Managing vegetation for the conservation of the Common Kestrel *Falco tinnunculus* on farmland in England

Claudia M. Garratt

School of Biology
Faculty of Science, Agriculture and Engineering

A thesis submitted for the degree of Doctor of Philosophy, October 2011,
Newcastle University

Table of Contents

List of Figures and Tables

Abstract

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Agricultural intensification is a major threat to a range of biodiversity. Of particular concern at a Europe-wide level are population declines of farmland birds, an index of whose numbers are taken to be an indicator of the health of biodiversity as a whole. In this thesis, I explore how to aid the populations of a declining farmland raptor, the Common Kestrel *Falco tinnunculus*, via the potential management of habitat, and then I link this to Agri-environment Scheme (AES) options, which require continual adaptive management to maximise their benefits.

AESs have been responsible for the creation of additional areas of grassland in the British countryside, which could be expected to be of benefit to the Kestrel; a grassland hunter. A study of foraging breeding Kestrels showed that they select habitats non-randomly with cut grass being selected over all other available habitats.

I then explored the potential effects of vegetation management on Kestrels' preferred prey, small mammals, in a manipulative field experiment. While most small mammals will leave an area of grassland after mowing, I show that a small number may remain as long as some form of cover – such as the cut grass – remains *in situ*.

I investigated the temporal effects on farmland bird species of cutting grass both within whole fields and on field margins, There was a rapid drop-off in use of grass fields after cutting suggesting a rapid decline in available resources. Field margins that were cut in strips were favoured by some species, although others preferred to use the longer, control, vegetation.

Overall I conclude that targeted cutting of small patches or strips throughout the year would create mosaics of different habitat and therefore be of benefit to a range of different declining farmland biodiversity, and could be a worthwhile addition to AES field margin prescriptions.

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Conference, so thanks Rob. Some of the data gathered with all of your help has not ended up being used for this thesis, but I have no doubt it will be useful in the future for helping efforts to conserve the Kestrel.

The northern England-based fieldwork owes an enormous debt of gratitude to Durham Wildlife Trust, in particular Jim Cokill and Mark Richardson, who again have shown interest in the project, and have been accommodating beyond what I could ever have expected. All the other extensive helpers have been acknowledged at the end of the relevant chapter.

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Chapter 1: General Introduction

Agriculture and its Effects on Biodiversity

Changes to the farmed landscape in the UK are nothing new (O'Connor & Shrubb 1986, Stoate *et al*. 2001). Prior to the 17th century open, common-land was widespread, and communal farming in open-field systems was likely to have been a predominant farming method since the creation of villages from the 9th century onwards (Hey *et al*. 2009). But then the enclosures movement started to pick up pace in the 16th and 17th centuries, leading to over 4000 Parliamentary Enclosures Acts being passed between 1750 and 1850, permitting the enclosing, by fencing or planting of hedges, of at least 6.8 million acres of farmland in that period alone (Hey *et al*. 2009). This had an immediate negative impact on populations of open-country bird species such as the Stone Curlew *Burhinus oedicnemus* and the Great Bustard *Otis tarda* (Fuller 2000), the latter of which has now been extinct in the UK for over 150 years (Osborne 2005), and the former is severely threatened (Green *et al.* 2000). However, this period and the associated rising prices for livestock products also saw the creation of new resources for some farmland birds, as fodder crops such as clovers began to be widely planted (Fuller 2000, Hey *et al.* 2009). The planting of hedgerows during the enclosures, and the associated creation of field margin habitats, will also have significantly increased the area of these habitats on lowland English farms, creating new ecological niches - certainly 20 species usually classified as woodland birds are commonly found in hedgerows and other margin features on farmland (O'Connor & Shrubb 1986, Fuller *et al.* 1995). The hedgerows almost certainly also facilitate dispersal of species through farmed landscapes by acting as corridors (Fuller 2000). However, roughly half of these hedges have been lost since 1947 (Barr & Parr 1994) as Britain has undergone the latest period of agricultural change – the intensification of the late 20th century.

While changes in agricultural practice *per se* may be nothing new, what is new is the sheer pace and scale of these changes in the latter half of the 20th century, when the process of agricultural intensification really began in earnest. Technological advances, a leap in population density and the associated drive to increase food production since the Second World War have led to ever more intensive farming methods (Krebs *et al*. 1999, Fuller 2000, Evans & Green 2007). This has been typified by two broad types of change: widespread loss of semi-natural vegetation such as hedgerows, woodland and semi-natural grassland within farmland, and changes in the techniques of crop and grass production (Fuller *et al.* 1995), with technological advancement being the main cause underlying these changes (Chamberlain *et al.* 2000, Fuller 2000). The key changes which have taken place in British agriculture in the past 30 years are:-

- Increased mechanisation, leading to increased efficiency of harvest and less grain spillage.
- Removal of hedgerows, improvement of rough grassland and drainage of wet meadows.
- Changes in crops, comprising for example increases in the amount of wheat and oilseed rape, and decreases in oats, barley and potatoes.
- A move away from mixed farming and towards monoculture.
- Increased use of agrochemicals, most notably pesticides and inorganic fertilisers.
- A simplification of crop rotations, reduction of under-sown leys and a switch from spring-sown to autumn-sown crops.
- A switch in grassland management from hay to silage production, leading to earlier and multiple cuts.
- Changes in livestock farming, with numbers of sheep increasing between the mid 1970s and 1990.

(Taken from Chamberlain *et al.* 2000, Fuller 2000, and Newton 2004. For a more detailed breakdown of the intensification processes which have taken place in UK farming since the early 1960s, see Chamberlain *et al.* 2000, and for the probable effects of these changes on farmland birds and other taxa see Fuller 2000 and Stoate *et al.* 2001). Many of these processes are linked, for example the increase in mechanisation has prompted the removal of hedgerows to allow large machines to work more quickly and efficiently. It also enables rapid, large-scale application of fertilisers and pesticides, which in turn has reduced the need for fertility-building grass leys, more complex crop rotations, and mixed farming systems to provide manure, and it has enabled the autumn sowing of cereal crops (Fuller 2000).

It is now widely accepted that the increase in agricultural intensification and the concurrent decline in farmland biodiversity are linked (e.g. see Krebs *et al.* 1999, Wilson 1999, Chamberlain *et al.* 2000, Fuller 2000, Donald *et al.* 2001, Stoate *et al.* 2001, Benton *et al.* 2003, Burel *et al.* 2004, Newton 2004), the most likely mechanisms being reduction in suitable habitat and food supply, and direct mortality caused by farming practices such as increased mechanisation and use of pesticides (Fuller *et al.* 1995, Wilson 1999, Chamberlain *et al.* 2000, Fuller 2000, Newton 2004). This change has affected species which rely primarily on farmland from all taxa, (Krebs *et al.* 1999, Wilson 1999, Stoate *et al.* 2001, Burel *et al.* 2004), but has been particularly marked for farmland birds - although this is possibly also partly a function of increased detectability of population changes for birds due to the wealth of demographic information we have for this taxon in the UK. Long term monitoring of bird populations by the British Trust for Ornithology (BTO) has been in place since 1962, first as the Common Bird Census (CBC) (Marchant *et al.* 1990, Fuller *et al.* 1995, Siriwardena *et al.* 1998) then from 1994 until present as the Breeding Bird Survey (BBS) (Risely *et al.* 2010). During the period of this monitoring, many of the birds associated with farmland have declined either in numbers or in the size of their ranges, or both (Marchant *et al.* 1990, Fuller *et al.* 1995, Siriwardena *et al.* 1998, Donald *et al.* 2001, Gregory *et al.* 2004, Newton 2004). These changes first became apparent in the mid to late 1970s, which coincides with the period *c.* 1970-1985 which saw the most rapid intensification of agriculture (Fuller *et al.* 1995, Chamberlain *et al.* 2000, Fuller 2000). It has been estimated that ten million breeding individuals from ten species of farmland birds have disappeared from the British countryside within the 20 years between 1979 and 1999 (Krebs *et al.* 1999). As discussed above, many of the agricultural changes are strongly correlated, and often closely interlinked, which makes discerning the effect of one individual change on bird populations difficult – it seems there is no single mechanism underlying the declines in farmland bird populations (Chamberlain *et al.* 2000, Fuller 2000). However, this issue has caused much public interest in recent times, and led to the initiation of a Government Public Service Agreement designed to protect biodiversity, which was in place for ten years up until June 2010. Its progress was measured with 5 key indicators, one of which was “biodiversity as indicated by changes in wild breeding bird populations in England, as a proxy for the health of wider biodiversity” (Anon 2007, 2009). This indicator is an aggregate index of wild bird populations – dubbed the ‘Skylark Index’ in the popular press – one facet of which is the Farmland Bird Index (FaBI), whereby the population trends of nineteen bird species living on British farmland were designated by the UK government as a ‘Quality of Life’ indicator (Vickery *et al.* 2004). This index fell by over 40% between 1970 and the late 1990s (Grice *et al.* 2007). Some species, such as the Great Bustard, the Wryneck *Jynx* *torquilla* L., the Corncrake *Crex crex* and the Red-backed Shrike *Lanius collurio*, may already have been lost from British farmland for good (Gregory *et al.* 2004, Evans & Green 2007, Pomeroy *et al.* 2008). Others, such as the Stone Curlew and the Cirl Bunting *Emberiza cirlus* are clinging on in small pockets of habitat where the farming system has effectively been geared towards single species conservation (Aebischer *et al.* 2000, Peach *et al.* 2001, Grice *et al.* 2007). These 'narrow and deep' Special Projects can be considered to be one of the success stories – at a local scale at least - of Agri-environment Schemes (AESs) in the UK (Grice *et al.* 2007).

Agri-environment Schemes

Agri-environment Schemes are the principal strategy being introduced throughout Western Europe to achieve a range of environmental conservation objectives on farmland (Ovenden *et al.* 1998, Berendse *et al*. 2004, Bradbury *et al*. 2004, Kleijn *et al.* 2006), and they are the main mechanism by which the UK government hopes to reverse the population declines of farmland species (Smallshire *et al.* 2004, Vickery *et al.* 2004). In the UK, the Environmental Stewardship scheme (ES) has replaced earlier schemes such as the Environmentally Sensitive Areas scheme (ESA) - which was the first UK AES launched in 1987 - and the Countryside Stewardship Scheme (CSS) initiated in 1991 (Ovenden *et al.* 1998), although some farmland is still managed under CSS agreements. Agri-environment agreements are designed in part to promote biodiversity on farmland by compensating farmers for any financial loss associated with managing their land in a manner which benefits biodiversity and the environment (Ovenden *et al.* 1998, Grice *et al.* 2007). The downward trend in populations of farmland birds seen in the UK has been mirrored elsewhere in Europe (Donald *et al.* 2001) resulting in twenty percent of EU farmland being under some kind of agri-environment agreement (Kleijn *et al.* 2001). However, despite the enormous cost of these EU schemes (estimated to be around 24 Billion Euros spent between 1994 and 2002, (Kleijn & Sutherland 2003), their effectiveness in increasing wildlife populations remains largely unproven (e.g. see Kleijn *et al.* 2001, Bradbury & Allen 2003, Kleijn & Sutherland 2003, Kleijn *et al.* 2006, Wilson *et al.* 2007, Davey *et al.* 2010). ES - the current most prevalent UK scheme - is made up of two tiers: the 'broad and shallow' Entry Level Scheme (ELS) is an approach designed to encourage as many farmers as possible to adopt simple environmental management options. In terms of biodiversity, it aims to offer some benefits to as wide a suite of species and habitats as possible. The higher tier is the ‘narrow and deep’ Higher Level Scheme (HLS), where very specific targeted measures are put in place – often to benefit one or a very small number of priority species or habitats. Entry to HLS is competitive, the prescriptions and application process more complicated, but the financial rewards greater (Vickery, Chamberlain & Noble 2005, Evans & Green 2007, Davey *et al.* 2010, Natural England 2010a & b). Due to its outcome-driven approach, it is from this latter type of scheme that many of AESs’ successes have come (Aebischer *et al.* 1983, Peach *et al.* 2001). The success of narrow and deep conservation measures such as the Special Projects highlights one of current probable failings of AESs however; different species have different requirements and therefore some of the very specific measures put in place to rescue one species may be in conflict with the needs of many other declining farmland species (Berendse *et al.* 2004, Vickery *et al.* 2004, Whittingham 2007). Conversely, the less targeted broad and shallow schemes such as ELS are often applied in a fairly haphazard way, to geographically-separate small patches of land, or in areas where target species are absent (Whittingham 2007), and there is generally a bias in uptake towards the easier to implement field boundary options (Davey *et al.* 2010). One thing is clear – AESs need continual monitoring, management and adjustment in order to maximise their benefits for as wide a range of species and habitats as possible. What is needed is a holistic approach. Failing taking a huge technological step backwards and farming as we did 60 years ago, which is neither possible nor practical, then increasing heterogeneity at a field, farm and landscape scale, to provide habitats for as wide a range of taxa as possible while still ensuring plentiful food production, must surely be the goal, particularly as loss of heterogeneity in the farmed landscape is one of the most noticeable overall effects of the intensification of the last 60 years (Benton *et al.* 2003).

The Kestrel as a Study Species

Kestrel Ecology

One of the 19 farmland birds included in the Wild Bird Indicator is the Eurasian (or Common) Kestrel *Falco tinnunculus* - Britain’s commonest raptor. It is a relatively small, but highly visible falcon of open grassland, and can be found almost anywhere there are places to nest and a plentiful supply of food available (Village 1990), and it has therefore always been closely associated with farmland. Traditionally they nest on ledges, in cavities or abandoned stick nests, although they are becoming more and more associated with anthropogenic – and on occasion even urban – sites, and will nest in farm buildings, and have even been known to nest in window boxes of high rise buildings (Village 1990). Kestrels are early summer, single-brood breeders, who lay four to six eggs on average and usually fledge *c.* three to four young per successful pair (Village 1990), but recently productivity has been identified as their probable most limiting demographic factor (Vickery *et al.* 2004). Several factors are at play to mediate breeding season success, including age and experience of the breeding pair, and food supply limitation (Village 1990). Kestrels are adaptable hunters and will eat almost anything they can catch, including birds, frogs, lizards, large insects and earthworms (Village 1990). Other more unusual remains are also occasionally found in nests (e.g. moles, juvenile magpie and a baby weasel – Village 1990 and personal observation), but their preferred prey is small mammals, particularly diurnal field voles of the genus *Microtus* (the Short-tailed Field Vole *Microtus agrestis* is the only member of this genus in the UK). This is most likely due to their large size in comparison to many other common prey items (Bird *et al.* 1982, Korpimäki 1984), as foraging theory states that size can be used in most cases by foragers as a reliable proxy for energetic value (Stephens and Krebs 1986), and therefore taking larger prey for the same hunting effort is energetically advantageous. Like most generalist raptors though Kestrels are opportunistic, preying on whatever is most readily available (Bird *et al.* 1982, Korpimäki 1985) – they often switch to naïve fledgling birds in spring and early summer for example - but only usually when voles are scarce do Kestrels rely heavily on other prey (Korpimäki 1985, 1986, Village 1990, Korpimäki & Norrdahl 1991). This preference for *Microtus* voles is demonstrated in numerous studies in which Kestrel demography can be shown to cycle with that of their small mammal prey (e.g. see Village 1982, Korpimäki 1984, Korpimäki & Norrdahl 1991).

Microtus agrestis and its Continental counterpart the Common Vole *Microtus arvalis* are associated with rough grassland habitats (Tapper 1976, Village 1990, Bellamy *et al.* 2000, Tattersall *et al.* 2000). Their numbers (along with those of many other small mammals) have declined throughout Europe as the intensification of agriculture has lead to a decrease in uncropped land such as rough field margins and hedges on farms, and more intensive management of both pasture – including greater grazing and mowing pressure – and arable land (Aschwanden *et al*. 2007, MacDonald *et al*. 2007). The primary problems facing small mammals on farms are therefore the same as those facing many other declining farmland species – the loss of food sources and suitable habitat. Of all the small mammals which occur most frequently in the Kestrel's diet on farmland, only the Wood Mouse *Apodemus sylvaticus* is commonly found within the crop, and this species is largely nocturnal (Shore *et al.* 2005, Butet *et al.* 2006). As such, other open, uncropped areas where small mammals can be found - such as road verges - are increasingly important hunting areas for diurnal raptors such as the Kestrel (Meunier *et al*. 2000). A primary focus of AESs is to provide food for farmland birds, either by providing previously absent food or by improving access to existing resources (Whittingham 2007), and in light of this it could be expected that some of the current AES measures should be of benefit to Kestrels and their small mammal prey. The replanting and maintenance of hedges, the creation of low input grassland, and the provision of wide grassy margins and buffer strips around arable and pasture crops (Natural England 2010) are just three of the measures which should, on the face of it, increase suitable habitat for small mammals – and therefore suitable hunting habitat for raptors – on farmland. Indeed, a 2005 study by Aschwanden and colleagues showed that small mammal summer densities were eight times higher on Ecological Compensation Area wildflower and herbaceous strips than on low intensity meadows and artificial grassland, so it seems likely that the increase in suitable grassland habitat will have led to an increase in small mammal populations on AES farms. Based in part on these considerations, a modelling exercise which was carried out prior to the inception of the pilot ELS in 2003 predicted that the Kestrel was one of the species most likely to benefit from the new scheme (Grice *et al*. 2007).

Recent Declines

Kestrels are currently amber-listed in the UK and are considered, due to their historical moderate declines (20% decline 1995-2008, Risely *et al*. 2010), to be a species of conservation concern at a Europe-wide level (Eaton *et al*. 2009). There are probably *c*. 35,000 breeding pairs of Kestrels in the UK currently. This number stood at nearer 55,000 in 2007 (Robinson 2005), but the population has gone through more alarming declines in recent times, declining by around 36% between 2008 and 2009 (Riseley *et al*. 2010). According to BBS data, the patterns of decline differ throughout the UK (British Trust for Ornithology 2011), making the processes at work – and there are sure to be many - even harder to unravel. Additionally, as for many other farmland birds, the factors which initiated the declines may differ from those that are responsible for their continuation (Fuller 2000).

Foraging Theory

The fact that Kestrels are continuing to decline on farmland in the face of measures which could be expected to help them indicates that some aspects of the habitat being created via AESs are still sub-optimal for Kestrels in terms of foraging suitability. Optimal foraging models predict the foraging behaviour of animals based on the assumption that natural selection has resulted in animals that forage so as to maximise fitness (Pyke *et al*. 1977). Conventional foraging models maximise net rate of energy gain while foraging; foraging both costs and provides energy - there must be a balance between the energy gained from prey, and the energy expenditure of finding and handling that prey - therefore under ideal circumstances there is a net energy gain. Hence, foraging theory predicts that it is not only prey abundance, but also accessibility which governs where predators choose to hunt, as this is the primary factor affecting their intake rate. Intake rate is determined by the rate at which food is eaten, but is mediated by many factors such as disturbance, predation risk and food accessibility. More energy (‘maximisation’) is assumed to be better, as it leaves ‘spare’ energy for non-feeding activities such as reproducing and territory defence. Basic prey and patch models therefore assume a currency of “maximising long term average rate of energy intake”. Rate maximisation can be achieved either by maximising the energy taken in over a fixed period of time, or by minimising the time it takes to gain a fixed amount of energy (Schoener 1971 via Stephens and Krebs 1986). Which foraging decisions are better for a forager can vary depending on which of these strategies is employed.

Rate maximising depends on many factors, or ‘constraints’. Constraints are the elements of a model defining all the factors which limit and define the relationship between the currency and decision variables; the ‘limitations’, so to speak. These can be i) intrinsic to the animal; ii) extrinsic on the animal. Vegetation length and density potentially constitutes both an intrinsic and extrinsic constraint on Kestrel foraging. It is an extrinsic constraint from the environment, but it interacts with the intrinsic constraints for a Kestrel that it is a visually guided hunter and therefore needs to see its prey.

Modelling is always a trade-off between being a simple guide, or being more complex and realistic, but clumsy. As such, the most basic conventional foraging models have definite limitations, including: i) there are other factors beyond time and energy which influence foraging behaviour – wariness, territorial defence i.e., which are not taken into account in the basic models; ii) the models are static – the forager's state (hunger for example) is not taken into account; iii) models do not allow forager to use (adapt behaviour in response to) information gained while foraging. These problems are at least partially addressed in several later extensions to the basic models, which allow modelling of some slightly more complex and realistic scenarios. One example is central-place models - extensions of the basic prey and patch models which make predictions about how patches should be used if the forager has to keep returning to the central place (i.e. nest) after each foraging trip, and how far from the central place items should be attacked. Kestrels, during the breeding season at least, are 'single-prey loading' central-place foragers (Orians & Pearson 1979 via Stephens and Krebs 1986), meaning that on each foraging expedition they hunt for a single prey item before returning to the nest with it. For a single-prey loading forager, total within patch time is the sum of two components: within patch search time, and prey manipulation time. When a forager must search within a patch, being unselective about prey taken can reduce search time, and therefore the patch-use tactic 'be unselective' will often be the rate-maximising choice (this is never the case in the simpler models). We see this in nature with Kestrels, who are highly adaptable and opportunistic hunters. This tactic may well change with distance from the central place – a single-prey loader may be unselective when patches are close, but select for larger prey (prey with a higher energy value) when patches are far away – i.e. switch between unselectivity and selectivity. As a general rule, a rate-maximising forager should be unselective below some critical travel time, but size selectivity increases with distance from the central place. Orians and Pearson (1979) gave a general principle for rate maximising prey models: "For short travel times, superiority of prey hinges on energy per unit handling time. For long travel times, superior prey are those of higher energy, regardless of handling time."

For simplicity, as foraging theory is just one background aspect of my study rather than the main thrust of it, the above information was taken almost in entirety from Stephens and Krebs 1986. However, this book is in itself in part a review and consolidation of many prior studies of foraging theory, and as such more extensive detail can be found in Schoener 1971, Emlen 1973, Pulliam 1974, Werner & Hall 1974, Charnov 1976a & b and others.

Aims of Thesis

In an attempt to understand more about the autecology of Kestrels on farmland and to develop potential AESs, this Natural England funded project looked at which habitats were selected for foraging during the breeding season by Kestrels, and the spatial and temporal responses of small mammals and other farmland birds to vegetation management. A 2009 study by Douglas *et al*. showed that cutting strips or patches into arable field margins can benefit foraging by breeding Yellowhammers *Emberiza citrinella* at certain times of the year. I wanted to investigate whether this process could also benefit foraging Kestrels, for whom it has been suggested that mowing may have a positive effect on their distribution (Aschwanden *et al*. 2005, Whittingham & Devereaux 2008). I aimed to develop realistic recommendations for uncropped vegetation management regimes as part of agri-environment measures, which will ensure there is always some level of cover for small mammals, invertebrates and other vital prey items in AES field margins, but making sure they are accessible to the foraging Kestrels. This should balance availability and accessibility, and ensure a more constant supply of prey for the birds on farmland. It is expected that this will involve greater heterogeneity in grass margins (both spatially and temporally), which as a knock-on effect should have benefits for a wide range of farmland biodiversity. The principles of this study will be applicable across a broad range of species and ecosystems.

Thesis Outline

Figure 1.1 overleaf shows a conceptual model of the ecology of the Kestrel, and some of the different biotic and abiotic factors which can impact its productivity and survival, and therefore overall population growth or decline. This thesis concentrated on the influence of habitat on Kestrel population dynamics, via interactions with prey; i.e. the green boxes in the presented flow diagram.

