Investigating Conservation Management Interventions for the Benefit of Farmland Breeding Waders of Marginal Agricultural Grasslands in the UK

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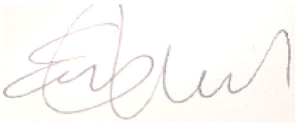
DECLARATION OF AUTHORSHIP

I hereby declare that this thesis has been composed by myself and embodies the results of my own research. Where appropriate, I have acknowledged the nature and extent of work carried out in collaboration with others. All research material has been acknowledged and cited.

Signed: 

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PHD ABSTRACT

Agriculture is the principal land use throughout Europe and agricultural intensification has been implicated in large reductions in biodiversity, with the negative effects on birds particularly well documented. The Northern Lapwing (*Vanellus vanellus*) and the Eurasian Curlew (*Numenius arquata*) are farmland breeding wader species where changes in farming practices has reduced the suitability and quality of breeding habitat, as well as the availability of large, soft-bodied soil prey which these birds depend upon leading to severe declines in population size that warrants them both listed on the Red List of Birds of Conservation Concern in the UK. Lowland areas, where agricultural intensification has generally been most pronounced, have been worst affected. However, more recently declines in marginal upland areas, previously considered refuges for breeding wader populations, have been identified. In this thesis I investigate conservation management interventions for the benefit of farmland breeding waders of marginal agricultural grasslands in the UK. Upland areas receive high levels of rainfall and are characterised by typically peaty soils with a low buffer capacity and a high organic matter content that results in these areas being particularly prone to localised acidification over time. I use field-scale trials to experimentally test the use of lime as a potential novel conservation tool to amend acidic soil pH, thus increasing earthworms (Annelida), an important prey item of Lapwing and Curlew. I show that lime use increases soil pH, although the effect varies with time, and that earthworm numbers increased following lime with the largest effect size observed two years after liming. Above ground, I show that there was no effect of liming on the vegetation of grazed grasslands, and no effect on beetles (Coleoptera) or spiders (Aranea). Additionally, I show that wader densities were 50% higher on lime treatment plots, and that wader intake rate of large prey items was greater on lime treatment than control three years following the application of lime. Furthermore, I show strong woodland edge effects on four wader species over a distance of several hundreds of metres in the Badenoch and Strathspey, an important stronghold for these birds. This is particularly relevant in light of land-use policy of tree planting to mitigate climate change committed to by the UK Government. The results presented here in this thesis will help contribute to the development of conservation strategies for waders. My results suggest that lime use could be an important tool to mitigate localised acidification of enclosed upland grassland where earthworm numbers may have declined, alongside other well established habitat managements for these birds, and that buffer zones should be placed of at least 200m between tree planting and important wader breeding habitat.

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CHAPTER 1 | GENERAL INTRODUCTION

1.1 GLOBAL BIODIVERSITY DECLINES

Biodiversity loss over the last fifty years is occurring at an alarming rate and is being reported across many different taxa globally (Barnosky *et al.*, 2011; Ceballos *et al.*, 2015). For example, of the 128,500 species assessed for the International Union for Conservation of Nature (IUCN) Red List of Threatened Species, approximately 40% of amphibians, 34% of conifers, 33% of reef corals, and 14% of birds are threatened with extinction (IUCN, 2021). Indeed the rate of species extinction is estimated to be 1,000 times higher than that of the background (pre-human) rate (Ceballos *et al.*, 2015; De Vos *et al.*, 2015). In addition, contractions in species ranges and declines in population abundance have occurred globally with an estimated population decline of 60% reported between 1970 and 2014 by the Living Planet Index (WWF, 2018). This loss in biodiversity has been linked to habitat loss and fragmentation (e.g. due to agricultural expansion), overexploitation of natural resources (e.g. unsustainable logging and fishing), terrestrial and marine pollution (e.g. excess fertiliser use and marine litter), non-native invasive species, and more recently through global impacts of climate change (OECD, 2012; Ehrlich and Ehrlich, 2013; SCBD, 2014b). Biodiversity is responsible for providing many ecosystem services which society depends upon such as provisioning food (e.g. game and fish meat), raw materials (e.g. wood) or freshwater; regulating climate or water quality; supporting soil formation and nutrient cycling; and cultural benefits such as recreation or spiritual and mental well-being. Halting biodiversity loss, and restoring degraded ecosystems, is therefore an ambitious but vital target which requires both local and global attention.

Sustainable development and halting biodiversity loss are central aims of the Convention on Biological Diversity (CBD) and the United Nations Sustainable Development Goals, in which member states agree to achieve targets to address the underlying causes of biodiversity loss (SCBD, 2010, 2011; United Nations, 2015). However, outcomes of these to date have been poor: neither the 2010 CBD target nor most of the 2020 Aichi biodiversity targets were reached, and there was no clear progress on "slowing down and eventually reversing" the loss of biological diversity around the world (Buchanan *et al.*, 2020). This is worrying when we consider the continued human population growth and therefore the increased resources required such as food production, consumption of energy (e.g. fuel), clean water and air, and living space; these growing demands raise serious concerns for biodiversity (SCBD, 2010; Ehrlich and Ehrlich, 2013). Over 70% of the Earth's ice-free land area is already affected by human use, with agricultural land accounting for a large proportion of this (FAOSTAT, 2003). Agricultural expansion and intensification, currently the dominant pressures on terrestrial biodiversity, are expected to increase to reach the demands of feeding the growing population (SCBD, 2014a). The sustainable management of agriculture is therefore one of the key 2020 Aichi target (strategic goal A: Target 7). In developing countries, agriculture is often linked to expansion and direct habitat loss such as the removal of natural forests which are an important and rich habitat for biodiversity (Gibson *et al.*, 2011; Hansen *et al.*, 2013). Conversely in Europe, agriculture already dominates the landscape and intensification of the farmed land, rather than expansion, is often linked to biodiversity loss. Management of land already in farmland is often the target for conservation strategies in Europe (Boitani and Sutherland, 2015).

1.2 AGRICULTURE IN EUROPE AND BIODIVERSITY DECLINES

Within Europe, over half of the European landscape is under agricultural management, and approximately 70% of land in the UK (Batáry *et al.*, 2015; DEFRA 2012). Therefore, much of the European countryside is agricultural, with very little “natural” habitat remaining. Due to its dominance, and because extensive agriculture has occurred for millennia within Europe, agriculture is also an important habitat for many species with over half of all European biodiversity dependent on land which is used as farmland for at least some part of their life cycle (Kleijn and Sutherland, 2003; European Environmental Agency (EEA), 2018). As a result, much conservation in Europe has focussed on farmland biodiversity and the management of traditional agricultural practices (Sutherland, 2004). The abandonment of agricultural land is generally seen as a threat to biodiversity (Queiroz *et al.*, 2014) as the loss of large wild grazers from Europe leads to the natural succession of the habitat into scrub and forest which is detrimental to species of open landscapes common on farmland (Thomas, 1991).

The implementation of the Common Agricultural Policy (CAP) in 1957, is considered to be a key driver of the intensification of farmland within Europe post World War II (Robson, 1997; Krebs *et al.*, 1999) which has been implicated in widespread biodiversity declines across a wide range of taxa including birds (Tucker and Heart, 1994; Chamberlain *et al.*, 2000; Donald, Green and Heath, 2001; Vickery *et al.*, 2001; Donald *et al.*, 2006; EBCC, 2017) butterflies and moths (Heath, Pollard and Thomas, 1984; Thomas, 1995; Warren *et al.*, 1997; Maes and Van Dyck, 2001; Conrad *et al.*, 2004, 2006; New, 2004; Reynolds and Perrins, 2010; van Strien *et al.*, 2019; Hallmann *et al.*, 2017), bees (Goulson, Lye and Darvill, 2008; Potts *et al.*, 2010; Woodcock *et al.*, 2016), arachnids (Aebischer, 1991; Nyffeler and Bonte, 2020), and annual arable wildflowers (Schumacher, 1987; Andreasen, Stryhn and Streibig, 1996). Although agricultural intensification has taken many forms in different countries and encompasses a wide range of components, there has been a general trend for widespread drainage of grassland within Europe, a switch to more productive grass species, higher livestock densities, and increased frequencies of cutting of meadows (e.g. Fuller, 1987). Within arable systems there has been an increase in chemical input from nitrogen fertilisers and pesticides, a loss of non-farmed habitat such as field margins, ponds and hedges, changes to the timing of farming management including sowing and harvesting, and a decrease in the variety of crop types planted (e.g. Deckers *et al.*, 2005). Overall, these changes have contributed to large increases in production, but have resulted in reduced habitat amount and quality for many species (Vickery *et al.*, 2001; Newton, 2004, 2020). Contrasting land use changes have also occurred, predominantly due to less intensive traditional practices becoming no longer profitable, resulting in the abandonment of farmland and reforestation particularly in Mediterranean regions (e.g. Debussche, Lepart and Dervieux, 1999). Consequentially, intensification of the farmed landscape has seen an overall loss in small farms and mixed farming systems, a geographically polarised landscape and an increasingly simplified and uniform habitat with widespread wildlife losses (e.g. Stoate, 1996; Maton *et al.*, 1997; Chamberlain *et al.*, 2000; Tilman *et al.*, 2001; Robinson and Sutherland, 2002; Wilson, Evans and Grice, 2009).

1.3 FARMLAND BREEDING WADERS

1.3.1 Population declines and conservation status

Farmland breeding waders are a group of wading birds (order: Charadriiformes) associated with farmed land within Europe and in the context of this thesis includes the following five species common on farmland in the United Kingdom: the Northern Lapwing (Vanellus vanellus); Eurasian Curlew (Numenius arquata); Eurasian Oystercatcher (Haematopus ostralegus); Common Snipe (Gallinago gallinago); and Common Redshank (Tringa tetanus) hereafter referred to as Lapwing, Curlew, Oystercatcher, Snipe, and Redshank respectively. All of these birds have been listed as either red or amber species on the UK list of birds of conservation concern (Eaton et al., 2015; Harris et al., 2020) due to long-term and severe population declines and range contractions within the UK (Balmer et al., 2013). The magnitude of decline has been particularly large for both Lapwing and Curlew which have declined by more than 40% in the UK between 1995 and 2019 according to the BTO/JNCC/RSPB Breeding Bird Survey (BBS) report from survey year 2019 (Harris et al., 2020). Due to COVID-19, the number of BBS squares surveyed in the most recent report for 2020 were lower than usual meaning that estimates should be treated with caution and were only available for England (rather than UK wide) and for a few species (Harris et al., 2021). The rate of population decline of Curlew is the highest recorded across the species global range, which is concerning as the UK harbours an estimated 19-27% of the Curlew's global breeding population, thus the Curlew is a species of high conservation priority in the UK (Brown et al. 2015). Aside from Russia and the Netherlands, Lapwing declines in the UK are one of the most severe in the species

Global range (Birdlife International, 2021). Other waders of note which are found within the UK and use farmland to some extent, but are not targets of this thesis, include: European Golden plover (Pluvialis apricaria), Stone curlew (also known as the Eurasian Thick-knee; Burhinus oedicnemus); Ringed plover (Charadrius hiaticula); Dunlin (Calidris alpina); and Woodcock (Scolopax rusticola). Furthermore, the Black-tailed Godwit (Limosa limosa) is a schedule 1 bird species which has seen steep breeding population declines over the past 50 years within Europe and is labelled as Near Threatened by the IUCN (2016), and is therefore an important farmland breeding wader species of conservation concern (Kentie et al., 2016). However, very few pairs breed in the UK, although the UK is an important wintering ground for this species in wetland areas (Prater, 1975; Rehfisch et al., 2003).

Farmland breeding waders have a broad geographical distribution within the Palearctic, with a large proportion of the global population breeding within Europe (Table 1.1). Significant declines in waders have occurred globally over recent decades in many parts of the world (Stroud et al., 2006; Butchart et al., 2010) as well as in several key breeding populations on agricultural land within Europe (Piersma, 1986; Tucker and Heart, 1994; Norris *et al.*, 1998; Thorup, 2004; Wilson, Ausden and Milsom, 2004; Stroud *et al.*, 2006; Teunissen and Soldaat, 2006; Brown *et al.*, 2015; Birdlife International, 2017b; Harris *et al.*, 2020). As of such, many wader species are of conservation concern with IUCN extinction categories of Near Threatened or Vulnerable at the Global and European scale (Table 1.1). However, Redshank and Snipe are both species of extremely large ranges globally, and are listed as IUCN species of Least Concern despite evidence to suggest population trends are decreasing they are not sufficiently rapid for the threshold of Vulnerable (>30% decline over ten years or three generations). The primary driver of wader declines is low breeding productivity, mainly due to high nest and chick predation rates that arise through numerous factors including land use change and degradation of habitat (Roodbergen, Werf and Hötker, 2012; Douglas *et al.*, 2014; Samantha E. Franks *et al.*, 2018; Roos *et al.*, 2018).

In the UK, declines have occurred widely across the range of habitats in which waders occur (Sim et al., 2005; Wilson et al., 2005; Harris et al., 2020), although lowland enclosed farmland in England and Wales where agricultural intensification has been most pronounced has experienced the most severe and long-term declines (Baines, 1990; O'Brien, Tharme and Jackson, 2002; Wilson et al., 2005; Shrubb, 2007). In light of these declines in the lowlands, UK upland areas are considered to be critically important for supporting breeding populations of declining wader species (Balmer et al., 2013; Siriwardena et al., 2016). Marginal agricultural land in these upland areas of the UK are typically less intensively farmed. In Scotland, these birds are still common and widespread species of the upland countryside, and were largely considered to have escaped the losses seen in lowland areas (Baines, 1990; O'Brien, Tharme and Jackson, 2002; O'Brien and White, 2003; Sheldon et al., 2004; Wilson et al., 2005; Sim et al., 2005; Forrester et al., 2007; Shrubb, 2007; Smart et al., 2008; Balmer et al., 2013; Bell and Calladine, 2017). However, in recent decades substantial declines in wader populations have also occurred within upland areas (Taylor and Grant, 2004; Risely et al., 2011; Bell and Calladine, 2017; Harris et al., 2020), although moorland and rough grassland where predator control is carried out as part of grouse moor management still hold high densities of breeding waders (Fletcher et al., 2010; Douglas et al., 2014; Franks et al., 2017).

