Nesting and brood-rearing opportunities for farmland birds in and around Miscanthus and short rotation coppice biomass crops

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I confirm that this thesis is my own work other than where explicitly referenced. Data for Chapters 2, 3 and 4 were collected with two MSc students, Edward Boyle and Kathleen Killen under my supervision. Jenny Bright at the RSPB kindly provided data previously collected in a 2009 Natural England study, which was analysed in Chapter 2.

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Abstract

Reaching the EU target for renewable energy is likely to encourage a rapid expansion of biomass crop production in the UK. This expansion could pose a considerable threat to farmland birds, via the loss of suitable nesting habitat for ground-breeding birds or by decline in availability and accessibility of foraging resource for other farmland birds. Alternatively, the low-input management of biomass crops may provide benefits over more intensively managed arable crops. The potential impacts of biomass expansion on nesting and foraging opportunities were therefore investigated by examining the nesting success of lapwings (*Vanellus vanellus*) in Miscanthus, and the faecal components of songbirds nesting in boundaries of Miscanthus and short rotation coppice (SRC) willow fields.

Despite similar nest densities and field occupancy across crop types, lapwing clutches experienced lower hatching success in Miscanthus fields than in arable crops. This result was mainly influenced by increased losses in 2012; a particularly wet breeding season. In years of already unfavourable breeding conditions, nests in Miscanthus may therefore be more vulnerable and suffer higher predation rates than those in arable crops.

More invertebrates were found in SRC crop than in Miscanthus, and this was reflected in the faeces of whitethroat (*Sylvia communis*) chicks nesting in the boundaries of SRC and Miscanthus fields. Despite lower availability of Coleoptera in Miscanthus fields than SRC, this was not reflected in faeces of whitethroats or yellowhammers (*Emberiza citronella*), suggesting birds nesting in Miscanthus may supplement chick diet with food found elsewhere.

Converting land to biomass crop cultivation may pose some risk to farmland birds, but while nesting and foraging opportunities may be limited in Miscanthus crop, SRC provides a more abundant foraging resource. The extent of any risk will depend on the crops being replaced, the position of biomass crops within the landscape and the extent of biomass crop uptake throughout the UK. It is vital that government energy targets are met in a sustainable manner, which could be achieved if the expansion of the industry is managed sensitively.

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1 Introduction

1.1 Biomass crops

Arguably the most pressing issue facing the planet is that of climate change. It is no longer perceived as an issue just for ecologists and conservationists concerned with protecting endangered species, but as the grave threat it poses to all aspects of life. Direct effects of temperature increases, sea level rises and increased frequency of extreme weather events such as droughts and floods will be seen on human health, migration, economy, food security and international relations (Cohen & Thompson 2012; IPCC 2014; WHO 2014). In addition, the species extinctions (Thomas *et al.* 2004; Sekercioglu *et al.* 2008; Pimm 2008), changes in phenology and migration events (Fitter & Fitter 2002; Walther *et al.* 2002), and contraction, expansion or shifts in species' ranges that are expected consequences of climate change (Parmesan *et al.* 1999; Parmesan & Yohe 2003; Root *et al.* 2003; Thomas, Franco, & Hill 2006) will impact on humans (Loeb *et al.* 1997; Walther *et al.* 2002; IPCC 2014). The full extent of these impacts are not yet known, given the complex interactions between species; e.g. local extinction of any one species may have knock-on effects for the food-web from which it has been removed, causing disruption throughout the ecosystem that could not have been predicted (Loeb *et al.* 1997; Walther *et al.* 2002).

Limiting climate change in the future requires reduction of greenhouse gas emissions. Recently, these emissions (of carbon dioxide; CO_2, nitrous oxide; N_2O and methane; CH_4) have been the highest in history, "and are extremely likely to have been the dominant cause of the observed warming since the mid-20th century" (IPCC 2014). As the energy sector is the main contributor of greenhouse gas emissions (*loc cit*), particular attention has been paid to how reductions in this area can be achieved. Investment in renewable energies is therefore an integral part of government energy strategies around the world (Gasparatos, Stromberg, & Takeuchi 2011), acting to reduce our reliance on fossil fuels. Global interest in the production of energy from agricultural sources, be this in liquid form from sugars, starches and vegetable oils (first generation bioenergy) or solid biomass from agro-forestry and grass crops (second generation bioenergy) is gathering momentum (Eggers *et al.* 2009; Gasparatos, Stromberg, & Takeuchi 2011). The removal of CO_2 during the

growth of these crops results in lower carbon emissions throughout the entire life cycle of the energy production process. The demand for these 'green' energies is therefore driven by issues of climate change, as well as those of energy security, rising energy prices and rural development (Parris 2004).

Currently, biomass contributes 10% to the global energy supply, mostly from traditional sources such as burning wood, charcoal and straw, whilst only 2% of this comes from modern bio-energy providing heat, power and transport (Slade *et al.* 2011). If global primary energy demand doubles as predicted, the contribution from biomass sources will be required to increase, which must be environmentally and economically sustainable. There are currently many areas of concern regarding the extent to which biomass can contribute to the energy demand; mainly stemming from the sustainability of increasing crop yields and the associated agricultural intensification, competition with food supply, and environmental damage caused by land use change (Slade *et al.* 2011; Gasparatos, Stromberg, & Takeuchi 2011).

The EU has committed to a target of a 20% share of renewable energies in its overall energy consumption by 2020 (Defra 2007a). Reaching this requires rapid expansion of energy crop production across the member states. To contribute to this target, the UK has agreed that 15% of its total energy, and 12% of its heat energy is to be generated from renewable sources (Insitute for European Environmental Policy 2008; European Commission 2009). To achieve this ambitious target, it has been suggested that domestic biomass will play a key role (Rokwood 2014; DECC 2012), with agricultural residues and energy crops providing the greatest increase to domestic bioenergy supply by 2030 (DECC 2012). In England, the area planted with biomass crops increased more than seven times from less than 2000ha in 2003 to an estimated 15000 ha in 2008 (Haughton *et al.* 2009), and Defra (2007a) suggested this could increase to a potential 1.1 million ha by 2020.

The two main crops grown for energy in the UK are Miscanthus (*Miscanthus x giganteus*) and short rotation coppice (SRC). SRC in the UK is mostly willow, *Salix* spp., but can also include poplar, hazel, silver birch, sycamore, sweet chestnut and lime. For the purposes of this study, SRC refers solely to SRC willow. *Miscanthus x giganteus* was introduced to Denmark from Japan in the 1930s, and since the 1980s has undergone several programmes of field trials to investigate its potential as a

biomass crop (Lewandowski *et al.* 2000). It uses the C_4 pathway and as such tends to produce higher yields than C_3 plants, although this can also raise the temperature required to initiate growth (Lewandowski *et al.* 2000). The fact that Miscanthus is highly adaptable to different environments, whilst being sterile (owing to its triploid genotype), means it can be cultivated in a range of different non-native areas, whilst posing low risk of invasion (Lewandowski *et al.* 2000). Miscanthus is a perennial grass, planted as rhizomes in the spring and harvested each year, after which it grows rapidly, reaching heights of up to 3m by July (Anderson, Haskins, & Nelson 2004). Translocation of nutrients from rhizomes to shoots in spring and back again at the end of the growing season reduces the demand for nitrogen applications (Beale & Long 1997; Lewandowski *et al.* 2000; Karp *et al.* 2009). As an introduced species, Miscanthus is not yet vulnerable to any significant invertebrate pest populations in the UK (Semere & Slater 2007a) and as such, insecticide treatments are not needed, although there may be some potential for pests to feed on it (Huggett, Leather, & Walters 1999; Bradshaw *et al.* 2010).

SRC is planted as cuttings and produces woody shoots which are harvested every two to three years (Sage, Cunningham, & Boatman 2006). Breeding programmes over the last 20 years involving hundreds of willow species with differing life history traits and pest and disease resistance have led to yield increases of 8-143% in the UK (Keoleian & Volk 2005). Although SRC is host to a number of insect pest species, notably blue and brown willow beetles, *Phratora vulgatissima* and *Galerucella lineola* (Sage *et al.* 1999; Cunningham *et al.* 2006), it has a high damage threshold; severe defoliation can occur before effects on yield are seen (Bach 1994; Sage 1998; Bjorkman *et al.* 2000). There is therefore little incentive to employ chemical pest control (Dauber, Jones, & Stout 2010), rather alternative methods are advised for reasons of practicality, economics and biodiversity (Sage & Tucker 1997). Other than in the early stages of establishment, weed control on Miscanthus or SRC is unnecessary and not commercially viable due to the effort and machinery required for its deployment (Sage 1998; Anderson, Haskins, & Nelson 2004). After two to three years, the crop canopy closes completely, which acts to shade out the understory, and can prevent weed growth altogether (Bullard 2000; Defra 2007b).

Both crops have the potential for high biomass production within a short space of time following harvest or coppicing, with few chemical inputs required. As a result, Miscanthus and SRC differ greatly from traditional arable crops in their requirements, management and structure (see Appendix I for examples of biomass crop structure), and thus the potential wildlife they can support. The associated change in land use from arable to energy crop production may therefore be of concern (Anderson, Haskins, & Nelson 2004). This has even greater import against the background of biodiversity decline currently being experienced on UK farmland, particularly that of birds (e.g. Fuller *et al.* 1995; Siriwardena *et al.* 1998; Chamberlain *et al.* 2000; Gregory *et al.* 2005; Boatman, Parry, & Bishop 2007; Butler, Vickery, & Norris 2007; Stoate *et al.* 2009).

The magnitude of any impacts of changing land use on farmland birds will depend on the extent of the conversion (Anderson, Haskins, & Nelson 2004; Anderson & Fergusson 2006; Gove & Bradbury 2010). Currently, the amount of land used for biomass production in the UK is fairly low relative to the total area of agricultural land; 0.06 to 0.1% of agricultural land depending on which estimate is used (Defra 2007a; RELU 2009; NNFCC 2012; Defra *et al.* 2014). This will have to expand in future if biomass is to make any significant contribution to the energy sector (Lindegaard 2013).

Theoretically, 3,120,173 ha of land in England is suitable for energy crop planting, which represents 23.9% of the total land area (Karp *et al.* 2009). The amount of land actually used for energy crop cultivation will be limited by infrastructure, economics, policy and environmental demands. For example, transport of biomass to the end user will impose both a financial and environmental cost, thus a sourcing radius of 25km has often been suggested, and in fact was required under the Energy Crops Scheme (Thomas, Bond, & Hiscock 2013), but this stipulation was later altered to “a reasonable distance” (Natural England 2013a). Based on current locations of potential end users, this alters the amount of land that could be used for energy crop production to 1,998,435 ha (Thomas, Bond, & Hiscock 2013). Demand and supply chain infrastructure will affect the amount of land dedicated to energy crops, as without these, there will be a reluctance to invest money and land in energy crop production (NNFCC 2012). The industry is still

regarded as fairly novel, thus many farmers perceive it as potentially risky and are reluctant to change without seeing successful examples nearby to follow (Convery *et al.* 2012). Investment in infrastructure, education and promotion of the industry and increased energy demand could, however, lead to considerable expansion of the energy crop market.

A key barrier to the growth of the domestic biomass industry is lack of policy support. In England, the production of biomass crops was supported by the Energy Crops Schemes (Natural England 2013a). These covered the period 2000-2006, during which time farmers received up to £1000 ha^{-1} planted (Defra 2003); and 2007-2013, when farmers received a grant covering 40-50% of the establishment costs (Mawhood 2015). Despite these schemes, the total planted area has fallen since 2009 to an estimated 10000-12000 ha (NNFCC 2012; Defra 2013; Table 1.1), with uptake being considerably below expectations (Lindegaard 2013).

Table 1.1 The area of land planted with Miscanthus and SRC in the UK under two energy crop schemes (ECS1 and ECS2). (Source: NNFCC 2012).

	Total area (ha) planted under ECS1 (2000-2006)		Total area (ha) planted under ECS2 (2006-2012)	
Region	Miscanthus	SRC	Miscanthus	SRC
East Midlands	1890	609	333	264
East of England	381	76	93	40
North East	0	228	0	0
North West	63	125	0	0
South East	305	257	95	18
South West	1036	31	387	11
West Midlands	859	27	374	0
Yorkshire and Humber	1843	464	216	89
Subtotal	6377	1817	1498	422
Total			10114	

The current Energy Crops Scheme ended in 2013, with planting covered until 2015, but beyond that, no decision has been taken to continue this support in the future (Natural England 2013b). Indirect support continues from the Renewables

Obligation (RO) and the Renewable Heat Incentive (RHI), which provide payments to plants and domestic systems that are fuelled by biomass crops (DECC 2012; Mawhood 2015). These schemes have the potential to enhance market demand for biomass, the benefits of which may then trickle down to growers (Lindegaard 2013), but this is likely to be insufficient to achieve the growth predicted by the biomass strategy (Defra 2007a). To meet our energy targets, however, the role of domestic biomass supply must increase (DECC 2012, Rokwood 2015), thus although the current rate of planting may appear stagnant, with appropriate policy support, this could increase rapidly. This is further discussed in Chapter 5.

1.2 Agricultural intensification and farmland birds

The decline in farmland birds is well documented, showing both population decreases and range contractions. Populations of the 19 species that make up the farmland bird indicator have halved over the last forty years, whilst overall breeding bird populations have changed little (Fuller *et al.* 1995; Defra 2014). Although these declines mostly occurred between the late 1970s and early 1990s, the trend has continued with an on-going decline of 10% between 2007 and 2012 so that in 2013, breeding farmland bird populations in the UK were at their lowest level ever recorded (Defra 2014).

The declines have coincided with periods of agricultural intensification (Fuller *et al.* 1995; Chamberlain *et al.* 2000), and have been most severe in countries with the most intensive agriculture (Donald, Green, & Heath 2001; Wilson, Evans, & Grice 2009), whilst recoveries have occurred following the introduction of organic farming (Hole *et al.* 2005) and set-aside introduction (Henderson *et al.* 2000; Wilson, Evans, & Grice 2009), both of which can be seen as reversals in intensification. Advancements have been made in three main areas to improve productivity of the farmed landscape; in machinery, agrochemicals and plant breeding technology. Development of machinery such as tractors and combine harvesters led to a reduction in mixed farming, the disappearance of hedgerows and other non-farmed features, and a shift towards larger fields (for reviews see Boatman, Parry, & Bishop 2007; Stoate *et al.* 2009; Wilson, Evans, & Grice 2009). The resultant loss of habitat heterogeneity has important negative implications for farmland birds (Fuller *et al.*

1995; Siriwardena *et al.* 2000). The fact that horses were no longer needed for farming operations meant it was no longer essential for every farm to have cereal fields to provide feed, thus an important seed resource was lost (Wilson, Evans, & Grice 2009).

The advent of agrochemicals had the direct effect of reducing the food available to birds in the form of seeds, weeds and invertebrates (Gregory, Gibbons, & Donald 2004; Hart *et al.* 2006) but also led to disrupted breeding opportunities of ground-nesting birds as fertiliser use increased the frequency and advanced the onset of grass cutting (Wilson, Evans, & Grice 2009). The development of hardier crops, in combination with pre-emergent herbicides and more efficient methods of harvesting and drilling led to the switch from spring to autumn sowing. This led to a reduction in the area of over-winter stubbles whilst increasing sward height during the breeding season thereby reducing the suitability of important nesting habitat (Devereux *et al.* 2004; McCracken & Tallowin 2004; Wilson, Evans, & Grice 2009). In addition, a shift to earlier harvest dates, increased land drainage and intensification of grasslands have been detrimental to farmland birds (Vickery *et al.* 2001; Newton 2004). These key changes in agricultural practice have caused bird declines through the depletion of food resources (Potts 1986; Campbell *et al.* 1997; Brickle *et al.* 2000; Benton *et al.* 2002; Boatman *et al.* 2004) and degradation of habitat (Galbraith 1988b; Chamberlain *et al.* 1999; Donald *et al.* 2001; Newton 2004). Both these mechanisms will be discussed further in sections 1.4.2 and 3.1.1.

As pressure on agriculture increases and changes in land use continue, the threat to biodiversity is apparent. Target 7 of the Convention on Biological Diversity set out the goal of “ensuring that by 2020 areas under agriculture, aquaculture and forestry are managed sustainably, ensuring conservation of biodiversity” (Defra 2011). As part of this, the EU aims to halt declines in farmland birds by 2020 (Bradbury & Kirby 2006; Henderson *et al.* 2012). If these targets are to be reached, it is essential that any threats posed by changing land use be investigated.

1.3 Birds in biomass crops

Research into the impacts of energy crop cultivation on biodiversity has thus far been fairly limited considering the scale of biomass production anticipated, and has

focussed on species richness and densities in these crops (Dauber, Jones, & Stout 2010). More is known of the biodiversity effects of SRC than Miscanthus as it is the more established of the two, but overall, positive effects of biomass production are suggested (Cunningham *et al.* 2004; Sage, Cunningham, & Boatman 2006; Dauber, Jones, & Stout 2010). Low management intensity, lack of soil disturbance and reduced agricultural inputs have been cited as reasons for the higher breeding densities, abundances and diversity of bird species found in SRC and Miscanthus than in cereals (Berg 2002; Semere & Slater 2007b; Bellamy *et al.* 2009). A large proportion of the studies reviewed by Dauber *et al*. (2010) are, however, of limited sample size, lack controls, and are conducted on experimental plots rather than those for large-scale commercial production, so extrapolating these results is unwise.

Effects are likely to vary between species and through time, so the rapidly changing structure of SRC and Miscanthus may attract a greater range of species throughout the breeding season than traditional crops (Sage *et al.* 2010). There are concerns that this could be to the detriment of those birds characteristic of more open landscapes, e.g. lapwing, skylark (*Alauda arvensis*), meadow pipit (*Anthus pratensis*) and corn bunting (*Miliaria calandra*) as these will be replaced by scrub and woodland birds as the crops mature (Anderson, Haskins, & Nelson 2004; Gove & Bradbury 2010). Those that are most often found in developing SRC during the breeding season are widespread species such as blackbird (*Turdus merula*), wren (*Troglodytes troglodytes*) and robin (*Erithacus rubecula*), but red-listed reed buntings (*Emberiza schoeniclus*) and song thrush (*Turdus philomelos*) are also often found in the crop (Kavanagh 1990; Göransson 1994; Sage & Robertson 1996; Coates & Say 1999). It has been suggested, however, that biomass crops in the first few years of establishment or immediately following harvest could be beneficial to ground-nesting birds (Anderson, Haskins, & Nelson 2004; Bellamy *et al.* 2009). The potential impacts of biomass expansion on ground-nesting birds are investigated in Chapter 2, wherein the nesting opportunities available to lapwings (*Vanellus vanellus*) in biomass crops are assessed.

The foraging potential of biomass crops has been poorly examined; despite some work investigating the invertebrate populations supported by biomass crops, the extent to which this food supply is accessible and used by birds has yet to be

examined, other than one small scale study and observational work (Sharples 1997; Sage, Cunningham, & Boatman 2006). Low herbicide applications in biomass crops allow a range of ground vegetation to develop, (Sage & Robertson 1994), providing food and habitat for invertebrates that further benefit from a lack of insecticide treatments. Higher invertebrate species richness has been recorded for both SRC and Miscanthus compared with cereal crops (Sage & Tucker 1997; Dauber, Jones, & Stout 2010). Given that a diverse and abundant invertebrate diet is crucial for chick growth and development (O'Connor & Shrubb 1986; Moreby & Stoate 2001), this suggests potential benefits to birds foraging in biomass crops (Dauber, Jones, & Stout 2010). There is evidence, however, that these benefits depend on crop type and management. While SRC supports a diverse and numerous invertebrate fauna, regardless of weed content (Sage & Tucker 1997; Rowe *et al.* 2011), it is mainly the weeds within Miscanthus that contribute to the invertebrate composition of the crop rather than the canopy itself (Semere & Slater 2007a; Bellamy *et al.* 2009).

An area of concern for the future development of biomass crops is that as the crops become established and are more intensively managed, the potential benefits to farmland birds may be negated as the canopy closes over, and shades out weeds resulting in low invertebrate densities, especially in Miscanthus (Bellamy *et al.* 2009; Gove & Bradbury 2010). It is not merely the abundance of food that will determine the quality of the habitat, but its accessibility (Devereux *et al.* 2004), thus raising questions over the foraging potential of established Miscanthus and SRC crops. The presence of ground flora within the crops may also provide nesting opportunities for some birds (e.g. reed bunting and willow warbler; Sharples 1997), or enhance structural complexity thus attracting more birds, so the absence of a weed layer in future commercial crops would be detrimental (Sage & Robertson 1996). The foraging potential of biomass crops is investigated in Chapter 4, as the availability of invertebrate food within biomass crops is examined, and compared with the dietary intake of two farmland birds, whitethroat (*Sylvia communis*) and yellowhammer (*Emberiza citronella*).

1.4 Lapwings (*Vanellus vanellus*)

The lapwing is Britain's most numerous and widely distributed breeding wader but it has suffered significant declines over the last 40 years. This has led to its red-listed conservation status (Eaton *et al.* 2009) and its inclusion in the UK Government's Biodiversity Action Plan (BAP) as a priority species. The decline has been attributed to reduced reproductive output caused by deterioration in breeding habitat (Newton 2004; Gregory *et al.* 2005), raising concerns over the future of this species in the face of continued alterations to the farming landscape (Galbraith 1988b; Peach, Thompson, & Coulson 1994; Fuller *et al.* 1995; Henderson *et al.* 2002; Newton 2004).

1.4.1 Breeding ecology and habitat requirements

Lapwings arrive on breeding grounds from as early as mid-January still largely in their foraging flocks, which disperse a month later (Cramp & Simmons 1983; Parish 1996). Eggs (usually four) are laid in scrapes built by the male, are incubated by both parents, and hatch 25 days after clutch completion (Sheldon, Chaney, & Tyler 2007). Chicks leave the nest shortly after hatching, sometimes within hours and are led by the adults to foraging grounds (Johansson & Blomqvist 1996) and are considered fledged at 35 days old (Shrubb 2007). First clutches can be laid from late March, with the peak in breeding activity usually in mid-April (Shrubb 2007). A single-brooded species, although occasional double brooding can occur (Blomqvist & Johansson 1994; Parish, Thompson, & Coulson 1997), lapwings will lay up to four replacement clutches after failure, hence breeding can continue to June (Berg, Lindberg, & Kallebrink 1992; Wilson, Evans, & Grice 2009).

The majority of lapwing nest failures are due to predation, regardless of habitat, although nests in arable and improved grasslands also suffer losses to agricultural activities, to a lesser extent (Galbraith 1988b; Baines 1990; Shrubb 2007). To minimise predation risk, a combination of strategies are used, namely early visual detection of predators (Galbraith 1988b; Shrubb 2007), egg and chick crypsis (Šálek & Cepáková 2006), and co-operative nest defence (Berg 1993; Šálek & Šmilauer 2002), where the predator is mobbed by several individuals and excluded

from the territory. The efficacy of the first two of these strategies will depend on the nesting habitat (Whittingham & Evans 2004; Evans 2004; Wilson, Whittingham, & Bradbury 2005). To ensure an unobstructed view from the nest, lapwings require short sward height, which is reflected in the negative relationship between sward height and presence of nests or breeding pairs (Milsom *et al.* 2000; Hart *et al.* 2002; Nicholls 2010) and an avoidance of swards above 10cm (Durant *et al.* 2008; Eglington *et al.* 2009).

Sward height may also influence breeding site choice through its effect on chicks. Taller vegetation impedes access to foraging sites and reduces foraging rate of chicks by disrupting their detection of prey (Devereux *et al.* 2004; Butler & Gillings 2004; Bradbury & Kirby 2006), as such lapwing chicks tend to be found foraging in short vegetation (Beintema *et al.* 1991; Durant *et al.* 2008). Additionally, tall, dense vegetation retains more moisture following rainfall or periods of frost or dew, which could be energetically costly for chicks that are unable to thermo-regulate (Whittingham & Evans 2004; Butler, Bradbury, & Whittingham 2005). It has, however, been suggested that fields with areas of taller vegetation in which chicks can hide, are ideal nesting habitat and improve fledging success (Devereux *et al.* 2004; Butler & Gillings 2004). Variation in sward height also aids nest crypsis (Newton 2004; MacDonald & Bolton 2008b) and increases abundance and diversity of invertebrate prey in close proximity to the nest (Vickery *et al.* 2001; McCracken & Tallowin 2004).

Food availability is an important facet of breeding habitat selection, affecting the survival of chicks, both directly and indirectly via the foraging efficiency of adults. Good food availability during egg formation may lead to larger eggs being laid, with consequently higher hatch rates and better growth and fitness of chicks (Lislevand *et al.* 2005). If during incubation, there is good foraging resource available, this minimises the amount of time adults spend off the nest looking for food, which may reduce mortality (Wright *et al.* 1988; Evans 2004). Unattended nests will be more at risk of predation and exposure to the elements, while increased traffic to and from the nest may attract attention (Evans 2004). Additionally, if adults have to fly to foraging grounds, this may also aid detection of nests (Berg 1993).

Exactly how important foraging availability is to nest site selection is unknown; Berg (1993; 2002) found the presence of foraging habitat exerted a stronger influence on territory establishment than the availability of nesting habitat, while Galbraith (1989) found habitat characteristics that increased clutch and chick crypsis thereby reducing the likelihood of detection by predators to be more important than food availability. Once hatched, chicks often move to foraging grounds nearby to optimise their food intake, but it is at this time that they are particularly vulnerable (within the first ten days of hatching; Galbraith 1988b), due to an inability to regulate body temp and poor feeding efficiency, but mainly the high risk involved in travelling to foraging grounds. The shorter the distance to the foraging grounds, the lower the risk; of chicks that had to cross intervening cereal fields to reach pasture, only 8.7% survived and higher chick survival rate was attributed to the fact that growth retardation by poor weather allowed chicks to stay in natal fields for foraging (Galbraith 1988b; Wilson, Vickery, & Browne 2001). Lapwings therefore express a strong selection for spring-tilled fields that are adjacent to grass fields (Wilson, Vickery, & Browne 2001). Variation in food availability had only minor effects on breeding success of lapwings in Upper Teesdale, but did result in slower growth rates of chicks, suggesting a more limiting factor at play, probably predation (Baines 1990).

The wetness of the habitat is an important factor in breeding success (Berg 1993). As a surface and sub-surface invertebrate feeder, soil penetrability, which is closely linked to wetness, largely determines food accessibility for lapwings (Eglington *et al.* 2008). Earthworms are a key component of lapwing diet, but these will retreat deeper into the soil and out of the reach of beaks of many waders as the ground dries (Beintema *et al.* 1991). Surface water may introduce aquatic invertebrates as a resource, and may also retard vegetation growth, thus improving foraging opportunities (Ausden, Sutherland, & James 2001). It has even been suggested that mammalian predators avoid travelling over wet soil to enter fields (Smart *et al.* 2006). It is not surprising, therefore, that lapwing territories often contain a high proportion of flooded tillage and grassland (Berg 1993). This implies adults may be selecting nest sites based primarily on foraging requirements of the chicks, as broods with access to wet features and moist soil experience higher success rates than those in dry sites (Eglington *et al.* 2008; Bellebaum & Bock 2008).

Taking all these requirements into account, the optimal breeding habitat for lapwings contains a short, sparse heterogeneous sward with access to wet features (Smart *et al.* 2013). A number of studies have documented higher breeding densities and success on arable than other habitat types (e.g. Chamberlain & Crick 2003; Sheldon *et al.* 2004) although Galbraith (1988b) and Baines (1989) found grass to be more productive, probably due to greater loss to farming operations in arable habitats. Within arable habitats, breeding densities and success rates are consistently higher in spring cereals (Galbraith 1988b; Wilson, Vickery, & Browne 2001), whilst autumn-sown crops tend to be avoided, showing the importance of habitat structure for lapwings, given this is the only way in which these crops differ (Galbraith 1988b; Berg *et al.* 2002). Habitat requirements do, however, change throughout the breeding season, from the short, open swards of the natal ground, to damp invertebrate-rich foraging grounds. This was emphasised in a recent study of lapwings in the Avon Valley, in which no habitat feature exerted a constant effect throughout the breeding season. As an example, variation in sward structure positively influenced pair densities and the survival rate of nests but negatively affected the survival rate of broods (Nicholls 2010). This indicates the need for a diverse mosaic of habitats, to enhance overall reproductive productivity (Redfern 1982; Berg 1993; Nicholls 2010).

1.4.2 Reasons for lapwing decline

The decline in the reproductive output of lapwings in the UK (especially marked in 1953-1990) has coincided with a period of significant agricultural intensification, and has been attributed to the associated decline in habitat and food supply (Newton 2004). Predation has also increased over this time, which has reduced productivity (Chamberlain & Crick 2003; Roodbergen, Werf, & Hötker 2012), but such is the interaction between agricultural intensification and predation rates (Evans 2004; Roodbergen, Werf, & Hötker 2012) that separating the effects of these two factors and attributing the lapwing declines to just one cause is very challenging.

One of the key ways in which agricultural intensification has affected lapwing productivity is via changing sward structure and height. Autumn sown crops reach an unsuitable height much earlier in the breeding season than those sown in the

spring, and lack sward heterogeneity (Galbraith 1988b; Wilson, Vickery, & Browne 2001). The switch to autumn sowing has therefore been detrimental to lapwings, both by reducing the area of suitable nesting habitat, and by curtailing the length of the breeding season, thus minimising the opportunity for replacement clutches (Galbraith 1988b; Wilson, Vickery, & Browne 2001).

Additionally, increased use of fertiliser and improvement of meadows and grasslands leads to dense, homogenous swards, unsuitable for breeding and foraging (Hudson, Tucker, & Fuller 1994; Vickery *et al.* 2001; McCracken & Tallowin 2004). High fertiliser inputs also tend to lead to a reduction in prevalence of larger invertebrates, which are more profitable prey items (McCracken & Tallowin 2004). This, combined with more frequent cutting caused by more rapid growth, may explain why breeding densities and success are lower on improved pasture than rough grassland (Galbraith 1988b). It should be noted, however, that unmanaged grasslands are likely to be too dense, thus making potentially valuable food resource inaccessible, so moderate management is preferred (McCracken & Tallowin 2004). Similarly, grazing keeps swards to the optimum height for lapwings, but increases the risk of trampling, so moderate stocking densities are optimal (Mason & Macdonald 1999; Vickery *et al.* 2001).

Improved drainage is a particularly important factor in the decline of wading birds, as earthworms may retreat earlier than would naturally occur, causing problems for birds that may synchronise breeding for optimum food availability, as well as making soils impenetrable (Beintema *et al.* 1991; Eglington *et al.* 2008). One of the most significant changes in land use has been the loss of mixed farming, to be replaced by areas of homogenous, fast-growing crops. This therefore reduces the availability of feeding habitats adjacent to breeding habitats and does not satisfy the lapwing preference for heterogeneity at the field and landscape scale (Siriwardena *et al.* 2000; Benton, Vickery, & Wilson 2003).

It has been suggested that although the initial decline in lapwing populations is attributable to agricultural intensification, it is predation that has prevented recovery of the population, by reducing reproductive output (Milsom 2005; MacDonald & Bolton 2008a; b). Predation rates have increased across Europe over the last 40 years, whilst periods of predator exclusion have proved beneficial to populations

(Roodbergen, Werf, & Hötker 2012). Misidentification of the causes of nest loss and overrepresentation of Western European populations, where nest protection is more common, may, however, have introduced minor bias to the dataset. When investigating the cause of this increase in predation, it is clear that it is intrinsically linked with changing agricultural practices and the associated altered landscape. Agricultural intensification leads to higher predation rates via a direct impact on predator densities such as red fox (*Vulpes vulpes*), stoats (*Mustela erminea*), buzzards (*Buteo buteo*) and corvids (Evans 2004; Wallander, Isaksson, & Lenberg 2006), and an indirect effect of increased vulnerability of prey in degraded or less suitable nesting habitat (Roodbergen, Werf, & Hötker 2012).

Change in land use can particularly increase the vulnerability of nests of ground-breeding birds, as the accompanying increased sward homogeneity and removal of cover reduces nest crypsis (Whittingham & Evans 2004), while the reduction of invertebrate food availability in homogenous landscapes (Wilson, Evans, & Grice 2009), may lead to adults foraging further afield with greater risk of detection (Evans 2004). Fragmentation of suitable habitat and the associated edge effects will aid accessibility of nests (Seymour *et al.* 2003; MacDonald & Bolton 2008b), while the concentration of birds into the few pockets of high-quality habitat remaining may attract high densities of opportunistic predators (Donald *et al.* 2002; Eglington *et al.* 2009). Equally, fragmenting habitat may disrupt nest aggregations, thus losing the significant advantages of semi-colonial nest defence (Berg, Lindberg, & Kallebrink 1992). The increasing prevalence of anthropogenic food sources, such as landfill and road-side carrion may also inflate predator densities to above those normally sustained naturally, subjecting prey to higher predation rates than usual (Courchamp, Langlais, & Sugihara 2000). It is important when analysing the impact of changing land use on productivity of farmland birds to therefore include an assessment of how this change may influence predation rates, given that these may not only be altered in the new habitat, but also in the remaining fragments of unchanged habitat.

