparrow nests (Chapter 5). This suggests that the increased food supply in organic farms will not directly affect these aspects of reproductive success, at least for Song Sparrows. Few studies have examined whether organic farming improves reproductive success. Bradbury et al (2000) found no significant

difference in reproductive success for Yellowhammers (*Emberiza citrinella*) nesting in hedgerows adjacent to organic and conventional fields in the UK. However, Yellowhammers did tend to start breeding earlier in organic farmland than conventional farmland, which can allow more breeding attempts during the nesting season, leading to increased annual reproductive success. Lokemon and Beiser (1997) studied birds nesting in organic, minimum-tillage and conventional fields in North Dakota and showed no difference in nest success. Similarly Wilson et al. (1997) studying Skylarks (*Alauda arvensis*), and Kragten and de Snoo (2007) studying Lapwings (*Vanellus vanellus*), found no difference in nest success between organic and conventional farms. However, because all three of these studies were based on nests built within crop fields, the major cause of failure was destruction of nests by farm machinery, so the role of food supply may be less important in these cases. In addition, both Skylarks and Lapwings are precocial, meaning that the chicks leave the nest soon after hatching, so estimates of nest survival do not include feeding of the nestlings and may under-emphasize the role of food supply in reproductive success. Further study is needed to understand whether increased food abundance in organic farmland directly increases reproductive success of breeding birds. However, even if increased food abundance does not directly affect chick mass or nest survival, it may still benefit birds nesting in organic farmland through increased fledgling or chick survival, or increased adult survival during the breeding season.

Diet and foraging habitats of Song Sparrows

The stable isotope analysis I used in chapter 4 allowed me to quantitatively assess the relative importance of different invertebrates to nestling diet, and to assess the importance of different habitats to Song Sparrows foraging for nestlings. Stable isotope analysis is advantageous because it provides information on the food sources that have been incorporated into the animal's tissue (Hobson and Clark 1992; Hobson and Bairlein 2003; Pearson et al. 2003; Bearhop et al. 2003), compared to traditional methods which give a snapshot of food items fed to the nestling, and can be biased by differences in digestibility among food items.

To use a stable isotope mixing model to quantify relative importance of food sources or foraging habitats, the food sources and habitats must have different isotopic signatures (Gannes et al. 1997, Phillips and Greg 2001). In chapter 3, I showed that carbon and nitrogen stable isotopes could be used to distinguish between invertebrates captured in crop fields and those captured in hedgerows and hayfields. This is the first published study I know of to report $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values of invertebrates captured in hedgerows, and one of the first to report $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values of invertebrates captured in soybean fields. A better understanding of how birds use crop fields for foraging will help inform studies of pesticide exposure, and crop pest predation by birds (e.g. Tremblay et al. 2001). As well as being useful in studying ecology of farmland birds, this finding will also inform studies in movement and food webs of invertebrates in farmland (e.g. Prasifka et al. 2004, Prasifka and Heinz 2004, Schallhart et al. 2009).

I showed that hoppers and caterpillars were the most important food sources for Song Sparrow nestlings. Knowing which invertebrates are most important to farmland breeding birds will not only inform studies of food supply for farmland birds (e.g. Green 1984, Chapter 4), but will also help us to understand how crop types and farming practices, especially pesticides, affect invertebrates that are most important to breeding birds (e.g. Westbury et al. 2011, Hoste-Danylow et al. 2010).

In chapter 4 I found that Song Sparrows obtain a higher proportion of nestling food from semi-natural habitats, and a lower proportion of food from crop fields than expected based on habitat availability. Hedgerow habitat in particular seems to be important to nesting Song Sparrows; even at nests with very low semi-natural cover (<7%), where the only non-crop habitat within 100 m of the nest was hedgerow, Song Sparrows still obtained an average of 57% of nestling diet from semi-natural habitat sources. The role of hedgerows in providing nesting habitat, and protection from predators has been much studied in the UK (reviewed by Hinsley and Bellamy 2000), and work in North America has also suggested that hedgerows provide important bird breeding habitat in farmland (reviewed by Mineau and McLaughlin 1996, Jobin et al. 2001). However, the role of hedgerows in providing foraging habitat has not been well studied, although hedgerows have often been found to support higher invertebrate abundances than adjacent fields (e.g. Thomas et al. 1991; Thomas and Marshall 1999; Varchola and Dunn 2001; Meek et al. 2002; Brickle and Peach 2004; Grübler et al. 2008, Chapter 5). My work expands previous work on the importance of

hedgerows to ecological interactions in farmland in North America (Shalaway 1985, Best et al. 1995, reviewed by Mineau and McLaughlin 1996, Jobin et al. 2001, Maisonneuve et al. 2001, Varchola and Dunn 2001, Silva and Prince 2008, Kirk et al. 2011) by providing evidence for the role of hedgerows as foraging habitat as well as nesting habitat for farmland birds. This is actually an advantage to farm managers, if hedgerows can be managed to provide both nesting and foraging habitat for birds, with minimal change to farm operations in the adjacent fields. However, hedgerows continue to be removed from farmland in eastern Canada, despite their ecological and agronomic benefits (Boutin et al. 2001, personal observation). This trend will not be reversed, unless either farmers come to understand the value of hedgerows through education, or incentive programs or some other form of recognition is put in place to compensate farmers for retaining existing hedgerows and encouraging creation of new hedgerows or shelterbelts.

Food supply and reproductive success

I found that neither food availability nor availability of semi-natural foraging habitat affected chick size or the number of fledglings in Song Sparrows. These variables have been shown to be sensitive to food availability in other sparrows in farmland (Brickle et al. 2000, Morris et al. 2005, Hart et al. 2006). In addition, nest survival was not affected by the availability of semi-natural foraging habitat close to the nest. Zquette et al. (2006B) found that supplemental food increases nest survival in Song Sparrows, presumably because fed adults spend less time

foraging, and more time attending and defending the nest (Rastogi et al. 2006). This suggests that even when nests are surrounded by crop fields, Song Sparrows can still find enough food to maintain nest survival. I did not measure seasonal productivity of Song Sparrows, because we were not able to follow Song Sparrow pairs through the breeding season. In addition, attempts to radio-track fledglings were unsuccessful, so I do not have estimates of fledgling survival. Therefore it is possible that food availability affects annual productivity of Song Sparrows, through reductions in the number of breeding attempts or reductions in fledgling survival. However, based on my results, I have not found a negative impact of food availability or foraging habitat on farmland nesting Song Sparrows. This suggests that current agricultural practices in eastern Ontario are not detrimental to the reproductive success of Song Sparrows, and that availability of hedgerows which provide foraging and nesting habitat are more likely than food supply to drive population trends of farmland Song Sparrows in eastern Ontario.

In chapter 5, I argue that the most likely reason for a lack of effect of food supply and foraging habitat on Song Sparrow reproductive success is the relatively low rate of insecticide use in the study region. If this is correct, then my results can only be extended to regions of higher insecticide use, such as southern Ontario, with caution. It is possible that even at levels of higher insecticide use, Song Sparrows could still find enough invertebrate food in hedgerows to support levels of reproductive success, but further study is needed to confirm this. However, organic cropping systems are likely to become even

more important in providing foraging and breeding habitat in regions with higher insecticide use.

I suggest that my finding that availability of food and foraging habitat do not affect reproductive success of Song Sparrows in this region can probably be extended to other farmland birds that breed in hedgerows and other field margins. Birds such as American Robins (*Turdus migratorius*, Sallabanks and James 1999), Brown Thrashers (*Toxostoma rufum*, Cavitt and Haas 2000), Savannah Sparrows (*Passerculus sandwichensis*, Wheelwright and Rising 2008), Common Grackles (*Quiscalus quiscula*, Peer and Bollinger 1997) and Gray Catbirds (*Dumetella carolinensis*, Smith et al. 2011) eat primarily invertebrates during the season and forage on the ground and in hedgerows (personal observation, Jobin et al. 2001). Further research into use of hedgerows and other field border types as foraging habitat, including which types of border support the highest abundances of invertebrates and of foraging birds, and how management actions such as mowing and removing overhanging vegetation affects bird use and invertebrate abundance is needed to clarify the importance of hedgerows as foraging to farmland breeding birds.

Management recommendations

While hedgerows will not benefit all farmland birds (e.g. Bobolinks, *Dolichonyx oryzivorus*, avoid hedgerows, Bollinger and Gavin 2004), their importance to the birds that use them for breeding and foraging is clear (Chapter 4, Chapter 5 Shalaway 1985, Best et al. 1995, reviewed by Mineau and

McLaughlin 1996, Jobin et al. 2001, Maisonneuve et al. 2001, Varchola and Dunn 2001, Silva and Prince 2008, Kirk et al. 2011). Simply recommending hedgerow conservation however, is not enough. To conserve hedgerows in eastern Ontario, and other agricultural regions, it will be necessary to work with farmers to understand why farmers remove hedgerows and how this can be avoided, to educate farmers and landowners on the value of hedgerows and, where necessary, to develop compensation schemes to encourage farmers to maintain and perhaps replant hedgerows or shelterbelts. In addition, research is needed into which types or structures of hedgerows are most important in providing foraging and nesting habitat, as well as to investigate optimal hedgerow densities that promote abundance and richness of farmland birds.

If insecticide use increases in eastern Ontario, there is a danger that invertebrate abundance will be reduced to a level that will no longer support farmland bird reproductive success. Maintaining low levels of insecticide inputs is the best option for maintaining bird populations in eastern Ontario. As the effects of climate change increase, it is likely that eastern Ontario will face increased pressure from crop pests. Working with farmers to promote integrated pest management and the use of narrow-spectrum insecticides wherever possible will help to minimize potential negative effects of increased insecticide use.

Conclusion

Although I have shown that conventional cropping systems reduce food abundance for farmland breeding birds compared to organic cropping systems, I conclude that current farmland management in eastern Ontario is fairly benign for Song Sparrows, and potentially other hedgerow-breeding birds, due to low insecticide inputs. Therefore, as for grassland breeding birds (Hartley 2007), eastern Ontario has the potential to support regional populations of farmland birds associated with hedgerows. However, this will only continue if we can work together with farmers to achieve hedgerow conservation and continued low insecticide inputs in farmland.

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Appendix A: Invertebrate taxa identified from pitfall traps in organic and conventional soybean fields in eastern Ontario. Taxa marked with an asterisk were excluded when calculating the nestling food index, which includes potential Song Sparrow nestling food, and excludes taxa associated with aquatic habitats.

Group	Class or Order	Suborder or Family	Common Name
Insects	Archaeognatha		jumping bristletails
	Coleoptera	Cantharidae	soldier beetles
		Carabidae	ground beetles
		Cleridae	checkered beetles
			predaceous diving
			beetles
		Dytiscidae *	click beetles
		Elateridae	lady beetles
		Coccinellidae	fireflies
		Lampyridae	leaf beetles
		Chrysomelidae	plate-thigh beetles
		Eucinetidae	pill beetles
		Byrrhidae	parasitic flat back
			beetles
		Passandridae	tumbling flower beetles
		Mordellidae	rove beetles
		Staphylinidae	sap beetles
		Nitidulidae	scarab beetles
		Scarabaeidae	carrion beetles
		Silphidae	snout beetles
		Curculionidae	tiger beetles
		Cicindelinae	beetle larvae
			unidentified beetles
	Diptera	Brachycera	
	Diptera	Nematocera	
	Hemiptera	Auchenorrhyncha	hoppers etc.
	Hemiptera	heteroptera	typical bugs
	Hemiptera	Sternorrhyncha*	aphids etc.
	Hymenoptera	Formicidae *	ants
	Hymenoptera	other Hymenoptera *	sawflies, wasps, bees
	Lepidoptera	Lepidoptera adults	butterflies and moths
	Lepidoptera	Lepidoptera larvae	caterpillars
	Orthoptera	Caelifera	grasshoppers etc.
	Orthoptera	Ensifera	crickets etc.
	Unknown		unidentified larvae
	Unknown		unidentified insects

Appendix A: Continued

Group	Class or Order	Suborder or Family	Common Name
Other invertebrates			
	Acari *		mites and ticks
	Araneae		true spiders
	Chilopoda		centipedes
	Collembola		springtails
	Diplopoda		millipede
	Gastropoda *		slugs
			snails
	Isopoda		pillbugs
	Oligochaeta *		earthworms
	Opiliones		harvest spiders

Appendix B: Number of pitfall traps recovered at each site.

Habitat	Pair	Type	Month	Number of traps used in analysis
crop	1	conventional	June	6
crop	1	conventional	July	8
crop	1	organic	June	0
crop	1	organic	July	7
crop	2	conventional	June	7
crop	2	conventional	July	7
crop	2	organic	June	7
crop	2	organic	July	8
crop	3	conventional	June	6
crop	3	conventional	July	7
crop	3	organic	June	8
crop	3	organic	July	7
crop	4	conventional	June	7
crop	4	conventional	July	8
crop	4	organic	June	6
crop	4	organic	July	6
crop	5	conventional	June	8
crop	5	conventional	July	8
crop	5	organic	June	8
crop	5	organic	July	6
crop	7	conventional	June	7
crop	7	conventional	July	8
crop	7	organic	June	5
crop	7	organic	July	8
crop	8	conventional	June	6
crop	8	conventional	July	7
crop	8	organic	June	8
crop	8	organic	July	6
crop	10	conventional	June	8
crop	10	conventional	July	7
crop	10	organic	June	6
crop	10	organic	July	6
crop	11	conventional	June	8
crop	11	conventional	July	6
crop	11	organic	June	1
crop	11	organic	July	7
edge	1	conventional	June	7
edge	1	conventional	July	6

Appendix B continued

Habitat	Pair	Type	Month	Number of traps used in analysis
edge	1	organic	June	3
edge	1	organic	July	8
edge	2	conventional	June	7
edge	2	conventional	July	8
edge	2	organic	June	7
edge	2	organic	July	8
edge	3	conventional	June	4
edge	3	conventional	July	3
edge	3	organic	June	7
edge	3	organic	July	7
edge	4	conventional	June	7
edge	4	conventional	July	7
edge	4	organic	June	7
edge	4	organic	July	7
edge	5	conventional	June	6
edge	5	conventional	July	8
edge	5	organic	June	8
edge	5	organic	July	5
edge	7	conventional	June	8
edge	7	conventional	July	7
edge	7	organic	June	8
edge	7	organic	July	8
edge	8	conventional	June	5
edge	8	conventional	July	6
edge	8	organic	June	8
edge	8	organic	July	8
edge	10	conventional	June	8
edge	10	conventional	July	5
edge	10	organic	June	3
edge	10	organic	July	8
edge	11	conventional	June	8
edge	11	conventional	July	7
edge	11	organic	June	3
edge	11	organic	July	0
hedge	1	conventional	June	5
hedge	1	conventional	July	7
hedge	1	organic	June	7
hedge	1	organic	July	7
hedge	2	conventional	June	8

Appendix B continued

Habitat	Pair	Type	Month	Number of traps used in analysis
hedge	2	conventional	July	8
hedge	2	organic	June	7
hedge	2	organic	July	8
hedge	3	conventional	June	8
hedge	3	conventional	July	5
hedge	3	organic	June	7
hedge	3	organic	July	8
hedge	4	conventional	June	5
hedge	4	conventional	July	8
hedge	4	organic	June	8
hedge	4	organic	July	7
hedge	5	conventional	June	8
hedge	5	conventional	July	7
hedge	5	organic	June	9
hedge	5	organic	July	6
hedge	7	conventional	June	7
hedge	7	conventional	July	7
hedge	7	organic	June	7
hedge	7	organic	July	7
hedge	8	conventional	June	7
hedge	8	conventional	July	7
hedge	8	organic	June	7
hedge	8	organic	July	8
hedge	10	conventional	June	6
hedge	10	conventional	July	7
hedge	10	organic	June	8
hedge	10	organic	July	7
hedge	11	conventional	June	7
hedge	11	conventional	July	7
hedge	11	organic	June	8
hedge	11	organic	July	0

Appendix C: Management practice questionnaire used to compare management of conventional and organic soybean fields.

Site Type	Category	Question
Organic only	Organic history	How many years has farm been organic?
Both	Planting date	What was the planting date of soybean in the study field?
Both	Fertilization	Was fertilizer applied to this field between harvest in 2008 and 2009? If so, when was it applied? What type of fertilizer? Amount of fertilizer (application rate)?
Both	Tillage	Was the field tilled between harvest in 2008 and 2009? What type of tillage was used? When was the field tilled? How deep did the machinery penetrate the soil?
Both	Seed treatment	Were treated seeds used, and if so what was the treatment?
Both	Pesticide use	Was any kind of pesticide applied to the field? If so, when was the application? What was applied? What was the application rate?
Both	Field history	What was the previous crop in the study field? Can you describe the usual crop rotation in the study field?

Appendix D: Responses to management questionnaire

Management Type	Pair	Planting date	Fertilizer Type	Dates applied	Spring till dates	method	depth
conventional	1	5/9/2009	none		before planting	cultivated	
conventional	2	5/15/2009	none		none	NA	NA
conventional	3	5/20/2009	none		5/10/2009	disked	3 inches
conventional	5	6/2/2009	none		5/31/2009	cultivated	2-3 inches
conventional	7	5/25/2009	none		5/23/2009	cultivated	4 inches
					5/24/2009	cultivated	4 inches
conventional	8	6/4/2009	unknown		before planting	combine	
				in spring before discing			
conventional	10	unknown	artificial		before planting	disked	
organic	1	6/5/2009	unknown		5/21/2009 several times before planting	ploughed cultivated	
organic	2	6/7/2009	none		none	NA	NA
organic	3	6/5/2010	none		5/24/2009 6/8/2009 6/24/2009 7/6/2009	ploughed in clover tine weed cultivated cultivated	

Appendix D continued

Management Type	Pair	Planting date	Fertilizer Type	Dates applied	Spring till dates	method	depth
organic	4	6/4/2009	GSR calcium. Spray: calcium diluted in water.	6/4/2009	twice before planting	rototilled twice before planting	6 inches
				6/25/2009	3 times during soybean development	tine weed	negligible
					1 week after last tine weed	row cultivator	1.5-2 inches
organic	5	6/4/2009	liquid calcium	fall and spring	several times before planting	cultivated	
organic	7	5/23/2009	none		before planting	disked, cultivated, disced, harrowed	
organic	8	5/18/2009	none		5/16/2009	disked	1-2 inches
					5/17/2009	cultivated	2 inches

Appendix D continued

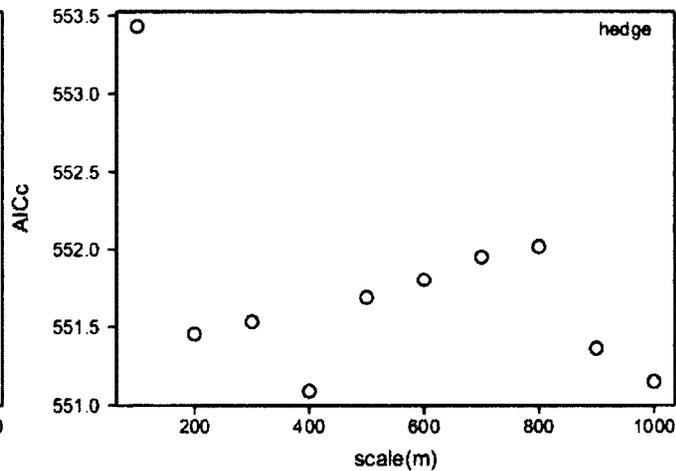
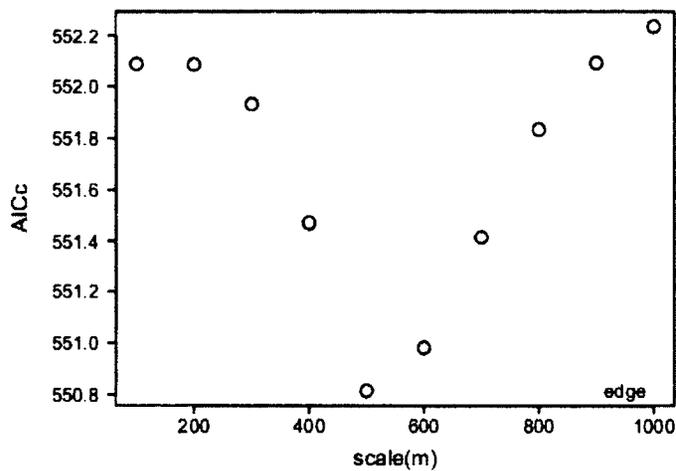
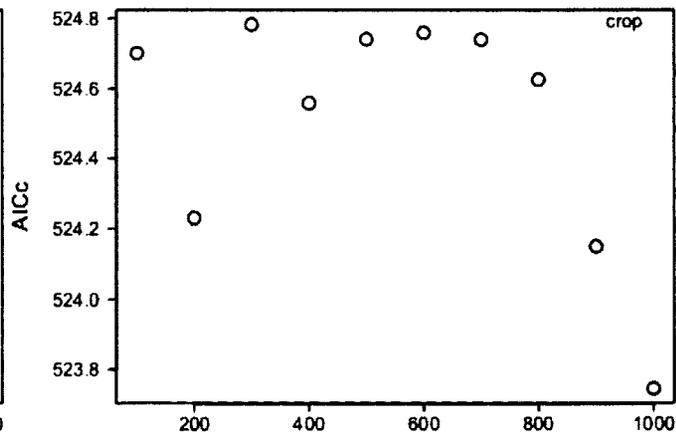
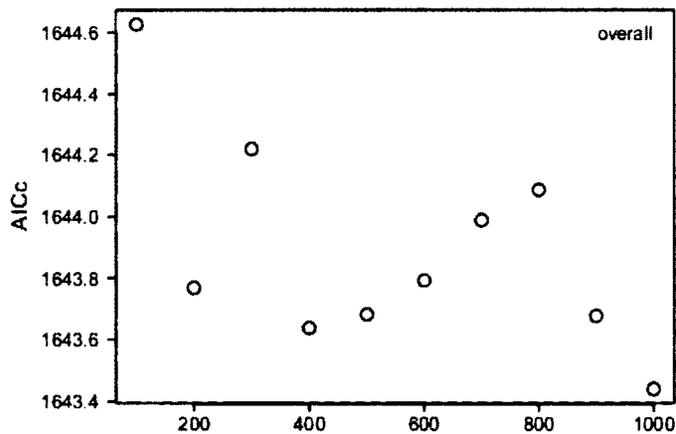
Management Type	Pair	Fall till dates	method	depth	Herbicide type	Dates applied	Quantity
conventional	1	fall 2008	heavy disc		round up	5/30/2009	
conventional	2	fall 2008	disc ripped		clean sweep 3 weeks after planting	5/5/2009	
conventional	3	fall 2008	ripped		Pursuit	June 18/19 2009	165 ml/acre
					Venture	June 18/19 2009	0.25 ml/acre
conventional	5				Round-up & Assure	6/24/2009	Round-up: 0.8 L/acre, Assure: "recommended rate"
conventional	7				Touchdown Total (for Round-up ready soybeans)	6/24/2009	1.4 L/acre
conventional	8				round up	30-35 days after planting	1.4L/acre
conventional	10				round up	3 weeks after planting	
organic	1				NA		
organic	2				NA		
organic	3				NA		
organic	4				NA		
organic	5	fall 2008	disc and chisel	6 inches	NA		

ploughed							
organic	7						NA

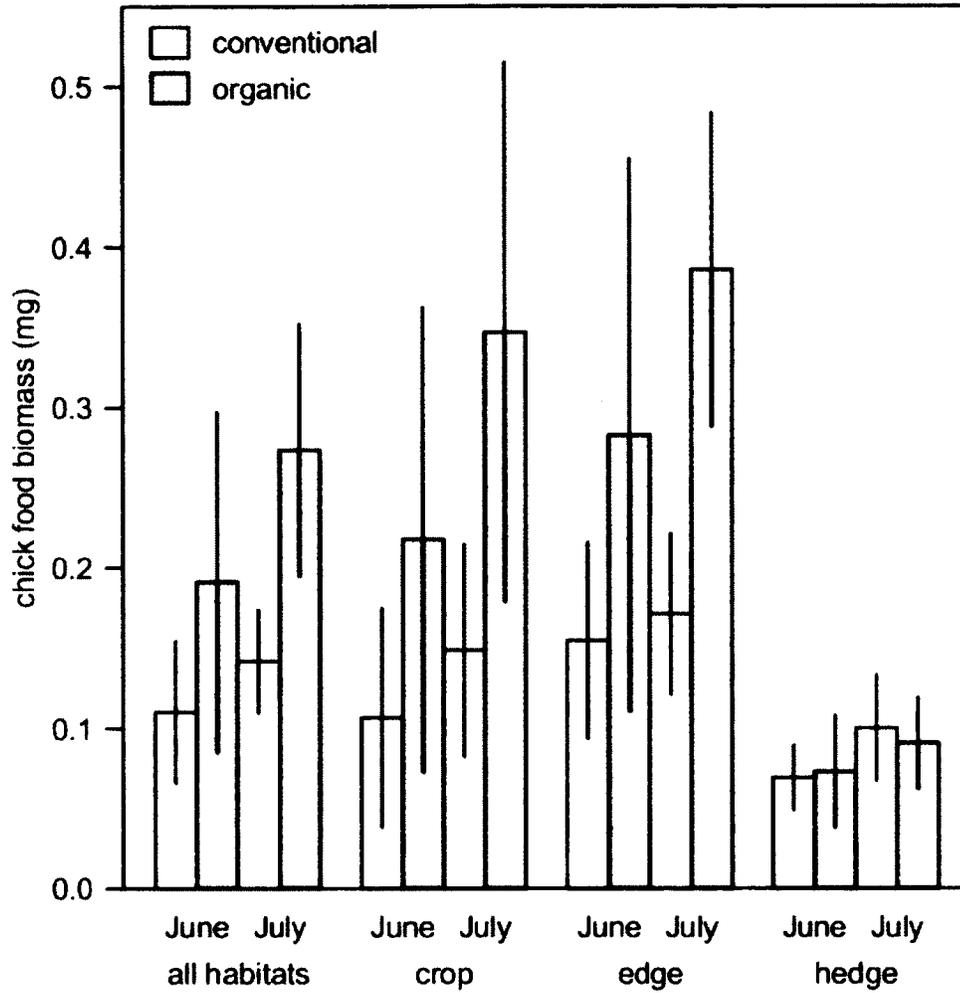
Appendix D continued

Management Type	Pair	Fall till dates	method	depth	Herbicide type	Dates applied	Quantity
organic	8				NA		
organic	10	fall 2008	ploughed		NA		

Appendix E: Graphs showing scale of effect tests for chick food biomass measured overall (in all 3 habitats), soybean crop fields, soybean field edges and hedgerows adjacent to soybean fields. Relationship between chick food biomass is regressed against crop cover at each scale, and the scale with the lowest AIC_c is selected as the scale at which crop cover has the largest effect on chick food biomass.