The magnitude of declines in England breeding wader populations between 1995 and 2019 are estimated as 30% for Lapwing, 31% for Curlew, and 43% for Redshank, while there has been an estimated increase in breeding populations of Oystercatcher (49%) and Snipe (27%) (Harris et al., 2020). For Scotland, the magnitude of declines in breeding wader populations between 1995 and 2019 differ and are estimated as 39% for Oystercatcher, 56% for Lapwing, 59% for Curlew, and an increase in Snipe of 22%, while numbers of Redshank recorded in BBS squares are too low for estimates to be calculated (Harris et al., 2020). Therefore, despite Scotland previously being considered a refuge for wader species, the overall percentage declines of Lapwing and Curlew since 1995 have been greater in Scotland than England.

Table 1.1. Summary table showing the five farmland breeding wader species global breeding population size (calculated as the number of global individuals/3 where estimates on population size at the individual level were available following methods of BirdLife International (2004)), estimated number of breeding pairs within Europe (including the European population of % of the global breeding pairs using the midpoint values where a range in estimates is provided); estimated number of breeding pairs in the UK (Massimino *et al.*, 2019; including the UK population of % of European breeding pairs calculated using the midpoint where a range in estimate is provided); the IUCN red list status at both the Global and European regional assessment level (www.iucnredlist.org accessed February 14th 2021); and status in the UK's Red List of Birds of Conservation Concern shown by colour of the species where red = red listed; and orange = amber listed (Eaton *et al.*, 2015).

Species	Global breeding pairs	European breeding pairs (% of Global population)	UK breeding pairs (% of European population)	Global & European regional assessment status
Curlew	284,333 – 433,333 [a]	212,000 – 292,000 [e] (c.67%)	68,000 (c.27%)	Near Threatened; Vulnerable [g]
Lapwing	1,866,666 – 3,500,000[b]	1,590,000 – 2,580,000[e] (c.78%)	140,000 (c.7%)	Near Threatened; Vulnerable [g]
Oystercatcher	308,333 – 343,333 [c]	292,000 – 425,000 [f j]	110,000 (c.31%)	Near Threatened; Vulnerable [g]
Redshank	433,333 – 1,033,333 [d]	340,000 – 484,000 [e] (c.56%)	25,000 (c.6%)	Least Concern; Least Concern [h]
Snipe	5,000,000 - 9,666,666 [i]	2,670,000 – 5,060,000[e] (c.6%)	80,000 (c.2%)	Least Concern; Least Concern [i]

Sources:

[a] (Wetlands International, 2016)

[b] (Wetlands International, 2012)

[c] (Wetlands International, 2019)

[d] (Wetlands International, 2015)

[e] (Birdlife International, 2015)

[f] (Burfield and Bommel, 2004)

[g] (Birdlife International, 2017b, 2017a)

[h] (BirdLife International, 2016)

[i] (Birdlife International, 2019)

[j] please note that due to taxonomic splits in Oystercatcher into three subspecies and then back into two subspecies the breeding population estimates do not match up and therefore a proportion was not calculated (see del Hoyo *et al.*, 2014; Ehmke, 2015; Hayman, Marchant and Prater, 1986; Marchant and Higgins, 1993).

1.3.2 Breeding ecology and habitat requirements

The five species of waders in this thesis occur on several habitats within the UK, although they are most commonly associated with farmland during the breeding season, particularly on agricultural grasslands (Galbraith, 1988b; Berg, 1993; O'Brien and Murray, 1998; Sheldon *et al.*, 2004). For example, 96% of breeding lapwing populations within England and Wales, and 85% of the Scottish population are recorded on agricultural land (Shrubb and Lack, 1991). The nesting habitat of Curlew is more often associated with moorlands; however, enclosed pasture fields are an important foraging habitat for both adults and chicks.

Waders are ground nesting birds which typically raise a single brood of chicks each breeding season. Waders lay around four eggs, although two to three is more common with Oystercatcher, often within a simple scrape and most waders will lay a replacement clutch if the nest is lost during the incubation period; however, if chicks are lost a further attempt at breeding is not usually made (Klomp, 1954; Mullarney *et al.*, 1999). The incubation period varies between 25 – 34 days and usually begins once all eggs have been lain to ensure hatching at a similar time. Waders select nest sites with open views often in relatively flat, large fields. Waders tend to avoid areas with perches (e.g. trees) and field boundaries that restrict their ability to scan for predators (Elliot, 1985; Milsom *et al.*, 2000; Wallander, Isaksson and Lenberg, 2006; Shrubb, 2007). Early detection of predators is an important strategy for ground nesting birds which either mob or distract predators away from their nests (Elliot, 1985). Mobbing of predators as a nest defence in lapwing is more successful in larger colonies (Berg, Lindberg and Gunnar, 1992). In addition, birds often select habitats in which they are less conspicuous to avoid predation, with nest sites offering some degree of camouflage to the eggs, incubating birds and chicks. Such habitat includes tussocky vegetation to provide cover and shelter, and areas with a mixture of vegetation, bare ground and damp patches which provide camouflage as well as also providing important invertebrate foraging areas (Baines, 1988, 1990; Galbraith, 1989; Vickery *et al.*, 2001; Shrubb, 2007).

Wader chicks are precocial and can feed themselves within a few hours of hatching. They are led to suitable foraging areas by their parents, sometimes over large distances if suitable habitat is not present in the nesting site (Cramp and Simmons, 1983; Galbraith, 1988b; Sheldon *et al.*, 2004). Chicks are brooded at night and periodically during the day (Shrubb, 2007). Fledging varies but is around 35 - 40 days after hatching, although newly fledged birds are often still dependent on their parents for a few days after they can fly. Waders are relatively long lived birds, with a typical lifespan of about five years for Lapwing, eleven for curlew, twelve for oystercatcher, four for redshank and three years for snipe (Robinson, 2005). Waders typically breed at two years old, with the exception of oystercatcher at four and snipe at two.

The diet of waders is variable. Both the adults and chicks feed on a broad range of arthropods including soil invertebrates such as earthworms (Lumbricidae) and dipteran larva (particularly Tipulidae), as well as above ground invertebrates including beetles (Coleoptera), flies (Diptera), moths (Lepidoptera), ants (Hymenoptera), spiders (Aranea), and woodlice (Isopoda) (Cramp and Simmons, 1983; Berg, 1993). Waders are strongly associated with wet features and rely on these areas and moist soil to provide their invertebrate prey (Berg, 1993; Mckeever, 2003; Smart *et al.*, 2006; Eglington, 2008; Eglington *et al.*, 2010; Rhymer *et al.*, 2010; Schmidt *et al.*, 2017). A higher water table helps to move soil invertebrates closer to the surface and patches of bare wet muddy features created by standing water can provide a mosaic of both nesting habitat and important foraging areas (Ausden and Hirons, 2002; Milsom *et al.*, 2002; Ausden *et al.*, 2003; Eglington *et al.*, 2010). Earthworms are an important component of the diet of Lapwing and Curlew, likely due to their high calorific and protein content (Högstedt, 1974; Barnard and Thompson, 1985; Galbraith, 1989; Baines, 1990; Beintema *et al.*, 1991; Berg, 1993; Paoletti *et al.*, 2002; Sheldon, 2002; Mckeever, 2003; Shrubb, 2007; Watkins, 2007; Navedo *et al.*, 2020). This is particularly the case early in the breeding season where lapwings are found feeding in fields with high earthworm densities, which is thought to be linked to birds getting into condition for breeding, an energetically demanding period (Högstedt, 1974; Galbraith, 1989; Baines, 1990; Berg, 1993; Mckeever, 2003; Watkins, 2007). For female lapwings, body condition influences eggs size and thus chick weight, which has implications for chick survival (Galbraith, 1988c; Blomqvist and Johansson, 1995). The length of the pre-laying period is inversely correlated with the abundance of earthworms (Högstedt, 1974), suggesting earthworms play an important role in lapwing breeding success; eggs are laid earlier in areas that are particularly earthworm rich, providing them with ample time to lay a replacement clutch if required. Earthworms are also important for lapwing chick development (Beintema *et al.*, 1991; Sheldon, 2002), with positive relationships between the number of earthworm chaetae found in chick faeces and chick age, growth rates and body condition (Sheldon, 2002). Older chicks may find it easier to consume earthworm prey than younger chicks, which may be incapable of catching earthworms with their small bills. Indeed, chicks older than 12 days have been observed foraging in relatively earthworm rich areas within fields, whereas the opposite was true for younger chicks (Watkins, 2007). It has also been suggested that this difference in chick diet is because older chicks need to consume more earthworms to meet their increasing energy demands as they approach fledging (Beintema *et al.*, 1991).

Earthworms also feature in the diets of snipe, redshank, and oystercatcher (Beintema *et al.*, 1991). For Snipe, the dry weight of prey ingested between April and June was largely earthworms (61%), with the remainder of prey being made up of tipulid larvae (24%) and a wide variety of surface-active and aquatic prey (Hoodless, Ewald and Baines, 2007). The earthworm diet of Curlew is predominantly made up of smaller earthworms on the soil surface epigeic earthworms), despite the length of the Curlew bill which makes it possible for birds to access food items along a broader depth range within the soil profile (Navedo *et al.*, 2020). This may be the result of a trade-off between energy intake rate and foraging effort. Similarly, reconstruction of Lapwing diet from faecal samples have shown that in terms of biomass, these small epigeic earthworms make up the higher proportion of prey items taken during the day; however, at night the proportion of earthworms that make up the diet are largely deep burrowing earthworms (anecic earthworms) which come up to the surface to feed at night (Gillings and Sutherland, 2007).

For both coastal and inland grasslands within Europe, livestock grazing is a key driver of habitat quality for waders although the sward preference for each species varies. Lapwing prefer a short sward whereas species like Curlew and Redshank are more often associated with longer swards with a preference for a higher proportion of tussocks/rush cover within fields than Lapwing (Baines, 1988; Treweek *et al.*, 1997; Valkama, Robertson and Currie, 1998; Tichit, Durant and Kernéïs, 2005; Smart *et al.*, 2006; Durant *et al.*, 2007; Shrubb, 2007)**.** Grazing by cattle typically results in a more varied vegetation structure than sheep due to the mechanism of grazing (e.g. cows tearing off vegetation with the tongue compared to sheep biting/nibbling with the teeth). Additionally, there are behavioural differences in grazing between livestock, with cattle foraging around dung patches and having a "less fussy" palate compared with sheep that results in a more diverse grassland habitat including patches of tall and short vegetation attractive to waders (Rook *et al.*, 2004; Tichit, Durant and Kernéïs, 2005; Smart *et al.*, 2006). Livestock grazing also influences the invertebrates within a field, as different species feed on the dung of livestock, and the hooves of cattle can produce a more varied ground micro-topography (i.e. ground bumpiness). Sheep grazing or high livestock densities of cattle can lead to compaction of soil. As waders are opportunistic foragers which hunt visually for invertebrate prey the sward structure and soil penetrability of fields are important for the accessibility of prey items (Green, Hirons and Cresswell, 1990; Butler and Gillings, 2004; Devereux *et al.*, 2004, 2006; Smart *et al.*, 2006).

1.3.3 Negative effects of agricultural intensification

Agricultural intensification can impact bird populations in two ways: directly, through the loss of habitat or by increased mortality; and indirectly, by altering food resources or the quality of nesting sites (Fuller and Gough, 1999). For farmland breeding waders, the recent intensification of agricultural land use has, in general, made conditions less suitable for breeding waders and reduced invertebrate food availability

(Shrubb, 2007; Wilson, Vickery and Pendlebury, 2007), as well as increasing mortality from changes in mechanical operations and higher livestock numbers, and through higher densities of predators which can limit breeding success and site occupancy (Grant *et al.*, 1999; Evans, 2004; Macdonald and Bolton, 2008; Teunissen *et al.*, 2008; Roodbergen, Werf and Hötker, 2012; Douglas *et al.*, 2014; Manton *et al.*, 2016; Roos *et al.*, 2018).

For example, land drainage of agricultural grassland areas has led to taller more uniform swards, and drier ground reducing the suitability of pasture for nesting (Klomp, 1954; Milsom *et al.*, 2000; Vickery *et al.*, 2001). Widespread land drainage is likely to have also significantly reduced the abundance and availability of invertebrate prey (Baines, 1990; Taylor and Grant, 2004; Wilson, Evans and Grice, 2009), as well as reduced the detectability of prey due to the taller, more uniform swards (Devereux *et al.*, 2004). The improvement of grassland fields through reseeding with more productive leys and from increased fertiliser use has also resulted in faster growing swards which facilitated increased stocking densities resulting in high rates of nest destruction by trampling (Beintema and Muskens, 1987; Pakanen, Luukkonen and Koivula, 2011). There has also been a move to sheep grazing in upland sites, resulting in a loss of a diverse sward produced by cattle grazing. In addition, the compaction of soil can also occur from sheep grazing or high livestock densities of cattle, with some evidence that compacted grassland soils may be less biodiverse than uncompacted soils (Roovers, Baeten and Hermy, 2004) and that compaction can lead to a reduction in some soil-pupating larvae of *Lepidoptera* (butterflies and moths; (Roach and Campbell, 1983). This may affect waders due to the effect of compaction on soil-dwelling prey, or reduce accessibility of soil probing (Smart *et al.*, 2006; Clarke *et al.*, 2007).