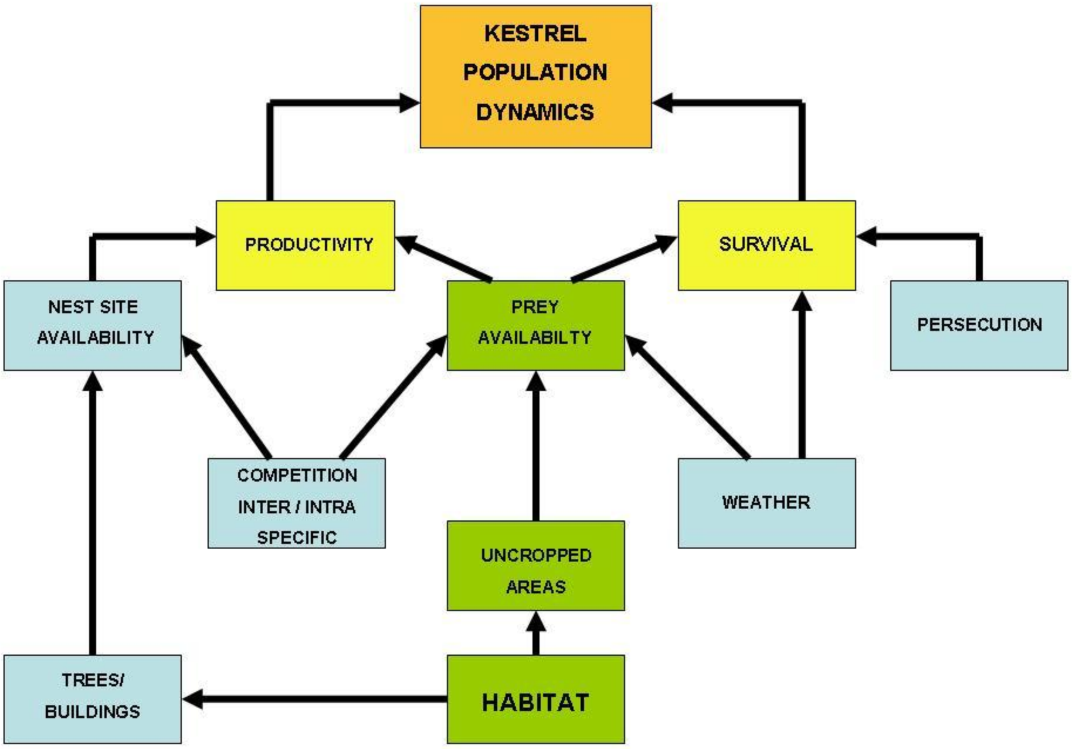

Figure 1.1. Conceptual model of the factors influencing Kestrel population dynamics. The focus of this thesis is the pathway highlighted in green.

Chapter 2 looks at foraging habitat selection by breeding Kestrels on lowland farmland in England, using vantage point observations of Kestrel pairs during the breeding season, over a period of three years at six different sites in northeast England. Which habitats are most frequently selected for hunting are analysed to give a broad idea of which habitats are preferred by Kestrels for hunting. Shorter grass – specifically that which had been recently cut - was the most preferred habitat and so I went on in future chapters to investigate how management of grassland would affect not only Kestrels but other farmland bird species (which would be affected if AESs designed to change vegetation structure were implemented) as well as the key prey of the Kestrel. In Chapter 3 I investigate the spatial and temporal effects of grass cutting on movements of small mammals, using live trapping and vegetation manipulation with agricultural machinery. Chapter 4 then looks at the temporal patterns of use by birds in general, of recently cut grass fields (the most selected habitat in chapter 2) on farmland. Some species are preferentially drawn to recently mown grass/pasture, we look at why this is, and analyse how long the presumed beneficial effects last. Data were gathered using transect and vantage point surveys, both before and after cutting operations at 33 grass fields in northern England. Chapter 5 then tests a possible AES recommendation for grass margins, to see what the likely effects will be on other farmland birds which preferentially use this habitat on farms. The final chapter reviews and discusses the key findings of the thesis, and provides recommendations for adjustments to the current management of uncropped margins on AES farms, to increase their usefulness to a range of birds, but in particular raptors.

The data for Chapter 2 were collected by a range of fieldworkers over three years prior to my PhD. I collated the data, analysed it and wrote up the chapter (and published paper that resulted from this chapter). Data for Chapter 4 were collected by a Masters student that I co-supervised. I helped with the statistical analysis and re-wrote her write-up into a paper format for publication (and for this thesis chapter). I collected the data, analysed it and wrote up the results from Chapters 3 and 5.

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Chapter 2: Foraging Habitat Selection by Breeding Common Kestrels *Falco tinnunculus* on Lowland Farmland in England

Claudia M Garratt [1], Martin Hughes [2], Giselle Eagle [3], Tanya Fowler [4], Philip V Grice [5] and Mark J Whittingham [1]

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[1] Newcastle University, School of Biology, Ridley Building, Claremont Road, Newcastle, NE1 7RU
[2] Resting Hill, 5 Birches Nook Road, Stocksfield, Northumberland, NE43 7PA
[3] WWT Washington Wetland Centre, Pattinson, Washington, Tyne and Wear, NE38 8LE.
[4] 37 Western Way, Darras Hall, Newcastle upon Tyne, NE20 9AS
[5] Natural England, Northminster House, Peterborough, PE11UA

Short title: Habitat selection by breeding Kestrels.

Abstract

Capsule Breeding Common Kestrels prefer to forage over recently cut grass than all other habitat types on farmland.

Aims To identify foraging habitat and prey of Common Kestrels during the breeding season.

Methods We observed seven pairs of Common Kestrels during the breeding season over three years, using fixed vantage point observations. We recorded foraging attempts, and habitat and prey data for where the birds chose to forage. Compositional Analysis was used to establish use of each habitat category relative to that habitat's availability within the Kestrels' observed foraging ranges.

Results We found that Kestrels select habitats non-randomly, with cut grass (≤ 5cm, all cut less than two weeks previously) being the most used relative to availability. Prey taken varied with grass height: the ratio of mammals to invertebrates was greater on cut grass (4.36 mammals: 1 invertebrate) than on longer, uncut grass swards (1.73 mammals: 1 invertebrate).

Conclusion Our results highlight the importance of areas of long and short grass in close juxtaposition, to provide conditions suitable for prey and access to them, respectively. The creation and maintenance of such small-scale habitat heterogeneity will be the key to maximising the benefits of English agri-environment schemes for Common Kestrels.

Introduction

The populations of many bird species which live primarily on European farmland (‘farmland birds’) have declined markedly over the past four decades, primarily as a consequence of agricultural intensification (Krebs *et al*. 1999, Donald *et al*. 2001, Wilson *et al*. 2009). The recovery of farmland bird populations has, therefore, emerged as a key objective for biodiversity conservation at both a UK and European level and reversing the decline in farmland birds in England even featured as a governmental target for ten years, up to June 2010. Progress with this target was measured by the Farmland Bird Index (FaBI); a composite index made-up from the breeding population trends of 19 widespread farmland bird species, one of which is the Common Kestrel *Falco tinnunculus* (hereafter referred to as Kestrel). In 2009, the FaBI for England was at just 47% of its 1970 level (Defra 2010). The principal means for reversing the decline in farmland birds is Agri-environment Schemes (AESs), such as Environmental Stewardship (Natural England 2010 a, b). This paper focuses on understanding the habitat requirements of Kestrels on English farmland to help to inform the appropriate design and deployment of AESs.

Kestrels in the UK are largely reliant on farmland (Village 1990). Their UK populations have declined by one fifth since 1994 (Risely *et al.* 2010), and the combined results of bird monitoring programmes from across Europe suggest that numbers fell as a whole by nearly one third during the period 1990-2006 (Pan-European Common Bird Monitoring Scheme 2009). The species is, therefore, regarded as a conservation concern in both a UK and European context (Eaton *et al.* 2009), and it has been identified amongst a suite of 14 bird species that should be considered the most important targets for restorative action on lowland farmland in England (Vickery *et al.* 2004).

Kestrels prefer to forage on grassland rather than other farmland habitats (Aschwanden *et al*. 2005). AESs have resulted in large amounts of potentially suitable Kestrel foraging habitat, in the form of grass margins and arable buffer strips, being created in the UK countryside: in England, for example, approaching 25,000 ha of grass margins had been created as part of AES agreements by March 2005 (Grice *et al.* 2007). The subsequent launch of Environmental Stewardship in 2005 resulted in a further *c.* 37,000 ha of grass margins and buffer strips being built up progressively over the following five years (although some of the area within the predecessor AESs will have been lost or transferred into Environmental Stewardship). Despite the large uptake of this particular AES option (Davey *et al.* 2010) and, therefore, the increase in potentially suitable foraging habitat, the latest British Trust for Ornithology (BTO) Breeding Bird Survey (BBS) results suggest that while Kestrel populations have started to stabilise in the UK since the mid 1990s, they have failed to recover to pre-decline levels (BTO 2009), and they underwent a further 36% decline from 2008 to 2009 (Risely *et al.* 2010).

Kestrels are visually guided, diurnal hunters (Village 1990, Aschwanden *et al.* 2005) and foraging theory predicts that it is not prey abundance but availability that determines where predators hunt (Stephens & Krebs 1986). So longer (or more dense) grassland that conceals prey may be of lower quality for Kestrels than shorter swards, even if prey abundance is higher in the longer and denser grassland (Arlettaz *et al.* 2010). A recent UK study found little relationship between prey abundance and bird distribution in grassland, and found that instead, sward height seemed to be one of the most important factors governing where some species of birds chose to forage (Atkinson *et al*. 2005, Whittingham & Devereux 2008). More specifically, two studies suggested that mown grassland was preferred by foraging Kestrels (Aschwanden *et al*. 2005, Whittingham & Devereux 2008). However, few studies have fully explored the foraging selection made by Kestrels with regard to various habitat types, and we here build on previous findings (Aschwanden *et al.* 2005) by increasing sample size and specifically investigating prey captures within different habitat types. We also focus on a known number of Kestrel pairs during the breeding season (as opposed to wintering birds [Whittingham & Devereux 2008]) and on collecting data on all habitats visited around the nest.

We studied the breeding season foraging behaviour of Kestrels at six sites in northern England over a period of three years, aiming to quantify the relative importance of grassland to other habitats for foraging Kestrels. In addition, by recording differences in selection of cut and uncut grassland, as well as different prey types taken in different habitats, we also studied the importance of grassland management for these birds.

Methods

Study sites

We observed seven breeding pairs of Kestrels between April and August in 2006, 2007 and 2008, at six different sites in northeast England. Three sites were in Northumberland (Ordnance Survey grid referencess: NY9681, NZ0765, NZ0066), two sites in Tyne & Wear (both NZ1758) and two pairs from the same nest site (one in 2006 and one in 2008) in County Durham (NZ3248). All pairs used natural nest-sites, except at the latter site, which used a nestbox.

Pairs 1 and 7 (the only pairs to use a nestbox) were sited on a nature reserve, consisting of a series of interlinked ponds and scrapes, newly re-seeded grasslands and wildflower meadows, and a larger area of young (about 15 years old) native woodland. The area surrounding the nature reserve was a mixture of farmland (a mixture of arable and grazing – mainly cattle), industrial units and housing. Pair 2 was in an upland livestock area - predominantly sheep - with the majority of the surrounding habitat made up of a mixture of improved grassland and semi-natural grazing. Pair 3 nested in a small wooded dene (small steep-sided valley), surrounded by arable farmland. Pairs 4 and 5 were sited at a 160-ha 18th-century landscaped forested garden, surrounded by arable farmland, human habitation, and woodland. Pair 6 nested on a ruined castle backed by a large broadleaved dene, and the surrounding area was mixed farmland.

Habitat measurements

Habitat types and availability were quantified in a radius around each nest site. Initially these habitats were classified immediately (i.e. one or two days) before the first observation of the foraging behaviour. This was carried out on radius of up to 1.5 km surrounding the nest site. The actual availability for each pair was determined by the farthest observed foraging trip made by that pair (x); thus x was used as the radius for the circle around the nest in the Compositional Analysis (see below). Habitats were classified visually into five broad categories according to what they represent to a foraging Kestrel. These categories were:

1. **Uncut grass** – comprising short grass (<20 cm), long grass (>20 cm), meadow (>50 cm), field margin and rough buffer categories.

2. **Cut grass** – comprising recently cut (within one week) hay crops, and mowed meadow strip categories. All about 5 cm or less in height.
3. **Arable use** – comprising arable crops and ploughed fields.
4. **Woodland** – comprising conifer, broadleaf and mixed woodland, and young plantation with trees up to a maximum of 4.5 m high.
5. **Other** – comprising water, reed swamp, anthropogenic features and unknown categories.

Within the grassland habitats, grass height was estimated by eye and where necessary confirmed using the drop disk method (Stewart *et al.* 2001), allowing us to distinguish between cut, short (<20 cm), long (>20 cm) and meadow (>50 cm) grass. Cut grass was always ≤5 cm. Acetate overlays and Romer cards laid over 1:10,000 scale Ordnance Survey maps were used to calculate the total area in hectares each habitat type comprised within the radius around each nest and, therefore, the availability of each habitat type within the Kestrels' observed usual foraging range. Each time the recorder made Kestrel foraging observations they noted any major changes to previously categorised habitats (e.g. silage cutting, crop harvesting). Habitat availability used in the analysis was calculated as the mean of the percentage of each habitat category across observations.

Foraging behaviour

Foraging behaviour of adults at each site was quantified by observations from fixed vantage points, several hundred metres away from the nest, for between 6.5 and 8 hours per observation session (mean number of hours observation per nest = 67.1 [range 50-80 hours] – a total of 470 hours across all seven nests), by different observers in each of the three years. Where the Kestrels flew out of sight they were also followed by car where possible. Foraging data were collected in 2006 (Pair 7) for 50 hours over ten days in June, in 2007 data were collected from 80 hours of observation per nest between 14 May and 31 July (Pairs 4-6) and in 2008 were collected from 80 hours of observation per nest from 15 June to 31 July (Pairs 1-3). Every hunting attempt made by the breeding pair over each of the habitat types during the observation period was recorded. A hunting attempt was defined as a bird making an attempt to capture prey (usually by landing on the ground), hovering over a habitat was not classed as an attempt. The outcome of these hunting attempts varied (successful or unsuccessful), and where successful we attempted to record the prey taken (bird, mammal, or invertebrate) but this was not always possible. The prey taken was not recorded for one nest where it was difficult to obtain close enough views of the nest and the surrounding habitats to identify any prey with any reliability.

Statistical analysis

Data were analysed using statistical software R (version 2.10.0) (R Development Core Team 2009). We analysed the habitat data using a Compositional Analysis (package *adehabitat*, [Calenge 2006]), which compares use of habitat categories relative to their proportional availability. This enabled us to determined two levels of habitat selection by foraging Kestrels. First, we determined whether Kestrels used all the available habitats surrounding each nest randomly with an overall test statistic (Wilks' Lambda). Second, we went on to compare the selection (i.e. use divided by availability) of each of the five different habitats with each other to rank the relative selection of these habitats by our Kestrel pairs. We used a G-test for goodness-of-fit to investigate whether prey captured was random with respect to habitat. We carried out three G-tests; one for all prey summed together and then separate tests for invertebrates and mammals (note 0.01 was added to all values to permit the G-test calculation due to some observed values being equal to zero). The data for birds were included in the 'all prey' category, but excluded from individual G-test analysis due to very small sample size. For all tests, a significance level of 0.05 was used.

Results

Habitat

The size of foraging area recorded for each Kestrel pair ranged from 78.54 ha for Pair 7, to 708.64 ha for Pair 2 (Appendix 2.A). For all seven pairs, uncut grass formed by far the largest proportion of the grassland in their foraging area, with Pair 2 having 87.96% uncut grass compared with just 0.99% cut grass. The largest proportion of foraging area for two of the pairs (Pair 3 and Pair 6) was made up of arable land, at 62.8% and 43.59% respectively, and Pair 4's site comprised more woodland (45.84%) than any other habitat type. Pair 7 had a considerably higher proportion of 'other' category habitat within their usual foraging range than all the other pairs. This is because Pair 7's recorded foraging range out from the central point of the nest was relatively small, and the nest was located on the edge of a large body of water, next to an industrial estate.

Foraging behaviour

Compositional Analysis demonstrated that Kestrels showed non-random use of foraging habitats surrounding their nests (randomisation test; Wilks' Lambda = 0.026, $P = 0.02$) (details of foraging for each pair given in Appendix 2.B). Relative to availability, the most preferred habitat was cut grass followed by uncut grass and woodland, and then both arable and 'other' were used far less than was available (Figs. 2.1 & 2.2, Table 2.1).

Table 2.1. The ranking of the five different habitats against each other.

	Uncut grass	Cut grass	Arable	Woodland	Other	Rank
Uncut grass	0	−−−	+	+	+++	2
Cut grass	+++	0	+++	+++	+++	1
Arable	−	−−−	0	−	+++	2
Woodland	−	−−−	+	0	+++	2
Other	−−−	−−−	−−−	−−−	0	5

Cut grass was significantly preferred to all other habitats, while all other habitats were significantly preferred to 'other'. The rank scores were derived by adding together the significant scores, e.g. cut grass scored '+++' four times, thus a score of + 4 (and rank of 1) because it was significantly preferred to all four other habitats, whereas uncut grass was significantly preferred to one habitat and significantly avoided relative to one habitat so it received an overall score of zero, similar to two other habitats which were thus all given a rank of 2 jointly.

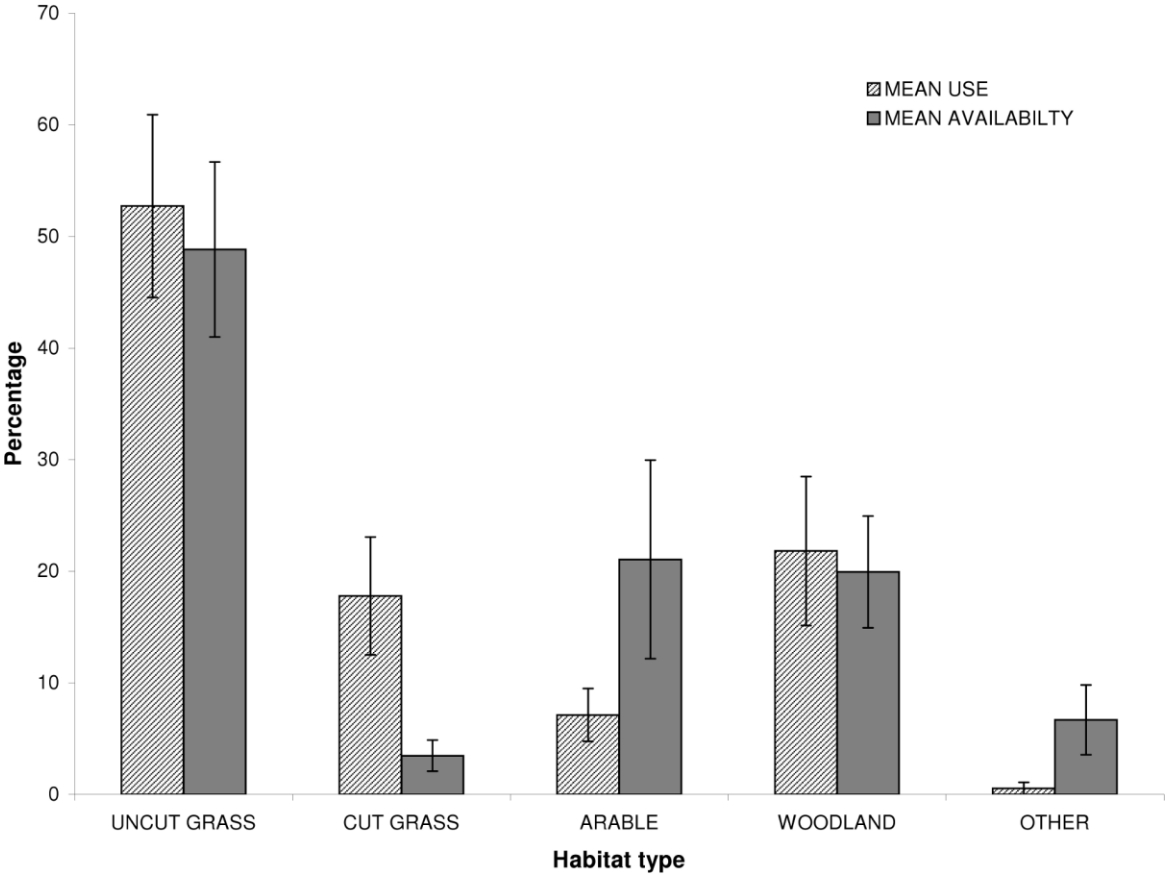

Figure 2.1. Comparison of mean habitat use (mean of proportion of forages in each habitat type by each Kestrel pair) and mean habitat availability (mean proportion of habitat type available in each Kestrels' home range) averaged across the seven pairs. For example, on average across the seven pairs 56% of foraging attempts were made in uncut grass and uncut grass comprised 48% of the 'available' habitats surrounding the nests. Cut grass is clearly the habitat selected most relative to its availability.

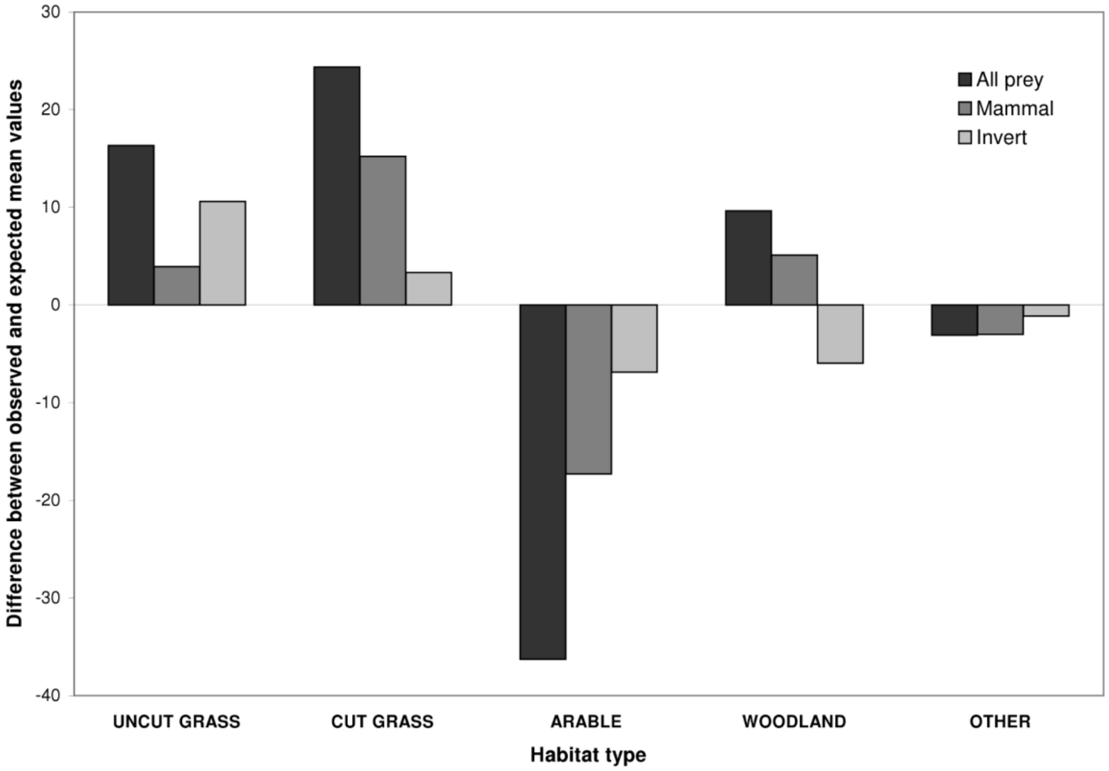

Figure 2.2. The difference between expected and observed prey captures, for different identified prey types across the five habitat categories. A positive value indicates that more prey were captured in that habitat type than would be expected by chance, e.g. if 100 mammals were observed to be captured across all habitats and habitat 'A' comprised 10% of the habitat then the number expected in that habitat would be 10 but the observed may be 20 indicating more mammals were captured in that habitat than would be expected. The figure shows that grassland was the most important habitat type for both mammals and invertebrates. For the purposes of this analysis, birds were excluded from individual analysis as the sample size was too small, but they are included in the 'all prey' category.

Prey

The total numbers of prey items captured by each of the six pairs during observation periods were as follows: 22, 40, 24, 38, 22 and 28. Thus, prey captures were distributed reasonably equally across the six pairs, and any patterns reported are not driven disproportionately by a subset of pairs. All observed prey captures were non-randomly distributed with respect to habitat availability (G-test statistic with 4 df in each case; mammals: $G = 90.80$; invertebrates: $G = 41.21$; all prey summed together: $G = 137.88$, $P < 0.001$ in all cases). The difference between observed and expected values for the prey taken in each habitat type (Fig. 2.2) shows that more prey were captured in grassland (both in uncut and cut grass) and woodland than would be expected from these habitat types' availability. The difference between observed and expected prey captures shows that the ratio of mammals to invertebrates taken changes with habitat type. In cut grass, 4.36 mammals were taken for each invertebrate, whereas in the longer, uncut grass, the ratio was 1.73 mammals to each invertebrate (Table 2.2).

Table 2.2. Captures of different types of prey in the different habitats, summed across all six pairs.

	Prey type				
Habitat type	Bird	Invert	Mammal	Unclear	Total
Uncut Grass	2 (2%)	26 (26%)	45 (45%)	27 (27%)	100
Cut Grass	4 (14%)	4 (14%)	17 (61%)	3 (11%)	28
Arable	0 (0%)	0 (0%)	1 (100%)	0 (0%)	1
Woodland	10 (24%)	0 (0%)	21 (50%)	11 (26%)	42
Other	0 (0%)	0 (0%)	0 (0%)	3 (100%)	3
Total	16 (9%)	30 (17%)	84 (48%)	44 (26%)	174

The proportions of each type of prey taken in each habitat are shown in parentheses. From this, we can see that invertebrates are more prevalent as a prey item in uncut grass than they are in any other habitat type, whereas birds form a higher proportion of the total prey taken in woodland (24%) than they do in either grassland category.