Appendix F: Mean chick food biomass in soybean fields in June and July. Data are averaged across 9 conventional and 9 organic fields. Bars show means \pm SE



Appendix G: Sample sizes of each taxon in each trapping location. Where manure status is not given (corn – other or hay – other), we were not able to obtain enough information from the farmer to determine the fertilization regimen of the field, and the field was excluded from the manure analysis.

Taxon	Trapping location	n
caterpillars	corn	3
caterpillars	hay	4
caterpillars	hedge	28
crickets	corn - manured	3
crickets	corn - not manured	2
crickets	corn - other	1
crickets	hay - not manured	2
crickets	hedge	14
crickets	soy	4
ground beetles	corn - manured	25
ground beetles	corn - not manured	10
ground beetles	corn - other	3
ground beetles	corn edge	33
ground beetles	hay - manured	6
ground beetles	hay - not manured	6
ground beetles	hay - other	6
ground beetles	hay edge	14
ground beetles	hedge	36
ground beetles	soy	13
ground beetles	soy edge	15
hoppers	corn - manured	4
hoppers	corn - not manured	2
hoppers	corn - other	1
hoppers	hay - manured	3
hoppers	hay - not manured	4
hoppers	hay - other	3
hoppers	hedge	36
hoppers	soy	2
spiders	corn - manured	23
spiders	corn - not manured	7
spiders	corn - other	3
spiders	corn edge	31
spiders	hay - manured	4
spiders	hay - not manured	6

Appendix G continued:

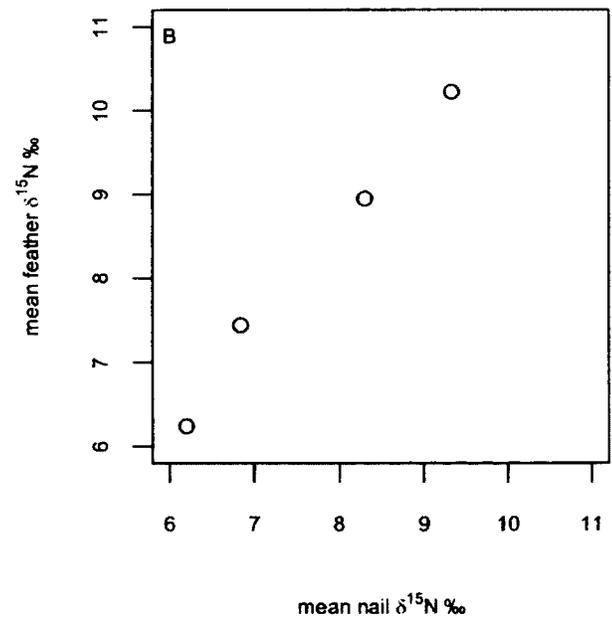
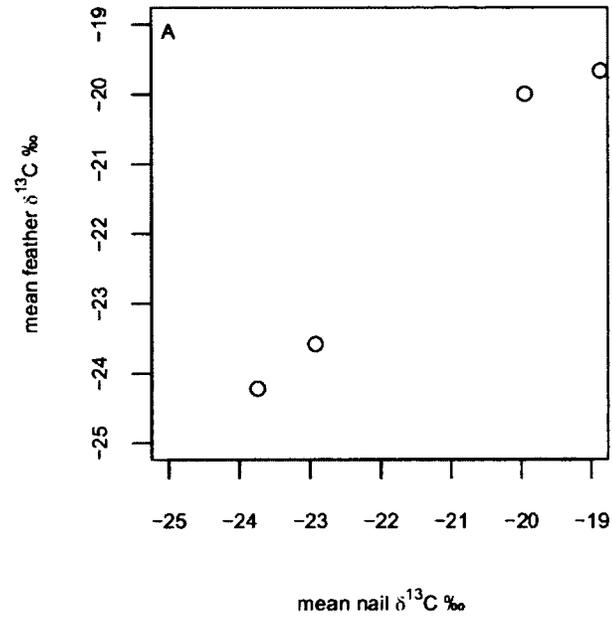
Taxon	Trapping location	n
spiders	hay - other	5
spiders	hay edge	15
spiders	hedge	49
spiders	soy	13
spiders	soy edge	15

Appendix H: Isotope values of food sources used in mixing model to predict diet of nestling Song Sparrows (*Melospiza melodia*).

Food source	Mean $\delta^{13}\text{C}$ ‰ (SD)	Mean $\delta^{15}\text{N}$ ‰ (SD)	Mean C:N (SD)	<i>n</i> *
Semi-natural habitats				
Beetle larvae	-24.6 (1.2)	6.7 (2.6)	3.9 (0.3)	8
Caterpillars	-26.9 (1.2)	3.5 (1.2)	4.3 (0.6)	20
Crickets	-24.9 (3.5)	4.7 (1.2)	3.7 (0.5)	12
Ground beetles	-23.4 (3.2)	7.2 (2.0)	4.4 (0.4)	27
Hoppers	-26.1 (0.9)	2.7 (1.6)	4.1 (0.4)	22
Spiders	-23.8 (1.3)	8.7 (0.9)	3.7 (0.4)	27
Cropped habitats				
Crickets	-20.9 (3.3)	5.6 (1.6)	3.8 (0.9)	12
Ground beetles	-20.3 (1.8)	8.4 (1.5)	4.3 (0.3)	26
Hoppers	-24.4 (1.8)	3.2 (1.4)	3.9 (0.3)	6
Spiders	-20.6 (1.9)	10.0 (1.3)	3.6 (0.3)	25
All habitats				
Sap beetles	-12.9 (1.7)	8.0 (1.8)	4.7 (0.4)	30

* Where multiple samples of the same taxa were analysed from the same field or hedgerow section, these were averaged before the overall mean $\delta^{13}\text{C}$ or $\delta^{15}\text{N}$ was calculated. Therefore *n* represents the number of fields and hedgerow sections from which a taxa was sampled, not the number of samples analysed.

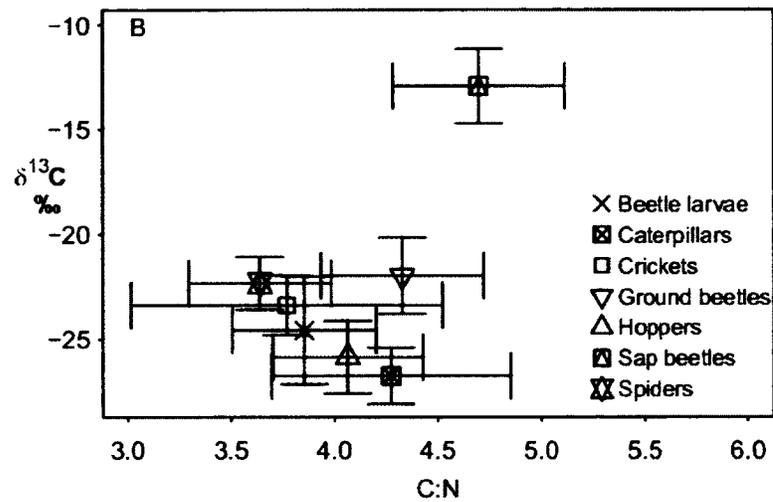
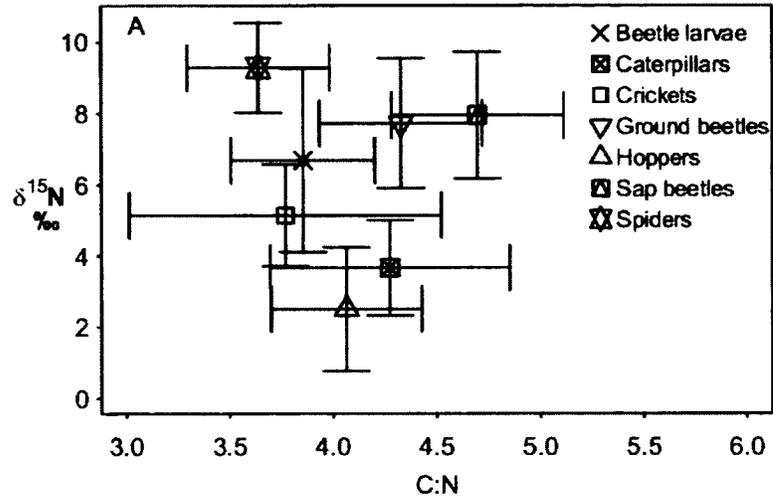
Appendix I: Correlation between a) mean claw $\delta^{13}\text{C}$ and mean feather $\delta^{13}\text{C}$ and b) mean claw $\delta^{15}\text{N}$ and mean feather $\delta^{15}\text{N}$ from four Song Sparrow (*Melospiza melodia*) nests.



Appendix J: Equations for predicting mean nest feather $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ from mean Song Sparrow (*Melospiza melodia*) claw $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$.

	Intercept (SE)	claw estimate (SE)
$\delta^{13}\text{C}$	-0.39 (2.11)	1.00 (1.00)
$\delta^{15}\text{N}$	-1.17 (0.67)	1.22 (0.09)

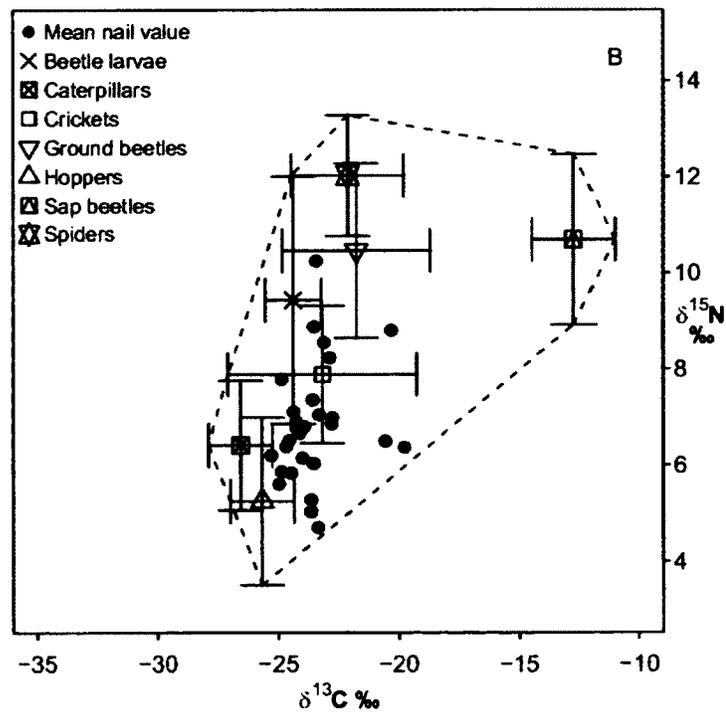
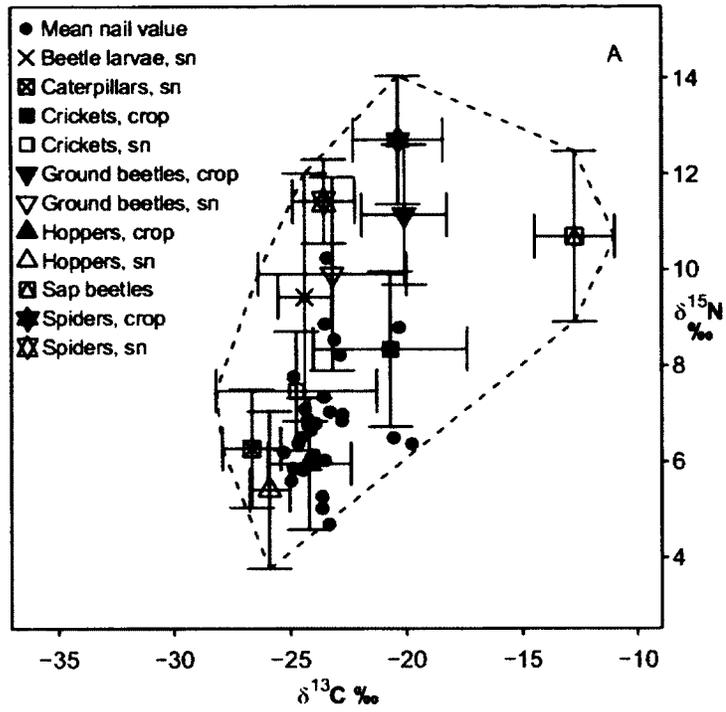
Appendix K: Biplot showing relationship between mean C:N ratios and a) mean $\delta^{15}\text{N}$ and b) mean $\delta^{13}\text{C}$ of invertebrate food sources (mean \pm 1 SD).



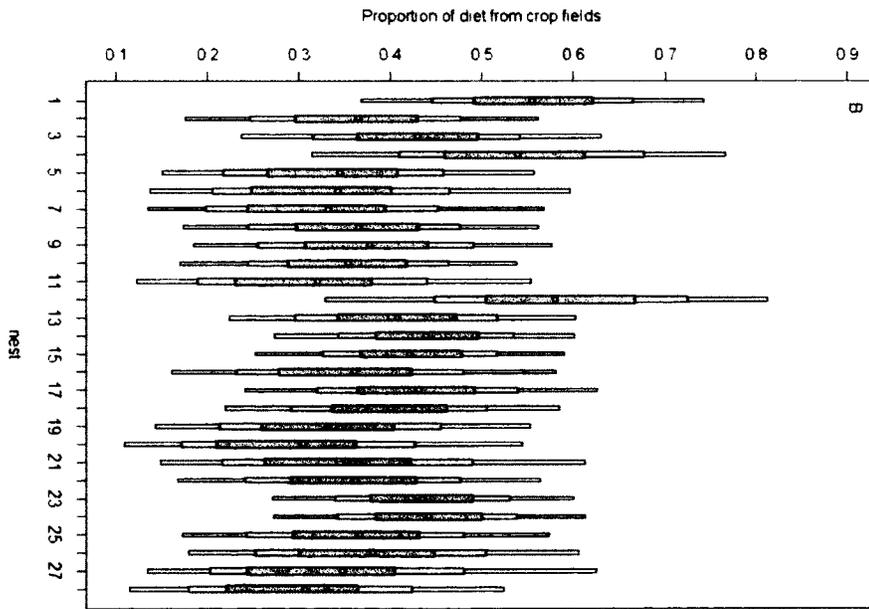
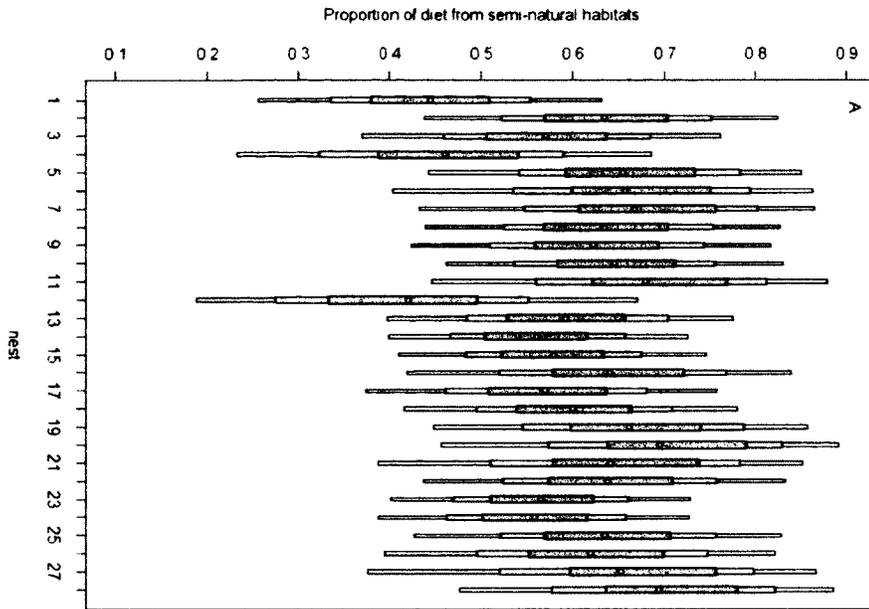
Appendix L: Mean $\delta^{13}\text{C}$ ‰ and $\delta^{15}\text{N}$ ‰ of claws taken from nestling Song Sparrows (*Melospiza melodia*) in 28 nests.

Nest	Mean $\delta^{13}\text{C}$ ‰ (SD)	Mean $\delta^{15}\text{N}$ ‰ (SD)	Number of chicks sampled
1	-19.8 (0.1)	8.1 (0.1)	3
2	-23.7 (0.2)	6.6 (0.1)	2
3	-22.3 (0.03)	6.6 (0.1)	2
4	-20.1 (0.3)	6.3 (0.2)	2
5	-24.0	6.3	1
6	-23.1 (0.2)	5.1 (0.3)	2
7	-24.0 (0.1)	5.7 (0.04)	2
8	-23.9 (0.2)	6.8 (0.1)	2
9	-23.4 (0.1)	6.5 (0.00)	2
10	-24.4 (0.01)	7.3 (0.04)	2
11	-24.4 (0.2)	5.7 (0.1)	2
12	-19.3 (0.6)	6.2 (0.1)	2
13	-22.8 (0.0)	6.7 (0.1)	2
14	-22.9 (0.3)	9.3 (0.1)	5
15	-23.0 (0.2)	8.2 (0.2)	5
16	-23.5 (0.2)	6.0 (0.2)	4
17	-22.3 (0.1)	6.7 (0.2)	4
18	-23.1 (0.1)	7.0 (0.1)	2
19	-24.2 (0.1)	6.2 (0.2)	2
20	-24.5 (0.1)	5.5 (0.2)	4
21	-23.1 (0.1)	5.3 (0.3)	5
22	-23.7 (0.2)	6.5 (0.3)	4
23	-22.6 (0.1)	7.9 (0.2)	2
24	-22.4 (0.2)	7.7 (0.2)	4
25	-23.6 (0.3)	6.4 (0.3)	3
26	-23.0 (0.2)	5.9 (0.4)	5
27	-22.9 (0.1)	4.8 (0.2)	3
28	-24.8 (0.1)	6.0 (0.2)	3

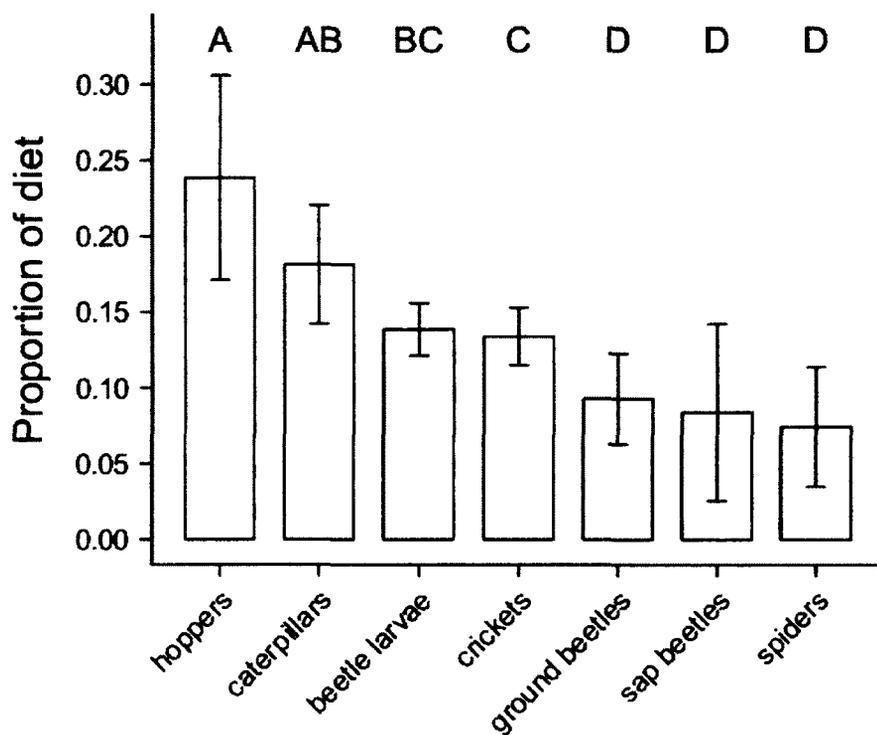
Appendix M: Predicted mean $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values for feathers of nestlings in 28 Song Sparrow (*Melospiza melodia*) nests, together with a) mean $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values (± 1 SD) of invertebrate food sources collected from adjacent crop fields (solid symbols) and semi-natural and forage habitats (sn, open symbols) and b) mean $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values of invertebrate food sources averaged over all habitats. Food sources have discrimination factors added to them, to account for the concentration of heavier isotopes at higher trophic levels ($\delta^{13}\text{C}$: 0.2 ‰, $\delta^{15}\text{N}$: 2.74 ‰, Kempster et al. 2007). The dotted line shows the mixing polygon bounded by the food sources entered into the mixing model.