For lapwing which nest on both pasture and arable fields, the change from spring to autumn sowing in arable systems has reduced the availability of preferred nesting habitat, as autumn sown crops become too tall and dense by spring to be used for breeding (Sheldon *et al.*, 2004). The intensity of agricultural operations on arable land in spring has also increased, leading to higher rates of nest destruction from agricultural machinery (Wilson, Evans and Grice, 2009). In addition, faster growing varieties of crops and high fertiliser application rates mean that spring sown crops quickly become too tall to be used, shortening the period that a field is suitable for nesting lapwing. A decline in mixed farming and a move to polarisation of agriculture in the UK to arable mainly in the south and east, and pasture in the north and west, have further affected waders, particularly lapwing as the suitability of arable fields for nesting depends on the proximity of chick rearing habitat such as damp grassland. Loss of mixed farming has therefore been detrimental as chicks that have to travel larger distances between the nest site and foraging habitat suffer from increased mortality (Galbraith, 1988a; Blomqvist and Johansson, 1995; Shrubb, 2007; Stoate *et al.*, 2009).

The recent intensification of agricultural land has also been linked to increased predation risk of waders (e.g. Roos *et al.*, 2018). Many generalist predators of farmland birds have increased such as the Red fox, *Vulpes vulpes* and corvids (Marchant *et al.*, 1990; Baillie *et al.*, 2002; Macdonald and Bolton, 2008; Fletcher *et al.*, 2010). Predation is a key factor in determining wader breeding success (Grant *et al.*, 1999; Teunissen *et al.*, 2008; Eglington *et al.*, 2009). Habitat changes can lead to shortened breeding seasons and less opportunity to replace predated nests, therefore increasing the sensitivity of breeding success to nest predation rates. For example widespread drainage of lowland wet grasslands in the UK has reduced the number of nesting opportunities for Snipe and likely contributed to observed population declines (Green, 1988).

Food availability may also be linked to predation rates of waders as hungry chicks may increase their mobility and thus their visibility to predators, leading to a reduction in survival rates (Swennen, 1989). Similarly, food shortages may also lead to increased predation rates by altering adult behavioural patterns as parents spend less time being vigilant for detecting predators and more time foraging (Lima and Dill, 1990; Dall and Witter, 1998; Lilliendahl, 2000; Evans, 2004). The combination of increased predation and reduced food availability may therefore have increased starvation-induced mortality as a consequence of the costs of predator avoidance behaviour. Experimental studies of lethal predator removal have indicated beneficial effects of wader nest survival, further supporting the hypothesis that predation may be limiting waders (Bolton *et al.*, 2007; Fletcher *et al.*, 2010; Roos *et al.*, 2018). The landscape of agricultural land can also influence predation. For example, the structure of linear features such as fences and hedges, which make agricultural processes easier to undertake, may also act as landscape features which allow mammalian predators such as foxes to move into fields and more easily find nests (but see Eglington *et al.*, 2009), and predators can harbour in patches of fragmented woodland in the landscape.

1.3.4 Agri-environment schemes (AES)

Environmental measures, such as Agri-Environment Schemes (AES), were introduced into the CAP to help reverse the impacts of agricultural intensification on farmland biodiversity by providing financial incentives to deliver environmental goods from agricultural management (Vickery *et al.*, 2004). Uptake of schemes has been high, with approximately 20% of EU farmland within some form of AES (Kleijn and Sutherland, 2003; Donald and Evans, 2006). However, the effect of these schemes on biodiversity conservation has been mixed (e.g. Kleijn *et al.*, 2001; Peach *et al.*, 2001; Breeuwer *et al.*, 2009; Santana *et al.*, 2014; Batáry *et al.*, 2015; Sanderson *et al.*, 2016; Heldbjerg, Sunde and Fox, 2018; Leyrer *et al.*,

2018) and is poorly monitored (Kleijn and Sutherland, 2003). Nevertheless, when appropriately designed and targeted (Evans, Armstrong-Brown and Grice, 2002), AES have the potential to deliver large-scale benefits to wildlife populations on agricultural land (e.g. Aebischer *et al.*, 2000; Peach *et al.*, 2001; Bradbury *et al.*, 2004; Fuller, Hinsley and Swetnam, 2004; Kleijn *et al.*, 2004; Vickery *et al.*, 2004; Perkins *et al.*, 2011) and some of the best AES success stories come from farmland birds (Aebischer *et al.*, 2000; Peach *et al.*, 2001; O'Brien, Green and Wilson, 2006; Bretagnolle *et al.*, 2011; Perkins *et al.*, 2011). The cost of AES is expensive, with approximately a budget of €363 billion for the period 2014 - 2020 (Navarro and López-Bao, 2018). AES therefore represent a major source of nature conservation funding within the European Union and ensuring the cost-effectiveness of schemes to deliver their biodiversity targets remains a key priority within Europe (Kleijn and Sutherland, 2003; Sutherland *et al.*, 2006).

Targeted conservation measures in the form of AES for the benefit of farmland breeding waders within Europe has been built on the extensive research into their ecology and responses to agricultural intensification (Sheldon, 2002), with particular conservation effort within the Netherlands and the UK. AES prescriptions predominantly focuses on two broad prescriptions for grassland management; 1) the manipulation and timings of grazing and cutting to reduce chick mortality and nest loss from livestock trampling or mechanical operation (Teunissen and Soldaat, 2006) and to create a desired sward structure (Beintema and Muskens, 1987; Vickery *et al.*, 2001), and 2) the manipulation of the water table and creation of non-linear wet features (i.e. scrapes) to improve foraging habitat by increasing invertebrate prey (Eglington *et al.*, 2010) and to attract breeding waders to fields (Smart *et al.*, 2006; Eglington, 2008; Fisher *et al.*, 2011). In both cases, farmers typically enter into management agreements on a field-by-field basis. In Scotland support for predator control is also available but is restricted to Sites of Special Scientific Interest (SSSIs) and Natura sites (Scottish Government, 2014) and only when sufficient evidence is provided of an adverse effect of predators on ground nesting birds (Fletcher *et al.*, 2010; Malpas *et al.*, 2013). Despite this high conservation effort, the evidence for wader AES suggests that current management has not been successful at reversing or halting the trend of population decline at the national or European level, although it has been able to slow the rate of decline at the local level for farmland breeding waders in some cases through increased nest success and abundance (Ausden and Hirons, 2002; Kleijn and van Zuijlen, 2004; Ottvall and Smith, 2006; Sheldon, Chaney and Tyler, 2007; O'Brien and Wilson, 2011; Rickenbach *et al.*, 2011; Samantha E Franks *et al.*, 2018).

There is a large body of research on the negative effects of intensified management of UK lowland wet grassland and of meadow fields in the Netherlands on waders, as well as research quantifying the impact of conservation measures on breeding waders in these two habitats (Kleijn and van Zuijlen, 2004; Smart *et al.*, 2006; Eglington *et al.*, 2007, 2009, 2010; Kleijn *et al.*, 2010). However, specific AES schemes for upland grasslands do not exist despite the fact that wader populations are also declining here. The existing measures applicable to upland grasslands focus solely on manipulating vegetation, stocking rates and hydrology, but with the exception of soil moisture, do not consider the soil properties or how this impacts on the bird's foraging resource. Food availability has been suggested as playing an important role in farmland bird declines and given the importance of earthworms in the diet of waders, current environmental prescriptions may be missing an important element from their toolkit of measures.

1.3.5 Woodland edge effects

There is increasing evidence of "woodland edge effects" on waders where the displacement of breeding waders occurs due to behavioural avoidance of nesting in enclosed fields or near woodland patches (e.g. Wallander, Isaksson and Lenberg, 2006) resulting in previously suitable areas becoming unsuitable for waders following planting of conifer plantations. Indeed, several studies have reported lower abundances of waders when adjacent to commercial conifer plantations on unenclosed moorland (e.g. Stroud, Reed and Harding, 1990; Pearce-Higgins *et al.*, 2009), and lower abundance of Lapwing nests in lowland wet grasslands than expected by chance within 500 m of woodland patches (Bertholdt *et al.*, 2017). Woodland edge effects also include changes in predator density as mature woodlands may harbour predators resulting in increased predator densities and higher predator activity in the ground adjacent to woodlands which has been reported to have an effect on waders up to 1km from the edge of the woodland (Valkama and Currie, 1999; Amar *et al.*, 2011; Douglas *et al.*, 2014; Wilson *et al.*, 2014).

Woodland edge effects could have serious consequences on farmland breeding waders as the planting of trees and natural regeneration of forest is likely to increase due to government targets to combat climate change by the sequestration of carbon dioxide (CO_2; Scottish Forestry, 2020a). This will result in a large shift in habitat type, potentially replacing and fragmenting previously open landscape once suitable for waders and increasing predators (Ratcliffe, 2007; Valkama and Currie, 1999; Amar *et al.*, 2011; Douglas *et al.*, 2014; Wilson *et al.*, 2014). The current evidence base on the effects of woodland expansion in the UK is dominated by research on non-native conifer plantations (for review see Burton *et al.*, 2018), and for waders in particular, have focussed on the effects in unenclosed moorland or peatland habitat rather than farmland (e.g. Douglas *et al.*, 2014; Wilson *et al.*, 2014). Research on the edge effects of native broadleaved woodland on waders is therefore an important area of research currently missing which will be vital in helping to target tree planting schemes. This will be particularly important in the UK as limited land space is available and it is likely that upland areas which are less productive will be targeted for planting and natural regeneration, which are also areas typically characteristic of suitable wader habitat. This could have impacts on the development of future AES policy, which will be an important tool in conserving waders, for suggesting recommendations on how close tree planting near wader habitat should be.

1.4 LIME USE AS A CONSERVATION TOOL FOR THE BENEFIT OF FARMLAND BREEDING WADERS

1.4.1 pH and soil acidification

Acidity is measured along a pH scale which is the measure of the concentration of hydrogen ions (H+) within a solution following a negative logarithmic relationship (Equation 1.1). Therefore, low pH values equate to an acidic solution (high H+ concentration), and high pH values an alkaline solution (low H+ concentration). Small changes in pH equate to a large change in H+, for example a change of 1 pH unit corresponds to a ten-fold change in H+ concentration. When pH of soil is measured, it is the ratio of H+ within the soil solution, rather than the solid soil itself, although the two are strongly related.

Equation 1.1. The pH anti log equation.

$$pH = -\log_{10}[H^{+}]$$

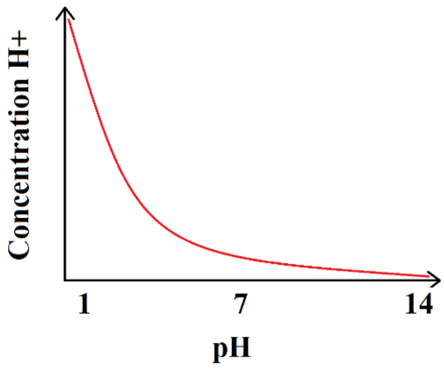


Figure 1.1. Cartoon representation of the pH scale showing a negative exponential relationship between hydrogen ion (H^+) concentration and increasing pH value, where lower values of pH have greater concentrations of H^+ (more acidic).

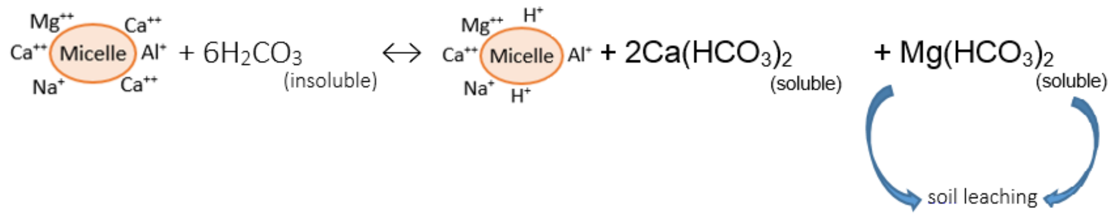
Soil pH decreases (becomes more acidic) with the leaching of calcium and other base cations from soil resulting in a change (increase) in the ratio of the H+ ions (Rowell and Wild, 1985; Equation 1.2). Ions such as calcium, magnesium and aluminium, as well as hydrogen, are present within soil and move between a state of being soluble and insoluble, depending on whether or not they are associated with soil particles, or dissolved in the soil solution. When ions are in their soluble state they are susceptible to leaching and being lost from the soil. Hydrogen ions have a higher affinity and can dislodge other ions from their soluble state. This is a natural process which can occur through respiration in the soil from micro-organisms and decomposition. With time the gradual loss of compounds results in the ratio with H+ shifting. In this way, strongly acidic soils can also be deficient in important soil nutrients. Soils have a natural buffer capacity to this form of leaching, which varies depending on the base rock and soil type. For example, clay soils have a high buffer capacity which means that it takes longer for a significant change in pH to occur and therefore pH is more stabilised over time. Soils which receive high levels of rainfall and have a low buffering capacity (e.g. sandy soils) are particularly prone to acidification.

Equation 1.2. A chemical equation representing the natural acidification process of soil. A) Carbon dioxide produced from respiring soil micro-organisms, plant roots and the decomposition of organic matter react with water in the soil to produce carbonic acid. B) Carbonic acid reacts with insoluble minerals and clay particles present in the soil (e.g. Calcium, Magnesium, and Aluminium). In this example carbonic acid is reacting with Calcium compounds in the soil. In the soluble state these compounds are susceptible to leaching and can be lost from the soil resulting in the ratio with hydrogen ions shifting. C) Cartoon representation of equation B, showing the compounds associated with soil micelles (small clay particles). Note that for all 3 equations, these are constantly in flux moving between the two states.