Discussion

Breeding Kestrels in our study significantly preferred to forage over cut grass than all other habitats available to them, despite this habitat type generally making up a relatively small proportion of their observed foraging range (mean = 3.5%). For four of the pairs, grassland of any kind was not the habitat most available to them in the immediate foraging area around their nest (Appendix 2.A). Thus, Kestrels select foraging habitat non-randomly, and seem to travel to areas where accessibility to ‘higher value’ prey is enhanced. However, it should also be noted that it is difficult to see foraging birds in woodland, and so the number of hunting attempts in this habitat may be underrepresented. The prey-type analysis (Fig. 2.2, Table 2.2) showed that in longer, uncut grass, invertebrates are more prevalent as a prey item than they are in any of the other habitat types, whereas in cut grass the difference between observed and expected prey captures is more strongly driven by mammals. However, owing to the relatively small sample size, both of prey captures and numbers of Kestrel pairs, these results may not be robust and further study is needed.

Why does grass height influence the ratio of invertebrates to mammals? Many types of invertebrate prefer an uncut sward and there is usually an abundance of arthropods found in the ‘field layer’ at the top of the sward (Morris 2000), presumably making this the easier prey item for foraging Kestrels to take in areas of longer uncut grass. However, shorter swards are likely to enhance access to small mammals, such as voles. The selection of shorter, recently cut swards may be explained by the energy gained from consuming invertebrates versus small mammals: many invertebrates will have to be caught and consumed to provide the same nutritional value as one small mammal (Rudolph 1982, Korpimäki 1984, Village 1990, Wiebe & Bortolotti 1994). The decline of Kestrels on farmland has been largely attributed to the adverse effects of agricultural intensification on their foraging habitats and on populations of small mammals, particularly voles *Microtus* spp. which are their major prey (Aschwanden *et al*. 2005, 2007).

All of the cut grass category within our study was agricultural sward that had been shortened by mowing. In all cases this will have been within the preceding two weeks, as observation visits to individual nests were never more than two weeks apart. Owing to high regrowth rates at the latitude of this study and at this time of year (rates of 60 – 100 kg dry matter/ha/day in May, and 50 - 80 kg dry matter/ha/day from June to August [Rural NI 2010]) it is likely that cut grass functionally becomes short grass relatively rapidly. For Common Starlings *Sturnus vulgaris,* it has been shown that capture efficiency of their soil-dwelling invertebrate prey was 71% greater on newly (within one hour) mown sward, than on sward mown to the same height 48 hours previously, although they did not spend any more time foraging on the former compared with the latter (Devereux *et al.* 2006). We expect that the effects of such changes in micro-habitat caused by cutting would be analogous for Kestrel prey species, and that generally the advantages of freshly cut grass may be quite fleeting. However, although we do not have exact dates for the cutting operations, all cut grass was still only a height of 5 cm or less, and the time that had elapsed since cutting (within the two-week timeframe) did not appear to affect the foraging Kestrels’ selection of this habitat. It seems probable that the main initial attraction of freshly cut grass to foraging birds is due to disturbance effects caused by the cutting itself - the flushing out and sudden exposure of both foliar invertebrates (Vickery *et al*. 2001) and small mammals

(Hansson 1977, Village 1990), both of which will be taken by Kestrels. Kestrels will also take earthworms, which generally surface in response to seismic vibrations in the soil (Kaufmann 1986, Mitra *et al*. 2009) and are, therefore, likely to be drawn to the surface by the noise and vibration caused by agricultural mowing. It is probable, however, that prey which is initially exposed when long sward is cut will, over time, disperse to areas where conditions are more favourable. Voles generally show a strong preference for longer, more dense swards (Jacob & Brown 2000, Arlettaz *et al.* 2010), and Edge *et al.* (1995) found that vole densities decline by approximately 50% after cutting, regardless of initial densities. This would imply that the benefits for Kestrels of cutting operations would be limited and relatively short lived. However, it has also been shown that some resident adult Common Voles *Microtus arvalis* do not leave their territory, even after substantial alteration to the habitat caused by mowing or harvesting (Jacob & Hempel 2003), which would mean that targeted cutting of patches or strips of grass margins could be a useful tool for increasing accessibility to voles for Kestrels whilst not necessarily overly affecting their abundance.

Although the requirements of birds on farmland are often very specific, conflicting requirements are rare (Vickery *et al.* 2004), but they do occur. With their selection of cut sward for hunting over, Kestrels may be one example for whom some of their breeding season requirements come into direct conflict with the requirements of birds for which some current management prescriptions for grass margins are designed. AES margins are currently not permitted to be cut until mid July (Natural England 2010a, b, Vickery *et al.* 2004), after the breeding season is over, owing to the possible presence of ground nesting birds. However, if cut grass patches are valuable for Kestrels as would appear to be the case, then during the breeding season is when improved accessibility to their small mammal prey would be of the most benefit for productivity, which has been put forward as the probable most limiting demographic factor for Kestrels (Vickery *et al.* 2004). One potential option is to manage grass margins with both long and short swards in close proximity to benefit both invertebrate and mammal populations, whilst enhancing accessibility for their predators. This has been shown to be beneficial to foraging Yellowhammers *Emberiza citrinella* on farmland (Douglas *et al.* 2009) and work is on-going to investigate its potential benefits for Kestrels.

Conclusions

Grassland requires careful management to provide both conditions suitable for invertebrates and mammals (i.e. longer grass swards), and accessibility for Kestrels (shorter swards). This work shows that habitat heterogeneity at small spatial scales is important and adds to the range of studies highlighting habitat heterogeneity as a key feature of habitat management at a range of scales (e.g. Benton *et al*. 2003, Wilson *et al*. 2005, Whittingham *et al*. 2007). We plan future work to focus on the details of the timing and extent of grass management to benefit Kestrels, to inform the future design and deployment of AESs.

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Appendix 2.A. Area of each habitat type and the proportion of the usual foraging range which this comprises for each kestrel pair.

	Pair 1		**Pair 2**		**Pair 3**		**Pair 4**		**Pair 5**		**Pair 6**		**Pair 7**	
Habitat	**Area (ha)**	**% Foraging Range**	**Area (ha)**	**% Foraging Range**	**Area (ha)**	**% Foraging Range**	**Area (ha)**	**% Foraging Range**	**Area (ha)**	**% Foraging Range**	**Area (ha)**	**% Foraging Range**	**Area (ha)**	**% Foraging Range**
Uncut grass	166.32	59.26	623.32	87.96	101.68	27.11	55.92	42.47	60.04	55.20	40.00	36.00	26.592	33.858
Cut grass	15.00	5.34	7.04	0.99	0.92	0.25	3.40	2.58	2.76	2.54	1.80	1.62	8.658	11.024
Arable	42.04	14.98	0.00	0.00	235.56	62.80	0.00	0.00	17.20	15.81	48.44	43.59	8.040	10.236
Woodland	50.04	17.83	59.52	8.40	25.04	6.68	60.36	45.84	28.28	26.00	15.96	14.36	16.079	20.472
Other	7.24	2.58	18.76	2.65	11.92	3.18	12.00	9.11	0.48	0.44	4.92	4.43	19.171	24.409
Total	**280.64**	**100.00**	**708.64**	**100.00**	**375.12**	**100.00**	**131.68**	**100.00**	**108.76**	**100.00**	**111.12**	**100.00**	**78.540**	**100.000**

Appendix 2.B. The number of foraging attempts observed in each habitat type for each kestrel pair, and the proportion of each pair's total foraging attempts which occurred in each habitat type.

	Pair 1		**Pair 2**		**Pair 3**		**Pair 4**		**Pair 5**		**Pair 6**		**Pair 7**	
Habitat	**Number of Forages**	**% Forages**	**Number of Forages**	**% Forages**	**Number of Forages**	**% Forages**	**Number of Forages**	**% Forages**	**Number of Forages**	**% Forages**	**Number of Forages**	**% Forages**	**Number of Forages**	**% Forages**
Uncut grass	52.00	55.32	109.00	77.86	63.00	85.14	13.00	25.00	16.00	43.24	17.00	42.50	4.00	40.00
Cut grass	18.00	19.15	22.00	15.71	3.00	4.05	17.00	32.69	2.00	5.41	3.00	7.50	4.00	40.00
Arable	15.00	15.96	0.00	0.00	4.00	5.41	0.00	0.00	5.00	13.51	2.00	5.00	1.00	10.00
Woodland	9.00	9.57	9.00	6.43	4.00	5.41	20.00	38.46	14.00	37.84	18.00	45.00	1.00	10.00
Other	0.00	0.00	0.00	0.00	0.00	0.00	2.00	3.85	0.00	0.00	0.00	0.00	0.00	0.00
Total	**94.00**	**100.00**	**140.00**	**100.00**	**74.00**	**100.00**	**52.00**	**100.00**	**37.00**	**100.00**	**40.00**	**100.00**	**10.00**	**100.00**

Chapter 3: Should We Stay or Should We Go Now? What Happens to Small Mammals When Grass is Mown, and the Implications for Birds of Prey.

Claudia M. Garratt[1], Jeroen Minderman[2] & Mark J. Whittingham[1]

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[1]School of Biology, Newcastle University, Newcastle upon Tyne, UK

[2]Institute of Biological and Environmental Sciences, University of Stirling, Stirling, FK9 4LA

Short title: Grass cutting and small mammals

NB: An appendix has been added (Appendix 3.C) showing Likelihood Ratio Tests for the models reported in this chapter.

Abstract

Capsule Some small mammals continue to use areas of grassland after cover is reduced by mowing.

Aims To find out if some small mammals will continue to use areas of grassland after a) the vegetation is cut, and b) grass cuttings are removed.

Methods We live-trapped five grassland plots on a nature reserve in northern England then halved each plot and cut the grass on the treatment half, before trapping again. The grass cuttings were then removed, and a final trapping session carried out. The data were then modelled using Generalized Linear Mixed Models (GLMMs).

Results Although mowing grass causes a rapid decline in small mammal captures, a small proportion of captures (20-27%) still occurred in patches of mown grass immediately after cutting. This proportion more than halved again when the cut grass that was initially left *in situ* was removed.

Conclusions We (1) conclude that although grass cutting may decrease the presence of small mammals, a minority remained in cut areas provided some form of cover was present, (2) suggest that leaving cut grass in situ may effectively provide such cover. These findings are discussed in light of Agri-environment Scheme (AES) options (e.g. grass margins) that could be improved further to benefit birds of prey. In line with previous studies, we suggest that creating mosaics of shorter grass near to cover – e.g. long grass or woodland/hedges - is likely to provide accessible food for predators of small mammals but also maintain healthy small mammal populations.

Introduction

Agriculture is the largest driver of extinction risk in birds (Green *et al.* 2005). European Agri-environment Schemes (AESs) are designed, in part, to benefit biodiversity on farms (Kleijn *et al.* 2006, Whittingham 2007) and have existed since 1987 in England (Ovenden *et al.* 1998). Some AESs have substantially benefitted biodiversity (e.g. Peach *et al.* 2001**,** Albrecht *et al.* 2007, Perkins *et al.* in press) although some taxa benefit more than others (e.g. Aebischer *et al.* 2000, Bradbury & Allen 2003, Kleijn *et al.* 2006) and some schemes have shown no demonstrable benefit to target species or overall biodiversity at all (e.g. Kleijn *et al.* 2001, Kleijn *et al.* 2004, Davey *et al.* 2010). Recent work has suggested that adaptive management is a key tool to improve the biodiversity benefits from AESs (Perkins *et al.* in press). Adaptive management involves ‘learning through doing’ (Convention on Biological Diversity 2004) and thus encourages, in this context, modification of AESs to improve their benefit for wildlife as new information comes to light.

Within the English AESs grass margins have proven to be one of the most popular management options (Davey *et al.* 2010), and this has led to an increase of *c.* 62 000 ha in the area of grassland on farms in England since the mid 80s (Grice *et al.* 2007, Garratt *et al.* 2011). If wide and densely vegetated enough, grass margins should enhance numbers of small mammals on arable farmland (Shore *et al.* 2005, Aschwanden *et al.* 2007), a key resource for Kestrels *Falco tinnunculus* L. (Cavé 1968, Village 1990) and many other birds of prey (Glue 1974, Baker & Brooks 1981, Korpimaki 1984**,** Korpimäki & Norrdahl 1991, Redpath & Thirgood 1999, Koks *et al.* 2007). However, along with other raptors (e.g. Barn Owl *Tyto alba*, Risely *et al.* 2010) Kestrel numbers are still declining (Risely *et al.* 2010, Garratt *et al.* 2011). These continued declines may indicate that the quality of grass margins in AES can be improved to further benefit raptors that rely on small mammals. In this study, we wished to explore possibilities for doing this.

The balance between availability and accessibility of prey determines the quality of a foraging area (Stephens & Krebs 1986). Previous work has shown that Kestrels foraging on farmland prefer short grass above all other habitats available to them (Ashwanden *et al.* 2005, Whittingham & Devereux 2008, Garratt *et al.* 2011) presumably as it increases accessibility to their prey. The same has also been shown for Long-eared Owls *Asio otus* (Aschwanden *et al.* 2005), Rough-legged Buzzards *Buteo* *lagopus* and Red-tailed Hawks *Buteo jamaicensis* (Baker & Brooks 1981). Thus the value of grassland to foraging raptors may be determined to a large extent by the length of the vegetation, suggesting that cutting or mowing could represent a beneficial management option.

However, while cutting grass may improve access to small mammal prey for raptors, evidence suggests that small mammals prefer longer, dense grass or more densely vegetated habitats generally, with evidence existing of such associations for a range of species: e.g. Field Voles *Microtus* spp, (Birney *et al.* 1976, Hansson 1977, Bellamy *et al.* 2000, Tattersall *et al.* 2000), Bank Voles *Myodes glareolus* Schreber (formerly *Clethrionomys glareolus*) (Kikkawa 1964, Dickman & Doncaster 1987, Shore *et al.* 2005, Butet *et al.* 2006) and Common Shrews *Sorex araneus* L. (Dickman & Doncaster 1987, Butet *et al.* 2006, Aschwanden *et al.* 2007), although this association is less strong for shrews which are fairly ubiquitous (Churchfield *et al.* 1997). Some studies suggest that grass which is mowed regularly probably only provides a temporary small mammal habitat (Lemen & Clausen 1984, Sheffield *et al.* 2001, Edge *et al.* 1995), and mowing appears to cause “whole population" movements out of the habitat in the case of *Microtus agrestis* L. (Hansson 1977). However, other studies suggest either that small mammals may not be strongly affected by mowing (e.g. Jacob 2003, Jacob & Hempel 2003) or that some subdominant individuals remain in the cut grass (Meunier *et al.* 1999, Jacob & Brown 2000).

Thus, in spite of previous work on the subject, the effect of cutting grass on the distribution of small mammals remains unclear. Clearly, the value of cutting or mowing of grassland as a management tool to benefit foraging raptors depends on fully understanding its effect on small mammals. In this paper, we present the results of a field experiment in which we manipulate the height of grass patches by cutting. Using live trapping, we test both the temporal and spatial effect of cutting on small mammal distribution. Ultimately we aim to provide recommendations for the management of grassland to improve habitat suitability for declining raptor species that rely on small mammals.

Materials and Methods

Study site

The experimental plots were located in a grassland nature reserve in County Durham, northeast England (54° N 49’ 52.7’’ -1°W 29’ 44’’). The reserve primarily consists of immature, artificial grassland on the site of a former opencast coal mine, containing areas of newly reseeded grasslands and immature native species woodlands planted from 1996 onwards. The grassland is dominated by Ryegrass *Lolium* spp., and wildflowers such as Clover *Trifolium* spp., vetches *Vicia* spp., Birdsfoot Trefoil *Lotus corniculatus* L. and Yellow Rattle *Rhinanthus minor* L.

There were five experimental plots within the site; the closest two were 10 m apart and separated by a tarmac path and a ditch, the others were all separated by a minimum of 93.6 m. Three of the experimental plots were small, fenced, managed meadows (two were cattle grazed over winter, and one was cut each year for hay). The other two were in an unfenced area of rough grassland which had undergone no management for the previous five years, and had only been cut once since being established in 1997. The mean width of plot was 36.96 m (range: 30.4 – 45.2 m) and the mean length 73.61 m (range: 40 – 121.25 m).

Trapping Protocol

Trapping was carried out over nine days in August 2009. Sward height in both the precut/control patches and treatment patches post-cutting was established by choosing ten random sites on each plot and obtaining vegetation height using a tape measure. Mean vegetation height in the experimental plots prior to treatment was 68.3 cm (range 31 – 114.1 cm). Each experimental plot was divided in half (randomly assigned by tossing a coin) to give treatment (to be cut) and control (to be left long), ensuring as far as possible that it was divided in such a way as to make the two halves equivalent in terms of surrounding habitat.

The trap-lines were placed a third of the way between the centre line and the boundary in either direction. Traps were spaced evenly along the trap-line, with the first one being the same distance from the boundary fence as it was from the next trap (Fig. 3.1). Fifty Longworth small mammal traps were put out for each trapping session - ten traps per experimental plot – five in the treatment half, and five in control. Traps contained hay as bedding, and wild bird seed mix, dried mealworms and carrot as bait and moisture source. All traps were set at a sensitivity of 4 g. There was no pre-baiting prior to trapping, as there would be no time to pre-bait after cutting and we wanted the experimental protocol to be the same before and after treatment.

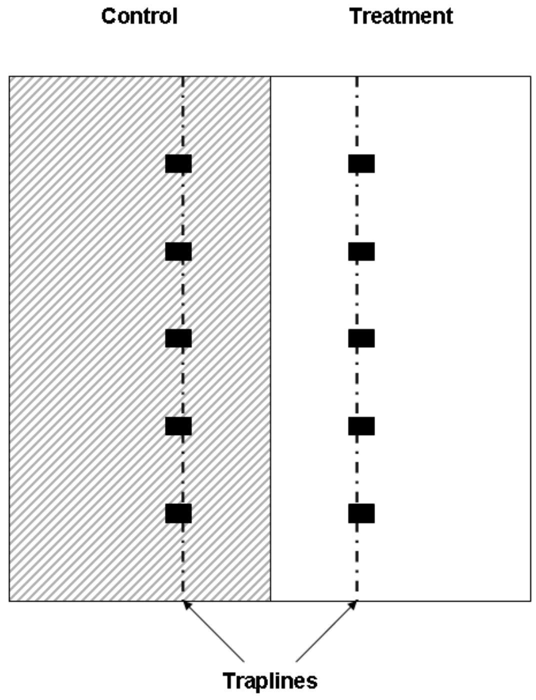

Figure 3.1. Diagram of one trap plot, with traps represented by black rectangles. There were five individual trap plots within the experiment, each containing ten Longworth traps, therefore 50 traps in total; 25 in treatment and 25 in control.

Trapping took place in four sessions before and after experimental grass cutting (Fig. 3.2). Each trapping session consisted of four rounds of trap checks at five hourly intervals:

Session 1: pre-cutting. The traps were placed out at 10:00, with the first check at 15:00, and they were then checked through the night (at 20:00 and 01:00) before being checked and disarmed at 06:00 the following morning. Immediately following the first trapping session the traps were removed and the grass in the treatment halves of the plots was cut to a mean height of 9.3 cm (range 3.9-14.8 cm) using tractor-towed agricultural mowing machinery (other than in plot 4, which was cut by hand and the cut grass immediately harvested - see below). The traps were replaced empty and locked in their original positions immediately following cutting. The cut grass was left *in situ* and used as cover for the traps in all but one of the plots (plot 4).

Session 2: 20h after cutting. The traps were baited and armed at 10:00 the morning following cutting (*c*. 20h after cutting). Traps were then checked at five hourly intervals over the next 24 hours, as before. Following this trapping session, traps were again emptied, locked and left in place.

Session 3: four days after cutting. This session was run as session 2 and was started 48h after finishing the latter, to see if there was a change in the pattern of small mammal captures with time after treatment. After the final trap round at 06:00, the traps were lifted before baling of the cut grass which took place the following day. Traps were then replaced and covered with a small amount of cut grass, and left empty and disarmed.

Session 4: 24h after removal of cut grass. A final trapping session was carried out, commencing at 10:00 on the morning following the removal of the cut grass, to see if the removal of the vegetation had any effect on captures (Fig. 3.2).

To decouple the potentially confounding effects of treatment and time, in one plot (no. 4) the cut grass was immediately collected prior to session 2, rather than left *in situ*. Unfortunately there was insufficient data to test the effects of session using this plot alone, and therefore data from this plot has not been used in analysis of the post-cutting stages.

The experiment was repeated in September 2010, but this time cutting the opposite side of each plot to the one which had been cut in 2009. This was to control for any possible effects of location of the side which had been cut previously. Due to several unforeseen problems, this repeat experiment did not yield sufficient results for analysis and is therefore not considered further, but details can be found in Appendix 3.A.

Statistical analysis

Although traps were checked at five-hour intervals, analyses were carried out on a sub-set of the data using only the data from the 06:00 trap rounds. This approach reduced pseudo-replication in the trapping data; i.e. the individual in trap A could not be the same individual as the one in trap B during a single trap round. Although this does not rule out that the individual could have been present twice in the subsets used to test models 2-4 (see below), the time schedule of the experiment did not allow for individual marking. We used the data from the 06:00 trap rounds because this was the time of day with the largest number of captures across the period of the whole experiment.

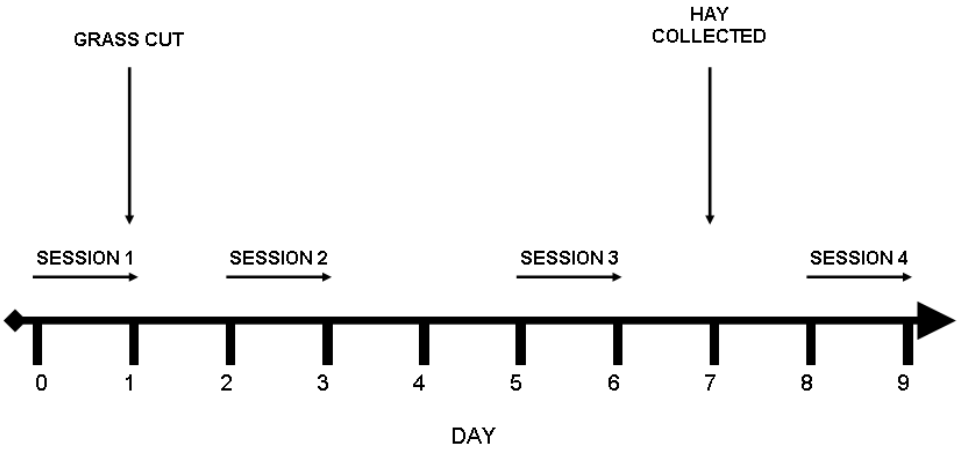

Figure 3.2. Timeline of the experiment, with day zero being the first day on which trapping (session 1) commenced. Models 2, 3 and 4 presented in the results model the changes in capture probabilities between sessions 1& 2 at days 0 and 2, sessions 2 & 3 at days 2 and 5, and sessions 3 & 4 at days 5 and 8.

We used Generalized Linear Mixed Models (GLMMs) to model small mammal presence (probability of capture), with a binomial error structure and a logit link function. Plot was included in all models as a random effect, and treatment (i.e. cut or not cut, hay removed or not removed) and trapping session (see above and Fig. 3.2) were both included as fixed factors. Where appropriate, models included an interaction term between treatment and session, which tests whether the difference in capture probability between treatment and control plots is different between the sessions.

Because we set out to test a number of discrete *a priori* hypotheses, we ran separate models on the subsets of our data, detailed below. All models bar Model 1 (which only models session 1) are pair-wise comparisons of each session with the next consecutive session, including an interaction term for treatment and session.

Model 1: Tests the hypothesis that here is no difference between the probability of capturing small mammals in control and treatment patches prior to grass cutting. This model included data from the first trapping session (pre-cutting) and all five plots used.

Model 2: Tests the hypothesis that there is no effect of grass cutting on capture probability of small mammals. This model included data from session 1 (pre-cutting) and session 2 (post-cutting) and excluded plot 4 (see above). Cut grass was left *in situ* in all of the four plots included in the analysis.

Model 3: Tests the hypothesis that there is no difference in capture probability of small mammals 24h and 96h after cutting. This model included data from session 2 (24h post-cutting) and session 3 (96h post-cutting) and excluded plot 4 (see above). Cut grass was left *in situ* in all four plots included in this model.