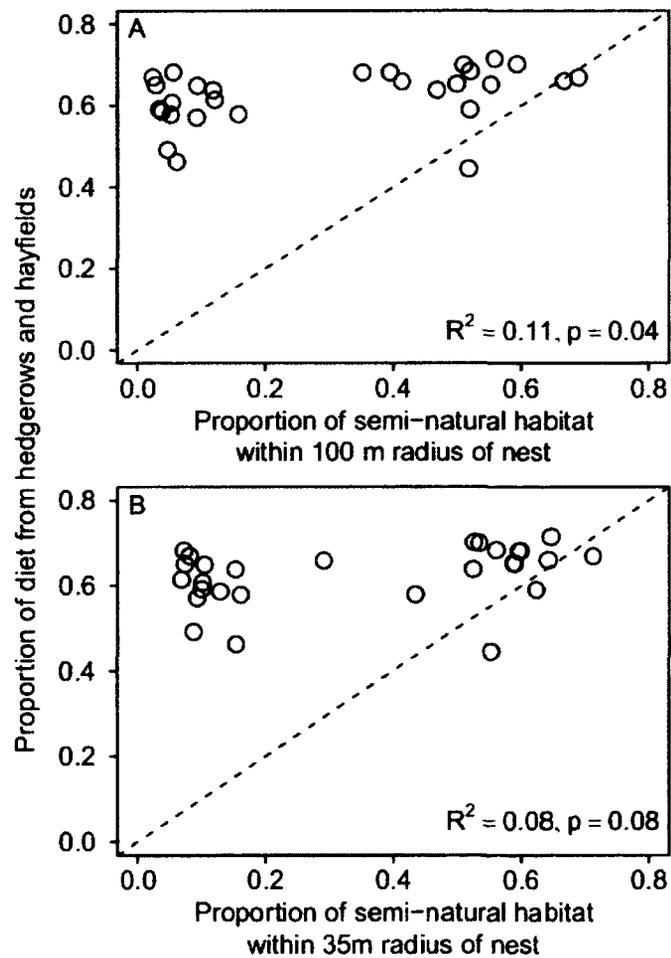
Appendix N: Predicted proportion of Song Sparrow (*Melospiza melodia*) nestling diet obtained from a) semi-natural habitats and b) cropped habitats near 28 nests. The median (white circle), 50% (dark grey), 75% (mid grey) and 95% (light grey) credibility intervals of the predicted contribution of each food source to the diet are shown. The 95% credibility interval is the portion of the distribution that has a 95% probability of containing the median.



Appendix O: Median estimated proportion of diet from invertebrate food sources for 28 Song Sparrow (*Melospiza melodia*) nests, generated from mixing model shown in figure S2b, which uses predicted feather $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values as estimate of Song Sparrow $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$. Error bars show ± 1 SD. Letters indicate significant differences at $p < 0.05$ (Kruskal-Wallis, $X^2_7 = 126.36$, $p < 0.001$).



Appendix P: Correlation between the proportion of semi-natural cover within a 100 m of Song Sparrow (*Melospiza melodia*) nests and the median proportion of the diet of nestlings estimated to come from semi-natural habitats. Data are, generated from mixing model shown in figure S2a, which uses predicted feather $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values as estimate of Song Sparrow $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$. The line shows the 1:1 relationship expected if nestling food is obtained from semi-natural habitat in proportion to local habitat availability.



Appendix Q: Invertebrate taxa identified from pitfall traps and D-Vac suction sampling at Song Sparrow nests in hedgerows in eastern Ontario. Taxa marked with an asterisk were excluded when calculating the nestling food index, which includes only potential nestling food.

Group	Class or Order	Suborder or Family	Common Name
Insects	Coleoptera		
		Anthicidae	ant-like flower beetles
		Bruchinae	bean weevil
		Cantharidae	soldier beetles
		Carabidae	ground beetles
		Cleridae	checkered beetles
		Corylophidae	minute fungus beetles
		Cryptophagidae	silken fungus beetles
		Dytiscidae *	predaceous diving beetles
		Elateridae	click beetles
		Eucnemidae	false click beetles
		Histeridae	hister beetles
		Hydrophilidae *	water scavenger beetles
		Coccinellidae	lady beetles
		Lampyridae	fireflies
		Latridiidae	minute scavenger beetles
		Chrysomelidae	leaf beetles
		Leiodidae	round fungus beetles
		Megalopodidae	
		Meloidae	blister beetles
		Mordellidae	tumbling flower beetles
		Orsodacnidae	
		Staphylinidae	rove beetles
		Nitidulidae	sap beetles
		Scarabaeidae	scarab beetles
		Scydmaenidae	ant-like stone beetles
		Silphidae	carrion beetles
		Silvanidae	silvanid flat bark beetles
		Curculionidae	snout beetles
		Cantharidae	soldier beetles

Appendix Q continued:

Group	Class or Order	Suborder or Family	Common Name
		Lucanidae	stag beetles
		Cicindelinae	tiger beetles
			beetle larvae
			unidentified beetles
	Dermaptera		earwigs
	Diptera	Brachycera	
	Diptera	Nematocera	
	Hemiptera	Auchenorrhyncha	hoppers etc.
	Hemiptera	heteroptera	typical bugs
	Hemiptera	Sternorrhyncha	aphids etc.
	Hymenoptera	Formicidae *	ants
	Hymenoptera	other Hymenoptera *	sawflies, wasps, bees
	Hymenoptera	Symphyta larvae	sawfly larvae
	Lepidoptera	Lepidoptera adults	butterflies and moths
	Lepidoptera	Lepidoptera larvae	caterpillars
	Mecoptera	Mecoptera *	scorpionfly
	Neuroptera	Neuroptera	net-winged insects
	Orthoptera	Caelifera	grasshoppers etc.
	Orthoptera	Ensifera	crickets etc.
		Siphonaptera *	
	Siphonaptera		fleas
	Unknown		unidentified larvae
	Unknown		unidentified insects
Other invertebrates			
	Acari *		mites and ticks
	Araneae		true spiders
	Chilopoda		centipedes
	Collembola		springtails
	Diplopoda		millipede
	Gastropoda *		slugs
			snails
	Isopoda		pillbugs
	Oligochaeta *		earthworms
	Opiliones		harvest spiders

Appendix R: Parameter estimates for regressions of invertebrate dry mass (mg) on length, measured in 2 mm intervals.

The equation is presented as $\ln(\text{mass}) = \ln(a) + b \cdot \ln(\text{length})$.

Group	Taxon	ln (a)	SE	b	SE	adjusted R ²	n	minimum size ^A (mm)	maximum size ^B (mm)
Insecta									
	Adult insects ^C	-10.07	0.23	2.25	0.10	0.78	128	0-2	26.1-28
	Adult beetles ^D	-9.35	0.32	2.10	0.14	0.77	67	0-2	26.1-28
	Larvae ^E	-11.53	0.39	2.24	0.14	0.88	36	2.1-4	46.1-48
Orthoptera									
	Caelifera (shorthorned grasshoppers)	-10.67	1.01	2.37	0.40	0.83	8	4.1-6	22.1-24
	Ensifera (crickets)	-11.61	0.23	2.73	0.11	0.99	7	2.1-4	14.1-16
Hemiptera									
	Heteroptera (true bugs)	-9.35	0.63	1.54	0.31	0.82	7	0-2	12.1-14
	Auchenorrhyncha (leafhoppers, treehoppers, planthoppers, spittlebugs)	-8.59	0.57	1.22	0.33	0.72	6	0-2	12.1-14
Lepidoptera									
	Lepidoptera larvae (caterpillars)	-12.73	0.52	2.60	0.17	0.92	22	2.1-4	46.1-48
Coleoptera									
	Coleoptera larvae	-11.00	1.58	1.99	0.70	0.59	7	2.1-4	14.1-16
	Carabidae (ground beetles)	-9.49	0.35	2.23	0.14	0.95	15	0-2	26.1-28

Appendix R Continued

Group	Taxon	ln (a)	SE	b	SE	adjusted R ²	n	minimum size ^A (mm)	maximum size ^B (mm)
	Staphylinidae (rove beetles)	-10.30	0.69	1.96	0.30	0.81	11	0-2	20.1-22
	Scarabaeidae (scarab beetles)	-10.65	0.47	2.89	0.19	0.97	8	4.1-6	22.1-24
	Elateridae (click beetles)	-9.93	0.26	2.29	0.12	0.98	7	2.1-4	16.1-18
	Chrysomelidae (leaf beetles)	-8.72	0.87	2.00	0.53	0.77	5	0-2	10.1-12
	Curculionidae (snout beetles)	-8.50	0.40	2.09	0.19	0.94	9	0-2	16.1-18
Diptera									
	Brachycera (deer flies, house flies and other short-horned flies)	-10.52	0.55	2.30	0.26	0.93	8	0-2	16.1-18
	Nematocera (midges, mosquitos and other long- horned flies)	-11.16	0.38	2.06	0.19	0.95	7	0-2	16.1-18
Diplopoda									
	Millipedes	-12.13	0.42	2.66	0.14	0.95	18	4.1-6	38.1-40
Chilopoda									
	Centipedes	-13.78	0.55	2.92	0.20	0.95	11	4.1-6	26.1-28
Isopoda									
	Pillbugs	-11.14	0.19	2.74	0.09	0.99	7	2.1-4	14.1-16
Araneae									
	True spiders	-10.03	0.35	2.59	0.19	0.97	7	0-2	12.1-14
Opiliones									
	Harvest spiders	-8.62	0.38	1.86	0.24	0.93	5	0-2	8.1-10

^A Minimum size of invertebrates used to obtain estimated dry weight; equations should not be used to predict biomass of invertebrates smaller than this. If invertebrates smaller than this were found in the samples, the most appropriate generalized equation was used to predict dry mass.

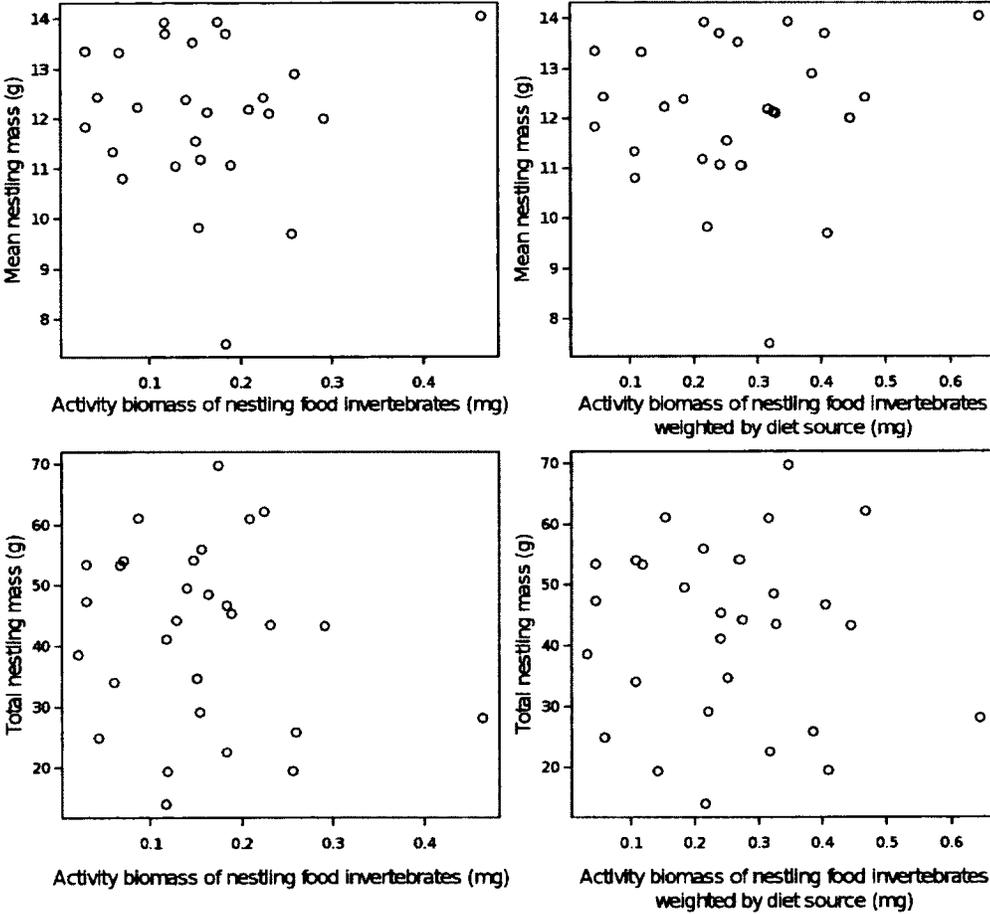
^B Maximum size of invertebrates used to obtain estimated dry weight; equations should not be used to predict biomass of invertebrates larger than this. If invertebrates larger than this were found in the samples, the most appropriate generalized equation was used to predict dry mass.

^C includes all adult insects listed below plus Formicidae (ants), Sternorrhyncha) aphids, Collembola (springtails), Lampyridae (fireflies), Coccinellidae (ladybird beetles), Nitidulidae (sap beetles), Cicindelidae (tiger beetles). Too few size categories of these additional groups were collected to generate taxon-specific equations.

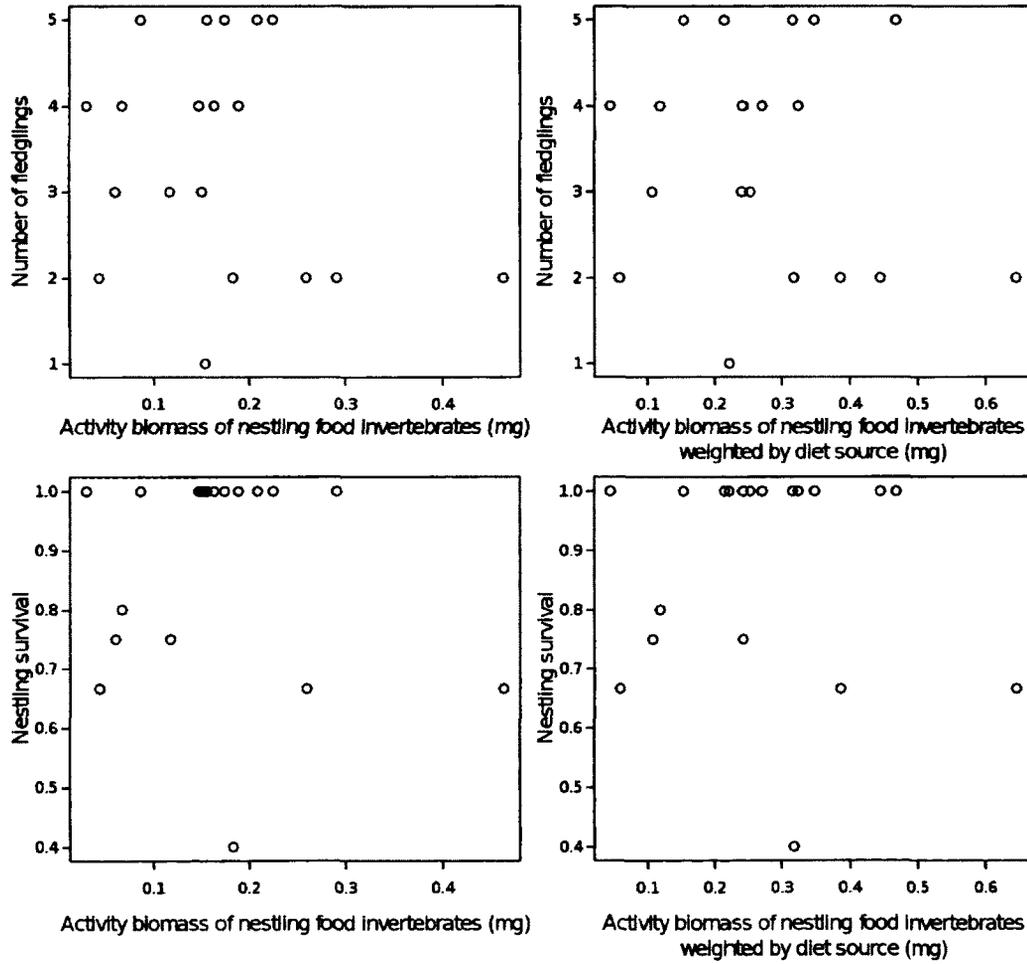
^D includes all adult beetles listed below plus Lampyridae (fireflies), Coccinellidae (ladybird beetles), Nitidulidae (sap beetles), Cicindelidae (tiger beetles). Too few size categories of these additional groups were collected to generate taxon-specific equations.

^E includes Lepidoptera larvae, Coleoptera larvae and unidentified larvae

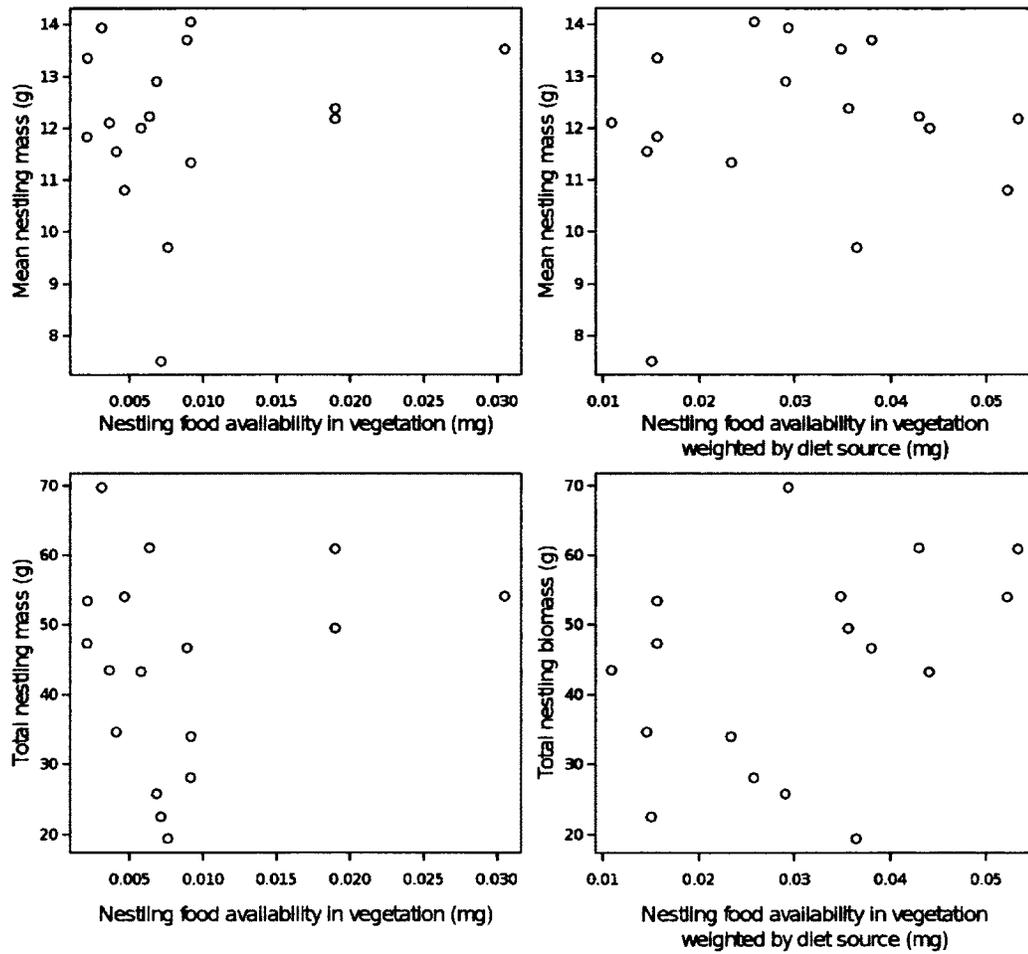
Appendix S: Biplot showing activity biomass of nestling food invertebrates measured in pitfall traps, with mean and total nestling mass in Song Sparrow nests. Upper left panel is duplicated from Fig 5.2 to allow easy comparison.



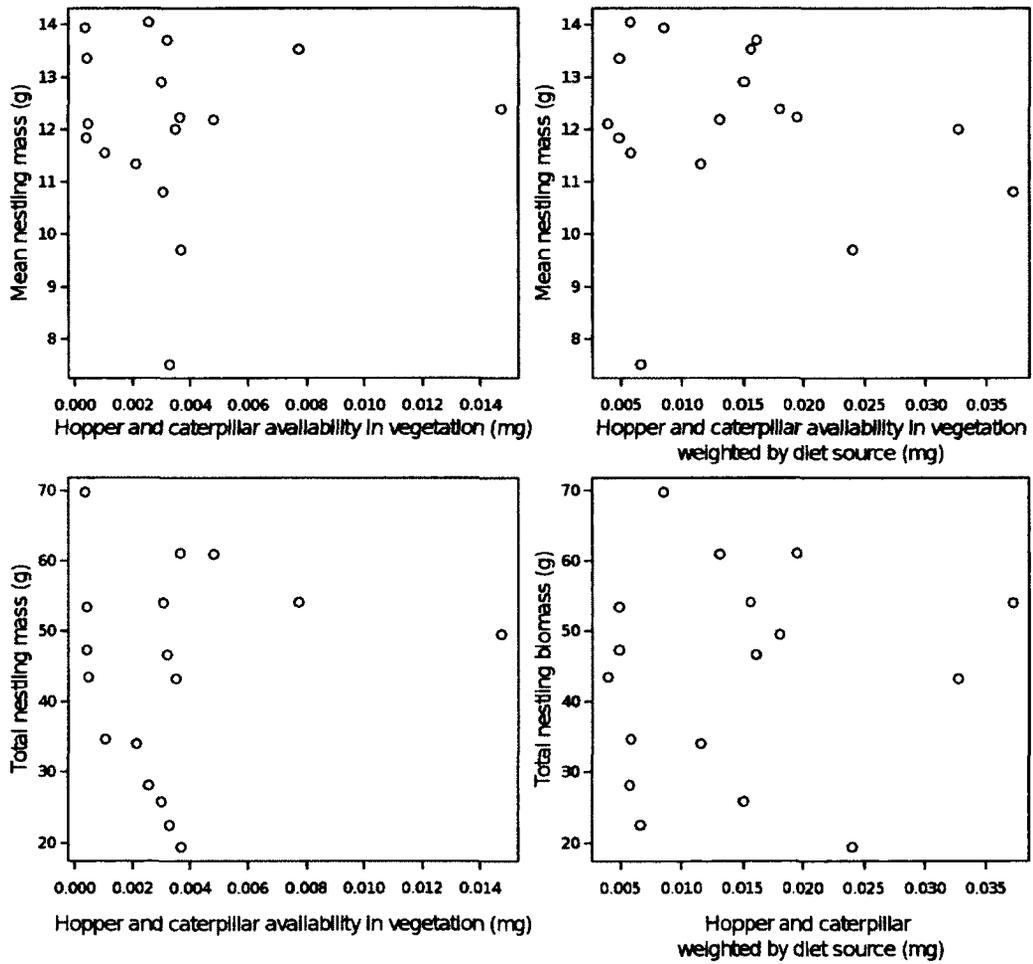
Appendix T: Biplot showing activity biomass of nestling food invertebrates measured in pitfall traps, with numbers of fledglings and nestling survival in Song Sparrow nests.