1.5 Dietary analysis

A key factor contributing to the quality of nesting habitat is the foraging opportunities it provides. Identification and analysis of items in the diet is therefore an important

part of studies investigating habitat quality. In addition to examining the hatching success afforded by biomass crops for birds nesting within them (Chapters 2 and 3), an investigation of the foraging potential of biomass crops for passerines nesting in their vicinity was also conducted (Chapter 4). Various methods have been used to identify dietary components of birds, which vary in their degree of invasiveness. At the most extreme, stomach samples can be obtained from dead birds (e.g. Gaye-Siessegger 2014), but this is inappropriate for most studies, particularly those involving endangered species. Emetics have been used to induce vomiting, thus providing partially digested samples. This method has proved useful in identifying dietary components (Robert, Cloutier, & Laporte 1997; Valera & Barrios 1997), but its effectiveness varies depending on the type of emetic used and the species involved (Lederer & Crane 1978). The quality of data produced from this method can be limited by the condition of the regurgitated items, and the amount of food regurgitated which may in turn be affected by the time of day and its effects on gizzard contents (Lederer & Crane 1978; Robert, Cloutier, & Laporte 1997). Additionally, the use of such emetics can be harmful to birds, and may increase mortality both directly and indirectly (Lederer & Crane 1978; Johnson *et al.* 2002). Another invasive technique used is that of neck collars. These are ligatures placed around the neck of chicks, preventing food being swallowed so it can then be extracted and analysed. This presents risks of handling the birds to fit the collars, then of ensuring the collars can be removed before fledging (Poulsen & Aebischer 1995). Changes in provisioning behaviour may also result (Johnson, Best, & Heagy 1980 but Poulsen & Aebischer 1995). The collars should only be used for short periods to minimise disturbance, and as such only provide information on one or two meals (Moreby & Stoate 2000). The collar must also be fitted accurately to ensure it does not result in strangulation but is tight enough to collect all food items; sometimes small insects may still pass through (Poulsen & Aebischer 1995). The advantage of this technique, however, is the accuracy of identification and quantification of food provisioning it allows, as the samples obtained are yet to be digested and are usually whole, save for the removal by the parent of wings, legs and elytra in some cases (Poulsen & Aebischer 1995; Moreby & Stoate 2000).

Due to the potential harm caused by the methods described above, non-invasive techniques are often preferred in studies of dietary composition. Monitoring

of food provisioning to nests, whether directly or using cameras can provide useful information for cavity nesters or nest-box populations (e.g. Cowie & Hinsley 1988; Kleintjes & Dahlsten 1992). In such cases, dietary items can be identified down to order at least and quantities of each can be determined (Cowie & Hinsley 1988), although this will depend on the location of the nest (Poulsen & Aebischer 1995). There is usually a trade-off between the financial and time costs involved in video analysis, limiting sample sizes thus making this option impractical for many studies. Monitoring can, however, prove useful when used in tandem with other methods (Poulsen, Sotherton, & Aebischer 1998).

Faecal sampling is perhaps the most widely used method of dietary analysis (e.g. Green 1984; Ralph, Nagata, & Ralph 1985; Stoate, Moreby, & Szczur 1998; Poulsen, Sotherton, & Aebischer 1998; Moreby & Stoate 2001). Faeces can easily be collected in large quantities at no harm to the birds, and contain undigested prey fragments from a collection of foraging trips that can then be identified under the microscope (Moreby & Stoate 2000). There are, however, problems with this method arising from differences in digestibility. Small and soft-bodied items, such as Hymenoptera, Hemiptera, Collembola and Diptera leave very few, if any, undigested remains and are therefore likely to be under-reported in studies of dietary analysis using this method (Moreby & Stoate 2000). Conversely, the presence of hard, undigested items such as mandibles (Larvae), fangs and chelicera (Araneae), or characteristic leg parts (Coleoptera and Hemiptera) is likely to lead to over-estimation of these prey (Poulsen, Sotherton, & Aebischer 1998). Counting the number of these characteristic fragments in a sample gives an indication of the minimum number of a particular order eaten, although in some cases only presence or absence of the order can be confirmed (Moreby & Stoate 2000). Identification beyond order is rarely achieved, although sometimes family and genus can be determined (Moreby & Stoate 2001). Aside from morphological identification, faecal samples can also be used to determine dietary composition by applying molecular techniques. Identification of invertebrate prey using DNA extracted from bird faeces is a developing field (Deagle *et al.* 2010; Pompanon *et al.* 2012; Joo & Park 2012; Jarman *et al.* 2013). Again there are problems caused by differential digestion, but accurate information regarding the species and diversity of prey items can be obtained, and should be treated as semi-quantitative (Pompanon *et al.* 2012).

Enhanced knowledge of DNA extraction techniques and the factors that affect degradation of samples (e.g. environmental factors, laboratory treatments and storage conditions), as well as the persistence of DNA in the gut, are likely to lead to improved accuracy of this method (Oehm *et al.* 2011).

In this study, dietary composition of whitethroat and yellowhammer chicks was analysed by means of morphological identification of faecal remains (see Chapter 4). This was deemed to be the most practical, cost effective method for the purpose of this study, as despite issues of bias caused by differential digestion, it has yielded similar results to other methods in comparison studies. Faecal components are therefore thought to be representative of chick diet, if not completely comprehensive (Poulsen & Aebischer 1995; Sharples 1997).

1.6 Aims

To ensure expansion of the biomass crop industry is carried out in a sustainable way, the impacts of the associated change in land use should be investigated. This PhD aimed to do this by assessing the quality of habitat available for farmland birds nesting both in and around biomass crops. This involved:

- Examining the impacts of biomass crop production on ground-nesting birds, namely the lapwing (Chapters 2 and 3).
- Investigating the foraging potential of biomass crops for songbirds nesting in the vicinity (Chapter 4).
- Using this information to assess the overall risk to farmland birds of expansion of the biomass industry, and to discuss the implications for future policy (Chapter 5).

2 The breeding ecology of lapwings (*Vanellus vanellus*) in biomass crops

2.1 Abstract

Reaching the EU target for renewable energy is likely to encourage a rapid expansion of biomass crop production in the UK. These crops differ significantly from arable crops in terms of their management and structure, and thus will bring about changes to the farming landscape. Given their vulnerability to changing land use, this expansion could pose a considerable threat to ground-nesting birds. The suitability of Miscanthus energy crops was therefore examined as a nesting habitat for lapwings, a priority conservation species in the UK. Although there was no evidence of lapwings avoiding nesting in Miscanthus fields, clutches laid in this crop experienced lower hatch success than those laid in arable crops. This result was mainly influenced by increased losses in 2012, a particularly wet breeding season. This suggests in years of already unfavourable breeding conditions, nests in Miscanthus may be more vulnerable than those in traditional crops. Higher predation rates facilitated by the change in crop structure are discussed as a likely cause of reduced hatch success in Miscanthus. Long-term monitoring is required to account for inter-year variation and to determine the risk posed by energy crop expansion to ground-nesters. This is the first evidence of reduced breeding success in these crops.

2.2 Introduction

Agricultural intensification has contributed to a decline in farmland birds, due to deterioration in habitat and foraging opportunities. Ground-nesting birds, such as waders, are particularly vulnerable to changes in land use. Increased sward uniformity, loss of mixed farming and improved drainage, all typical traits of agricultural intensification, have led to a reduction in suitable nesting and foraging habitat (Chapter 1; Wilson *et al.* 1997; Chamberlain *et al.* 2000; Newton 2004; Wilson, Whittingham, & Bradbury 2005). Increased livestock densities with associated risks of nest trampling (Fuller *et al.* 1995; Henderson *et al.* 2002), increased vulnerability to predators (Chapter 1; Roodbergen, Werf, & Hötker 2012),

and increased risk of losses to agricultural operations have all combined to reduce the breeding success of ground-nesting birds. Indeed, evidence from French populations of farmland birds suggests ground-nesters may be more affected by agricultural production intensity than others (Bas, Renard & Jiguet 2009 but see Siriwardena *et al.* 1998).

The lapwing (*Vanellus vanellus*) is one such ground nesting species that has experienced a decline in numbers alongside agricultural intensification, with a continuous decline since the 1950s (Wilson, Vickery, & Browne 2001) and an accelerated decline from the mid-1980s. Between 1987 and 1998 there was a 49% decline in England and Wales (Wilson, Vickery, & Browne 2001) and according to the most recent BTO survey, the UK breeding population has declined by a further 41% in the period 1995-2011 (Risely *et al.* 2013). The pattern of decline has, however, varied with region, with the greatest decline in Southwest England and Wales (Figure 2.1), due to the polarisation of farming into pastoral lands in the west and arable in the east (Wilson, Vickery, & Browne 2001). Significant range contractions have occurred, with the result that 68% of the UK population was found in the northwest of England in 1998, compared with 40% in 1987 (Wilson, Vickery, & Browne 2001). The wintering population has also declined sharply in recent years (Holt *et al.* 2012), and the decreases in all regions of Europe make it one of the most strongly declining species in Europe (Baillie *et al.* 2014). In the UK, the latest population estimate is 140 000 pairs (Musgrove *et al.* 2013).

It is a reduction in breeding success that seems to have caused this decline in the lapwing population (Galbraith 1988b; Peach, Thompson, & Coulson 1994; Besbeas *et al.* 2002), while declines of other farmland birds are ascribed to reduced adult survival (Gregory *et al.* 2005). A recent review of the population trends of five meadow breeding wader species in Europe, including lapwing, found no overall decline in adult survival (Roodbergen, Werf, & Hötker 2012), in fact Peach *et al*. (1994) suggest adult lapwing survival has increased since the 1960s. Instead, a strong reduction in chick survival over the last 40 years has been observed (Roodbergen, Werf, & Hötker 2012). In 1994, the number of fledglings required of each breeding pair per year to replace annual adult losses was estimated at 0.83-097

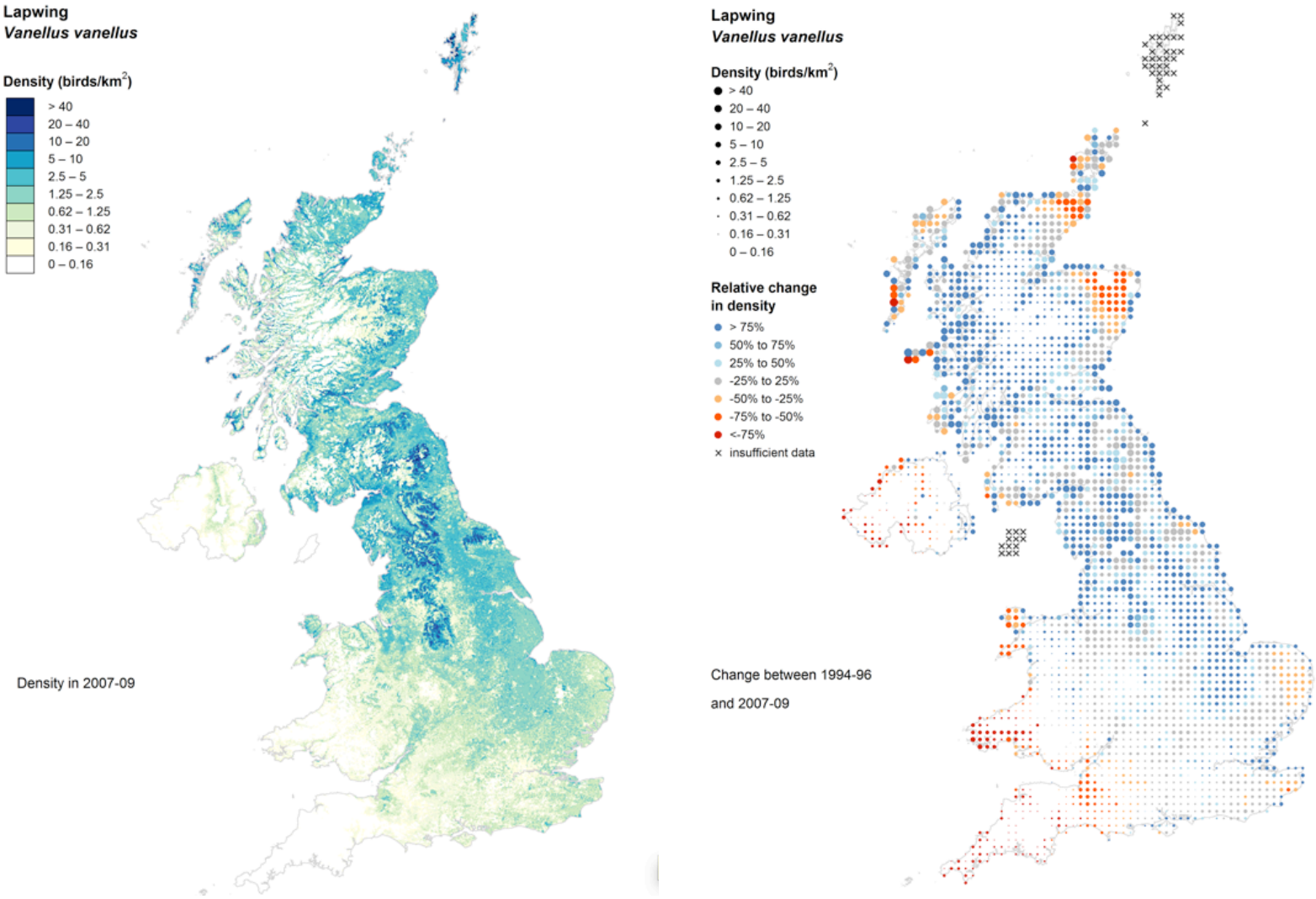


Figure 2.1 Lapwing density in the UK in 2007-09 (left) and change in lapwing densities between 1994-6 and 2007-9 (right). BTO British Bird Survey (BBS) data http://www.bto.org/volunteer-surveys/bbs/latest-results/maps-population-density-and-trends, downloaded 6/1/14

(Peach, Thompson, & Coulson 1994). This figure was met by only eight out of twenty-four European populations, according to a review by the authors. A more recent estimate suggests an average of 0.6-0.8 fledglings is required per pair per year (MacDonald & Bolton 2008b), but a mean of only 0.4 fledglings per pair was estimated in a review of 162 studies undertaken between 1996 and 2006 (Roodbergen, Werf, & Hötker 2012). Even in fallow plots provided in agri-environment schemes to enhance the breeding success of ground-nesting birds, lapwings only achieved 0.38-0.48 fledglings per pair in the South East in 2010-2011 (GWCT 2013).

Much research has been carried out examining the breeding ecology of lapwings across different landscapes to elucidate these mechanisms of decline. As such, characteristics of their optimum breeding habitat have been identified as open, short, heterogeneous sward with access to wet features (Galbraith 1988b; Wilson, Whittingham, & Bradbury 2005; Eglington *et al.* 2009; Smart *et al.* 2013). Preference for spring tillage has often been shown (Galbraith 1988b; Wilson, Vickery, & Browne 2001), followed by grassland, while autumn tillage and ley grass are avoided (Galbraith 1988b; Berg 1993; Wilson, Vickery, & Browne 2001). Proximity to grassland is also an important factor in nest-site selection because of the foraging opportunities it presents; spring tillage adjacent to grassland is more likely to hold breeding lapwings than spring tillage with no grassland nearby (Wilson, Vickery, & Browne 2001). Given the direct association of ground-nesting birds with the field habitat and therefore their vulnerability to changes in land use, it is vital that potential risks of further change are assessed, especially given the government's commitment to halt biodiversity loss on farmland (European Commission 2011).

A key EU target that 20% of overall energy consumption should be supplied from renewable energy sources (Defra 2007a) may conflict with this commitment. A promising and popular option for domestic supply of this renewable energy is from dedicated crops, thus a rapid increase in energy crop production is expected. In the UK, potentially 1.1 million ha (6% of agricultural land) was projected to be under energy crop production by 2020 (Defra 2007a). Although it is unlikely that this figure will be achieved (increasing the amount of planting by 20% each year would result in

only 0.04 million ha of land dedicated to energy crops; NNFCC 2012), there is the potential for this to contribute a significant change in land use in the future.

Miscanthus (*Miscanthus x giganteus*) and short rotation coppice (SRC) willow (*Salix spp.*) are the two main crops grown for energy production in the UK, both producing a tall dense canopy within months of harvest (see Appendix I for examples of biomass crop structure). Once established, yields are largely unaffected by the presence of pests or weeds, thus the need for chemical applications is reduced (Sage 1998; Defra 2007b). In addition, both crops' efficient use and re-distribution of nutrients throughout the growing cycle greatly minimise annual fertiliser requirements (Beale & Long 1997; Karp *et al.* 2009). Although this lack of chemical inputs and disturbance may be beneficial to birds associating with the crop (Haughton *et al.* 2009), there is concern that lapwings may be at a disadvantage, with their preference for open habitats, and short vegetation (Gove & Bradbury 2010). It is likely that Miscanthus and SRC will become unsuitable as a nesting habitat for lapwings fairly quickly and this might lead to curtailment of the breeding season, or even total avoidance of these crops (Anderson, Haskins, & Nelson 2004). Additionally, the density of biomass swards may impede access to foraging sites, thus affecting nest-site selection and chick survival (Galbraith 1988b).

The impact of a reduced nest success rate or shorter breeding season will only be of concern if overall productivity is affected; a high success rate at the start of the breeding season may mitigate for any reduction in the number of nesting attempts (e.g. skylarks; Donald 2004). The patchy nature of some fields may help to lower predation rates, as nesting could continue in open areas with nearby longer swards providing chicks with cover in which to hide (Butler & Gillings 2004; Devereux *et al.* 2004), whilst also increasing nest crypsis (Evans 2004; MacDonald & Bolton 2008b). It is therefore possible that the conversion of land to biomass crop production may not be detrimental to farmland birds but benign or even beneficial.

Evidence of the impacts of biomass cultivation on lapwing populations has so far come from studies examining overall bird densities and species richness, which have shown mixed effects. Lapwings have been found in both SRC and Miscanthus (Sage 1998; Sage, Cunningham, & Boatman 2006; Semere & Slater 2007b; Sage *et al.* 2010; Bright *et al.* 2013), and higher lapwing densities have been observed in

Miscanthus than winter wheat, although it is worth noting that the Miscanthus fields used were fairly early in establishment, with relatively high weed cover (Bright *et al.* 2013). In contrast, a previous study found no lapwings in Miscanthus (Sage *et al.* 2010), although this is likely to be due to regional differences in abundances. No lapwings were found in South West England in either study, but the later study also surveyed Lincolnshire, where the differences in lapwing abundance between crop type could be seen (Sage *et al.* 2010; Bright *et al.* 2013).

Whilst lapwing nesting attempts have been casually observed in both crops (Semere & Slater 2004, 2007b; Bright *et al.* 2013), investigations of breeding ecology and success of any species in either crop have not been carried out in the UK, save for one small scale study into the foraging potential of SRC, which monitored use of the crop by birds breeding within it (Sharples 1997). Dhondt *et al.* (2007) found 21 bird species nested in SRC plantations of various ages in New York, but the breeding success of ground-nesting birds, which are likely to be particularly vulnerable to energy crop expansion, has not yet been investigated. There are concerns that any benefits of biomass crop production so far observed will diminish with the loss of open patches and valuable weed cover typical of increasing age and commerciality of the crop (Gove & Bradbury 2010; Dauber, Jones, & Stout 2010). Loss of weeds from Miscanthus fields may be particularly detrimental as it is these, rather than the crop itself that support invertebrate prey (Semere & Slater 2007a). Due to the relative novelty of the market, studies thus far have investigated fairly small scale plantations, all early in establishment, so it is important that effects continue to be monitored as the crops age (Gove & Bradbury 2010; Dauber, Jones, & Stout 2010).

Attempts to mitigate for the effects of changing land use on ground-nesting birds have been made through the implementation of various options within agri-environment schemes, with varying degrees of success. An increase in breeding densities and success has been observed on fallow plots and specifically created areas of wet habitat (Sheldon, Chaney, & Tyler 2007; Wilson, Evans, & Grice 2009), indicating the importance of habitat features to these birds. Benefits are dependent on the species of concern (Smart *et al.* 2013) and level of uptake by farmers (Wilson, Vickery, & Pendlebury 2007). The variation in efficacy of these schemes emphasises the importance of thorough investigation of the impacts of land use change prior to its

occurrence, to enable implementation of appropriate protective measures rather than attempting to mitigate after the event, which may be considerably less effective.

The distinctive call and aerial displays of the male lapwing make it an easily recognisable species, synonymous with the arable landscape and popular with farmers and the general public (Cocker & Mabey 2005; Shrubb 2007). The size and anti-predator mobbing behaviour of lapwings make their nests easier to find than those of other ground-nesting birds, such as skylarks or meadow pipits (pers. obs.). The presence of breeding lapwings in an area correlates with the presence of other species; their anti-predator mobbing behaviour deters predators and decreases the amount of time other birds have to allocate to vigilance behaviour (Eriksson & Gotmark 1982; Berg *et al.* 2002). Additionally the presence of lapwings may act as an indicator of good quality habitat beneficial to other species (Eriksson & Gotmark 1982; Stillman *et al.* 2006; MacDonald & Bolton 2008a; Chamberlain *et al.* 2009). Efforts to conserve lapwings are therefore likely to benefit other species (Berg *et al.* 2002). Almost half the European population of lapwings is found in the UK, with only 1% of this occurring outside farmland (Wilson, Vickery, & Browne 2001). By assessing impacts of an increase in biomass crop cultivation on lapwings at this early stage, it is hoped that policy may be informed and appropriate measures taken to minimise conflict with biodiversity goals.

2.3 Aims

Many uncertainties exist regarding the viability of biomass crops as nesting habitat, but given the role productivity plays in lapwing declines, it is vital the impact of biomass crop production on breeding success is assessed. The following questions were therefore asked:

- Do lapwings nest in recently cut biomass crops?
- If so, how successful are these nesting attempts compared with those in arable crops?
- Does the timing of the breeding season vary between crop types?
- Are lapwing chicks able to forage in biomass crops, i.e. do they persist in the crop post-hatching?

2.4 Methods

2.4.1 Fieldwork and study sites

Fieldwork was conducted in Lincolnshire and Nottinghamshire, in a number of Miscanthus, SRC and arable fields (Figure 2.2) between March and August of 2011, 2012 and 2013. (For examples of biomass structure, see Appendix I). This area is "at the forefront of the UK's Woody Biomass market" (Wright & Leivers 2008), with approximately 5,500 hectares in the East Midlands in 2007, largely due to a number of coal-firing power stations co-firing biomass (e.g. at Cottam, near Retford; and Drax, Selby), as well as the widespread installation of biomass boilers both within the private and public sectors, under schemes such as Nottinghamshire County Council's "Woodheat Project", and "The Lincolnshire County Council LIGHT" project (Wright and Leivers 2008). Information provided by the biomass producers in the area; Renewable Energy Growers Ltd (ReGro), Coppice Resources Ltd (CRL) and International Energy Crops (IEC), and data from a previous study (Bright *et al.* 2013) were used to identify a number of sites of recently cut Miscanthus and SRC (i.e. since the preceding November). Fields were 2-10 ha in size, not adjacent to any built-up areas and biomass crops were typical of commercial crops (i.e. at least four years old, without large non-crop patches). In 2011 and 2012, the presence of lapwings in preliminary visits at the start of the season, or their presence in a previous study (Bright *et al.* 2013) was used to select fields, to maximise the number of nests found. In 2013, the study opened out to include more fields, irrespective of lapwing presence, to investigate field selection by the birds. Control fields were sugar beet (*Beta vulgaris*), or cereal (mainly wheat; *Triticum spp* and barley; *Hordeum vulgare*), and were located on the same farm as the biomass crop fields, wherever possible. These fields were not adjacent to the selected biomass crop fields to avoid any problems of assigning birds to fields.

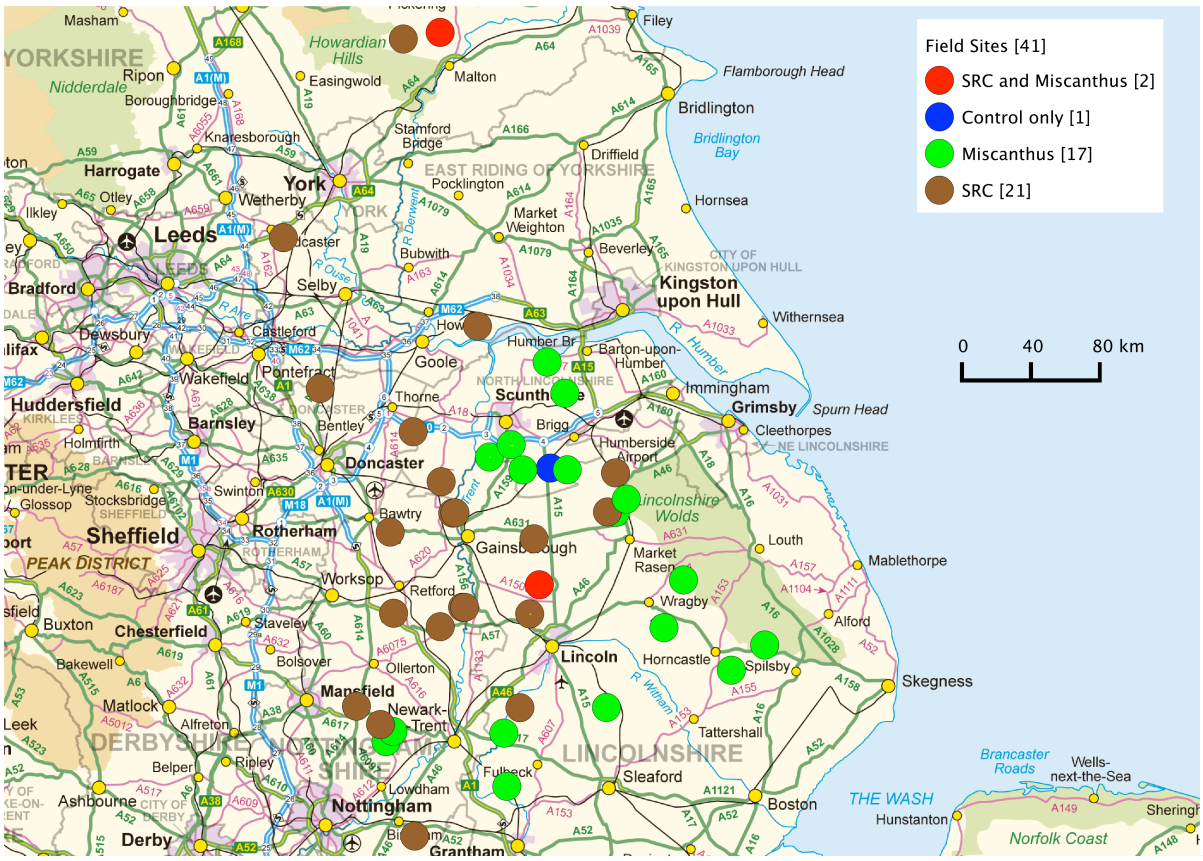


Figure 2.2 Location of sites used 2011-2013. Mostly the same Miscanthus sites were used each year, while SRC sites varied each year due to harvesting regimes. Control fields used were on the same site as biomass crop fields, with one exception, which only provided control fields.

Fields were first visited in mid-March, and the numbers of lapwings seen, evidence of breeding activity and the location of territorial males/sitting females were recorded. Fields that contained lapwings or in which breeding was suspected were then monitored regularly (at least once a week) to confirm breeding. If no lapwings were present, the field was revisited a fortnight later to confirm the absence of lapwings, when visits would then be reduced to solely the monthly vegetation surveys. Nest presence was recorded in all fields once a month in addition to the nest monitoring. Early in the survey period when crops remained short, or in patchy areas, female lapwings could be observed sitting on nests and an accurate estimate of nest location obtained. Where this was not possible other behaviours were used to identify nests and determine breeding stage, e.g. scrape building, mobbing and displaying by the male during nest building and incubation, alarm calling by both parents when chicks present (A. Hoodless pers. comm.; Ferguson-Lees, Castell, &

Leech 2011). Once an approximate assessment of nest locations had been obtained, one observer approached each location guided by a second observer who provided directions (by mobile telephone) from a field edge vantage point. If after 10 minutes the nest had not been found, the observer returned to the vantage point and after a further ten minutes, made a second attempt to find the nest. Once found, the nest location was logged in a hand-held GPS (Garmin Etrex), clutch size recorded, and the weight (±0.1 g), length (±0.1 mm) and breadth (±0.1 mm) of each egg measured using a Pesola balance and calipers. This enabled estimation of expected hatch dates using the equation developed by (Galbraith 1988b):

$$\text{Egg volume } (cm^3) = \text{length (cm)} \times \text{breadth}^2 \text{ (cm)} \times 0.457$$

$$\text{Density } (g\ cm^{-3}) = \text{mean egg weight (g)/mean egg volume}$$

$$\text{Days until hatching} = (150.84 \times \text{density}) - 140.68 \text{ with a mean error of 2.3 days}$$

Nests were subsequently checked every four to five days (more frequently closer to expected hatch date) to ascertain outcome. This usually involved monitoring from a distance with binoculars; only if there was no sign of an incubating adult was the nest visited and contents checked. Approaching nests has not been shown to affect clutch survival (Galbraith 1987; Fletcher, Warren, & Baines 2005), although frequent visits may increase predation risk (Hart *et al.* 2002). No visits took place in rain or low temperatures to ensure neither eggs nor chicks were unduly exposed (Shrubb 2007). The presence of small egg fragments (c. 1 mm in size) in the nest lining was assumed to indicate a successful hatch, whilst a completely empty nest or the presence of large pieces of egg shell in or near the nest were taken as indications of predation (Green, Hawell, & Johnson 1987; Galbraith 1988b; Hart *et al.* 2002). A nest was recorded as successful if at least one egg hatched. When nests were found hatched or no longer active, vegetation height next to the nest was recorded. As chicks leave the nest shortly after hatching, nests were only followed to this point. First egg dates were calculated as 32 days before hatch date (assuming a clutch size of four, two days between each egg, and a further 25 days to incubate; Berg, Lindberg, & Kallebrink 1992; Seymour *et al.* 2003; Sheldon, Chaney, & Tyler 2007). Hatch dates estimated from the egg volume calculation of Galbraith (1988b) were occasionally inaccurate (see Figure 2.9) so to minimise error,

measurements were not collected in windy conditions, which resulted in some eggs not being measured at all. Therefore the equation for estimating hatch date was only used as a guide for fieldworkers for when to expect hatching, and not for calculation of lay dates. All nest searching and subsequent monitoring followed the BTO Nest Recorder's code of conduct (BTO 2003).

Surveys of crop height, percentage crop cover and weed cover were carried out in all SRC and Miscanthus fields (regardless of whether or not lapwings were present) in April/May, May/June and June/July to assess the changing characteristics of the crop. Ten samples were taken either side of a diagonal transect down the centre of the field, two of which were taken in the edge of the crop. Crop canopy cover was estimated as a percentage of leaf or stem area in a 2x2m quadrat at each location, and the height of a stem in that quadrat was measured. Weed cover in 10x1m quadrats was estimated using the DAFOR scale (Dominant >50%, Abundant 25-50%, Frequent 10-25%, Occasional 1-10% and Rare <1%; Sage *et al.* 2010).

Finally, the length of time chicks remained in natal fields after hatching was monitored. The chicks were not ringed or radio-tagged, thus chicks could not be identified as belonging to particular nests, instead persistence was measured for each field as the number of days chicks were observed to be present, after the last nest in that field had hatched. Fields were visited weekly after hatching and the presence of chicks was recorded for each field. Chick presence was determined either by direct observation of the chicks, or using behavioural cues from the adults. In contrast to the response to disturbance during the incubation stage, when chicks are present adults react to disturbance by staying on the ground and calling to the chicks; a call very distinct from the usual mobbing call (A. Hoodless pers. comm.). Where possible, the age of chicks was determined, (Figure 2.3) to help in identifying the nest of origin. When disturbed, chicks react to the adult calls by sitting flat on the ground, relying on their camouflage for protection. As a result, finding and therefore ageing chicks by eye proved very difficult hence the reliance on adult behaviour to determine chick presence.

Alongside lapwing nest data, efforts were also made to find skylark nests. This proved unsuccessful, with only seven nests being found in total, but further details of the methods and results of these searches can be found in Appendix II.

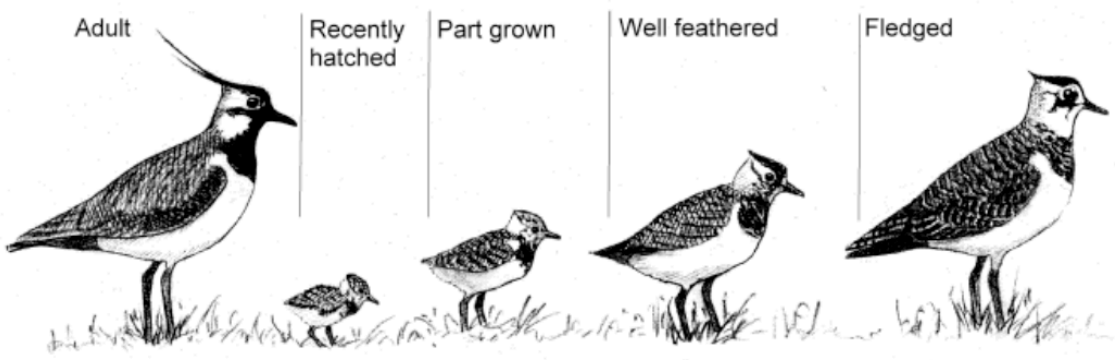


Figure 2.3 Guide to ageing lapwing chicks. Recently hatched = less than 1 week (presence of egg tooth refines this to c.24 hours), part grown = 2 weeks old, well feathered = 3 weeks old and fledged = 28-35 days old. Taken from Bradshaw (2011).

2.4.2 Data analysis

The choice of nesting habitat was investigated by modelling the number of fields containing nests as a proportion of the total fields surveyed for each crop type in each year, using binomial errors. The response variable was the maximum number of fields containing nests each year, at any point in the survey period. To investigate the effects of crop characteristics on site choice, nest presence in biomass crops was modelled as a binary outcome in a generalised linear model (GLM) with binomial errors. The effect of crop type on nest density was examined by using the maximum number of nests at any one time as the response variable, with log of the survey area as an offset to convert counts to densities, with Poisson errors. Explanatory variables were crop type and year. For a subset of nests in biomass crops, additional variables of crop height, crop cover and weed cover for each survey were modelled against peak density.

To further examine nest site choice in 2013, measurements of crop height next to the nest (recorded as soon as possible after hatch or last active date) were compared with the mean crop height for the rest of the field using a Wilcoxon rank statistic. The two measurements were taken within one week of each other and mean crop height was taken as the mean of five measurements taken at random points throughout the field (either those at the monthly vegetation survey, or on a separate occasion).