A) $$CO_2 + H_20 \leftrightarrow H_2CO_3$$

B) $$CaCO_{3(insoluble)} + H_2CO_3 \leftrightarrow Ca(HCO_3)_{2(soluble)}$$ soil leaching

C)



Soils also become acidified due to the deposition of acids. Acid deposition, in the form of acid rain resulting from the emissions of sulphur dioxide (SO_2) and nitrogen oxides (NO_x) from industrial pollution, was a strong driver in the acidification of habitats, particularly of waterbodies and woodland soils, prior to the Clean Air Act of 1993; however there is evidence to suggest that this form of anthropogenic acidification in the UK is now low (Carey *et al.*, 2008; Emmett *et al.*, 2010; Kirk, Bellamy and Lark, 2010). Agricultural management also results in the addition of acids to the soil (e.g. urea from livestock and use of nitrogenous fertilisers), resulting in soils becoming increasingly acidic over time (Rowell and Wild, 1985).

1.4.2 Decline in lime use in Great Britain

Soil pH influences the availability of soil nutrients, with more acidic soils inhibiting the intake of many major plant nutrients such as nitrogen, phosphorus and potassium, reducing plant growth. In addition, soil acidification leads to soil degradation and the potential release of toxic metals and aluminium into the soil water and plants. The application of agricultural lime fertiliser ($CaCO_3$) can be used to amend acidic soil and maintain soil pH by mopping up excess hydrogen ions and is considered good practice on agricultural systems for maintaining optimal growing conditions (MAFF, 1969; Wilkinson, 1998; Spaey, Traon and Joas, 2012). However, the production and application of agricultural lime fertiliser (henceforth "lime") has gradually declined in the UK following the abolition of Government subsidy in 1977/78 (Church, 1985; Skinner and Todd, 1998; Wilkinson, 1998; McCallum *et al.*, 2016) with a decline of over 70% from peak sales in the late 1960s/early 1970s (see Figure A1 in Appendix). Similar trends have also been reported within Europe of a decline in the overall production of lime by c.30% between 2007 and 2015 due to changes in demand, with only 1.9% of the lime produced in the EU used in the agricultural industry in 2018 compared to 3% in 2014 (European Comission, 2018; EuLA, 2020).

Although the importance of lime for maintaining soil quality and crop yields in arable soils is well known, grassland areas tend to be neglected as the effect of not liming is less obvious. Nevertheless, over time grassland soils with lower pH values tend to lead to the colonisation of plant species which are generally less vigorous and digestible, reducing the number of livestock which can be supported and leading to rank fields (Forbes *et al.*, 1980). The recommended optimum soil pH for grassland fields is pH 6.0 on mineral soils and around pH 5.5 for organic soils (DEFRA, 2010; Sinclair, Crooks and Coull, 2014). However, the Professional Agricultural Analysis Group (PAAG), which publishes annual reports on soil samples taken from agricultural fields in the UK, have shown that over 50% of grassland soil samples have a pH below 6.0 and c.20% below pH 5.5 (PAAG, 2020). Although it should be noted that the PAAG soil samples are not randomised and may not be a true representation of UK fields, several other studies have also reported a trend in declining soil pH between the 1970s and early 2000s (Goulding and Annis, 1998) particularly in the West and North of the UK (Skinner and Todd, 1998; Baxter, Oliver and Archer, 2006).

1.4.3 Implications of soil pH for wader conservation

The uplands of Scotland and Northern England support a high proportion of the UK's breeding population of waders and are dominated by mixed livestock grazed grasslands (Wilson *et al.*, 2005). These areas receive high levels of rainfall and are characterised by typically peaty soils with a low buffer capacity and a high organic matter content that results in these areas being particularly prone to acidification over time (White, 2006; Aitkenhead *et al.*, 2012). This could have implications on wader populations as soil pH influences soil invertebrates, such as earthworms which are less abundant in acidic soil conditions (Edwards, Hendrix and Arancon, 1996). Indeed, in Scottish grasslands the distribution of earthworm species found in the topsoil (endogeic) and deep burrowing (anecic) species are thought to be strongly limited by low pH (Boag *et al.*, 1997). Given the importance of earthworms in the diet of waders, this could pose a serious threat. The importance of food in determining breeding success in birds is widely acknowledged and a reduction in food abundance and availability as a result of changes in agriculture management is one of the main factors contributing to lapwing decline in Britain (Vickery *et al.*, 2001; Wilson, Vickery and Browne, 2001). It is possible that historical liming subsidies which reduced soil acidity may have had important benefits in supporting farmland breeding waders on upland grasslands.

There is a vast body of literature on the negative effects of acidification of habitats in parts of central Europe and America, from acid rain. Declines in fish, invertebrates and tree seeds have been recorded, and implicated in the decline in abundance and reproduction of birds associated with the acidified habitats (Eriksson, 1986; Ormerod and Tyler, 1986; McNicol, Bendell and Ross, 1987; Alvo, Hussell and Berrill, 1988; Ormerod *et al.*, 1988; Graveland, 1990; Graveland and Drent, 1997; Tilgar, Mänd and Mägi, 2002). Significant correlations between high levels of acid deposition and the absence of breeding Wood Thrush (*Hylocichla mustelina*) in America (Hames *et al.*, 2002) also add to the evidence to suggest that the acidification of habitats may play a role in the local decline of bird species reliant on food resources sensitive to pH. Liming has been used to mitigate the effects of acidic deposition on forest soils and add calcium back into the soil with the goal of promoting tree growth since the 1980s and is a common and widespread practice in some parts of Europe, predominantly the Netherlands, Germany, France, Czech, Finland and Sweden (Auclerc *et al.*, 2012; Reid and Watmough, 2014). The primary aim of this management is to prevent forest dieback and improve the production of wood biomass for harvesting. However, ecological studies have also been conducted on the effects of liming on forest invertebrates with mixed results, partly because the effect of lime may vary over time and therefore depends on the length of the study as well as the amount applied, target pH, soil type, lime use and starting pH e.g.(Buckton and Ormerod, 1997; Korenko *et al.*, 2008; Sinclair, Crooks and Coull, 2014). Snails are one group which have shown positive effects of liming in forests, leading to lime use being proposed as a technique for restoring forest ecosystems impacted by acidic deposition for the benefit of birds which prey on snails (Pabian *et al.*, 2012). Snail abundance is positively correlated with calcium content (and to a lesser extent, pH) of the litter layer of the forest soil, with liming benefitting great tits, *Parus major*, in the Netherlands (Graveland and Van Der Wal, 1996) and Ovenbirds, *Seiurus aurocapilla*, in America through the increase in snails brought about by liming (Pabian and Brittingham, 2007; Pabian *et al.*, 2012).

The evidence for the positive effect of liming on the abundance, biomass and species richness of earthworms has also been shown by several studies of acidified forests with positive effects on earthworms that can persist over tens of years (Robinson *et al.*, 1992; Edwards, Hendrix and Arancon, 1996; Deleporte and Tillier, 1999; Davidson *et al.*, 2004; Potthoff *et al.*, 2008; Ampoorter *et al.*, 2011). For example, inoculation studies of the earthworm species *Allolobophora caliginosa* into limed versus un-limed plots in acidified forests reported higher abundances and biomass in the treatment plots several years after liming had occurred thought to be caused by an increase in cocoon production and immigration rates (Huhta, 1979; Robinson *et al.*, 1992). Similarly, liming of acidified forests in North America and of Sugar Maple stands (*Acer saccharum*) in Canada has led to the colonisation of earthworm species, primarily *Lumbricus terrestris*, which were previously absent and increased earthworm reproductive output due to the improved soil conditions (Moore, Ouimet and Bohlen, 2013; Homan *et al.*, 2016). It is therefore plausible that similar effects on earthworms following lime application to upland grasslands in the UK would be expected and may therefore benefit farmland breeding waders. However, to date, there have been relatively few studies considering the effect of lime management on earthworms of upland grassland in the UK (e.g. NERC Soil Biodiversity Programme at Sourhope, Scotland; Usher *et al.*, 2006) and those that have were at the plot scale (Bishop, 2003; Bishop *et al.*, 2008; McCallum, 2012) and had mixed results in terms of species level effects. A few studies have considered field use by lapwing and lime use on temperate grasslands. Field use of Lapwing and Black-tailed Godwit have been observed to increase following lime application, which was coupled with an observed increase in earthworm abundance (Brandsma, 2004). Furthermore, McCallum *et al.*, (2018) observed that the density of breeding Lapwings was 52% higher on fields which had been limed as part of re-seeding grass leys following fodder crop management than fields which had not undergone this process at a single farm within Scotland. The positive effects on field soil pH and higher Lapwing densities than control fields persisted for up to/about seven years. On a larger spatial scale, a positive additive effect was observed between the abundance of Lapwing and soil pH on upland farmland in Scotland, further suggesting that Lapwing numbers are influenced by soil pH, potentially due to the availability of earthworm prey (McCallum *et al.*, 2015).

1.5 THESIS RATIONAL AND RESEARCH OBJECTIVES

The current literature provides observational evidence that localised acidification can affect birds by reducing the availability of invertebrate prey, with some evidence to suggest that this may be amended by lime use, although experimental data is scarce or has produced mixed results. Few studies of lime use and soil acidification on earthworms have been conducted on agricultural grasslands, and even fewer in the context of farmland birds. However, recent research by McCallum et al., (2015, 2016) has suggested that there may be a strong link between soil pH and Lapwing which has previously been overlooked in conservation management interventions. It is therefore possible that this management could be a simple conservation intervention for the benefit of waders and other farmland birds which rely on earthworm prey. This would be a particularly attractive management option, as lime management also has an economic benefit to farmland by restoring soil pH to optimum conditions for grass production and could therefore provide both conservation and economic gains. Experimental trials of lime use are needed to test the observational studies on lapwing and to determine if benefits (or negatives) of lime use occur on other wader species and non-target groups to inform and develop the appropriate conservation management.

Therefore, the overarching aim of this thesis is to quantify the effect of lime use on marginal upland agricultural grassland for the benefit of farmland breeding waders as a conservation tool, as well as to investigate the current AES options available for the benefit of waders. The first three data chapters of this thesis addresses the effect of lime use on marginal agricultural grassland examining the effects of waders from the bottom up, i.e. from the use of lime on soil condition and soil invertebrates (Chapter 2), on vegetation and soil surface invertebrates (Chapter 3), and on wader abundance and feeding metrics (Chapter 4). The final data chapter in this thesis is a case study of waders in the Badenoch and Strathspey and investigates the impacts of AES measures on wader densities through proxy habitat variables (Chapter 5). The hypotheses tested in each chapter are as follows:

- Chapter 2: This chapter aims to provide experimental evidence, using a four year before-after-control impact (BACI) study covering a large geographical area, to test the hypothesis that the use of lime can mitigate acidified pH, leading to increases in earthworm abundance and thus improve foraging potential of fields for bird species which prey on earthworms.
- Chapter 3: This chapter aims to test the effect of lime use on other non-target organisms, namely beetles (Coleoptera) and spiders (Araneae). For new AES measures to be successful, it is important to ensure that they do not negatively affect other taxa. Additionally, changes to vegetation from lime use may impact foraging and accessibility of prey to waders, which would reduce the effectiveness of lime as a conservation tool.
- Chapter 4: lime use increases wader field use and wader feeding success. This chapter aims to test the effect of lime use on the foraging behaviour and density of four wader species.
- Chapter 5: This chapter aims to use farmland breeding wader survey data between 2000 to 2015 in the Badenoch and Strathspey to investigate habitat associations, test for an effect of year on density, and to test for woodland edge effects.

CHAPTER 2 | EFFECTS OF LIME MANAGEMENT ON EARTHWORMS IN UPLAND AGRICULTURAL GRASSLANDS: IMPLICATIONS FOR FARMLAND BREEDING WADERS.

2.1 ABSTRACT

Changes in agricultural management and intensification have been linked to the observed declines of biodiversity loss within Europe across a wide range of taxa, with evidence for the negative effect of modern agriculture practices on farmland birds being particularly well studied. In the United Kingdom (UK), one change in farming management, which is often not considered in the context of biodiversity benefits, is the use of lime fertiliser. This has seen a 70% decline in use since the 1970s alongside a documented gradual acidification of agricultural soils, particularly in the north and west of the UK dominated by upland agricultural grassland. Lime fertiliser amends acidic soil condition and is used in agricultural systems to improve fields for crop and grass production. Acidic soil conditions also have an effect on soil biodiversity, in particular earthworms, which are a key food item for many high priority conservation species of concern in the UK, notably the Northern Lapwing (*Vanellus vanellus*) and Eurasian Curlew (*Numenius arquata*). In this chapter, I investigate the relationship between earthworms and soil properties of marginal upland agricultural grasslands in the UK across nine farms (n = 19 fields). I experimentally test the effect of lime fertiliser ($CaCO_3$) on fields which have acidic soil pH following a before-after-control-impact (BACI) experimental design running for four years (2014, 2015, 2016, and 2017) on both the soil properties and on earthworms. Soil pH was identified as one of the most important predictors of total earthworm abundance, with a marked additive effect of soil pH, with earthworms ten times as abundant at high (pH 6.0) as at the low (pH 4.0) extremes of soil pH recorded in this chapter. Lime treated soil in the BACI experiment was, on average, a higher pH (more neutral) in comparison to control soil in years 2015, 2016 and 2017 but not in the year 2014 prior to lime application. Earthworm abundance also showed a significant interaction effect between Treatment and Year, with the largest effect seen between lime treatment and control (non-limed) in the second sampling year after lime was applied to treatment plots. The results of this study suggests that active management to improve soil condition via the application of lime fertiliser on agricultural grasslands with low (acidic) soil pH has the potential to be a simple and effective method to increase earthworm abundance. Targeted lime management of grassland fields may therefore have the capacity to be included amongst the tools available by conservationists and land managers to benefit farmland birds of high conservation concern which rely on earthworm prey, by increasing the foraging potential of fields. This is especially important in areas of important nesting/foraging habitat particularly where the underlying geology has poor buffering capacity and high levels of rainfall which can lower soil pH through leaching.