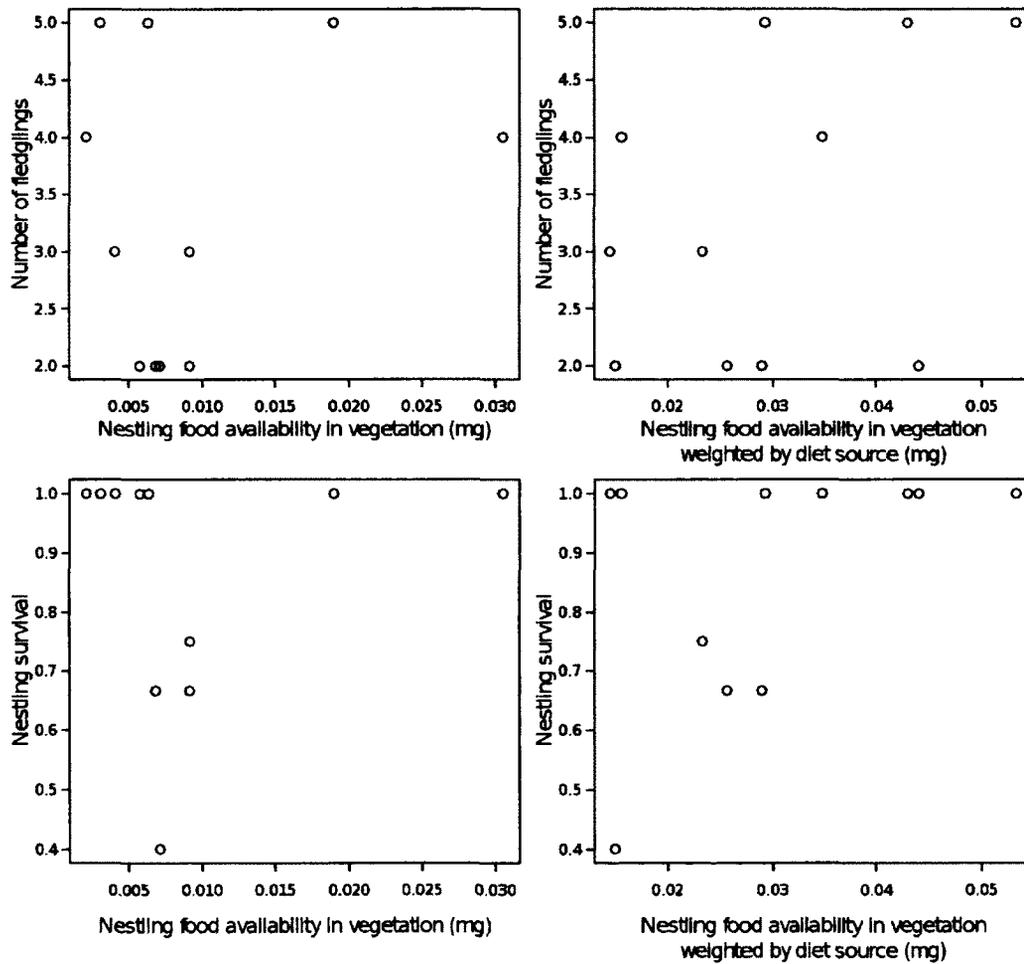
Appendix U: Biplots showing nestling food availability in the vegetation measured using a D-Vac, with mean and total nestling mass in Song Sparrow nests. Upper left panel is duplicated from Fig. 5.2 to allow easy comparison



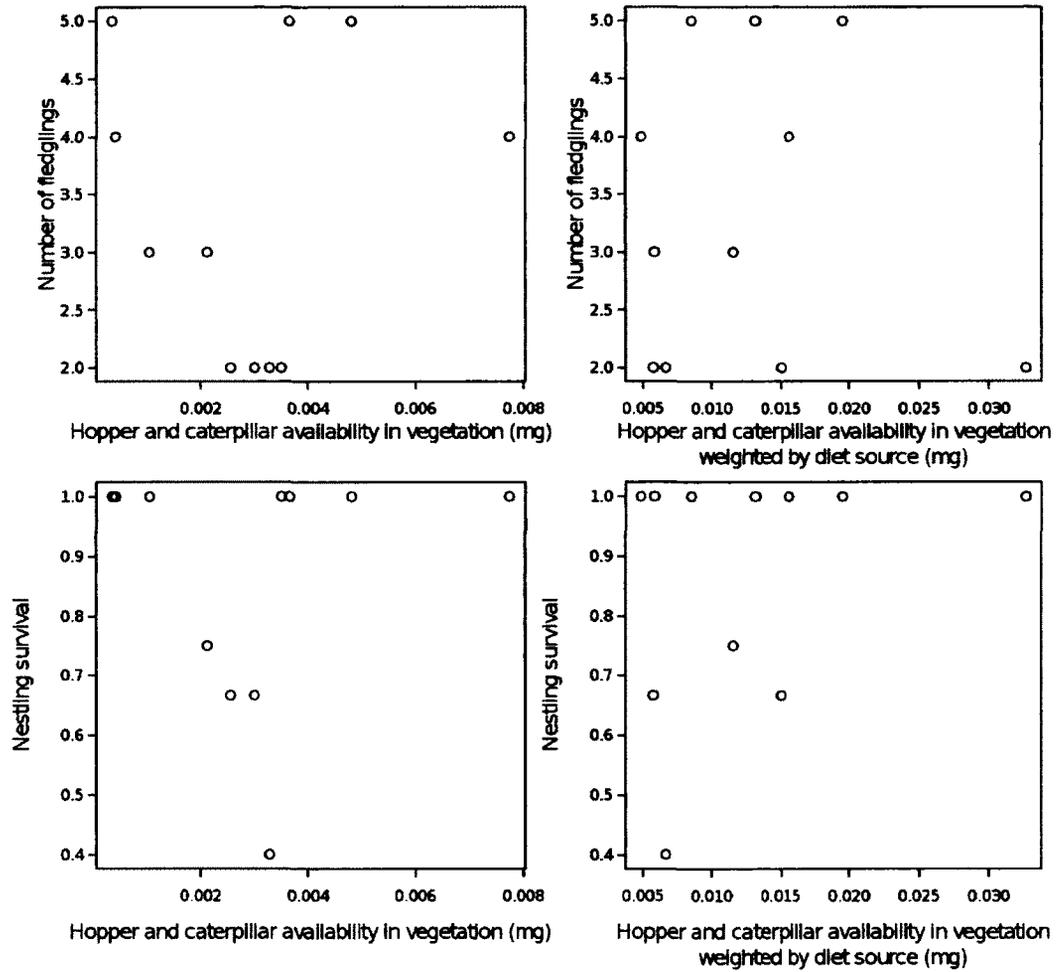
Appendix V: Biplots showing hopper and caterpillar availability in the vegetation measured using a D-Vac with mean and total nestling mass in Song Sparrow nests.



Appendix W: Biplots showing nestling food availability in the vegetation measured with a D-Vac and number of fledglings and nestling survival in Song Sparrow nests.



Appendix X: Biplots showing hopper and caterpillar availability in the vegetation measured using a D-Vac with number of fledglings and nestling survival in Song Sparrow nests.



Appendix Y: Support for logistic exposure models predicting daily survival rates of Song Sparrow nests. All 51 candidate models are shown.

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Model	Log likelihood ^A	K ^B	AIC _c ^C	ΔAIC _c ^D	w _i ^E
nest stage	-173.596	2	351.202	0.000	0.225
seminatural cover + nest stage	-173.149	3	352.320	1.118	0.129
date + nest stage	-173.477	3	352.976	1.774	0.093
brood parasitism + nest stage	-173.567	3	353.156	1.954	0.085
date + seminatural cover + nest stage	-172.939	4	353.914	2.712	0.058
seminatural cover + brood parasitism + nest stage	-173.145	4	354.326	3.123	0.047
date + date2 + nest stage	-173.369	4	354.774	3.572	0.038
year + nest stage	-173.393	4	354.821	3.618	0.037
date + brood parasitism + nest stage	-173.441	4	354.917	3.714	0.035
brood parasitism*nest stage	-173.549	4	355.133	3.930	0.032
date + date2 + seminatural cover + nest stage	-172.806	5	355.666	4.464	0.024
date + seminatural cover + brood parasitism + nest stage	-172.933	5	355.919	4.717	0.021
seminatural cover + year + nest stage	-172.993	5	356.039	4.837	0.020
date + year + nest stage	-173.247	5	356.547	5.345	0.016
date + date2 + brood parasitism + nest stage	-173.300	5	356.653	5.450	0.015
null	-177.328	1	356.660	5.458	0.015
year + brood parasitism + nest stage	-173.378	5	356.808	5.606	0.014
date + seminatural cover + year + nest stage	-172.758	6	357.591	6.389	0.009
date + date2 + seminatural cover + brood parasitism + nest stage	-172.783	6	357.641	6.438	0.009

Appendix Y continued

Model	Log likelihood ^A	K ^B	AIC _c ^C	Δ AIC _c ^D	w _i ^E
seminatural cover	-176.938	2	357.886	6.683	0.008
seminatural cover + year + brood parasitism + nest stage	-172.992	6	358.059	6.857	0.007
date + date2 + year + nest stage	-173.070	6	358.215	7.013	0.007
date + year + brood parasitism + nest stage	-173.224	6	358.523	7.321	0.006
brood parasitism	-177.325	2	358.661	7.459	0.005
date	-177.328	2	358.667	7.465	0.005
seminatural cover*nest stage	-176.372	3	358.766	7.564	0.005
date + date2 + seminatural cover + year + nest stage	-172.566	7	359.231	8.029	0.004
date*nest stage	-176.791	3	359.604	8.402	0.003
global	-172.756	7	359.613	8.410	0.003
date + seminatural cover	-176.930	3	359.881	8.678	0.003
seminatural cover + brood parasitism	-176.936	3	359.894	8.691	0.003
date + date2 + year + brood parasitism + nest stage	-173.011	7	360.121	8.918	0.003
year	-177.216	3	360.453	9.250	0.002
date + date2	-177.299	3	360.620	9.417	0.002
date + brood parasitism	-177.325	3	360.672	9.469	0.002
global with date2	-172.547	8	361.222	10.019	0.002
seminatural cover + year	-176.850	4	361.735	10.532	0.001
date + date2 + seminatural cover	-176.887	4	361.809	10.607	0.001
date + seminatural cover + brood parasitism	-176.929	4	361.893	10.690	0.001
date + year	-177.215	4	362.466	11.264	0.001
year + brood parasitism	-177.216	4	362.467	11.264	0.001

Appendix Y continued:

Model	Log likelihood ^A	K ^B	AIC _c ^C	ΔAIC _c ^D	w _i ^E
date + date2 + brood parasitism	-177.291	4	362.617	11.415	0.001
date + seminatural cover + year	-176.839	5	363.732	12.529	0.000
seminatural cover + year + brood parasitism	-176.843	5	363.740	12.538	0.000
date + date2 + seminatural cover + brood parasitism	-176.887	5	363.827	12.624	0.000
date + date2 + year	-177.168	5	364.390	13.187	0.000
date + year + brood parasitism	-177.215	5	364.484	13.281	0.000
date + date2 + seminatural cover + year	-176.779	6	365.634	14.431	0.000
date + seminatural cover + year + brood parasitism	-176.834	6	365.742	14.539	0.000
date + date2 + year + brood parasitism	-177.164	6	366.404	15.201	0.000
date + date2 + seminatural cover + year + brood parasitism	-176.779	7	367.658	16.455	0.000

^A -2 times the log-likelihood estimator

^B number of parameters

^C Akaike's information criterion, corrected for small sample sizes

^D Change in AIC_c relative to model with lowest AIC_c

^E Akaike weight

Appendix Z: Raw chick food biomass data.

Sample	Type	Month	nestling food biomass	mean residue	mean weed cover	mean plant species richness
C1-A-C3	con	June	0.406	25.0	12.1	15.5
C1-A-C4	con	June	0.499	25.0	12.1	15.5
C1-B-C1	con	June	0.321	25.0	12.1	15.5
C1-B-C2	con	June	0.161	25.0	12.1	15.5
C1-B-C3	con	June	0.135	25.0	12.1	15.5
C1-B-C4	con	June	0.192	25.0	12.1	15.5
C1-A-C1 R3	con	July	0.176	15.6	25.1	15.5
C1-A-C2 R3	con	July	0.152	15.6	25.1	15.5
C1-A-C3 R3	con	July	0.078	15.6	25.1	15.5
C1-A-C4 R3	con	July	0.082	15.6	25.1	15.5
C1-B-C1 R3	con	July	0.070	15.6	25.1	15.5
C1-B-C2 R3	con	July	0.071	15.6	25.1	15.5
C1-B-C3 R3	con	July	0.137	15.6	25.1	15.5
C1-B-C4 R3	con	July	0.215	15.6	25.1	15.5
C10-A-C1	con	June	0.012	22.6	3.6	31.0
C10-A-C2	con	June	0.008	22.6	3.6	31.0
C10-A-C3	con	June	0.020	22.6	3.6	31.0
C10-A-C4	con	June	0.010	22.6	3.6	31.0
C10-B-C1	con	June	0.015	22.6	3.6	31.0
C10-B-C2	con	June	0.019	22.6	3.6	31.0
C10-B-C3	con	June	0.030	22.6	3.6	31.0
C10-B-C4	con	June	0.004	22.6	3.6	31.0
C10-A-C1 R2	con	July	0.139	20.0	6.4	31.0
C10-A-C2 R2	con	July	0.056	20.0	6.4	31.0

Appendix Z continued:

Sample	Type	Month	nestling food biomass	mean residue	mean weed cover	mean plant species richness
C10-A-C3 R2	con	July	0.060	20.0	6.4	31.0
C10-A-C4 R2	con	July	0.148	20.0	6.4	31.0
C10-B-C1 R2	con	July	0.101	20.0	6.4	31.0
C10-B-C2 R2	con	July	0.143	20.0	6.4	31.0
C10-B-C3 R2	con	July	0.089	20.0	6.4	31.0
C11-A-C1	con	June	0.009	8.3	0.1	24.0
C11-A-C2	con	June	0.019	8.3	0.1	24.0
C11-A-C3	con	June	0.025	8.3	0.1	24.0
C11-A-C4	con	June	0.020	8.3	0.1	24.0
C11-B-C1	con	June	0.015	8.3	0.1	24.0
C11-B-C2	con	June	0.065	8.3	0.1	24.0
C11-B-C3	con	June	0.012	8.3	0.1	24.0
C11-B-C4	con	June	0.029	8.3	0.1	24.0
C11-A-C1 R2	con	July	0.416	8.4	0.8	24.0
C11-A-C2 R2	con	July	0.452	8.4	0.8	24.0
C11-A-C3 R2	con	July	0.296	8.4	0.8	24.0
C11-B-C2 R2	con	July	0.117	8.4	0.8	24.0
C11-B-C3 R2	con	July	0.082	8.4	0.8	24.0
C11-B-C4 R2	con	July	0.090	8.4	0.8	24.0
C2-A-C1	con	June	0.009	59.2	0.5	27.0
C2-A-C3	con	June	0.015	59.2	0.5	27.0
C2-A-C4	con	June	0.039	59.2	0.5	27.0
C2-B-C1	con	June	0.028	59.2	0.5	27.0
C2-B-C2	con	June	0.011	59.2	0.5	27.0

Appendix Z continued:

Sample	Type	Month	nestling food biomass	mean residue	mean weed cover	mean plant species richness
C2-B-C3	con	June	0.020	59.2	0.5	27.0
C2-B-C4	con	June	0.059	59.2	0.5	27.0
C2-A-C2 R2	con	July	0.105	30.0	17.9	27.0
C2-A-C3 R2	con	July	0.147	30.0	17.9	27.0
C2-A-C4 R2	con	July	0.221	30.0	17.9	27.0
C2-B-C1 R2	con	July	0.204	30.0	17.9	27.0
C2-B-C2 R2	con	July	0.236	30.0	17.9	27.0
C2-B-C3 R2	con	July	0.155	30.0	17.9	27.0
C2-B-C4 R2	con	July	0.181	30.0	17.9	27.0
C3-A-C2	con	June	0.073	61.3	0.0	27.5
C3-A-C3	con	June	0.121	61.3	0.0	27.5
C3-A-C4	con	June	0.069	61.3	0.0	27.5
C3-B-C1	con	June	0.126	61.3	0.0	27.5
C3-B-C3	con	June	0.430	61.3	0.0	27.5
C3-B-C4	con	June	0.129	61.3	0.0	27.5
C3-A-C1 R2	con	July	1.049	52.3	7.3	27.5
C3-A-C3 R2	con	July	1.925	52.3	7.3	27.5
C3-A-C4 R2	con	July	0.646	52.3	7.3	27.5
C3-B-C1 R2	con	July	0.638	52.3	7.3	27.5
C3-B-C2 R2	con	July	0.641	52.3	7.3	27.5
C3-B-C3 R2	con	July	0.485	52.3	7.3	27.5
C3-B-C4 R2	con	July	0.054	52.3	7.3	27.5
C4.1-A-C1	con	June	0.129	5.9	1.5	32.5
C4.1-A-C2	con	June	0.161	5.9	1.5	32.5

Appendix Z continued:

Sample	Type	Month	nestling food biomass	mean residue	mean weed cover	mean plant species richness
C4.1-A-C4	con	June	0.022	5.9	1.5	32.5
C4.1-B-C1	con	June	0.013	5.9	1.5	32.5
C4.1-B-C2	con	June	0.009	5.9	1.5	32.5
C4.1-B-C3	con	June	0.028	5.9	1.5	32.5
C4.1-B-C4	con	June	0.064	5.9	1.5	32.5
C4.1-A-C1 R2	con	July	0.434	3.0	36.1	32.5
C4.1-A-C2 R2	con	July	0.208	3.0	36.1	32.5
C4.1-A-C3 R2	con	July	0.084	3.0	36.1	32.5
C4.1-A-C4 R2	con	July	0.308	3.0	36.1	32.5
C4.1-B-C1 R2	con	July	0.130	3.0	36.1	32.5
C4.1-B-C2 R2	con	July	0.310	3.0	36.1	32.5
C4.1-B-C3 R2	con	July	0.182	3.0	36.1	32.5
C4.1-B-C4 R2	con	July	0.152	3.0	36.1	32.5
C5-A-C1	con	June	0.326	7.1	0.3	21.5
C5-A-C2	con	June	0.169	7.1	0.3	21.5
C5-A-C3	con	June	0.243	7.1	0.3	21.5
C5-A-C4	con	June	0.260	7.1	0.3	21.5
C5-B-C1	con	June	0.234	7.1	0.3	21.5
C5-B-C2	con	June	0.218	7.1	0.3	21.5
C5-B-C3	con	June	0.323	7.1	0.3	21.5
C5-B-C4	con	June	0.251	7.1	0.3	21.5
C5-A-C1 R2	con	July	0.027	11.3	0.3	21.5
C5-A-C2 R2	con	July	0.024	11.3	0.3	21.5
C5-A-C3 R2	con	July	0.084	11.3	0.3	21.5

Appendix Z continued:

Sample	Type	Month	nestling food biomass	mean residue	mean weed cover	mean plant species richness
C5-A-C4 R2	con	July	0.042	11.3	0.3	21.5
C5-B-C1 R2	con	July	0.033	11.3	0.3	21.5
C5-B-C2 R2	con	July	0.042	11.3	0.3	21.5
C5-B-C3 R2	con	July	0.063	11.3	0.3	21.5
C5-B-C4 R2	con	July	0.017	11.3	0.3	21.5
C7-A-C1	con	June	0.043	44.2	0.5	20.5
C7-A-C2	con	June	0.038	44.2	0.5	20.5
C7-A-C3	con	June	0.014	44.2	0.5	20.5
C7-A-C4	con	June	0.009	44.2	0.5	20.5
C7-B-C1	con	June	0.002	44.2	0.5	20.5
C7-B-C2	con	June	0.004	44.2	0.5	20.5
C7-B-C4	con	June	0.039	44.2	0.5	20.5
C7-A-C1 R2	con	July	0.027	29.2	4.0	20.5
C7-A-C2 R2	con	July	0.021	29.2	4.0	20.5
C7-A-C3 R2	con	July	0.029	29.2	4.0	20.5
C7-A-C4 R2	con	July	0.055	29.2	4.0	20.5
C7-B-C1 R2	con	July	0.139	29.2	4.0	20.5
C7-B-C2 R2	con	July	0.147	29.2	4.0	20.5
C7-B-C3 R2	con	July	0.026	29.2	4.0	20.5
C7-B-C4 R2	con	July	0.035	29.2	4.0	20.5
C8-A-C1	con	June	0.038	82.5	14.0	15.0
C8-A-C2	con	June	0.143	82.5	14.0	15.0
C8-A-C3	con	June	0.101	82.5	14.0	15.0
C8-B-C1	con	June	0.127	82.5	14.0	15.0

Appendix Z continued:

Sample	Type	Month	nestling food biomass	mean residue	mean weed cover	mean plant species richness
C8-B-C3	con	June	0.134	82.5	14.0	15.0
C8-B-C4	con	June	0.155	82.5	14.0	15.0
C8-A-C1 R2	con	July	0.251	79.4	10.5	15.0
C8-A-C2 R2	con	July	0.139	79.4	10.5	15.0
C8-A-C3 R2	con	July	0.247	79.4	10.5	15.0
C8-A-C4 R2	con	July	0.144	79.4	10.5	15.0
C8-B-C1 R2	con	July	0.069	79.4	10.5	15.0
C8-B-C3 R2	con	July	0.324	79.4	10.5	15.0
C8-B-C4 R2	con	July	0.278	79.4	10.5	15.0
O1-A-C1 R3	org	July	0.268	1.0	66.5	26.0
O1-A-C2 R3	org	July	0.102	1.0	66.5	26.0
O1-A-C3 R3	org	July	0.058	1.0	66.5	26.0
O1-A-C4 R3	org	July	0.392	1.0	66.5	26.0
O1-B-C2 R3	org	July	0.127	1.0	66.5	26.0
O1-B-C3 R3	org	July	0.197	1.0	66.5	26.0
O1-B-C4 R3	org	July	0.094	1.0	66.5	26.0
O10-A-C2	org	June	0.069	8.0	2.2	33.0
O10-A-C3	org	June	0.001	8.0	2.2	33.0
O10-A-C4	org	June	0.012	8.0	2.2	33.0
O10-B-C1	org	June	0.001	8.0	2.2	33.0
O10-B-C2	org	June	0.052	8.0	2.2	33.0
O10-B-C3	org	June	0.044	8.0	2.2	33.0
O10-A-C2 R2	org	July	0.190	15.0	34.8	33.0
O10-A-C3 R2	org	July	0.115	15.0	34.8	33.0

Appendix Z continued:

Sample	Type	Month	nestling food biomass	mean residue	mean weed cover	mean plant species richness
O10-A-C4 R2	org	July	0.270	15.0	34.8	33.0
O10-B-C1 R2	org	July	0.411	15.0	34.8	33.0
O10-B-C2 R2	org	July	0.297	15.0	34.8	33.0
O10-B-C3 R2	org	July	0.070	15.0	34.8	33.0
O11-B-C1	org	June	0.343	17.1	1.4	27.0
O11-A-C1 R2	org	July	0.311	18.1	35.9	27.0
O11-A-C2 R2	org	July	0.128	18.1	35.9	27.0
O11-A-C3 R2	org	July	0.325	18.1	35.9	27.0
O11-A-C4 R2	org	July	0.241	18.1	35.9	27.0
O11-B-C2 R2	org	July	0.102	18.1	35.9	27.0
O11-B-C3 R2	org	July	0.151	18.1	35.9	27.0
O11-B-C4 R2	org	July	0.317	18.1	35.9	27.0
O2-A-C1	org	June	0.053	93.0	2.5	31.5
O2-A-C2	org	June	0.154	93.0	2.5	31.5
O2-A-C3	org	June	0.164	93.0	2.5	31.5
O2-A-C4	org	June	0.239	93.0	2.5	31.5
O2-B-C1	org	June	0.070	93.0	2.5	31.5
O2-B-C3	org	June	0.207	93.0	2.5	31.5
O2-B-C4	org	June	0.183	93.0	2.5	31.5
O2-A-C1 R2	org	July	0.491	90.2	11.3	31.5
O2-A-C2 R2	org	July	0.725	90.2	11.3	31.5
O2-A-C3 R2	org	July	0.288	90.2	11.3	31.5
O2-A-C3 R2	org	July	0.684	90.2	11.3	31.5
O2-B-C1 R2	org	July	0.546	90.2	11.3	31.5

Appendix Z continued:

Sample	Type	Month	nestling food biomass	mean residue	mean weed cover	mean plant species richness
O2-B-C2 R2	org	July	0.692	90.2	11.3	31.5
O2-B-C3 R2	org	July	0.349	90.2	11.3	31.5
O2-B-C4 R2	org	July	0.638	90.2	11.3	31.5
O3-A-C1	org	June	0.158	12.4	0.0	26.5
O3-A-C2	org	June	0.162	12.4	0.0	26.5
O3-A-C3	org	June	0.079	12.4	0.0	26.5
O3-A-C4	org	June	0.088	12.4	0.0	26.5
O3-B-C1	org	June	0.248	12.4	0.0	26.5
O3-B-C2	org	June	0.247	12.4	0.0	26.5
O3-B-C3	org	June	0.140	12.4	0.0	26.5
O3-B-C4	org	June	0.107	12.4	0.0	26.5
O3-A-C2 R2	org	July	0.410	2.6	55.0	26.5
O3-A-C3 R2	org	July	0.238	2.6	55.0	26.5
O3-A-C4 R2	org	July	0.153	2.6	55.0	26.5
O3-B-C1 R2	org	July	0.196	2.6	55.0	26.5
O3-B-C2 R2	org	July	0.340	2.6	55.0	26.5
O3-B-C3 R2	org	July	0.219	2.6	55.0	26.5
O3-B-C4 R2	org	July	0.240	2.6	55.0	26.5
O4-A-C1	org	June	0.347	26.3	0.3	22.5
O4-A-C2	org	June	0.098	26.3	0.3	22.5
O4-A-C3	org	June	0.309	26.3	0.3	22.5
O4-B-C1	org	June	0.132	26.3	0.3	22.5
O4-B-C2	org	June	0.086	26.3	0.3	22.5
O4-B-C4	org	June	0.151	26.3	0.3	22.5

Appendix Z continued:

Sample	Type	Month	nestling food biomass	mean residue	mean weed cover	mean plant species richness
O4-A-C1 R2	org	July	0.144	22.5	52.5	22.5
O4-A-C2 R2	org	July	0.783	22.5	52.5	22.5
O4-B-C2 R2	org	July	0.364	22.5	52.5	22.5
O4-B-C3 R2	org	July	0.185	22.5	52.5	22.5
O4-B-C4 R2	org	July	0.286	22.5	52.5	22.5
O4-B-C4 R2	org	July	0.532	22.5	52.5	22.5
O5-A-C1	org	June	0.061	30.6	4.3	18.5
O5-A-C2	org	June	0.154	30.6	4.3	18.5
O5-A-C3	org	June	0.169	30.6	4.3	18.5
O5-A-C4	org	June	0.211	30.6	4.3	18.5
O5-B-C1	org	June	0.210	30.6	4.3	18.5
O5-B-C2	org	June	0.108	30.6	4.3	18.5
O5-B-C3	org	June	0.171	30.6	4.3	18.5
O5-B-C4	org	June	0.266	30.6	4.3	18.5
O5-A-C2 R2	org	July	0.034	9.0	29.7	18.5
O5-A-C4 R2	org	July	0.322	9.0	29.7	18.5
O5-B-C1 R2	org	July	0.116	9.0	29.7	18.5
O5-B-C2 R2	org	July	0.241	9.0	29.7	18.5
O5-B-C3 R2	org	July	0.086	9.0	29.7	18.5
O5-B-C4 R2	org	July	0.319	9.0	29.7	18.5
O7-A-C1	org	June	0.092	17.1	1.4	36.5
O7-A-C2	org	June	0.020	17.1	1.4	36.5
O7-B-C1	org	June	0.118	17.1	1.4	36.5
O7-B-C3	org	June	0.044	17.1	1.4	36.5

Appendix Z continued:

Sample	Type	Month	nestling food biomass	mean residue	mean weed cover	mean plant species richness
O7-B-C4	org	June	0.095	17.1	1.4	36.5
O7-A-C1 R2	org	July	0.240	1.0	18.0	36.5
O7-A-C2 R2	org	July	0.042	1.0	18.0	36.5
O7-A-C3 R2	org	July	0.264	1.0	18.0	36.5
O7-A-C4 R2	org	July	0.384	1.0	18.0	36.5
O7-B-C1 R2	org	July	0.138	1.0	18.0	36.5
O7-B-C2 R2	org	July	0.074	1.0	18.0	36.5
O7-B-C3 R2	org	July	0.139	1.0	18.0	36.5
O7-B-C4 R2	org	July	0.082	1.0	18.0	36.5
O8-A-C1	org	June	0.013	8.1	0.1	23.0
O8-A-C2	org	June	0.059	8.1	0.1	23.0
O8-A-C3	org	June	0.029	8.1	0.1	23.0
O8-A-C4	org	June	0.022	8.1	0.1	23.0
O8-B-C1	org	June	0.106	8.1	0.1	23.0
O8-B-C2	org	June	0.196	8.1	0.1	23.0
O8-B-C3	org	June	0.026	8.1	0.1	23.0
O8-B-C4	org	June	0.187	8.1	0.1	23.0
O8-A-C1 R2	org	July	1.179	3.4	19.8	23.0
O8-A-C2 R2	org	July	0.897	3.4	19.8	23.0
O8-A-C3 R2	org	July	0.765	3.4	19.8	23.0
O8-B-C1 R2	org	July	1.288	3.4	19.8	23.0
O8-B-C3 R2	org	July	0.533	3.4	19.8	23.0
O8-B-C4 R2	org	July	1.031	3.4	19.8	23.0
C1-A-E1	con	June	0.260	25.0	12.1	15.5

Appendix Z continued:

Sample	Type	Month	nestling food biomass	mean residue	mean weed cover	mean plant species richness
C1-A-E2	con	June	0.419	25.0	12.1	15.5
C1-A-E3	con	June	0.261	25.0	12.1	15.5
C1-B-E1	con	June	0.342	25.0	12.1	15.5
C1-B-E2	con	June	0.424	25.0	12.1	15.5
C1-B-E3	con	June	0.052	25.0	12.1	15.5
C1-B-E4	con	June	0.287	25.0	12.1	15.5
C1-A-E1 R3	con	July	0.097	15.6	25.1	15.5
C1-A-E2 R3	con	July	0.086	15.6	25.1	15.5
C1-A-E3 R3	con	July	0.158	15.6	25.1	15.5
C1-B-E1 R3	con	July	0.149	15.6	25.1	15.5
C1-B-E2 R3	con	July	0.294	15.6	25.1	15.5
C1-B-E4 R3	con	July	0.185	15.6	25.1	15.5
C10-A-E1	con	June	0.146	22.6	3.6	31.0
C10-A-E2	con	June	0.024	22.6	3.6	31.0
C10-A-E3	con	June	0.164	22.6	3.6	31.0
C10-A-E4	con	June	0.113	22.6	3.6	31.0
C10-B-E1	con	June	0.425	22.6	3.6	31.0
C10-B-E2	con	June	0.324	22.6	3.6	31.0
C10-B-E3	con	June	0.230	22.6	3.6	31.0
C10-B-E4	con	June	0.122	22.6	3.6	31.0
C10-A-E2 R2	con	July	0.191	20.0	6.4	31.0
C10-A-E3 R2	con	July	0.121	20.0	6.4	31.0
C10-A-E4 R2	con	July	0.100	20.0	6.4	31.0
C10-B-E1 R2	con	July	0.068	20.0	6.4	31.0

Appendix Z continued:

Sample	Type	Month	nestling food biomass	mean residue	mean weed cover	mean plant species richness
C10-B-E2 R2	con	July	0.052	20.0	6.4	31.0
C11-A-E1	con	June	0.124	8.3	0.1	24.0
C11-A-E2	con	June	0.255	8.3	0.1	24.0
C11-A-E3	con	June	0.028	8.3	0.1	24.0
C11-A-E4	con	June	0.014	8.3	0.1	24.0
C11-B-E1	con	June	0.112	8.3	0.1	24.0
C11-B-E2	con	June	0.160	8.3	0.1	24.0
C11-B-E3	con	June	0.307	8.3	0.1	24.0
C11-B-E4	con	June	0.134	8.3	0.1	24.0
C11-A-E1 R2	con	July	0.521	8.4	0.8	24.0
C11-A-E2 R2	con	July	0.261	8.4	0.8	24.0
C11-A-E3 R2	con	July	0.958	8.4	0.8	24.0
C11-A-E4 R2	con	July	0.232	8.4	0.8	24.0
C11-B-E2 R2	con	July	0.040	8.4	0.8	24.0
C11-B-E3 R2	con	July	0.512	8.4	0.8	24.0
C11-B-E4 R2	con	July	0.285	8.4	0.8	24.0
C2-A-E1	con	June	0.129	59.2	0.5	27.0
C2-A-E2	con	June	0.077	59.2	0.5	27.0
C2-A-E3	con	June	0.080	59.2	0.5	27.0
C2-B-E1	con	June	0.260	59.2	0.5	27.0
C2-B-E2	con	June	0.115	59.2	0.5	27.0
C2-B-E3	con	June	0.222	59.2	0.5	27.0
C2-B-E4	con	June	0.063	59.2	0.5	27.0
C2-A-E1 R2	con	July	0.745	30.0	17.9	27.0

Appendix Z continued:

Sample	Type	Month	nestling food biomass	mean residue	mean weed cover	mean plant species richness
C2-A-E2 R2	con	July	0.137	30.0	17.9	27.0
C2-A-E3 R2	con	July	0.202	30.0	17.9	27.0
C2-A-E4 R2	con	July	0.273	30.0	17.9	27.0
C2-B-E1 R2	con	July	0.482	30.0	17.9	27.0
C2-B-E2 R2	con	July	0.334	30.0	17.9	27.0
C2-B-E3 R2	con	July	0.183	30.0	17.9	27.0
C2-B-E4 R2	con	July	0.271	30.0	17.9	27.0
C3-A-E2	con	June	0.020	61.3	0.0	27.5
C3-A-E3	con	June	0.112	61.3	0.0	27.5
C3-A-E4	con	June	0.195	61.3	0.0	27.5
C3-B-E1	con	June	0.206	61.3	0.0	27.5
C3-A-E2 R2	con	July	0.692	52.3	7.3	27.5
C3-A-E3 R2	con	July	1.652	52.3	7.3	27.5
C3-B-E4 R2	con	July	0.459	52.3	7.3	27.5
C4.1-A-E1	con	June	0.034	5.9	1.5	32.5
C4.1-A-E2	con	June	0.047	5.9	1.5	32.5
C4.1-A-E3	con	June	0.077	5.9	1.5	32.5
C4.1-A-E4	con	June	0.111	5.9	1.5	32.5
C4.1-B-E1	con	June	0.024	5.9	1.5	32.5
C4.1-B-E3	con	June	0.012	5.9	1.5	32.5
C4.1-B-E4	con	June	0.010	5.9	1.5	32.5
C4.1-A-E1 R2	con	July	0.268	3.0	36.1	32.5
C4.1-A-E2 R2	con	July	0.158	3.0	36.1	32.5

Appendix Z continued:

Sample	Type	Month	nestling food biomass	mean residue	mean weed cover	mean plant species richness
C4.1-A-E4 R2	con	July	0.180	3.0	36.1	32.5
C4.1-B-E1 R2	con	July	0.090	3.0	36.1	32.5
C4.1-B-E2 R2	con	July	0.056	3.0	36.1	32.5
C4.1-B-E3 R2	con	July	0.237	3.0	36.1	32.5
C4.1-B-E4 R2	con	July	0.171	3.0	36.1	32.5
C5-A-E1	con	June	0.133	7.1	0.3	21.5
C5-A-E2	con	June	0.182	7.1	0.3	21.5
C5-A-E3	con	June	0.287	7.1	0.3	21.5
C5-A-E4	con	June	0.406	7.1	0.3	21.5
C5-B-E1	con	June	0.270	7.1	0.3	21.5
C5-B-E2	con	June	0.558	7.1	0.3	21.5
C5-A-E1 R2	con	July	0.146	11.3	0.3	21.5
C5-A-E2 R2	con	July	0.235	11.3	0.3	21.5
C5-A-E3 R2	con	July	0.159	11.3	0.3	21.5
C5-A-E4 R2	con	July	0.087	11.3	0.3	21.5
C5-B-E1 R2	con	July	0.199	11.3	0.3	21.5
C5-B-E2 R2	con	July	0.102	11.3	0.3	21.5
C5-B-E3 R2	con	July	0.130	11.3	0.3	21.5
C5-B-E4 R2	con	July	0.316	11.3	0.3	21.5
C7-A-E1	con	June	0.156	44.2	0.5	20.5
C7-A-E2	con	June	0.098	44.2	0.5	20.5
C7-A-E3	con	June	0.051	44.2	0.5	20.5
C7-A-E4	con	June	0.054	44.2	0.5	20.5
C7-B-E1	con	June	0.124	44.2	0.5	20.5

Appendix Z continued:

Sample	Type	Month	nestling food biomass	mean residue	mean weed cover	mean plant species richness
C7-B-E2	con	June	0.039	44.2	0.5	20.5
C7-B-E3	con	June	0.144	44.2	0.5	20.5
C7-B-E4	con	June	0.014	44.2	0.5	20.5
C7-A-E1 R2	con	July	0.063	29.2	4.0	20.5
C7-A-E2 R2	con	July	0.108	29.2	4.0	20.5
C7-A-E3 R2	con	July	0.113	29.2	4.0	20.5
C7-A-E4 R2	con	July	0.292	29.2	4.0	20.5
C7-B-E1 R2	con	July	0.303	29.2	4.0	20.5
C7-B-E2 R2	con	July	0.154	29.2	4.0	20.5
C7-B-E3 R2	con	July	0.214	29.2	4.0	20.5
C8-A-E3	con	June	0.012	82.5	14.0	15.0
C8-A-E4	con	June	0.048	82.5	14.0	15.0
C8-B-E1	con	June	0.024	82.5	14.0	15.0
C8-B-E2	con	June	0.121	82.5	14.0	15.0
C8-B-E3	con	June	0.100	82.5	14.0	15.0
C8-A-E2 R2	con	July	0.123	79.4	10.5	15.0
C8-A-E4 R2	con	July	0.151	79.4	10.5	15.0
C8-B-E1 R2	con	July	0.032	79.4	10.5	15.0
C8-B-E2 R2	con	July	0.037	79.4	10.5	15.0
C8-B-E3 R2	con	July	0.220	79.4	10.5	15.0
C8-B-E4 R2	con	July	0.030	79.4	10.5	15.0
O1-A-E1	org	June	0.134	17.1	1.4	26.0
O1-A-E3	org	June	0.291	17.1	1.4	26.0
O1-B-E1	org	June	0.405	17.1	1.4	26.0

Appendix Z continued:

Sample	Type	Month	nestling food biomass	mean residue	mean weed cover	mean plant species richness
O1-A-E1 R3	org	July	0.325	1.0	66.5	26.0
O1-A-E2 R3	org	July	0.201	1.0	66.5	26.0
O1-A-E3 R3	org	July	0.395	1.0	66.5	26.0
O1-A-E4 R3	org	July	0.516	1.0	66.5	26.0
O1-B-E1 R3	org	July	0.218	1.0	66.5	26.0
O1-B-E2 R3	org	July	0.518	1.0	66.5	26.0
O1-B-E3 R3	org	July	0.107	1.0	66.5	26.0
O1-B-E4 R3	org	July	0.456	1.0	66.5	26.0
O10-A-E3	org	June	0.107	8.0	2.2	33.0
O10-A-E4	org	June	0.107	8.0	2.2	33.0
O10-B-E4	org	June	0.067	8.0	2.2	33.0
O10-A-E1 R2	org	July	0.235	15.0	34.8	33.0
O10-A-E2 R2	org	July	0.442	15.0	34.8	33.0
O10-A-E3 R2	org	July	0.658	15.0	34.8	33.0
O10-A-E4 R2	org	July	0.393	15.0	34.8	33.0
O10-B-E1 R2	org	July	0.896	15.0	34.8	33.0
O10-B-E2 R2	org	July	0.396	15.0	34.8	33.0
O10-B-E3 R2	org	July	0.585	15.0	34.8	33.0
O10-B-E4 R2	org	July	0.352	15.0	34.8	33.0
O11-A-E1	org	June	0.085	17.1	1.4	27.0
O11-B-E1	org	June	0.025	17.1	1.4	27.0
O11-B-E3	org	June	0.083	17.1	1.4	27.0
O2-A-E1	org	June	0.085	93.0	2.5	31.5
O2-A-E2	org	June	0.112	93.0	2.5	31.5

Appendix Z continued:

Sample	Type	Month	nestling food biomass	mean residue	mean weed cover	mean plant species richness
O2-A-E3	org	June	0.097	93.0	2.5	31.5
O2-A-E4	org	June	0.176	93.0	2.5	31.5
O2-B-E1	org	June	0.528	93.0	2.5	31.5
O2-B-E3	org	June	0.239	93.0	2.5	31.5
O2-B-E4	org	June	0.211	93.0	2.5	31.5
O2-A-E1 R2	org	July	0.652	90.2	11.3	31.5
O2-A-E2 R2	org	July	0.566	90.2	11.3	31.5
O2-A-E3 R2	org	July	1.041	90.2	11.3	31.5
O2-A-E4 R2	org	July	0.439	90.2	11.3	31.5
O2-B-E1 R2	org	July	0.433	90.2	11.3	31.5
O2-B-E2 R2	org	July	0.865	90.2	11.3	31.5
O2-B-E3 R2	org	July	0.252	90.2	11.3	31.5
O2-B-E4 R2	org	July	0.665	90.2	11.3	31.5
O3-A-E1	org	June	0.033	12.4	0.0	26.5
O3-A-E2	org	June	0.162	12.4	0.0	26.5
O3-A-E3	org	June	0.163	12.4	0.0	26.5
O3-A-E4	org	June	0.420	12.4	0.0	26.5
O3-B-E1	org	June	0.371	12.4	0.0	26.5
O3-B-E2	org	June	0.118	12.4	0.0	26.5
O3-B-E4	org	June	0.397	12.4	0.0	26.5
O3-A-E1 R2	org	July	0.202	2.6	55.0	26.5
O3-A-E2 R2	org	July	0.338	2.6	55.0	26.5
O3-A-E3 R2	org	July	0.315	2.6	55.0	26.5
O3-A-E4 R2	org	July	0.179	2.6	55.0	26.5

Appendix Z continued:

Sample	Type	Month	nestling food biomass	mean residue	mean weed cover	mean plant species richness
O3-B-E1 R2	org	July	0.036	2.6	55.0	26.5
O3-B-E2 R2	org	July	0.242	2.6	55.0	26.5
O3-B-E4 R2	org	July	0.287	2.6	55.0	26.5
O4-A-E1	org	June	0.144	26.3	0.3	22.5
O4-A-E2	org	June	0.691	26.3	0.3	22.5
O4-A-E3	org	June	0.108	26.3	0.3	22.5
O4-A-E4	org	June	0.224	26.3	0.3	22.5
O4-B-E1	org	June	0.152	26.3	0.3	22.5
O4-B-E2	org	June	0.086	26.3	0.3	22.5
O4-B-E3	org	June	0.108	26.3	0.3	22.5
O4-A-E2 R2	org	July	0.162	22.5	52.5	22.5
O4-A-E3 R2	org	July	0.520	22.5	52.5	22.5
O4-A-E4 R2	org	July	0.477	22.5	52.5	22.5
O4-B-E1 R2	org	July	0.329	22.5	52.5	22.5
O4-B-E2 R2	org	July	0.489	22.5	52.5	22.5
O4-B-E3 R2	org	July	1.123	22.5	52.5	22.5
O4-B-E4 R2	org	July	0.158	22.5	52.5	22.5
O5-A-E1	org	June	0.058	30.6	4.3	18.5
O5-A-E2	org	June	0.091	30.6	4.3	18.5
O5-A-E3	org	June	0.108	30.6	4.3	18.5
O5-A-E4	org	June	0.083	30.6	4.3	18.5
O5-B-E1	org	June	0.052	30.6	4.3	18.5
O5-B-E2	org	June	0.045	30.6	4.3	18.5
O5-B-E3	org	June	0.083	30.6	4.3	18.5

Appendix Z continued:

Sample	Type	Month	nestling food biomass	mean residue	mean weed cover	mean plant species richness
O5-B-E4	org	June	0.082	30.6	4.3	18.5
O5-A-E1 R2	org	July	0.519	9.0	29.7	18.5
O5-A-E3 R2	org	July	0.040	9.0	29.7	18.5
O5-A-E4 R2	org	July	0.335	9.0	29.7	18.5
O5-B-E3 R2	org	July	0.126	9.0	29.7	18.5
O5-B-E4 R2	org	July	0.285	9.0	29.7	18.5
O7-A-E1	org	June	0.177	17.1	1.4	36.5
O7-A-E2	org	June	0.103	17.1	1.4	36.5
O7-A-E3	org	June	0.135	17.1	1.4	36.5
O7-A-E4	org	June	0.029	17.1	1.4	36.5
O7-B-E1	org	June	0.133	17.1	1.4	36.5
O7-B-E2	org	June	0.229	17.1	1.4	36.5
O7-B-E3	org	June	0.370	17.1	1.4	36.5
O7-B-E4	org	June	0.110	17.1	1.4	36.5
O7-A-E1 R2	org	July	0.128	1.0	18.0	36.5
O7-A-E2 R2	org	July	0.581	1.0	18.0	36.5
O7-A-E3 R2	org	July	0.112	1.0	18.0	36.5
O7-A-E4 R2	org	July	0.067	1.0	18.0	36.5
O7-B-E1 R2	org	July	0.295	1.0	18.0	36.5
O7-B-E2 R2	org	July	0.061	1.0	18.0	36.5
O7-B-E3 R2	org	July	0.165	1.0	18.0	36.5
O7-B-E4 R2	org	July	0.469	1.0	18.0	36.5
O8-A-E3	org	June	0.077	8.1	0.1	23.0
O8-A-E4	org	June	0.129	8.1	0.1	23.0

Appendix Z continued:

Sample	Type	Month	nestling food biomass	mean residue	mean weed cover	mean plant species richness
O8-B-E3	org	June	0.135	8.1	0.1	23.0
O8-B-E4	org	June	0.144	8.1	0.1	23.0
O8-B-E1	org	June	0.158	8.1	0.1	23.0
O8-A-E1	org	June	0.197	8.1	0.1	23.0
O8-B-E2	org	June	0.342	8.1	0.1	23.0
O8-A-E2	org	June	0.500	8.1	0.1	23.0
O8-B-E3 R2	org	July	0.010	3.4	19.8	23.0
O8-B-E4 R2	org	July	0.079	3.4	19.8	23.0
O8-A-E2 R2	org	July	0.172	3.4	19.8	23.0
O8-B-E1 R2	org	July	0.245	3.4	19.8	23.0
O8-B-E2 R2	org	July	0.306	3.4	19.8	23.0
O8-A-E3 R2	org	July	0.416	3.4	19.8	23.0
O8-A-E4 R2	org	July	0.697	3.4	19.8	23.0
O8-A-E1 R2	org	July	1.656	3.4	19.8	23.0
C1-A-H2	con	June	0.165	25.0	12.1	15.5
C1-B-H1	con	June	0.041	25.0	12.1	15.5
C1-B-H2	con	June	0.058	25.0	12.1	15.5
C1-B-H3	con	June	0.012	25.0	12.1	15.5
C1-B-H4	con	June	0.025	25.0	12.1	15.5
C1-A-H1 R3	con	July	0.091	15.6	25.1	15.5
C1-A-H2 R3	con	July	0.069	15.6	25.1	15.5
C1-A-H3 R3	con	July	0.069	15.6	25.1	15.5
C1-A-H4 R3	con	July	0.100	15.6	25.1	15.5
C1-B-H2 R3	con	July	0.248	15.6	25.1	15.5

Appendix Z continued:

Sample	Type	Month	nestling food biomass	mean residue	mean weed cover	mean plant species richness
C1-B-H3 R3	con	July	0.437	15.6	25.1	15.5
C1-B-H4 R3	con	July	0.215	15.6	25.1	15.5
C10-A-H1	con	June	0.023	22.6	3.6	31.0
C10-A-H2	con	June	0.015	22.6	3.6	31.0
C10-A-H3	con	June	0.045	22.6	3.6	31.0
C10-A-H4	con	June	0.007	22.6	3.6	31.0
C10-B-H1	con	June	0.053	22.6	3.6	31.0
C10-B-H3	con	June	0.100	22.6	3.6	31.0
C10-A-H1 R2	con	July	0.035	20.0	6.4	31.0
C10-A-H2 R2	con	July	0.031	20.0	6.4	31.0
C10-A-H3 R2	con	July	0.049	20.0	6.4	31.0
C10-A-H4 R2	con	July	0.053	20.0	6.4	31.0
C10-B-H1 R2	con	July	0.034	20.0	6.4	31.0
C10-B-H2 R2	con	July	0.020	20.0	6.4	31.0
C10-B-H3 R2	con	July	0.022	20.0	6.4	31.0
C11-A-H1	con	June	0.080	8.3	0.1	24.0
C11-A-H2	con	June	0.171	8.3	0.1	24.0
C11-A-H3	con	June	0.032	8.3	0.1	24.0
C11-A-H4	con	June	0.011	8.3	0.1	24.0
C11-B-H2	con	June	0.044	8.3	0.1	24.0
C11-B-H3	con	June	0.041	8.3	0.1	24.0
C11-B-H4	con	June	0.043	8.3	0.1	24.0
C11-A-H1 R2	con	July	0.007	8.4	0.8	24.0
C11-A-H3 R2	con	July	0.056	8.4	0.8	24.0

Appendix Z continued:

Sample	Type	Month	nestling food biomass	mean residue	mean weed cover	mean plant species richness
C11-A-H4 R2	con	July	0.026	8.4	0.8	24.0
C11-B-H1 R2	con	July	0.043	8.4	0.8	24.0
C11-B-H2 R2	con	July	0.042	8.4	0.8	24.0
C11-B-H3 R2	con	July	0.013	8.4	0.8	24.0
C11-B-H4 R2	con	July	0.042	8.4	0.8	24.0
C2-A-H1	con	June	0.108	59.2	0.5	27.0
C2-A-H2	con	June	0.075	59.2	0.5	27.0
C2-A-H3	con	June	0.032	59.2	0.5	27.0
C2-A-H4	con	June	0.082	59.2	0.5	27.0
C2-B-H1	con	June	0.098	59.2	0.5	27.0
C2-B-H2	con	June	0.033	59.2	0.5	27.0
C2-B-H3	con	June	0.074	59.2	0.5	27.0
C2-B-H4	con	June	0.063	59.2	0.5	27.0
C2-A-H1 R2	con	July	0.079	30.0	17.9	27.0
C2-A-H2 R2	con	July	0.106	30.0	17.9	27.0
C2-A-H3 R2	con	July	0.017	30.0	17.9	27.0
C2-A-H4 R2	con	July	0.018	30.0	17.9	27.0
C2-B-H1 R2	con	July	0.043	30.0	17.9	27.0
C2-B-H2 R2	con	July	0.079	30.0	17.9	27.0
C2-B-H3 R2	con	July	0.047	30.0	17.9	27.0
C2-B-H4 R2	con	July	0.096	30.0	17.9	27.0
C3-A-H1	con	June	0.055	61.3	0.0	27.5
C3-A-H2	con	June	0.069	61.3	0.0	27.5
C3-A-H3	con	June	0.281	61.3	0.0	27.5

Appendix Z continued:

Sample	Type	Month	nestling food biomass	mean residue	mean weed cover	mean plant species richness
C3-A-H4	con	June	0.145	61.3	0.0	27.5
C3-B-H1	con	June	0.029	61.3	0.0	27.5
C3-B-H2	con	June	0.127	61.3	0.0	27.5
C3-B-H3	con	June	0.017	61.3	0.0	27.5
C3-B-H4	con	June	0.066	61.3	0.0	27.5
C3-A-H1 R2	con	July	0.140	52.3	7.3	27.5
C3-A-H2 R2	con	July	0.124	52.3	7.3	27.5
C3-A-H3 R2	con	July	0.061	52.3	7.3	27.5
C3-A-H4 R2	con	July	0.146	52.3	7.3	27.5
C3-B-H4 R2	con	July	0.042	52.3	7.3	27.5
C4.1-A-H1	con	June	0.049	5.9	1.5	32.5
C4.1-A-H3	con	June	0.027	5.9	1.5	32.5
C4.1-B-H1	con	June	0.006	5.9	1.5	32.5
C4.1-B-H3	con	June	0.086	5.9	1.5	32.5
C4.1-B-H4	con	June	0.024	5.9	1.5	32.5
C4.1-A-H1 R2	con	July	0.053	3.0	36.1	32.5
C4.1-A-H2 R2	con	July	0.042	3.0	36.1	32.5
C4.1-A-H3 R2	con	July	0.010	3.0	36.1	32.5
C4.1-A-H4 R2	con	July	0.062	3.0	36.1	32.5
C4.1-B-H1 R2	con	July	0.024	3.0	36.1	32.5
C4.1-B-H2 R2	con	July	0.035	3.0	36.1	32.5
C4.1-B-H3 R2	con	July	0.040	3.0	36.1	32.5
C4.1-B-H4 R2	con	July	0.042	3.0	36.1	32.5
C5-A-H1	con	June	0.131	7.1	0.3	21.5

Appendix Z continued:

Sample	Type	Month	nestling food biomass	mean residue	mean weed cover	mean plant species richness
C5-A-H2	con	June	0.051	7.1	0.3	21.5
C5-A-H3	con	June	0.069	7.1	0.3	21.5
C5-A-H4	con	June	0.057	7.1	0.3	21.5
C5-B-H1	con	June	0.148	7.1	0.3	21.5
C5-B-H2	con	June	0.210	7.1	0.3	21.5
C5-B-H3	con	June	0.080	7.1	0.3	21.5
C5-B-H4	con	June	0.326	7.1	0.3	21.5
C5-A-H1 R2	con	July	0.037	11.3	0.3	21.5
C5-A-H2 R2	con	July	0.033	11.3	0.3	21.5
C5-A-H3 R2	con	July	0.046	11.3	0.3	21.5
C5-A-H4 R2	con	July	0.007	11.3	0.3	21.5
C5-B-H1 R2	con	July	0.078	11.3	0.3	21.5
C5-B-H2 R2	con	July	0.025	11.3	0.3	21.5
C5-B-H3 R2	con	July	0.055	11.3	0.3	21.5
C7-A-H1	con	June	0.015	44.2	0.5	20.5
C7-A-H2	con	June	0.033	44.2	0.5	20.5
C7-A-H3	con	June	0.004	44.2	0.5	20.5
C7-A-H4	con	June	0.004	44.2	0.5	20.5
C7-B-H2	con	June	0.175	44.2	0.5	20.5
C7-B-H3	con	June	0.068	44.2	0.5	20.5
C7-B-H4	con	June	0.008	44.2	0.5	20.5
C7-A-H1 R2	con	July	0.019	29.2	4.0	20.5
C7-A-H2 R2	con	July	0.071	29.2	4.0	20.5
C7-A-H3 R2	con	July	0.043	29.2	4.0	20.5

Appendix Z continued:

Sample	Type	Month	nestling food biomass	mean residue	mean weed cover	mean plant species richness
C7-A-H4 R2	con	July	0.044	29.2	4.0	20.5
C7-B-H1 R2	con	July	0.021	29.2	4.0	20.5
C7-B-H3 R2	con	July	0.034	29.2	4.0	20.5
C7-B-H4 R2	con	July	0.016	29.2	4.0	20.5
C8-A-H1	con	June	0.065	82.5	14.0	15.0
C8-A-H3	con	June	0.003	82.5	14.0	15.0
C8-A-H4	con	June	0.017	82.5	14.0	15.0
C8-B-H1	con	June	0.124	82.5	14.0	15.0
C8-B-H2	con	June	0.048	82.5	14.0	15.0
C8-B-H3	con	June	0.104	82.5	14.0	15.0
C8-B-H4	con	June	0.173	82.5	14.0	15.0
C8-A-H1 R2	con	July	0.111	79.4	10.5	15.0
C8-A-H2 R2	con	July	0.198	79.4	10.5	15.0
C8-A-H3 R2	con	July	0.060	79.4	10.5	15.0
C8-A-H4 R2	con	July	0.077	79.4	10.5	15.0
C8-B-H1 R2	con	July	0.309	79.4	10.5	15.0
C8-B-H2 R2	con	July	0.173	79.4	10.5	15.0
C8-B-H3 R2	con	July	0.038	79.4	10.5	15.0
O1-A-H1	org	June	0.064	17.1	1.4	26.0
O1-A-H2	org	June	0.077	17.1	1.4	26.0
O1-A-H3	org	June	0.081	17.1	1.4	26.0
O1-A-H4	org	June	0.114	17.1	1.4	26.0

Appendix Z continued:

Sample	Type	Month	nestling food biomass	mean residue	mean weed cover	mean plant species richness
O1-B-H1	org	June	0.172	17.1	1.4	26.0
O1-B-H3	org	June	0.131	17.1	1.4	26.0
O1-B-H4	org	June	0.137	17.1	1.4	26.0
O1-A-H1 R3	org	July	0.089	1.0	66.5	26.0
O1-A-H2 R3	org	July	0.048	1.0	66.5	26.0
O1-A-H3 R3	org	July	0.037	1.0	66.5	26.0
O1-A-H4 R3	org	July	0.035	1.0	66.5	26.0
O1-B-H1 R3	org	July	0.116	1.0	66.5	26.0
O1-B-H3 R3	org	July	0.054	1.0	66.5	26.0
O1-B-H4 R3	org	July	0.124	1.0	66.5	26.0
O10-A-H1	org	June	0.007	8.0	2.2	33.0
O10-A-H2	org	June	0.036	8.0	2.2	33.0
O10-A-H3	org	June	0.020	8.0	2.2	33.0
O10-A-H4	org	June	0.082	8.0	2.2	33.0
O10-B-H1	org	June	0.085	8.0	2.2	33.0
O10-B-H2	org	June	0.023	8.0	2.2	33.0
O10-B-H3	org	June	0.004	8.0	2.2	33.0
O10-B-H4	org	June	0.077	8.0	2.2	33.0
O10-A-H1 R2	org	July	0.005	15.0	34.8	33.0
O10-A-H2 R2	org	July	0.154	15.0	34.8	33.0
O10-A-H3 R2	org	July	0.065	15.0	34.8	33.0
O10-A-H4 R2	org	July	0.031	15.0	34.8	33.0
O10-B-H1 R2	org	July	0.048	15.0	34.8	33.0
O10-B-H3 R2	org	July	0.264	15.0	34.8	33.0

Appendix Z continued:

Sample	Type	Month	nestling food biomass	mean residue	mean weed cover	mean plant species richness
O10-B-H4 R2	org	July	0.007	15.0	34.8	33.0
O11-A-H1	org	June	0.026	17.1	1.4	27.0
O11-A-H2	org	June	0.183	17.1	1.4	27.0
O11-A-H3	org	June	0.033	17.1	1.4	27.0
O11-A-H4	org	June	0.022	17.1	1.4	27.0
O11-B-H1	org	June	0.065	17.1	1.4	27.0
O11-B-H2	org	June	0.031	17.1	1.4	27.0
O11-B-H3	org	June	0.240	17.1	1.4	27.0
O11-B-H4	org	June	0.080	17.1	1.4	27.0
O2-A-H1	org	June	0.269	93.0	2.5	31.5
O2-A-H2	org	June	0.089	93.0	2.5	31.5
O2-A-H3	org	June	0.095	93.0	2.5	31.5
O2-A-H4	org	June	0.204	93.0	2.5	31.5
O2-B-H1	org	June	0.413	93.0	2.5	31.5
O2-B-H3	org	June	0.147	93.0	2.5	31.5
O2-B-H4	org	June	0.170	93.0	2.5	31.5
O2-A-H1 R2	org	July	0.215	90.2	11.3	31.5
O2-A-H2 R2	org	July	0.150	90.2	11.3	31.5
O2-A-H3 R2	org	July	0.096	90.2	11.3	31.5
O2-A-H4 R2	org	July	0.123	90.2	11.3	31.5
O2-B-H1 R2	org	July	0.236	90.2	11.3	31.5
O2-B-H2 R2	org	July	0.206	90.2	11.3	31.5
O2-B-H3 R2	org	July	0.100	90.2	11.3	31.5
O2-B-H4 R2	org	July	0.136	90.2	11.3	31.5

Appendix Z continued:

Sample	Type	Month	nestling food biomass	mean residue	mean weed cover	mean plant species richness
O3-A-H2	org	June	0.118	12.4	0.0	26.5
O3-A-H3	org	June	0.082	12.4	0.0	26.5
O3-A-H4	org	June	0.098	12.4	0.0	26.5
O3-B-H1	org	June	0.099	12.4	0.0	26.5
O3-B-H2	org	June	0.375	12.4	0.0	26.5
O3-B-H3	org	June	0.060	12.4	0.0	26.5
O3-B-H4	org	June	0.183	12.4	0.0	26.5
O3-A-H1 R2	org	July	0.167	2.6	55.0	26.5
O3-A-H2 R2	org	July	0.095	2.6	55.0	26.5
O3-A-H3 R2	org	July	0.027	2.6	55.0	26.5
O3-A-H4 R2	org	July	0.065	2.6	55.0	26.5
O3-B-H1 R2	org	July	0.017	2.6	55.0	26.5
O3-B-H2 R2	org	July	0.030	2.6	55.0	26.5
O3-B-H3 R2	org	July	0.059	2.6	55.0	26.5
O3-B-H4 R2	org	July	0.041	2.6	55.0	26.5
O4-A-H1	org	June	0.159	26.3	0.3	22.5
O4-A-H2	org	June	0.033	26.3	0.3	22.5
O4-A-H3	org	June	0.149	26.3	0.3	22.5
O4-A-H4	org	June	0.276	26.3	0.3	22.5
O4-B-H2	org	June	0.129	26.3	0.3	22.5
O4-B-H2	org	June	0.062	26.3	0.3	22.5
O4-B-H3	org	June	0.114	26.3	0.3	22.5
O4-B-H4	org	June	0.120	26.3	0.3	22.5
O4-A-H1 R2	org	July	0.020	22.5	52.5	22.5

Appendix Z continued:

Sample	Type	Month	nestling food biomass	mean residue	mean weed cover	mean plant species richness
O4-A-H2 R2	org	July	0.103	22.5	52.5	22.5
O4-A-H3 R2	org	July	0.105	22.5	52.5	22.5
O4-A-H4 R2	org	July	0.448	22.5	52.5	22.5
O4-B-H1 R2	org	July	0.059	22.5	52.5	22.5
O4-B-H2 R2	org	July	0.010	22.5	52.5	22.5
O4-B-H4 R2	org	July	0.337	22.5	52.5	22.5
O5-A-H1	org	June	0.065	30.6	4.3	18.5
O5-A-H2	org	June	0.138	30.6	4.3	18.5
O5-A-H3	org	June	0.119	30.6	4.3	18.5
O5-A-H4	org	June	0.077	30.6	4.3	18.5
O5-A-H4	org	June	0.047	30.6	4.3	18.5
O5-B-H1	org	June	0.051	30.6	4.3	18.5
O5-B-H2	org	June	0.034	30.6	4.3	18.5
O5-B-H3	org	June	0.024	30.6	4.3	18.5
O5-B-H4	org	June	0.064	30.6	4.3	18.5
O5-A-H1 R2	org	July	0.136	9.0	29.7	18.5
O5-A-H2 R2	org	July	0.044	9.0	29.7	18.5
O5-A-H3 R2	org	July	0.029	9.0	29.7	18.5
O5-B-H1 R2	org	July	0.104	9.0	29.7	18.5
O5-B-H2 R2	org	July	0.055	9.0	29.7	18.5
O5-B-H4 R2	org	July	0.064	9.0	29.7	18.5
O7-A-H1	org	June	0.063	17.1	1.4	36.5
O7-A-H2	org	June	0.053	17.1	1.4	36.5
O7-A-H3	org	June	0.003	17.1	1.4	36.5

Appendix Z continued:

Sample	Type	Month	nestling food biomass	mean residue	mean weed cover	mean plant species richness
O7-A-H4	org	June	0.072	17.1	1.4	36.5
O7-B-H2	org	June	0.105	17.1	1.4	36.5
O7-B-H3	org	June	0.054	17.1	1.4	36.5
O7-B-H4	org	June	0.006	17.1	1.4	36.5
O7-A-H1 R2	org	July	0.059	1.0	18.0	36.5
O7-A-H3 R2	org	July	0.024	1.0	18.0	36.5
O7-A-H4 R2	org	July	0.024	1.0	18.0	36.5
O7-B-H1 R2	org	July	0.247	1.0	18.0	36.5
O7-B-H2 R2	org	July	0.096	1.0	18.0	36.5
O7-B-H3 R2	org	July	0.016	1.0	18.0	36.5
O7-B-H4 R2	org	July	0.036	1.0	18.0	36.5
O8-A-H1	org	June	0.082	8.1	0.1	23.0
O8-A-H2	org	June	0.048	8.1	0.1	23.0
O8-A-H4	org	June	0.094	8.1	0.1	23.0
O8-B-H1	org	June	0.020	8.1	0.1	23.0
O8-B-H2	org	June	0.107	8.1	0.1	23.0
O8-B-H3	org	June	0.113	8.1	0.1	23.0
O8-B-H4	org	June	0.037	8.1	0.1	23.0
O8-A-H1 R2	org	July	0.010	3.4	19.8	23.0
O8-A-H2 R2	org	July	0.037	3.4	19.8	23.0
O8-A-H3 R2	org	July	0.063	3.4	19.8	23.0
O8-A-H4 R2	org	July	0.062	3.4	19.8	23.0
O8-B-H1 R2	org	July	0.036	3.4	19.8	23.0
O8-B-H2 R2	org	July	0.048	3.4	19.8	23.0

Appendix Z continued:

Sample	Type	Month	nestling food biomass	mean residue	mean weed cover	mean plant species richness
O8-B-H3 R2	org	July	0.132	3.4	19.8	23.0
O8-B-H4 R2	org	July	0.051	3.4	19.8	23.0

Appendix AA: $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values of invertebrates captured in farmland habitats.

species	nest	Field	Habitat	$\delta^{13}\text{C}$ (‰)	$\delta^{15}\text{N}$ (‰)	Julian date	year	site
GRD	918	b.corn	Corn	-21.3	8.4	74	2007	bei
SPI	918	b.corn	Corn	-18.44	9.31	74	2007	bei
GRD	943	cars.n	Corn	-19.24	6.01	93	2007	cars
ENS	943	cars.n	Corn	-27.13	6.24	93	2007	cars
HOP	943	cars.n	Corn	-27.14	-0.02	93	2007	cars
SPI	943	cars.n	Corn	-24.96	9.1	93	2007	cars
GRD	943	cars.s	Corn	-25.23	7.04	93	2007	cars
ENS	943	cars.s	Corn	-26.72	4.35	93	2007	cars
HOP	943	cars.s	Corn	-28.16	-0.5	93	2007	cars
GRD	916	cea.corn	Corn	-24.76	10.5	55	2007	cea
GRD	956	cea.corn	Corn	-22.88	9.12	34	2008	cea
GRD	982	cea.corn	Corn	-24.16	7.67	44	2008	cea
GRD	991	cea.corn	Corn	-19.06	7.02	59	2008	cea
GRD	A6	cea.corn	Corn	-25.78	8.49	62	2007	cea
CAT	916	cea.corn	Corn	-24.03	7.63	55	2007	cea
CAT	956	cea.corn	Corn	-24.59	5.56	34	2008	cea
CAT	A6	cea.corn	Corn	-24.91	5.76	62	2007	cea
ENS	916	cea.corn	Corn	-22.09	7.57	55	2007	cea
HOP	916	cea.corn	Corn	-21.22	4.48	55	2007	cea
HOP	A6	cea.corn	Corn	-24.66	3.42	62	2007	cea
SPI	916	cea.corn	Corn	-23.55	10.49	55	2007	cea
SPI	956	cea.corn	Corn	-22.98	11.41	34	2008	cea
SPI	982	cea.corn	Corn	-21.52	11.29	44	2008	cea
SPI	991	cea.corn	Corn	-20.88	11.36	59	2008	cea
SPI	992	cea.corn	Corn	-21.38	11.21	59	2008	cea

Appendix AA continued:

species	nest	Field	Habitat	$\delta^{13}\text{C}$ (‰)	$\delta^{15}\text{N}$ (‰)	Julian date	year	site
SPI	A6	cea.corn	Corn	-23.09	9.94	62	2007	cea
GRD	956	cea.soy	Corn	-23.56	7.64	34	2008	cea
GRD	982	cea.soy	Corn	-20.79	8.75	44	2008	cea
GRD	991	cea.soy	Corn	-20.88	9.36	59	2008	cea
GRD	992	cea.soy	Corn	-17.73	8.27	59	2008	cea
SPI	956	cea.soy	Corn	-21.93	11.32	34	2008	cea
SPI	982	cea.soy	Corn	-19.24	10.8	44	2008	cea
SPI	992	cea.soy	Corn	-18.3	10.64	59	2008	cea
GRD	F1	f1	Corn	-19.84	8.67	38	2006	mcf
GRD	F11	f1	Corn	-21.33	8.1	66	2006	mcf
GRD	F6	f1	Corn	-19.83	9.01	38	2006	mcf
GRD	F7	f1	Corn	-20.25	8.88	38	2006	mcf
SPI	F1	f1	Corn	-22.03	9.24	38	2006	mcf
SPI	F13	f1	Corn	-20.11	11.72	72	2006	mcf
SPI	F6	f1	Corn	-22.62	10.35	38	2006	mcf
SPI	F7	f1	Corn	-18.38	10.65	38	2006	mcf
GRD	928	f2	Corn	-20.08	8.63	75	2007	mcf
GRD	F1	f2	Corn	-25.2	7.41	38	2006	mcf
GRD	F13	f2	Corn	-22.88	8.23	72	2006	mcf
GRD	F6	f2	Corn	-24.14	8.18	38	2006	mcf
GRD	F9	f2	Corn	-23.4	9.26	54	2006	mcf
ENS	928	f2	Corn	-18.9	6.8	75	2007	mcf
SPI	928	f2	Corn	-20.33	10.76	75	2007	mcf
SPI	F1	f2	Corn	-20.78	13.55	38	2006	mcf
SPI	F13	f2	Corn	-21.57	11.48	72	2006	mcf

Appendix AA continued:

species	nest	Field	Habitat	$\delta^{13}\text{C}$ (‰)	$\delta^{15}\text{N}$ (‰)	Julian date	year	site
SPI	F6	f2	Corn	-23.22	9.83	38	2006	mcf
SPI	F9	f2	Corn	-22.52	10.56	54	2006	mcf
GRD	H1	hs	Corn	-17.35	11.58	33	2006	hui
HOP	781	mb.for.corn	Corn	-26.02	2.96	39	2007	mb
GRD	785	mb.ncorn	Corn	-22.7	10.72	41	2007	mb
SPI	785	mb.ncorn	Corn	-22.17	11.79	41	2007	mb
HOP	785	mb.scorn	Corn	-21.51	3.27	41	2007	mb
GRD	MB10	mb2	Corn	-15.36	5.56	89	2006	mb
GRD	MB2	mb2	Corn	-21.64	8.3	33	2006	mb
GRD	MB4	mb2	Corn	-18.47	8.24	51	2006	mb
SPI	MB4	mb2	Corn	-23.16	9.04	51	2006	mb
GRD	954	n1	Corn	-24.56	8.52	33	2008	noo
GRD	990	n1	Corn	-16.04	8.97	55	2008	noo
GRD	N14	n1	Corn	-15.04	6.95	77	2006	noo
GRD	N3	n1	Corn	-20.89	9.08	36	2006	noo
SPI	954	n1	Corn	-25.14	8.97	33	2008	noo
SPI	990	n1	Corn	-20.4	9.44	55	2008	noo
SPI	N14	n1	Corn	-17.43	7.29	77	2006	noo
GRD	921	n4	Corn	-25.18	7.51	33	2008	noo
GRD	1886	n4	Corn	-15.92	11.16	79	2008	noo
GRD	N12	n4	Corn	-18.8	11.18	67	2006	noo
GRD	N15	n4	Corn	-15.33	7.12	87	2006	noo
ENS	1886	n4	Corn	-19.38	8.9	79	2008	noo
SPI	1886	n4	Corn	-16.37	12.08	79	2008	noo
SPI	N12	n4	Corn	-19.87	10.62	67	2006	noo

Appendix AA continued:

species	nest	Field	Habitat	$\delta^{13}\text{C}$ (‰)	$\delta^{15}\text{N}$ (‰)	Julian date	year	site
SPI	N15	n4	Corn	-18.7	12.98	87	2006	noo
GRD	N9	noo-s.corn	Corn	-23.39	10.94	51	2006	noo
GRD	Nfledge	noo-s.corn	Corn	-20.2	8.45	51	2006	noo
SPI	N9	noo-s.corn	Corn	-22.88	12.21	51	2006	noo
SPI	Nfledge	noo-s.corn	Corn	-22.37	10.99	51	2006	noo
SPI	NS4	ns-s	Corn	-18.99	9.6	65	2006	ns
GRD	915	pc	Corn	-15.96	4.65	71	2007	pat
GRD	939	pc	Corn	-16.97	6.27	24	2008	pat
ENS	915	pc	Corn	-17.39	4.58	71	2007	pat
HOP	915	pc	Corn	-25.74	0.65	71	2007	pat
SPI	915	pc	Corn	-20.75	10.48	71	2007	pat
SPI	939	pc	Corn	-22.54	12.65	24	2008	pat
GRD	962	f1	Hay	-23.05	7.79	52	2008	mcf
HOP	962	f1	Hay	-27.5	2.13	52	2008	mcf
SPI	962	f1	Hay	-23.91	8.22	52	2008	mcf
GRD	F11	f3	Hay	-17.75	7.77	66	2006	mcf
SPI	F11	f3	Hay	-25.64	8.23	66	2006	mcf
GRD	786	han.hay	Hay	-25.82	6.14	46	2007	han
GRD	787	han.hay	Hay	-26.2	6.04	46	2007	han
GRD	959	han.hay	Hay	-27.24	6.33	38	2008	han
CAT	959	han.hay	Hay	-28.35	2.82	38	2008	han
HOP	786	han.hay	Hay	-27.46	1.64	46	2007	han
HOP	787	han.hay	Hay	-27.78	2.26	46	2007	han
HOP	959	han.hay	Hay	-27.1	0.91	38	2008	han
SPI	786	han.hay	Hay	-26.83	7.3	46	2007	han