To measure hatch success, the logistic-exposure method developed by Shaffer (2004) was used (see Chapter 3 for a discussion of different nesting success methods using data from this study). The logistic-exposure method uses the nest monitoring interval as the sampling unit and is based on a GLM with binomial error distribution and a modified link function: $g(\theta) = \log_e (\theta^{1/t}/[1 - \theta^{1/t}])$, where θ is the survival estimate for the monitoring interval and t is the interval length in days.

The logistic-exposure method was applied to mixed-effects models to investigate hatching success in all three years, with field and site as random effects to avoid pseudoreplication arising from multiple nests in the same field and fields within the same site. Very little variation was attributable to site, so this was omitted from final models. Several models were run to investigate the effects of other explanatory variables besides crop type, including, field size (ha), distance to boundary (m), distance to the nearest nest (m) and nest density (no of nests per ha, log transformed). To avoid over-parameterisation of the models, variables were added to the model one by one, to reach a maximal model and the significance of each was tested by a process of stepwise deletion. Terms were deleted and subsequent models compared via ANOVAs; only those variables whose removal resulted in significantly different models ($p<0.05$) were retained. Terms were removed in this way until the minimum adequate model was reached (Crawley 2007). The models were also carried out for a subset of nests in biomass crops alone, but in addition to the variables mentioned previously, distance to the nearest sward (m) was also investigated. Finally, the effect of vegetation characteristics on hatch success was analysed for the same subset of nests in biomass crops. Each nest had three measures of crop height, percentage crop cover and weed cover (one for each monthly survey), and different combinations of these were modelled and tested by stepwise deletion. To cope with then non-normal distribution of errors, crop height was log transformed, and percentage crop cover arcsine transformed (Crawley 2007). The DAFOR weed cover scale was converted to a numeric scale where Dominant = 5, Rare = 1. The logistic-exposure method produced estimates of daily survival rates, which could then be converted to hatch success by raising them to the power of 32, the average length of egg-laying and incubation (Berg, Lindberg, & Kallebrink 1992).

Egg size can act as a determinant and reflection of fitness; larger eggs are more likely to hatch than smaller eggs, and to produce larger chicks that have a higher chance of survival to fledging than smaller chicks (Galbraith 1988a; Blomqvist, James, & Gotmark 1997). Egg size can reflect parental fitness and foraging availability at the time of egg formation (Blomqvist, James, & Gotmark 1997). Thus variation in egg size across crop types was examined to account for any variation in hatching success. Mean egg length and breadth was compared by means of Wilcoxon rank sum tests, but not volume as this decreases with proximity to hatch date and clutches were measured at different ages Galbraith (1988b). The inaccuracies in hatch predictions mentioned below likely arose from errors in egg weight measurements, thus affecting egg volume calculations.

Variation in breeding onset between crop types was tested by comparing first egg dates in different crop types by a Wilcoxon rank sum test. First egg dates were calculated as 32 days before hatch date. Hatch date was observed directly or estimated as the midpoint between the last visit when eggs were still present and the next visit when eggs had hatched. Hatch dates could also be predicted from the egg density calculation of Galbraith (1988b) to enable first egg dates to be calculated for unsuccessful nests. To assess the accuracy of using the density calculation to calculate first egg dates, this method was used to predict first egg dates of those clutches of 'known' first egg date, i.e. successful nests. Observed (calculated from the observed or midpoint hatch date) and predicted first egg dates differed by up to 15 days (Figure 2.10). Due to this discrepancy, first egg dates calculated using the density equation were considered unreliable, therefore only successful nests were used in analysis of nest timings.

The hypothesis that Miscanthus fields are likely to become denser and less weedy as they become more established as commercial crops was examined by comparing the vegetation characteristics surveyed in the current study with data obtained in a previous study (Bright *et al.* 2013), when the average age of fields was 2.3 ± 0.1 years. Only crop height, percentage cover and weed cover measured in June surveys from fields used in both studies were analysed. The effect of year was determined via mixed-effects models, with site as random effect. All statistical analyses were performed in R (R Development Core Team 2011).

2.5 Results

Table 2.1 Distribution of lapwing nests between control fields (n=32), and biomass crop fields (n=42) in 2011-2013. Biomass fields were harvested between the November and March preceding surveying.

	Crop	Nests (Success)	Fields (Occupied)	Field Size (ha)
2011	Miscanthus	4 (3)	12 (4)	6.65 ± 0.70
	SRC	4 (4)	8 (4)	6.82 ± 0.99
	Control	9 (5)	13 (4)	6.77 ± 0.77
	Total	17 (12)	33 (12)	6.74 ± 0.45
2012	Miscanthus	18 (7)	12 (5)	6.85 ± 0.80
	SRC	2 (2)	10 (3)	6.56 ± 1.05
	Control	22 (17)	10 (6)	7.47 ± 0.92
	Total	42 (26)	32 (14)	6.95 ± 0.52
2013	Miscanthus	21 (17)	16 (4)	7.13 ± 0.66
	SRC	3 (3)	6 (3)	7.17 ± 0.79
	Control	12 (10)	12 (7)	7.79 ± 0.48
	Total	36 (30)	34 (14)	7.37 ± 0.37

The increase in field size through the years was not significant ($F_{2,33}$=1.19, p=0.3), nor was there any difference in field size across crop types ($F_{2,33}$=1.11, p=0.4).

2.5.1 Nest distribution

A total of 95 nests were found across 40 fields in the three breeding seasons between 2011 and 2013. The distribution of these nests is summarised in Table 2.1; many fields were used in multiple years, so over the three seasons, nests were found in 33 different fields; 17 controls and 16 biomass crop fields. Different SRC fields were used every year due to the 3-year harvesting cycle; only recently cut fields were sampled. The wet weather at the end of 2012 and start of 2013 (Met Office 2013) led to delays in harvesting SRC and Miscanthus sites in the final year of fieldwork. Some SRC fields were left unharvested altogether and thus had to be excluded from the study in 2013.

The proportion of fields surveyed that contained nests did not vary with crop type (the greatest difference was between beet and Miscanthus fields (z=-1.82, p=0.07, Figure 2.4), nor was there any effect of year (p=0.72). The proportion of occupied fields decreased on subsequent visits in the same year (z=-4.06, p<0.0001).

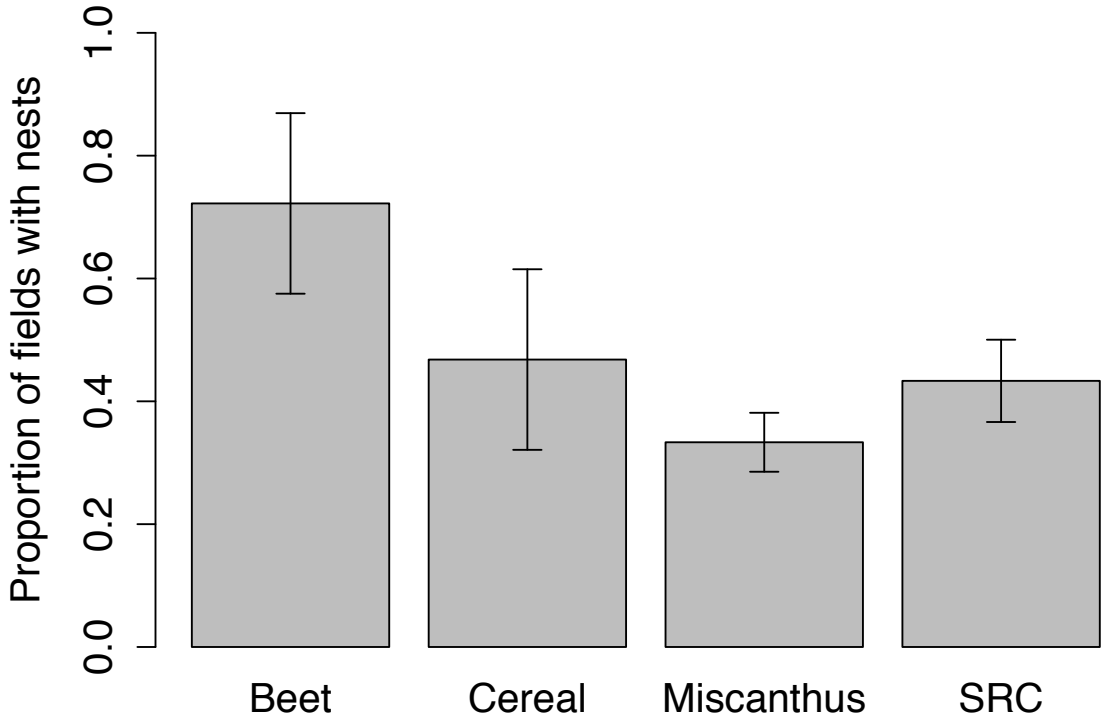


Figure 2.4 Proportion of fields containing nesting lapwing according to crop type, with standard errors. Sugar beet n=9; Cereal n=26; Miscanthus n=40; SRC n=24. Differences between crop types were not significant (p>0.05 in all cases).

In biomass crops, nests are more likely in crops of shorter sward height (z=-3.3, p=0.001, Figure 2.5) or lower percentage crop cover (z=-2.48, p=0.01). Crop cover and crop height are correlated (R^2=0.6) so were not tested for an additive effect in models to avoid multicollinearity. Neither the degree of weed cover (p=0.2) nor field size (p=0.88) had any effect on nest presence.

There was no effect of crop type (t_{99}=-0.10, p=0.92, t_{99}=-1.29, p=0.20 for Miscanthus and SRC respectively or year (p=0.44), on nest densities (Figure 2.6). For nests in biomass crops, neither weed cover nor crop cover affected density

(p=0.6 and 0.1 respectively), but nest density was lower in fields of taller crop (t_{55}=-2.1, p=0.03).

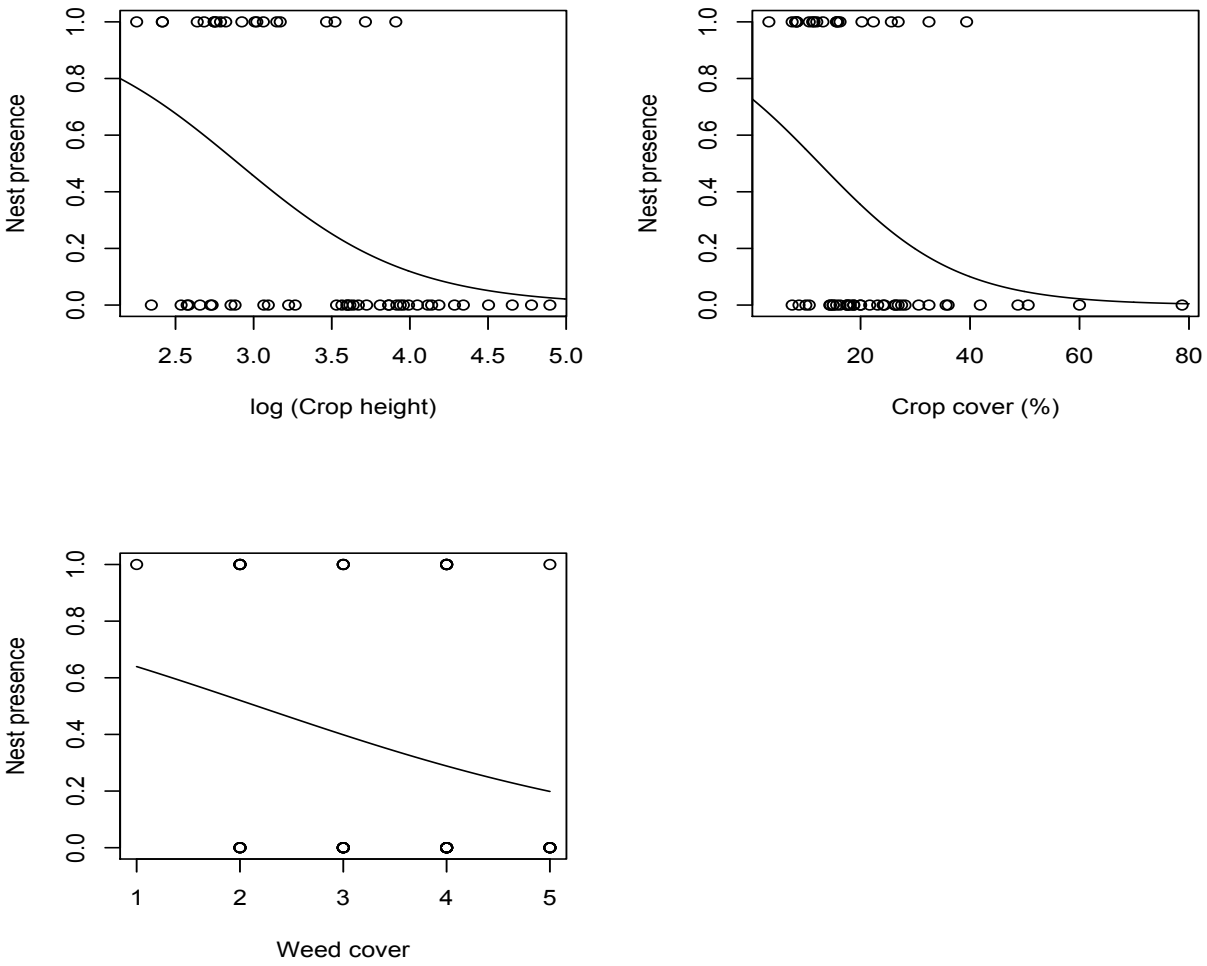


Figure 2.5 The effect of vegetation characteristics on the likelihood of biomass fields containing lapwing nests. Nest presence was the occurrence of nests at any point in the three-month survey period (April-June). Nest presence was negatively affected by both crop height (z=-3.3, p=0.001) and percentage crop cover (z=-2.48, p=0.01), but was not affected by weed cover (p=0.2).

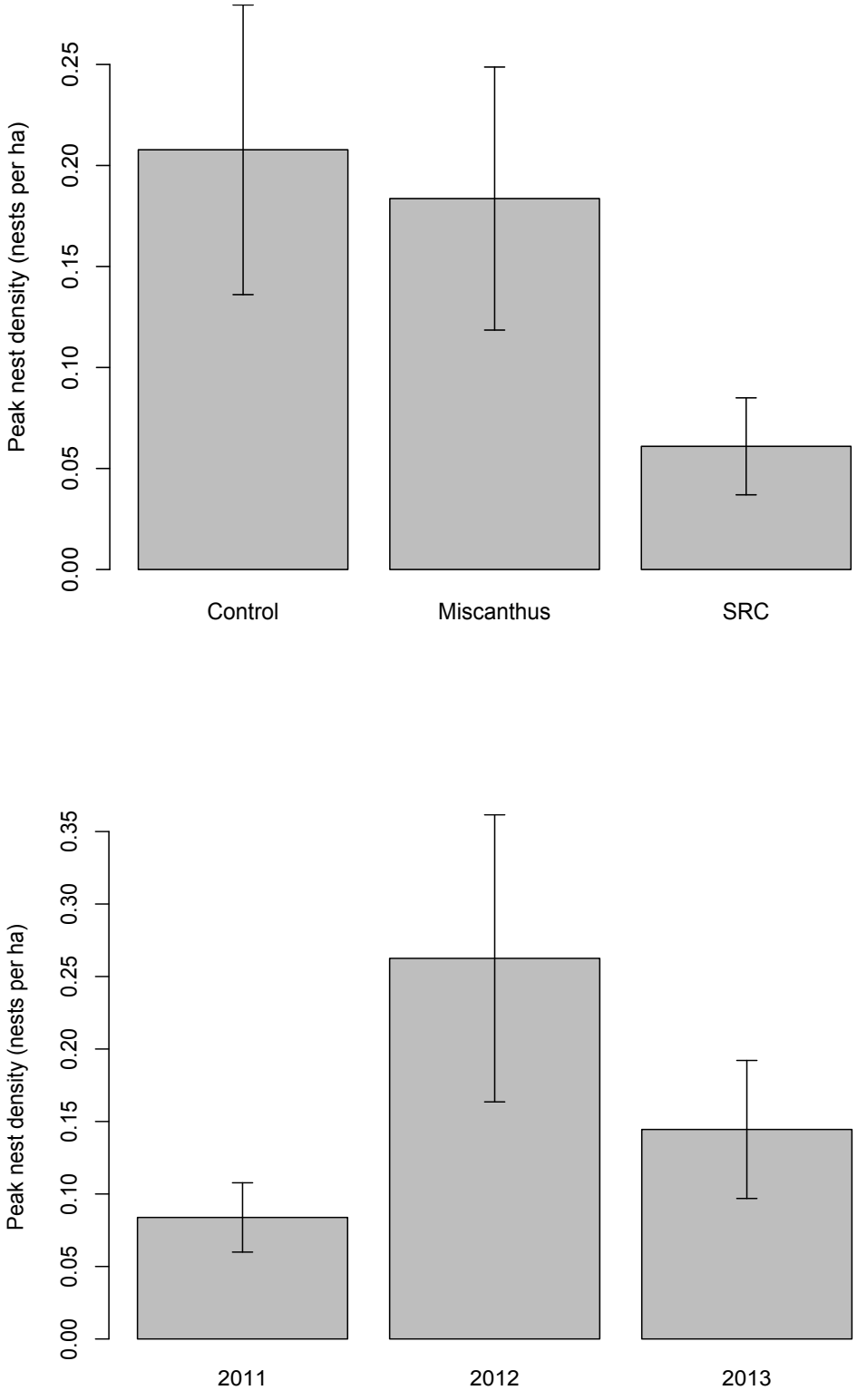


Figure 2.6 Lapwing nest density per crop type (top) and year (bottom), with standard errors. Density was measured as the maximum number of nests per hectare at any one time. Differences between crop type and years were not significant (p>0.1 in all cases).

2.5.2 Hatch success

When comparing across all years, all nine nests in the SRC crops were successful and therefore excluded from analysis. Nests in Miscanthus were less successful than those in control crops (z=-2.06, p=0.04), whilst higher success rates were achieved in 2013 than in other years (z=2.17, p=0.03; z=2.18, p=0.03 in comparisons with 2011 and 2012 respectively, Figure 2.7).

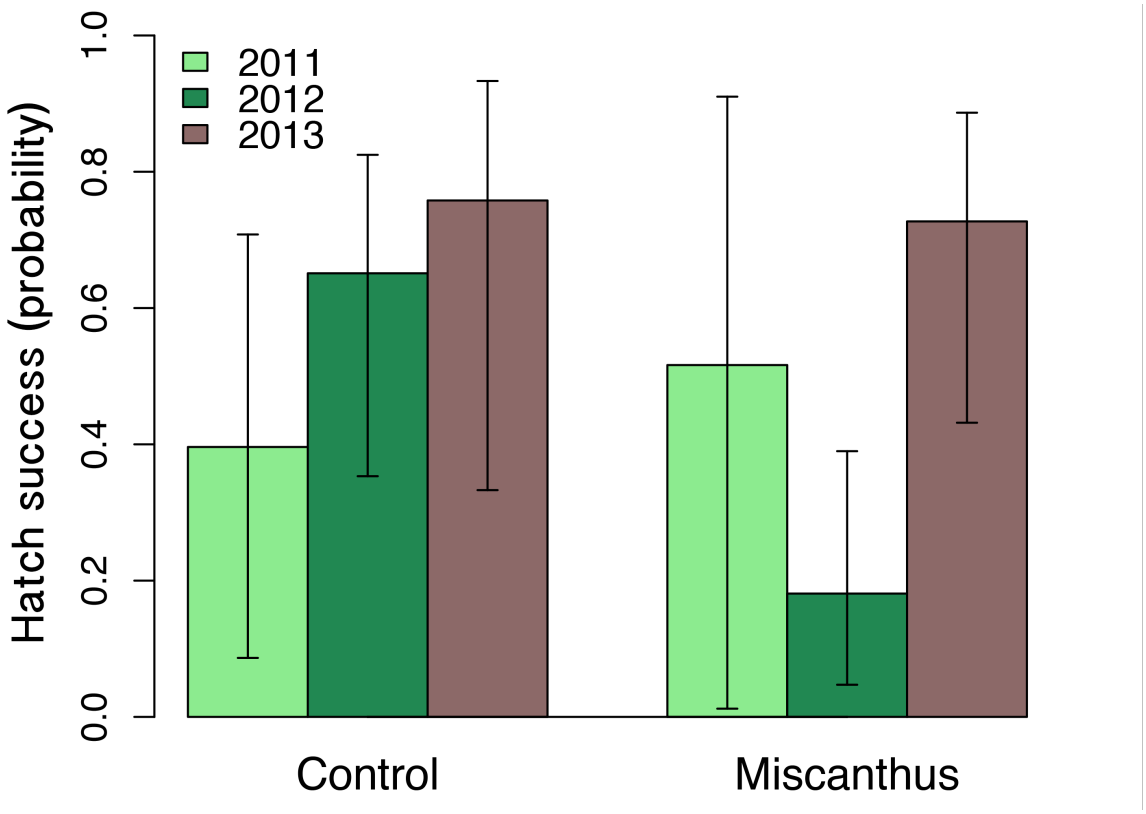


Figure 2.7 Hatch success of lapwing nests calculated using the logistic-exposure method of Shaffer in a generalised linear model structure, with standard errors. Control n=9, 22, 12, Miscanthus n=4, 18, 21 in 2011, 2012 and 2013 respectively. Difference between crop types: z=-2.06, p=0.04. Difference between 2013 and 2011 z=2.17, p=0.03, and between 2013 and 2012 z=2.18, p=0.03.

Hatching success increased with field size (z=2.9, p=0.004). Field size did not differ by year or crop type ($F_{2,33}$=1.19, p=0.3 and $F_{2,33}$=1.11, p=0.4 respectively). There was no effect of distance to boundary (x^2=0.82, p=0.2) or nest density on hatch success (x^2=0.02, p=0.9). As distance to the nearest nest was negatively correlated with nest density ($F_{(1,93)}$ =138.8, p=0.0008, R^2=0.7), only one of these was included in the model at any one time.

For nests in biomass crops, hatching success increased with greater proximity to the nearest sward (z=2.4, p=0.01; Table 2.2). Overall, 24 out of 44 nests were located in Miscanthus swards or willow stands; 45% in 2012 and 63% in 2013. There were no effects of vegetation characteristics on hatching success; for each nest, three measures of crop height, percentage crop cover and weed cover in each of three breeding seasons; all gave p values >0.05. There was no difference in mean crop height adjacent to nests compared with the mean crop height of the whole field (W=623, p=0.3), measured within one week of each other. The difference between crop height at the nest and the mean crop height of the whole field ranged from 20.4cm taller at the nest to 21cm shorter at the nest and in 26 out of 38 cases, the crop nearest the nest was shorter than the mean crop height of that field (Table 2.3).

Table 2.2 Nest characteristics of nests in biomass crops.

	No. of nests	Mean distance to sward (cm)	Mean nest density	Mean distance to nearest nest (m)
Success	29	18.1 ± 6.38	0.84 ± 0.11	84.1 ± 21.68
Failure	15	53.3 ± 15.98	1.27 ± 0.20	138.8 ± 48.50
Total	44	30.0 ± 7.23	0.99 ± 0.10	102.7 ±21.87

Table 2.3 Height of biomass crop at nests and mean crop height of the field in which they were found, in 2012 and 2013.

	Nests	Mean crop height nearest nest (cm)	Mean crop height per field (cm)
Success	27	25.16 ± 3.12	28.77 ± 3.31
Failure	11	28.54 ± 5.96	28.27 ± 4.86
All	38	26.14 ± 2.77	28.63 ± 2.71

There were no differences in egg size (length and breadth) between the nests of control or Miscanthus fields or between 2012 or 2013 (p>0.5 in all cases; nests in SRC and 2011 were excluded due to small sample size), although there is a greater variance in egg length in Miscanthus fields than in control fields (D=0.4, p=0.01). Eggs from Miscanthus tended to be longer, this difference was marginally significant

(t=1.98, p=0.05). There were no differences in egg size between successful and unsuccessful clutches (p>0.4 for both length and breadth).

There were no differences in lay dates between control and biomass fields (W=325.5, p=0.2, Figure 2.8). Eggs were laid significantly earlier in 2012 (3rd April ± 3 days) than both 2011 (27th April ± 4 days, W=263.5, p<0.0001) and 2013 (21st April ± 3 days, W=141.5, p<0.0001, Figure 2.9). It should be noted that fieldwork did not commence in 2011 until mid-April, so some early nests may have been missed.

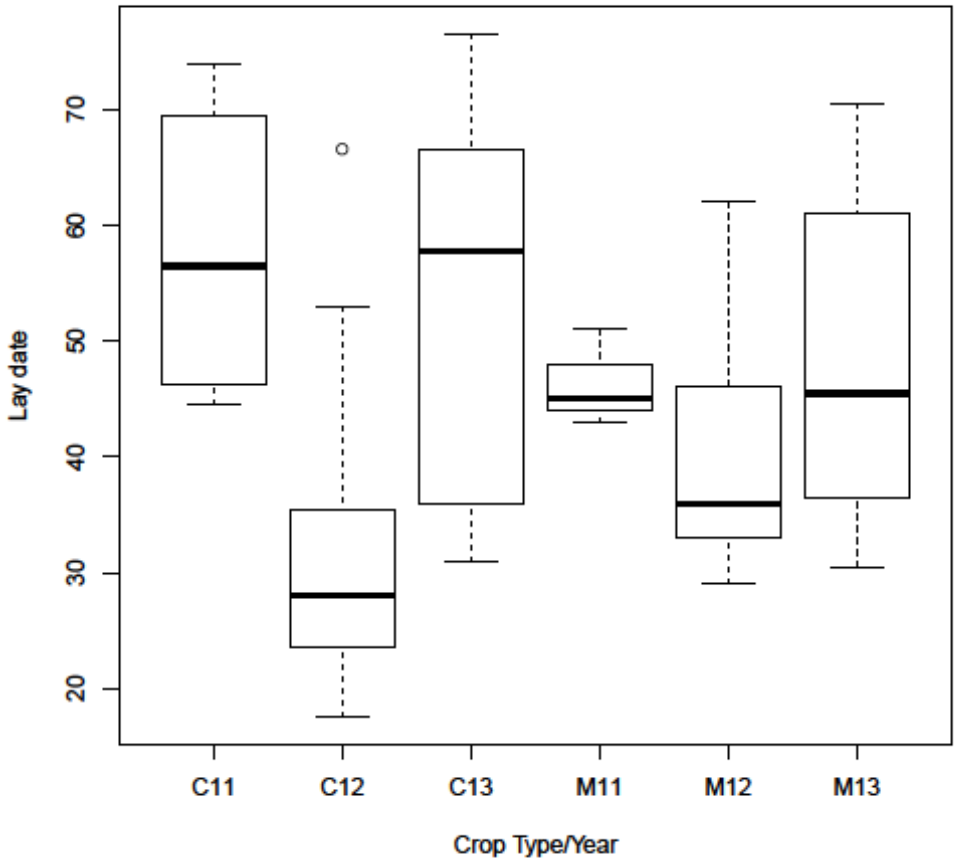


Figure 2.8 Lay dates per crop type and year. C= Control, M=Miscanthus. 11, 12 and 13 = 2011, 2012 and 2013 respectively. Day 0=1st March. First egg date was estimated only for successful nests as 32 days before the observed hatch date. n=5, 17, 10 for C11, C12, C13; n=3, 7, 17 for M11, M12, M13. The difference between crop types was not significant (W=325.5, p=0.2), but the difference between 2012 and other years was significant (2012 vs 2011 W=263.5, p<0.0001, 2012 vs 2013 W=141.5, p<0.0001).

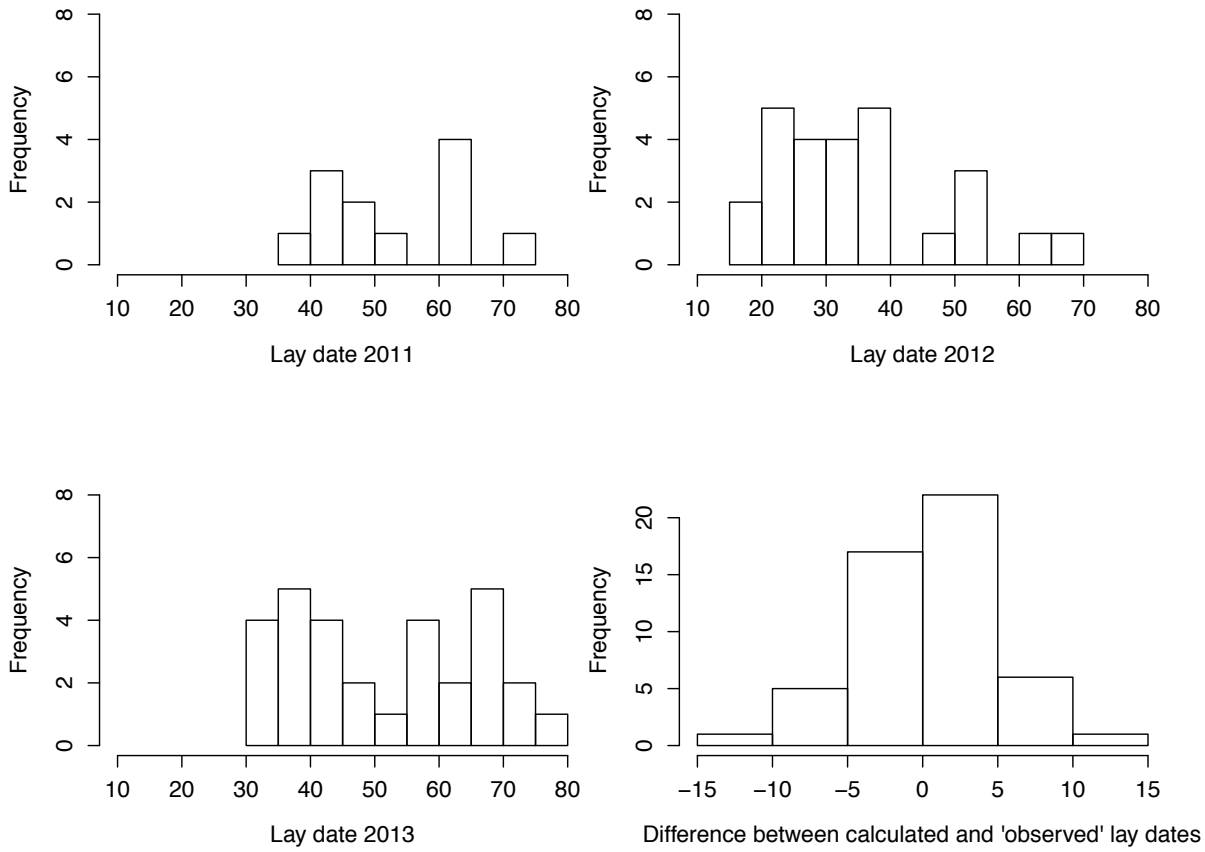


Figure 2.9 First egg dates (lay date) for each year 2011-2013. Day 0=1st March. First egg date was estimated only for successful nests as 32 days before the observed hatch date. Differences between these "observed" lay dates and those calculated using Galbraith's (1988b) equation for loss of density are shown. 2011 n= 12, 2012 n= 26, 2013 n= 30. The difference in lay dates between 2012 and other years was significant (2012 vs 2011 W=263.5, p<0.0001, 2012 vs 2013 W=141.5, p<0.0001).

2.5.3 Chick persistence

In 2013, fields continued to be monitored after nests had hatched to determine the length of time chicks remained in the natal field. Chicks were observed in 12 fields, although in two Miscanthus fields, the final nests hatched at least 18 days after chicks were last seen in the fields, so these nests were treated as separate units. Any chicks observed after these final nests hatched could be allocated to these nests with near certainty, making n=14.

The mean length of time chicks remained in control fields was 15.42 ± 5.2 days, in Miscanthus 12.12 ± 4.5 days and in SRC 8.5 ± 4 days. Given the small sample sizes, these data were not analysed statistically, but are presented here to indicate possible differences in suitability as foraging habitat between crop types (Figure 2.10).

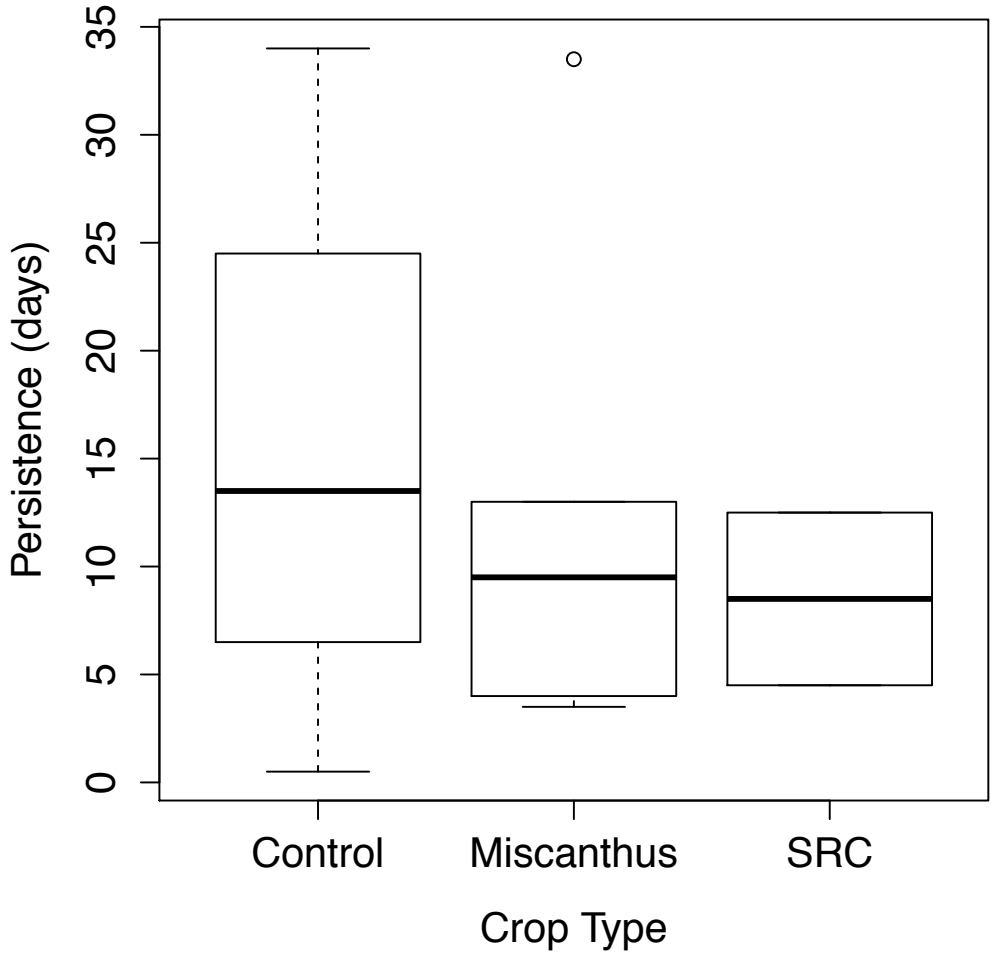


Figure 2.10 Number of days chicks were observed to persist in natal fields post-hatching. Control n=6, Miscanthus n=6, SRC n=2.