Model 4: Tests the hypothesis that there is no effect of the removal of cut grass cover on the probability of capturing small mammals in the treatment plots. This model included data from session 3 (post-cutting, cut grass left *in situ*) and session 4 (post-cutting, cut grass removed) and excluded plot 4.

All analysis were performed using R (version 2.9.1) (R Development Core Team 2009) and the package *lme4* v.0.99875.9 (Bates 2008). We present the parameter estimate (b) ± 1 standard error and the test statistic (z) in the results, and for all tests a significance level of 0.05 was used.

Results

Over all trap rounds, sessions and in both treatment and control plots, we caught a total of N = 264 small mammals. This total was made up of six species of small mammal (see Appendix 3.B). The numbers trapped in the 06:00 periods across the experiment (i.e. data used in the statistical modelling presented below) were as follows: Common Shrew (N=65), Field Vole (N=6), Wood Mouse *Apodemus sylvaticus* L. (N=3), Pygmy Shrew *Sorex minutus* L. (N=1), Bank Vole (N=1), and Water Shrew *Neomys fodiens* Pennant (N=1). Apart from one Bank Vole, all of the 06:00 captures in the treatment plots post-treatment were Common Shrews (Appendix 3.B). Before cutting 30% (N=3) of small mammals were captured in control and 70% (N=7) in treatment patches. However, once the grass was cut only 27% (N=4) of captures were in the treatment patches. This proportion had dropped by around 25% 48 hours later with 20% (N=3) caught in the cut grass. Upon removal of the grass, this proportion dropped a further 70%, to only 6% (N=1) (Table 3.1).

Table 3.1. The distribution of the 06:00 small mammal captures between control and treatment plots.

	Session 1	Session 2	Session 3	Session 4	Total
Control	3(30%)	11(73%)	12(80%)	16(94%)	42
Treatment	7(70%)	4 (27%)	3 (20%)	1 (6%)	15
Total	10	15	15	17	57

Data from across all sessions of the experiment, and for replicates 1, 2, 3 & 5 only. The proportion of captures which were in treatment drops by 61% between sessions 1 & 2, by 25% between sessions 2 & 3, and by 70% between sessions 3 & 4.

Model 1: There was no significant difference in the probability of capture between control and treatment plots before grass cutting (b = < 0.001 ± 0.611, z =< 0.001, P = 0.999). This result did not change if plot 4 was excluded from the model (b = 1.115 ± 0.782, z = 1.426, P = 0.154).

Model 2: The interaction term between treatment and session was significant (b = -2.821 ± 1.08, z = -2.614, P = 0.009). This indicates grass cutting affects the probability of capturing small mammals: this probability increased between sessions one and two on the control patches, but the opposite occurred in treatment (cut grass) patches (Fig. 3.3a).

Model 3: There was no significant effect of the interaction term between treatment and session (b = -0.587 ± 1.083, z = -0.542, P = 0.588), thus there was no difference in the probability of capturing small mammals on the treatment plots between the two sessions following grass cutting, but prior to removal of the cut grass. Removal of the interaction term showed that the there was significantly greater probability of capture on uncut grass than cut grass (b = -1.985 ± 0.542, z = -3.664, P < 0.001). However, the probability of capturing small mammals did not change significantly with time/session (b < -0.001 ± 0.52, z = 0.000, P > 0.999) (Fig. 3.3b).

Model 4: There was no significant effect of the interaction term between treatment (cut or not cut) and session (cut grass *in situ* or removed) (b = -2.257 ± 1.437, z = -1.570, P = 0.116). Although there was no significant change in the probability of capturing small mammals on the treatment plots between the two sessions, the probability of capture tended to decrease in the treatment patches, and increase in the control patches (Fig. 3.3c). Removal of the interaction term showed that there was a significantly greater probability of capture on uncut grass than cut grass (b = -3.175 ± 0.649, z = -4.895, *P* < 0.001). However, the probability of capturing small mammals did not change significantly with session (i.e. presence or absence of cut grass cover; b = 0.346 ± 0.59, z = 0.586, *P* = 0.558)

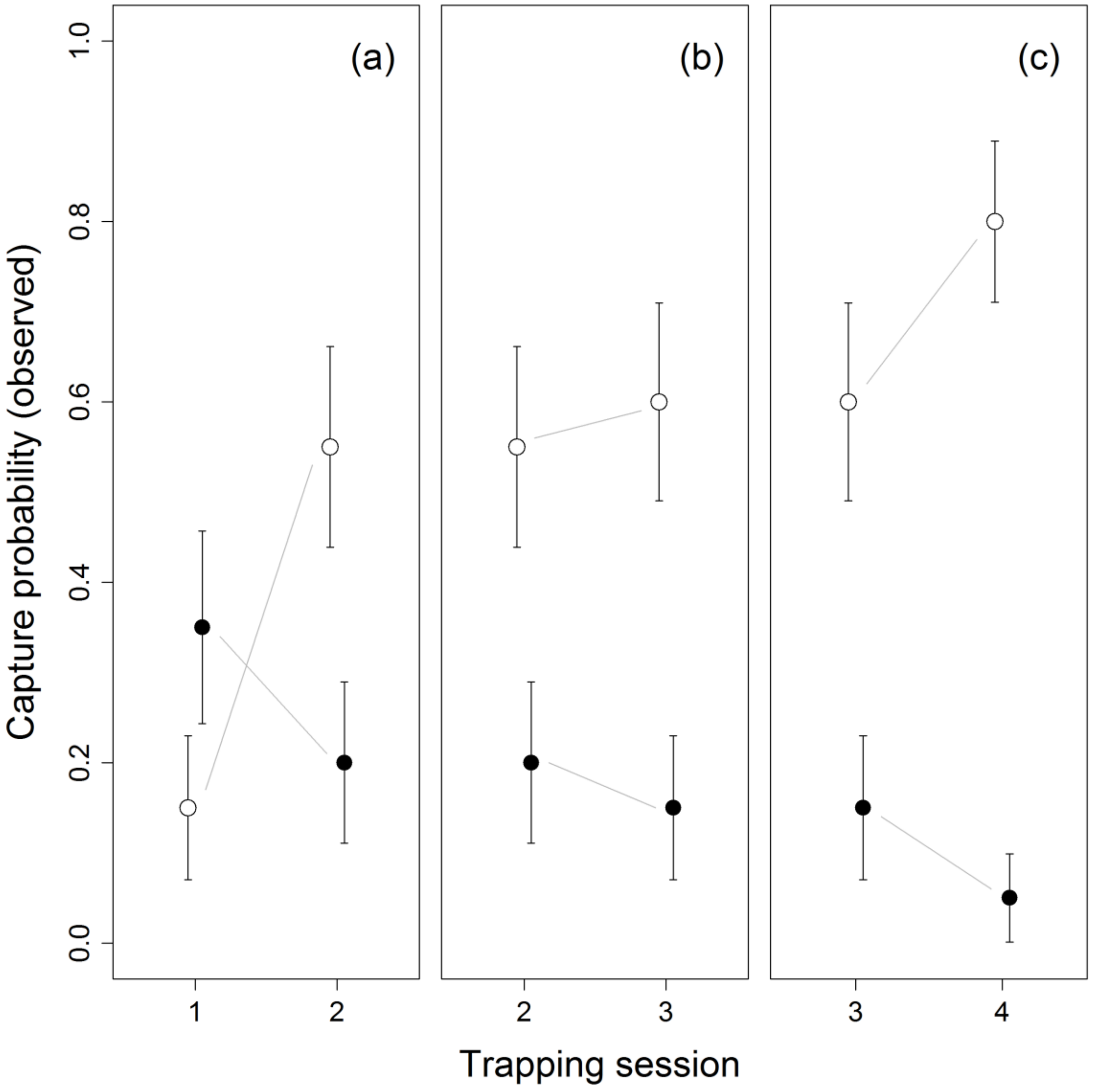

Figure 3.3. Observed mean capture probabilities in treatment (filled circles) and control (open circles) patches (a) before and 24h after cutting (sessions 1-2, model 2), (b) 24h after cutting and 96h after cutting (sessions 2-3, model 3), and (c) before and after removal of the cut grass (sessions 3-4, model 4). Error bars represent +/- 1 SE

Discussion

Our results show that grass cutting significantly affects the probability of capturing small mammals, as demonstrated by the change in capture probabilities in cut and uncut areas. These results are significant even though data has only been used from one of four trap rounds carried out at each stage of the experiment (06:00), thereby reducing the data set by 75%, but reducing issues of pseudo-replication. Our findings support the idea that grass cutting causes rapid movement of small mammals out of the cut areas (i.e. the proportion of captures which were in treatment plots had declined by 61% 24 hours after cutting), but that a small proportion will remain in these cut areas. This contrasts with some earlier studies; e.g. Hanson (1977) where whole populations of *M. agrestis* were found to leave areas where the grass had been cut, but supports other studies which have shown that at least a proportion of a range of small mammals will remain after grass cutting (e.g. Edge *et al.* 1995, Jacob 2003, Jacob & Hempel 2003).

Cutting grass on farmland may therefore be a good management strategy to aid birds of prey. It has been shown that Kestrels prefer to forage over cut grass rather than all other habitat types available to them on farmland (Ashwanden *et al.* 2005, Whittingham & Devereux 2008, Garratt *et al.* 2011), and this is likely to be largely due to increased visibility of their prey. Kestrels hunt by sight, and therefore thick vegetative cover is almost certainly disadvantageous. Additionally, the urine and faeces trails left by small mammals to mark their runways are visible in ultraviolet light, which it is thought can be detected by some diurnal birds of prey; including Rough-legged Buzzards (Koivula & Viitala 1999) and Kestrels (Viitala 1995). This potentially allows foraging raptors to rapidly assess an area for small mammal abundance and thereby assess the profitability of a particular hunting patch, provided the grass is short enough to see the trails (Viitala *et al.*1995, Koivula & Viitala 1999). It would seem likely that it also makes any small mammals remaining in the area easier to actually catch.

As discussed above, the non-zero capture probabilities following treatment suggest that at least some small mammals are found in patches of freshly cut grass (at least for eight - nine days following cutting). It is worth stressing that because of our study design, we cannot distinguish between individuals that stayed in the treatment patches, and individuals that returned or moved into the area from outside. This is important, because the presence of baited traps *per se* could explain the latter pattern. However, even if this is the case it suggests that some small mammals will continue to use an area of cut grass provided it still contains a resource they require – such as a food source – or due to territorial pressures on dispersal, as optimal un-mowed habitats surrounding the mowed areas are likely to resist immigrants if already saturated (Hansson 1977). This suggestion is further supported by Jacob and Hempel (2003), who found that radio-collared Common Voles *Microtus arvalis* Pallas did not leave their territories even after substantial habitat alteration by mowing.

Furthermore, our results stress the importance of the presence of cover to small mammals, as demonstrated by the difference between the effects of cutting and subsequent grass removal (Fig. 3.3c). We found that following removal of the cut grass from patches that had previously been cut, capture probabilities dropped even further. Although this result was not significant, it should be noted that the limited number of captures (see Table 3.1) is likely to have reduced the power of this particular analysis. However, as can be seen from the graph (Fig. 3.3c), the pattern of capture probabilities between sessions 3 and 4 are strongly suggestive of an increased effect of treatment, as is the change in proportion of captures which were in treatment vs. control, which drops from 20% at session 3 to just 6% at session 4 – a drop of 70% (Table 3.1). This is not particularly surprising, and while not powerful enough to be significant adds further weight to the wealth of research which shows how important cover is to small mammals (e.g. Birney *et al.* 1976, Hansson 1977, Hansson 1982, Ostfeld 1985, Pusenius & Viitala 1993). These findings strongly suggest that the majority of small mammals which continue to use a patch after grass cutting would no longer do so if all remaining cover (i.e. the cut grass) is also removed. We therefore suggest that grass cutting *per se* is not as important a determinant of the effect of mowing on small mammals, as whether the cut grass is left *in situ*.

The fact that captures post-cutting increased in the control patches while decreasing in the treatment patches (Fig. 3.3) would seem to suggest emigration by some of the resident small mammals to the nearby control patches. It is also interesting to note that almost all of the captures in treatment patches post-cutting were Common Shrews. The foraging ecology of shrews, which have very high energy demands and need to eat every few hours in order to survive (Barnard & Brown 1984), puts different pressures on them than those on other small mammals. Thus, it may not be so easy for shrews to leave habitats which contain resources they need in response to changes in suitability of such habitats. Indeed, during two of the daylight trap rounds shrews were observed moving around in the cut grass patches, as well as being caught. Most of the previous work on the effects of mowing practices on small mammals has concentrated on rodents; particularly microtines, and therefore further work on the responses of shrews specifically to mowing and the removal of cover could be useful, particularly because they – due to their relative abundance and high daytime activity rates – may be a very important component of the diet of birds of prey.

It is important to note that our results should not be extrapolated into encouraging cutting of large areas of grass simultaneously. Instead leaving patches of taller grass or other habitat (such as nearby woodland or hedgerow) to create conditions for long term survival of small mammal populations is likely to be useful for maintaining healthy mammal populations on which their predators rely. Bank Voles are most commonly found in hedges in agricultural areas, and along with Wood Mice are very abundant in this habitat, whereas Field Voles prefer grass-dominated habitats at field boundaries, therefore increasing the area of both these habitats should in theory increase small mammal abundance and diversity on farmland (Butet *et al.* 2006).

The current study provides evidence that not all small mammals leave areas of grass that have been cut. This suggests that cutting grass may be a valid option to improve habitat suitability for birds of prey (such as Kestrels) by increasing accessibility to small mammal prey. However, this idea needs to be tested by cutting of grass patches (e.g. grass margins or strips, or patches as in our experiment) and measuring the use of such areas by birds of prey. Furthermore, the longer-term effects of such cutting on small mammal populations would need further study. Nevertheless our study suggests that provided small mammals are present in grassland in the first place, and as long as some form of cover is provided (e.g. the cut grass), then mowing is a management tool that is likely to improve access to preferred prey of several diurnal raptor species, some of which continue to decline in the UK.

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Appendix 3.A

	Session 1 (pre-cut)		Session 2 (20h post-cut)	
Trap round	Treatment	Control	Treatment	Control
15:00	-	-	-	-
20:00	-	-	-	2
01:00	-	-	-	-
06:00	-	-	-	3

Capture data for the 2010 repeat experiment. Methodology followed that of 2009. There were no captures prior to the cutting treatment, no captures in treatment sections post-cutting, and only five captures in control post cutting (3 Common Shrews *S. araneus* and 2 Pygmy Shrews *S. minutus*). The experiment was therefore abandoned after Session 2 due to insufficient data.

Appendix 3.B

Tables showing the total captures of each of the six species of small mammals caught, across all of the sessions of the experiment, for all five replicates and for each of the four trap rounds carried out at each session. Captures from replicates 1, 2, 3 & 5 and from the 06:00 trap rounds only were used in the statistical analysis of the results.

		Session 1								
		15:00		20:00		01:00		06:00		
Replicate	Species	Treatment	Control	Treatment	Control	Treatment	Control	Treatment	Control	Total
1	*S. araneus*		1	1				1		3
	S.minutus					1				1
2	*S. araneus*		2		1	1	2		2	8
	M.agrestis			1						1
	M.glareolus	1								1
	A.sylvaticus			1				1		2
3	*S. araneus*	3		3		2		3		11
	M.agrestis			1		1		1		3
	M.glareolus	1								1
	S.minutus		1							1
4	*S. araneus*		2	1	3		3	1	5	15
	M.glareolus		1							1
5	*S. araneus*			1	1		1		1	4
	S.minutus							1		1
	Total	5	7	9	5	5	6	8	8	53

		Session 2								
		15:00		20:00		01:00		06:00		
Replicate	Species	Treatment	Control	Treatment	Control	Treatment	Control	Treatment	Control	Total
1	*S. araneus*		1		3		3	1	3	11
	S.minutus	1								1
2	*S. araneus*	1	3	1	3		1		3	12
	M.agrestis				1		1			2
3	*S. araneus*	2	1	2	4		2	2	2	15
	M.glareolus							1		1
	A.sylvaticus								1	1
	S.minutus	1	1							2
4	*S. araneus*		4		4	1	1	1	3	14
	M.agrestis		1		1				1	3
5	*S. araneus*		2	1			1			4
	M.agrestis				1					1
	A.sylvaticus								1	1
	Total	5	13	4	17	1	9	5	14	68

		Session 3								
		15:00		20:00		01:00		06:00		
Replicate	Species	Treatment	Control	Treatment	Control	Treatment	Control	Treatment	Control	Total
1	*S. araneus*	1	3	1	1	1	2	1	2	12
2	*S. araneus*	1	3		4		3	1	3	15
	M.agrestis						1		2	3
3	*S. araneus*	2		2	2	1	1	1	3	12
	M.glareolus	1								1
4	*S. araneus*	1	2	2	1	1	4	2	2	15
	M.agrestis				2					2
5	*S. araneus*		3		3		3		2	11
	S.minutus				1					1
	Total	6	11	5	14	3	14	5	14	72

		Session 4								
		15:00		20:00		01:00		06:00		
Replicate	Species	Treatment	Control	Treatment	Control	Treatment	Control	Treatment	Control	Total
1	*S. araneus*	1	3	1	3	1	4		5	18
2	*S. araneus*		3		4		4		4	15
	M.agrestis		1				1			2
	N.fodiens				1				1	2
3	*S. araneus*		1		2		3	1	3	10
4	*S. araneus*		2		3	1	1	2	2	11
	M.agrestis								1	1
5	*S. araneus*		2		4		4		3	13
	Total	1	12	1	17	2	17	3	19	72

Appendix 3.C

Likelihood Ratio Tests comparing the fit of models with a given factor included to those with the factor excluded. Presented are AIC, BIC and log-likelihood of each model, and a chi-squared test testing the significance of the change in likelihood, thereby testing whether inclusion of the factor improves the fit of the model.

Model 1a: pre-cut. No replicate 4. Testing the effect of removing ‘treatment’ on the fit of the model

	Df	AIC	BIC	logLik	Chisq	Chi df	Pr(>Chisq)
Without treat	2	48.987	52.365	-22.493			
With treat	3	48.806	53.873	-21.403	2.181	1	0.140

Model 2: The first two sessions, with interaction term. Testing the effect of removing ‘treatment x time’ on the fit of the model. Time reference category is session 1.

	Df	AIC	BIC	logLik	Chisq	Chi df	Pr(>Chisq)
Without treat x time	4	104.842	114.370	-48.421			
With treat x time	5	99.536	111.450	-44.768	7.306	1	0.007

Model 3: mid two sessions, with interaction term. Testing the effect of removing ‘treatment x time’ on the fit of the model. Time reference category is session 2.

	Df	AIC	BIC	logLik	Chisq	Chi df	Pr(>Chisq)
Without treat x time	4	98.005	107.530	-45.003			
With treat x time	5	99.713	111.620	-44.856	0.293	1	0.589

Model 3a: mid two sessions, no interaction term. Testing the effect of removing ‘treatment’ on the fit of the model. Time reference category is session 2.

	Df	AIC	BIC	logLik	Chisq	Chi df	Pr(>Chisq)
Without treat	3	111.006	118.150	-52.503			
With treat	4	98.005	107.530	-45.003	15.001	1	<0.001

Model 3a: mid two sessions, no interaction term. Testing the effect of removing ‘time’ on the fit of the model. Time reference category is session 2.

	Df	AIC	BIC	logLik	Chisq	Chi df	Pr(>Chisq)
Without time	3	96.005	103.15	-45.003			
With time	4	98.005	107.53	-45.003	0	1	1

Model 4: end two sessions, with interaction term. Testing the effect of removing ‘treatment x time’ on the fit of the model. Time reference category is session 3.

	Df	AIC	BIC	logLik	Chisq	Chi df	Pr(>Chisq)
Without treat x time	4	82.123	91.651	-37.061			
With treat x time	5	81.277	93.187	-35.638	2.846	1	0.092

Model 4a: end two sessions, no interaction term. Testing the effect of removing ‘treatment’ on the fit of the model. Time reference category is session 3.

	Df	AIC	BIC	logLik	Chisq	Chi df	Pr(>Chisq)
Without treat	3	113.473	120.620	-53.737			
With treat	4	82.123	91.651	-37.061	33.350	1	<0.001

Model 4a: end two sessions, no interaction term. Testing the effect of removing ‘time’ on the fit of the model. Time reference category is session 3.

	Df	AIC	BIC	logLik	Chisq	Chi df	Pr(>Chisq)
Without time	3	80.468	87.614	-37.234			
With time	4	82.123	91.651	-37.061	0.3449	1	0.557

Chapter 4: Creating Ephemeral Resources: How Long Do the Beneficial Effects of Grass Cutting Last for Birds?

Carly T. Peggie, Claudia M. Garratt & Mark J. Whittingham

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School of Biology, Newcastle University, Newcastle upon Tyne, UK

Short title: Bird response to grass field cutting

NB: An appendix has been added (Appendix 4.A) showing Likelihood Ratio Tests for the models reported in this chapter.

Abstract

Capsule Beneficial effects of cutting grass are relatively short-lived for a range of bird species.

Aims To investigate how cutting affects a range of birds occurring on farmland, how long these effects last and whether there is any effect of the timing of cutting operations.

Methods We surveyed birds on 33 grass fields on 3 farms in northern England, both before and after agricultural cutting operations. The data were then modelled using Generalized Linear Mixed Models (GLMMs).

Results Species relying on below-ground invertebrates (e.g. corvid spp.) prefer cut swards, whilst some species make greater use of longer grass swards (e.g. Pheasant). However, the key finding of this study was the rapid decline in use of grass fields following cutting by a range of bird species. Use of fields after cutting declined by 50% by day 11 for corvids, by day eight for aerial feeders (e.g. hirundines), by day six for gulls and by day 4 for Kestrels. The date of cutting was significant for corvids, but not for any of the other groups studied.

Conclusions We recorded mainly common, generalist species that make use of agricultural grassland. The foraging behaviour of these species is similar to other, rarer or declining species, and so our findings can be extrapolated to a range of farmland birds. However, it is important to note that while some species of bird apparently benefit from grass cutting, other studies indicate direct negative effects of cutting on some species not found in our surveys.

Introduction

Farmland biodiversity has been in decline since the mid 20th century, and this has been linked to the increasing intensity of agriculture during that period (Benton *et al.* 2003). Farmland birds have been the subject of much agro-ecological research, and there is clear evidence linking changes in agricultural practice to declines in farmland bird populations across Europe (e.g. Donald, Green & Heath 2001, Gregory *et al.* 2005, Donald *et al.* 2006). Agri-environment Schemes (AESs) have been introduced in many countries throughout Europe, in part, at least, to lessen these negative effects on biodiversity (Whittingham 2007). Within the category ‘farmland birds’ are many species which rely mainly on grassland and the management options for grassland birds within AESs have been informed by several recent studies (e.g. Perkins *et al.* 2000, Vickery *et al.* 2001, Barnett *et al*. 2004, Atkinson *et al.* 2004, 2005, Buckingham *et al*. 2004, 2006, Whittingham & Devereux 2008). Grassland sward management has consistently come out as a key management tool to manipulate predation risk, and also invertebrate food abundance and accessibility, by enabling birds to more readily access below-ground grassland invertebrates (Atkinson *et al.* 2004, Buckingham *et al.* 2006). Changes to predation risk and food abundance/accessibility all combine to alter intake rates, which ultimately alter the quality for patch choice (e.g. Stephens & Krebs 1986, Lima & Dill 1990, Whittingham *et al.* 2004).

Some popular AES options focus on the creation and maintenance of new grassland habitats, such as grass margins, buffer strips and low input grassland (Natural England 2010 a & b), but once in place these newly created grassland areas need careful management in order to maximise their benefits for wildlife. We know that changing sward height alters its attractiveness to a range of bird species (e.g. Perkins *et al.* 2000, Atkinson *et al.* 2004, 2005, Whittingham & Devereux 2008, Douglas *et al.* 2009), but it is not clear over what time period short grass acts as high-quality habitat for the birds which use it. Such information is useful if we are to adapt current AES sward management prescriptions to maximise the usefulness of those swards for as wide a number of species as possible.

Conversely, we sometimes need to understand bird habitat usage in order to discourage their use of an area, such as the large patches of grasslands at airports (e.g. Brough & Bridgman 1980, Blackwell *et al*. 2009). Birds are a primary risk group for causing aircraft crashes (e.g. Blackwell *et al*. in review), and so discouraging the use of grasslands by some larger species, such as corvids, gulls, and wildfowl, and even smaller species such as Common Starlings *Sturnus vulgaris*, is important to minimise strike risk.