Appendix AA continued:

species	nest	Field	Habitat	$\delta^{13}\text{C}$ (‰)	$\delta^{15}\text{N}$ (‰)	Julian date	year	site
SPI	787	han.hay	Hay	-26.21	7.31	46	2007	han
SPI	959	han.hay	Hay	-25.21	7.72	38	2008	han
GRD	953	han.wheat	Hay	-24.89	7.27	32	2008	han
CAT	953	han.wheat	Hay	-28.07	2.23	32	2008	han
HOP	953	han.wheat	Hay	-25.53	1.12	32	2008	han
SPI	953	han.wheat	Hay	-23.45	10.34	32	2008	han
GRD	MB3	mb.for.hay	Hay	-21.41	8.12	33	2006	mb
GRD	775	mb.hay	Hay	-17.67	5.57	37	2007	mb
GRD	MB10	mb.hay	Hay	-16.39	4	89	2006	mb
GRD	MB4	mb.hay	Hay	-24.31	8.33	51	2006	mb
CAT	775	mb.hay	Hay	-27.43	2.17	37	2007	mb
HOP	775	mb.hay	Hay	-26.86	3.24	37	2007	mb
SPI	MB10	mb.hay	Hay	-21.53	7.65	89	2006	mb
SPI	MB4	mb.hay	Hay	-24.98	8.87	51	2006	mb
GRD	993	mcv.e	Hay	-26.37	5.19	66	2008	mcv
CAT	993	mcv.e	Hay	-28.44	2.7	66	2008	mcv
ENS	993	mcv.e	Hay	-27.81	3.09	66	2008	mcv
HOP	993	mcv.e	Hay	-26.57	1.78	66	2008	mcv
SPI	993	mcv.e	Hay	-24.84	7.07	66	2008	mcv
GRD	782	n2	Hay	-27.74	6.03	50	2007	noo
GRD	925	n2	Hay	-27.12	6.59	83	2007	noo
GRD	954	n2	Hay	-25.79	7.48	33	2008	noo
GRD	N14	n2	Hay	-15.34	6.16	77	2006	noo
GRD	N3	n2	Hay	-26.55	6.7	36	2006	noo
ENS	925	n2	Hay	-26.09	4.39	83	2007	noo

Appendix AA continued:

species	nest	Field	Habitat	$\delta^{13}\text{C}$ (‰)	$\delta^{15}\text{N}$ (‰)	Julian date	year	site
HOP	782	n2	Hay	-27.53	0.91	50	2007	noo
HOP	925	n2	Hay	-26.39	0.54	83	2007	noo
HOP	954	n2	Hay	-27.38	1.08	33	2008	noo
SPI	782	n2	Hay	-25.92	7.19	50	2007	noo
SPI	925	n2	Hay	-19.13	9.73	83	2007	noo
SPI	954	n2	Hay	-24.34	9.16	33	2008	noo
SPI	990	n2	Hay	-19.25	11.62	55	2008	noo
SPI	N3	n2	Hay	-25.25	6.96	36	2006	noo
GRD	N9	noo-s.hay	Hay	-26.3	8.1	51	2006	noo
GRD	Nfledge	noo-s.hay	Hay	-26.63	7.69	51	2006	noo
SPI	N9	noo-s.hay	Hay	-25.62	9.42	51	2006	noo
CAT	976	Hedge.b	Hedge	-25.9	4.01	33	2008	bei
CAT	A1	Hedge.b	Hedge	-25.09	3.06	30	2007	bei
ENS	918	Hedge.b	Hedge	-23.28	4.28	74	2007	bei
HOP	776	Hedge.b	Hedge	-26.98	1.85	49	2007	bei
HOP	A1	Hedge.b	Hedge	-25.52	4.86	30	2007	bei
HOP	918	Hedge.b	Hedge	-24.33	1.6	74	2007	bei
SPI	776	Hedge.b	Hedge	-23.92	9.52	49	2007	bei
SPI	A1	Hedge.b	Hedge	-23.6	10.79	30	2007	bei
SPI	976	Hedge.b	Hedge	-21.69	9.13	33	2008	bei
SPI	918	Hedge.b	Hedge	-16.9	8.96	74	2007	bei
GRD	943	Hedge.cars	Hedge	-26.41	7.3	93	2007	cars
ENS	943	Hedge.cars	Hedge	-27.33	4.15	93	2007	cars
HOP	943	Hedge.cars	Hedge	-25.79	0.12	93	2007	cars
SPI	943	Hedge.cars	Hedge	-25.49	8.92	93	2007	cars

Appendix AA continued:

species	nest	Field	Habitat	$\delta^{13}\text{C}$ (‰)	$\delta^{15}\text{N}$ (‰)	Julian date	year	site
GRD	916	Hedge.cea	Hedge	-24.45	12.34	55	2007	cea
GRD	991	Hedge.cea	Hedge	-16.72	9.83	59	2008	cea
GRD	A6	Hedge.cea	Hedge	-16.39	10.17	62	2007	cea
CAT	A6	Hedge.cea	Hedge	-27.1	3.7	62	2007	cea
CAT	916	Hedge.cea	Hedge	-26.83	4.57	55	2007	cea
CAT	982	Hedge.cea	Hedge	-26.54	5.32	44	2008	cea
CAT	956	Hedge.cea	Hedge	-24.36	5.44	34	2008	cea
HOP	991	Hedge.cea	Hedge	-27.45	1.36	59	2008	cea
HOP	982	Hedge.cea	Hedge	-26.33	1.58	44	2008	cea
HOP	956	Hedge.cea	Hedge	-26.12	1.79	34	2008	cea
HOP	916	Hedge.cea	Hedge	-25.05	4.56	55	2007	cea
HOP	A6	Hedge.cea	Hedge	-24.7	2.07	62	2007	cea
SPI	956	Hedge.cea	Hedge	-23.54	9.87	34	2008	cea
SPI	916	Hedge.cea	Hedge	-23.42	8.23	55	2007	cea
SPI	991	Hedge.cea	Hedge	-22.3	11.65	59	2008	cea
SPI	982	Hedge.cea	Hedge	-21.97	8.79	44	2008	cea
SPI	A6	Hedge.cea	Hedge	-17.22	10.8	62	2007	cea
GRD	F6	Hedge.f1	Hedge	-25.43	8.15	38	2006	mcf
GRD	F9	Hedge.f1	Hedge	-24.68	9.54	54	2006	mcf
GRD	F1	Hedge.f1	Hedge	-24.32	7.96	38	2006	mcf
GRD	928	Hedge.f1	Hedge	-23.7	10.94	75	2007	mcf
GRD	962	Hedge.f1	Hedge	-22.16	7.5	52	2008	mcf
CAT	F13	Hedge.f1	Hedge	-26.19	0.82	72	2006	mcf
CAT	F1	Hedge.f1	Hedge	-23.99	5.91	38	2006	mcf
ENS	962	Hedge.f1	Hedge	-26.19	5.09	52	2008	mcf

Appendix AA continued:

species	nest	Field	Habitat	$\delta^{13}\text{C}$ (‰)	$\delta^{15}\text{N}$ (‰)	Julian date	year	site
ENS	928	Hedge.f1	Hedge	-23.74	3.51	75	2007	mcf
HOP	962	Hedge.f1	Hedge	-26.59	1.65	52	2008	mcf
HOP	F13	Hedge.f1	Hedge	-25.43	7.12	72	2006	mcf
HOP	F9	Hedge.f1	Hedge	-24.83	3.39	54	2006	mcf
HOP	928	Hedge.f1	Hedge	-24.54	1.32	75	2007	mcf
SPI	F13	Hedge.f1	Hedge	-25.45	7.31	72	2006	mcf
SPI	F1	Hedge.f1	Hedge	-25.43	9.13	38	2006	mcf
SPI	928	Hedge.f1	Hedge	-24.62	6.34	75	2007	mcf
SPI	F9	Hedge.f1	Hedge	-24.28	8.99	54	2006	mcf
SPI	962	Hedge.f1	Hedge	-23.22	8.08	52	2008	mcf
SPI	F6	Hedge.f1	Hedge	-22.42	12.11	38	2006	mcf
GRD	F7	Hedge.f3	Hedge	-21.56	9.45	38	2006	mcf
GRD	F11	Hedge.f3	Hedge	-21.23	8.36	66	2006	mcf
CAT	F11	Hedge.f3	Hedge	-27.18	3.45	66	2006	mcf
HOP	F7	Hedge.f3	Hedge	-24.19	2.87	38	2006	mcf
SPI	F7	Hedge.f3	Hedge	-24.43	8.18	38	2006	mcf
SPI	F11	Hedge.f3	Hedge	-22.2	9.68	66	2006	mcf
GRD	787	Hedge.han	Hedge	-24.08	10.38	46	2007	han
CAT	786	Hedge.han	Hedge	-28.3	2.43	46	2007	han
CAT	959	Hedge.han	Hedge	-28.25	2.94	38	2008	han
CAT	787	Hedge.han	Hedge	-27.76	3.2	46	2007	han
HOP	959	Hedge.han	Hedge	-27.43	1.93	38	2008	han
HOP	786	Hedge.han	Hedge	-27.26	2.68	46	2007	han
HOP	787	Hedge.han	Hedge	-25.77	1.61	46	2007	han
SPI	959	Hedge.han	Hedge	-25.96	7.78	38	2008	han

Appendix AA continued:

species	nest	Field	Habitat	$\delta^{13}\text{C}$ (‰)	$\delta^{15}\text{N}$ (‰)	Julian date	year	site
SPI	787	Hedge.han	Hedge	-25.17	9.09	46	2007	han
SPI	786	Hedge.han	Hedge	-23.26	8.52	46	2007	han
GRD	H1	Hedge.hui	Hedge	-21.79	8.97	33	2006	hui
CAT	H1	Hedge.hui	Hedge	-25.23	4.78	33	2006	hui
HOP	H1	Hedge.hui	Hedge	-25.08	6.27	33	2006	hui
SPI	H1	Hedge.hui	Hedge	-24.1	8.41	33	2006	hui
GRD	MB10	Hedge.mb	Hedge	-27.5	5.31	89	2006	mb
GRD	775	Hedge.mb	Hedge	-26.56	7.17	37	2007	mb
GRD	MB4	Hedge.mb	Hedge	-26.5	4.8	51	2006	mb
CAT	MB3	Hedge.mb	Hedge	-27.78	1.49	33	2006	mb
CAT	MB4	Hedge.mb	Hedge	-27.61	6.72	51	2006	mb
CAT	775	Hedge.mb	Hedge	-24.93	7.38	37	2007	mb
CAT	MB2	Hedge.mb	Hedge	-24.74	1.9	33	2006	mb
HOP	775	Hedge.mb	Hedge	-26.91	4.41	37	2007	mb
HOP	MB10	Hedge.mb	Hedge	-26.01	2.77	89	2006	mb
SPI	MB2	Hedge.mb	Hedge	-25.84	8.78	33	2006	mb
SPI	775	Hedge.mb	Hedge	-24.84	8.55	37	2007	mb
SPI	MB10	Hedge.mb	Hedge	-24.78	6.16	89	2006	mb
SPI	MB4	Hedge.mb	Hedge	-19.84	8.95	51	2006	mb
GRD	MB3	Hedge.mb.for	Hedge	-26	5.34	33	2006	mb
SPI	MB3	Hedge.mb.for	Hedge	-25.47	8.43	33	2006	mb
GRD	785	Hedge.mbtrack	Hedge	-28.12	4.12	41	2007	mb
CAT	785	Hedge.mbtrack	Hedge	-27.38	2.32	41	2007	mb
HOP	785	Hedge.mbtrack	Hedge	-25.96	-1.02	41	2007	mb
SPI	785	Hedge.mbtrack	Hedge	-23.62	9.05	41	2007	mb

Appendix AA continued:

species	nest	Field	Habitat	$\delta^{13}\text{C}$ (‰)	$\delta^{15}\text{N}$ (‰)	Julian date	year	site
GRD	993	Hedge.mcv	Hedge	-27	5.5	66	2008	mcv
CAT	993	Hedge.mcv	Hedge	-26.82	1.86	66	2008	mcv
ENS	993	Hedge.mcv	Hedge	-27.18	2.69	66	2008	mcv
HOP	993	Hedge.mcv	Hedge	-26.84	1.24	66	2008	mcv
SPI	993	Hedge.mcv	Hedge	-24.56	7.54	66	2008	mcv
GRD	954	Hedge.n1	Hedge	-22.43	7.15	33	2008	noo
GRD	782	Hedge.n1	Hedge	-21.16	5.22	50	2007	noo
GRD	925	Hedge.n1	Hedge	-20.02	6.63	83	2007	noo
GRD	N14	Hedge.n1	Hedge	-13.82	6.39	77	2006	noo
CAT	N3	Hedge.n1	Hedge	-25.3	2.4	36	2006	noo
ENS	925	Hedge.n1	Hedge	-15.02	7.23	83	2007	noo
HOP	782	Hedge.n1	Hedge	-26.33	2.31	50	2007	noo
HOP	954	Hedge.n1	Hedge	-26.27	1.74	33	2008	noo
HOP	925	Hedge.n1	Hedge	-26.19	2.76	83	2007	noo
HOP	N14	Hedge.n1	Hedge	-25.82	0.52	77	2006	noo
HOP	N3	Hedge.n1	Hedge	-25.69	4.2	36	2006	noo
SPI	N3	Hedge.n1	Hedge	-24.55	7.3	36	2006	noo
SPI	925	Hedge.n1	Hedge	-23.77	7.15	83	2007	noo
SPI	954	Hedge.n1	Hedge	-23.67	9.26	33	2008	noo
SPI	N14	Hedge.n1	Hedge	-23.46	7.41	77	2006	noo
SPI	782	Hedge.n1	Hedge	-23.41	7.88	50	2007	noo
GRD	767	Hedge.n2	Hedge	-23.6	8.68	37	2007	noo
CAT	767	Hedge.n2	Hedge	-26.19	5.16	37	2007	noo
HOP	767	Hedge.n2	Hedge	-26.82	4.48	37	2007	noo
SPI	767	Hedge.n2	Hedge	-23.24	9.76	37	2007	noo

Appendix AA continued:

species	nest	Field	Habitat	$\delta^{13}\text{C}$ (‰)	$\delta^{15}\text{N}$ (‰)	Julian date	year	site
GRD	N15	Hedge.n3	Hedge	-24.65	10.4	87	2006	noo
GRD	921	Hedge.n3	Hedge	-24.41	8.1	33	2008	noo
GRD	N13	Hedge.n3	Hedge	-23.92	9.41	72	2006	noo
CAT	1886	Hedge.n3	Hedge	-27.51	2.95	79	2008	noo
CAT	N13	Hedge.n3	Hedge	-26	3.63	72	2006	noo
ENS	921	Hedge.n3	Hedge	-26.86	3.22	33	2008	noo
ENS	N13	Hedge.n3	Hedge	-26.03	4.22	72	2006	noo
ENS	1886	Hedge.n3	Hedge	-18.96	7.91	79	2008	noo
HOP	N13	Hedge.n3	Hedge	-26.25	2.97	72	2006	noo
HOP	921	Hedge.n3	Hedge	-25.69	4.13	33	2008	noo
HOP	N15	Hedge.n3	Hedge	-25.59	4.14	87	2006	noo
HOP	1886	Hedge.n3	Hedge	-25.57	1.62	79	2008	noo
SPI	N13	Hedge.n3	Hedge	-24.94	7.85	72	2006	noo
SPI	N15	Hedge.n3	Hedge	-23.01	9.4	87	2006	noo
SPI	1886	Hedge.n3	Hedge	-22.38	8.8	79	2008	noo
SPI	921	Hedge.n3	Hedge	-21.23	9.19	33	2008	noo
GRD	N6	Hedge.n8	Hedge	-28.46	3.17	35	2006	noo
ENS	N6	Hedge.n8	Hedge	-27.63	5.71	35	2006	noo
SPI	N6	Hedge.n8	Hedge	-24.52	7.49	35	2006	noo
GRD	N12	Hedge.n9	Hedge	-25.31	9.29	67	2006	noo
SPI	N12	Hedge.n9	Hedge	-22.14	10.46	67	2006	noo
CAT	Nfledge	Hedge.noo-s	Hedge	-26.51	6.83	51	2006	noo
CAT	N9	Hedge.noo-s	Hedge	-24.88	5.57	51	2006	noo
ENS	Nfledge	Hedge.noo-s	Hedge	-26.13	4.4	51	2006	noo
HOP	NS4	Hedge.ns-n	Hedge	-25.31	1.68	65	2006	noo

Appendix AA continued:

species	nest	Field	Habitat	$\delta^{13}\text{C}$ (‰)	$\delta^{15}\text{N}$ (‰)	Julian date	year	site
GRD	939	Hedge.pat	Hedge	-20.8	7.91	24	2008	pat
GRD	915	Hedge.pat	Hedge	-20.44	7.05	71	2007	pat
HOP	915	Hedge.pat	Hedge	-24.82	2.68	71	2007	pat
SPI	915	Hedge.pat	Hedge	-24.39	7.58	71	2007	pat
SPI	939	Hedge.pat	Hedge	-21.71	11.16	24	2008	pat
GRD	994	Hedge.pin	Hedge	-25.68	8.33	68	2008	pin
GRD	974	Hedge.pin	Hedge	-15.3	6.02	25	2008	pin
CAT	994	Hedge.pin	Hedge	-29.13	2.66	68	2008	pin
SPI	994	Hedge.pin	Hedge	-22.59	8.52	68	2008	pin
SPI	974	Hedge.pin	Hedge	-18.3	9.66	25	2008	pin
ENS	774	Hedge.r1	Hedge	-27.1	6.12	39	2007	rat
HOP	774	Hedge.r1	Hedge	-26.01	5.81	39	2007	rat
GRD	779	Hedge.r2	Hedge	-23.62	7.93	47	2007	rat
CAT	779	Hedge.r2	Hedge	-28.35	4.97	47	2007	rat
ENS	779	Hedge.r2	Hedge	-24.28	5.01	47	2007	rat
SPI	779	Hedge.r2	Hedge	-24.72	7.62	47	2007	rat
SPI	R1	Hedge.r2	Hedge	-23.45	8.83	26	2006	rat
GRD	R5	Hedge.r3	Hedge	-18.26	2.54	65	2006	rat
GRD	R7	Hedge.r3	Hedge	-15.96	3.34	73	2006	rat
CAT	R7	Hedge.r3	Hedge	-25.88	4.78	73	2006	rat
ENS	R7	Hedge.r3	Hedge	-25.79	4.5	73	2006	rat
HOP	R7	Hedge.r3	Hedge	-26.3	4.43	73	2006	rat
SPI	R7	Hedge.r3	Hedge	-24.83	8.84	73	2006	rat
SPI	R5	Hedge.r3	Hedge	-24.07	8.08	65	2006	rat
GRD	776	b.soy	Soy	-21.72	6.98	49	2007	bei

Appendix AA continued:

species	nest	Field	Habitat	$\delta^{13}\text{C}$ (‰)	$\delta^{15}\text{N}$ (‰)	Julian date	year	site
SPI	776	b.soy	Soy	-15.25	8.97	49	2007	bei
SPI	A1	b.soy	Soy	-22.66	10.46	30	2007	bei
GRD	916	cea.soy	Soy	-21.55	11.45	55	2007	cea
GRD	A6	cea.soy	Soy	-19.98	9.04	62	2007	cea
HOP	A6	cea.soy	Soy	-24.91	3.97	62	2007	cea
SPI	916	cea.soy	Soy	-16.62	10.44	55	2007	cea
SPI	A6	cea.soy	Soy	-18.56	10.27	62	2007	cea
GRD	767	n1	Soy	-16.22	9.56	37	2007	noo
GRD	782	n1	Soy	-20.51	7.25	50	2007	noo
GRD	925	n1	Soy	-17.03	7.89	83	2007	noo
ENS	925	n1	Soy	-15.1	6.06	83	2007	noo
HOP	782	n1	Soy	-25.23	4.51	50	2007	noo
SPI	782	n1	Soy	-17.11	10.26	50	2007	noo
SPI	925	n1	Soy	-17.95	10.61	83	2007	noo
GRD	921	n3	Soy	-17.9	7.9	33	2008	noo
GRD	N13	n3	Soy	-21.01	11.03	72	2006	noo
SPI	921	n3	Soy	-22.77	9.9	33	2008	noo
SPI	1886	n3	Soy	-17.66	11.33	79	2008	noo
GRD	994	pin.n	Soy	-20.46	8.11	68	2008	pin
ENS	994	pin.n	Soy	-25.46	3.62	68	2008	pin
SPI	994	pin.n	Soy	-22.09	8.31	68	2008	pin
GRD	994	pin.s	Soy	-17.97	6.18	68	2008	pin
CAT	994	pin.s	Soy	-27.72	1.69	68	2008	pin
ENS	994	pin.s	Soy	-24.14	4.98	68	2008	pin
SPI	994	pin.s	Soy	-16.17	9.88	68	2008	pin

Appendix AA continued:

species	nest	Field	Habitat	$\delta^{13}\text{C}$ (‰)	$\delta^{15}\text{N}$ (‰)	Julian date	year	site
GRD	779	r5	Soy	-23.4	7.74	47	2007	rat
GRD	R5	r5	Soy	-18.1	4.43	65	2006	rat
GRD	R7	r5	Soy	-15.16	3.42	73	2006	rat
ENS	779	r5	Soy	-21.94	4.28	47	2007	rat
SPI	779	r5	Soy	-24.07	8.8	47	2007	rat
SPI	R1	r5	Soy	-23.91	8.77	26	2006	rat
SPI	R7	r5	Soy	-19.89	8.66	73	2006	rat

Appendix AB: Data on Song Sparrow (sosp) nestlings and cover of semi-natural habitat at nests in farmland in eastern Ontario.

nest	Mean nestling mass (g)	Total nestling mass (g)	brood parasitism	brood reduction	sosp clutch size	sosp fledglings	sosp nestling survival	Semi-natural cover within 100 m radius of nest (%)
774	43.28	12.00	1	para	2	2	1	9.36
776	53.40	13.35	0	n	4	4	1	4.66
779	29.11	9.83	1	para	1	1	1	2.43
781	22.50	7.50	0	y	5	2	0.4	35.28
782	61.04	12.23	0	n	5	5	1	52.05
784	57.18	10.38	0	n	5	5	1	49.98
785	25.80	12.90	0	n	3	2	0.66666667	11.82
787	34.00	11.33	0	n	4	3	0.75	66.65
915	46.68	13.70	1	para	2	0	0	59.34
916	49.52	12.38	0	n	5	0	0	2.07
921	54.03	10.81	0	n	5	NA	NA	6.05
925	60.90	12.18	0	n	5	5	1	51.77
928	69.68	13.94	0	n	5	5	1	5.39
937	28.00	9.33	0	y	5	0	0	36.03
939	34.64	11.55	0	n	3	3	1	51.19
943	28.10	14.05	0	n	3	2	0.66666667	1.66
956	44.20	11.05	0	n	4	NA	NA	15.84
958	53.30	13.33	0	n	5	4	0.8	53.34
959	54.10	13.53	0	n	4	4	1	93.08
960	49.04	10.48	1	para	NA	NA	NA	69.32
962	19.40	9.70	0	n	2	0	0	62.75

Appendix AB continued:

nest	Mean nestling mass (g)	Total nestling mass (g)	brood parasitism	brood reduction	sosp clutch size	sosp fledglings	sosp nestling survival	Semi-natural cover within 100 m radius of nest (%)
968	47.90	11.98	0	n	4	NA	NA	27.53
974	19.28	NA	1	para	4	0	0	2.91
976	41.10	13.70	0	n	4	3	0.75	15.92
982	43.48	12.10	1	para	3	NA	NA	1.88
990	13.93	13.93	0	y	5	NA	NA	11.07
992	38.56	NA	1	para	NA	NA	NA	2.12
994	62.10	12.42	0	n	5	5	1	34.37
995	48.50	12.13	0	n	4	4	1	48.92
1886	55.90	11.18	0	n	5	5	1	4.05
1887	45.34	11.07	0	n	4	4	1	2.67
1888	31.88	12.60	1	para	NA	1	NA	5.79
A12	47.32	11.83	0	n	4	0	0	4.28
A6	24.86	12.43	0	n	3	2	0.66666667	1.79

Appendix AC: Food availability at Song Sparrow nests in farmland in eastern Ontario. Activity biomass is measured with pitfall traps, and nestling food availability with a D-Vac suction sampler. Food availability by diet source is weighted by the proportion of nestling food predicted to come from cropped (39%) and semi-natural habitats (61%), using a stable isotope mixing model of Song Sparrow nestling diet.

nest	Activity biomass of nestling food invertebrates	Activity biomass of nestling food invertebrates weighted by diet source	Nestling food availability	Hopper and caterpillar availability	Nestling food availability weighted by diet source	Hopper and caterpillar availability weighted by diet source
774	0.2912	0.4450	0.0058	0.0035	0.0440	0.0327
776	0.0305	0.0453	0.0021	0.0004	0.0156	0.0049
779	0.1539	0.2214	NA	NA	NA	NA
781	0.1833	0.3180	0.0071	0.0033	0.0151	0.0066
782	0.0872	0.1542	0.0063	0.0037	0.0430	0.0195
785	0.2592	0.3865	0.0068	0.0030	0.0290	0.0150
787	0.0604	0.1074	0.0092	0.0021	0.0233	0.0116
915	0.1836	0.4061	0.0089	0.0032	0.0380	0.0161
916	0.1401	0.1846	0.0190	0.0147	0.0356	0.0180
921	0.0708	0.1082	0.0046	0.0031	0.0521	0.0372
925	0.2090	0.3161	0.0190	0.0048	0.0533	0.0131
928	0.1746	0.3486	0.0031	0.0004	0.0293	0.0085
939	0.1508	0.2520	0.0041	0.0010	0.0146	0.0058
943	0.4629	0.6453	0.0092	0.0026	0.0257	0.0058
956	0.1287	0.2739	NA	NA	NA	NA
958	0.0672	0.1185	NA	NA	NA	NA

Appendix AC continued:

nest	Activity biomass of nestling food invertebrates	Activity biomass of nestling food invertebrates weighted by diet source	Nestling food availability	Hopper and caterpillar availability	Nestling food availability weighted by diet source	Hopper and caterpillar availability weighted by diet source
959	0.1474	0.2698	0.0305	0.0077	0.0348	0.0156
962	0.2558	0.4100	0.0076	0.0037	0.0365	0.0240
974	0.1186	0.1417	NA	NA	NA	NA
976	0.1176	0.2406	NA	NA	NA	NA
982	0.2315	0.3283	0.0036	0.0005	0.0109	0.0040
990	0.1168	0.2169	NA	NA	NA	NA
992	0.0215	0.0315	NA	NA	NA	NA
994	0.2251	0.4681	NA	NA	NA	NA
995	0.1634	0.3244	NA	NA	NA	NA
1886	0.1561	0.2144	NA	NA	NA	NA
1887	0.1889	0.2412	NA	NA	NA	NA
A12	0.0305	0.0453	0.0021	0.0004	0.0156	0.0049
A6	0.0436	0.0589	NA	NA	NA	NA

Appendix AD: Nest survival data for Song Sparrow nests in farmland in eastern Ontario.