2.5.4 Comparison of crop characteristics between years

The Miscanthus crop development showed a trend of greater sward height in earlier years of monitoring with an accompanying lower crop cover and greater weed cover (Figure 2.11). Miscanthus crop was significantly taller in 2009 and 2011 than 2012 and 2013 (2009 compared with 2012: t=2.37, p=0.02 and 2013: t=2.64, p=0.01; 2011 compared with 2012: t=2.30, p=0.03 and 2013: t=2.48, p=0.02). Percentage crop cover was lower in 2009 than in all other years (t=-4.72, p=0.0007; t=-4.19,

p=0.0008; t=-2.00, p=0.05 compared with 2011, 12 and 13 respectively). Lastly, weed cover was higher in 2009 than in all other years (t=2.71, p=0.01; t=2.81, p=0.009 and t=3.09, p=0.005 compared with 2011, 12 and 13 respectively).

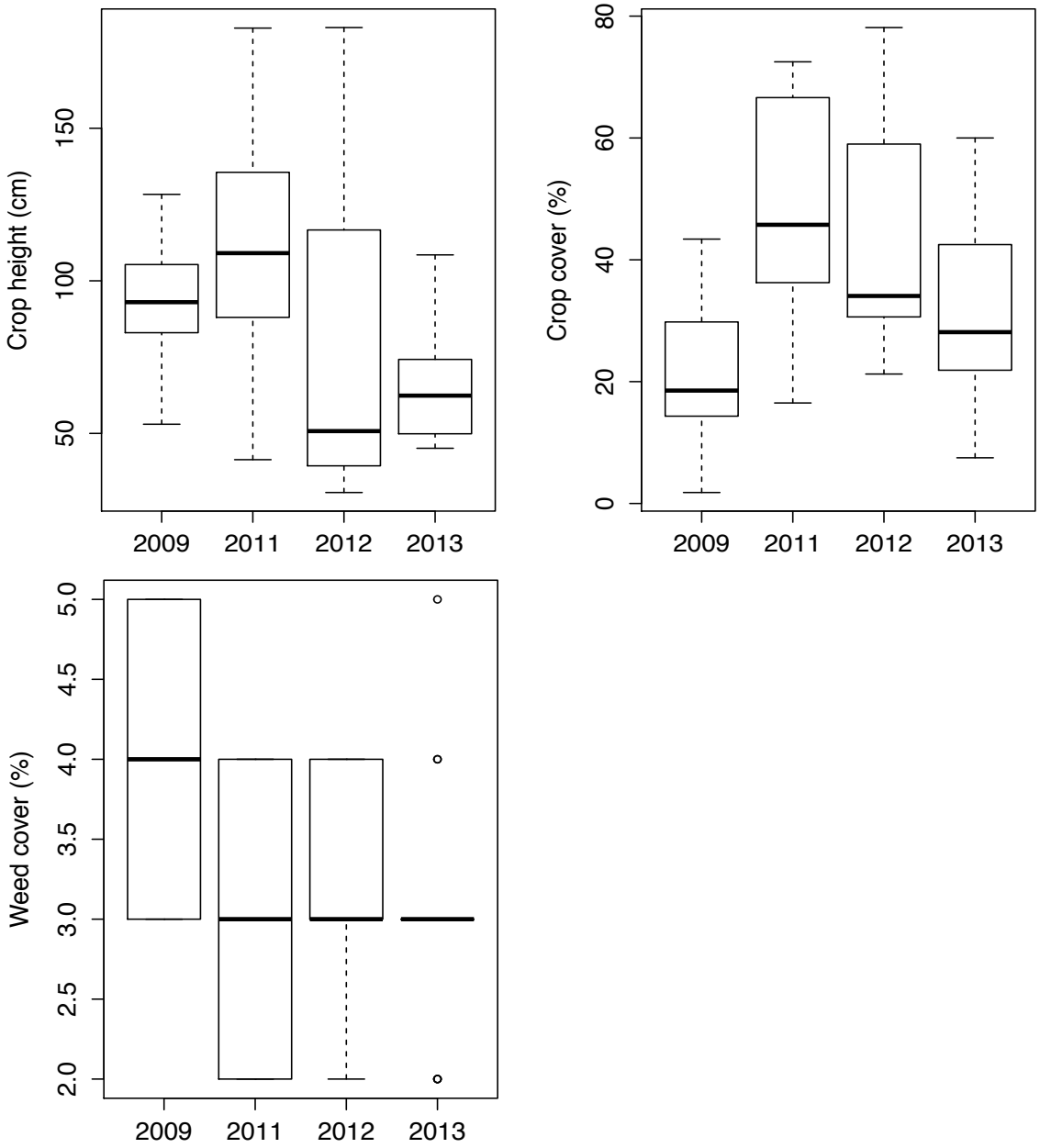


Figure 2.11 Comparison of Miscanthus characteristics between years. 2009 data from Bright *et al.* (2013), n=10. Crop height: 2009 vs 2012 t=2.37 p=0.02; 2009 vs 2013 t=2.64, p=0.01; 2011 vs 2012 t=2.30 p=0.03; 2011 vs 2013 t=2.48, p=0.02. Crop cover: 2009 vs 2011 t=-4.72, p=0.0007; 2009 vs 2012 t=-4.19, p=0.0008; 2009 vs 2013 t=-2.00, p=0.05. Weed cover: 2009 vs 2011 t=2.71, p=0.01, 2009 vs 2012 t=2.81, p=0.009, 2009 vs 2013 t=3.09, p=0.005.

2.6 Discussion

Lapwing nests were distributed fairly evenly between biomass and traditional arable crops; there was no evidence of avoiding Miscanthus or SRC willow as nesting habitat. Although a lower proportion of Miscanthus fields were occupied by lapwings than cereal and beet fields, this difference was not significant; perhaps indicating the need for a larger sample of fields in future studies to increase the detectability of a significant difference in distribution among crop types. Lapwings appeared to prefer nesting in beet to other crop types, but not significantly so. Nests here appeared to be more exposed, but perhaps the bare soil offered good nest camouflage whilst providing adults with a clear, unobstructed view of approaching predators (Vickery *et al.* 2001; Wilson, Whittingham, & Bradbury 2005; Sheldon, Chaney, & Tyler 2007).

Nests in Miscanthus experienced lower hatching success than those in traditional arable crops. It is likely that the height and sward structure of the crop disrupts anti-predator behaviour by impeding the detection of predators from the nest (Galbraith 1988b; Whittingham & Evans 2004; Sheldon *et al.* 2004; Bright *et al.* 2013). Although no correlation was found between crop characteristics and hatching success, lapwings avoided fields of taller swards with greater crop cover, suggesting that the birds select optimum habitat so that the effect on hatch success is benign. A similar pattern was found by Nicholls (2010), where crop height negatively affected nest presence but within the range selected, crop height actually had a positive effect on hatching success.

Despite the reduced nesting success in Miscanthus, there is no evidence from this study of lapwings avoiding this crop as a nesting habitat. Given that site fidelity is often influenced by previous experience and success (Thompson *et al.* 1994; Wilson, Vickery, & Browne 2001), lapwings may be expected to distribute themselves accordingly, but this was not observed in the current study. Berg (2002) did not find any effect of previous hatching success or indeed of any factors affecting success on site selection, and in fact noted a lower success rate on the lapwings' preferred habitat. Bright *et al.* (2013) observed higher densities of lapwings in Miscanthus than control crops, although these comparisons were with winter wheat, which is likely to be avoided anyway due to the density and homogeneity of the sward (Wilson,

Vickery, & Browne 2001). It is also not known whether the birds observed in that study were using the crop as a breeding or foraging ground.

One possible reason for adult lapwings continuing to nest in Miscanthus despite it affording lower rates of hatching success, could be that it acts as an ecological trap, whereby it appears suitable at the beginning of the breeding season, when nest sites are being selected, but it quickly becomes too tall and dense for efficient predator defence, once clutches have already been laid (Bright *et al.* 2013). In the current study, lapwing nest density decreased over time, and fields of taller Miscanthus with greater crop cover tended to be avoided, indicating the potential for the crop to become unsuitable. Given the expected increased density of more established commercial crops (Gove & Bradbury 2010), supported by the current study, this suggests Miscanthus may be unsuitable as a nesting habitat in the future.

The vast majority of nest losses in this study were attributed to predation as evident in many cases from the egg remains (large egg fragments found in or near the nest or the nest completely empty of contents; A. Hoodless, pers. comm.). Failure due to agricultural operations, identified by complete disappearance of the nest (Sheldon, Chaney, & Tyler 2007), and nest abandonment were rarely observed, although if any nests were predated having already been abandoned, this would not have been differentiated from straightforward predation. Predation is often cited as the main cause of nest failures among lapwings, in studies across the UK and Europe (Galbraith 1988b; Seymour *et al.* 2003; Teunissen *et al.* 2008), and failure due to predation has increased in recent decades, being the main cause of nest failure in the 1990s (Chamberlain and Crick 2003). Assuming therefore that predation was the main cause of nest failure, this suggests greater vulnerability of nests in Miscanthus than those in other crops, most likely due to the impeded detection of predators in the dense crop. Additionally, the extra cover and low input management may support more invertebrates and plants, thus attracting more predators (Sage & Tucker 1997; Donald *et al.* 2002; Dauber, Jones, & Stout 2010). No information was gathered regarding predator identity; it was hoped that predator identification from egg remains could be carried out in 2013, along with a more thorough investigation of predation risk and its contribution to lapwing hatch success, but the high success rates and lack of variation made this impossible. Cameras and

temperature loggers are increasingly used to identify predators, to at least attribute nest losses to nocturnal (mammalian) or diurnal (avian) predators. Most losses are thought to occur at night (Stillman *et al.* 2006), with foxes accounting for up to 90% of predation events (Teunissen *et al.* 2008), therefore monitoring of predator densities during the day will be of limited value, but in tandem with surveys of other predator evidence, such as fox scats, could provide valuable insight into the level of risk.

Predation has recently received considerable attention as perhaps a more important contributor to reduced reproductive output of lapwings than changing land use *per se*, but it is important to note the interaction between these two mechanisms of decline, highlighted by the higher failure and therefore predation rates of nests in Miscanthus in this study. By altering sward structure and landscape heterogeneity, land use change disrupts predator detection and increases vulnerability of nests, thus facilitating increased predation rates (Whittingham & Evans 2004; MacDonald & Bolton 2008a). It should be argued, therefore, that without the change in land use, predation may not become such a limiting factor to ground-nesting bird populations.

Lapwings defend their nests by mobbing approaching predators, and benefit from co-operative defence of nests within aggregations (Baines 1990; Berg, Lindberg, & Kallebrink 1992; Šálek & Šmilauer 2002; Seymour *et al.* 2003; Eglington *et al.* 2009). In the current study, birds were often observed joining others to mob predators at neighbouring nests, but the positive relationship between nest density and hatching success often documented was not seen in the current study (see also Galbraith 1988b). Hatch success increased with increasing field size. Although control fields were slightly bigger than Miscanthus fields and size increased each year, which may explain greater success in controls and in 2013, these differences in field size were not significant. The increased success of nests in larger fields is likely to be a result of these nests being further from boundaries, and thus at lower predation risk (Stillman *et al.* 2006; MacDonald & Bolton 2008a), although no effect of distance to boundary was detected. Another possible cause of nests in 2013 being more successful than in other years could be the later onset of breeding; mean first egg date was 18 days later in 2013 than 2012. Nest timings could only be calculated with accuracy for successful nests, so it is not known whether there was any effect of timing on success rates. Eggs laid earlier may experience cooler

temperatures and therefore require longer incubation periods thus increasing the length of time exposed to predators (Lack 1947; Smith 1989). Cooler temperatures may also raise the energetic requirements of the parent birds (Williams 1996; Thomson, Monaghan, & Furness 1998), so they will either spend more time off the nest, leaving the eggs at risk, or there will be increased traffic to and from the nest, thereby drawing attention to the nest and increasing predation risk (Wiebe & Martin 2000; Lislevand *et al.* 2005). Later nests may also be at an advantage of greater food availability, so adults do not need to forage far or for long periods, reducing the time spent off the nest (Beintema *et al.* 1991; Evans *et al.* 1997).

Rather than the crop acting as an ecological trap, it could be that the birds are continuing to nest in Miscanthus because there are other factors that make it an attractive crop in which to nest. Perhaps, as has been suggested for skylarks, (Donald *et al.* 2002), nest sites are not selected purely based on maximising nest survival rates, but on other factors that may lead to greater productivity. Although lapwings are single-brooded species, the opportunity to lay replacement clutches is important to mitigate for low hatching success (Cramp & Simmons 1983). Given the potential curtailment of the breeding season in biomass crops due to their rapid growth, it seems unlikely that Miscanthus offers more opportunities to re-nest following failures than in control crops. There was however, no evidence of this curtailment of breeding season in this study, as first egg dates did not differ between crop types.

A diverse sward structure with areas of open crop is important for allowing access to invertebrates, whilst an undisturbed crop may be home to larger invertebrates and thus a more valuable food resource (McCracken & Tallowin 2004). The frequency of tussocks within grass fields has been shown to be related to the distribution of nesting lapwings (Milsom *et al.* 2000), with birds preferring to nest in fields with tussocks than without, perhaps because the tussocks increased heterogeneity, made nests less visible to predators and provided cover for chicks (Baines 1990; Milsom *et al.* 2000). Ideally therefore, the breeding habitat should contain areas of shorter crop for nesting and foraging, and taller areas for cover (Vickery *et al.* 2001; van der Wal & Palmer 2008). This is where Miscanthus could be beneficial to lapwings, as its patchy, weedy nature may lead to improved foraging

opportunities (Bellamy *et al.* 2009; Dauber, Jones, & Stout 2010), whilst aiding nest and chick crypsis (Whittingham & Evans 2004). The fact that hatching success increased with increasing proximity to the sward, suggests a reduced predation risk near swards, similar to the reduced predation observed within arable crops rather than tramlines (Donald *et al.* 2002). Additionally, nesting on a raised position within a sward may afford incubating adults a better view of approaching predators. As the crop grows, however, this may become a less viable option as shoots grow up through the nest, causing potential damage. The proportion of nests found in Miscanthus swards rather than open patches increased between 2012 and 2013 which may indicate the development of a predator avoidance strategy by birds nesting within the crop. This is only based on two years of data so it would be worthwhile monitoring nest location within Miscanthus long-term to see if lapwings are indeed adapting their behaviour to adjust to this relatively new crop.

Lapwings could be nesting in Miscanthus despite lower hatch success if they experience some other benefit in the crop. There is no evidence of fitness benefits originating from the egg stage, as egg size did not vary among crop type, although there is a suggestion of eggs from Miscanthus fields being longer than others. Increased sample sizes and an investigation of egg volume would be of value to investigate the possibility of larger eggs being laid in Miscanthus fields.

Foraging habitat is of great importance in determining territory establishment (Galbraith 1988b), both in terms of providing food for the adults during the incubation period, and for chicks after hatching. Chicks are at greatest risk of predation when travelling from the nest to foraging grounds (Galbraith 1988b) so if chicks hatching in Miscanthus were able to persist in the natal ground longer than those hatching in control fields, this may confer increased fledging success, thereby explaining the apparent non-avoidance of Miscanthus. This hypothesis requires sufficient foraging resource to allow persistence in the natal ground. Conversely, shorter persistence time may be expected in Miscanthus if the sward structure is unsuitable for foraging or predator avoidance. Thus it was hoped that by comparing the length of time chicks remain in fields after hatching between crop types, some assessment of the value of biomass crops to chicks was possible but due to the small sample sizes involved, no conclusions can be drawn. This would be an interesting area to explore in future

studies, by expanding sample sizes and ringing or radio-tagging chicks to determine foraging destinations. Some assessment of food availability and accessibility within different crop types would also be beneficial to examine the potential benefits to chicks.

Although no interaction between crop type and year was detected, the difference in success between crop types appears mainly to be the result of low hatching success in Miscanthus in 2012; success rates in control crops did not vary significantly between years. Between year variations are commonly found in studies of nest success (e.g. Stillman *et al.* 2006; Smart *et al.* 2013) either as a result of general stochasticity of the system, or due to weather conditions affecting the birds directly or indirectly via habitat and food availability (Milsom *et al.* 2000). Spring 2012 was one of the wettest on record (especially in the East Midlands, where field sites were located), following a mild February-March (Met Office 2013). Whilst nesting for many resident species started early, the wet weather delayed the season for many migrants (BTO 2012). This could have implications for lapwings, if they started nesting early, but were then adversely affected by the prolonged wet period, via increased failure rates and reduced capacity to lay replacement clutches. In contrast, late spring 2013 was much cooler and drier, following a wet start (Met Office 2013), which is likely to have delayed the breeding season, thus the weather may explain much of the inter-year variation. It is important to remember that productivity is not a result of the breeding ground environment alone, but may also be affected by foraging habitat and winter conditions, e.g. the dry winter of 2011 may have made it more difficult for female lapwings to reach peak condition in time for the 2012 breeding season (A. Hoodless, pers. comm.). This emphasises the importance of long-term monitoring; as nesting success may vary considerably between years, and different crop types may exert different effects on success each year.

Presented here is evidence of reduced hatching success for lapwings nesting in Miscanthus compared with traditional arable crops, but no indications of avoidance of Miscanthus as nesting habitat, or of curtailment of breeding season in Miscanthus. Success varied between years, thus long term monitoring of lapwing habitat choice and nesting success in biomass crops is required to fully assess the risk of biomass crop expansion. If breeding in Miscanthus does indeed make lapwing nests more

vulnerable in some years, this has important implications for future expansion of the industry (see Chapter 5 for further discussion).

2.7 Further work

Long-term monitoring of lapwing hatching success in biomass would be useful to determine patterns that are perhaps obscured by yearly differences. When investigating variation in breeding success across crop types, it is important to look at a range of years. Breeding conditions should not be looked at in isolation; conditions throughout the rest of the year should also be taken into account (Siriwardena, Baillie, & Wilson 1998; Bradbury *et al.* 2000; Newton 2004). Long-term monitoring would also be beneficial to detect any behavioural adaptations to nesting in Miscanthus, as it becomes more established. These adaptations may result in the avoidance of the crop altogether, or the development of alternative nesting strategies such as nesting within swards to increase nest crypsis.

Examining nesting success for the whole breeding season would be beneficial to determine whether Miscanthus confers any fitness advantages post-hatching that may explain the non-avoidance of the crop despite lower hatch success. Estimates of fledging success could be obtained by radio-tagging chicks, which would also serve to identify foraging destinations of chicks. Ideally, adult birds would also be ringed or marked (Galbraith 1987) to ascertain whether the number of clutches produced per pair is affected by crop type, and if replacement clutches have a higher success rate, as found by Berg, Lindberg, & Kallebrink (1992). Re-nesting is rare after failure of broods, so chick rather than egg survival is likely to be the more important determinant of productivity (Stillman *et al.* 2006). With these data, estimates of productivity can be made (number of chicks fledged per pair) and compared between habitats. Additionally, looking at the whole breeding period will help to identify the stage at which offspring are most vulnerable (as eggs or chicks), and whether this varies between habitats, thus allowing more targeted management options to be implemented.

Access to foraging resource within the crop is expected to be impeded by the structure of Miscanthus (Gove & Bradbury 2010; Sage *et al.* 2010), but this may be mitigated if the resource is of high quality or if the habitat affords protection from

predators. Whilst evidence for the former is fairly inconclusive (Semere & Slater 2007a; Bellamy *et al.* 2009; Dauber, Jones, & Stout 2010), the latter could be true of Miscanthus due to its greater sward heterogeneity than control crops. Invertebrate sampling of the crop could be carried out to determine the quality of the resource, and to investigate its availability to lapwing chicks, faecal samples of chicks could be taken. Given the difficulties of capturing and sampling lapwing chicks (pers. obs.), the technique used by Hitchcock (2010) of taking faecal samples from human-imprinted pheasant chicks could be used. Although these samples may not be representative of lapwing chick diet, they would give some indication of what is available to chicks foraging on the ground within biomass crops. Chicks are at their most vulnerable when moving to foraging grounds (Galbraith 1988b), so it would be of value to examine the proximity of each field to potential foraging grounds.

3 Comparison of methods to estimate hatching success

3.1 Abstract

When examining the factors that affect nest success, it is important to consider how the method used to estimate success may influence results. Many different methods of estimating nest success have been developed, each varying in their accuracy, ease of application and therefore use in the literature. Using the data collected in Chapter 2 to investigate the hatching success of lapwings in different crop types, a comparison of three of the most widely used methods of estimating hatching success was carried out, to indicate the differences that may result from method choice. These methods were the apparent estimator, the Mayfield method and the more recently developed logistic-exposure method. The apparent estimator (the number of nests that hatched as a proportion of total nests in the study) produced estimates consistently higher than the other two methods used. The Mayfield and logistic-exposure methods both incorporate the length of time for which the nests are observed, thereby correcting for bias caused by nests that are found late or remain undiscovered. The variation in estimates produced by the different methods would have led to quite different conclusions regarding the hatching success of lapwings; using Mayfield and logistic-exposure estimates, hatching success varied with year and crop type, but using the apparent estimator, these differences were not significant. This illustrates the influence of estimation method on results, and thus the importance of method choice and consideration of these issues when comparing nest success in the literature.

3.2 Introduction

The rate of nest success is usually defined as the proportion of nests in a study that fledge at least one young. This rate has long been used to investigate differences in productivity between species or age groups, or to reflect differences in habitat quality, management regime (Hazler 2004; Johnson 2007a). There are, however, many different methods that can be used to calculate and compare nest success rates, each with their own limitations; so many in fact, that a 168-page document was

produced to accompany a workshop held in 2005 that aimed to evaluate the different methods and guide biologists as to which methods are the most appropriate (Jones & Geupel 2007).

The most straightforward and intuitive way to measure nest success is to calculate the number of successful nests as a proportion of the total number of nests found. The problem with this apparent estimator, however, is that it assumes that all nests have an equal chance of being found, which is rarely the case in studies of wild populations. Some nests will be missed from the dataset completely as they failed before being discovered and are simply not found at all, whilst others that are found later in the incubation period will inflate the success rate, as they will have a higher chance of surviving to hatching than those found shortly after laying (Mayfield 1975; Johnson 2007a).

To overcome these issues, Mayfield (1975) introduced the concept of exposure days, whereby a daily survival rate for the nest is calculated based on the number of days over which a nest is observed. An estimate of nest success can then be calculated from this daily survival rate. This method has been widely used and is applied in many studies; Macdonald & Bolton (2008) reviewed 57 papers that used nest daily failure rates as calculated by Mayfield estimates to determine success of wader nests in Europe. Often the aim of nest success studies is to assess the factors that affect survival, which is where the Mayfield method has its drawbacks. Separate estimates of nest survival could be calculated for nests in each factor level and compared, but this requires enough nests in each treatment to make robust comparisons, and cannot be applied to continuous explanatory variables (Hazler 2004).

Various methods based on logistic regression have been developed to allow investigation of multiple predictor variables, and those that are continuous (e.g. Aebischer 1999; Hazler 2004). Many of these methods, together with the basic Mayfield estimate require assumptions to be made about the timing of nest failure and how to handle nests of unknown outcome. These assumptions determine the total exposure days and thus affect the resulting estimates (Manolis, Andersen, & Cuthbert 2000; Shaffer 2004). For example, should exposure days of nests of unknown fate contribute to the dataset? This then raises the question of when should

the exposure period end; when the nest was last observed to be active or halfway between the day the nest was last observed to be active and the last day it was checked? Should termination dates be extrapolated from cues such as expected hatch date? Once these questions have been answered, should the same rules then apply to successful and failed nests, or will the exposure period vary? Clearly there are many different combinations possible, as illustrated by the nine different approaches used by 22 researchers employing the Mayfield estimator alone, each of which introduce bias to varying degrees (Manolis, Andersen, & Cuthbert 2000). Ideally, methods should be standardised so that one approach is used consistently to allow more transparent comparisons across studies.

Until such a time is reached, efforts should be made to minimise the bias in estimates. This may be achieved by keeping the interval between nest visits small (ideally daily), or by adopting a method that does not require assumptions of termination dates (either success or failure). The logistic-exposure method (Shaffer 2004) was developed to overcome this issue by only requiring the length and outcome of each observation interval. Shaffer (*loc cit*) argues that the method developed by Aebischer (1999) is merely a special case of the logistic-exposure method, in which visits to nests occur daily therefore exact failure date is known (Shaffer 2004). This logistic-exposure method is gaining popularity, having been used in various recent papers (Peak, Thompson, & Shaffer 2004; Winter *et al.* 2004; Toral & Figuerola 2012). The results of the Mayfield estimator and logistic-exposure method are often comparable, thus indicating the continued value of Mayfield estimator in nest success studies due to its simplicity, but implementation of the logistic-exposure method allows more complex modelling and the inclusion of random effects into mixed-effects models, a key advantage over the Mayfield estimator (Rotella, Dinsmore, & Shaffer 2004; Lloyd & Tewksbury 2007).

When investigating factors that affect nest success, it is important therefore to be aware of the different methods of measuring nest success and the impact the choice of method can have on estimates. In the discussion of how conversion of land from arable production to biomass cultivation may affect the hatch success of lapwings, the logistic-exposure method was used to estimate hatching success. This was deemed the most appropriate method for the task, as it accounts for the length

of observation period and allows for the inclusion of continuous explanatory variables within mixed-effects models, as well as being relatively straightforward to implement (Chapter 2). The data collected also presented an opportunity to examine additional methods of estimating hatch success and to compare the results with those obtained from the logistic-exposure models used in Chapter 2.

3.3 Aims

To highlight the effect that choice of method can have on estimates of nest success, analysis of the hatch success data collected in Chapter 2 was carried out using three methods of estimating hatch success. The following methods were compared:

a) logistic regression of nest outcome (analogous to the apparent estimator),

b) Mayfield estimates of daily survival rates,

c) the logistic-exposure method of Shaffer (2004).

3.4 Methods

Lapwing nests were found and monitored in a number of Miscanthus, SRC and arable fields in Lincolnshire and Nottinghamshire. On discovery, the location of nests was recorded using a hand-held GPS (Garmin Etrex), and clutch size, weight (±0.1 g), length (±0.1 mm) and breadth (±0.1 mm) of each egg recorded. This enabled calculation of egg density (mean egg weight/mean egg volume; where volume = length x breadth^2 x 0.457) and estimation of expected hatch dates using the equation developed by Galbraith (1988b); days until hatching = 150.84 x density - 140.68 (± 2.3 days). Nests were checked every 4-5 days, or more frequently as expected hatch date approached, and were recorded as successful (at least one egg hatched), unsuccessful (predated, abandoned or destroyed) or active (eggs present and warm). The presence of small egg fragments (c. 1 mm in size) in the nest lining was assumed to indicate a successful hatch, whilst a completely empty nest or the presence of large pieces of egg shell in or near the nest were taken as indications of predation (Green, Hawell, & Johnson 1987; Galbraith 1988b; Hart *et al.* 2002). For further details of fieldwork methods, see Chapter 2.

3.4.1 Data analysis

Three methods of quantifying nest success were used: a logistic regression of nest outcome, (analogous to the apparent estimator); Mayfield estimates; and the logistic-exposure method. The apparent estimator models nest fate as a binary outcome in relation to crop type by a generalised linear model (GLM) with quasibinomial error structure. To take into account the length of time in which nests had been under observation, exposure days were incorporated as a covariate (e.g. Pasitschniak-Arts, Clark, & Messier 1998), although this method has been criticised as it assumes an additive effect of exposure period (Shaffer 2004). The logistic-exposure method developed by Shaffer (2004) uses the nest monitoring interval as the sampling unit and is based on a GLM with binomial error distribution and a modified link function: $g(\theta) = \log_e (\theta^{1/t}/[1 - \theta^{1/t}])$, where θ is the survival estimate for the monitoring interval and t is the interval length in days. GLMs were therefore run to produce estimates of hatch success, using the apparent estimator, or Daily Survival Probabilities (DSPs), using the logistic-exposure method.

To arrive at Mayfield estimates of hatching success, DSPs are calculated manually for each group of nests. Where hatch or failure date was unknown, the midpoint was used between the last day the nest was checked and the last day the nest was observed to be active to calculate exposure days (Manolis, Andersen, & Cuthbert 2000). For each group of nests (i.e. in each crop type), DSPs were calculated by dividing the total number of nest failures by the total number of exposure days and subtracting this from one. Standard errors of these estimates were calculated according to Johnson (1979; Eq. 1), and groups were compared using the test statistic developed by Hensler & Nichols (1981; Eq. 2).

Eq. 1
$$Variance(v) = \frac{1}{\left(\frac{(Exposure)^3}{(Exposure - losses) \times losses} \right)}$$

Eq. 2

$$z = \frac{\left|DSP_1 - DSP_2\right|}{\sqrt{{v_1}^2 + {v_2}^2}}$$

Given the treatment of pseudoreplication within a mixed model structure is not possible for the Mayfield estimate, the comparison of all three nest success methods was initially carried out on data from 2012 alone. This year was chosen in particular because it showed sufficient variation in hatch success. Similarly, the Mayfield method does not allow the examination of continuous explanatory variables, so in initial investigations of the impact of success method choice on estimates, only the effect of crop type was examined.

For the purposes of illustration, hatch success across all three years was estimated, using the apparent estimator and Mayfield methods and compared to the results obtained in Chapter 2 using the logistic-exposure method. It should be noted that the Mayfield estimates may be affected by the presence of multiple nests in the same fields year after year, so strict comparisons cannot be made using data from all years. Hatching success measured by the logistic-exposure and apparent estimator methods was examined using mixed-effects models, with field and site as random effects to avoid pseudoreplication arising from multiple nests in the same field and fields within the same site. Very little variation was attributable to site, so this was omitted from final models.

Unlike the Mayfield method for estimating nest success, both the apparent estimator and logistic-exposure methods enable the comparison of continuous effects within a generalised linear model structure. Several models using these two methods were therefore run using all years' data to investigate the effects of other explanatory variables besides crop type, including, field size (ha), distance to boundary (m), and nest density (no of nests per ha, log transformed). To avoid over-parameterisation of the models, variables were added to the model one by one, to reach a maximal model and the significance of each was tested by a process of stepwise deletion. Terms were deleted and subsequent models compared via

ANOVAs; only those variables whose removal resulted in significantly different models ($p<0.05$) were retained. Terms were removed in this way until the minimum adequate model was reached (Crawley 2007).

Both Mayfield and logistic-exposure methods produced estimates of daily survival rates, which could then be converted to hatch success by raising them to the power of 32, the average length of egg-laying and incubation (Berg, Lindberg, & Kallebrink 1992). These could then be compared with the hatch success rates produced by the logistic regression model (apparent estimator).

3.5 Results

In 2012, lower daily survival rate is observed in Miscanthus crops compared with control crops of barley and beet irrespective of analysis method (2012: Mayfield method $z=-2.27$, $p=0.02$; logistic-exposure method $z=-2.56$, $p=0.01$). The GLM, analogous to the apparent estimator method, showed the same pattern of lower success in Miscanthus than controls when testing success rate over the whole egg stage as the response variable ($z=-2.06$, $p=0.04$). Nesting success did not differ between cereal and beet fields ($p=0.08$, beet nests=13 in 2 fields, cereal nests=10 in 4 fields), so these were kept as one control category. SRC nests were excluded from the analysis as only two were found in 2012, both of which successfully hatched. Estimates of hatching success produced by the two methods incorporating observation period (Mayfield and logistic-exposure methods) were very similar, the maximum difference in estimates being 3.3%. The GLM of nest outcome with observation days as a covariate, produced estimates of hatching success 7.2-21.2% higher than estimates produced by the other methods (Figure 3.1).