2.2 INTRODUCTION

Declines in the distribution and abundance of wildlife associated with agricultural ecosystems have been documented throughout Europe across a wide range of taxa including birds (Tucker and Heart, 1994; Chamberlain *et al.*, 2000; Donald, Green and Heath, 2001; Vickery *et al.*, 2001; Donald *et al.*, 2006), butterflies and moths (Heath, Pollard and Thomas, 1984; Thomas, 1995; Warren *et al.*, 1997; Maes and Van Dyck, 2001; Conrad *et al.*, 2004, 2006; New, 2004; Reynolds and Perrins, 2010; van Strien *et al.*, 2019), beneficial insects (Aebischer, 1991; Biesmeijer *et al.*, 2006; Attwood *et al.*, 2008; Le Féon *et al.*, 2010) and annual arable wildflowers (Schumacher, 1987; Andreasen, Stryhn and Streibig, 1996) to name a few. The increasing body of evidence suggests that the processes of agricultural intensification have been responsible for many of these losses (e.g. Krebs *et al.*, 1999; Aebischer *et al.* 2000; Robinson and Sutherland, 2002; Newton, 2004; Vickery *et al.*, 2004; Stoate *et al.*, 2009; Foley *et al.*, 2011). Although agricultural intensification has taken many forms in different countries and encompasses a wide range of components, there has been a general trend for widespread drainage of grassland within Europe, a switch to more productive grass species, higher livestock densities, and increased cutting of hay and silage. Within arable systems there has been an increase in chemical input from nitrogen fertilisers and pesticides, a loss of non-farmed habitat such as field margins, ponds and hedges, changes to the timing of farming management including sowing and harvesting, and a decrease in the variety of crop types planted. Consequentially, intensification of the farmed landscape has seen an overall loss in small farms and mixed farming systems, a geographically polarised landscape and an increasingly simplified and uniform habitat with widespread wildlife losses (e.g. Stoate, 1996; Maton *et al.*, 1997; Chamberlain *et al.*, 2000; Tilman *et al.*, 2001; Robinson and Sutherland, 2002; Wilson, Evans and Grice, 2009).

The introduction of the European Union's (EU's) the Common Agriculture Policy (CAP) in the 1960s, has been pivotal in driving the intensification of farmland within Europe (Robson, 1997). However, increasing concern over the negative effect of intensification on the environment, particularly with the observed sharp decline in many farmland species such as birds, has resulted in several reforms of the CAP since 1992, which have aimed to reduce pressures on the environment (Beintema, Dunn and Stroud, 1997). One of the outcomes of this has been the development of agri-environment schemes (AES) in which financial support is provided to EU Member States to design and implement agri-environment measures under the Member State's Rural Development Programme. AES are mandatory for national and regional administrations, but voluntary for farmers to uptake. Payments are provided to farmers who undertake environmentally friendly practices and act to counterbalance any potential loss of profit, with a range of measure available which may concentrate on low-intensity production, organic farming, soil conservation, crop rotation, or the enhancement of habitat for biodiversity on farmland. Uptake has been high in the EU with c.25% of farmland under some form of AES agreement with farmers, and an estimated €23 billion spent on AES between 2007 and 2013 (Science for Environment Policy, 2017a). However, to date, the evidence for the effectiveness of AES to reduce or halt biodiversity losses has been mixed (e.g. Kleijn *et al.*, 2001, 2004; Kleijn and Sutherland, 2003; Verhulst, Kleijn and Berendse, 2006; Wilson, Vickery and Pendlebury, 2007; Blomqvist, Tamis and Snoo, 2009; Wilson, Evans and Grice, 2009; Batary *et al.*, 2010; Princé, Moussus and Jiguet, 2012; Pe'er *et al.*, 2014). Although there is some evidence of AES implementation yielding recovery in farmland wildlife populations at the local scale (e.g. Aebischer, N.J., Evans, A.D., Grice, P.V. & Vickery, 2000; Peach *et al.*, 2001; Vickery *et al.*, 2004; Yela *et al.*, 2006; Carvell *et al.*, 2007; Maes, Musters and De Snoo, 2008; Perkins *et al.*, 2011) declines are still occurring nationally and studies which quantify long-term population responses to AES over the larger scale are scarce (e.g. Peach *et al.*, 2001; Wilson, Vickery and Pendlebury, 2007; Haye *et al.*, 2010). The success of AES may be being limited by the scale (temporal and spatial) of the implementation (O'Brien and Wilson, 2011), poor design of measures that vary in their effectiveness (Smart *et al.*, 2013) or that the current options are missing specific habitat requirements of the target species (Redpath *et al.*, 2010; Fuentes-Montemayor, Goulson and Park, 2011). Therefore, well conducted monitoring of the target species is essential to test the effectiveness of management on the ground and to inform adaptive prescriptions which can be tailored and improved over time and across a species' range (Salafsky *et al.*, 2002; Memmott *et al.*, 2010), as well as the need for carefully constructed field-tests of novel solutions to identify future prescriptions. Maximising the effectiveness of AES to deliver conservation outcomes is a key policy challenge (Sutherland *et al.*, 2006) which needs to maintain close links between policy and science to justify the resources required (Batáry *et al.*, 2015).

Farmland birds are an ideal study group for investigating AES options due to the extensive research which has been conducted within Europe on the effects of modern agricultural management on birds and their ecological requirements (Donald, Green and Heath, 2001; Newton, 2004; Donald *et al.*, 2006; Wretenberg *et al.*, 2007; Wilson, Evans and Grice, 2009; Traba and Morales, 2019). Some generalist species including Common Woodpigeon (*Columba palumbus*) and Eurasian Jackdaw (*Corvus mondedula*) have seen an increase in abundance since 1970 due in part to the increased availability of food from the change to autumn sowing in arable systems. Other groups such as Northern Lapwing (*Vanellus* vanellus), Eurasian Curlew (*Numenius arquata*), Black-tailed Godwit (*Limosa* limosa) and Eurasian Oystercatcher (*Haematopus ostralegus*), more commonly referred to as farmland breeding waders, have observed dramatic and long-term declines, both in range and population size and are classified as globally Near Threatened with extinction (Birdlife International, 2020). Drainage and agricultural conversion of wetlands has resulted in the loss of around half of global wetlands since 1900 and is the main cause of the global population decline of waders (Dugan, 1993; Wilson, Evans and Grice, 2009; IUCN, 2015). However, within Europe these birds are considered species of the wider countryside and depend on farmland for breeding and foraging habitat, particularly agricultural grasslands (Piersma, 1986). The fragmentation and loss of breeding habitat on farmland associated with agricultural intensification is a key factor in the European population declines of waders. Intensification practices detrimental to waders on grassland include (but are not exclusively limited to) the drainage of wet grasslands, increased livestock densities, and changes in the timings and frequency of agricultural operations (Vickery *et al.*, 2001; Wilson, Vickery and Pendlebury, 2007) which have reduced both the availability and security of suitable nesting habitat and the availability of large, soft-bodied soil arthropod prey which these birds depend upon (Newton, 2004; Wilson, Evans and Grice, 2009). Predation has also been shown to be having an adverse effect on farmland breeding waders and may be suppressing populations in some locations (Macdonald and Bolton, 2008; Teunissen *et al.*, 2008; Roodbergen, Werf and Hötker, 2012; Douglas *et al.*, 2014; Manton *et al.*, 2016; Roos *et al.*, 2018), with higher densities of predators of waders often linked with fragmented agricultural landscapes (Valkama and Currie, 1999; Cervinka *et al.*, 2011). Consequentially, AES prescriptions have been developed specifically to target farmland breeding waders, with particular conservation effort within the Netherlands and the United Kingdom (UK), both of which hold internationally important breeding populations of waders (Thorup, 2004). Conservation management options include manipulation of stocking rates and mowing dates to reduce loss of nests and chicks (Beintema and Muskens, 1987; Green, 1988), the creation of wet features such as shallow pools and scrapes for improved foraging habitat (Smart *et al.*, 2006; Eglington *et al.*, 2010), and more recently support for predator control is available as a capital item in Scotland on a restricted basis at Sites of Special Scientific Interest (SSSIs) and Natura sites only (Scottish Government, 2014) when sufficient evidence is provided of an adverse effect of predators on ground nesting birds (Fletcher *et al.*, 2010; Malpas *et al.*, 2013). Despite this conservation effort, the evidence for wader AES suggests that current management has not been successful at reversing or halting the trend of population decline at the national or European level, although it has been able to slow the rate of decline at the local level for farmland breeding waders in some cases through increased nest success and abundance (Ausden and Hirons, 2002; Kleijn and van Zuijlen, 2004; Ottvall and Smith, 2006; Sheldon, Chaney and Tyler, 2007; O'Brien and Wilson, 2011; Rickenbach *et al.*, 2011; Samantha E Franks *et al.*, 2018).

One aspect of agricultural management which has received relatively little attention in the context of conservation management for farmland breeding waders is soil condition, despite the effect that agricultural management can have on soil properties, vegetation and consequentially soil invertebrates (Webb *et al.*, 2001; Bardgett *et al.*, 2005; Cole, Buckland and Bardgett, 2005; White, 2006). Although there has been considerable research on farmland bird habitat associations few studies consider soil pH (e.g. McCallum *et al.*, 2015; Johannesdottir *et al.*, 2018) or soil organic matter content (e.g. Gilroy *et al.*, 2008, 2010; McCallum *et al.*, 2015). However, the association between waders and soil moisture is well established e.g.(Devereux *et al.*, 2004; Verhulst, Kleijn and Berendse, 2006; Vanderhoff and Eason, 2008; Olsson and Rogers, 2009; Rhymer *et al.*, 2010; Berndt and Hölzel, 2012) particularly for Lapwing which rely on wet habitats as important foraging sites for invertebrate prey (Berg, 1993; Mckeever, 2003; Verhulst, Kleijn and Berendse, 2006; Eglington *et al.*, 2007, 2010; Eglington, 2008; Rhymer *et al.*, 2010). Nonetheless, soil pH is one of the main determinants of soil biodiversity and functioning (Edwards, Hendrix and Arancon, 1996; Cole *et al.*, 2006; Fierer and Jackson, 2006; Griffiths *et al.*, 2011) and is relatively simple to manipulate either with liming agents to amend acidic soil or with acidifying agents such as sulphur to amend alkaline soil. The diversity and abundance of many soil invertebrate are sensitive to soil pH (Edwards, Hendrix and Arancon, 1996; Cole *et al.*, 2006) and constitute an important food resource for a wide range of farmland birds including gulls, thrushes, starlings, corvids and waders (Barnard and Thompson, 1985; Tucker, 1992; Wilson, Evans and Grice, 2009). For example, the presence of earthworms is linked to the breeding season success rates of the song thrush (*Turdus philomelos*) on lowland farmland in the UK, a bird species whose populations have rapidly declined in England (Gruar, Peach and Taylor, 2003). Earthworms make up the bulk of soil fauna biomass of temperate grasslands, accounting for up to 75% (Bardgett and Cook, 1998) and are more abundant under neutral soil conditions (Edwards, Hendrix and Arancon, 1996). At low pH values, earthworms may not only be less numerous but can also go into diapause more rapidly upon the onset of dry conditions (Doekson and van Wingerden, 1964).

Although opportunistic feeders with an eclectic diet, earthworms have been identified as a key food resource for Lapwing and Curlew (Barnard and Thompson, 1985; Galbraith, 1989; Baines, 1990; Beintema *et al.*, 1991; Berg, 1993; Sheldon, 2002; Shrubb, 2007; Watkins, 2007; Navedo *et al.*, 2020). Earthworms are high in nutrients including protein, calcium and iron (Paoletti *et al.*, 2002) and are therefore an important food item for birds getting into condition during the pre-breeding season (Högstedt, 1974; Galbraith, 1989; Berg, 1993) as well as for chick development (Beintema *et al.*, 1991; Sheldon, 2002). Although the length of the Curlew bill means that it can access earthworms and invertebrate food items along a broader depth range within the soil profile, a trade-off between energy intake rate and effort may exist as Curlew have been observed to preferentially select smaller earthworms on the soil surface, which are known as epigeic earthworms (Navedo *et al.*, 2020). Similarly, reconstruction of Lapwing diet from faecal samples have shown that in terms of biomass, small epigeic earthworms make up the higher proportion of prey items taken during the day; however, nocturnally this switches to large deep burrowing earthworms (anecic earthworms) which come up to the surface to feed at night (Gillings and Sutherland, 2007). For Snipe, the dry weight of prey ingested between April and June constituted 61% of earthworms, with the remainder of prey being made up of tipulid larvae (24%) and a wide variety of surface-active and aquatic prey (Hoodless, Ewald and Baines, 2007). Earthworms also feature in the diets of redshank, oystercatcher and the chicks of ruff, *Philomachus pugna,* (Beintema *et al.*, 1991). Lapwing chicks have also been observed foraging in locations with higher densities of one species of earthworm, *Allolobophora chlorotica*, which is an acid-intolerant species of earthworm, when compared with randomly selected locations in the same field in an upland grassland farm within Scotland (McCallum *et al.*, 2016). However, earthworms are likely to play a smaller role in the diet of very young chicks as the smaller bills may not be capable of catching earthworms. Indeed, Watkins (2007) observed that chicks older than 12 days foraged in relatively earthworm rich areas within fields, whereas the opposite was true for younger chicks. Similarly, Beintema *et al.*, (1991) identified an increase in the number of earthworms consumed by Lapwing chick with increasing chick age, despite decreasing availability of earthworms as the breeding season progressed suggesting the importance of earthworms in chick development and growth.