Site	nest	interval number	julian date	exposure (d)	Year	age of nest (d)	survival	brood parasitism	nest stage
hui	h1	1	27	3	2006	16	1	0	y
hui	h2	1	43	3	2006	.	1	0	e
hui	h2	2	46	8	2006	.	0	0	e
hui	h3	1	46	6	2006	.	0	0	e
MB	mb1	1	19	5	2006	.	1	0	e
MB	mb1	2	24	4	2006	.	0	0	e
MB	mb10	1	75	5	2006	6	1	0	e
MB	mb10	2	80	5	2006	11	1	0	e
MB	mb10	3	85	4	2006	16	1	0	y
MB	mb2	1	19	5	2006	6	1	0	e
MB	mb2	2	24	4	2006	11	1	0	e
MB	mb2	3	28	5	2006	15	1	0	y
MB	mb2	4	33	3	2006	20	1	0	y
MB	mb3	1	28	5	2006	15	1	0	y
MB	mb3	2	33	1	2006	20	0	0	y
MB	mb4	1	28	8	2006	2	1	0	e
MB	mb4	2	36	7	2006	10	1	0	e
MB	mb4	3	43	4	2006	17	1	0	y
MB	mb6	1	47	5	2006	.	1	0	e
MB	mb6	2	52	3	2006	.	0	0	e
MB	mb8	1	70	5	2006	10	1	0	e
MB	mb8	2	75	5	2006	15	0	0	y
MB	mb9	1	75	5	2006	0	0	0	e
mcf	mcf1	1	16	3	2006	6	1	0	e

Appendix AD continued:

Site	nest	interval number	julian date	exposure (d)	Year	age of nest (d)	survival	brood parasitism	nest stage
mcf	mcf1	2	19	6	2006	9	1	0	e
mcf	mcf1	3	25	3	2006	15	1	0	e
mcf	mcf1	4	28	2	2006	18	1	0	y
mcf	mcf1	5	30	4	2006	20	0	0	y
mcf	mcf10	1	34	2	2006	12	1	0	e
mcf	mcf10	2	36	3	2006	14	0	0	y
mcf	mcf13	1	59	7	2006	8	1	0	e
mcf	mcf13	2	66	5	2006	15	1	0	y
mcf	mcf13	3	71	1	2006	20	1	0	y
mcf	mcf13	4	72	1	2006	21	1	0	y
mcf	mcf14	1	67	4	2006	.	0	0	e
mcf	mcf2	1	16	3	2006	.	1	0	e
mcf	mcf2	2	19	6	2006	.	0	0	e
mcf	mcf3	1	16	3	2006	.	1	0	e
mcf	mcf3	2	19	6	2006	.	1	0	e
mcf	mcf3	3	25	3	2006	.	1	0	e
mcf	mcf3	4	28	6	2006	.	0	0	e
mcf	mcf4	2	19	6	2006	1	1	0	e
mcf	mcf4	3	25	9	2006	7	0	0	e
mcf	mcf6	1	28	6	2006	16	1	0	y
noo	n1	1	17	6	2006	.	0	0	e
noo	n12	1	52	5	2006	7	1	0	e
noo	n12	2	57	2	2006	12	1	0	e
noo	n12	3	59	8	2006	14	1	0	e

Appendix AD continued:

Site	nest	interval number	julian date	exposure (d)	Year	age of nest (d)	survival	brood parasitism	nest stage
noo	n13	1	57	2	2006	8	1	0	e
noo	n13	2	59	8	2006	10	1	0	e
noo	n13	3	67	1	2006	18	1	0	y
noo	n13	4	68	2	2006	19	1	0	y
noo	n15	1	67	3	2006	5	1	0	e
noo	n15	2	70	4	2006	8	1	0	e
noo	n15	3	74	4	2006	12	1	0	e
noo	n15	4	78	2	2006	16	1	0	y
noo	n15	5	80	2	2006	18	1	0	y
noo	n15	6	82	1	2006	20	1	0	y
noo	n18	1	74	3	2006	14	1	0	y
noo	n18	2	77	3	2006	17	1	0	y
noo	n18	3	80	1	2006	20	0	0	y
noo	n2	1	23	4	2006	.	0	0	e
noo	n3	1	23	4	2006	10	1	0	e
noo	n3	2	27	3	2006	14	1	0	e
noo	n3	3	30	6	2006	17	1	0	y
noo	n5	1	30	6	2006	9	1	0	e
noo	n5	2	36	4	2006	15	1	0	y
noo	n5	3	40	3	2006	19	1	0	y
noo	n7	1	44	8	2006	5	1	0	e
noo	n7	2	52	5	2006	13	1	0	e
noo	n7	3	57	2	2006	18	1	0	y
noo	n7	4	59	4	2006	20	1	0	y

Appendix AD continued:

Site	nest	interval number	julian date	exposure (d)	Year	age of nest (d)	survival	brood parasitism	nest stage
NS	ns1	1	35	3	2006	.	0	0	e
NS	ns4	1	44	7	2006	5	1	0	e
NS	ns4	2	51	6	2006	12	1	0	e
NS	ns4	3	57	3	2006	18	1	0	y
NS	ns4	4	60	3	2006	21	1	0	y
rat	r1	1	17	5	2006	10	1	0	e
rat	r1	2	22	5	2006	15	1	0	y
rat	r4	1	39	7	2006	.	0	0	e
rat	r7	1	59	7	2006	6	1	0	e
rat	r7	2	66	6	2006	13	1	0	e
rat	r7	3	72	1	2006	19	1	0	y
rat	r7	4	73	1	2006	20	1	0	y
SM	sm1	1	16	3	2006	5	1	0	e
SM	sm1	2	19	6	2006	8	1	0	e
SM	sm1	3	25	4	2006	14	1	0	e
SM	sm1	4	29	4	2006	18	1	0	y
noo	758	1	7	1	2007	3	1	0	e
noo	758	2	8	4	2007	4	1	0	e
noo	758	3	12	6	2007	8	1	0	e
noo	758	4	18	3	2007	14	1	0	e
noo	758	5	21	2	2007	17	0	0	y
mcf	761	1	65	3	2007	5	1	0	e
mcf	761	2	68	5	2007	8	1	0	e
mcf	761	3	73	4	2007	13	1	0	e

Appendix AD continued:

Site	nest	interval number	julian date	exposure (d)	Year	age of nest (d)	survival	brood parasitism	nest stage
mcf	761	4	77	1	2007	17	1	0	y
mcf	761	5	78	4	2007	18	0	0	y
car	762	1	10	4	2007	.	0	0	e
bei	764	1	19	3	2007	.	0	0	e
car	765	1	25	4	2007	.	1	1	e
car	765	2	29	5	2007	.	0	1	e
bei	766	1	9	5	2007	.	0	0	e
noo	767	1	18	5	2007	4	1	0	e
noo	767	2	23	2	2007	9	1	0	e
noo	767	3	25	5	2007	11	1	0	e
noo	767	4	30	4	2007	16	1	0	y
noo	767	5	34	3	2007	20	0	0	y
noo	768	1	18	5	2007	.	0	0	e
rat	769	1	18	5	2007	.	1	0	e
rat	769	2	23	3	2007	.	0	0	e
mcf	773	1	28	6	2007	6	1	0	e
mcf	773	2	34	4	2007	12	1	0	e
mcf	773	3	38	2	2007	16	1	0	y
mcf	773	4	40	4	2007	18	1	0	y
rat	774	1	30	4	2007	10	1	1	e
rat	774	2	34	3	2007	14	1	1	y
rat	774	3	37	2	2007	17	1	1	y
rat	774	4	39	3	2007	19	1	1	y
MB	775	1	30	4	2007	16	1	0	y

Appendix AD continued:

Site	nest	interval number	julian date	exposure (d)	Year	age of nest (d)	survival	brood parasitism	nest stage
MB	775	2	34	3	2007	20	1	0	y
bei	776	1	28	2	2007	2	1	0	e
bei	776	2	30	3	2007	4	1	0	e
bei	776	3	33	5	2007	7	1	0	e
bei	776	4	38	6	2007	12	1	0	e
bei	776	5	44	2	2007	18	1	0	y
bei	776	6	46	3	2007	20	0	0	y
han	778	1	34	3	2007	0	0	0	e
rat	779	1	39	6	2007	10	1	1	e
rat	779	2	45	1	2007	16	1	1	y
rat	779	3	46	3	2007	17	1	1	y
ced	780	1	22	7	2007	8	1	0	e
ced	780	2	29	5	2007	15	0	0	y
MB	781	1	22	4	2007	4	1	0	e
MB	781	2	26	4	2007	8	1	0	e
MB	781	3	30	4	2007	12	1	0	e
MB	781	4	34	3	2007	16	1	0	y
MB	781	5	37	2	2007	19	1	0	y
MB	781	6	39	1	2007	21	1	0	y
MB	781	7	40	1	2007	22	1	0	y
MB	781	8	41	1	2007	23	1	0	y
noo	782	1	25	3	2007	0	1	0	e
noo	782	2	28	2	2007	3	1	0	e
noo	782	3	30	3	2007	5	1	0	e

Appendix AD continued:

Site	nest	interval number	julian date	exposure (d)	Year	age of nest (d)	survival	brood parasitism	nest stage
noo	782	4	33	4	2007	8	1	0	e
noo	782	5	37	3	2007	12	1	0	e
noo	782	6	40	5	2007	15	1	0	e
noo	782	7	45	1	2007	20	1	0	y
noo	782	8	46	3	2007	21	1	0	y
MB	784	1	26	4	2007	4	1	0	e
MB	784	2	30	4	2007	8	1	0	e
MB	784	3	34	3	2007	12	1	0	e
MB	784	4	37	2	2007	15	1	0	y
MB	784	5	39	3	2007	17	1	0	y
MB	784	6	42	3	2007	20	0	0	y
MB	785	1	26	4	2007	4	1	0	e
MB	785	2	30	4	2007	8	1	0	e
MB	785	3	34	3	2007	12	1	0	e
MB	785	4	37	2	2007	15	1	0	y
MB	785	5	39	2	2007	17	1	0	y
MB	785	6	41	3	2007	19	1	0	y
han	786	1	28	6	2007	5	1	0	e
han	786	2	34	3	2007	11	1	0	e
han	786	3	37	7	2007	14	1	0	e
han	786	4	44	2	2007	21	1	0	y
han	787	1	28	6	2007	2	1	0	e
han	787	2	34	3	2007	8	1	0	e
han	787	3	37	7	2007	11	1	0	e

Appendix AD continued:

Site	nest	interval number	julian date	exposure (d)	Year	age of nest (d)	survival	brood parasitism	nest stage
han	787	4	44	2	2007	18	1	0	y
han	787	5	46	1	2007	20	1	0	y
han	787	6	47	2	2007	21	1	0	y
pat	915	1	47	6	2007	1	1	1	e
pat	915	2	53	5	2007	7	1	1	e
pat	915	3	58	7	2007	12	1	1	e
pat	915	4	65	1	2007	19	1	1	y
pat	915	5	66	1	2007	20	0	1	y
ced	916	1	50	3	2007	14	1	0	e
ced	916	2	53	2	2007	17	1	0	y
ced	916	3	55	2	2007	19	0	0	y
noo	917	1	63	4	2007	.	1	0	e
noo	917	2	67	5	2007	.	0	0	e
bei	918	1	65	2	2007	8	1	0	e
bei	918	2	67	4	2007	10	1	0	e
bei	918	3	71	2	2007	14	1	0	y
bei	918	4	73	1	2007	16	1	0	y
bei	918	5	74	1	2007	17	0	0	y
mcf	923	1	73	5	2007	0	1	0	e
mcf	923	2	78	4	2007	5	0	0	e
noo	925	1	75	2	2007	18	1	0	y
noo	925	2	77	1	2007	20	1	0	y
mcf	928	1	58	7	2007	1	1	0	e
mcf	928	2	65	3	2007	8	1	0	e

Appendix AD continued:

Site	nest	interval number	julian date	exposure (d)	Year	age of nest (d)	survival	brood parasitism	nest stage
mcf	928	3	68	5	2007	11	1	0	e
mcf	928	4	73	4	2007	16	1	0	y
mcf	928	5	77	1	2007	20	1	0	y
pat	929	1	58	7	2007	1	0	0	e
car	930	1	59	4	2007	.	1	1	e
car	930	2	63	8	2007	.	0	1	e
MB	940	1	65	6	2007	.	1	0	e
MB	940	2	71	6	2007	.	1	0	e
MB	940	3	77	8	2007	.	0	0	e
car	943	1	79	6	2007	7	1	0	e
car	943	2	85	6	2007	13	1	0	e
car	943	3	91	2	2007	19	1	0	y
car	943	4	93	2	2007	21	1	0	y
bei	A1	1	28	1	2007	20	0	0	y
bei	A12	3	44	5	2007	13	1	0	y
bei	A12	4	49	3	2007	18	0	0	y
pat	A3	1	38	6	2007	.	0	0	e
noo	A4	1	39	6	2007	12	1	0	e
noo	A4	2	45	4	2007	18	0	0	y
noo	A5	1	40	4	2007	0	1	1	e
noo	A5	2	44	5	2007	4	1	1	e
noo	A5	3	49	3	2007	9	1	1	e
noo	A5	4	52	6	2007	12	0	1	e
ced	A6	1	40	4	2007	1	1	0	e

Appendix AD continued:

Site	nest	interval number	julian date	exposure (d)	Year	age of nest (d)	survival	brood parasitism	nest stage
ced	A6	2	44	6	2007	5	1	0	e
ced	A6	3	50	3	2007	11	1	0	e
ced	A6	4	53	2	2007	14	1	0	y
ced	A6	5	55	2	2007	16	1	0	y
ced	A7	1	44	6	2007	6	1	0	e
ced	A7	2	50	3	2007	12	1	0	e
ced	A7	3	53	2	2007	15	0	0	y
noo	921	2	16	5	2008	3	1	0	e
noo	921	3	21	3	2008	8	1	0	e
noo	921	4	24	4	2008	11	1	0	e
noo	921	5	28	6	2008	15	1	0	e
noo	921	6	34	1	2008	21	1	0	y
noo	921	7	35	1	2008	22	1	0	y
noo	937	1	12	4	2008	7	1	0	e
noo	937	2	16	5	2008	11	1	0	e
noo	937	3	21	2	2008	16	1	0	y
noo	937	4	23	2	2008	18	1	0	y
noo	937	5	25	3	2008	20	0	0	y
MB	938	3	21	2	2008	3	1	0	e
MB	938	4	23	5	2008	5	1	0	e
MB	938	5	28	4	2008	10	0	0	e
pat	939	1	14	6	2008	8	1	0	e
pat	939	2	20	4	2008	14	1	0	y
pat	939	3	24	3	2008	18	1	0	y

Appendix AD continued:

Site	nest	interval number	julian date	exposure (d)	Year	age of nest (d)	survival	brood parasitism	nest stage
pat	939	4	27	3	2008	21	1	0	y
pat	941	1	14	6	2008	3	1	0	e
pat	941	2	20	4	2008	9	1	0	e
pat	941	3	24	3	2008	13	1	0	e
pat	941	4	27	3	2008	16	1	0	y
pat	941	5	30	3	2008	19	0	0	y
ced	942	1	14	6	2008	.	0	0	e
car	944	1	16	5	2008	11	1	0	e
car	944	2	21	4	2008	16	0	0	y
car	952	1	16	5	2008	.	0	0	e
han	953	1	14	6	2008	2	1	0	e
han	953	2	20	3	2008	8	1	0	e
han	953	3	23	6	2008	11	1	0	e
han	953	4	29	3	2008	17	0	0	y
noo	954	1	16	5	2008	3	1	0	e
noo	954	2	21	3	2008	8	1	0	e
noo	954	3	24	4	2008	11	1	0	e
noo	954	4	28	4	2008	15	1	0	e
noo	954	5	32	3	2008	19	0	0	y
ced	956	1	20	4	2008	6	1	0	e
ced	956	2	24	5	2008	10	1	0	e
ced	956	3	29	4	2008	15	1	0	y
ced	956	4	33	3	2008	19	1	0	y
MB	957	1	21	2	2008	10	1	0	e

Appendix AD continued:

Site	nest	interval number	julian date	exposure (d)	Year	age of nest (d)	survival	brood parasitism	nest stage
MB	957	2	23	5	2008	12	1	0	e
MB	957	3	28	4	2008	17	1	0	y
MB	957	4	32	3	2008	21	1	0	y
MB	958	1	21	2	2008	9	1	0	e
MB	958	2	23	5	2008	11	1	0	e
MB	958	3	28	4	2008	16	1	0	y
MB	958	4	32	1	2008	20	1	0	y
MB	958	5	33	2	2008	21	1	0	y
han	959	1	23	6	2008	5	1	0	e
han	959	2	29	3	2008	11	1	0	e
han	959	3	32	3	2008	14	1	0	y
han	959	4	35	3	2008	17	1	0	y
han	959	5	38	3	2008	20	1	0	y
han	960	1	23	2	2008	17	1	1	y
pat	961	1	27	3	2008	.	1	0	e
pat	961	2	30	5	2008	.	0	0	e
mcf	962	1	42	4	2008	9	1	0	e
mcf	962	2	46	5	2008	13	1	0	y
mcf	962	3	51	1	2008	18	1	0	y
mcf	962	4	52	3	2008	19	1	0	y
rat	965	1	51	4	2008	.	0	0	e
car	966	1	56	3	2008	.	1	1	e
car	966	2	59	3	2008	.	0	1	e
car	967	1	68	5	2008	.	0	0	e

Appendix AD continued:

Site	nest	interval number	julian date	exposure (d)	Year	age of nest (d)	survival	brood parasitism	nest stage
noo	968	1	83	1	2008	13	1	0	e
noo	968	2	84	5	2008	14	1	0	y
noo	968	3	89	2	2008	19	1	0	y
mcf	969	1	82	4	2008	.	1	0	e
mcf	969	2	86	3	2008	.	0	0	e
pin	972	1	20	5	2008	.	0	0	e
pin	973	1	20	5	2008	16	0	0	y
pin	974	1	20	5	2008	.	1	1	e
pin	974	2	25	3	2008	.	0	1	y
pin	975	1	20	5	2008	9	1	0	e
pin	975	2	25	3	2008	14	1	0	y
pin	975	3	28	2	2008	17	0	0	y
bei	976	1	20	3	2008	9	1	0	e
bei	976	2	23	5	2008	12	1	0	e
bei	976	3	28	2	2008	17	1	0	y
cla	978	1	22	6	2008	.	0	0	e
ced	979	1	24	5	2008	2	1	0	e
ced	979	2	29	7	2008	7	1	0	e
ced	979	3	36	5	2008	14	1	0	e
ced	979	4	41	3	2008	19	0	0	y
ced	981	1	29	7	2008	.	1	1	e
ced	981	2	36	5	2008	.	1	1	e
ced	982	1	29	7	2008	4	1	1	e
ced	982	2	36	5	2008	11	1	1	e

Appendix AD continued:

Site	nest	interval number	julian date	exposure (d)	Year	age of nest (d)	survival	brood parasitism	nest stage
ced	982	3	41	2	2008	16	1	1	y
ced	982	4	43	1	2008	18	1	1	y
ced	982	5	44	3	2008	19	0	1	y
car	989	1	52	4	2008	1	1	1	e
car	989	2	56	3	2008	5	1	1	e
car	989	3	59	3	2008	8	1	1	e
car	989	4	62	3	2008	11	0	1	e
noo	990	1	40	3	2008	6	1	0	e
noo	990	2	43	8	2008	9	1	0	e
noo	990	3	51	4	2008	17	1	0	y
noo	990	4	55	1	2008	21	1	0	y
noo	990	5	56	1	2008	22	0	0	y
ced	992	1	56	2	2008	.	1	1	y
mcv	993	1	56	7	2008	8	1	0	e
mcv	993	2	63	3	2008	15	1	0	y
mcv	993	3	66	2	2008	18	0	0	y
pin	994	1	66	2	2008	18	1	0	y
pin	994	2	68	3	2008	20	1	0	y
noo	995	1	67	4	2008	4	1	0	e
noo	995	2	71	6	2008	8	1	0	e
noo	995	3	77	6	2008	14	1	0	y
noo	995	4	83	1	2008	20	1	0	y
noo	996	1	43	8	2008	3	1	0	e
noo	996	2	51	4	2008	11	1	0	e

Appendix AD continued:

Site	nest	interval number	julian date	exposure (d)	Year	age of nest (d)	survival	brood parasitism	nest stage
noo	996	3	55	3	2008	15	1	0	e
noo	996	4	58	5	2008	18	1	0	y
noo	998	1	43	8	2008	.	0	0	e
noo	1886	1	67	4	2008	11	1	0	e
noo	1886	2	71	2	2008	15	1	0	y
noo	1886	3	73	3	2008	17	1	0	y
rat	1887	1	68	3	2008	1	1	0	e
rat	1887	2	71	4	2008	4	1	0	e
rat	1887	3	75	7	2008	8	1	0	e
rat	1887	4	82	4	2008	15	1	0	y
rat	1887	5	86	1	2008	19	1	0	y
cla	1888	1	68	5	2008	10	1	1	e
cla	1888	2	73	3	2008	15	1	1	y
cla	1888	3	76	2	2008	18	1	1	y