In this study we show the relationship between temporal aspects of cutting and use of grassland by a range of birds (see Methods and Materials for list of study groups), and demonstrate that the drop-off in use of cut fields is rapid. As a secondary aim we also repeat earlier work investigating the effects of cutting *per se* on grassland use by farmland birds in the summer and our results concur with earlier work. It is important to note that past studies have shown that some species (not found in our study) are negatively affected by grass cutting and so our study needs to be interpreted in light of these studies (see conclusions).

Methods and Materials

CTP surveyed 33 grass fields at three farms (4, 20 and 9 fields at each farm respectively) in County Durham, England, over 70 days from mid June to August, 2009. The mean field size was 8.47 ± 1.5 (1 se) ha (range: minimum 1 ha, maximum 26 ha). Twenty-nine of the fields were permanent pasture (they had been grass fields for more than 5 years), and four fields were grass leys (they had been ploughed less than 5 years ago).

Surveys were not conducted during wet and/or windy weather, due to the effects of those conditions on visibility, hindering bird identification and observed activity (Bibby *et al.* 1992). All surveys were carried out between 10.00 and 18.30 (BST) to avoid periods when birds were either leaving or arriving at roost sites. All fields were cut for either silage or hay during the observation period (at farm 1: three fields cut between 25 July and 1 August with one field cut on 24 June; at farm 2: all fields cut between 27 July and 7 August; at farm 3: all fields cut between 28 July and 17 August), so the use of fields by foraging birds was recorded on each field both before and after grass cutting.

Vegetation height measurements

Pre-cut sward height was measured during the first survey of each field and post-cut sward height was measured during the first survey following cutting. Ten random sites were chosen on each field and height was obtained using a tape measure, and then the mean for each field was used to calculate the data presented in the results.

Pre-cut bird surveys

An average of 2.2 visits (maximum visits = 4, minimum = 2) were made to each of the 33 fields before they were cut. All visits were at least one day apart except for two fields which were visited twice on the same day (early and late in the day).

Field surveys consisted of walking transects from one side of the field to the opposite side (the starting point for the first transect in each field was chosen randomly). Transects were used to avoid bias in the observability of birds between fields with differing grass length, as birds hiding in the long grass were flushed by the observer. The transects were approximately 50 m apart so that the observer visited every part of the field to within 25 m. Birds that were either flushed from the grass by the observer, foraging in the grass or foraging over the grass (e.g. Common Kestrels *Falco tinnunculus* hovering and aerial feeders flying low (<5 m) over the field to forage) were identified to species and their numbers recorded (the size of large flocks was estimated as accurately as possible). Birds flying over fields but not landing on them (other than in the circumstances outlined above), or those at the edge of the field (e.g. in hedges) were not recorded. Although the three study farms were separated geographically, within each farm some of the surveyed fields were adjacent to one another. In order to minimize double-counting of birds, the observer noted the direction taken by birds which were flushed during the course of the surveying of each field, and thus avoided (as far as possible) re-counting the same individual in adjacent fields surveyed on the same visit.

Post-cut bird surveys

An average of 4.8 visits (maximum visits = 9, minimum = 3) were made to each of the 33 grass fields after they were cut for harvest. Those surveys carried out whilst grass was being cut (0 days post-cut), turned, or baled did not involve transects, but instead vantage point observations were made from a point nearby. Otherwise, transects every 50 m were carried out and birds recorded as detailed above. If livestock were introduced to a field in the intervening period between observation visits, then no more surveys of that field were carried out to avoid conflicting effects due to grazing.

Statistical analysis

We grouped species into functional units for analysis. We did this for two reasons. First, numbers of some species were too few to analyse independently. Secondly, and more importantly, some bird species feed in similar ways; for example, hirundines and Common Swifts *Apus apus* all feed on aerial insects and so it was sensible to group them together. Likewise, corvids all feed mainly on ground-dwelling and below-ground invertebrates. Thirdly, because some species often occur in mixed species flocks, each species is not likely to be statistically independent of other similar species (e.g. the presence of foraging Eurasian Jackdaws *Corvus monedula* L. is likely to attract Rooks *Corvus frugilegus* L. and Carrion Crows *Corvus corone* L.). Thus it makes ecological sense to group the species recorded into six functional units for analysis: (1) ‘aerial feeders’ (Barn Swallows *Hirundo rustica* L., House Martins *Delichon urbicum*, Common Swifts); (2) ‘corvids’ (Carrion Crows, Eurasian Jackdaws, Rooks and Black-billed Magpies *Pica pica* L.); (3) ‘gulls’ (Black-headed Gulls *Larus ridibundus* L., Common Gulls *Larus canus*, L.); (4) ‘pigeons’ (Common Wood Pigeons *Columba palumbus* L., Eurasian Collared Doves *Streptopelia decaocto* L.); (5) ‘Common Kestrels’; (6) ‘Common Pheasants’ *Phasianus colchicus* L.

There is a potential lack of independence because some of the fields at each farm were located in blocks bordering one another, and so the numbers of birds on any given field could affect numbers on nearby fields. However, this issue was minimized by making repeat visits to the same fields (see above), such that use of a given field was more likely to be picked up over multiple visits. In addition, this source of lack of independence is likely to cause Type 2 error (absence of an effect of birds selecting shorter grass fields when there is actually one present) and so the results we report are likely to be robust.

Generalized Linear Mixed Models (GLMMs) were used to analyse the data in R (version 2.9.1) (R Development Core Team 2009) using the package *lme4* v.0.99875.9 (Bates *et al.* 2008). The response variable was the number of occasions that each functional group was recorded in a field, and the number of surveys was specified as the binomial denominator [with a binomial error structure with logit link function (Crawley 2007), e.g. if corvids were present on two visits out of six then two was specified as the response variable and six as the denominator]. The assumption within our analysis is that birds can make an independent choice of fields on each visit. The method we use has been previously shown to correlate number of occurrences (e.g. species ‘a’ was recorded on field 'b' on two out of six visits) with total count of birds on a field (e.g. 14 individuals of species 'a' recorded on field 'b' across all six visits; see Perkins *et al.* 2000, Moorcroft *et al.* 2002). This method gives a good indication of 'use' of a particular field. Whilst we acknowledge that this is not perfect, it was not possible to analyse total species counts because of highly skewed data. Many birds are social foragers and the decision of where to feed is influenced by where its flock-mates are. This renders count data non-independent and its analysis would violate statistical test assumptions.

The term 'field' was included in all models as a random effect to allow within-field comparisons, while controlling for variation resulting from site-specific parameters. Two models were fitted to each of the six functional groups. Model A included 'cutting', fitted as a two-way factor (before cutting was coded as '1', and after cutting was coded as '2') to account for the effects of manipulation of grass height. Model B was fitted in each instance with just data on post-cutting. Thus the factor 'cutting' was not included in Model B, but an additional two variables were incorporated into Model B: 'timing of visit' was incorporated into the model to investigate whether the probability of occurrence of birds on plots changed with time since mowing; 'date of cutting' was included, with the earliest cut field given a value of '1' and the number of days between that and the cutting of each subsequent field then added (thus a field cut 10 days after the earliest field was assigned '11'). This was to investigate whether there was any difference in use of fields depending on whether they were cut earlier or later in the season. In addition a two way factor for the presence or absence of 'bales' in the field and a three-level factor for 'farm' was also included in Model B. The former was fitted as it was noted that birds often made use of bales as perches.

The significance of each predictor in the analyses of both model A and B was assessed using the change in deviance (ΔD), which is distributed asymptotically as χ^2, on removal of each term from a model including all predictors (i.e. a full model, Whittingham *et al.* 2006). The fit of the model to the assumptions of a binomial distribution can be approximated by comparing the ratio of residual deviance / residual degrees of freedom; ratios close to one indicate a reasonable fit to the data, whereas ratios greater than 2.5 indicate a poor, overdispersed fit (Crawley 1993). The data did not need to be corrected for overdispersion in any of the models due to low residual deviance to residual degrees of freedom ratios (highest ratio was 1.23). All probabilities quoted are two-tailed. Means and standard errors are presented in the form mean ± 1 standard error.

Results

The mean pre-cut sward height over all fields was 89.2 cm (maximum mean height = 129.5 cm, minimum mean height = 69.0 cm), and the mean post-cut sward height over all fields was 4.6 cm (maximum mean height = 5.8 cm, minimum mean height = 3.6 cm).

During the surveys, 21 bird species (3958 individuals) were recorded (Table 4.1). Some species did not have sufficient data to be analysed (e.g. Skylark *Aluda arvensis*) (Table 4.1). Species with similar foraging behaviour and body size were grouped into functional groups for analysis (see Table 4.2, and Methods and Materials for a description of the groups and which bird species they contain).

Effect on birds of cutting grass

Aerial feeders were more frequently recorded on fields before cutting occurred than after ($\chi^2 = 53.01$, 1 df, $P < 0.001$), as were Pheasants ($\chi^2 = 34.40$, 1 df, $P < 0.001$). Gulls, corvids and pigeons showed the reverse pattern and were recorded significantly more frequently after cutting (gulls: $\chi^2 = 47.80$, 1 df, $P < 0.001$, corvids: $\chi^2 = 72.60$, 1 df, $P < 0.001$, pigeons: $\chi^2 = 11.10$, 1 df, $P < 0.001$). There were more Kestrels present during post-cut observations, but this was not significant ($\chi^2 = 2.01$, 1 df, $P = 0.160$).

Table 4.1. The total number of individuals of each species observed pre-cut and post-cut, with mean number per survey in brackets.

	Total number (mean no. per visit)	
Species	Pre-cut	Post-cut
Greylag Goose *Anser anser*	0 (0)	5 (0.032)
Kestrel *Falco tinnunculus*	2 (0.027)	9 (0.057)
Grey Partridge *Perdix perdix*	0 (0)	2 (0.013)
Pheasant *Phasianus colchicus*	24 (0.324)	0 (0)
Lapwing *Vanellus vanellus*	0 (0)	10 (0.063)
Curlew *Numenius arquata*	0 (0)	6 (0.038)
Black-headed Gull *Larus ridibundus*	0 (0)	149 (0.943)
Common Gull *Larus canus*	0 (0)	1038 (6.570)
Wood Pigeon *Columba palumbus*	0 (0)	34 (0.215)
Collared Dove *Streptopelia decaocto*	0 (0)	4 (0.025)
Common Swift *Apus apus*	12 (0.162)	7 (0.044)
Skylark *Alauda arvensis*	5 (0.068)	4 (0.025)
Barn Swallow *Hirundo rustica*	207 (2.797)	129 (0.816)
House Martin *Delichon urbicum*	7 (0.095)	2 (0.013)
Dunnock *Prunella modularis*	0 (0)	1 (0.006)
Mistle Thrush *Turdus viscivorus*	0 (0)	5 (0.032)
Magpie *Pica pica*	0 (0)	11 (0.070)
Jackdaw *Corvus monedula*	0 (0)	405 (2.563)
Carrion Crow *Corvus corone*	0 (0)	116 (0.734)
Rook *Corvus frugilegus*	0 (0)	1763 (11.158)
Tree Sparrow *Passer montanus*	1 (0.014)	0 (0)
Total	258	3700

For example, 207 Barn Swallows in total were counted on the pre-cut surveys, with an average of 2.80 Swallows recorded per survey visit per field. A total of 74 pre-cut surveys were carried out in contrast to 158 post-cut surveys thus the mean number per visit per field is a better indicator of differences between treatments than total numbers.

Table 4.2. The number of times each species was present out of 74 pre-cut surveys and 158 post-cut surveys and the same number expressed as a proportion in brackets.

Species/group	Pre-cut	Post-cut
Aerial feeders	49 (0.66)	31 (0.20)
Pheasant	14 (0.19)	0 (0)
Gulls	0 (0)	53 (0.34)
Corvids	0(0)	74 (0.47)
Kestrel	2 (0.03)	9 (0.06)
Pigeons	0 (0)	13 (0.08)

For example, one or more aerial feeders were recorded on 49 occasions pre-cut out of 74 visits, giving a proportion of 49/74 (0.66 or 66%).

How did the date on which the field was cut and time since cutting affect birds?

Corvids were recorded more frequently on fields that were cut earlier in the year (date of cutting) (χ^2 = 4.72, 1 df, P = 0.03) and there was a non-significant trend in the same direction for gulls (χ^2 = 3.26, 1 df, P = 0.07). However, the date on which the field was cut did not affect the probability of field use by any other group: aerial feeders (χ^2 = 0.57, 1 df, P = 0.45), Kestrels (χ^2 = 0.33, 1 df, P = 0.56) and pigeons (χ^2 = 0.63, 1 df, P = 0.42) (there was insufficient data to model the effect on Pheasants).

Time since cut had a significant negative effect on aerial feeders (χ^2 = 11.21, 1 df, P < 0.001), gulls (χ^2 = 51.81, 1 df, P < 0.001), corvids (χ^2 = 24.5, 1 df, P < 0.001), and Kestrels (χ^2 = 7.12, 1 df, P = 0.008) (Fig. 4.1). Time since cut did not have a significant effect on the presence of pigeons (χ^2 = 0.04, 1 df, P = 0.85) and again there was insufficient data to analyse the data on Pheasants.

The best-fit relationship was then plotted for those relationships which were significant, to examine how quickly this drop off in use occurred (Fig. 4.2). Using this best-fit line, we calculated on which day since cutting began use dropped by more than 50% from the starting level, and found this occurred for corvids by day 11, aerial feeders by day 8, gulls by day 6, and Kestrels by day 4.

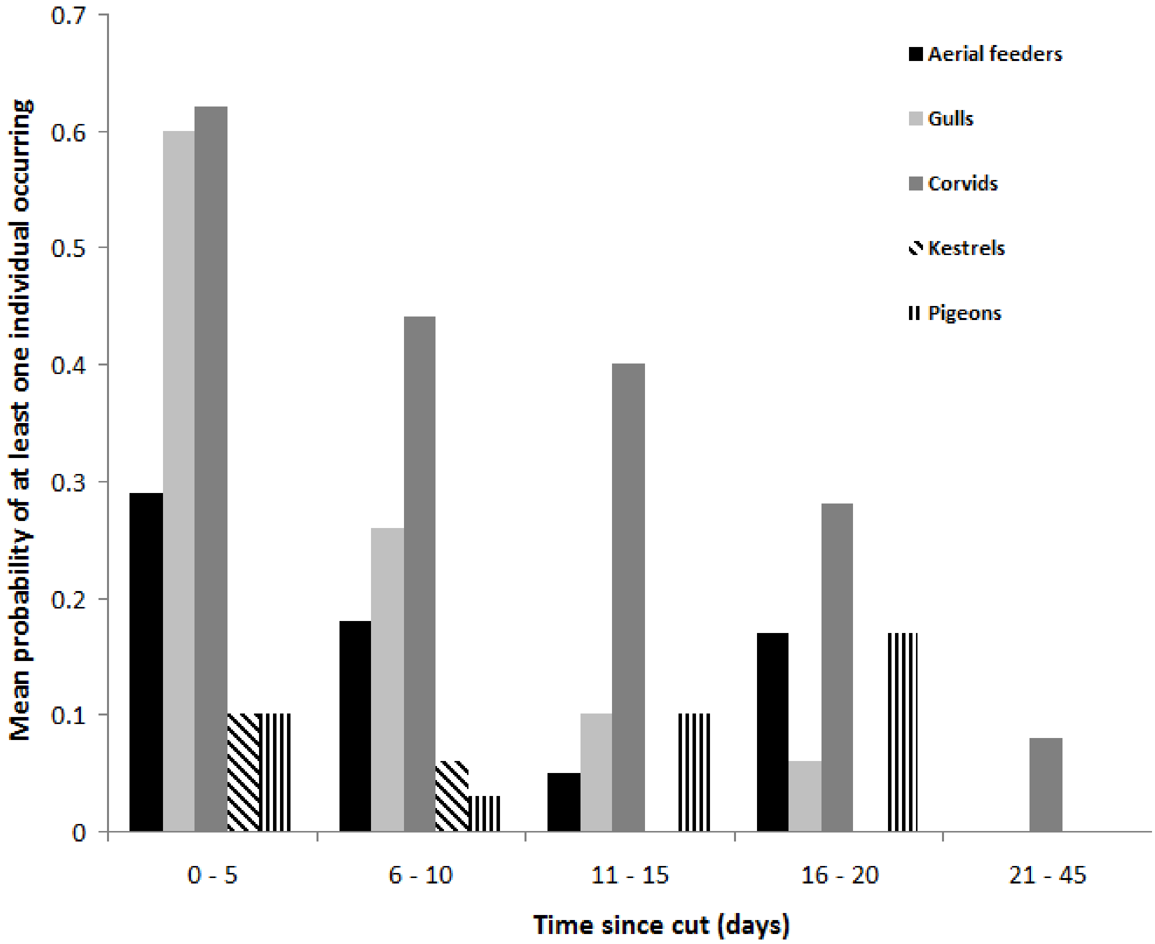

Figure 4.1. The effect of time since mowing on the presence of different birds during post-cut surveys. Mean values are the mean number of occurrences per survey over each time period. E.g. aerial feeders occurred in 21 of the 73 post-cut surveys carried out between days 0 to 5, which gives a mean value of 0.29 aerial feeders per survey for that time period. Note that although statistical analysis was performed using the actual number of days since the field was mown for ease of presentation we have shown the data here based on grouping into five day periods (except the last category which was summed into a 25 day period due to the scarcity of data for that period).

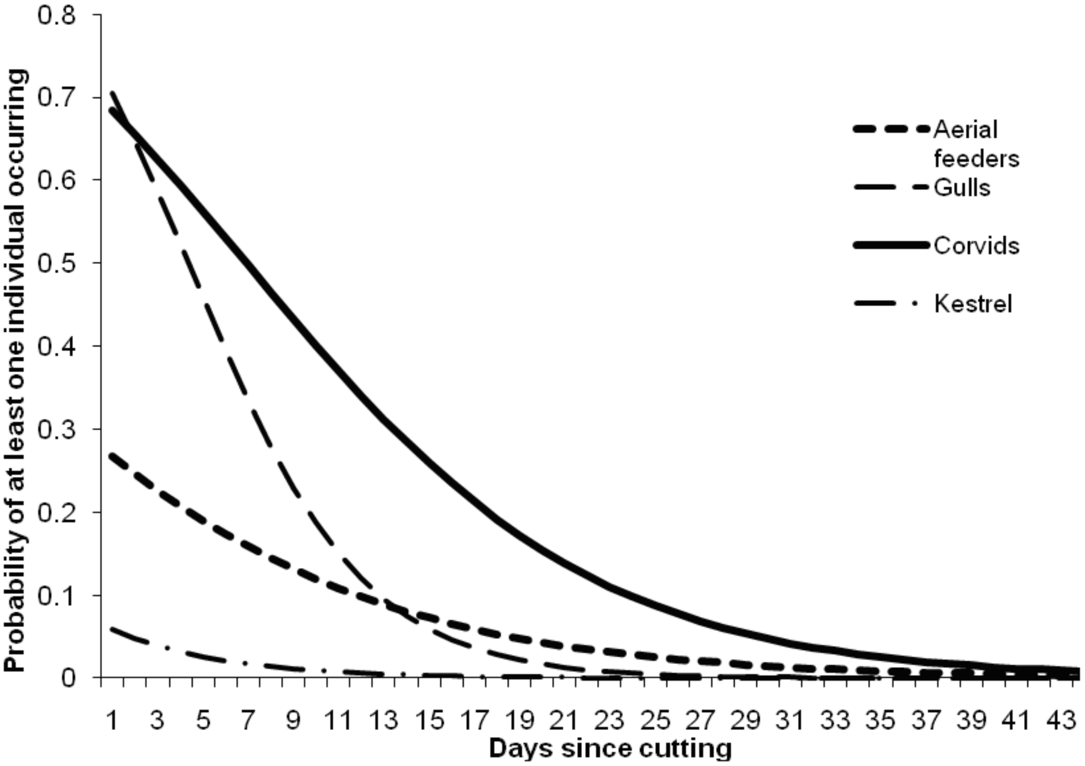

Figure 4.2. Effect of time since cutting on the likelihood of recording four functional groups of birds. All relationships shown were significant (see results) and the back-transformed parameter estimates of those relationships are plotted here over the range of time since cutting from which data were recorded.

Discussion

This study concurs with earlier studies reporting preferences for short grass by a range of species. These earlier studies showed corvids, pigeons and Kestrels preferred short grass swards (Perkins *et al.* 2000, Atkinson *et al.* 2005, Whittingham & Devereux 2008, Garratt *et al.* 2011) but did not investigate the time period over which the effects of grass cutting attract foraging birds. The key result reported here is the rapid drop off in field use by many bird species after cutting (Fig. 4.2).

Why do some species prefer shorter grass whilst others show the reverse pattern?

There are three key factors which affect choice of foraging location: predation risk, food accessibility and food abundance. The ways in which each of these factors is likely to influence the results from our study are considered and discussed.

For many small birds, perceived predation risk is a principle factor affecting which habitats are selected for foraging activities (Whittingham & Evans 2004, Whittingham *et al.* 2004, Whittingham & Devereux 2008). Predation risk differs depending on escape strategy (fleeing or crypsis). Shorter grass provides easier detection of predators, and is preferred by species that flee upon detection of a predator. Long grass provides somewhere to hide, and is generally preferred over short grass by those species which use crypsis as an anti-predator strategy, for example Pheasants (Dowell 1990).

Prey abundance is an important determinant of patch choice, but is often mediated by accessibility to influence actual food availability and intake rates. For example a patch containing some prey items which are relatively exposed is likely to be chosen over a patch containing a greater number of prey items which are difficult to access (e.g. due to dense vegetation: Whittingham and Markland 2002). However, for one group in our study prey abundance is likely to be the key factor determining intake rate. Aerial feeders (e.g. swifts, and hirundines such as Barn Swallows and House Martins) forage on flying (mainly foliar) invertebrates, which are likely to be more abundant in long, established grassland (Ambrosini *et al*. 2002, Atkinson *et al*. 2004). Cutting can result in a temporary flush of invertebrate availability, and hirundines will often congregate on freshly cut hay or silage (Vickery *et al*. 2001), but our results show this effect to be reasonably short-lived, and there is a significantly lower occurrence of aerial feeders with time since cutting (Fig. 4.2).

In contrast to aerial feeders, many species of farmland bird feed on surface invertebrates such as beetles, below-ground soil invertebrates such as earthworms and *Tipulid* spp. (e.g. corvids, gulls), or small mammals (e.g. Kestrels). The intake rate of birds which feed on these prey is determined not just by their abundance but by their accessibility. Experiments with Common Starlings foraging on below-ground invertebrates on grassland swards are useful to demonstrate the principle. Devereux and colleagues (2004) showed that prey intake rates by Starlings are higher on short swards (3 cm) than on longer swards (13 cm), and that this is likely to be due to a combination of greater food accessibility and lower predation risk due to improved detectability of predators. In our study we found that both gull and corvid species occurred more often on cut grass. Because of their size, neither of these groups is likely to be concerned with predation risk. Instead they are more likely to be responding to changes in accessibility of their prey, e.g. earthworms, *Lumbricus* spp. Similarly, Kestrels also showed a trend for greater occurrence on shorter swards in line with previous studies (Sheffield *et al.* 2001, Aschwanden *et al.* 2005, Whittingham & Devereux 2008, Garratt *et al.* 2011). Kestrels are diurnal hunters (Village 1990) which either hover over or perch overlooking grass in order to visually detect their prey, most commonly small mammals such as voles *Microtus* spp., and small birds (the latter particularly in June/July). However, they will also take large surface and foliar invertebrates, and more rarely amphibians and reptiles (Village 1990). Reduction of cover is likely to increase accessibility to many of these prey items for Kestrels (Preston 1990, Viitala *et al.* 1995, Jacob & Brown 2000, Jacob & Hempel 2003, Garratt *et al*. 2011).

The preference for shorter swards by Common Wood Pigeons is likely to be due to their preference for grazing on low-growing, small-leaved weeds such as clover *Trifolium* spp., and they have been shown in several studies to prefer short swards maintained by either mowing, grazing or trampling (Kenward & Sibly 1977, Brough & Bridgman 1980, Buckingham & Peach 2005).

Why does use of cut grass decline rapidly after cutting?