The process of soil acidification can occur naturally from leaching of calcium and other cations resulting in a change in the ratio of hydrogen ions following heavy rainfall, but this process is also intensified by anthropogenic activity such as acid deposition from acid rain, and as a result of the addition of nutrients and ammonia from livestock excretion and nitrogen based fertilisers under agricultural management (Gasser, 1985; Rowell and Wild, 1985; Johnston, Goulding and Poulton, 1986). The uplands of Scotland and Northern England, which support a high proportion of the UKs breeding population of waders which have suffered dramatic declines on lowland agricultural systems (Wilson *et al.*, 2005), receive high levels of rainfall and are characterised by typically peaty soils with a low buffer capacity and a high organic matter content that results in these areas being particularly prone to acidification over time (White, 2006; Aitkenhead *et al.*, 2012). In Scottish grasslands the distribution of earthworm species found in the topsoil (endogeic) and deep burrowing (anecic) species are thought to be strongly limited by low pH (Boag *et al.*, 1997). Soil pH also influences the availability of soil nutrients, with more acidic soils inhibiting the intake of many major plant nutrients such as nitrogen, phosphorus and potassium, reducing plant growth. In addition, soil acidification leads to soil degradation and the potential release of toxic metals and aluminium into the soil water and plants. The application of agricultural lime fertiliser ($CaCO_3$) can be used to amend acidic soil and maintain soil pH by mopping up excess hydrogen ions and is considered good practice on agricultural systems for maintaining optimal growing conditions (MAFF, 1969; Wilkinson, 1998; Spaey, Traon and Joas, 2012). However, the production and application of agricultural lime fertiliser (henceforth "lime") has gradually declined in the UK following the abolition of Government subsidy in 1977/78 (Church, 1985; Skinner and Todd, 1998; McCallum *et al.*, 2016) with a decline of over 70% from peak sales in the 1960s (Wilkinson, 1998). Similar trends have also been reported within Europe of a decline in the overall production of lime by c.30% between 2007 and 2015 due to changes in demand, with only 1.9% of the lime produced in the EU used in the agricultural industry in 2018 compared to 3% in 2014 (European Comission, 2018; EuLA, 2020). Although the importance of lime for maintaining soil quality and crop yields in arable soils is well known, grassland areas tend to be neglected as the effect of not liming is less obvious. Nevertheless, whilst acidity under grasslands may appear to have little effect, over time the productive grass and clover species generally become replaced by less vigorous and less digestible species at lower pH values which can reduce the number of livestock which can be supported and lead to rank fields (Forbes *et al.*, 1980). The recommended optimum soil pH for grassland fields is pH 6.0 on mineral soils and around pH 5.5 for organic soils (DEFRA, 2010; Sinclair, Crooks and Coull, 2014). However, the Professional Agricultural Analysis Group (PAAG), which publishes annual reports on soil samples taken from agricultural fields in the UK, consistently shows that over 50% of grassland soil samples have a pH below 6.0 and c.20% below pH 5.5 (PAAG, 2020). Although it should be noted that the PAAG soil samples are not randomised and may not be a true representation of UK fields, several other studies have also reported a trend in declining soil pH between the 1970s and early 2000s (Goulding and Annis, 1998) particularly in the West and North of the UK (Skinner and Todd, 1998; Baxter, Oliver and Archer, 2006). This may have important implications for the conservation of farmland breeding waders as localized acidification, potentially related to a reduction in lime use of upland grassland systems (Kuylenstierna and Chadwick, 1991; Baxter, Oliver and Archer, 2006; Goulding, 2016) may have reduced the quality of foraging habitat due to lower abundances of earthworms in less neutral soil conditions (Edwards, Hendrix and Arancon, 1996).

Amending acidic soil pH through the application of lime has shown some positive results as a tool for improving earthworm populations of acidified coniferous forests (damaged through air pollution) in parts of central Europe with positive effects on earthworms that can persist over tens of years (e.g. Deleporte and Tillier, 1999; Potthoff *et al.*, 2008). For example, several inoculation studies of the earthworm species *Allolobophora caliginosa* into limed versus un-limed plots in acidified forests reported higher abundances and biomass in the treatment plots several years after liming had occurred thought to be caused by an increase in cocoon production and immigration rates (Huhta, 1979; Robinson *et al.*, 1992). Similarly, liming of acidified forests in North America and of Sugar Maple stands (*Acer saccharum*) in Canada has led to the colonisation of earthworm species, primarily *Lumbricus terrestris*, which were previously absent and increased earthworm reproductive output due to the improved soil conditions (Moore, Ouimet and Bohlen, 2013; Homan *et al.*, 2016). Similar effects on earthworms could therefore be expected when lime is applied to upland grasslands in the UK. However, there have been relatively few studies considering the effect of lime management on earthworms of upland grassland in the UK (e.g. NERC Soil Biodiversity Programme at Sourhope, Scotland; Usher *et al.*, 2006) and those that have were at the plot scale (Bishop, 2003; Bishop *et al.*, 2008; McCallum, 2012) and had mixed results in terms of species level effects. There are 31 known species of earthworm in the UK (Sherlock, 2018) which can be separated into three ecological functional groups (Bouché, 1977): epigeic earthworms which are small surface dwelling earthworms that break down surface organic matter content; endogeic earthworms which live and feed in the top soil creating non-permanent horizontal burrows; and anecic earthworms which are long lived, deep vertical burrowing species. Different earthworm species will have different tolerances to soil pH, and earthworms within different ecological groups will also be influenced differently by the neutralising effect of lime, which will be greatest on the ground surface and in the top few centimetres of soil only, particularly when lime is applied as a surface dressing rather than mixed into the soil through conventional tillage processes which are known to be detrimental to both epigeic and anecic earthworms (Briones and Schmidt, 2017). Indeed, where increases in earthworm abundance following liming have been reported the primarily increases are in epigeic species (Deleporte and Tillier, 1999; Bishop, 2003; Bishop *et al.*, 2008; McCallum *et al.*, 2016), although smaller increases in both endogeic and anecic species have also been found (Bishop, 2003; Potthoff *et al.*, 2008) with the main mechanism behind the higher earthworm numbers following liming thought to be the increase in soil pH rather than the addition of calcium (Springett and Syers, 1984).

Field use of waders have also been observed to increase following lime application on upland grasslands in the UK of Lapwing and Black-tailed Godwit, which was coupled with an observed increase in earthworm abundance (Brandsma, 2004). Furthermore, McCallum *et al.*, (2018) observed that the density of breeding Lapwings was 52% higher on fields which had been limed as part of re-seeding grass leys following fodder crop management than fields which had not undergone this process at a single farm site within Scotland. The positive effects on field soil pH and higher Lapwing densities than control fields was reported to persist for seven years. On a larger spatial scale, a positive association between the abundance of Lapwing and soil pH in upland farmland has been found in species distribution models for Scotland, further suggesting that Lapwing numbers are influenced by soil pH, probably due to the availability of earthworm prey (McCallum *et al.*, 2015). Overall, this suggests that there may be a strong link between soil pH and farmland breeding waders which has previously been overlooked in conservation management interventions. It is possible that historical liming subsidies which reduced soil acidity may have had important benefits in supporting farmland breeding waders on upland grasslands, and therefore that this management could be a simple and novel conservation intervention for the benefit of waders and other farmland birds which rely on earthworm prey if it was shown to be able to improve foraging conditions at the field scale. This would be a particularly attractive management option, as lime management also has an economic benefit to farmland by restoring soil pH to optimum conditions for grass production and could therefore provide both conservation and economic gains; however, large scale experimental trials to re-enforce the observational studies on lime use is required in order to inform and develop the appropriate conservation management.

In this chapter I investigate the relationship between earthworms and soil properties from nine farms (n = 19 fields) consisting of agricultural grasslands of marginal uplands in the North of the UK, which are areas characterised by typically peaty soils with a low buffer capacity and a high organic matter content that results in these areas being particularly prone to acidification over time. Specifically this observational study will help to test the hypothesis (1) that earthworm abundance is positively correlated with soil pH. In order to determine if lime use may be beneficial for farmland breeding waders, I will also experimentally test the effect of lime management on both soil properties and on earthworm abundance using controlled field experiments following a before-after-control-impact (BACI) design of surface lime dressing (no-till) spanning a four year period at seven agricultural farms (n = 15 fields) across a broad geographical range and soil types. BACI designs help to control for differences among sites and for influences that all sites are subjected to which are out of our control when conducted in the natural world, such as the environment, and will therefore provide greater statistical power to find a general effect of lime treatment on our response. Specifically the BACI experiment will test the hypothesis (2) that lime treated soil has a higher (more neutral) soil pH than control (non-limed) soil, and (3) that lime treated soil has higher earthworm abundance than control (non-limed) soil. Furthermore, it was anticipated that a lag effect of one year would occur between the use of lime treatment and an effect in the response of both soil properties and earthworm abundance due to the time it takes lime to neutralise acidity, and therefore that an effect would not be detected in soil pH or earthworm abundance until the second sampling year post the application of lime. The results of this chapter will help to provide empirical evidence on the effect of lime management as a potential tool to manipulate soil condition for the benefit of waders and other farmland birds on upland grassland sites prone to acidification where earthworms constitute a major component of their diet.

Specifically, this chapter addresses the following three research questions:

1. What is the relationship between earthworm species composition and abundance with soil properties (pH, moisture and organic matter)?
2. Does lime applied as a surface dressing (no-till) amend sub-optimal soil pH, and what is the magnitude of effect over time?
3. What is the effect of lime on earthworm abundance, and are there detectable differences in response at the species or ecological grouping level?

The first of these questions is addressed using data collected spring 2014 to avoid any bias from the experimental manipulation of soil pH following lime application in autumn 2014, and is referred to as the observational study throughout this chapter. The subsequent two research questions (hypothesis 2 and 3) use four survey years of data (2014, 2015, 2016 and 2017) from the BACI liming experiment.

2.3 METHODS

2.3.1 Study area

The study was conducted on marginal upland agricultural grasslands covering a broad geographical range across the UK with a total of ten farms (n = 20 fields) sampled in the first year of the study (Figure 2.1). However, only nine of these farms had earthworms sampled (n = 19 fields) and data on soil properties at the soil core level was only available for one field from farm I in Northern Ireland, thus formal analysis for the observation study consisted of nine farms (n = 18 fields) only (Table 2.1). There is no statutory definition for the uplands, but in this case it refers to ground that is elevated above the lowlands and between the hills with land that is less productive than lowland arable fields. Farming in this region is typically extensive, and primarily focussed on rearing sheep. The farms consisted of six agricultural farms and four farm reserves which were managed by the Royal Society for the Protection of Birds (RSPB). The farms consisted of mixed livestock grazing in marginal upland areas spanning five broad soil types (Table 2.2).

The BACI liming experiment was conducted on a subset of the farms due to unforeseen circumstances preventing the application of lime at some of the farms and fields, resulting in a total of 7 farms (n = 15 fields) receiving lime treatment. However, the number of farms and field surveyed each year varied due to logistical fieldwork constraints resulting in fewer fields and sites monitored in year 2 and 3 (Table 2.1). Fields selected for the liming experiment were permanent "in-bye" pastures greater than 4ha in size, with an acidic mean field soil pH of 5.4 or less (which is below that of the recommended optimum grass growing conditions), a uniform structure, no history of lime spreading in the previous 10 years and with a record of farmland breeding waders at the farm. Here "in-bye" refers to enclosed fields of land on upland farm sites. The term does not include unenclosed fields on the hill or rough moorland (known as "out-bye").

Table 2.1. Number of farms and fields that had soil cores sampled to survey earthworms and soil properties by year for **(A)** the observational study; and **(B)** the BACI liming experiment. Where Year 0 = baseline data prior to any application of lime treatment. Year 1 = data one year post the application of lime onto treatment plots. Year 2 = data two years post the application of lime onto treatment plots. Year 3 = data three years post the application of lime onto treatment plots.