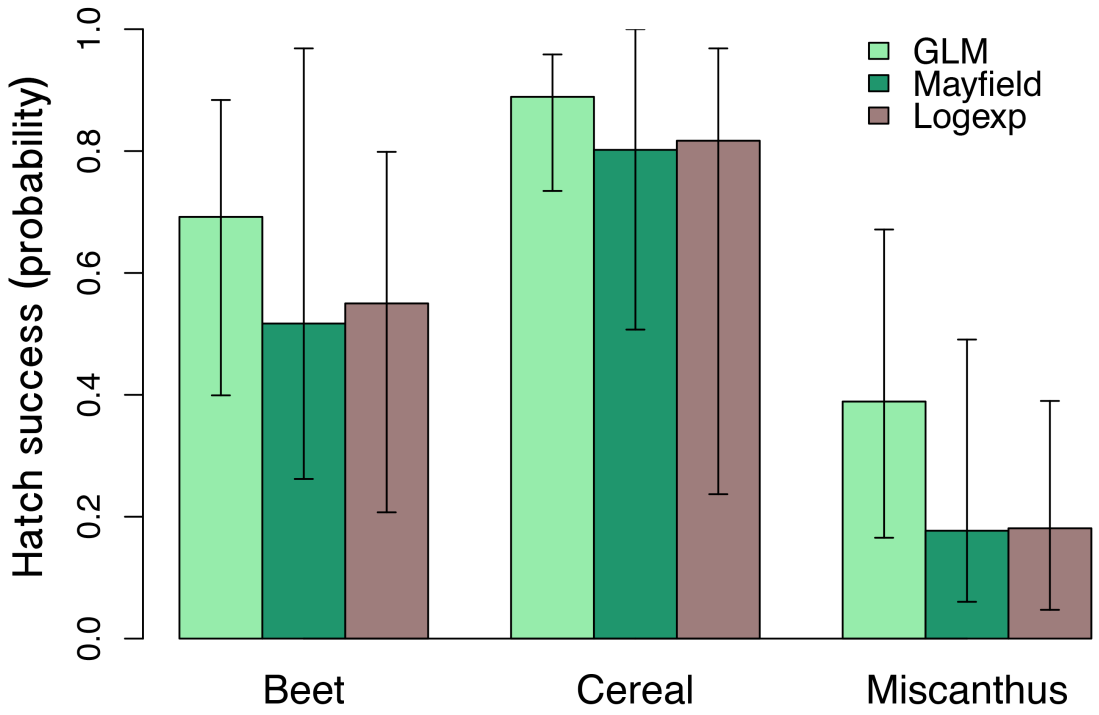
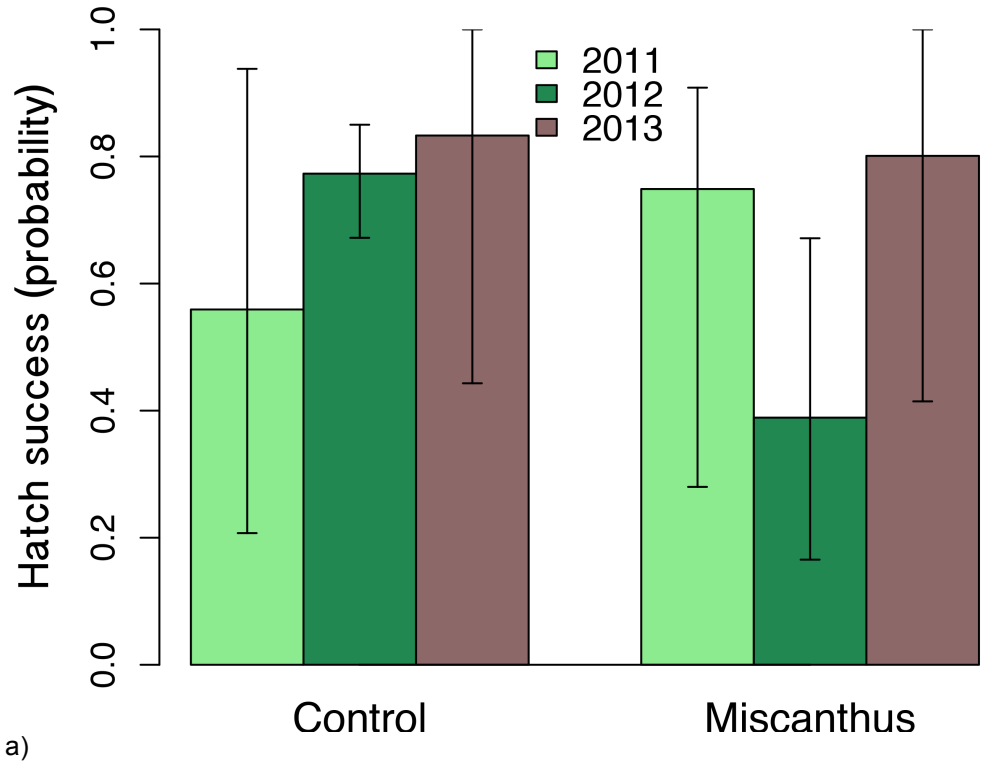
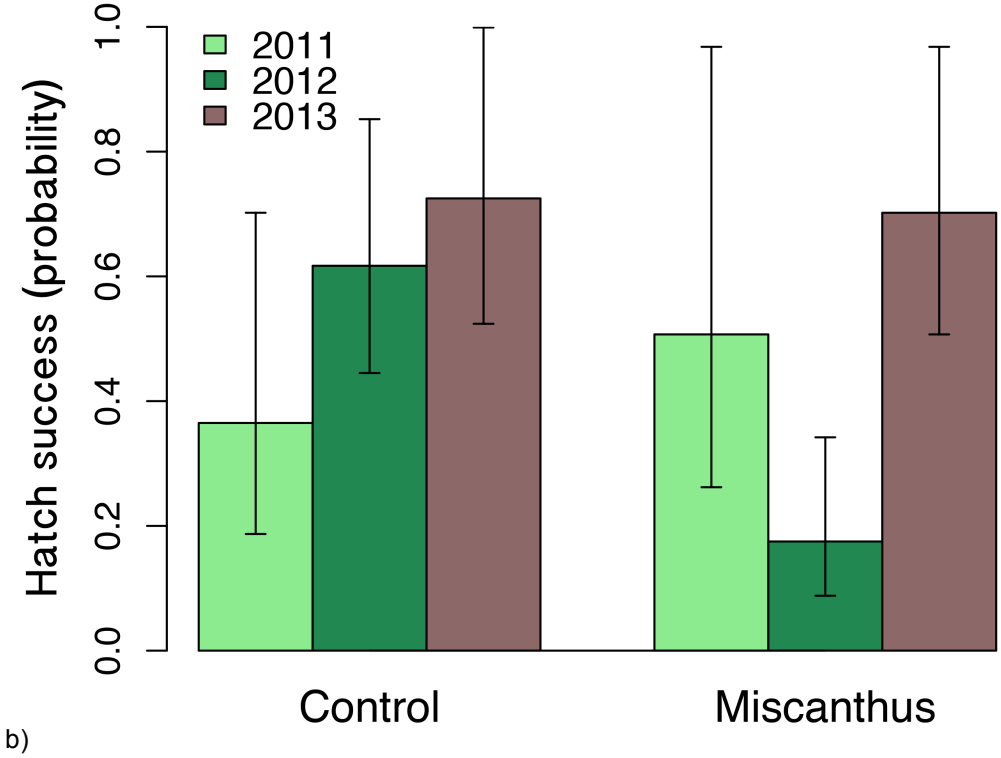


Figure 3.1 Hatch success of lapwing nests in 2012 calculated by three different methods, with standard errors. The GLM method is a logistic regression of outcome, analogous to the apparent estimator, Mayfield methods and logistic-exposure methods (Shaffer 2004; Logexp above) provided daily survival probabilities which were raised to the power of 32 (length of lapwing incubation period) to obtain hatch success rates presented here. Beet nests = 13, Cereal = 9, Miscanthus nests = 18. Hatch success was lower in Miscanthus using all three methods z=-2.06, p=0.04(GLM), z=-2.27, p=0.02 (Mayfield), z=-2.56, p=0.01 (logistic-exposure).

Hatch success estimates produced over the three years using the logistic-exposure method are reported in Chapter 2, but for ease of comparison are presented again here. The Mayfield and logistic-exposure methods again showed analogous trends, differing from tests using the GLM method to derive hatch success. Whereas nests in Miscanthus were less successful than those in control crops (Mayfield: z=-2.03, p=0.04; logistic-exposure: z=-2.06, p=0.04), and higher success rates were achieved in 2013 than in other years (Mayfield z= 2.09, p=0.04; logistic-exposure: z=2.17, p=0.03), neither of these differences was significant using the apparent estimator method (z=-1.2, p=0.2 and p=0.3 for crop type and year respectively, Figure 3.2, Table 3.1). Success rates increased with the length of observation period (z=2.4, p=0.01), which was longest in 2013. This could help to explain why according to the Mayfield and logistic-exposure methods (which account for length of observation), success was higher in 2013.



a)



b)

Figure 3.2 Hatch success of lapwing nests calculated using two methods. a) a simple GLM with observation days as a covariate (i.e. apparent estimator), where neither crop type nor year affected success (z=-1.2, p=0.2 and p=0.3). b) Mayfield method, where higher success rates were recorded in Control crops (z=2.03, p=0.04 and in 2013 z=2.09, p=0.04), with standard errors. The logistic-exposure method revealed a very similar pattern to the Mayfield method (see Figure 2.7 in Chapter 2). Control n= 9, 22, 12, Miscanthus n= 4, 18, 21 in 2011, 2012 and 2013 respectively.

Table 3.1 Lapwing hatch success in Miscanthus, and control fields, presented as daily survival probabilities (DSP) using three different methods.

Year	Crop	Nests (Successes)	Exposure Days	Apparent Estimator		Mayfield		Logistic-Exposure	
				DSP	SE	DSP	SE	DSP	SE
2011	Miscanthus	4 (3)	47.5	0.991	0.03	0.979	0.02	0.979	0.05
	SRC	4 (4)	50.5	1	-	1	-	1	-
	Control	9 (5)	131	0.982	0.03	0.969	0.02	0.971	0.01
	Total	17 (12)	229	0.989	0.02	0.978	0.01	0.980	0.01
2012	Miscanthus	18 (7)	208.5	0.971	0.01	0.947	0.02	0.948	0.02
	SRC	2 (2)	8	1	-	1	-	1	-
	Control	22 (17)	342	0.992	0.01	0.985	0.01	0.987	0.01
	Total	42 (26)	558.5	0.985	0.02	0.971	0.01	0.973	0.01
2013	Miscanthus	21 (17)	375.5	0.993	0.04	0.989	0.01	0.990	0.01
	SRC	3 (3)	48.5	1	-	1	-	1	-
	Control	12 (10)	222	0.994	0.04	0.990	0.01	0.991	0.01
	Total	36 (30)	646	0.994	0.01	0.991	0.01	0.991	0.01

Given the differences in hatching success estimates obtained by the apparent estimator compared to the other two methods, the apparent estimator was used to determine the effect of local habitat on hatching success. These estimates can then be compared with the logistic-exposure estimates calculated in Chapter 2. The results were largely similar; hatching success increased with field size ($z=2.8$, $p=0.03$; cf logistic exposure: $z=2.9$, $p=0.004$). Just as distance to boundary and nest density were not retained in the logistic-exposure model, these were also dropped from the model based on the apparent estimator (removing these variables from the models resulted in p values of 0.2 and 0.08 respectively). The length of the observation period was not retained in the model ($x^2=2.97$, $p=0.08$).

3.6 Discussion

Three different methods were used to compare nesting success in Miscanthus and control crops, over three years. The variation in estimates produced by different methods shows the importance of appropriate method selection for evaluating factors affecting nest success. Using a straightforward GLM to estimate nest success may seem the obvious way to approach the problem but this produces very different nest success estimates. The proportion of successful nests from the total number found is likely to overinflate nest survival, which is shown by the consistently higher estimates produced by this method, compared with Mayfield estimates and the logistic-exposure method. In the most extreme case, the apparent estimator produced a hatch success rate up to 24.8% higher than the rate estimated by the other two methods (74.9% success rate in Miscanthus in 2011 estimated by the simple GLM compared with 50.1% and 50.7% using the Mayfield and logistic-exposure methods respectively). Looking at the two methods that account for the bias caused by length of exposure time, these produce estimates that only differed by a maximum of 4.1% (for nests in Control fields in 2012, hatch success rate =61.7% and 65.8% using the Mayfield and logistic exposure methods respectively).

It has been argued that if the aim is to determine the factors that affect nest success rather than obtain accurate estimates of nest success, this apparent bias will be of limited concern, as it will be consistent across all factor levels and therefore will result in similar conclusions (Manolis, Andersen, & Cuthbert 2000). This reasoning is

supported by the results for 2012, as nests found in Miscanthus fields experienced lower hatching success than those in control fields, regardless of method used, although the effect sizes differ between methods. Examining the data across all three years, however, the results vary according to the method used; whilst the apparent estimator showed no difference in hatch success between years or crop type, those methods taking into account the length of exposure period suggest an effect of both crop and year. Given the length of observation period was retained as a significant variable in the apparent estimator model, this indicates the importance of incorporating observation days into estimates of nesting success. It must be noted, however, that the Mayfield method cannot account for pseudoreplication, thus using it to estimate hatch success over all three years, with the inclusion of nests from the same fields year after year may introduce bias.

Many other methods have been developed to analyse nest success (e.g. Aebischer 1999; Dinsmore, White, & Knopf 2002); a total of 23 different methods were identified by Johnson (2007b), though with varying employment beyond the scope of the original study. Approaches vary in the fieldwork effort involved and the software capabilities required, so the choice will depend on the objective of the study and the effect of disturbance on the species involved (Jehle *et al.* 2004; Rotella, Dinsmore, & Shaffer 2004; Johnson 2007b; Lloyd & Tewksbury 2007). The three methods compared here are perhaps the most widely used so by producing Mayfield and logistic-exposure estimates this will allow direct comparison with other estimates of lapwing nesting success in the literature. Although the Mayfield method is the most widely used approach, with variations in survival rate examined via t tests or ANOVAs, the logistic-exposure method is gaining recognition as a robust method that allows more complex modelling approaches and the inclusion of fixed effects.

The purpose of this chapter was to outline the influence that method choice can have on estimates of nest success. All three methods used here gave different estimates, depending on how the length of observation period was accounted for. The size of discrepancy between estimates varied, and in some instances led to different conclusions. The dataset used here was relatively small and involved nests that were checked frequently. For larger studies that involve nests that are harder to find and are checked at longer intervals, inaccuracies caused by bias in the method

chosen may build, affecting conclusions. Further comparisons with larger datasets would test this hypothesis and help determine the most appropriate method to use. Until a unified approach for nest success methods is reached, the biases in the different methods and how they may influence the conclusions drawn must be duly considered when trying to compare estimates across the literature.

4 Foraging potential of biomass crops for breeding songbirds

4.1 Abstract

Given the low management intensity and reduced chemical inputs biomass crops in the UK receive, they may provide good invertebrate resource for birds in the breeding season. Work thus far has suggested the invertebrate abundance and species richness of Miscanthus fields originate from the weed community therein, while the SRC crop itself hosts numerous invertebrates. This study aimed to investigate to what extent birds nesting in SRC and Miscanthus field boundaries foraged within the crops themselves (rather than in the weeds). Fewer invertebrates were found in Miscanthus crop than in SRC, but there was no difference in diversity between the two. Invertebrate abundance was lower in faecal samples recovered from whitethroat (*Sylvia communis*) chicks nesting in the boundaries of Miscanthus than those nesting around SRC. The most abundant items in whitethroat and yellowhammer (*Emberiza citronella*) faeces were Diptera, Larvae, Coleoptera and Arachnids, but fewer Coleoptera were found in sweep samples from Miscanthus than SRC, a difference which was not reflected in faeces. SRC may therefore provide a better foraging resource for nesting birds than Miscanthus. This is of concern if birds in Miscanthus have to increase foraging distance or are unable to maintain provisioning rate, resulting in chicks of poorer body condition and thus lower survival probability than those nesting elsewhere. Further work is suggested, and the implications for the expansion of biomass crop production are discussed.

4.2 Introduction

4.2.1 Invertebrate availability and its effects on farmland birds

Adequate food supply is crucial to the maintenance of animal populations (Lack 1954; Martin 1987; Newton 2004). This is evident in the recent decline of farmland birds (Chamberlain *et al.* 1999), as decreases in weed, seed and arthropod abundance on farms have been cited as drivers of this change (Gregory, Gibbons, & Donald 2004; Hart *et al.* 2006). The effects are seen all year round; winter food

availability affects the survival rates and hence subsequent breeding densities of birds (Lack 1954), whilst during the breeding season, food supply will directly affect reproductive success (Chamberlain *et al.* 1999; Potts and Aebischer 1991). These effects on reproductive success may be seen in the timing of breeding (Perrins 1970; Svensson & Nilsson 1995; Thomas *et al.* 2001), nestling condition (Potts 1986; Brickle *et al.* 2000; Boatman *et al.* 2004; Morris *et al.* 2005; Hart *et al.* 2006), growth rate (Potts 1986; Brickle *et al.* 2000; Southwood & Cross 2002; Hart *et al.* 2006) and starvation probability (Hill 1985; Potts & Aebischer 1995; Brickle *et al.* 2000; Boatman *et al.* 2004; Morris *et al.* 2005; Hart *et al.* 2006) all of which can contribute to future breeding densities (Virolainen 1984).

Of particular importance in the breeding season are invertebrates (Moreby and Stoate 2001), even to largely granivorous species, as they provide chicks with crucial protein required for growth and energy to resist chilling (O'Connor, 1984), with the result that chicks consume a higher proportion of invertebrates than adults (Holland *et al.* 2006). The variety and quality of the invertebrate supply will affect chick growth and survival (Potts 1986; Borg & Toft 2000; Southwood & Cross 2002); a diverse diet in the early stages of chick development ensures the elements required for tissue synthesis are obtained, and minimises oversupply of toxic components (O'Connor and Shrubb 1986). A high protein diet results in faster growth and better plumage formation than a diet low in protein (Southwood and Cross 2002). This requirement for different nutrients was used to explain the presence of spiders in the diet of tit chicks even in habitats rich in caterpillars, their preferred prey (O'Connor 1984). Similarly, a high abundance of leatherjackets are often found in the diet of starling chicks, reflecting their high abundance in the environment, while caterpillars are preferred, even when harder to obtain, likely reflecting some nutritional benefit (O'Connor & Shrubb 1986). It is not just nutritional qualities that factor into prey choice, but also physical properties; for example, if too many leatherjackets are consumed, moist faeces are produced that do not form distinct sacs, thus the faeces is harder to remove from the nest. As a result some remains in the nest, subjecting the chicks to a damp, cool environment, which may become fatal due to the chicks' inability to thermoregulate (Wilson, Evans, & Grice 2009). These requirements will vary with age and species of chick, as the variety in nestling diet is replaced by high

energy foods to meet maintenance costs once thermal independence is reached (O'Connor 1984).

Generally, the invertebrates most frequently observed in the diet of farmland birds are Coleoptera, Hempitera, Lepidoptera, Diptera, Aranea and Hymenoptera (Wilson *et al.* 1999; Holland *et al.* 2006). As already alluded to, this varies between species due to differences in foraging behaviour, nesting habitat and morphology, so some invertebrate groups will be more important to some birds than others. For example, Aranea and sawfly larvae are preferred by skylarks (Holland *et al.* 2006), and corn buntings respectively (Brickle and Harper 1999), whilst Coleoptera feature more in the diet of dunnock (*Prunella modularis*) than that of yellowhammer (*Emberiza citronella*) and whitethroat (*Sylvia communis*; Moreby & Stoate 2001).

Invertebrates have, however, suffered declines on farmland as a result of agricultural intensification (Wilson *et al.* 1999; Newton 2004; Boatman, Parry, & Bishop 2007), especially the loss of marginal habitats (Brickle *et al.* 2000), frequent ploughing (Wilson *et al.* 1999), cultivation of grasslands (Brickle *et al.* 2000; Vickery *et al.* 2001) and increased grazing, leading to a loss of botanical and structural diversity of swards (Tallowin, Brookman, & Santos 1995; Berg *et al.* 1997; Vickery *et al.* 2001). On intensively managed grasslands, for example, several prey groups (grasshoppers, ants, spiders, Lepidopteran larvae) important for a number of farmland bird species have disappeared (Wilson *et al.* 1999), and this has been cited as a direct cause of the population decline of the cirl bunting (*Emberiza cirlus*) (Evans *et al.* 1997).

The increased use of pesticides has received considerable attention as a leading cause of decline of invertebrates, and more recently for the knock-on effects on farmland birds (Brickle *et al.* 2000; Newton 2004). Pesticides may affect food resources available to birds via three routes (Boatman *et al.* 2004; Hart *et al.* 2006; Bright, Morris, & Winspear 2008); insecticides reducing abundance of invertebrates, herbicides reducing the availability of plant and seed food, and herbicides reducing the plant diversity for invertebrates and thus reducing invertebrate densities. This detrimental effect of pesticides on bird populations has only been shown categorically for grey partridge (*Perdix perdix*) (Potts 1986), although it has been hypothesised for

other species, e.g. skylark (Poulsen *et al.* 1998) and corn bunting (Brickle and Harper 1999).

What is clear is that many of the invertebrate species found in bird faeces have declined on British farmland (Brickle & Harper 1999; Benton *et al.* 2002) and that these invertebrates are more prevalent in the diets of declining rather than non-declining birds (e.g. Carabid and Lepidoptera larvae; Wilson *et al.* 1999). This is due to reduced food availability affecting nestling condition, growth and chick survival as has been shown in a number of species such as skylark, yellowhammer, corn bunting and grey partridge (Boatman *et al.* 2004; Morris *et al.* 2005; Hart *et al.* 2006, Potts and Aebischer 1991, Southwood and Cross 1969).

The effects may be more long term than merely affecting birds in the nest, as poor nestling condition caused by low invertebrate abundance can have knock-on effects for fledgling condition, social status and thus lifetime reproductive success (Bradbury *et al.* 2000). Food availability may affect the energy expenditure of parent birds as they exert more effort in foraging further and exploiting less profitable prey if the preferred items are unavailable (Tinbergen & Dietz 1994; Thomas *et al.* 2001; Hart *et al.* 2006). This may result in reduced overwinter survival for both adults and young or reduced fecundity in future years (Brickle *et al.* 2000; Bradbury *et al.* 2003; Hart *et al.* 2006).

4.2.2 Foraging potential of biomass crops

Due to the difficulties of application and the fact that the crops can tolerate a weed layer, without compromising growth, biomass fields receive little in the way of pesticide inputs as it is rarely economically viable (Sage 1998). This leads to important ground vegetation developing in the crop which provides food and habitat for diverse invertebrate communities crucial in chick diet (Moreby & Stoate 2001; Dauber, Jones, & Stout 2010). SRC hosts a wide diversity of invertebrate species (Kennedy & Southwood 1984; Sage & Robertson 1994; Sage & Tucker 1997), and higher invertebrate abundance and species richness has been recorded for both SRC and Miscanthus compared with arable crops (Bellamy *et al.* 2009; Dauber, Jones, & Stout 2010). Miscanthus supports a more diverse invertebrate community than reed canary grass, another perennial grass used for energy production in the

U.S. (Semere & Slater 2007a), and uncultivated reed *Phragmites australis* (Dauber, Jones, & Stout 2010).

There is evidence, however, that the higher species richness in biomass crops thus far observed is due to their increased patchiness and higher proportion of ground vegetation in comparison to more intensively managed land uses (Bellamy *et al.* 2009). The concern for the future is that as the crops become established and are more intensively managed themselves, the potential benefits to farmland birds may be negated as the canopy closes over, and shades out weeds resulting in low invertebrate densities (Bellamy *et al.* 2009; Gove & Bradbury 2010). Whilst SRC canopy holds high abundances of invertebrates, regardless of the weed content (Sage & Tucker 1997; Rowe *et al.* 2011), few insects are found in the Miscanthus crop itself (Semere & Slater 2007a; Bellamy *et al.* 2009).

Another potential future limitation to the apparent biodiversity benefits afforded by biomass crops, is the employment of chemical pest control. SRC is already host to several pest species that act to defoliate the crop, mainly the Chrysomelid leaf beetles (Kendall *et al.* 1996; Peacock, Harris, & Powers 2004). Should these become a problem affecting yields, pesticides may be applied, which would affect the invertebrate population in the crop. Alternative pest control methods may be more effective, however, such as biological control or applying chemicals only to the crop edge, thus mitigating this risk (Dauber, Jones, & Stout 2010).

It is not merely the diversity and abundance of invertebrates in a habitat that makes it a suitable foraging patch for farmland birds. Even if invertebrate prey is provided in sufficient quantity and quality, it must be accessible to make the habitat suitable for foraging birds. Dense swards of grassland have been cited as reducing suitability of a foraging patch, due to reduced accessibility (Odderskaer *et al.* 1997; Devereux *et al.* 2004), thus raising questions over the foraging potential of established biomass crops. The invertebrates concerned must also be those important in chick diet, so any increase in invertebrate richness in biomass crops must be framed in this context. For example, in a study of the invertebrate composition of different land uses, only Collembola and Chilopoda were more abundant in Miscanthus than in wheat, but neither of these are important prey for birds (Bellamy *et al.* 2009).

Although investigations of the invertebrate composition of biomass crops have linked findings to the known prey preferences of farmland birds, no work has thus far been carried out examining the diet of birds foraging in these crops, apart from one small scale study (Sharples 1997). That study examined the use of SRC by a number of different birds, and showed that birds nesting in the crop did forage within it, but due to the sample sizes involved, no statistical analyses were carried out. Nor has there been any work directly comparing the invertebrate composition of SRC with that of Miscanthus, despite comparisons of each with other land uses (Anderson, Haskins, & Nelson 2004; Rowe, Street, & Taylor 2009; Rowe *et al.* 2011). Given that native willow species in Britain in their natural form support more invertebrates than any other British tree (Kennedy & Southwood 1984), it is likely that SRC, which has already been shown to hold a diverse insect community (Sage & Tucker 1997), supports a greater abundance and diversity of invertebrates than non-native Miscanthus. This hypothesis is further supported by the fact that it is the weeds within Miscanthus that have apparently contributed to the high invertebrate abundances reported for these fields (Semere & Slater 2007a; Bellamy *et al.* 2009), as opposed to both crop and non-crop areas of SRC fields (Sage 1995, 1998; Sage & Tucker 1997). The foraging potential of biomass crops is therefore poorly understood, as limited work has been carried out in these crops, and investigations that have been undertaken have been in weedy crops. The aim of this study was therefore to assess to what extent birds nesting around biomass crops use them to forage.

The birds used in this study were whitethroats (*Sylvia communis*) and yellowhammers (*Emberiza citrinella*), both of which are included amongst the 19 species making up the Farmland Bird Indicator (Defra 2014). As the species most commonly found nesting in the study sites, besides generalist species such as blackbird (*Turdus merula*) or dunnock (*Prunella modularis*), yellowhammers and whitethroats were chosen to ensure data could be collected in sufficient quantities (Table 7.3 in Appendix III).

Yellowhammers are red listed in the UK (Eaton *et al.* 2009), due to the fact they have suffered a widespread moderate decline since the 1980s, and in 2000, the population was reported to be declining at a rate of 10% per year (Bradbury *et al.*

2000). They are resident in the UK, mostly wintering within 5km of natal ground (Lack 1986). In contrast, whitethroats are a migratory species, spending the breeding season in the UK (Lack 1986), and are amber listed (Eaton *et al.* 2009) due to their shallow recovery in recent years after the population fell by 70% in the late 1960s (Fuller *et al.* 1995; Siriwardena *et al.* 1998).

Both species nest in field boundaries, in scrubby vegetation, ditches and in the case of yellowhammers, low down in hedges (Mason 1976; Bradbury *et al.* 2000; Stoate & Szczur 2001b). The presence of territories of both species increases with the abundance of herbaceous vegetation in boundaries, and the width of these strips (Bradbury *et al.* 2000; Stoate & Szczur 2001b; Stoate, Morris, & Wilson 2001), probably due in part to enhanced nest concealment afforded by this vegetation (Dennis, Thomas, & Sotherton 1994). Additionally, these perennial grasses and herbaceous plants often support high densities of invertebrates important in the diet of yellowhammer and whitethroat, mainly Lepidoptera, Coleoptera and Diptera (Moreby & Stoate 2001). This may also explain the yellowhammer's preference for foraging in uncultivated field margins when provisioning nestlings (Stoate, Moreby, & Szczur 1998).

The exact cause of yellowhammer decline is unknown, but survival rates and breeding productivity that are too low to maintain stable populations have been cited (Siriwardena, Baillie, & Wilson 1998; Bradbury *et al.* 2000), although declining populations have exhibited stronger breeding performances, indicating complex interactions at play (Siriwardena *et al.* 2000). Food supply is an important contributing factor to survival rates, as it can affect both the condition of adults entering the breeding season, and post-fledging survival, in addition to future reproductive success (Bradbury *et al.* 2000). Although the cause of the whitethroat crash in 1969 was a severe drought in its wintering grounds (Winstanley, Spencer, & Williamson 1974), agricultural practices in the UK that make food and nesting habitat more unpredictable may be playing a part in its slow recovery. Current population trends for both species may therefore be attributed to agricultural intensification, leading to loss of vital herbaceous vegetation, via herbicide drift, loss of margins, filling or clearing of ditches (Stoate, Moreby, & Szczur 1998; Bradbury *et al.* 2000;

Hinsley & Bellamy 2000; Stoate & Szczur 2001b; Stoate, Morris, & Wilson 2001; Perkins *et al.* 2002).

4.3 Aims

To determine the foraging potential of biomass crops for birds nesting around them, the foraging resource of the crop and the extent to which birds used this resource was examined, via the following questions:

- What invertebrates are available in Miscanthus and SRC willow crops?
- On what do whitethroats and yellowhammers nesting in boundaries of biomass crop fields feed?
- Does diet reflect availability in the crop?
- Does diet differ between crop types?
- Is there therefore evidence that birds nesting in biomass boundaries forage within the biomass, and does this vary with crop type?

Faecal samples were collected from chicks nesting in biomass crop field edges and compared with invertebrate sampling of the field. Observations of parent birds were also carried out to investigate crop use. The suggestion that weeds contribute a large part to the invertebrate community of biomass crops (particularly for Miscanthus) led to the decision that only areas of crop should be included in the invertebrate surveys. This ensures greater relevance of this study to future scenarios, where it is likely the weed layers present in the crops will be reduced.

4.4 Materials and Methods

4.4.1 Faecal samples

Collecting faecal samples from chicks in nests in field boundaries took place in all three years, but in 2012, nest survival was so low that only 9 out of 27 nests produced chicks old enough to be sampled. A subsection of biomass crop fields was chosen from the broader lapwing nest work (Chapter 2) that contained suitable nesting habitat for a range of birds, i.e. hedgerows, grass margins, ditches and

mature trees (Lack 1992). In each year, three sites per crop type were selected, each consisting of up to three fields, so that 12 fields of each crop type were used. Four of the Miscanthus fields were used in both years but the remaining fields differed between 2011 and 2013.

Sites were visited throughout May, June and July, and field boundaries searched. Nests were located either by observation of adults carrying food or nesting material, or by using the less time-consuming 'tapping' method (J. Szczur, pers. comm.), whereby the hedge and ground vegetation were tapped with a length of bamboo cane at regular intervals in order to flush sitting females. Once a bird had been flushed, the area was carefully searched until the nest found. Nests were found at egg and chick stages, and nest contents were checked using a mirror attached to the end of a pole, to minimise disturbance. Nests were then marked with natural objects such as dead vegetation, sticks and stones where possible (Stoate & Szczur 2001a), but sometimes bamboo canes or pieces of coloured tape were placed nearby to act as markers for particularly obscured nests. Each nest was entered into a GPS system, before taking care to restore any trampled vegetation so as not to attract predators to the nest.

Faecal samples were collected every two to three days from chicks aged between three and seven days old; approaching chicks any older than this would result in forced fledging (Brickle and Harper 1999). Handling the chicks usually resulted in a sample being produced immediately, otherwise chicks were returned to the nest after ten minutes. Faecal sacs from the same nest on the same day were placed in a 25ml universal tube, labelled with species, nest number, number of sacs in the sample, and date collected. These were then frozen for later processing and analysis which followed Moreby (1988). Defrosted samples were washed in a 180μm mesh sieve to remove the uric acid and fine organic matter, leaving only solid, undigested fragments. Samples were then rinsed in ethanol to remove the water, labelled and stored. Birds were handled under Natural England licence numbers 20113116 and 20130757, and nest searching followed the BTO Nest Recorder's code of conduct (BTO 2003).

Field size and boundary score were also measured to determine the effect of these on invertebrate availability. Boundary indices were calculated for each field to

account for variation in habitat features, which could in turn affect invertebrate availability and accessibility. Boundary features were allocated a score, an extension of the system developed by (Poulsen, Sotherton, & Aebischer 1998) (Table 4.1) and these scores were multiplied by the length of each boundary type to obtain a boundary index for that field. Each boundary index was then divided by the area of the field to account for variation in field size.

4.4.2 Invertebrate sampling in the crop

Invertebrate abundance in each biomass field was sampled within two weeks of faecal sampling, using a sweep net. A transect running the diagonal length of the field was walked, and samples taken at ten randomly selected points. At each point, five sweeps of 180° were made over the crop, and the samples frozen for later analysis. In processing the samples, a small paintbrush was used to remove any vegetation, before storing them in 70% ethanol.

Table 4.1 Boundary index scoring system

Feature	Score
Adjacent crop, track, short grassy area	0
Tall grass, scrubby area, set-aside	1
Tall thin hedgerow, tall hedgerow with many gaps	2
Tall dense hedgerow, hedgerow with occasional mature trees	3
Tall dense hedgerow, hedgerow with many mature trees	4
Woodland	5

4.4.3 Invertebrate identification

Cleaned, defrosted crop and faecal samples were examined under a binocular microscope at 6-50x magnification on a Petri dish with a 1cm^2 grid on the bottom. Invertebrates from faecal and sweep samples were identified to order or family level, according to Ralph (1985), Moreby (1988) and Chinery (1993). Characteristic body parts that remain undigested could be identified from faecal samples and were used to quantify the minimum number of individuals contained in the sample by dividing the number found by the number of body parts per individual (Moreby 1988).

4.4.4 Foraging observations

In 2013, foraging trips made by breeding whitethroats and yellowhammers were observed, when chicks were aged at least 6 days old. Observations were carried out from a car located at least 20m from the nest, with a good view towards the nest and surrounding vegetation. Adults were observed leaving the nest and the foraging destination recorded, categorised as either within the biomass crop or any other destination. Where destination could not be determined, the foraging trip was discarded from the analysis. Observations at each nest were carried out for an hour, between 0700 and 1200, but were not conducted in periods of heavy rain or strong winds.