The occurrence of all species/groups with sufficient post-cut data for analysis decreased as the time since cut increased (Fig. 4.2). The temporal effects of grass cutting on the foraging behaviour of the birds in our study are likely to be primarily caused by the effects that cutting grass has on prey abundance and distribution. Tall grass supports greater arthropod diversity and abundance than short grass (Morris 2000, Vickery *et al*. 2001), and foliar invertebrates are likely to disperse quickly after cutting or are killed by the cutting operation (e.g. Humbert *et al*. 2010) which probably explains the trend shown by the aerial feeders (see above). However, the immediate effects of cutting on soil and surface invertebrates are likely to be less marked, with the most noticeable initial decline in abundance of the newly exposed prey probably due to depletion by foraging birds and other predators. Mowing of the grass will also alter the microclimate of the soil and sward, which is likely to cause invertebrates to disperse and their availability to decline naturally over time (Devereux *et al*. 2006). This decline in prey abundance will result in a decline in foraging activity of those birds that feed on them. Kestrel prey items, such as voles, will also have a relatively high availability in cut grass initially (whether dead or alive) and then decline (Edge *et al.* 1995, Garratt unpubl data) as they either move to find cover or are eaten. Many studies have found clear reactions to sudden changes in vegetation height in voles *Microtus* spp., such as dispersal and decreases in home range size (Hansson 1977, Edge *et al*. 1995, Sheffield *et al*. 2001, Jacob & Hempel 2003).

Cutting of grass fields on other farms surrounding our study fields could potentially affect our results, but we were unable to control for this as we did not collect information on the timing of cutting in the surrounding landscape. However, we could partially address this issue by examining whether there was a preference for fields that were cut earlier (which are likely to have represented a scarcer resource than later cut fields). Most of the fields on each of our three study farms were cut within one or two weeks of each other (with the exception of one field cut a month early – see Methods and Materials). The date that each field was cut was not significantly related to occurrence of any group except corvids (and a close to significant result for gulls) which tended to make more use of fields which were cut earlier. This suggests that the numbers of cut fields present in the surrounding area effects use of a particular field by corvids and gulls (i.e. the later a field is cut, the higher the likelihood of the availability of other cut fields on and around each farm). However, it is important to note that although we could not control for this issue entirely, the date of cutting was included in all of the models in the results section. Coordination of the timing of cutting by an individual farmer (or neighbouring farmers) could be used as a management tool to manipulate habitat quality for farmland birds.

Management implications

Conserving wild bird populations is a key conservation objective at a Europe-wide level (Gibbons 2000, Gregory *et al*. 2005, European Commission 2010 Directive 2009/147/EC) and in response to this, up until June 2010, a UK government target was in place to reverse the declines of the 19 farmland bird species on the ‘Farmland Bird Index’ (FaBI) by 2020. This target was selected by the UK government as a measurable surrogate for assessing the success of its policies which seek to conserve wildlife, in general, in the English countryside (Gregory *et al.* 2005). Four of the species included in our study are on the FaBI list (namely Rooks, Jackdaws, Common Wood Pigeons, and Kestrels). Thus our findings suggest management that is directly pertinent to this index, although of the four species Kestrel is the only one whose numbers are still declining.

It is important to note that our study sites did not support many species of conservation concern (with the notable exception of Grey Partridges *Perdix perdix* L., Swifts, Skylarks, Tree Sparrows *Passer montanus* L., Northern Lapwings *Vanellus vanellus* L. and Kestrels). However, some of the species which did occur in great enough numbers for analysis are functionally similar to other key species. For example, the corvids and gulls we recorded feed mainly on below-ground invertebrates, and so the patterns we report are likely to apply to other species that feed in a similar way and that are on the UK list of Birds of Conservation Concern (for example the red-listed Song Thrush *Turdus philomelos* and Common Starling, and the amber-listed Red-billed Chough *Pyrrhocorax pyrrhocorax*).

We acknowledge the important point that past studies have shown that some species (not found in our study) are negatively affected by grass cutting due to nest destruction and increased chick mortality (e.g. Corncrake *Crex crex*: Tyler *et al*. 1998; Black-tailed Godwit *Limosa limosa*: Schekkerman & Bientema 2007; Kleijn *et al*. 2010). These species are unlikely to nest in small patches of grassland such as would be found in grass margins (a key Entry Level Scheme management option, Natural England 2010a), but our results need to be interpreted in light of these studies as well as those showing benefits of grass cutting for some species, and the potential for negative effects needs to be borne in mind when implementing our findings.

The temporal effects shown in this study suggest the need to provide cut grass for foraging at different times throughout the summer months. Our results suggest that providing freshly cut grass patches every 2-3 weeks would be of benefit to a range of different farmland birds, and farmers or landowners could be recommended to carry out multiple cuts on grassland, or stagger the cuts for each field (logistics allowing) to provide a more constant food supply. We suggest our results should be applied in a bespoke manner by land mangers wishing to either deter use of grasslands (such as on airports) who may wish to retain longer swards (for a fuller exploration of this issue see Brough & Bridgman 1980 and Bradwell *et al*. in review) or those wishing to provide beneficial habitats for foraging birds on farmland by creating a mix of short and long grass in close juxtaposition (Aschwanden *et al.* 2005, Douglas *et al*. 2009, Garratt *et al.* 2011).

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Appendix 4.A

Likelihood Ratio Tests comparing the fit of models with a given factor included to those with the factor excluded. Presented are AIC, BIC and log-likelihood of each model, and a chi-squared test testing the significance of the change in likelihood, thereby testing whether inclusion of the factor improves the fit of the model.

Model 1: Testing the effect of removing 'treatment' on the fit of the model.

Model 1a: aerial feeders.

	Df	AIC	BIC	LogLik	ChiSq	Chi df	P
Without Treat	2	289.21	296.1	-142.6			
With Treat	3	238.12	248.46	-116.06	53.089	1	<0.001

Model 1b: pheasants

	Df	AIC	BIC	LogLik	ChiSq	Chi df	P
Without Treat	2	109.216	116.109	-52.608			
With Treat	3	76.834	87.174	-35.417	34.382	1	<0.001

Model 1c: gulls

	Df	AIC	BIC	LogLik	ChiSq	Chi df	P
Without Treat	2	253.35	260.25	-124.68			
With Treat	3	207.6	217.94	-100.8	47.754	1	<0.001

Model 1d: corvids

	Df	AIC	BIC	LogLik	ChiSq	Chi df	P
Without Treat	2	294	300.9	-145			
With Treat	3	223.41	233.75	-108.71	72.588	1	<0.001

Model 1e: pigeons

	Df	AIC	BIC	LogLik	ChiSq	Chi df	P
Without Treat	2	98.87	105.76	-47.435			
With Treat	3	89.767	100.11	-41.884	11.103	1	<0.001

Model 1f: kestrels

	Df	AIC	BIC	LogLik	ChiSq	Chi df	P
Without Treat	2	83.689	90.583	-39.845			
With Treat	3	83.684	94.024	-38.842	2.005	1	0.157

Model 2: testing the effect of removing 'cutdate' on the fit of the model.

Model 2a: corvids

	Df	AIC	BIC	LogLik	ChiSq	Chi df	P
Without cutdate	5	204.104	219.417	-97.052			
With cutdate	6	201.381	219.756	-94.69	4.723	1	0.03

Model 2b: gulls

	Df	AIC	BIC	LogLik	ChiSq	Chi df	P
Without cutdate	5	157.763	173.076	-73.881			
With cutdate	6	156.5	174.875	-72.25	3.263	1	0.071

Model 2c: aerial feeders

	Df	AIC	BIC	LogLik	ChiSq	Chi df	P
Without cutdate	5	137.947	153.260	-63.974			
With cutdate	6	139.377	157.753	-63.689	0.570	1	0.450

Model 2d: kestrels

	Df	AIC	BIC	LogLik	ChiSq	Chi df	P
Without cutdate	5	52.026	67.339	-21.013			
With cutdate	6	53.694	72.070	-20.847	0.332	1	0.565

Model 2e: pigeons

	Df	AIC	BIC	LogLik	ChiSq	Chi df	P
Without cutdate	5	93.473	108.790	-41.736			
With cutdate	6	94.847	113.220	-41.423	0.626	1	0.429

Model 3: testing the effect of removing 'time since cut' on the fit of the model.

Model 3a: aerial feeders

	Df	AIC	BIC	LogLik	ChiSq	Chi df	P
Without time	3	153.974	163.162	-73.987			
With time	4	144.763	157.013	-68.381	11.211	1	<0.001

Model 3b: gulls

	Df	AIC	BIC	LogLik	ChiSq	Chi df	P
Without time	3	207.040	216.230	-100.521			
With time	4	157.240	169.490	-74.618	51.805	1	<0.001

Model 3c: corvids

	Df	AIC	BIC	LogLik	ChiSq	Chi df	P
Without time	3	222.527	231.715	-108.264			
With time	4	200.026	212.276	-96.013	24.502	1	<0.001

Model 3d: kestrels

	Df	AIC	BIC	LogLik	ChiSq	Chi df	P
Without time	3	70.152	79.340	-32.076			
With time	4	65.033	77.284	-28.517	7.119	1	0.008

Model 3e: pigeons

	Df	AIC	BIC	LogLik	ChiSq	Chi df	P
Without time	3	89.588	98.775	-41.794			
With time	4	91.551	103.801	-41.775	0.037	1	0.848

Chapter 5: Use of Uncropped Arable Field Margins by Farmland Birds, After Management by Mowing.

Claudia M. Garratt & Mark J. Whittingham

School of Biology, Newcastle University, Newcastle upon Tyne, UK

Short title: Bird response to grass margin cutting

NB: An appendix has been added (Appendix 5.C) showing Likelihood Ratio Tests for the models reported in this chapter.

Abstract

Capsule: Cutting half-width strips adjacent to the crop in agri-environment field margins will potentially benefit a range of farmland birds.

Aims: To find out how birds use agri-environment grass margins, and how this use will be affected by the cutting of strips in the vegetation.

Methods: 38 transects around 18 fields on four farms had half-width strips mown adjacent to the crop along half their length, at the start of June 2010. All transects were visited either 6 or 7 times to survey for birds throughout June/early July; seven surveys spread throughout the month at three of the farms, and six surveys at the fourth farm. The data were modelled using Generalized Linear Mixed Models (GLMMs).

Results: Pheasants and bird species that feed on insects significantly preferred the cut strips to the uncut strips and control plots, whereas warblers showed a significant negative effect of treatment overall, and Reed Buntings significantly preferred control plots.

Conclusions: Providing some areas of cut grass in agri-environment field margins throughout the year will increase access to food resources for some species of farmland bird, and increase architectural heterogeneity generally.

Introduction

Agriculture has been identified as the most significant current threat to bird species worldwide, and it is likely that this trend will continue in the future (Donald *et al*. 2001, Green *et al*. 2005), with species of other taxa also affected by the current global extinction crisis (Thomas *et al*. 2004), probably driven most strongly - terrestrially at least - by land use change (Sala *et al*. 2000). The provision of un-cropped field margins (such as grass strips, wildflower strips and naturally regenerated weedy strips) on arable farms is an integral part of agri-environment measures designed to halt the loss of biodiversity caused in recent times by the intensification of agriculture (Krebs *et al*. 1999, Donald *et al*. 2001, Stoate *et al*. 2001, Butler *et al*. 2007). Thus the UK Biodiversity Action Plan (BAP) identifies arable field margins as one of the UK’s 45 priority habitats (Perkins *et al.* 2002; Grice *et al*. 2007).

The way in which boundaries of agricultural fields are managed has considerable bearing on their efficacy for the conservation of biodiversity on farmland. Boundary options, including margin strips, feature prominently in the prescriptions of many of Europe’s Agri-environment Schemes (AESs), and - due in part to the ease of implementation (Vickery *et al.* 2004) - they are currently one of the most popular options within English AESs (Butler *et al*. 2007, Henderson *et al.* 2007, Vickery *et al*. 2009, Davey *et al*. 2010). Current Entry Level Scheme (ELS) prescriptions for the management of 6 m margins state that cutting of margins should not take place until after mid July (Natural England 2010), which is primarily to protect ground nesting birds (Vickery *et al*. 2004, Phil Grice pers comm.), but also allows many wildflower species to seed (Vickery *et al*. 2004). In the case of birds, targeted cutting regimes have been shown to potentially benefit some species such as the Common Kestrel *Falco tinnunculus* (Aschwanden *et al*. 2005, Whittingham & Devereux 2008, Garratt *et al*. 2011), other raptors (Baker & Brooks 1981, Aschwanden *et al*. 2005), Yellowhammer *Emberiza citrinella* (Douglas *et al*. 2009, Perkins *et al*. 2002) and Common Starling *Sturnus vulgaris* (Devereux *et al*. 2006), and - along with selective sowing – can be a tool to maintain or even increase plant species richness (in the short term, at least) and prevent succession to a climax community (Gathmann *et al*. 1994, Perkins *et al*. 2002, Vickery *et al*. 2009, Woodcock *et al*. 2009, Smith *et al*. 2010). Plant species richness and structural heterogeneity increase invertebrate species richness and community complexity (Gathmann *et al*. 1994, Vickery *et al*. 2002, Vickery *et al*. 2009, Woodcock *et al*. 2009), which are likely to be of benefit to both granivorous and insectivorous farmland birds. In the short term, cutting is one of the quickest ways to increase structural and compositional heterogeneity in vegetation (Vickery *et al*. 2009), and it has been suggested that increasing habitat heterogeneity, at the scale of field, farm and landscape, will be the key to reversing the biodiversity declines (Perkins *et al*. 2000, Benton *et al*. 2003, Whittingham 2007, McMahon *et al*. 2008).

For the birds which have been shown to preferentially use areas of cut-grass, it seems likely that this is due to improved accessibility of their prey (Aschwanden *et al*. 2005, Douglas *et al*. 2009, Garratt *et al*. 2011). In the case of grassland hunting raptors such as the Kestrel, longer, more dense vegetation almost certainly holds a greater abundance of their most profitable and therefore preferred prey; small mammals (Arlettaz *et al*. 2010), but foraging theory states that availability is a better predictor than abundance of where predators chose to hunt (Stephens & Krebs 1986). The beneficial effects of cut-grass may be quite fleeting for Kestrels, as well as for other farmland birds (Devereux *et al*. 2006, Garratt *et al*. 2011, Peggie *et al*, in press), but it has been shown that when grass is cut a proportion of any small mammals present may remain in those areas as long as some element of cover (i.e. the grass cuttings) remains (Meunier *et al*. 1999, Jacob & Brown 2000, Jacob 2003, Jacob & Hempel 2003, Garratt *et al*. in review). Therefore, it seems possible that targeted cutting of patches or strips within AES grass margins, while leaving other areas long as reservoirs for prey, could be of benefit to Kestrels as well as for some other declining farmland specialists (Douglas *et al*. 2009). While the cutting of the outer 3m of margin is already permitted under ELS management, the late July timeframe for this management is possibly too late to be of much benefit to Kestrels as it is after the end of their breeding season, and it would appear that currently productivity is their principle limiting demographic factor (Vickery *et al*. 2004). However, before any change to the rules regarding cutting could be proposed, it is important to understand what the effect of any such measures would be on all of the birds that use and rely on AES margins within the farmed landscape. To this end, we here report the results of a split-plot design experiment looking at the use of margins under different cutting treatments by a range of guilds of farmland birds.

Our study builds on a 2007 study by Henderson *et al*., however there are several important differences, including the ‘split’ nature of the experimental plots (Fig. 5.1); in the Henderson study each plot represented a different treatment, and all treatments were recently sown - the vegetation in this study was either naturally regenerated, or sown so long ago that natural colonists had taken over. It was also cut in the summer rather than the spring or autumn, and to an average height of 10 cm rather than 15 cm; in their 2004 study Devereux and colleagues showed that swards mown to a height of 13 cm were treated as long grass by foraging starlings *Sturnus vulgaris*. Additionally, in the present study only birds actually using the margins strip itself were counted, as we wanted to find out if and how birds used the different areas of treatment and control.

Methods and Materials

Strip transect surveys took place in June and July of 2010, on four arable farms with AESs in Cambridgeshire, England (Appendix 5.A). AES margins on four (N=2) or five (N=2) fields on each farm contained between one and four transects per field (i.e. in total margins around 18 fields were sampled). Most of the margins were ELS 6 m margins (N=23), with the exception of the margins at Farm 1 (N=10) which were 6 m Countryside Stewardship Scheme (CSS), and those at Farm 2, which were a mixture of 3 m (N=5) and 6 m (N=1) CSS and 3 m ELS (N=3) margins. The most prevalent crop within fields used for this experiment was Wheat (N=8), followed by Barley (N=3), Bean (N=3), Beet (N=2) and Linseed (N=2). In all but two cases, where there was a farm track between the fields, the boundary between the margin and the next field was formed by a *c*. 2 m wide overgrown ditch. For the most part, the fields used on each farm were adjacent to one another, so margins which were on opposite sides of the same ditch were surveyed at least half an hour apart, to ensure that any use of the margin observed was an independent event rather than that the bird sighted in margin ‘B’ was there simply because it had previously been flushed from margin ‘A’. On the rare occasions this was not possible, where the bird that was flushed went was observed and if seen on the margin on the other side it was not counted. In this way we ensured that a bird’s presence on any individual treatment was not an artefact of activity by the surveyor.

Prior to cutting at the beginning of June 2010, margins were measured along their length, and at the same time checked for the presence of ground nesting birds; any margins where possible nesting birds were found were excluded from the experiment. On two of the farms, some transects stretched across two margins (i.e. around the corner of a field), whereas on the other two farms each margin constituted one transect. Transects did not necessarily take up the entire length of a margin, in which case the point where the transect ended was marked with a bamboo cane with a yellow marker attached, and any birds seen in the margin but beyond this marker were not counted. The area of each transect was then halved along its length to give treatment and control. Following this, a half width strip of grass adjacent to the crop in the treatment halves of the plots was cut by the farmer using tractor-towed agricultural mowing machinery, set to a cutting height of 3 inches (7.5 cm) (Fig. 5.1). Due to the unevenness of the ground, this resulted in a cut sward with a mean height of 10.2 cm (range 3.5-17.1 cm). The half width strip which was left un-cut next to the cut strip was also designated as treatment (‘uncut’), on the grounds that it was not totally undisturbed; there may have been an effect on the uncut section of treatment of reduction of area, due to half the margin width at that point being cut, and also of being adjacent to cut grass rather than just crop or ditch. The first survey of each farm was carried out within four days of the margin being cut, and then a further five (Farm 2) to six (all other farms) surveys were carried out, at various different times of day, over the next four and a half weeks (last survey between 23 (Farm 2) and 30 (Farm 3) days after cut). Surveys were carried out using a form of strip transect methodology after Conner & Dickson (1980), with each margin/pair of margins constituting a strip transect; transects were walked at the midline of the margin, ensuring sufficient noise disturbance to flush any birds hiding in the long grass. Any bird seen within treatment uncut, treatment cut or control sections of the margin was identified, noted and its position marked on a map. The distance between the bird and the observer was estimated, and the number of metres (0-3) perpendicular to the line being walked and, in the case of control, whether it was on the side of the transect nearer to the ditch or to the crop. Any obvious behaviour (i.e. foraging, perching etc.) was also noted.

Statistical analysis

Due to the problems of the lack of independence of transects due to several of them being located in the same fields, and also of overdispersion in the data, data from transects were grouped by field and many of the bird species sighted were not used in the final statistical analysis due to insufficient data points (see Appendix 5.B). Of those that were used, some species were grouped according to diet, behavioural parameters and habitat requirements. Hence the data for Yellow Wagtail *Motacilla flava* and Meadow Pipit *Anthus pratensis* were grouped into the broad category ‘insectivores’, and Whitethroat *Sylvia communis*, Reed Warbler *Acrocephalus scirpaceus* and Sedge

Warbler *Acrocephalus schoenobaenus* were grouped as ‘warblers’. Reed Bunting *Emberiza schoeniclus*, Common Pheasant *Phasianus colchicus* and Skylark *Alauda arvensis* were all included in the analysis as stand-alone groups. In the case of the two former species, they were the most commonly sighted during surveys and therefore had more data points than the others. Whilst the number of Skylark records was limited, they were more commonly sighted than many of the other species which were not included, and we felt that their importance as a target for conservation on farmland - and therefore for several AES measures - merited their inclusion as a stand-alone group in the statistical analysis. However, we acknowledge that the power of test for them will be considerably reduced.

We used Generalized Linear Mixed Models (GLMMs) to model presence (probability of sighting) of the different functional groups of farmland birds within different treatments on fields, with a binomial error structure and a logit link function. Field and farm were included in all models as nested random effects, and treatment (i.e. cut, uncut or control) was a fixed factor. An offset for area was also included as the treatments were not of equal size (Fig. 5.1). Number of days since cut was a continuous variable. We ran all models for all five of the species groups.

Model 1: Tests the hypothesis that there is no difference in the probability of sighting a bird in the treatment (uncut and cut combined) and control plots (Fig. 5.1, A+B vs. C).

Model 2: Tests the hypothesis that there is no difference in the probability of sighting a bird in the cut treatment compared to control plots (Fig. 5.1, A vs. C).

Model 3: Tests the hypothesis that here is no difference in the probability of sighting a bird in the uncut treatment compared to control plots (Fig. 5.1, B vs. C).

Model 4: Tests the hypothesis that here is no difference in the probability of sighting a bird in the cut treatment compared to uncut treatment plots (Fig. 5.1, A vs. B).

All models were also run a second time, including an interaction term between treatment and ‘day’ (days since cut), which tests whether the difference in sighting probability between treatment and control plots changes with increasing time since cutting. All analyses were performed using R (version 2.9.1) (R Development Core Team 2009) and the package *lme4* v.0.99875.9 (Bates 2008). We present the parameter estimate (b) ± 1 standard error and the test statistic (z) in the results, and for all tests a significance level of 0.05 was used.

Draper & Smith (1981) suggest that between five and ten replicates is needed for each predictor. Thus we had data from 18 fields (the unit of replication in this experiment) and we fitted either two or three predictors to our models as described above.

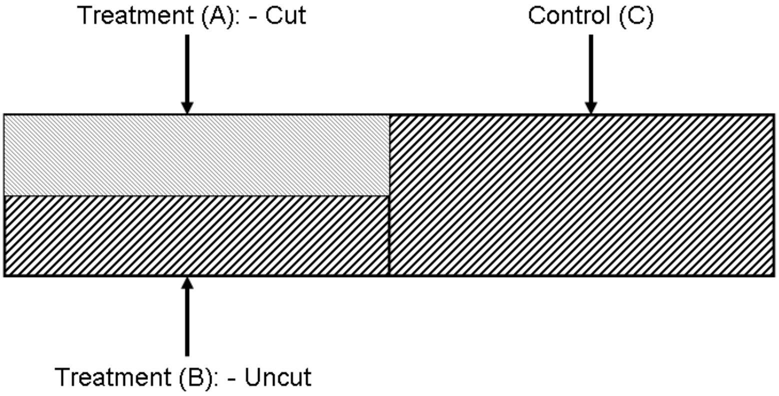

Figure 5.1. The split-plot design of the margin transects. A half width strip was cut adjacent to the crop (A), along half the length of the field margin. The other half-width section, adjoining the boundary of the field, was left long, to give treatment: ‘uncut’ (B). The remainder of the margin was also left long, to act as a control (C).

Results

A total of 263 birds, and 21 identified species, were sighted during the 27 surveys carried out at all farms (Appendix 5.B). Of these, many only occurred once, and were therefore excluded from statistical analysis. More birds were sighted in control than in either of the other treatments (N=118), but this is largely due to the high numbers of Reed Buntings (N=98) which showed a strong preference for the control plots (Model 1) and which were by far the most commonly sighted bird in the margins, with nearly four times as many sightings than the next most numerous species, the Reed Warblers (N=25) and Pheasants (N=24). If the data for Reed Buntings is excluded, there were more sightings of birds in the cut strips (N=65) than in control (N=58) or uncut (N= 42), although it is unlikely that these figures would be statistically significant.

Model 1 - Control vs. Treatment: (Table 5.1) Both Reed Buntings (b = -1.3 ± 0.326, z = -3.990, *P* < 0.001) and warblers (b = -0.89714 ± 0.33760, z = -2.657, *P* = 0.008) showed a significant negative response to treatment. Insectivores showed a near significant positive response to treatment (b = 1.71260 ± 0.88490, z = 1.935, *P* = 0.053), probably driven largely by their positive response to cut strips (as seen in Model 2).

Model 2 – Cut vs. Control: (Table 5.1) Insectivores (b = 2.755 ± 0.885, z = 3.113, *P* = 0.002) and Pheasants (b = 1.337 ± 0.536, z = 2.495, *P* = 0.013) showed a significant preference for cut plots over control plots, and Skylarks also showed a positive response to 'cut' that was near significant (b = 1.662 ± 0.955, z = 1.742, *P* = 0.082). Reed Buntings showed a significant negative response to cut plots (b = -0.89344 ± 0.411, z = -2.172, *P* = 0.03).

Model 3 – Uncut vs. Control: (Table 5.1) None of the results for this model were significant, but warblers showed a near-significant positive response to uncut plots (b = 0.632 ± 0.349, z = 1.813, *P* = 0.07).