Year	Number farms	Number fields	Number soil cores	Number earthworm species
(A)				
Year 0	9	18	360	14
(B)				
Year 0	7	15	300	13
Year 1	7	15	300	14
Year 2	6	13	260	15
Year 3	4	11	220	9

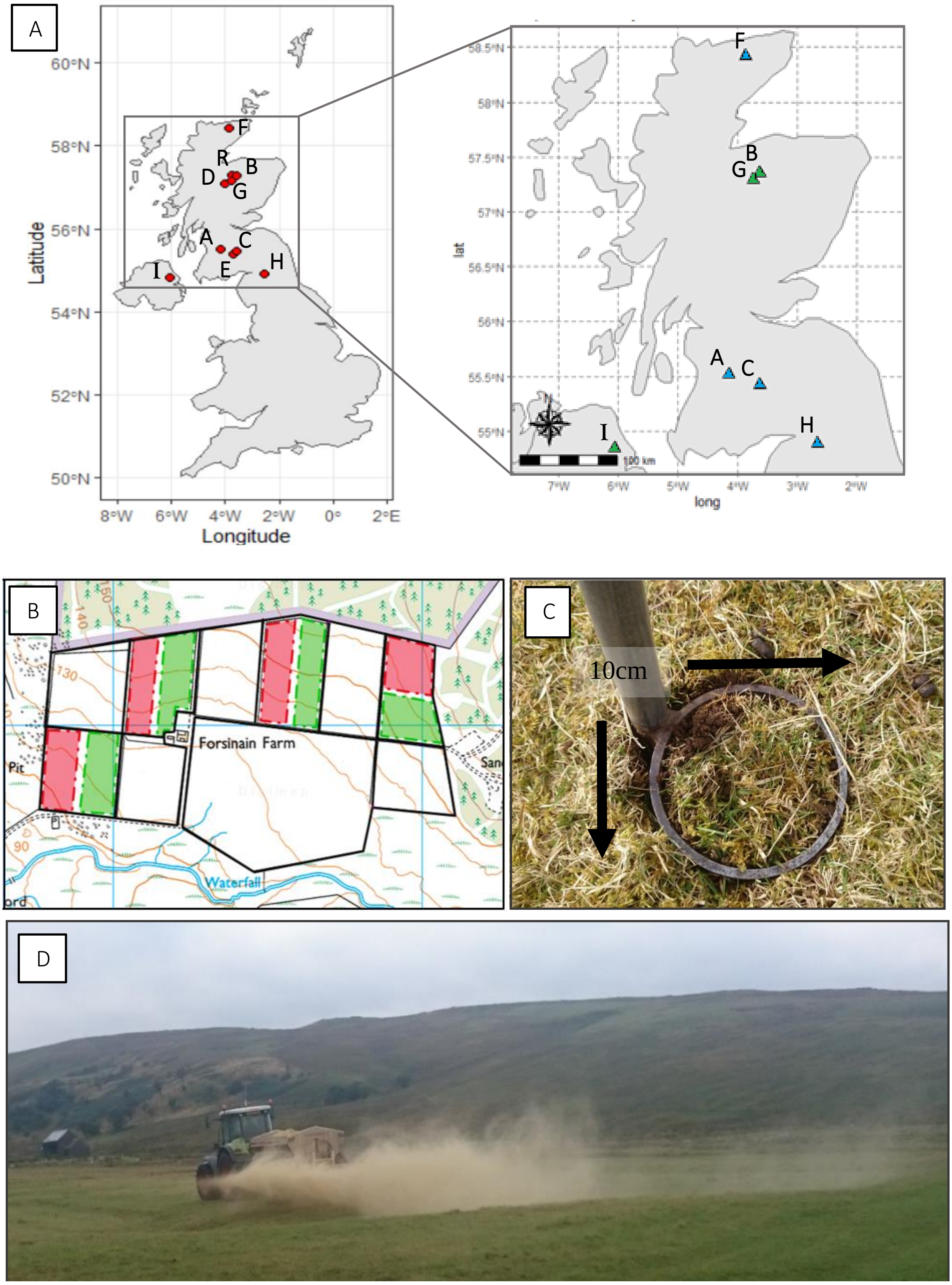


Figure 2.1. **(A)** Map of the United Kingdom (UK) showing location of the ten grassland farms (filled red circles) surveyed in the baseline year. Please note that the earthworm observation study was conducted on nine of the farms only (farm E excluded from formal analysis). Map insert shows the location of the seven farms involved in the BACI liming experiment (filled triangles) where blue = core farms surveyed in all four years; and green = farms surveyed for two or three years only (see Table 2.1). Farm codes shown in black text on the upper left of the symbol. **(B)** Ordnance Survey map of RSPB Forsinain Farm (farm F), showing the split plot design of the BACI liming experiment field trials with lime treatment (green) and control (red) plots with a buffer strip between on four of the farms in-bye grassland fields; **(C)** Photo of soil corer used to extract earthworms and for soil analysis; **(D)** Photo of lime being applied by a spreader (no till) onto a lime treatment plot at one of the liming experiment fields in autumn 2014 at RSPB Geltsdale Farm (farm H).

2.3.2 Experimental design of the BACI liming experiment

A split-plot before-after-control-impact (BACI) experimental design was implemented at the field scale with one half of each field (approx. 2ha) receiving an application of agricultural lime (treatment), and the other half receiving no lime (control). A non-limed buffer strip of 0.5 m was left between the two field halves to reduce contamination between plots (Figure 2.1C). Baseline soil analyses were carried out in 2014 prior to the application of any lime treatment. Soil samples were taken using a soil corer with diameter of 10.5cm to a depth of 10cm avoiding field boundaries and gates. Ten soil cores were taken from the control half and ten soil cores from the treatment half resulting in 20 soil cores across the 4ha. Each soil core was labelled and stored separately in a sealable plastic bag. I conducted soil analysis (pH and soil organic matter content) at the soil core level within the Soils Laboratory at Stirling University. However, to determine lime requirements for the field I followed methods recommended by SAC Consulting to ensure that the experiment of lime use would reflect that of typical agricultural practice. Therefore a mixed soil sample which consisted of a representative portion of each of the 20 soil cores from a field was mixed and sent to SAC Consulting for routine soil analysis (pH, Phosphate, Potash and Magnesium) for the seven agricultural farms in the study. The resulting SAC Consulting report on routine soil analysis at the field level were provided to the landowners for their own records with the pH results shown in brackets in Table 2.2 as a comparison to the mean pH calculated at the soil core level. Fields from RSPB reserves also had a field level pH value calculated from a mixed sample of all 20 soil cores but this was analysed at the Soils Laboratory at Stirling University due to expenses. To ensure that the pH results conducted in the Soil Laboratory from the RSPB fields were comparable to pH results conducted by SAC Consulting, pH was also measured for the agricultural fields in the Soils Laboratory with the methods used giving similar results (Table A 0-2 in Appendix). It is interesting to note that the mean pH value of fields calculated from the individual soil cores (as opposed to a mixed soil sample) differed from the pH values measured from a mixed soil sample. The mixed soil samples were on average 0.3 pH units higher than that of the mean. For consistency with real world advice that would be provided to landowners the mixed soil sample pH results were used to calculate lime requirements for each field; however it should be noted that in precision farming with the GPS of soil samples it would be possible to have very specific lime requirements targeted at specific areas of the field which would benefit greatest from pH amendment. The amount of lime needed depends on a variety of soil factors, including current soil pH, soil texture, and soil organic matter content, and also depends on the neutralizing value (NV) and particle size of the liming reagent applied (SRUC, 2014). Liming requirements (tonnes ha^{-1} of product with NV 50% CaO) therefore varied between fields and was provided by qualified agronomists at SAC Consulting based on the pH value calculated from the fields mixed soil sample and not from the mean pH of soil cores (Table 2.2). Please see Table A 0-3 in Appendix for table on lime requirements provided by SAC Consulting for more information. Landowners were responsible for the purchase of lime, hiring of contractors where applicable, and the spreading of lime on their own land outside the bird breeding season (autumn), although I was present at the time of spreading to ensure that only one field half was limed. The amount of lime applied in the study was lower than that recommended in some instances due to logistical issues, but in all instances, calcium carbonate (CaO) dust agricultural lime (calcitic) was applied as a surface dressing onto established grassland with no soil cultivation/tilling (Table A 0-4 in Appendix). The ability of lime to change the soil pH below the soil surface is very slow when it is applied as a surface dressing (approximately 18 months, pers comms with SAC Consulting Agronomists, 2014); however, the aim of the liming experiment was to test the effectiveness of a wildlife friendly and simple approach to liming rather than creating agriculturally optimum conditions, and to minimise disturbance to invertebrates as soil cultivation and tilling are known to be detrimental to earthworm populations (Briones and Schmidt, 2017). To prevent grass burning or lime being lost from heavy rainfall before it has the chance to react with the soil in a no till/no cultivation method, lime was applied in two stages where more than 4 tonnes per ha of lime was required (pers comms SAC Consulting Agronomist, 2014). Thus eight fields had their liming requirements applied over two consecutive years with 50% of the lime requirements applied in each year (i.e. one lime application in 2014, and a second lime application in 2015, Table 2.2). All other farming management such as livestock grazing or rush cutting was allowed to continue as usual across the entire field, allowing for direct comparison between the treatment and control plots. Fields were monitored annually following the application of lime which ranged between one to three years due to unforeseen circumstances (Table 2.1 & Table 2.2).

Table 2.2. Summary table and data structure for the BACI liming experiment across seven farms (n = 15 fields) with RSPB sites marked by *. Generalised soil type and soil texture collated from the James Hutton Institute's *Soil Information for Scottish Soil* open source database (SIFSS; http://sifss.hutton.ac.uk/SSKIB_Stats.php accessed 29th August 2019) and from the UK Soils Observatory open source database (http://www.ukso.org/ accessed 5th September 2019). Annual rainfall values shown are for the baseline year (2014) and were derived from the nearest weather station to each field from the Scottish Environment Protection Agency (SEPA; https://apps.sepa.org.uk/rainfall accessed 10th September 2019) open access *Rainfall Data for Scotland* and from the Met Office MIDAS Open: *UK Land Surface Stations Data 1853-current* (https://catalogue.ceda.ac.uk/uuid/dbd451271eb04662beade68da43546e1 accessed 25th August 2019). Mean field pH and soil organic matter (SOM) calculated from soil cores taken in the baseline year (raw data) with SAC consulting pH values shown in brackets where applicable. Lime requirements in tonnes ha^{-1} of liming product with a Neutralising Value (NV) of 50% calcium carbonate (CaO) as advised by SAC consulting. When >7 tonnes ha^{-1} lime required application of lime was conducted over two consecutive years.

Location	Farm code	Field code	Generalised soil type	Soil texture	Mean soil $pH_{(CaCl2)}$ ± SE	Mean SOM (%) ± SE	Total CaO applied (tonnes ha^{-1})	Number applications	Number years data collected	Annual rainfall (mm)
Scotland	A*	A01	Brown	Sandy loam to silty loam	4.9 ± 0.12 (5.1)	10.7 ± 2.08	8	2	4	1,455.8
		A02	Brown	Sandy loam to silty loam	4.9 ± 0.05 (5.2)	10.0 ± 0.77	8	2	4	1,455.8
		A03	Brown	Sandy loam to silty loam	5.0 ± 0.08 (5.4)	8.7 ± 0.59	8	2	4	1,455.8
		A04	Brown	Sandy loam to silty loam	4.9 ± 0.05 (5.1)	8.2 ± 0.35	8	2	4	1,455.8
	B	B05	Alluvial	Clay to sandy loam	4.8 ± 0.03 (5.3)	24.5 ± 1.21	4	1	2	838.7
	C	C06	Alluvial	Sandy loam to silty loam	5.0 ± 0.05 (5.4)	14.3 ± 0.54	4	1	4	2,142.4
		C07	Alluvial	Sandy loam to silty loam	4.8 ± 0.07 (5.0)	29.9 ± 2.65	4	1	4	2,142.4
	F*	F08	Peaty podzol	Sand to sandy loam	5.1 ± 0.04	32.8 ± 0.85	8	2	4	1,129.8
		F09	Peaty gleys	Sand to sandy loam	5.3 ± 0.04	32.3 ± 1.79	4	1	4	1,129.8
		F10	Peaty gleys	Sand to sandy loam	4.8 ± 0.04	33.6 ± 1.70	4	1	4	1,129.8
		F11	Peaty gleys	Sand to sandy loam	5.2 ± 0.04	43.1 ± 1.67	4	1	4	1,129.8
	G	G12	Alluvial	Clay to sandy loam	4.7 ± 0.04 (4.9)	41.4 ± 2.41	4	1	2	895.7
England	H*	H18	Peaty gleys	Clayey loam to silty loam	4.6 ± 0.07	14.1 ± 1.55	8	2	4	1,287.0
Ireland	I*	I16	Stagnosol	Clayey loam to silty loam	(5.3)	-	4	2	3	998.0
		I17	Stagnosol	Clayey loam to silty loam	(4.8)	-	4	2	3	998.0

2.3.3 Earthworms and soil sampling

Samples were taken using soil cores of 10 cm depth by 10.5 cm diameter (Figure 2.1D) as recommended by Laidlaw (2008) as a suitable sampling method for earthworms as food resources for waders. Samples were collected annually between March and April which coincides with periods of peak adult earthworm abundance (Gerard, 1967) and with the pre-breeding period and territory establishment of Lapwing (Galbraith, 1989). A total of ten soil cores were collected from each field half (n = 20 soil cores field^{-1}) from randomised sampling points generated using a sampling design tool in ArcGIS version 10.2.2 (Buja and Menza, 2013). Soil samples were collected from each farm in a single day in an effort to even out any variation introduced by weather or date at the field and farm scale. Soil cores were stored in sealed, and labelled plastic bags and processed in the Ecology laboratory at Stirling University. Data on earthworms and soil properties were recorded in the laboratory from the same field core, with the exception of soil moisture and soil penetrability which was calculated in the field within 10cm of the soil core location. Three measurements were taken with the soil moisture meter probe (HH2 moisture meter, SM200 moisture sensor, Delta-T Devices, Cambridge, England) and the mean value used in analysis. The ease of penetrability of the soil was determined by taken three recordings using a spring operated soil penetrometer which measures the soils resistance to vertical penetration (Davidson, 1965) and using the mean value in analysis. This consisted of a metal probe of 6.44mm in diameter pushed 3cm into the soil with an indicator scale measuring strength from 0 to 5.0 tons per square foot. This variable was chosen to give an indication of soil compaction which can affect earthworms, as well as provide an idea of the effort it would require to probe the soil with a beak. Both soil penetrability and soil moisture vary with weather conditions (i.e. rainfall) and therefore also capture climatic conditions which may influence earthworms. The heterogeneity of the ground surface or "bumpiness" (also referred to as ground micro-topography) was measured using a "bumpimeter" which is a piece of wood with six holes equally spaced drilled into it with two legs to place it against the ground. In this way it was possible to record distance in cm from a fixed point to the ground, and the standard deviation of the mean used as an index for how flat/bumpy the ground was.

Soil cores were hand-sorted to extract earthworms (Edwards, Hendrix and Arancon, 1996; Laidlaw, 2008) with only earthworm sections which contained the head counted to prevent double counting of broken parts from the same individual (see Barnard and Thompson (1985) for discussion). Soil cores were processed within 14 days of collection and stored within a fridge to preserve earthworm specimens before being sorted. Adult earthworms were identified to species level and the functional

Figure 2.2. Photograph of an earthworm showing the saddle (also known as clitellum) of an adult earthworm which is a key characteristic in identification to species level. The clitellum is used in reproduction and forms a cocoon which protects the earthworm egg.

ecological group recorded (Sims and Gerard, 1985; Sherlock, 2018). Where required, rare specimens were verified from experts at the Natural History Museum, London.