4.4.5 Statistical analysis

To determine the factors affecting the abundance and diversity of invertebrates found in sweep samples, the total number of invertebrates found per field and the number of orders represented in each field were used as the response variables. Crop type, year, Julian date, field size (log-transformed), and boundary-score (square-root) were the explanatory variables tested within a mixed-effects model structure with Poisson errors, with field and site as random effects to account for repeated visits. If the variation attributed to field or site was negligible, these effects were omitted in the interests of model simplification, and the models were run as simple generalised linear models. To avoid over-parameterisation of the models, variables were added to the model one by one, to reach a maximal model and the significance of each was tested by a process of stepwise deletion. Terms were deleted and subsequent models compared via ANOVAs; only those variables whose removal resulted in significantly different models ($p<0.05$) were retained. Terms were removed in this way until the minimum adequate model was reached (Crawley 2007).

To examine the invertebrate composition of sweep samples, invertebrates found were reduced to seven groups; the six most abundant Arachnid, Diptera, Larvae (all larvae of Coleoptera, Lepidoptera and Symphyta), Hymenoptera, Hemiptera, Coleoptera, and 'Other" containing Odonata, Gastropoda, Orthoptera, Dermaptera, Collembola, Thysanoptera, Psocoptera. Instead of single proportions for each invertebrate group, a vector of the number of invertebrates found in each of

these groups together with the remaining total of all other invertebrates found was produced. This resulted in seven response variables, each of which was tested within a generalised linear model with binomial error structure (Crawley 2007). This method was followed rather than compositional analysis of all groups together in a multivariate analysis (Aebischer, Robertson, & Kenward 1993) for ease of interpretation. The same explanatory variables that were used to test invertebrate abundance and diversity were used to test composition; additionally the number of invertebrates (log-transformed) and groups found in sweeps were also tested. The same method of stepwise deletion was followed as above until the minimum adequate model was reached. To further examine the invertebrate composition of sweep samples, principle component analysis (PCA) was carried out. By plotting the seven invertebrate groups in ordination space, visualisation of the invertebrate community can be achieved, by identifying the components that explain the most variation in the data. To test whether the invertebrate communities differed according to crop type, an analysis of similarity (anosim) was conducted.

Analysis of faecal samples followed the same methods used to analyse sweep samples. Faecal sacs were pooled for each brood, and the total number of invertebrates averaged per sac to provide a measure of invertebrate abundance. The abundance and diversity (measured as the number of orders represented in all faecal sacs) of invertebrates found in faeces were then tested as above. The number of faecal sacs collected from each brood was included as a covariate. For invertebrate diversity, the number of orders represented in sweep samples was also included as an explanatory variable. Mixed-effects models were used with site and field as random effects to account for repeated visits, and Poisson errors.

Faecal composition was analysed in the same way as sweep composition, using the same seven invertebrate groups. In addition to the explanatory variables used in the models explaining invertebrate abundance and diversity in faecal samples, the number of invertebrates (log-transformed) and groups found in faeces, the number of invertebrates (log-transformed), and groups found in sweep samples and the proportion of the group being analysed found in sweep samples (arc-sine transformed) were also incorporated in the models. Just as with the sweep samples,

the invertebrate composition of faecal samples was further examined using PCA and anosim, to determine whether the composition differed according to crop type.

Finally, the difference in composition between field and faecal samples was analysed using GLMs with binomial errors. Sampling technique was used as a two level explanatory variable (sweep or faecal). All statistical analyses were performed in R (R Development Core Team 2011).

4.5 Results

4.5.1 Invertebrate composition of sweep samples.

More invertebrates were collected from SRC fields than Miscanthus ($z=7.55$, $p<0.0001$; Table 4.2) and bigger fields yielded a greater abundance of invertebrates ($z=1.97$, $p=0.04$; Figure 4.1). Crop type did not predict invertebrate diversity (removal from model results in $p=0.59$), and yet the crop type affected abundance, which in turn predicts diversity ($z=2.66$, $p=0.008$; Figure 4.2).

Table 4.2 Abundance and diversity of invertebrates found in sweep samples taken from Miscanthus and SRC fields.

Crop	Fields	Mean no. invertebrates per ha	Mean no. of orders per ha
Miscanthus	12	8.44 ± 0.54	1.31 ± 0.09
SRC	12	20.44 ± 2.33	1.39 ± 0.13

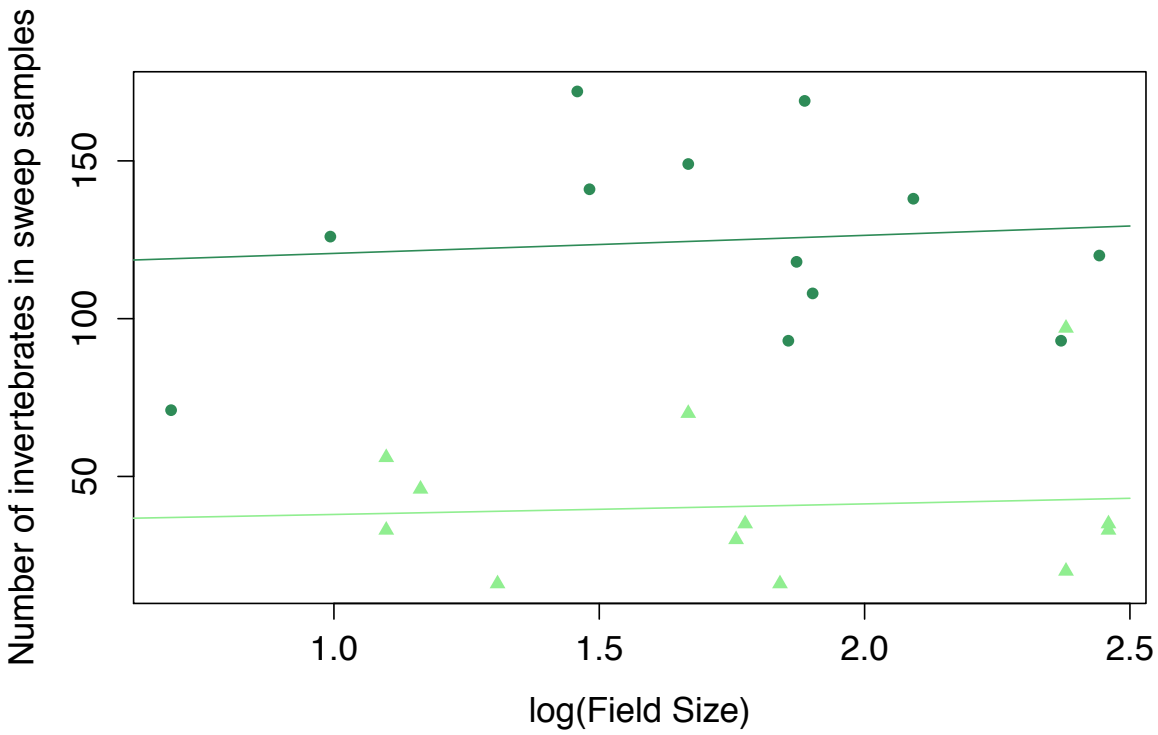


Figure 4.1 Invertebrate abundance in sweep samples in Miscanthus (light green triangles) and SRC (dark green circles) fields. The relationship with field size was significant (z=1.97, p=0.04) as was the difference between crop types (z=7.55, p=0.0001).

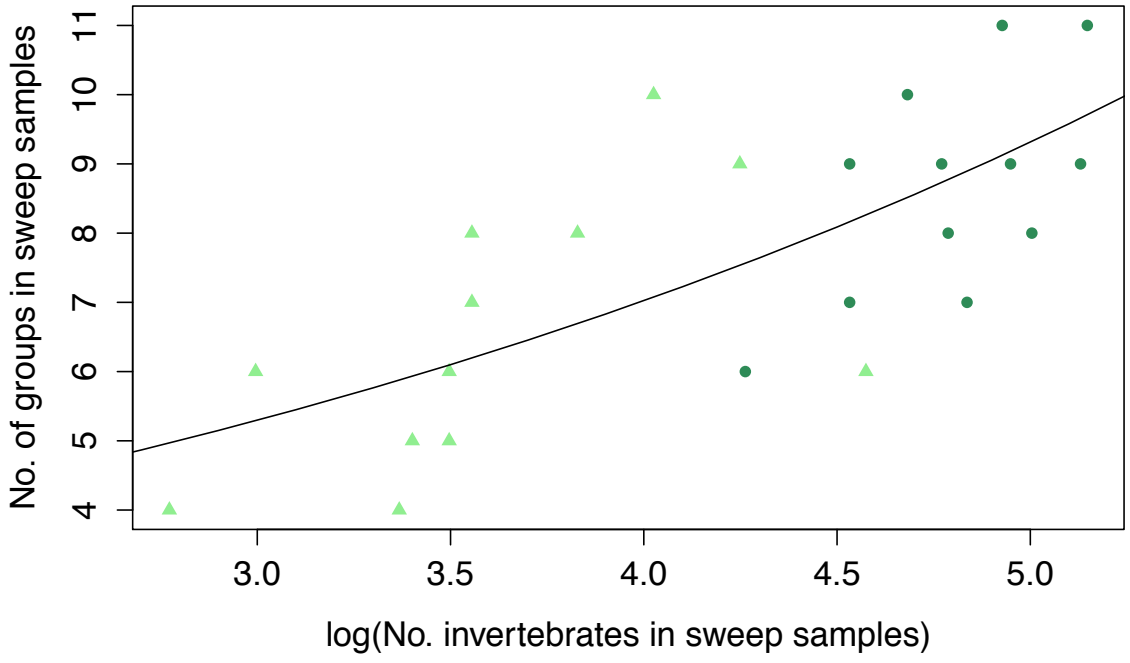


Figure 4.2 Invertebrate abundance and diversity in sweep samples in Miscanthus (light green triangles) and SRC (dark green circles) fields. Abundance increased with diversity (z=2.66, p=0.008).

Table 4.3 The effects of explanatory variables on the proportions of invertebrate orders found in sweep samples.

Order	SRC (effect compared with Miscanthus)	Field abundance	Field diversity	Field Size
Arachnid			+	
Diptera	-			
Larvae				
Hymenoptera	-	++		--
Hemiptera		-		
Coleoptera	+++			
Other				

Signs indicate size of the effect of removing explanatory variable from the model.
-/+ $p<0.05$; --/++ $p<0.01$; ---/+++ $p<0.001$

Further analysis of the composition of sweep samples (Table 4.3) reveals a higher proportion of Diptera and Hymenoptera in Miscanthus ($z=-2.36$, $p=0.02$ and $z=-2.14$, $p=0.03$), but a higher proportion of Coleoptera in SRC ($z=4.54$, $p=0.0001$). For the dataset as a whole, including both crop types, the proportion of Hymenoptera in sweep samples was positively correlated with the number of invertebrates found in sweep samples ($z=2.64$, $p=0.008$), while the reverse was true of Hemiptera ($z=-2.42$, $p=0.03$). Only Arachnids were affected by the diversity of invertebrates found in sweep samples, with proportionally more found in those samples containing a greater range of orders ($z=1.98$, $p=0.04$).

Under Principal Components Analysis, PC1 explained 45.5% of the variation in the invertebrate community found in sweep samples, PC2 explained 24.8%. Hymenoptera were positively correlated with PC1, while Diptera and "Other" invertebrates were negatively associated with PC1. Diptera were also negatively associated with PC2, while "Other" invertebrates were positively associated with PC2 (Figure 4.3). The invertebrate community of field samples varied with crop type, $R=0.19$, $p=0.008$ (Figure 4.4).

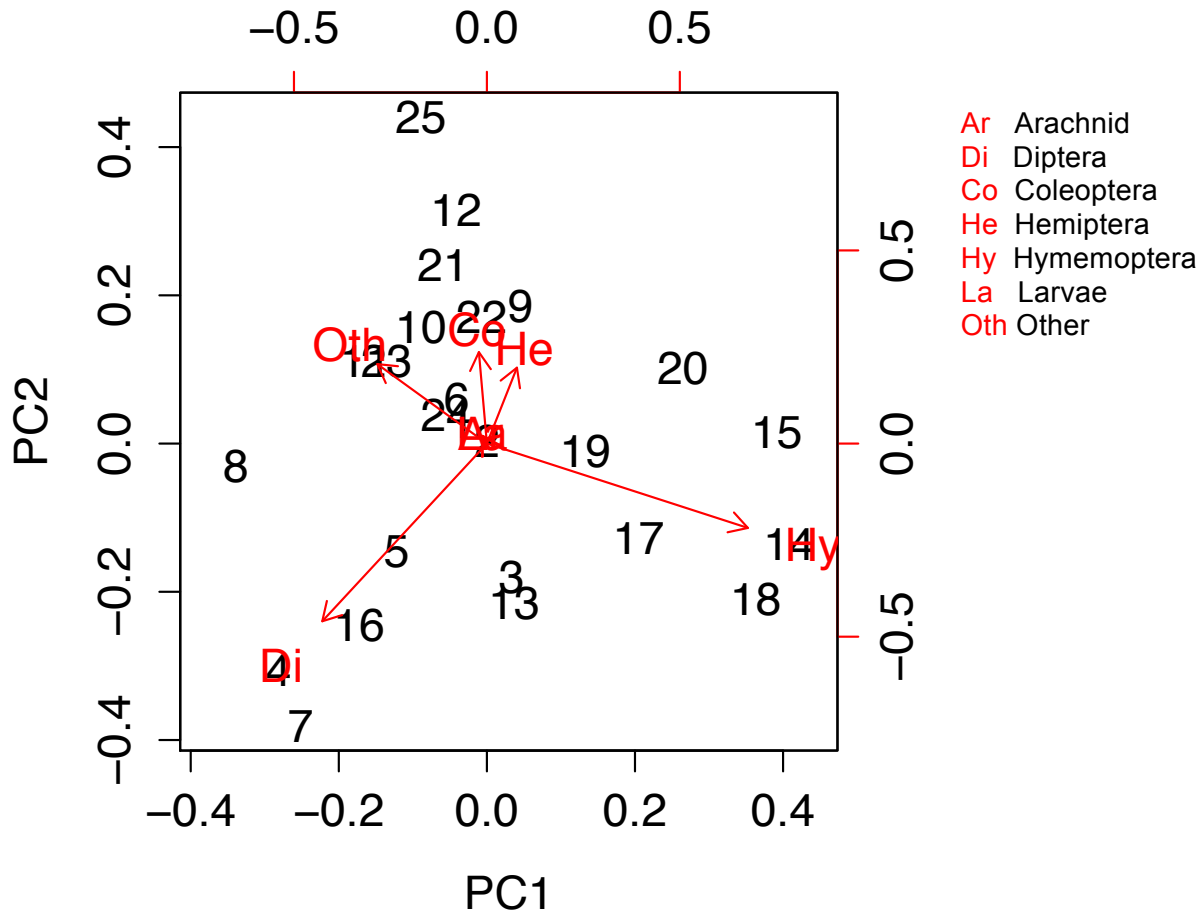


Figure 4.3 PCA biplot of invertebrates found in sweep samples taken from Miscanthus and SRC fields.

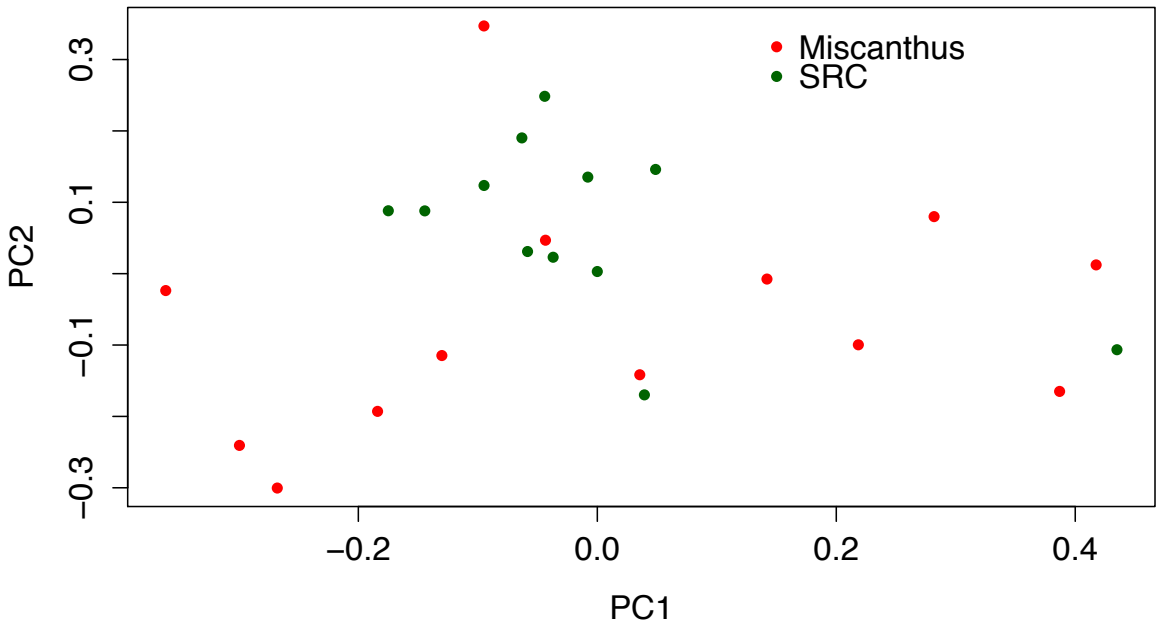


Figure 4.4 Scatterplot of the first two principle components from the PCA of the invertebrate community found in sweep samples.

4.5.2 Invertebrate composition of faecal samples

119 faecal sacs were collected from 52 whitethroat and yellowhammer nests over two years (Table 4.4). A minimum of 1097 invertebrates was identified from the fragments contained in these samples.

Table 4.4 Summary of invertebrate totals in whitethroat and yellowhammer faeces.

		Total faecal sacs collected	Total invertebrates	Mean no. of faecal sacs collected	Mean Invertebrates per sac	Mean orders per brood
WH	Miscanthus (16)	37	258	2.31	7.01 ± 0.86	5.63 ± 0.38
	SRC (8)	19	119	2.38	7.13 ± 1.30	6.25 ± 0.68
YH	Miscanthus (15)	34	245	2.27	7.83 ± 1.24	5.40 ± 0.59
	SRC (13)	29	475	2.23	14.75 ± 2.65	5.69 ± 0.50

4.5.3 Invertebrate abundance and diversity of whitethroat faeces

Faeces collected from whitethroat nests bordering SRC contained more invertebrates than those in Miscanthus field margins ($t=2.16$, $p=0.04$). An interaction term between crop type and the number of faecal sacs collected per brood was also retained in the model ($t=-3.38$, $p=0.003$). The mean number of invertebrates in whitethroat faeces increased with the number of invertebrates found in sweep samples ($t=2.17$, $p=0.04$; Figure 4.5), although when analysing crop types separately, this effect was only true for SRC ($t=4.08$, $p=0.01$ cf. $t=1.62$, $p=0.13$ in Miscanthus). Year was also retained in the model, with a greater number of invertebrates found in faeces in 2013 ($t=5.45$, $p<0.0001$), which was true of both SRC and Miscanthus ($p=0.02$ and $p=0.006$ respectively).

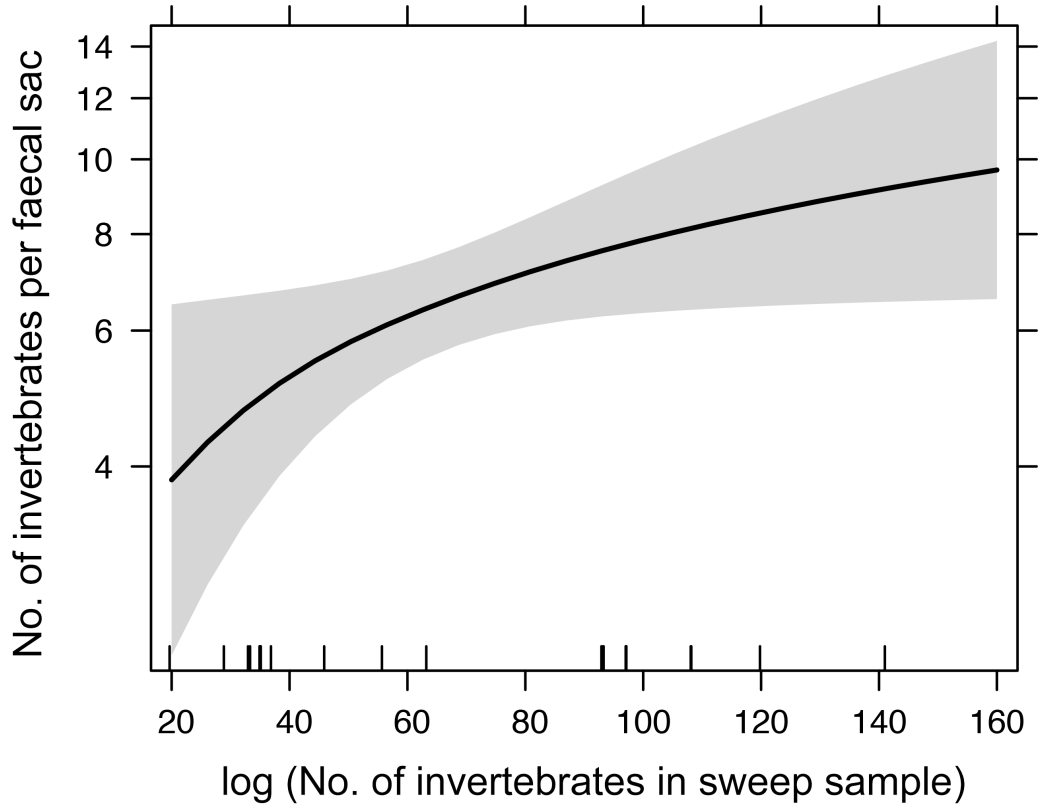


Figure 4.5 Number of invertebrates found in whitethroat faeces and sweep samples for both biomass crops combined. Fitted values are shown bounded by 95% confidence intervals. The relationship was significant (t=2.17, p=0.04).

There was no difference in invertebrate diversity of whitethroat faeces between crop types (removal from model p=0.69), but there was a positive relationship between the number of orders contained in whitethroat faeces and the number of orders obtained from sweep samples (t=2.93, p=0.008; Figure 4.6). Again, when analysing crop types separately this relationship was only apparent in faeces collected from SRC (t=4.47, p=0.003) and not Miscanthus (t=1.72, p=0.11).

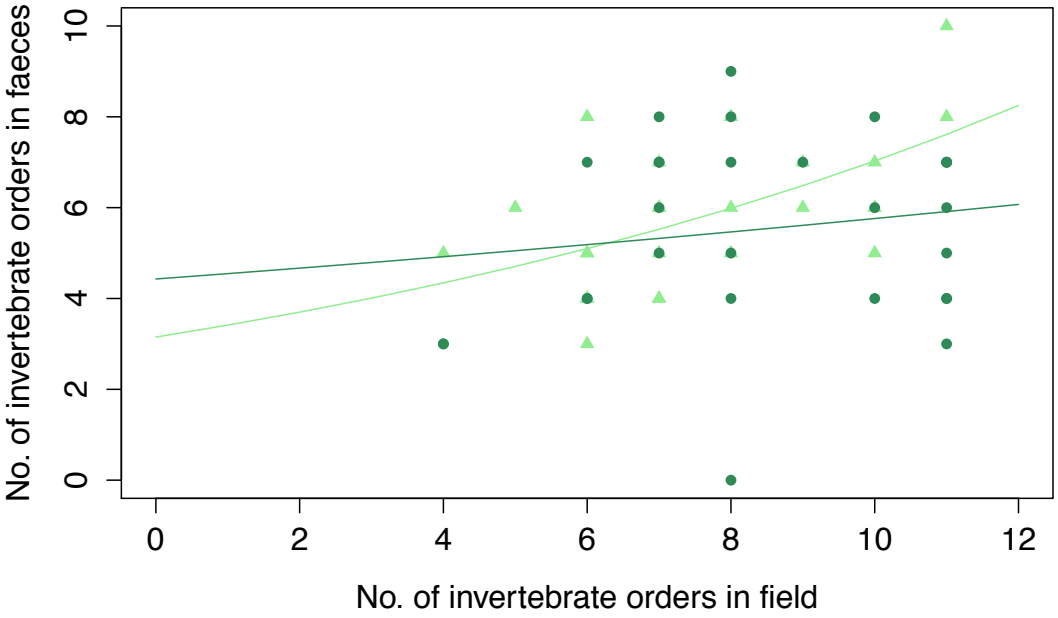


Figure 4.6. Invertebrate diversity in faecal and sweep samples, for whitethroat (light green triangles) and yellowhammer (dark green circles). The relationship between diversity in faecal and sweep samples was significant for whitethroat (t=2.93, p=0.008) but not for yellowhammer (t=1.86, p=0.08).

4.5.4 Invertebrate abundance and diversity of yellowhammer faeces

Crop type had no effect on invertebrate abundance in yellowhammer faeces (z=-1.47, p=0.14). Again there was an interaction between crop type and the number of faecal sacs collected (p=0.006). Invertebrate abundance decreased with date, so that faecal sacs collected later in the season contained fewer invertebrates than those collected earlier (z=-3.48, p=0.0005; Figure 4.7). The only variable retained in the model explaining the number of invertebrate orders found in faeces was Julian date (p=0.005), but in contrast to the relationship with invertebrate abundance, diversity increased with time (t=2.18 p=0.04; Figure 4.8).

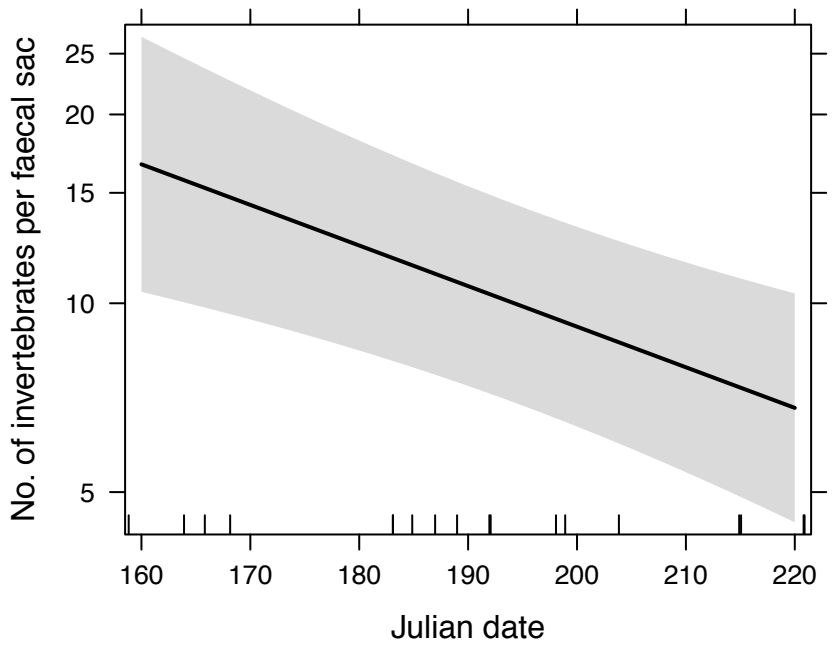


Figure 4.7 Invertebrate abundance in yellowhammer faeces over time. Fitted values are shown bounded by 95% confidence intervals. The relationship was significant (z=-3.48, p=0.0005).

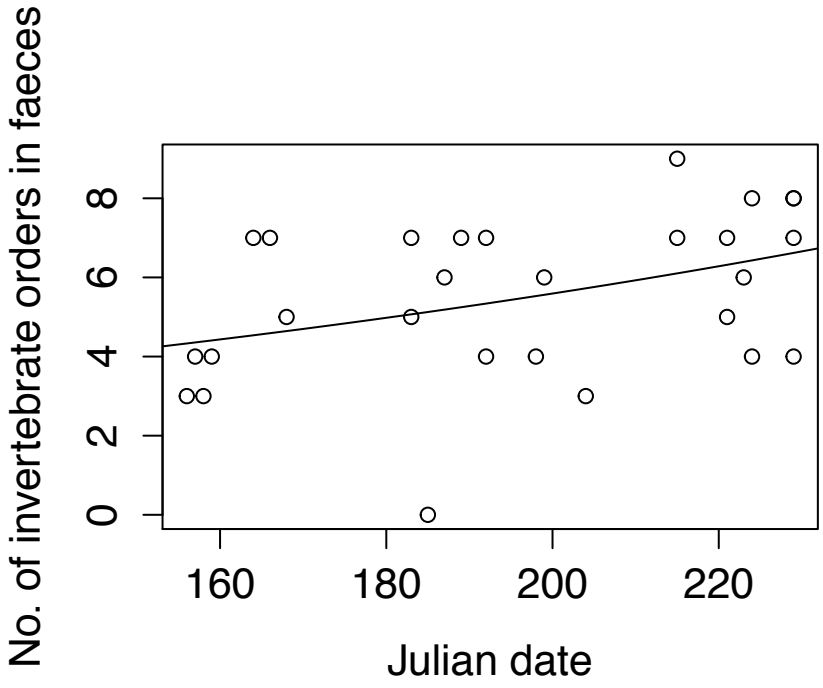


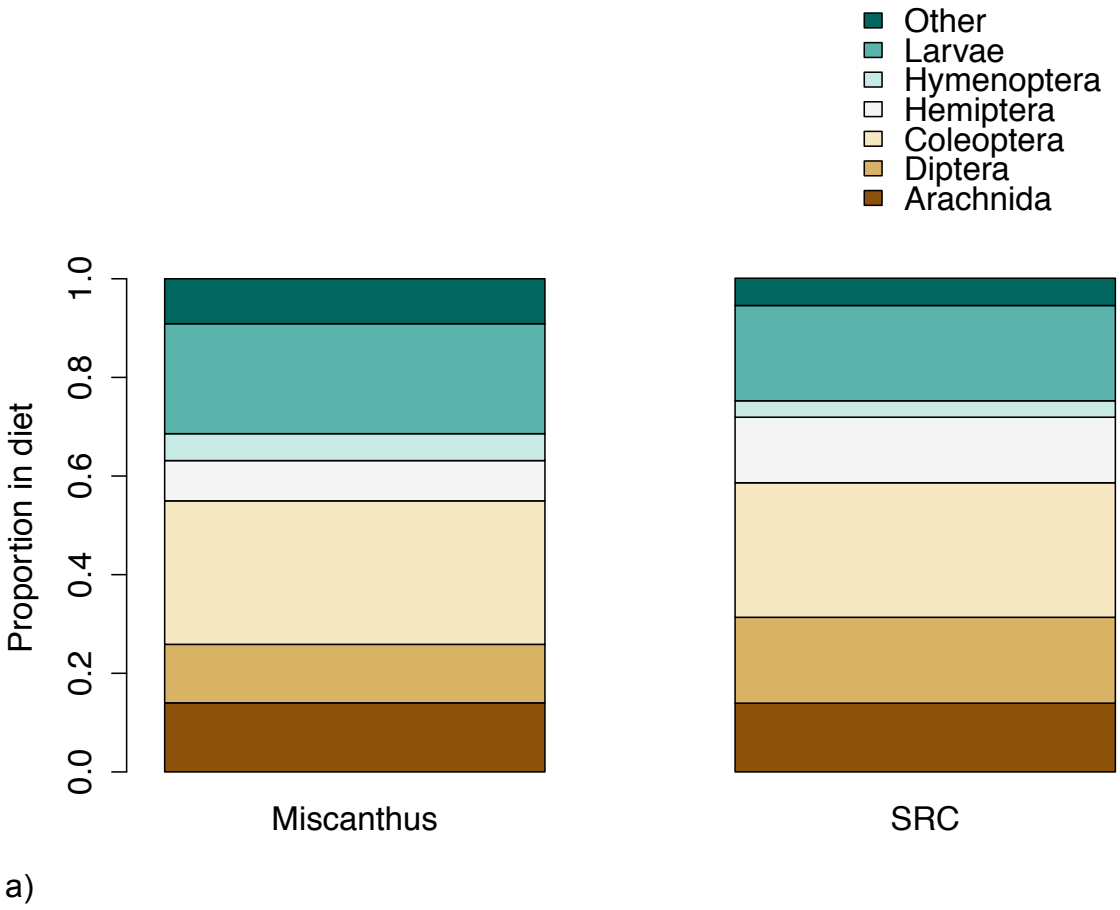
Figure 4.8 Number of invertebrate orders in yellowhammer faeces over time. The relationship was significant (t=2.18, p=0.04).