Model 4 – Cut vs. Uncut: (Table 5.1) Pheasants showed a significant negative response to uncut plots (b = -1.978 ± 0.819, z = -2.417, *P* = 0.0157), and insectivores a near significant negative response (b = -1.482 ± 0.768, z = -1.929, *P* = 0.054).

In all models, 'Day' (days since cut) was a significant or near-significant negative factor for insectivores and Reed Buntings, although it was a stronger factor for the former (Table 5.1, Fig. 5.2). The other three species groups showed no significant response to the number of days which had passed since cutting, although again Figure 5.2 shows a downward trend for the Warblers. The results for models including an interaction term were non-significant in all cases - other than a near-significant negative interaction between cut and day for insectivores (b = -0.162 ± 0.098, z = -1.652, *P* = 0.099) for Model 2 (Cut vs. Control) – and therefore will not be considered further in this study.

Table 5.1. Results of the four models for each of the five species groups.

	Insectivores			Reed Buntings			Pheasants			Skylarks			Warblers		
	Cut	Uncut	Day	Cut	Uncut	Day	Cut	Uncut	Day	Cut	Uncut	Day	Cut	Uncut	Day
Cont v Treat	Nr +	Nr +	- - -	- - -	- - -	-	0	0	0	0	0	0	- -	- -	0
Cut v Cont	+ +	/	- - -	-	/	Nr -	+	/	0	Nr +	/	0	0	/	0
Uncut v Cont	/	0	Nr -	/	0	Nr -	/	0	0	/	0	0	/	Nr +	0
Cut v Uncut	/	Nr -	- - -	/	0	0	/	-	0	/	0	0	/	0	0

The prefix ‘Nr’ denotes a near significant result (significant at the 10% level). The symbol ‘+’ denotes a positive response, and ‘-‘ a negative one, with one symbol indicating a significant result ($P < 0.05$), two symbols a strongly significant result ($P < 0.01$), and three symbols a very strongly significant result ($P < 0.001$). A ‘0’ shows no significance, and ‘/’ indicates that that factor was not applicable in that particular model, i.e. ‘cut’ was not applicable in a model looking at the effect of uncut vs. control.

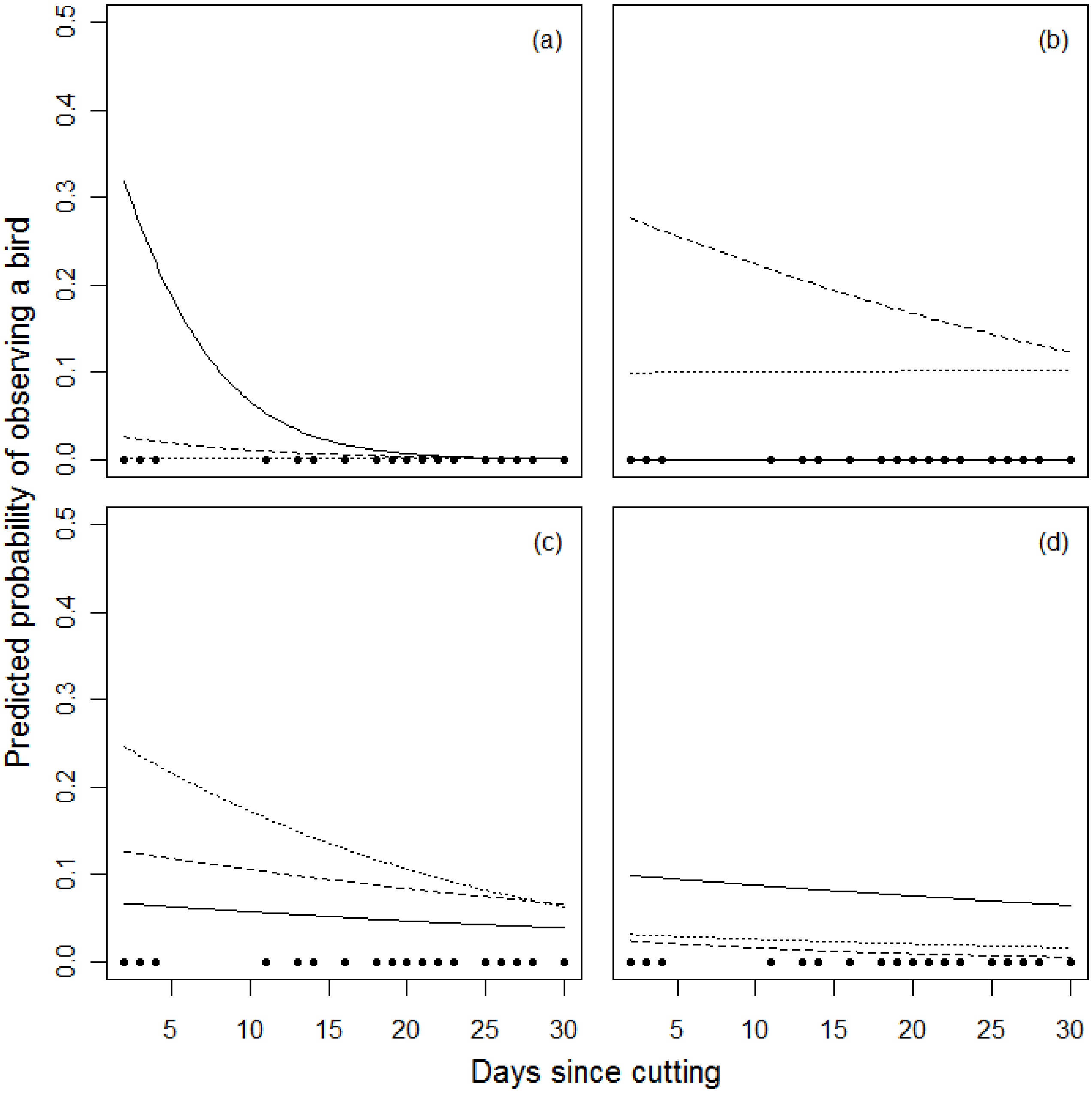

Figure 5.2. The predicted probability (%) of seeing: (a) an insectivore; (b) a warbler; (c) a Reed Bunting and (d) a Pheasant in: 'cut' (solid line); 'uncut' (dashed line) or 'control' (dotted line) at any given day during the survey period. At day zero i.e., there is a higher probability of seeing insectivores and Pheasants in 'cut' than in either of the other treatments, and a greater probability of seeing a Reed Bunting in 'control' than in either of the other treatments. Warblers show a preference for 'uncut' treatment. The predicted probability of seeing any of the groups (a-d) in any of the treatment declines throughout the experiment.

Discussion

As expected, and in line with other studies (e.g. see Atkinson *et al*. 2005), the effects of cutting differed widely for the various groups of birds studied. Insectivores – in this case Yellow Wagtails and Meadow Pipits, showed a significant preference for the cut strips compared to control plots and a near significant preference for the cut strips vs. the uncut strips. This reinforces the findings of Atkinson *et al*. 2004, that many species of insectivorous birds show a negative relationship with sward height despite the usually greater abundance of food resources in longer swards. There was also a near significant negative interaction with the passing of time, with the effect of this relationship declining as the number of days since cut increased. Pheasants and Skylarks also showed a preference for the cut strips compared to control plots, significant in the case of the Pheasant, and near significant for the Skylark – although in the latter instance small sample size will have decreased the power of test. Pheasants also significantly preferred cut plots to uncut treatments. Conversely, Reed Buntings were significantly negatively associated with the cut strips – indeed with treatment overall – and showed a significant preference for the control sections of transects. Henderson *et al*. 2007 grouped buntings and Skylarks for the purposes of analysis, but it would appear from these results that they may use margins differently, although the sample sizes in our study for Skylarks were small so this difference should be treated with caution. Warblers showed a significant negative effect of treatment overall, compared to control, but a near significant positive relationship with the uncut sections of treatment when compared to control, an interesting trend which would not have shown up without the split-plot design of this experiment. It is unclear whether this is just coincidence or a true result, and if the latter; why this might be, so it merits further research. As can be seen from Appendix 5.B, more of the species which were sighted - but in insufficient numbers to be included in the statistical analysis - were seen using the cut strips than either of the other treatments (including the only sighting of grass margin use by a Corn Bunting *Emberiza calandra*, a very strongly declining farmland specialist). The surveyor was never further away than 3 m from any bird hiding in the margin, and incidences of birds lying low and not being flushed under these conditions will have been highly improbable.

Day was a significant or near-significant negative factor in almost all of the models for both insectivores and Reed Buntings. In the case of the insectivores this is fairly intuitive, as over the course of the month that surveys took place the grass growth rates will have been relatively high, and therefore the positive effect of cutting seen for insectivores will have diminished fairly rapidly as their invertebrate prey dispersed and the grass grew. The temporal aspect of cut-grass as a resource was investigated by Peggie *et al*. (in press), and the benefits found to be relatively short-lived, although as discussed by Douglas *et al*. 2009, the cut patches will probably still be of use by increasing access to the food resources in the longer grass areas. The negative effect of day seen for Reed Buntings is a little harder to explain, as they showed a preference for control patches which on the face of it should not have changed a great deal with the passing of time. The most likely explanation for this would seem to be the decline in bird activity generally as the summer progresses – the end of the survey period took place in early July which was towards the end of the breeding season for many species of farmland birds, and therefore a decline in activity rates is perhaps to be expected.

The support that our results give to the argument that some species of farmland bird would benefit from having patches of cut-grass available to them in margins during their breeding season is clear. However, these proposed benefits are largely to do with provision of foraging opportunities, and the current prescriptions for cutting of margins are primarily concerned with birds which use the margins for nesting, which would almost certainly be adversely affected to some degree by earlier cutting. It should be noted though that around half of the species of declining farmland birds which are included on the FaBI are probably limited primarily by overwinter food supply rather than nest site availability (Newton 2004, Henderson *et al*. 2007). The scope of this study does not cover the effects that cutting strips adjacent to the crop would have on birds which nest within margins; it is suggested that birds which do nest in the margins (i.e Grey and Red-legged Partridge *Perdix perdix* and *Alectoris rufa*) like to nest nearer the hedgerow (Rands 1987**,** Douglas *et al*. 2009) so possibly would be relatively unaffected by - in the case of a 6 m margin for example - the 3 m adjacent to the crop being cut. Alternatively, it is possible that even if half the width of the margin adjacent to the hedge was left uncut as a refuge for ground nesting birds, the half next to the crop being cut could make the uncut areas a less optimal nesting habitat, due for instance to edge effects and a reduction of the area of un-cut habitat. This would need to be studied, and over a fairly long period of time, before the overall effect of earlier cutting on farmland birds could be discerned.

The effect of early cutting on birds is of course not the only consideration. There is a wealth of literature on the effects of different margin management regimes specifically on plant and invertebrate communities for example, but the associated interactions and therefore the knock-on effects on the higher trophic orders also has to be considered. Long term plant species richness and community composition is significantly affected and mediated by different cutting regimes. Smith and colleagues (2010) found that over the 13 year timeframe of their experiment plant species richness in managed arable field margins declined overall, although it increased initially in swards mown twice annually. However, they found that species composition of both sown and naturally regenerated swards (including loss of annuals from naturally regenerated margins), and control of pernicious weed species could all be modified by mowing, and that this effect varied with the timing and frequency of mowing. Therefore, targeted mowing of sections of margin could have added benefits for farmers by helping to control some pernicious weed species close to the crop – effectively creating a buffer strip. This could be particularly effective if cutting earlier in the season, as it would remove some of the more vigorous species such as Creeping Thistle *Circium arvense* from directly adjacent to the crop before they have had a chance to set seed. Smith and colleagues found lower instances of some plants, including Thistle, Nettle *Urtica dioica* and Couch Grass *Elytrigia repens*, in margins that had been cut twice a year (including a cut earlier in the season). Some other species such as Black Grass *Alopecurus myosuroides* which is particularly damaging to winter crops (Rothampsted Research 2011), have also been shown to be controllable by mowing (Smith *et al*. 2010), and therefore the management regime can be specifically targeted at dominant and troublesome species on any given farm. In terms of increasing heterogeneity, it also seems likely that differing treatment within margin will lead to margins which differ throughout both in sward architecture, and in species richness and composition, leading to a wider variety of available niches.

In the case of invertebrates, Gathmann and colleagues (1994) showed plant species richness to be greatly increased by cutting in early-successional set-aside fields, thus doubling species richness of bees. They found cutting of early-successional habitats seemed to benefit insects and plants in general, and that older grassland should show the greatest insect diversity when both mown and un-mown parts are present (Gathmann *et al*. 1994). Conversely, there are also several studies which suggest that invertebrates are in general negatively affected by cutting (Feber *et al*. 1996, Perkins *et* *al*. 2002, Humbert *et al*. 2010), but have been shown in the case of some genera to be less severely reduced by cutting earlier in the season (May rather than late-June, or July; Morris 1979, 1981, Morris & Lakhani 1979, Feber *et al*. 1996, Baines *et al*. 1998, Vickery *et al*. 2001, 2009). Some insectivores and species of raptor, such as the Kestrel, would probably benefit from having increased accessibility to their prey while provisioning for growing young, i.e. mid-May to late June, and recent evidence suggests that many grass swards become too dense and tall to allow birds to easily access their prey as the season progresses (Hart *et al.* 2006, Henderson *et al.* 2007, Douglas *et al.* 2009, Vickery *et al.* 2009). The timing of cutting is therefore extremely important for plants, invertebrates, and birds which use the margins. It should also be noted any adverse effects seen due to summer cutting are presumably already a factor for the currently allowed mid-July cutting timeframe.

Conclusions

What is clear from our work is that cutting strips into margin vegetation at various different times of the year, while leaving others untouched, provides a wide variety of habitats, with differing structure and composition of vegetation which alters foraging opportunities for the herbivores and predators which depend on the food resources they support. This study highlights the preferences of shorter vegetation by insectivores and Pheasants whilst warblers and Reed Buntings preferred longer vegetation.

Previous work suggests that the interactions between different margin inception and management techniques, and all the different species of plants, invertebrates, birds and mammals that use them are likely to be complex. Therefore, while we can draw inferences as to the effects (on weeds, invertebrates, birds etc.) of cutting a strip next to the crop from the multitude of literature on the subject (i.e. see Feber *et al*. 1996, Perkins *et al*. 2002, Douglas *et al*. 2009, Vickery *et al*. 2009, Smith *et al*. 2010), these effects will depend on what time of year the cutting takes place, and margin characteristics such as age and type; i.e. whether it is naturally regenerated or sown. Therefore to discern whether cutting of strips of margin earlier in the season than at present could have some beneficial effects for weed control, and some invertebrate and bird species as well as raptors, a long term study is needed where the recommended prescriptions are carried out on different types of margin, and the effect on a wide variety of taxa is then studied over a prolonged period of time (see Smith *et al*. 2010).

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Appendix 5.A

Survey sites

Farm 1 (EF)

Situated at 52°24’7.7’’N -0°0’4.56’’W (co-ordinates for the farmhouse) Seven surveys were carried out at 2, 14, 19, 21, 26 and 27 days. There were eight transects on five fields of; Wheat (N=3) and Barley (N=2), all of which were managed under Countryside Stewardship Scheme (CSS) with 6 m margins. Of those eight transects, two went around the corners of fields and therefore covered two margins (Transects 1 & 8). The average length of transect was 239 m (range 149-470 m). The field margins were originally sown, over five years ago (before the current farmer took over management of the site, therefore there is little information available) with a non-standard mix containing four to five different grasses including Cocksfoot *Dactylis glomerata* and Smooth Meadow Grass *Poa pratensis*, and various seed plants (farmer – personal communication). However, they were sown long enough ago that they have now been colonised by naturally occurring weeds. Margins of Field 1 were long grass (mean 118.7 cm) with few weeds. Other margins were very tall (mean 121.6 cm, range 53.6–157.7 cm) dense, weedy margins, made up mainly of grasses such as False Oatgrass *Arrhenatherum elatius*, Perennial Ryegrass *Lolium perenne*, Timothy-grass *Phleum pratense* and Common Wild Oat *Avena fatua*, along with Thistles *Circium arvense*, reeds (Poaceae family), Nettles *Urtica dioica*, Burdock *Arctium* spp. and umbellifers such as Hemlock *Conium maculatum*, and Common (*Heracleum sphondylium*) and Giant (*H. mantegazzianum*) Hogweed.

Farm 2 (QH)

Situated at 52°19’45.26’’N 0°5’38.08’’W (co-ordinates for the farmhouse). Six surveys were carried out at 2, 11, 16, 18, 21 & 23 days. There were seven transects (three of which went around corners of fields - Transects 6, 7 & 8) on four fields of; Bean (N=2) and Linseed (N=2), two of which were managed under Environmental Stewardship Entry Level Scheme (ELS), with naturally regenerated 3 m margins, and two under CSS, with some 3 m margins and one 6 m. The average length of transect was 442 m (range 285-621 m). The CSS margins were originally sown with a basic mix over five years ago, but they have now been largely colonised by un-sown naturally regenerating plants (farmer, pers comm.). The margins at this farm were all grassy and fairly short –

in some cases very short (*c*. 20-30 cm) and predominantly grass (including Cocksfoot, Meadow Grass *Poa trivialis* and Yorkshire Fog *Holcus lanatus*) but with a few weed species such as Thistle and Broad-leaved Dock *Rumex obtusifolius*.

Farm 3 (RM)

Situated at 52°30’31.63’’N 0°3’4.89’’W (co-ordinates for the barn at the centre of the survey area). Seven surveys were carried out at 3, 13, 20, 22, 27, 28 & 30 days. There were 14 transects on four fields of; Wheat (N=3) and Beet (N=1), all of which were managed under ELS with 6 m margins. The average length of transect was 213 m (range 79-450 m). The margins were originally sown with a basic grass mix over five years ago (farmer, pers comm.), and now consist largely of waist-high grasses (inc. False Oatgrass, Cocksfoot, Timothy-grass, Perennial Ryegrass and Smaller Cat’s-tail *Phleum bertolonii*) with some Thistles, Nettles, Broad-leaved Dock and umbelliferae (mean vegetation height 70.7 cm, range 30.1–121.2 cm).

Farm 4 (RH)

Situated at 52°27’9.56’’N -0°0’25.41’’W (co-ordinates for the farmhouse). Seven surveys were carried out at 4, 14, 19, 21, 25, 26 & 27 days. There were nine transects on five fields of; Wheat (N=2), Barley, Bean and Beet, all of which were managed under ELS with naturally regenerated 6 m margins. The average length of transect was 344 m (range 267-380 m). Two margins were predominantly grass *c*. 1.5 m in height (inc. Common Wild Oat, False Oat-grass and Smaller Cat’s-tail) the rest were very tall (range 82.1 – 151 cm), very dense vegetation made up mostly of Thistles, with some Common Hogweed, Broad-leaved Docks and large umbellifers. Reeds from the ditch extended out half the width of most margins.

Appendix 5.B

Species	Control	Treatment Cut	Treatment long	Total
Yellow wagtail *Motacilla flava*	1 (1)	7 (13)	1 (1)	9 (15)
Reed Bunting *Emberiza schoeniclus*	34 (60)	12 (14)	19 (24)	65 (98)
Sedge warbler *Acrocephalus schoenobaenus*	11 (15)	0	5 (6)	16 (21)
Reed warbler *Acrocephalus scirpaceus*	7 (11)	0	10 (14)	17 (25)
Whitethroat *Sylvia communis*	6 (8)	0	8 (10)	14 (18)
Skylark *Alauda arvensis*	2 (2)	5 (7)	0	7 (9)
Pheasant *Phasianus colchicus*	7 (8)	12 (14)	2 (2)	21 (24)
Meadow pipit *Anthus pratensis*	1 (1)	3 (6)	3 (3)	7 (10)
Kestrel *Falco tinnunculus*	0	1 (1)	0	1 (1)
Barn owl *Tyto alba*	0	2 (2)	0	2 (2)
Song thrush *Turdus philomelos*	0	1 (1)	0	1 (1)
Blackbird *Turdus merula*	0	1 (1)	0	1 (1)
Red-legged partridge *Alectoris rufa*	0	1 (2)	0	1 (2)
Grey partridge *Perdix perdix*	1 (2)	0	0	1 (2)
Carrion crow *Corvus corone*	1 (1)	2 (4)	0	3 (5)
Mallard *Anas platyrhynchos*	0	1 (2)	0	1 (2)
Wood Pigeon *Columba palumbus*	0	3 (3)	0	3 (3)
Corn bunting *Emberiza calandra*	0	1 (1)	0	1 (1)
Dunnock *Prunella modularis*	0	0	1 (1)	1 (1)
Wren *Troglodytes troglodytes*	0	1 (1)	0	1 (1)
Great tit *Parus major*	1 (1)	0	0	1 (1)
Unidentified	8 (8)	7 (7)	3 (5)	18 (20)
Total	80 (118)	60 (79)	52 (66)	192 (263)
Total visits	122	122	122	366

The transect data grouped by field (the unit of replication used in the analysis). The first number in each instance is the number of separate occasions each species was present within the different treatments (e.g. there were 27 repeated visits to 18 fields and thus of 486 possible occurrences reed buntings were seen on 65 occasions), with the count data (total number of individual birds sighted) in parentheses.

Appendix 5.C

Likelihood Ratio Tests comparing the fit of models with a given factor included to those with the factor excluded. Presented are AIC, BIC and log-likelihood of each model, and a chi-squared test testing the significance of the change in likelihood, thereby testing whether inclusion of the factor improves the fit of the model.

Model 1: Control vs. Treatment

Model 1a: Insectivores. Testing the effect of removing ‘day’ (time since cut) on the fit of the model

	Df	AIC	BIC	logLik	Chisq	Chi df	Pr(>Chisq)
Without day	4	121.890	137.510	-56.948			
With day	5	104.060	123.570	-47.029	19.838	1	<0.001

Testing the effect of removing ‘treatment’ on the fit of the model; treatment reference category is ‘Control’

	Df	AIC	BIC	logLik	Chisq	Chi df	Pr(>Chisq)
Without treat	4	107.130	122.740	-49.566			
With treat	5	104.060	123.570	-47.029	5.075	1	0.024

Model 1b: Warblers. Testing the effect of removing ‘day’ (time since cut) on the fit of the model

	Df	AIC	BIC	logLik	Chisq	Chi df	Pr(>Chisq)
Without day	4	261.050	276.660	-126.520			
With day	5	262.460	281.980	-126.230	0.581	1	0.446

Testing the effect of removing ‘treatment’ on the fit of the model; treatment reference category is ‘Control’

	Df	AIC	BIC	logLik	Chisq	Chi df	Pr(>Chisq)
Without treat	4	267.440	283.050	-129.720			
With treat	5	262.460	281.980	-126.230	6.970	1	0.008

Model 1c: Reed buntings. Testing the effect of removing ‘day’ (time since cut) on the fit of the model

	Df	AIC	BIC	logLik	Chisq	Chi df	Pr(>Chisq)
Without day	4	275.730	291.340	-133.860			
With day	5	273.870	293.380	-131.930	3.860	1	0.049

Testing the effect of removing ‘treatment’ on the fit of the model; treatment reference category is ‘Control’

	Df	AIC	BIC	logLik	Chisq	Chi df	Pr(>Chisq)
Without treat	4	287.800	303.410	-139.900			
With treat	5	273.870	293.380	-131.930	15.934	1	<0.001

Model 2: Cut vs. Control

Model 2a: Insectivores. Testing the effect of removing ‘day’ (time since cut) on the fit of the model

	Df	AIC	BIC	logLik	Chisq	Chi df	Pr(>Chisq)
Without day	4	89.919	103.908	-40.960			
With day	5	76.537	94.023	-33.268	15.382	1	<0.001

Testing the effect of removing ‘treatment’ on the fit of the model; treatment reference category is ‘Control’

	Df	AIC	BIC	logLik	Chisq	Chi df	Pr(>Chisq)
Without treat	4	88.590	102.578	-40.295			
With treat	5	76.537	94.023	-33.268	14.053	1	<0.001

Model 2b: Pheasants. Testing the effect of removing 'day' (time since cut) on the fit of the model

	Df	AIC	BIC	logLik	Chisq	Chi df	Pr(>Chisq)
Without day	4	129.560	143.550	-60.779			
With day	5	131.180	148.660	-60.589	0.380	1	0.537

Testing the effect of removing 'treatment' on the fit of the model; treatment reference category is 'Control'

	Df	AIC	BIC	logLik	Chisq	Chi df	Pr(>Chisq)
Without treat	4	136.120	150.110	-64.059			
With treat	5	131.180	148.660	-60.589	6.940	1	0.008

Model 2c: Skylark. Testing the effect of removing 'day' (time since cut) on the fit of the model

	Df	AIC	BIC	logLik	Chisq	Chi df	Pr(>Chisq)
Without day	4	69.390	83.379	-30.695			
With day	5	71.330	88.816	-30.665	0.060	1	0.806

Testing the effect of removing 'treatment' on the fit of the model; treatment reference category is 'Control'

	Df	AIC	BIC	logLik	Chisq	Chi df	Pr(>Chisq)
Without treat	4	73.596	87.585	-32.798			
With treat	5	71.330	88.816	-30.665	4.266	1	0.039

Model 2d: Reed bunting. Testing the effect of removing 'day' (time since cut) on the fit of the model

	Df	AIC	BIC	logLik	Chisq	Chi df	Pr(>Chisq)
Without day	4	187.410	201.400	-89.703			
With day	5	186.460	203.950	-88.231	2.945	1	0.086

Testing the effect of removing ‘treatment’ on the fit of the model; treatment reference category is ‘Control’

	Df	AIC	BIC	logLik	Chisq	Chi df	Pr(>Chisq)
Without treat	4	189.510	203.500	-90.757			
With treat	5	186.460	203.950	-88.231	5.052	1	0.025

Model 3: Uncut vs. Control

Model 3: Warblers. Testing the effect of removing ‘day’ (time since cut) on the fit of the model

	Df	AIC	BIC	logLik	Chisq	Chi df	Pr(>Chisq)
Without day	4	225.780	239.770	-108.890			
With day	5	227.090	244.580	-108.550	0.684	1	0.408

Testing the effect of removing ‘treatment’ on the fit of the model; treatment reference category is ‘Control’

	Df	AIC	BIC	logLik	Chisq	Chi df	Pr(>Chisq)
Without treat	4	228.310	242.300	-110.160			
With treat	5	227.090	244.580	-108.550	3.217	1	0.072

Model 4: Cut vs. Uncut

Model 4a: Pheasants. Testing the effect of removing ‘day’ (time since cut) on the fit of the model

	Df	AIC	BIC	logLik	Chisq	Chi df	Pr(>Chisq)
Without day	4	103.280	117.270	-47.641			
With day	5	104.940	122.420	-47.468	0.347	1	0.556

Testing the effect of removing 'treatment' on the fit of the model; treatment reference category is 'Cut'

	Df	AIC	BIC	logLik	Chisq	Chi df	Pr(>Chisq)
Without treat	4	111.750	125.740	-51.876			
With treat	5	104.940	122.420	-47.468	8.817	1	0.003

Model 4b: Insectivores. Testing the effect of removing 'day' (time since cut) on the fit of the model

	Df	AIC	BIC	logLik	Chisq	Chi df	Pr(>Chisq)
Without day	4	103.817	117.810	-47.909			
With day	5	83.702	101.190	-36.851	22.115	1	<0.001

Testing the effect of removing 'treatment' on the fit of the model; treatment reference category is 'Cut'

	Df	AIC	BIC	logLik	Chisq	Chi df	Pr(>Chisq)
Without treat	4	85.919	99.908	-38.960			
With treat	5	83.702	101.188	-36.851	4.217	1	0.040

Chapter 6: General Discussion

The results of this thesis have shown a preference by foraging Common Kestrels for shorter grass during the breeding season. The thesis then goes on to report a range of effects of manipulating vegetation height on other species of farmland birds and on key prey of Kestrels, and I go on here to explore the potential for adapting current AES options supported by this new evidence and other pertinent literature.