4.5.5 Invertebrate composition of whitethroat and yellowhammer faeces.

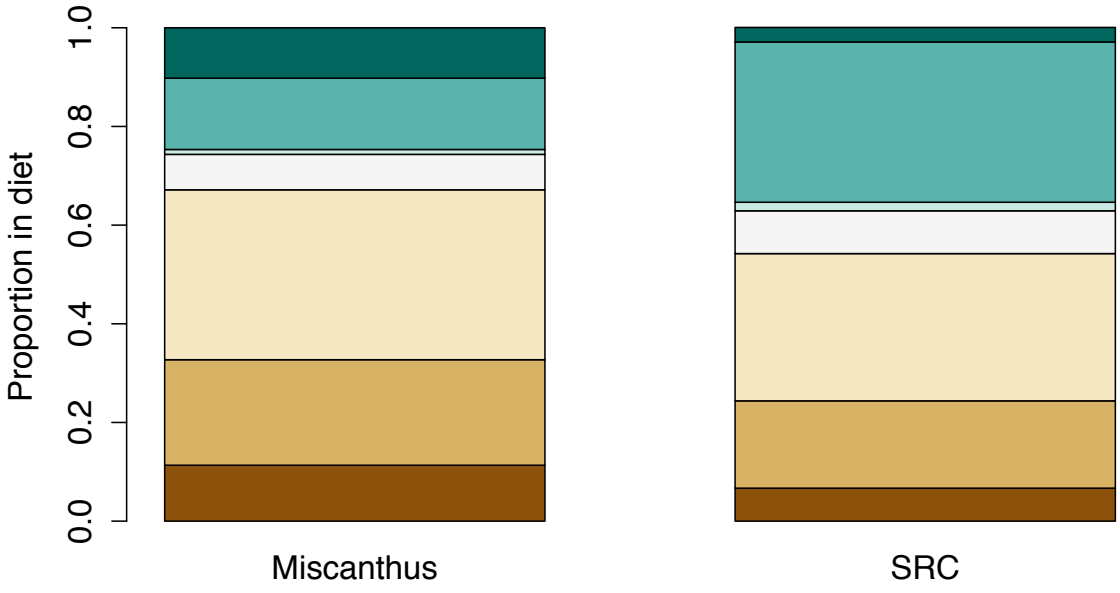
The composition of whitethroat and yellowhammer faeces is summarised in Figure 4.9, where the mean proportions of each order per brood are represented. For comparison with sweep samples, see Figure 4.14, and for further details including a summary of the overall proportions of total invertebrates found for each species, and the percentage of broods containing each order, see Appendix IV. The most abundant orders by proportion were Diptera, Larvae, Coleoptera and Arachnida, but the ranking of these top four orders varied between species and crop. Yellowhammer broods from SRC fields were the only faeces in which Diptera was replaced as the most abundant invertebrate order by Larvae. Odonata, Gastropoda and Dermaptera were not found in any faecal samples taken from nests bordering Miscanthus fields, but they did appear in whitethroat and yellowhammer faeces from SRC borders (with the exception of Gastropods and Odonata, which did not appear in any whitethroat faeces).

The effect of each explanatory variable on the proportion of each invertebrate order is summarised in Table 4.5. The size and direction of these effects varied between bird species and invertebrate orders. Proportionally more Hemiptera were found in whitethroat faeces collected from SRC than Miscanthus ($t=3.15$, $p=0.005$). Yellowhammer chicks in SRC boundaries produced faeces with a greater proportion of Larvae ($t=2.65$, $p=0.01$) but smaller proportions of Arachnids ($t=-3.00$, $p=0.006$), Coleoptera ($t=-4.55$, $p=0.0002$) and “Other” invertebrates ($t=-5.88$, $p<0.0001$) than those nesting in the boundaries of Miscanthus fields.

The abundance and diversity of invertebrates in faeces affected the proportional abundance of some orders. A higher number of invertebrates in faeces (invertebrate abundance) was associated with increased proportions of larvae in both whitethroat ($t=3.05$, $p=0.006$) and yellowhammer samples ($t=2.72$, $p=0.01$), and Diptera in whitethroat samples ($t=2.29$, $p=0.03$). The proportion of invertebrates falling into the “Other” category decreased in yellowhammer samples as the total number of invertebrates in faeces increased ($t=-3.19$, $p=0.004$). A greater proportion of Hymenoptera and “Other” invertebrates were found in yellowhammer faeces containing a greater diversity of invertebrates ($t=4.00$, $p=0.0005$; and $t=5.38$, $p<0.0001$), while fewer larvae were found in more diverse whitethroat faeces ($t=-2.72$



a)



b)

Figure 4.9 The composition of whitethroat (a) and yellowhammer (b) faeces collected from Miscanthus and SRC fields

Table 4.5 Summary of the effects of different explanatory variables on the proportions of invertebrates found in faeces.

Order	Whitethroat								
	SRC (effect compared with Miscanthus)	Field abundance	Field diversity	Faecal abundance	Faecal diversity	Field Size	Boundary Score	Date	Proportion of order in field
Arachnid									
Diptera				+					+
Larvae				++	--				-
Hymenoptera									
Hemiptera	++	-						+	
Coleoptera		+++					-	---	
Other		+++						+++	
	Yellowhammer								
Arachnid	--	++	--						
Diptera									
Larvae	++			++					
Hymenoptera					+++				---
Hemiptera						++		+	
Coleoptera	---	+	---			-			++
Other	---	+++		--	+++				

Signs indicate size of the effect of removing explanatory variable from the model, -/+ $p<0.05$; --/++ $p<0.01$; ---/+++ $p<0.001$

p=0.01).

The abundance and diversity of invertebrates found in sweep samples also affected the proportions of some orders found in faeces. Fields with higher invertebrate abundance yielded whitethroat faeces with proportionally more Coleoptera (t=3.23, p=0.004) and fewer Hemiptera (t=-2.84, p=0.01), and more Arachnids (t=2.66, p=0.05), Coleoptera (t=2.18, p=0.04) and “Other” invertebrates in yellowhammer faeces (t=4.63, p=0.0001). Fields with a greater diversity of invertebrate orders were associated with lower proportions of Arachnids (t=-2.99, p=0.007) and Coleoptera (t=-3.70, p=0.001) in yellowhammer faeces with no associated change in the proportions of any invertebrate order found in whitethroat faeces.

Composition of the invertebrate community found in whitethroat and yellowhammer faeces was examined using Principal Components Analysis. For whitethroats, PC1 explained 38.4% of the variation in whitethroat faecal composition, and 24.8% of the variance was explained by PC2. Diptera were positively associated with PC1. Coleoptera and “Other” invertebrates were positively associated with PC2, while Arachnids and Larvae were negatively associated with PC2 (Figure 4.10). The invertebrate community making up whitethroat faeces did not differ by crop type R=-0.11, p=1 (Figure 4.11).

For yellowhammer chick diet, PC1 explained 42% of the variation in yellowhammer faecal community, PC2 explained 34.4%. Diptera were negatively associated with PC1, and Coleoptera were negatively associated with PC2. Larvae were positively associated with both PC1 and PC2 (Figure 4.12). The invertebrate community of yellowhammer faeces did not differ with crop type, R=0.1, p=1 (Figure 4.13).

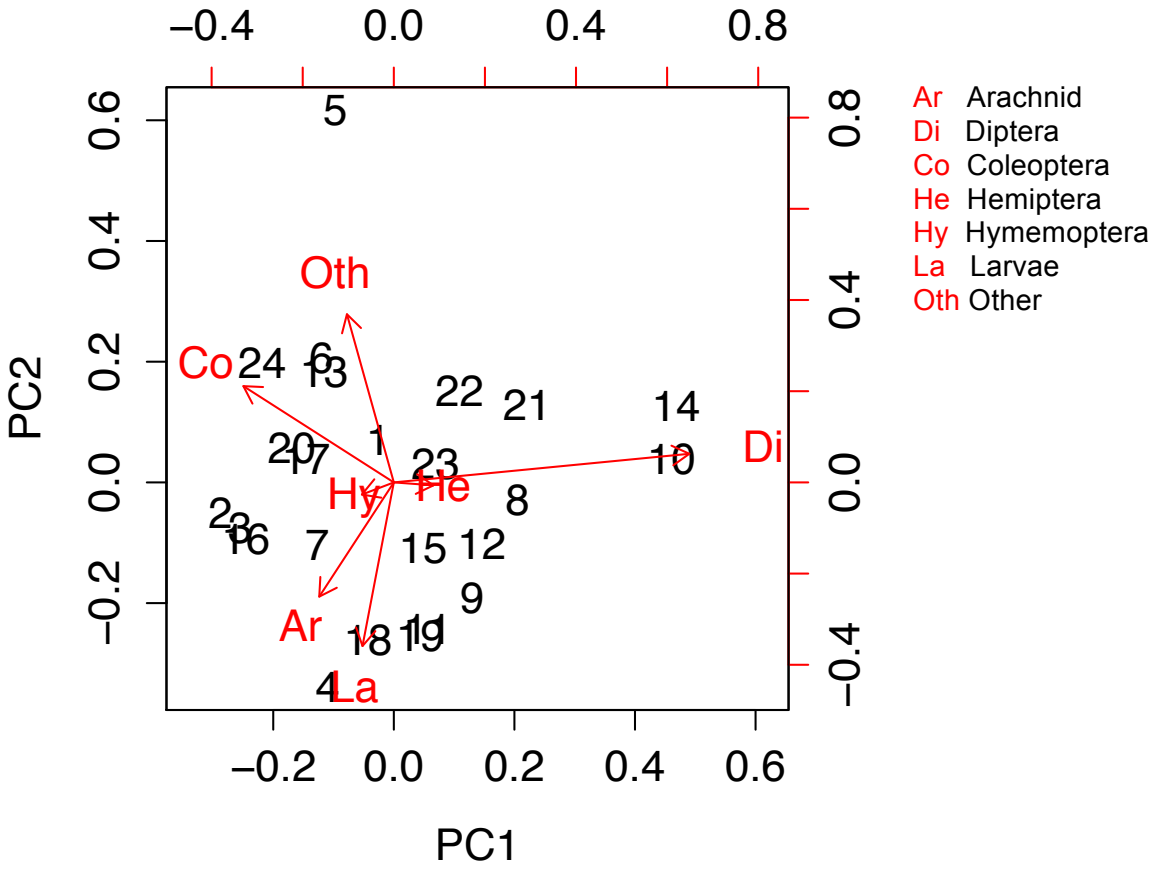


Figure 4.10 PCA biplot of invertebrates found in whitethroat chick faeces collected from boundaries of biomass fields

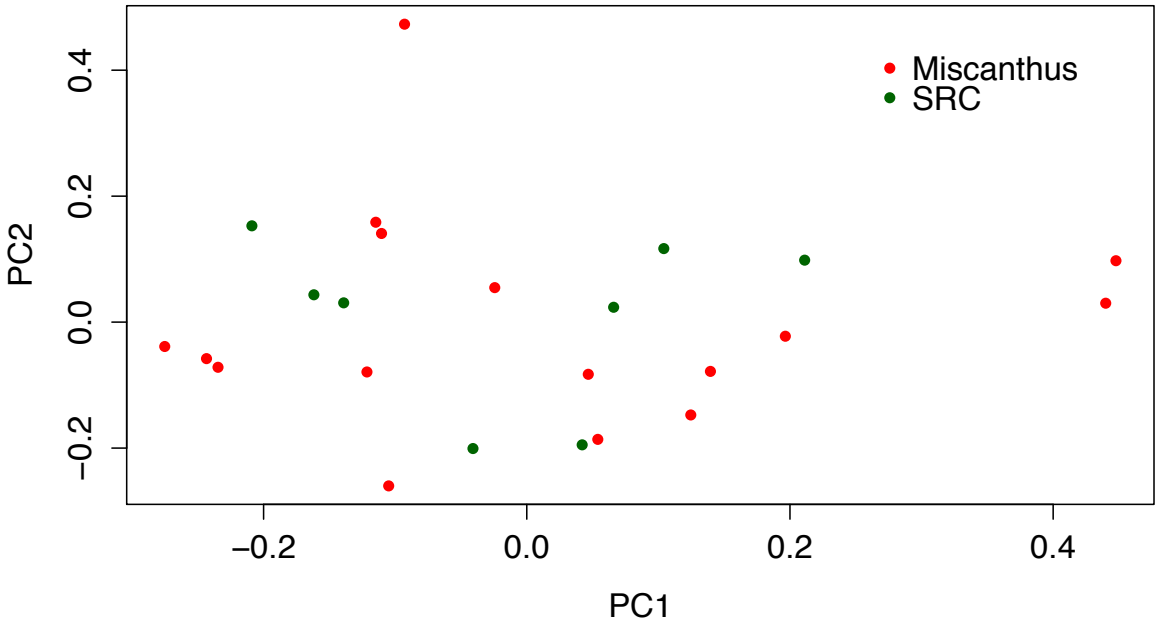


Figure 4.11 Scatterplot of the first two principle components from the PCA of the invertebrate community of whitethroat chick diet in Miscanthus and SRC

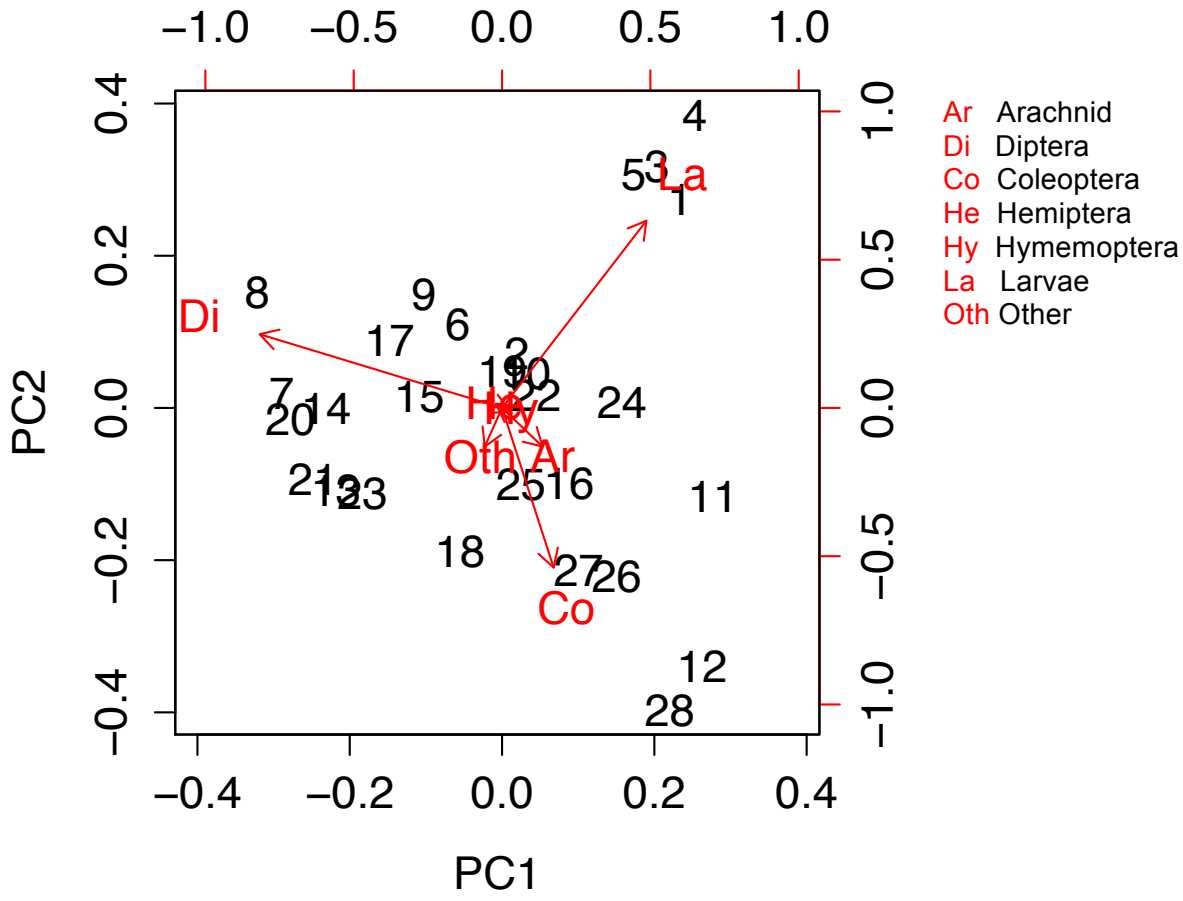


Figure 4.12 PCA biplot of invertebrates found in yellowhammer chick faeces collected from boundaries of biomass fields

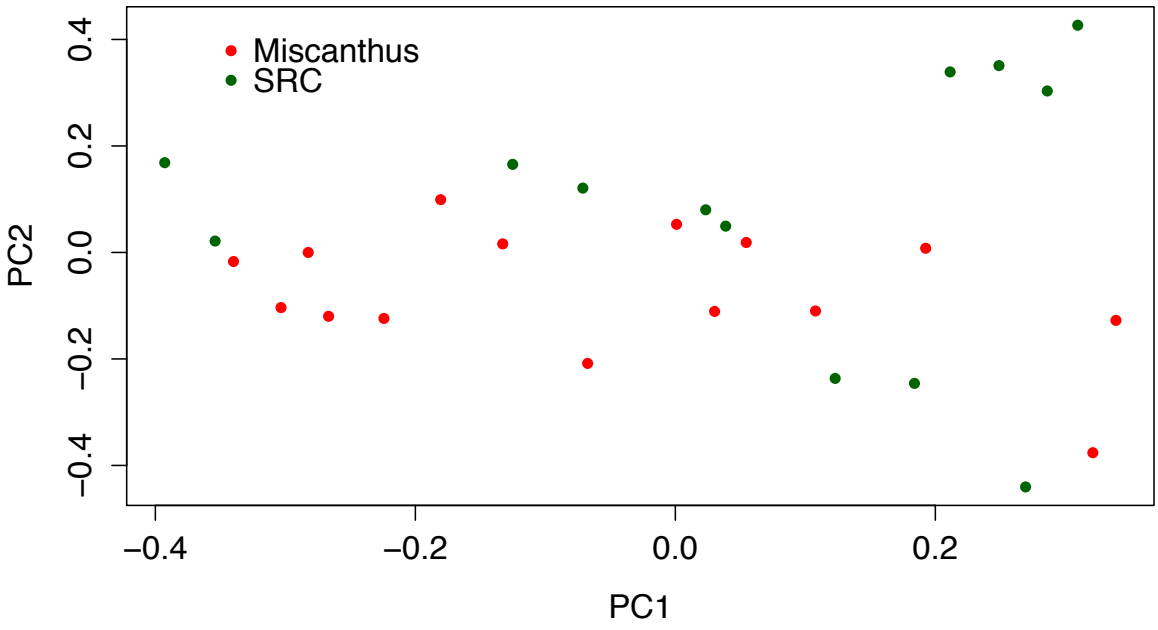


Figure 4.13 Scatterplot of the first two principle components from the PCA of the invertebrate community of yellowhammer chick diet in Miscanthus and SRC

4.5.6 Comparing invertebrate composition in faeces and field samples

For birds nesting near Miscanthus, Arachnids, Coleoptera and Larvae were found in higher proportions in faeces than in the crop for both yellowhammer (Arachnid: t=2.96, p=0.01; Coleoptera: t=5.12, p=0.0003; Larvae: t=3.41, p=0.004) and whitethroat (Arachnid: t=4.57, p=0.0001; Coleoptera: t=2.72, p=0.01; Larvae: t=4.70, p<0.0001; (Figure 4.14). Conversely, proportionally fewer Hymenoptera were found in Yellowhammer faecal samples than sweep samples (t=2.56, p=0.04), but this difference was not apparent in whitethroat faeces. No difference was found in the proportions of any other invertebrate order between faeces and field samples for birds nesting near Miscanthus.

For birds nesting near SRC, Arachnids and Larvae were found in higher proportions in faeces than in the crop for both yellowhammer (Arachnid: t=2.47, p=0.02; Larvae: t=7.68, p<0.0001) and whitethroat (Arachnid: 2.57, p=0.02; Larvae: 7.40, p<0.0001). Unlike the faeces of birds nesting near Miscanthus, the faeces of birds nesting near SRC did not differ from sweep samples in the proportion of Coleoptera making up the samples (t=-1.13, p=0.27 and t=0.1, p=0.92 and for yellowhammer and whitethroat respectively). As with yellowhammers nesting near Miscanthus, proportionally fewer Hymenoptera were found in faeces collected from yellowhammers nesting near SRC than in the crop itself (t=-2.34, p=0.03). Additionally, yellowhammer faeces collected from nests in SRC boundaries contained a lower proportion of those invertebrates falling into the “Other” category, than found in the crop (t=-3.18, p=0.004).

4.5.7 Foraging observations

Observations of foraging adults were carried out for 15 nests, in six Miscanthus fields and two SRC fields (Table 4.6). Although nest monitoring was carried out in 2012, the majority of these failed before chicks hatched or were old enough to be included in the observations. The nests in 2013 comprised eight whitethroat (four in Miscanthus, four in SRC), four yellowhammer (all in Miscanthus), and three reed bunting (two in Miscanthus, one in SRC). Given the small sample sizes involved, no statistical analyses of the observations were carried out, but for nests found in Miscanthus boundaries, on average 8% of foraging trips were made to the

Miscanthus, while 34% of foraging trips made by adults nesting in SRC boundaries had the SRC crop as their destination.

Table 4.6 Foraging observations of birds nesting in the boundaries of Miscanthus and SRC fields.

Crop	Nests	Total Visits observed	Total visits to Biomass observed	Mean proportion of visits to biomass per observation period
Miscanthus	10	158	12	0.08 ± 0.03
SRC	5	62	24	0.34 ± 0.15

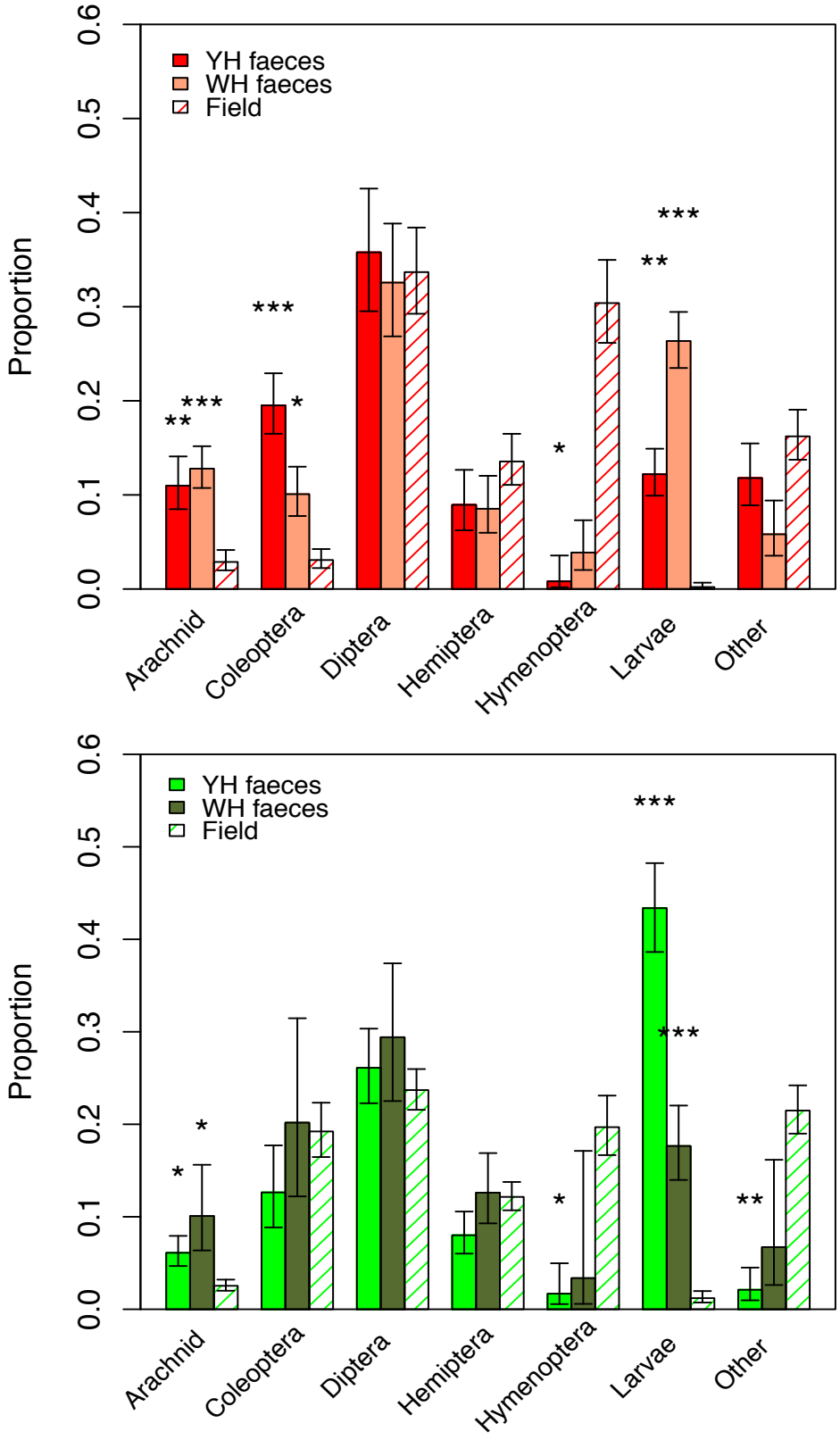


Figure 4.14 Invertebrate composition of field and faecal samples of yellowhammer (YH) and whitethroat (WH) in Miscanthus (red) and SRC (green) with standard errors. Significant differences between faecal and sweep samples are indicated by * p<0.05, ** p<0.01, ***p<0.001.

4.6 Discussion

More invertebrates were found in the faeces of whitethroat chicks nesting in the margins of SRC willow than those nesting in the margins of Miscanthus. This reflects availability in the crop surrounding them, as SRC fields contained a higher abundance of invertebrates. This is supported by the correlation between the number of invertebrates found in whitethroat faeces and the number of invertebrates found in sweep samples. The diversity of invertebrates found in whitethroat faeces was positively linked to invertebrate diversity of sweep samples, but there was no direct evidence of the diversity in the field or faeces varying with crop type.

4.6.1 Field composition

This is the first study to compare invertebrate composition of these two biomass crops. Previous work has suggested that while SRC supports a rich invertebrate community (Sage & Tucker 1997; Cunningham *et al.* 2006; Clapham 2011) it is only the weed element of Miscanthus fields rather than the crop itself that contributes an invertebrate resource for birds to exploit in the breeding season (Semere & Slater 2007a; Bellamy *et al.* 2009). As a non-native species, Miscanthus was predicted to support a less abundant and diverse invertebrate community than native crops, as has been shown in comparisons with native reed canary grass (Semere & Slater 2007a; Clapham 2011). The lower invertebrate abundance in Miscanthus compared with SRC observed here is in keeping with this prediction, although no such pattern in diversity was found. On average, a greater number of invertebrate orders was found in SRC fields, but this difference was not significant. The inclusion of more fields in future studies, or the calculation of a diversity index that takes into account the number of appearances of each order could perhaps further illuminate this (Clapham 2011).

In terms of the composition of sweep samples, Miscanthus samples contained proportionally fewer Coleoptera than SRC. The main pests of SRC are the leaf beetles in the Chrysomelidae family of Coleoptera, and as such, Coleoptera have been found to make up the majority of sweep samples from SRC in previous work (Sage & Tucker 1997; Sharples 1997; Cunningham *et al.* 2006). Hemiptera are also reported to be abundant in SRC (Sharples 1997; Cunningham *et al.* 2004), which

was not corroborated by this study. In Miscanthus, proportionally more Diptera and Hymenoptera were found than in SRC; these orders have previously been found to be among the most abundant in Miscanthus sweep samples (Bellamy *et al.* 2009; Clapham 2011).

4.6.2 Dietary preference

The most abundant invertebrate orders found in faecal samples of both yellowhammers and whitethroats were Diptera, Larvae, Coleoptera and Arachnida, with these four groups accounting for 76-86% of the diet, depending on which bird species and which crop were being considered. Fragments of these invertebrates were present in the faeces of most broods sampled (at least 70% of broods, although usually this figure was closer to 100%, especially for Diptera and Larvae, Appendix IV). The fact that these four groups were consistently the most abundant in the diet regardless of their availability in the crop, indicates a preference for these items. In particular, Arachnids and Larvae were found in much higher proportions in faeces than in biomass crops. This is also true for Coleoptera in Miscanthus samples. The birds therefore appear to be selecting these food items, a finding that is consistent with the literature (Holland *et al.* 2006; Table 4.7). Other studies of prey choice in these species found that Lepidoptera adults are also important, occasionally more so than spiders (Moreby & Stoate 2001; Holland *et al.* 2006), but not in this study. This may be due to the crop types involved; these earlier studies looked at feed preferences in traditional arable crops. The relatively low occurrence of Lepidoptera may also be a result of the difficulty in identifying their remains in faeces. The only characteristic feature remaining in faeces is scale, thus making it very difficult to quantify the number of Lepidoptera adults consumed (Moreby 1988; Moreby & Stoate 2000). It is possible that Lepidoptera adults are therefore underrepresented in this study, but this would be equally true of other studies using this technique (Moreby & Stoate 2000, 2001), so this is not enough to explain the variation between this study and the literature.

This is the first description of the diet of whitethroat and yellowhammer nesting near biomass crops. The only examination of faecal components of birds nesting in SRC did not involve yellowhammers or whitethroats, but did find that the diet of reed

SRC did not involve yellowhammers or whitethroats, but did find that the diet of reed bunting, willow warbler (*Phylloscopus trochilus*), sedge warbler (*Acrocephalus schoenobaenus*), reed warbler (*Acrocephalus scirpaceus*) and garden warbler (*Sylvia borin*) consisted mainly of Diptera, Larvae, Coleoptera and Hemiptera (Sharples 1997).

Table 4.7 Percentages of the most abundant invertebrate groups found in whitethroat (WH) and yellowhammer (YH) faeces in arable, Miscanthus and SRC fields, from the current study (a) and the literature (b)

(a)

Invert Order	WH Miscanthus	WH SRC	YH Miscanthus	YH SRC
Arachnida	14.0	13.9	10.6	6.7
Coleoptera	11.9	17.4	20.0	17.7
Diptera	29.1	27.3	32.1	29.8
Larvae	22.2	19.3	13.5	32.5
Lepidoptera	4.2	2.0	1.8	0.5

(b)

Invert Order	WH 1996 arable	WH 1997 arable	YH 1997 arable	YH 1993 arable	YH 1995 arable	YH Mixed cropping	YH review
Arachnida	16.0	16.6	9.5	10.6	10.4	11.6	26
Coleoptera	16.5	16.3	27.3	16.2	25.0	25.4	17
Diptera	11.2	12.6	23.4	16.3	18.0	31.6	34
Larvae	36.6	33.0	22.5	48.3	26.4	13.4	61
Lepidoptera	9.4	10.3	10.88	-	-	3.1	37
Reference	1	1	1	2	2	3	4

References: 1 – (Moreby & Stoate 2001); 2 - (Stoate, Moreby, & Szczur 1998); 3 - (Macleod *et al.* 2005); 4- (Holland *et al.* 2006)

In addition to dietary preference, the predominance of some groups over others, in a way that does not reflect availability in the crop, may be indicative of the adult birds foraging outside the crop. Parent birds need to adopt a foraging strategy

containing the right combination of nutrients (Wright *et al.* 1988). To maximise efficiency, adults should minimise the time spent foraging; time away from the nest may lead to a loss of energy for chicks through heat loss, and for adults by foraging further afield (Tinbergen & Dietz 1994; Tremblay *et al.* 2005). So if faced with two patches of equal value, birds should forage in the nearest patch (Krebs & Davies 1997; Whittingham 2006) but if prey is not available in sufficient quantity, they will forage elsewhere (Poulsen, Sotherton, & Aebischer 1998; Tremblay *et al.* 2005). The high proportions of Arachnids and Larvae in faeces relative to field samples may therefore reflect shortages in the field for which the adults compensate by obtaining food from sources external to the crops themselves.

In terms of differences between crops, i.e. whether one crop provides a better food resource for nesting birds than the other, evidence suggests SRC may be the superior crop. Sweep samples taken from SRC not only contained more invertebrates than Miscanthus, but they also comprised a greater proportion of Coleoptera, which is expected from previous work examining the crops separately (Sage & Tucker 1997; Sharples 1997; Cunningham *et al.* 2004; Bellamy *et al.* 2009; Clapham 2011). The multivariate analyses also indicated a difference in the invertebrate communities found in Miscanthus and SRC. Previous studies have suggested SRC may provide good foraging resource for birds (Sage & Robertson 1994; Dhondt *et al.* 2007), but the current study is the first to attempt to investigate this empirically.

The lower abundance of invertebrates in Miscanthus and the different composition of its invertebrate community may lead to birds nesting within it having to forage greater distances than those in SRC. The diet of birds nesting in SRC boundaries appears to be more representative of invertebrate availability in the crop than the diet of birds nesting in Miscanthus. Firstly, the correlation between invertebrate abundance in sweep and whitethroat faecal samples was only true of SRC samples and not Miscanthus. Secondly, Coleoptera proportions in faeces from Miscanthus did not reflect their low availability in the crop, instead being similar, if not higher than the proportion in faeces from SRC. It is likely therefore that birds nesting in boundaries of Miscanthus are supplementing their diet with food obtained from elsewhere, more so than birds nesting in the boundaries of SRC. This seems to be

supported by the foraging observations undertaken in this study. Only 8% of foraging visits observed were made to the Miscanthus, compared to 34% to SRC. Sharples (1997) observed willow warblers, sedge warblers and reed warblers feeding predominantly in the canopy of SRC. Evidence of parent birds adapting their foraging strategy to cope with variation in food availability according to crop type is also seen in the multivariate analyses. While the two biomass crops differed in the food they provide, indicated by the difference in invertebrate communities in sweep samples, no such clustering by crop type was evident in the faecal samples, suggesting the parent birds are either strongly selecting their preferred items or are supplementing their diet with food items found elsewhere.

Besides extending foraging distance and duration, another method of sustaining adequate energy provisioning rate is via prey switching. Ideally, adults should provide their chicks with a range of nutrients, some of which are crucial in growth and development. These nutritionally important prey items however, may not be the most energetically efficient, due to the rate at which they can be delivered to the nest (Wright *et al.* 1988). This rate will be determined by the item's availability and the search and handling time involved. In times of energy shortage, the short-term survival of the chicks becomes the priority, thus prey switching may occur, whereby preferred prey are switched for more abundant but less nutritional prey items (Tinbergen 1981; Wright *et al.* 1988). The energy shortage that may instigate this switch in foraging strategy occurs as a result of enlarged brood size, suboptimal foraging conditions, or unpredictability in resource (Wright *et al.* 1988). The latter two situations could apply to birds foraging in biomass crops, if the preferred prey is lacking. By being less selective in prey choice and increasing the diversity in the diet, birds may be able to sustain sufficient food supply from Miscanthus, reducing time spent foraging (Houston & McNamara 1985). In a more favourable environment, species are likely to become more specialist in their diet, as search time decreases relative to handling time (Pyke, Pulliam, & Charnov 1977). This may explain why proportionally more invertebrates in the 'Other' category appear to be eaten by birds nesting in Miscanthus than SRC. There was however, no difference in diversity of faecal components of birds nesting in different crop types.