Summary of Key Results

Some studies have suggested that foraging Common Kestrels preferentially chose mown grass as favoured hunting habitat (e.g. Aschwanden *et al.* 2005, Whittingham & Devereux 2008). The results of the work in Chapter 2 lend support to this argument, with Kestrels being shown to choose foraging habitat non-randomly. Compositional analysis comparing habitat use to availability showed that cut grass was significantly preferred to all other habitats available to the foraging Kestrels within their observed foraging range. This is most likely due to the increased accessibility of prey of higher nutritive value, such as small mammals – a probability that is further hinted at by prey analysis results showing that a greater proportion of small mammal to invertebrate prey was captured over recently cut grass compared to longer, uncut grass. However, it seems probable that the immediate effects of cutting grass are relatively fleeting, and that the newly exposed prey will, in time, disperse out of the cut-grass areas into more dense vegetation where they will be safer from predation. To test whether this is the case, in Chapter 3 I investigated how cutting grass to increase accessibility to small mammal prey for Kestrels would affect availability. Results of previous similar studies have been equivocal, but several found that not all small mammals leave an area when grass is cut (Meunier *et al*. 1999a, Jacob & Brown 2000, Jacob 2003, Jacob & Hempel 2003). My results support this, and show that while many animals do cease to be caught in grassland areas once the vegetation has been cut, a proportion either remain in or return to the cut area in order to continue to exploit resources. This was only the case, however, as long as the cut grass was left *in situ*. Once the hay – and therefore all cover – was removed, very few small mammals were captured in the treatment areas, implying that grass cuttings are considered sufficient cover for some small mammals to continue using an area either for resource or possibly territorial reasons. It was not clear from my experiment for how long this use would continue had the grass cuttings not been removed, or whether the areas of cut grass eventually just become short grass in a functional sense, almost completely uninhabited by small mammals, and if so how long post-cutting this process takes. However, it is obvious that the general composition of a grassland ecosystem changes with time since cutting, as new resources are revealed and then either depleted or dispersed.

The problem of prey accessibility does not affect Kestrels alone, and several other species of birds have been shown to prefer to forage over short or recently cut grass when it is available (e.g. Brough & Bridgman 1980, Perkins *et al.* 2000**,** 2002, Vickery *et al*. 2001, Atkinson *et al.* 2004, 2005, Devereux *et al*. 2006, Whittingham & Devereux 2008, Douglas *et al.* 2009). In Chapter 4, we looked at the temporal aspect of this relationship, and showed that the apparent beneficial effects for some bird species of grass cutting are relatively short lived, and that they decrease for different species at different rates. Of the birds observed, this drop off in use post-cutting occurred most rapidly for Kestrels, and least rapidly for corvids. Some of the bird species observed in our study which prefer to forage in short grass areas employ the same foraging techniques (i.e. ground-probing) as some of our most rapidly declining farmland specialists, and therefore it seems reasonable to extrapolate to them the presumed benefits of cutting to increase prey access. This being the case, cutting strips into uncropped areas on AES farms, and providing these freshly cut strips at various different times throughout the year from the breeding season (i.e. mid April) onwards could be supposed to be potentially beneficial to many different species of farmland bird. However, if there were to be a new AES management option – or a change to existing management options – it is important to know what the resultant effects on other species would be. Consequently, in Chapter 5 I tested the proposed measure on four farms by cutting half width strips into AES grass margins, to see how the species of farmland birds which were present responded. Insectivorous birds such as Meadow Pipits and Yellow Wagtails responded positively to the provision of cut areas – presumably again due to increased accessibility of prey. This is in line with the findings of Douglas *et al.* 2009 in relation to Yellowhammers, and also of Henderson *et al*. 2007 in relation to farmland birds generally. Pheasants and Skylarks also responded positively, whereas Reed Buntings and warblers preferred the longer, uncut grass – in the case of the warblers there was a non-significant tendency to prefer the uncut sections of treatment plots, whereas Reed Buntings significantly preferred control plots, although they were also occasionally also observed foraging in the cut strips.

Potential Use of Spring Cutting as an AES Option

My results, coupled with findings from other studies looking both at birds (e.g. Vickery *et al*. 2001, Aschwanden *et al*. 2005, Henderson *et al*. 2007, Douglas *et al*. 2009) and many other species of animals and plants (Gathmann *et al*. 1994, Feber *et al*. 1996, Perkins *et al*. 2002, Vickery *et al*. 2009), indicate that cutting could be a very useful cost-effective management tool to increase the benefits of uncropped areas on farmland throughout the year for a range of species. Several studies have already suggested that controlled cutting earlier in the season than is currently permitted would be likely to have some beneficial effects (Feber *et al*. 1996, Baines *et al*. 1998, Perkins *et al*. 2002, Douglas *et al*. 2009), and doing this in a mosaic fashion, cutting different areas at different times of the year, would create an architectural and compositional heterogeneity which would provide niches and habitats for a far wider range of species than are currently provided for under management rules for uncropped areas on AES farms. It is also a potentially useful control measure to allow farmers to insert a buffer between their crops and pernicious weed species which are often found in the margins – particularly naturally regenerated ones – and which in some cases set seed before the earliest date that is currently allowed for cutting (Perkins *et al*. 2002). It has been found that most weed seeds which disperse from the margin into the crop (81-97%) are captured within the first 4 m of the crop (De Cauwer *et al*. 2008), therefore a 3 m cut strip adjacent to the crop would be sufficient to prevent the majority of the seeds from the uncut section of margin from invading the adjacent crop.

In trying to decide what an AES option which is good for Kestrels might look like, a useful model is the verges of large roads, which are a favoured hunting habitat of Kestrels (Meunier *et al*. 2000). In the majority of cases, the verges of major roads are managed by having the *c*. 2 m closest to the road mown several times a year (Meunier *et al*. 2000) from the beginning of the growing season (around April) onwards and throughout the summer. The area of the verge further away from the roadside is left long. The tendency of Kestrels to hunt over these habitats (Meunier *et al*. 1999b, 2000, Bautista *et al*. 2004) might suggest that the small mammals which live there venture out of the longer grass where they are safer into the mown areas regularly enough to increase their accessibility and make the verges worthwhile hunting areas. Indeed, Meunier and colleagues (1999a) found Common Voles *Microtus arvalis* to be more numerous in the mown strip of road verges than either the unmown strips or the surrounding landscape during population peaks, and a similar result has been found for Wood Mice *Apodemus sylvaticus* (Bellamy *et al*. 2000). In light of this, I propose that cutting half the width of AES margins, adjacent to the crop, and staggering the cutting throughout spring, summer and autumn (i.e. cutting some margins within the farm one week, then some different margins a couple of weeks later and so on, so that there would always be areas of margins at different lengths/stages of growth). Each margin should be cut a maximum of twice in the year, as frequent cutting reduces plant diversity and species richness (Smith *et al*. 2010), and during the breeding season the margins would need to be walked to check for the presence of ground nesting birds prior to cutting. The half of the margin nearest the hedge/field boundary should be left long as a refuge for ground-nesting birds, and a reservoir for prey species, invertebrates, and plant species which need minimal disturbance (which will also allow them to set seed and persist in the margin). The issue of grass cuttings is a slightly more problematic one. In order to maximise continued small mammal use of the margins, at least some of the grass cuttings would need to be left *in situ*. This would probably also benefit some invertebrates (Baines *et al*. 1998, Smith *et al*. 2008). However, leaving grass cuttings in place also increases soil fertility and encourages dominant competitive plant species such as Nettle *Urtica dioica*, thereby suppressing other plant growth (Vickery *et al*. 2009, Smith *et al*. 2010). Other vegetation tends to perform poorly under a blanket of cover such as cut grass, and Smith and colleagues (2010) found that plant species richness in cut margins was significantly negatively affected by the leaving of grass cuttings. This would need to be investigated more fully in order to find a compromise between maximising cover for small mammals and invertebrate prey, and not preventing opportunities for less vigorous plants to grow, but again the road verges could be looked to for examples. As stated previously it would appear that small mammals continue to use the cut areas of verges, so managing margins in a similar way to verges would be unlikely to be disadvantageous to their benefits for Kestrels. In margins where none are currently present, the provision of perches would also be of potential benefit (upright fence poles at intervals i.e.), as these allow raptors to undertake “perch hunting”, which is less energetically costly than flight hunting – of particular importance in the winter which is often a period of food shortage for raptors

(Village *et al*. 1990) when energy reserves are at their lowest. The presence of fences from which to perch hunt has been proposed as another possible reason for the popularity of motorway verges as hunting grounds (Meunier *et al*. 2000).

Naturally, Kestrels and birds with similar requirements are not the only species for which current uncropped areas on farmland are managed. Many species with many different requirements rely on the field margins in arable landscapes, the interactions are complex (Table 6.1), and coming up with a design which benefits all of them and disadvantages none is an intractable problem. However, bearing this in mind, the idea of increasing heterogeneity in general becomes an even more compelling one – some birds, plants, animals and insects benefit from sowing, some from natural regeneration; some from earlier cutting, some from later cutting, and some from no cutting at all; some from removing grass cuttings, some from leaving them *in situ* (Table 6.1). However, there are some common patterns. In general, cutting in summer (i.e. late June/July) is more detrimental to plant and invertebrate diversity and species richness than cutting in the spring and/or autumn. The frequency of cutting (up to twice-annually) has less impact than the timing, but as frequency of cutting increases so do the deleterious effects. On the whole, sowing with a complex grass and wildflower mix has greater biodiversity benefits than sowing with simpler mixes or leaving margins to regenerate naturally, but natural regeneration provides useful larval host plants for some invertebrates and can allow persistence of rare arable weeds. Sowing tussocky grass mixes is good for small mammals, and provides nest-site opportunities for bumblebees. Therefore, by providing margins of many different types within each farm – i.e. sowing some with complex wildflower and forb mixes, some with tussocky grass mixes, and allowing some to regenerate naturally, and then cutting strips into different margins within-farm throughout the year, cutting each margin a maximum of twice, but minimising summer cutting, and leaving some – but not all – grass cuttings in place, some habitat would be provided for many niches, there would be an almost constant provision of recently mown areas, allowing the benefits of recently mown grass to be available to foragers throughout the summer, and there would always be some areas on farms where access to high quality prey was increased for Kestrels and other raptors. Plus as the vegetation began to grow in the cut areas, while new sections were cut elsewhere, vegetation of varying heights would be provided at a field and farm scale. Benton *et al*. (2003) and Whittingham (2007), amongst others, have suggested that increasing heterogeneity is key to increasing biodiversity on farmland, and in light of this having all AES farms cutting their uncropped areas at approximately the same time after mid July (when cutting is most damaging) would seem to be of limited use; it may control scrub encroachment and help to control some pernicious weeds, but provides little in terms of structural heterogeneity over an extended timescale.

Further Research

Several aspects of my study have hinted at interesting results, confirmation of which require more robust and targeted testing. More detailed study of which types of prey are taken by foraging raptors in different types of habitat, as addressed in a preliminary fashion in Chapter 2, would be useful and interesting. I would also like to continue the small mammal research, but with the individuals marked in order to make some assessment of what proportion of individuals present in an area will continue to use it after removal of vegetation by cutting; whether they are staying after cutting or immigrating back solely to exploit the food in the traps and whether the individuals present post-cutting are entirely different from the ones which were resident pre-cutting. By using PIT tags and automated readers, it may also be possible to deduce how the small mammals are using the area without the need for baited traps which, by their nature, will almost certainly change the mammals' behaviour to some degree. I would also like to investigate how long the mammals will continue using the area if grass cuttings are not removed at all – effectively testing the proposed AES option, preferably in a field margin setting. Another aspect of the proposed option which is not covered under the scope of this study is the effect of scale of uncropped areas on some species. Even if only half the width of the margin was cut, and the other half left long, the uncut sections might no longer be viable for some species if a particular threshold of required size of uncut area has not been reached once some of it is cut. There may also be increased edge effect, as there would be an increased edge to area ratio. This could make the remaining uncut margin sub optimal for some-species – of ground nesting birds e.g. – so the effects of scale on species which use the long areas of the margins would need investigating. I have seen some indication that this could be the case in the results of Chapter 5, showing that warblers and Reed Buntings appear to choose different 'sized' uncut areas. Therefore, robust testing of the proposed measures on a larger scale and over a longer timeframe is necessary, to see both if they could benefit Kestrels directly, and what the impacts would be on other species.

Another angle which I was keen to look at but was unable to within the scope of this project is the possibility of carrying out long term large-scale productivity studies on Kestrels breeding in nest boxes, and then using this along with historical data on Kestrel productivity (BTO nest record data for example) and land-use data to find out if there is any link between Kestrel productivity and the presence/quantity of AES farmland in the surrounding area.

As mentioned in the introduction to this thesis, while agricultural intensification generally, and breeding season productivity specifically (Vickery *et al*. 2004) are probably the principle driving forces behind Kestrel population declines, they are unlikely to be the sole reason and almost certainly several interconnected forces are at work. The winters of 2009 and 2010 experienced unusually low temperatures and heavy snowfall, and it seems probable that overwinter survival will have had a considerable effect, particularly for naïve young birds. I have also had many anecdotal suggestions from people that I have spoken to throughout the course of my project that persecution is a problem for Kestrels as well as for other raptors, although I have my doubts as to how great an effect this factor would be likely to be having on population numbers as a whole, and it seems unlikely that persecution levels would have increased sufficiently in recent times to explain the latest declines. Additionally, another interesting possibility given the nature of Kestrels as generalist predators, and fairly adaptable ones at that, is that the decline of many of the other birds on the FaBI could conceivably be having an effect on them. Kestrels will readily take small passerines, particularly when their preferred small mammal prey is less abundant (Village 1990), so therefore the concurrent declines in this valuable alternative food resource could also be having a negative impact on their numbers.

Conclusion

The provision of shorter vegetation alongside areas of longer vegetation in theory balances availability of prey with accessibility. This ‘kitchen-diner’ model of habitat provision is not a new idea (e.g. see Gathmann *et al*. 1994, Feber *et al*. 1996, Perkins *et al*. 2002, Vickery *et al*. 2004, Atkinson *et al*. 2005), but we have shown how this might apply to the Kestrel specifically. Although I have been unable to test directly how these proposed measures would affect Kestrels, the work in my thesis – as well as in work undertaken by others - is strongly suggestive that the provision of areas of cut grass next to areas of longer grass earlier in the season would be of benefit to them, as well as to other species of farmland birds which have similar foraging requirements. If carried out in a sympathetic way, there is also no reason why the implementation of these measures should adversely affect species with conflicting requirements, and certainly something needs doing to further increase structural and compositional heterogeneity in our farmed landscapes if we are to maximise the benefits of agri-environment schemes for a range of declining farmland species.

Establishment and management method (uncropped margins)	**Effects on plants**		**Effects on invertebrates**		**References**
Establishment	*Beneficial*	*Detrimental*	*Beneficial*	*Detrimental*	
Sown	Increase in abundance and species richness of perennial flowers (preferred by pollinators) from July onwards, although both decline steadily on sown plots over the long-term. Increases grass species richness. Beneficial for control of both perennial and annual weeds, although in the case of perennial weeds this beneficial effect is relatively short- term.	Lower abundance of annual flowers – annuals are excluded more rapidly from sown plots than naturally regenerated ones. Sowing with a simple grass mix is detrimental to plant species richness and diversity, and forb abundance compared to all other sowing regimes and natural regeneration.	Sowing with a grass & wildflower mix increases butterfly and bumblebee (and other nectar/pollen feeding flying inverts) abundance compared to sowing with a tussocky grass mix or natural regeneration. Sowing is generally beneficial for Meadow Brown butterfly *Maniola jurtina*, auchenorrhyncha and syrphid abundance, and spider abundance and species richness, compared to natural regeneration. Sowing with a tussocky grass mix benefits carabid & staphylinid beetles, spiders and small mammals, and provides nesting sites for bumblebees.		Feber *et al*. 1996 Baines *et al* 1998 Smith *et al*. 1994 Carvell *et al*. 2004 Meek *et al*. 2002 Asteraki *et al*. 2004 Smith *et al*. 2010
Unsown (naturally regenerated)	Garlic Mustard (important forbaceous larval host plant for some butterfly species) is more abundant in unsown plots. Natural regeneration also benefits rare arable weeds, which tend to be lost from sown plots.	Reduced plant species richness, and grass cover and diversity compared to sown swards. If left unmown, consistently more species poor than sown and/or mown plots.	Wolf Spiders prefer naturally regenerated swards (or sown with a tussocky grass mix) to swards sown with grass and wildflower mixes. Bumblebees prefer naturally regenerated margins to conservation headlands or crop, and trap nesting hymenoptera/bee species richness was found to be greater on naturally regenerated set-aside than on crops or sown fallow fields.	Harvestmen avoid naturally regenerated plots in autumn, in favour of all other sowing and cropping treatments.	Feber *et al*. 1996 Meek *et al*. 2002 Kells *et al*. 2001 Asteraki *et al*. 2004 Smith *et al*. 2010
Management					
Cutting generally	Increases plant species richness (in the short-term) on naturally regenerated and sown swards. It also affects plant species composition in the longer term, with	Decreases abundance of competitive, dominant plant species such as Thistles *Circium arvense*, Nettles *Urtica dioica* and Couch Grass *Elytrigia repens*, with	Increased plant species richness due to cutting of naturally regenerated early-successional set-aside was found to double the species richness of bees.	Reduces abundance and species richness of spiders. Effect of cutting in summer is worse than cutting in spring or autumn.	Baines *et al*. 1998 Smith *et al*. 2010 Gathmann *et al*. 1994

	frequency of cutting eventually being more important than timing. On sown swards, abundance of some perennials (inc. grasses such as Yellow Oat Grass *Trisetum flavescens*, Smooth Meadow Grass *Poa pratensis* and Crested Dog's-tail *Cynosurus cristatus* and forbs like Field Scabious, Ox-eye Daisy and Black Knapweed) increased. Field Scabious is lost completely from unmown plots.	abundance declining further with increased frequency of mowing. On naturally regenerated swards, mowing is ineffective for increasing plant diversity – only affects relative abundance.			
Cutting spring (April/May)	Perennial species richness and annual abundance increased on sown plots cut in spring and autumn. In the case of annuals, this cutting regime delayed their decline and increased opportunities for seed return.		No significant difference to butterfly abundance and species richness than not cutting at all.	Reduced species richness of phytophagous invertebrates on improved lowland mesotrophic grassland, although polyphagous inverts are unaffected. Slight, short-lived detrimental effect on heteroptera abundance and diversity, and auchenorrhycha abundance and species richness, on calcareous grassland.	Feber *et al*. 1996 Woodcock *et al* 2009 Morris & Lakhani 1979 Smith *et al*. 2010
Cutting summer (June/July)	Greater flower abundance in September than on uncut plots (following an initial drop), especially of annual flowers. Beneficial for control of annual weeds.	Lower abundance of perennial flowers in July (needed by some butterfly species) than on uncut plots. Nectar sources such as Common & greater knapweed, field scabious, thistle and Ox-eye daisy adversely affected. These are important in July and August for many butterfly species. Smaller Cat's-tail, Wild Oats (Avena spp) and Upright Hedge-parsley also adversely affected by summer cutting.		Decreases butterfly abundance and species richness. Reduced species richness of phytophagous invertebrates on improved lowland mesotrophic grassland, although polyphagous inverts unaffected. Strong and lasting detrimental effect on hemiptera (heteroptera & auchenorrhyncha) on calcareous grassland.	Feber *et al*. 1996 Woodcock *et al* 2009 Baines *et al*. 1998 Morris & Lakhani 1979 Smith *et al*. 2010
Cutting autumn (end of September)	Perennial species richness and annual abundance increased on sown plots cut in	Detrimental for Thistles – therefore good for controlling them.	No significant difference to butterfly abundance and species richness than not		Feber *et al*. 1996 Carvell *et al*. 2004 Smith *et al*. 2010

	spring and autumn. In the case of annuals, this cutting regime delayed their decline and increased opportunities for seed return.		cutting at all.		
Cutting more than once	Cutting twice annually found to increase plant species richness on both naturally regenerated and sown swards (which have been shown to retain more sown species than those cut once or not at all). Cocksfoot *Dactylis glomerata*, Yorkshire Fog *Holcus lanatus* and Bindweed *Convolvulus arvensis* more abundant in plots mown twice. Black Knapweed abundance declines with decreasing numbers of cuts. In the longer term, frequency of cuts more important determinant of plant species composition than timing of cuts.	Abundance of Thistles *Circium arvense*, Nettles *Urtica dioica* and Couch Grass *Elytrigia repens* found to decline further with cutting twice rather than once.		Cutting more than once a year (May & July) adversely affected monophagous and oligophagous invertebrate species richness on improved lowland mesotrphic grassland.	Woodcock *et al*. 2009 Smith *et al*. 2010
Leaving cut grass	Increases abundance of Nettles *Urtica dioica* (important forbaceous larval host plant for some butterfly species). It even counteracts the detrimental effects of increased mowing for this species. However, excessive abundance of nettles is undesirable for plant species richness.	Detrimental to less competitive plants, as increases soil fertility favouring competitive species, including pest species Black Grass *Alopecurus myosuroides* which was more abundant when hay was left lying than under all other treatments. On naturally regenerated swards, cutting twice and leaving hay lying did not decrease plant species richness significantly.	Increases species richness of spiders. Also benefits litter dwelling invertebrates.		Feber *et al*. 1996 Baines *et al* 1998 Smith *et al*. 2008 Smith *et al*. 2010

Table 6.1. Some of the interactions between field margin establishment and management methods for plants and invertebrates. For an overview of the same processes for birds, see this thesis and Vickery *et al*. 2009.

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