Further evidence of this widening of dietary intake to adapt to poorer foraging conditions is shown in this study. At low availability of invertebrates in the field, proportionally fewer Arachnids and Coleoptera were found in yellowhammer faeces, despite being preferred, meaning that more invertebrates must have been consumed from the other orders. For whitethroats, this was corroborated by the fact that Hemiptera proportions decreased as the total invertebrate abundance in field samples increased. Additionally, in faeces, proportions of Diptera and Larvae increased with the total number of invertebrates found in the faeces, suggesting an avoidance of these at lower abundances, when different invertebrates were taken. In yellowhammer faeces, when fewer invertebrates were consumed, a greater proportion of invertebrates from the 'Other' category were taken, which could also suggest diet diversification when food availability is low.

There are other factors to bear in mind when examining dietary composition. Weather conditions affect the distribution, activity and therefore availability of invertebrates, which may in turn be reflected in the diet of birds preying upon these invertebrates (Potts 1986; Honek 1997). The weather may also affect foraging behaviour; in cold, damp conditions, adults are likely to spend more time on the nest brooding chicks, thus food intake may be reduced (O'Connor 1984). Conversely, on warmer days, chicks may not require as much food or may require it in different proportions (Brickle & Harper 1999). This may explain the difference in faecal abundance between years (Grubb 1978; Petit *et al.* 1990).

An alternative explanation for this difference would be variation between observers identifying samples, but when a subsample of samples from 2011 were checked, no differences were found between observers. The abundance and diversity of invertebrates found in faeces was affected by date (abundance decreasing, diversity increasing), which again could be caused by weather conditions, or it could be due to the changing requirements of chicks as they develop. The nutritional value of invertebrates increases over time, as later instars may be more profitable than early ones, so a group that may have been ignored early on are taken with greater frequency as the season progresses (Sharples 1997). There was no effect of date on the abundance, diversity or composition of invertebrates captured in sweep samples, suggesting it is variation in foraging strategy and not

food availability that accounts for this effect of date on faecal components. Lastly foraging strategy and requirements vary between species; differences in bill structure, ability to catch insects or in the microhabitats used reduces competition between species and enables co-existence (Marchetti, Baldaccini, & Locatelli 1996). Variation is therefore seen between and within species although the diet of whitethroat and yellowhammer is fairly similar (Moreby & Stoate 2001). The current study does however, suggest that whitethroats and yellowhammers may differ in their foraging destinations. The correlation between invertebrate abundance in whitethroat faeces and that of sweep samples, while no such correlation was found in yellowhammer faeces, suggests yellowhammers may forage outside the biomass crop more than whitethroats. A difference in foraging distance between these species has previously been noted; while whitethroats obtained most of their food within 30m of the nest, yellowhammers foraged up to 300m away (Stoate, Moreby, & Szczur 1998; Stoate & Szczur 2001b).

4.6.3 Limitations and conclusions

When making comparisons of this type, it is important to bear in mind any bias caused by sampling technique. Sweep-netting used here samples canopy invertebrates alone, thus Diptera and Hymenoptera are well represented, Arachnids much less so (Standen 2000). The differences between field and faecal proportions may therefore be a product of sweep-sampling; spiders have been found in greater proportions in pit-fall samples than sweep net samples in Miscanthus (Bellamy *et al.* 2009). Neither Larvae nor Arachnids, however, have been described as particularly prevalent in Miscanthus or SRC crops, which is consistent with the results of this study. The contribution of weed content to the invertebrate composition of biomass crops has been discussed throughout this thesis; the intention of the current study was to compare dietary components with the invertebrate availability of the biomass crop itself, not the whole habitat. To determine whether the fields provide enough food, a combination of sampling techniques could be undertaken to obtain a picture of overall availability, which could then be differentiated into food availability in the crop, weeds and boundaries. More thorough foraging observations would also reveal exactly where birds were obtaining their food; in this study observations were carried out by one field worker, so only the first destination of birds was recorded. With more

observers or the use of cameras, more details could be determined such as other destinations within a foraging trip, duration of foraging trip and distance covered in foraging trip. Such information could then be used to assess whether birds nesting in Miscanthus are having to supplement their diet with food from elsewhere more frequently than those nesting in SRC, and if so, are they having to forage further afield, as has been shown previously for birds nesting in areas of low invertebrate abundance (Poulsen, Sotherton, & Aebischer 1998; Tremblay *et al.* 2005). Colour ringing the adult birds would aid observations, to more easily distinguish male from female and thus determine duration of foraging trip. Being able to recognise individuals would also allow identification of multiple broods by the same parents; some individuals may have foraging preferences which are continued with each nesting attempt, or foraging effort may vary with breeding attempt (e.g. Lislevand & Byrkjedal 2004).

The limitations involved with faecal sampling as a method of determining dietary composition have been discussed above (sections 1.6 and 3.6.2). Alternative methods include analysis of gut content, which raises ethical issues and cannot be used for studying species of conservation concern, or the use of emetics (Robert, Cloutier, & Laporte 1997; Valera & Barrios 1997) or neck collars to collect material before it is digested (Johnson, Best, & Heagy 1980; Poulsen & Aebischer 1995). These are highly invasive and may increase mortality or alter behaviour of both adults and chicks (Lederer & Crane 1978; Johnson *et al.* 2002). Faecal sampling is therefore the most appropriate method of diet determination, and has been shown to yield similar results to those achieved using neck collars (Poulsen & Aebischer 1995). Nevertheless, there are limitations involved with faecal identification, including difficulties in identifying remains, the existence of 'characteristic' body parts, and the complete digestion of soft-bodied items (Moreby & Stoate 2000). In this study, for example, the high proportion of Hymenoptera found in sweep samples relative to that found in faeces, may be explained in part by the under-representation of their body parts in faeces (Moreby and Stoate 2000).

To overcome this, some studies have calculated digestion rates wherein the number of fragments identified in faeces is related to the number of the item eaten (Custer & Pitelka 1975; Hitchcock 2010) to establish correction factors for different

species, as digestion rates vary between bird species. To calculate these factors, however, the number of prey items taken must be known, which is most achievable with captive birds, thus raising issues with the applicability of such factors to field studies. Another way of improving the accuracy of diet determination would be to carry out more detailed foraging observations. This would require the use of cameras to monitor nest visits and where possible, identify what items were being brought back to the nest (Cowie & Hinsley 1988; Kleintjes & Dahlsten 1992). The associated costs in terms of both time and money would have reduced the sample sizes in this study, thus rendering the use of cameras inappropriate for this initial study.

Finally, it should be noted that data are presented here as relative proportions, not absolute numbers, so, by their nature, a high proportion of one group leads to a low proportion of another. Thus a low proportion of e.g. Arachnids in the field may just be a result of the field samples containing more Hymenoptera, and does not necessarily mean Arachnids are in insufficient quantities in the field to support the preferences of farmland birds. This caveat is true of all studies that present data in this way and does not stop valuable insights being drawn.

Reduced food availability in Miscanthus may result in outright starvation or brood desertion, or, it may cause reduced body condition of adults or nestlings. If this impacts on the future reproductive success and therefore demography of the species, will this be of concern for the future of whitethroats and yellowhammers in the face of bioenergy expansion. Further work is needed to determine whether Miscanthus does present a poorer foraging environment than SRC and whether these crops differ from arable crops. Once this has been established, work should be carried out to determine whether changes in foraging quality will impact on bird populations. Even if birds nesting in Miscanthus boundaries are not able to obtain enough energy from the crop, there may be other advantages of nesting within it, such as increased protection from predators, or proximity to good foraging habitat. It is likely that crop management will be the most important factor influencing the foraging potential of biomass crops for farmland birds, as the presence of weed patches and abundant vegetation in margins or even strips within the crops will benefit invertebrate diversity and abundance (Sage 1998; Haughton *et al.* 2009).

The location of crops within the landscape will also affect the value of energy crops as foraging habitat, as dispersal from surrounding habitats may occur, and diversity of nearby vegetation will in turn influence invertebrate communities (Hanowski, Niemi, & Christian 1997; Dauber, Jones, & Stout 2010).

5 General discussion

5.1 Summary of findings

Presented here is the first examination of biomass crops as breeding and foraging habitat for birds in the UK. Two studies of the use of the crops by birds were carried out, one investigating the hatching success of lapwings nesting in Miscanthus, and one examining the prey availability and diet of songbirds nesting in the boundaries of biomass crops. By researching the nesting and foraging opportunities provided by biomass for a variety of species, greater insight into the potential impacts of large-scale conversion of land to biomass cultivation may be gained than looking at just one species or one habitat requirement.

Miscanthus and SRC willow provide nesting opportunities for a variety of species, including ground-nesting birds. The quality of these opportunities, however, is called into question. Lapwings nesting in Miscanthus experience lower hatch success than those nesting in arable crops. The associated change in sward structure as a result of conversion from arable to biomass crops is likely to make nests more vulnerable to predation. The indication that nests in Miscanthus may be especially vulnerable in years when the lapwings are already experiencing unfavourable conditions, e.g. bad weather leading to failure to reach adequate body condition, is of concern (see Chapter 2). Unpredictable weather, including both unseasonal dry and wet spells is likely to become increasingly common as a result of climate change (IPCC 2014), so these unfavourable years could be more frequent, resulting in greater losses in Miscanthus. For passerines nesting in the boundaries of biomass crops, there are indications that Miscanthus crop alone does not provide sufficient food resource for chick-rearing, or at least not in comparison to SRC. This could have important implications for the fitness and lifetime reproductive success of adult birds, if they are having to forage further afield to provide adequate resources (see Chapter 4). If the adults nesting in the boundaries of Miscanthus fields have to spend longer away from the nest than birds nesting elsewhere, their chicks could be of poorer body condition or be at greater risk of starvation or predation if they are not receiving food of sufficient quality or quantity (Tinbergen & Dietz 1994; Tremblay *et al.* 2005).

Nesting and brood-rearing opportunities for birds nesting in and around Miscanthus may therefore be of lower quality than birds nesting elsewhere; affording lower hatching success than arable crops and poorer foraging resource than SRC. For SRC, the evidence is less clear; it appears to be a better food resource than Miscanthus (and arable, Dauber, Jones, & Stout 2010) for passerines, but very few lapwing nests were found in SRC in this study, all of which hatched. Until a more extensive study involving many more SRC fields is carried out, we will not know whether the birds are avoiding SRC and the high hatch success rates are a result of sample size, or if they are genuine. What is clear, both from the current study and the wider literature, is the importance of structure of the crop (Semere & Slater 2007a; Bellamy *et al.* 2009; Gove & Bradbury 2010). Percentage crop cover, crop height and crop density have been shown to negatively influence nesting habitat preference of lapwings (Milsom *et al.* 2000; Hart *et al.* 2002; Durant *et al.* 2008; Nicholls 2010; and Chapter 2 of the current study) and impede accessibility to food resource (Devereux *et al.* 2004), while the weed content provides food and an element of habitat heterogeneity, important in nest and chick crypsis (Newton 2004; MacDonald & Bolton 2008b; Bellamy *et al.* 2009). In the future, as the biomass industry expands and crops become more established and commercial, crop height and percentage crop cover are likely to increase at the expense of the relatively high weed content and open patches observed currently in Miscanthus and SRC (Gove & Bradbury 2010). This could have significant impacts for nesting and brood-rearing opportunities, particularly in Miscanthus, where the weed content is the main contributor to the invertebrate community (Semere & Slater 2007a; Bellamy *et al.* 2009; Clapham 2011), thus raising questions about large-scale biomass conversion.

The next question to ask, therefore, is about the potential extent of biomass conversion, as any potential threat to biodiversity caused by changing land use will depend on this and the type of land being replaced (Gasparatos, Stromberg, & Takeuchi 2011). Estimates of the amount of land that could be made available for biomass production are varied, with NNFCC estimating a maximum of between 0.93 and 3.63 million ha (NNFCC 2012). A recent study using GIS constraints mapping suggested 3 million ha of UK land could be used for biomass production. This excluded areas that fell into any of the eleven land-cover types deemed unsuitable for conversion, such as existing woodland, BAP priority habitats, and areas of cultural

heritage. Environmentally sensitive landscapes and improved grassland were also excluded, as although these areas could be used for biomass planting, it is not currently recommended (see Karp *et al.* 2009 for a full list of constraints). Based on current end-user locations and a maximum sourcing radius of 25km, it is suggested that nearly 2 million ha are potentially available for Miscanthus planting alone (Thomas, Bond, & Hiscock 2013), which would be 12% of current agricultural land in the UK (Defra *et al.* 2014). The UK biomass strategy sets out a target of 1.1 million ha being made available for biomass cultivation by 2020 (Defra 2007a), but thus far, uptake has been far below potential (NNFCC 2012). Estimates of the current area used for biomass cultivation vary from 15 546 ha in 2007 (Defra 2007a), 17000 ha in 2009 (RELU 2009) to as low as 10000 ha (NNFCC 2012). Even if planting is increased by 20% per year, it is suggested that only an area of 0.04 million ha will be achieved by 2020 (NNFCC 2012). Reducing greenhouse gas emissions is likely to require a significant contribution from the biomass industry, so although the 2020 target is unlikely to be reached, this does not rule out large-scale land conversion in the long term.

Biomass uptake has thus far been below potential, due to a number of economic, legislative and technical barriers. A proportion of the Miscanthus and SRC planted under Energy Crop Schemes have been removed for reasons of finance and productivity (NNFCC 2012); several Miscanthus growers withdrew from the current study as they no longer found it financially viable. The relatively high cost of establishment can be prohibitive, particularly given the unpredictability of cost, yields and resulting net income (Sherrington, Bartley, & Moran 2008). The margins involved currently make biomass the less attractive option, although it has been said that biomass gives farmers an option to diversify to adapt to volatility (Convery *et al.* 2012). Aside from issues of profitability, focus groups have identified the relative novelty of biomass crops as a source of risk for growers. Farming decisions are often influenced by prior knowledge and experience, and by the experiences of neighbouring farms; given the insecurity of contracts, there is a reluctance to be the first in an area to try a new crop (Sherrington & Moran 2010; Convery *et al.* 2012). High profile failures of energy crop projects, such as Project Arable Biomass Renewable Energy (ARBRE), Winkleigh biomass energy generator and the North Wiltshire Biomass Energy Plant have not helped instil confidence (Boyle & Brown

2003; Upreti & Horst 2004; Lindegaard 2005). To use ARBRE as an example, this collapsed in 2002, leaving 35 farmers who had committed 1100 ha, stranded (Piterou *et al.* 2008). It has also been suggested that growing non-food crops goes against ingrained beliefs, particularly if it involves taking land out of food production (Convery *et al.* 2012). As well as these barriers, there are also a number of technical issues to overcome. Incompatibility of biomass boilers with Miscanthus (and sometimes SRC), lack of specialist planting and harvesting equipment, and insufficient infrastructure for processing are all standing in the way of rapid biomass expansion (NNFCC 2012). With increased investment and education, these problems can be minimised.

One of the main reasons for stagnancy in the energy crop industry is lack of confidence, caused by fluctuating policy support (Mawhood 2015). Unfortunately, it is due to the low uptake of Energy Crops Schemes, that they have been withdrawn, resulting in the current chicken-and-egg situation (Lindegaard 2013). The target area to be planted under ECS1 was 21700 ha, which rose to 40000 ha under ECS2, but only 12143 ha were planted under the two schemes (Defra & Government Statistical Service 2014). Thus only £10 million of the £76 million budget were spent (Mawhood 2015). Withdrawal of government support not only removes any financial incentive for growers, thus increasing the risk of returns, but it also impacts on the energy crop market through reducing demand and investment in the essential infrastructure, awareness and training requirements (NNFCC 2012; Rokwood 2014).

It is widely acknowledged, however, that in order to reach the ambitious target of 15% of the UK's energy coming from renewable sources by 2020, biomass should play a significant role (e.g. Aylott *et al.* 2010, DECC 2012, HM Government 2010). Without it, decarbonising the energy system would cost an additional £44 billion (DECC 2012). An increase in domestic supply would enhance fuel security and help alleviate concerns over displacement of food crops and sustainability of imported supplies (Rokwood 2014). Additionally, there is growing evidence of the multi-functional use of biomass, from the development of technologies and co-products that can be used in the non-energy sector (Volk *et al.* 2004; DECC 2012), to providing ecosystem benefits. These include carbon sequestration, flood mitigation through reduced soil erosion, and reductions in water contamination due to reduced run-off and nitrate leaching (DECC 2012; McCalmont *et al.* 2015; Milner *et al.* 2015,

Rokwood 2015). In particular, SRC plantations could be used for effective treatment of sewage and effluents, which would bring financial and fuel savings (Johnston *et al.* 2015, McCracken *et al.* 2014).

Despite the barriers to uptake and the current absence of policy support, there is potential for future expansion of biomass cultivation. Development of the industry suffered as a result of the lapse in grant support between 2006 and 2008, but then followed a period of recovery (NNFCC 2012). Advances and investment in planting, harvesting, processing and transport equipment by several companies (NNFCC 2012), increased publicity of the benefits of energy crops and stability in the supply chain, have all helped confidence in the industry to grow. Rising fuel prices and continued government commitment to renewable energies means that long term, expansion of the industry should continue (NNFCC 2012), as market demand increases. Provided policy support resumes, it seems likely that after a period of relatively slow development and confidence building, with further investment in infrastructure, there will come a tipping point into large-scale uptake of biomass crops. This could represent significant land use change in itself, but if expansion of the biomass industry reaches a point where land is being taken away from food production, this could lead to indirect land use change (ILUC) via the conversion of natural landscapes to meet food demands. Thus there is potential for rapid escalation of changing land use (Gasparatos, Stromberg, & Takeuchi 2011; NNFCC 2012), so it is important any effects of expansion on biodiversity are investigated at this early stage.

5.2 Further work

Before discussing policy implications of these findings, a number of opportunities for further work should be highlighted. Each chapter details specific work to be carried out to confirm and elucidate preceding results, but some general comments can be made here. Long-term monitoring of the study system would be greatly beneficial to determine the potential impacts of biomass expansion on farmland birds. 2012 was a particularly wet year, which saw high mortality reported across the country (BTO 2012, 2013). Not only did this preclude 2012 from the foraging potential study, as so few chicks survived to faecal-sampling stage, but it also likely affected hatch success

of lapwings nesting in Miscanthus, more so than in arable crops. This emphasises the importance of inter-year differences and the need for long-term monitoring to enable a more complete understanding of the mechanisms underway.

The main objective of this work was to assess the nesting and brood-rearing opportunities for birds nesting in and around biomass crops. In doing so, some idea of the potential impact of biomass expansion on these birds may be obtained, but only if these variations in hatching success or foraging potential translate to population changes. Although this might be suggested based on previous work stating the roles of reduced availability of nesting habitat and food in the declines of farmland birds (Newton 2004), it should not be assumed for this system. Instead, work should be carried out determining whether reduced hatching success of lapwings in Miscanthus results in an overall reduced productivity, which in turn leads to population declines. Likewise, does a lower quality food resource in Miscanthus influence fledging success, lifetime reproductive success and overall survival? By measuring success rates throughout the nesting cycle, vulnerable stages can be determined as can overall productivity for each habitat. Lastly, it is important to remember that population trends are not governed by productivity alone, but also by immigration and emigration (Stillman *et al.* 2006), thus any effects of habitat change may not be seen immediately in the population, or may affect other populations. In the case of Miscanthus, there are concerns that it may act as a breeding sink for Lapwings (Gove & Bradbury 2010). This work has focused on the impact of converting land to biomass crop production on breeding birds, as reduced productivity is a key factor in the decline of farmland birds (e.g. Peach, Thompson, & Coulson 1994; Bradbury *et al.* 2000; MacDonald & Bolton 2008b), but survival throughout the year will also contribute to population trends (Fuller *et al.* 1995; Newton 2004). An increase in biomass crop production will likely lead to a reduction in suitable wintering habitat for lapwings and other species characteristic of open landscapes (e.g. skylark *Alauda arvensis*), as these birds will avoid foraging in the tall, dense swards formed by Miscanthus and SRC (Milsom *et al.* 1998; Anderson, Haskins, & Nelson 2004; Sage *et al.* 2010). Conversely, these crops could present a new resource for other foraging birds, such as buntings and granivorous finches that are attracted by the cover provided and the weed resource within the crop (Bellamy

et al. 2009). Thus the impact of changing land use caused by the expansion of the energy crop industry should be investigated throughout the year.

5.3 Policy Implications

When discussing policy implications with regard to biomass crops, the most important issue to address is the absence of targeted energy crops schemes. It is clear that to reach our energy targets, domestic biomass supplies must increase significantly. To achieve this, growers must not only perceive biomass production as a financially viable option but also see government confidence in the industry (NNFCC 2012; Lindegaard 2013). Currently, planting biomass crops presents too much risk with too little reward, which must change if biomass is to make any significant contribution to the UK energy mix (Rokwood 2014). Uptake of grants offered by the Energy Crops Schemes was far below expectations for a number of reasons, so Rokwood (2014) and Crops for Energy Ltd (Lindegaard 2013) have developed recommendations to address these flaws in future schemes. The main focus of these recommendations is to reduce the investment risk for growers, which can be achieved by providing interim payments, helping to source alternative end users should one fail, and dedicating funding to the development of processing, storage and machinery technologies, in order to achieve a more robust infrastructure (Lindegaard 2013; Rokwood 2014). Other suggestions are to simplify the application process, making it less bureaucratic and prescriptive, and to fully review the costs and benefits of biomass to inform policy (Lindegaard 2013; Rokwood 2014).

Once the appropriate government support is in place, it is vital that any changes in land use are undertaken sustainably, especially given the fragility of the agricultural landscape. Policy measures should be introduced to encourage land conversion in a sustainable way, both environmentally and economically, and that is sensitive to increasing food production demands. One of the ways in which impacts of biomass expansion on biodiversity could be minimised is through regulation of the type of land conversion and the location in the landscape. Assessments of the potential area of land available for biomass tend to exclude some areas based on the environmental impacts of the conversion (Karp *et al.* 2009; Thomas, Bond, & Hiscock 2013). These areas include environmentally sensitive landscapes and improved

grassland. In this study, success rates were reduced in Miscanthus compared with beet and spring barley, but it may be that Miscanthus is replacing less profitable crops such as winter wheat (Bright *et al.* 2013). Equally if grassland or set-aside is being replaced, this is likely to have negative impacts on lapwings and other biodiversity (Anderson, Haskins, & Nelson 2004). Data on crops preceding Miscanthus and SRC are currently lacking, other than references to energy crops often being planted on the least productive land or former set-aside, and a reference in Bright *et al.*'s (2013) study to Miscanthus mostly replacing grass and wheat. Collecting information from growers regarding what crops precede biomass crop plantations would help to show current preferences. If demand for biomass increases however, it is likely that there will be great variation in the crop it replaces, thus decreasing the relevance of this issue. As an aside, it would be beneficial to obtain detailed cropping history of the land prior to biomass planting, as this would help to determine that the differences between crop types observed in this and future work are due only to crop type, and not the intrinsic productivity of the land.

The impacts of land conversion must also be thought of in terms of habitat mosaic effects, as biodiversity is influenced by the surrounding landscape (McCracken & Tallowin 2004; Eglington *et al.* 2009). To maximise yield and profitability, energy crop monocultures may arise (Gasparatos, Stromberg, & Takeuchi 2011), contravening the general recommendation for farmland mosaics and heterogeneous landscapes (Siriwardena *et al.* 2000). Additionally, there is evidence of reduced body mass of wader chicks hatched in monocultures, which are then at higher starvation and predation risk (Kentie *et al.* 2013). Rather than introducing vast energy crop monocultures, biomass crops could be used to introduce variety to otherwise homogenous landscapes (Hartman *et al.* 2011). The surrounding landscape will affect biodiversity within the biomass crops, as shown for bird and invertebrate communities (Christian *et al.* 1998; Berg 2002; Dauber, Jones, & Stout 2010). The location of biomass plantations within the agricultural landscape should therefore be taken into account in planting decisions.

If biomass expansion does increase to an extent where it may impact on biodiversity, different management options may be important to minimise the risk. Currently energy crops are not covered by any form of environmental stewardship

option, but it has been suggested that this could be of value, e.g. retaining areas of open crop within the plantation, or strips of beneficial weed cover within fields or in margins, that may benefit invertebrates (Engel, Huth, & Frank 2012).

The age of coppices will influence composition of bird communities using them, owing to the height and structure of the crops themselves and the ground vegetation within (Sage & Robertson 1996; Campbell *et al.* 2012). A combination of clones, age classes and harvesting regimes within a plantation may therefore ensure the biodiversity benefits of SRC are maximised (Göransson 1994; Sage & Robertson 1996; Dhondt *et al.* 2004, 2007; Campbell *et al.* 2012).

Similarly, bird use of Miscanthus may depend on the planting regime employed. A new method of precision planting has recently been developed, which is carried out manually to ensure more regular spacing than the more traditional method and results in a visibly different sward structure post-harvest (IEC 2013). This new method has been developed to increase yield, but it was noted in the current study that areas of precision planting tended to contain more lapwings than the traditionally planted area, which invariably held no birds at all (at three sites where adjacent fields were under different planting regimes). As planting using this technique increases, it would be interesting to examine whether this apparent preference exists and determine the causes underlying it; the thinner, more regular sward structure may afford better nest protection whilst allowing good visibility from the nest. This method, however, has only recently been developed (the first Miscanthus fields were planted in this way in 2010; IEC 2013) so the crops planted in this way are very early in their establishment. Thus it is possible that the benefits of this method may disappear with time as the crop becomes less weedy and more dense, especially as the aim of this technique is to increase yield by reducing the patchy nature of the crop (IEC 2013).

5.4 Conclusions

The purpose of this work was to examine the impacts biomass crop expansion could have on farmland birds, by assessing the quality of habitat that biomass provides. Two crucial aspects of breeding habitat were examined; the availability and quality of nesting sites within biomass crops for ground-nesting birds, and the food availability

within biomass crops for passerines nesting nearby. Biomass crops do provide nesting and foraging opportunities for a variety of species, but the quality of these opportunities varies with crop type and may be more influenced by external factors such as weather, than is the case in arable crops. The work indicates the need for long-term research into the issue and shows the importance of looking at a number of species. Previous work has suggested overall biodiversity benefits of biomass crops, in terms of species richness and abundance, but these benefits can be very species specific. By looking at the hatching success and diet of different species, a more detailed view of the quality of habitat can be obtained. With more information regarding the advantages or disadvantages of biomass for different species, appropriate policy measures can be put in place to help mitigate against any potential risk of biomass expansion to already vulnerable farmland birds.

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7 Appendices

Appendix I: Structure of biomass fields



Fig. 1 Miscanthus before harvest (March)



Fig. 2 Recently cut Miscanthus (March)



Fig. 3 Miscanthus after 2 months growth (May)



Fig. 4 Recently harvested SRC willow (March)



Fig. 5 SRC after several months' growth (July)

Appendix II Skylark nest data

The original aim of the work in Chapter 2 was to investigate the impact of biomass production on two ground-nesting species; lapwing and skylark. It soon became apparent however, that with the time and resources available, it would be better to focus all efforts on searching for lapwing nests as this proved the most fruitful. Using the methods stated below, only seven skylark nests were found across the three years (Table 7.2).

Methods

During field visits as part of the lapwing work, surveys were undertaken to determine the presence of skylarks. On arrival, fields were observed from a location at the perimeter for at least half an hour, followed by walking the length of the perimeter, pausing at regular points. Sketch maps were drawn showing the locations of singing male skylarks. Where evidence of breeding behaviour was observed (Table 7.1), these locations were watched for a prolonged period usually using a car as a mobile hide to minimise disturbance or by sitting along the field edge close to trees or vegetation to break up the observer's outline (A. Hoodless, personal communication). Once birds were seen repeatedly dropping down to the same area, it was marked with bamboo canes and searched until the nest found, or twenty minutes had passed, to minimise disturbance. Skylarks do not usually fly directly to the nest but instead drop down to the area and walk the remaining distance to the nest (Ferguson-Lees, Castell and Leech 2011). Sometimes the female will fly up from the nest to join the descending male, so this behaviour was looked for in particular to try and locate nests (*loc cit)*. Despite significant time and effort spent on searching for skylark nests, only seven nests were found (Table 7.2), four of which were found by accidental flushing of the incubating female, rather than the observation method outlined above. By 2013, skylark nest finding had been abandoned due to the lack of data.

Table 7.1. Behaviours looked for as evidence of breeding.

Breeding stage	Behaviour
Territory establishment/nest building	Males displaying, bringing nesting material, often directly to nest site.
Incubation	Short, inconspicuous foraging flights by female.
Chicks present	Both parents carrying food to nest, usually land away from nest and walk back or to different places if chicks have left the nest.

Adapted from Hitchcock (2010).

Table 7.2 Skylark nests found in Miscanthus. No nests were found in SRC.

Year	No. of nests found	Successes
2011	4	2
2012	2	0
2013	1	0

Appendix III Hedgerow nests

Chapter 4 examines the foraging potential of biomass crops for yellowhammers and whitethroats. These species were chosen as they were the most frequently found, but Table 7.3 details the other nests found. In 2013, it was decided that in order to obtain maximum data, search efforts would concentrate on yellowhammers and whitethroats; other nests were not recorded. Nests were found in 2012, but nest success was so low that insufficient faecal samples were collected.

Table 7.3 Nests found in the boundaries of biomass fields

		Miscanthus		SRC	
Year	Species	Nests found	Successes	Nests found	Successes
2011	Whitethroat (*Sylvia communis*)	8	6	2	2
	Yellowhammer (*Emberiza citronella*)	1	0	2	2
	Reed Bunting (*E. schoeniclus*)	1	1	0	0
	Blackbird (*Turdus merula*)	3	2	6	4
	Dunnock (*Prunella modularis*)	1	1	3	2
	Song Thrush (*T. philomelos*)	1	1	0	0
	Willow Warbler (*Phylloscopus trochilus*)	1	1	0	0
	Chaffinch (*Fringilla coelebs*)	0	0	1	1
	Total	**16**	**12**	**14**	**11**
2012	Whitethroat	5	2	1	0
	Yellowhammer	14	4	3	1
	Reed Bunting	1	0	0	0
	Willow Warbler	0	0	3	3
	Chiffchaff (*Phylloscopus collybita*)	1	1	0	0
	Total	**21**	**7**	**7**	**4**
2013	Whitethroat	13	8	5	5
	Yellowhammer	16	12	4	2
	Reed Bunting	7	6	3	2
	Total	**36**	**26**	**12**	**9**

Appendix IV: Composition of faecal samples

Each order is shown as an overall proportion of the total invertebrates found for each species, and as a mean of the proportions for each brood (Table 7.4). Also summarised is the percentage of broods containing each order.

Table 7.4 Composition of Whitethroat faeces. Shown below are the proportion of each order across all broods and per brood, taking the number of faecal sacs into account, and finally the percentage of broods that contained each order.

Crop	Invertebrate Order	Overall proportion	Mean proportion per brood ± se	Percentage broods
Miscanthus	Diptera	0.33	0.29 ± 0.05	100
	Larvae	0.26	0.22 ± 0.03	88
(37 samples	Arachnida	0.13	0.14 ± 0.03	88
from 16 nests)	Coleoptera	0.10	0.12 ± 0.03	75
	Hemiptera	0.09	0.08 ± 0.02	63
	Hymenoptera	0.04	0.05 ± 0.02	56
	Lepidoptera	0.02	0.04 ± 0.02	25
	Orthoptera	0.02	0.04 ± 0.02	25
	Unknown	0.02	0.01 ± 0.01	19
	Odonata	0	0	0
	Gastropoda	0	0	0
	Dermaptera	0	0	0
SRC	Diptera	0.29	0.27 ± 0.04	100
	Larvae	0.18	0.19 ± 0.03	100
(19 samples	Coleoptera	0.20	0.17 ± 0.06	75
from 8 nests)	Arachnida	0.10	0.14 ± 0.05	88
	Hemiptera	0.13	0.13 ± 0.01	100
	Hymenoptera	0.03	0.03 ± 0.02	25
	Lepidoptera	0.03	0.02 ± 0.01	38
	Dermaptera	0.02	0.02 ± 0.01	25
	Unknown	0.02	0.01 ± 0.01	25
	Odonata	0.01	0.01 ± 0.01	13
	Orthoptera	0	0	0
	Gastropoda	0	0	0

Composition of Yellowhammer faeces. Shown below are the proportion of each order across all broods and per brood, taking the number of faecal sacs into account, and finally the percentage of broods that contained each order.

Crop	Invertebrate Order	Overall proportion	Mean proportion per brood ± se	Percentage broods
Miscanthus	Diptera	0.36	0.32 ± 0.05	80
	Coleoptera	0.20	0.20 ± 0.04	93
(34 samples	Larvae	0.12	0.13 ± 0.03	73
from 15	Arachnida	0.11	0.11 ± 0.03	80
nests)	Hemiptera	0.09	0.07 ± 0.02	47
	Orthoptera	0.05	0.05 ± 0.02	53
	Lepidoptera	0.03	0.02 ± 0.01	33
	Hymenoptera	0.01	0.01 ± 0.01	13
	Unknown	0.01	0.005 ± 0.005	7
	Odonata	0	0	0
	Gastropoda	0	0	0
	Dermaptera	0	0	0
SRC	Larvae	0.43	0.32 ± 0.06	100
	Diptera	0.26	0.30 ± 0.06	92
(29 samples	Coleoptera	0.13	0.18 ± 0.05	92
from 13	Hemiptera	0.08	0.09 ± 0.02	77
nests)	Arachnida	0.06	0.07 ± 0.01	77
	Hymenoptera	0.02	0.02 ± 0.01	38
	Dermaptera	0.01	0.01 ± 0.01	15
	Gastropoda	0.01	0.01 ± 0.01	23
	Lepidoptera	0.002	0.01 ± 0.01	8
	Odonata	0.002	0.004 ± 0.004	8
	Orthoptera	0.002	0.002 ± 0.002	8
	Unknown	0	0	0