Identification of earthworms requires mature characteristics such as the clitellum (also known as the saddle, Figure 2.2) which is part of the reproductive system, and therefore identification of immature and juvenile earthworms was not possible. However, all earthworms in the Lumbricus genus have a characteristic head structure known as a tanylobic head in which the first segment of the earthworm's head looks like it is split into three parts due to the structure of the prostomium which covers the mouth of the earthworm (Figure 2.3). Therefore it was possible to discriminate juvenile and immature stage earthworms without a fully developed clitellum as an earthworm in the Lumbricus genus or not in the Lumbricus genus based on their head structure. All Lumbricus earthworm species are epigeic with the exception of *Lumbricus friendi* which is a rare and anecic earthworm species. In analysis of earthworm ecological groups, juvenile and immature stage earthworms with a tanylobic head were therefore included in the epigeic group. In addition, *Allolobophora chlorotica* produces a lot of yellow coelomic fluid when handled and also have a distinct yellow "neck" band towards the head end which is often present in immature/juvenile worms (Figure 2.4). This species is also found in two colour morphs: pale pink and pale green, with the green morph having a preference for slightly wetter soil conditions than the pink morph (Sims and Gerard, 1985; Sherlock, 2018). Although there is some evidence to suggest that the colour of the green morph is usually not visible until after sexual maturity and the development of the clitellum at about a year old (Satchell, 1967) in some instances the green colour is visible in young worms older than 6 months and greater than 2cm in size (Kalmus, Satchell and Bowen, 1955). In cases where juvenile and immature earthworms were a green colour and produced yellow fluid they were recorded as the green morph *A.chlorotica* even though the classical characteristics used in earthworm keys (i.e. the saddle) were not visible.

Soil pH was recorded from the same soil core as the extracted earthworms. Soil was air dried for a minimum of two weeks and milled through a 2mm sieve prior to analysis. A subsample of 10g was used for chemical analysis with measurements of pH taken using a digital pH meter (pH209, Hanna Instruments, Woonsocket, Rhode Island, USA), calibrated using buffers pH 7 and pH 4. Measurements of pH were recorded using a 1:5 soil-to-water ratio with 25ml of distilled and de-ionised water (H_2O). Although this is an established technique, measurements of pH in water are subject to a number of uncertainties which are related to the effects of changes in the ionic strength of the soil-water suspension on ions sorbed to the soil particles. One of the main consequences of this is the "dilution effect", whereby different pH values are measured depending on the soil:water ratio in the suspension from the same sample. Therefore, a second pH measurement was recorded from the same soil sample following the addition of 1ml of calcium chloride ($CaCl_2$) to the 25ml of distilled H_2O creating a 0.01M $CaCl_2$ solution. Measurement of pH in $CaCl_2$ yield more consistent and repeatable results under laboratory conditions; although the pH value recorded in $CaCl_2$ is typically lower than that measured in water (White, 1969). The 0.01M $CaCl_2$ solution is less affected by changes in the ionic concentration of the soil solution and simulates the soil solution found in the field better than pure water as it has approximately the same ionic strength as the soil solution in agricultural temperate soils and has been recommended as a more appropriate medium for soil pH measurement (Schofield and Taylor, 1955;

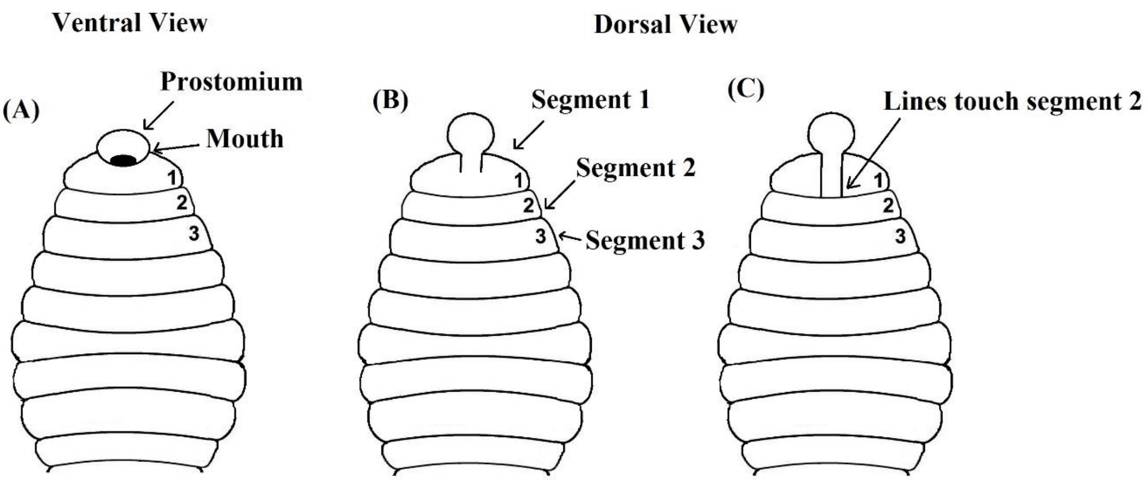


Figure 2.3. Cartoon image of an earthworm head. Image shows **(A)** ventral view of the prostomium and mouth hole on the first segment of an earthworm; and **(B)** and **(C)** the two characteristics that the prostomium "hinge" can take when viewed dorsally under a light microscope which are classified as **(B)** epilobic (crease lines of the prostomium do not touch the second earthworm segment) and **(C)** tanylobic (both crease lines touch the second earthworm segment).

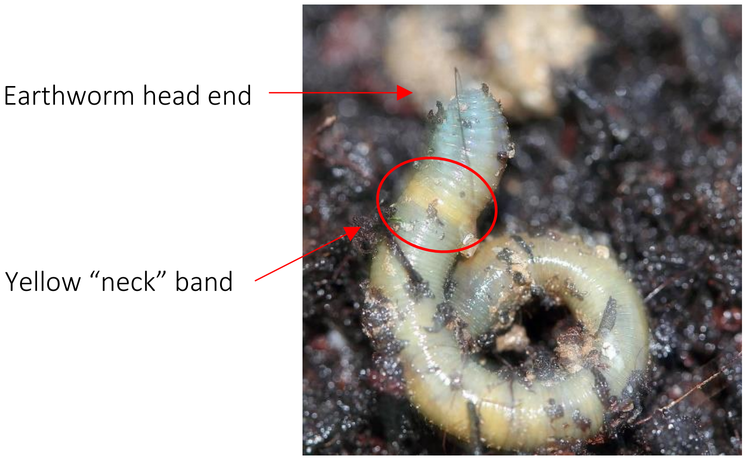


Figure 2.4. Image of the green morph of *Allolobophora chlorotica* showing the distinct green colouration and the characteristic yellow “neck” band towards the head end. Image by Holger Casselmann and adapted from: https://commons.wikimedia.org/wiki/File:Allolobophora_chlorotica_HC2.jpg

Blake *et al.*, 1999). Furthermore, the methods followed by SAC Consulting measure soil pH in a 0.01M $CaCl_2$ solution resulting in consistency between the soil samples analysed in the University of Stirling, with that of the baseline data recorded by SAC consulting and values provided to landowners. All data analysis in this study therefore used the pH value measured in the 0.01M $CaCl_2$ solution.

Soil organic matter content was derived from drying 10g of the sieved and air-dried soil for 4 hours at 105°C, before burning at 425°C overnight. The soil sample was weighed before and after to calculate the soil organic matter content as a percentage (i.e. the loss of organic matter on ignition known as LOI).

2.3.4 Environmental data

Environmental data such as soil type and soil texture were collated from the James Hutton Institute’s *Soil Information for Scottish Soil* open database (SIFSS; http://sifss.hutton.ac.uk/SSKIB_Stats.php accessed 29th August 2019) and from the UK Soils Observatory open database (http://www.ukso.org/ accessed 5th September 2019) respectively (see Table 2.2 & Table 2.3). Weather variables including soil temperature at 10cm depth, and rainfall (mm) were obtained at the annual, seasonal and monthly scale from the nearest weather station to each of the farms from the Scottish Environment Protection Agency (SEPA; https://apps.sepa.org.uk/rainfall accessed 10th September 2019) open access *Rainfall Data for Scotland* and from the Met Office MIDAS Open: *UK Land Surface Stations Data 1853-current* (https://catalogue.ceda.ac.uk/uuid/dbd451271eb04662beade68da43546e1 accessed 25th August 2019). Time series plots of weather data were used in the exploratory data phase to identify if there were any obvious trends or extreme weather events which may have influenced the survey years. Trends in soil type and soil texture were plotted against response variables in the exploratory phase but were highly correlated with farm (.e.g. Brown soil type was only for farm A; Stagnosol soil type only for farm J, see Table 2.3). Therefore it was felt that farm captured the variation observed in soil type and texture so these two variables were not included in model building.

Table 2.3. Summary table of (A) response, and (B) predictor variables used to investigate the relationship of earthworm abundance with soil properties, and to test the effect of lime treatment on soil pH and earthworm abundance.

Variable	Unit of Replication	Method of data collection
(A)		
Earthworm count	Soil core	Sorted by hand (soil core range: 0 – 40; soil core mean: 6.5)
Soil pH ($CaCl_2$)	Soil core	Measured in the laboratory using 0.01M $CaCl_2$ (soil core range: pH 3.4 - 6.6; soil core mean: pH 4.9)
(B)		
Soil organic matter (%)	Soil core	Measured in the laboratory from loss of ignition (LOI; soil core range: 2.5 – 93.0%; soil core mean: 25.2%)
Soil moisture (%)	Soil core	Recorded in the field using soil moisture probe (soil core range: 15.4 – 90.0%; soil core mean: 66.0%)
Ground bumpiness (i.e. ground micro-topography)	Soil core	Measured using a "bumpimeter" with six values recorded of the distance from a fixed point to the ground, and the standard deviation of the mean used as an index for how flat/bumpy the ground was (soil core range: 0.0 – 3.0, mean: 0.8)
Soil penetrability	Soil core	Measured using a spring operated soil penetrometer with three values recorded and using the mean value in analysis. The metal probe was pushed 3cm into the soil with an indicator scale measuring strength from 0 to 5.0 tons per square foot (soil core range: 1.1 – 5.4, mean: 3.4)
Elevation (m)	Farm	Extracted from Ordnance Survey Digital Terrain map using 50-m grid (site range: 188.0 – 303.0m; site mean: 230.6m)
Soil Type	Field	Extracted from The James Hutton Institute Soil Information For Scottish Soils (SIFSS) – Compsoil 1:250,000 soil polygon layer (5 categories: Brown; Alluvial; Peaty Podzol; Peaty Gley; Stagnosol)
Soil Texture	Field	Extracted from the UK Soils Observatory - Soil Texture (simple) 1:50K soil polygon layer (4 categories: Sandy loam to silty loam; Clayey loam to silty loam; Sand to sandy loam; Clay to sandy loam)
Farm	Farm	Recorded from the farm site that sampling occurred (10 categories: Site A to Site J)
Year	Year	Recorded from the survey year that sampling occurred with the baseline year before any lime was applied as Year 0 (4 categories: Year 0; Year 1; Year 2; Year 3)
Treatment	Field half	Recorded from the liming experiment field trials (2 categories: control; treatment)

2.3.5 Statistical analysis

2.3.5.1 Observational study: Earthworm community summary descriptions

Several summary descriptors and indices were used to provide an overview of the earthworm community in the baseline year of the study across the nine farms (n= 18 fields) where earthworms were sampled. These comprised the total number of individual earthworms recorded (abundance), earthworm species richness and Simpson's Diversity Index (D) and Simpson's measure of evenness (E) of the community (Magurran, 2004) calculated at the farm scale.

In order to graphically represent and compare earthworm species composition between farms (communities) at the field scale a *non-metric multidimensional scaling* (NMDS) approach was taken. Unlike other ordination techniques, such as Principal Coordinates Analysis, NMDS uses rank orders instead of Euclidean distances which are sensitive to total abundances, and was fitted using a Bray-Curtis dissimilarity calculation. A Wisconsin double standardization was applied to square root transformed data in order to reduce influence of large counts (Bray and Curtis, 1957). The acceptability of the NMDS was checked graphically using a Shepard diagram to ensure that there was linearity between the ordination distance and the Bray-Curtis distance, and by its Stress value. The Stress value is a "goodness of fit" measures which provides a measure of the degree to which the distance between samples in reduced dimensional space corresponds with the actual multivariate distance between the samples. Lower Stress values indicate greater conformity and therefore are desirable. A general rule of thumb is that Stress values < 0.2 are acceptable for biological interpretation of the ordination (Kruska, 1964; Clarke, 1993; Dexter, Rollwagen-Bollens and Bollens, 2018); however, this will be considered alongside other complementary criteria to evaluate the ordination fit such as the Shepherd plot mentioned above as the Stress value can be influenced by the number of samples, species and dimensions in the ordination (Clarke, 1993; Vanderklift, Ward and Jacoby, 1996; Dexter, Rollwagen-Bollens and Bollens, 2018).

2.3.5.2 Observational study: The relationship between soil properties and earthworms

A generalised linear mixed effects model (GLMM) was used to test the relationship between earthworm abundance and soil properties (hypothesis 1). Data for this model was from the first year of the project only (2014), prior to the application of lime treatment, resulting in a dataset with 360 observations (soil cores) from nine farm sites (n = 18 fields, Table 2.1). Total earthworm count per soil core was fitted as the response variable, with soil pH (continuous), organic matter content (continuous), soil moisture (continuous) and soil penetrability (continuous) as the potential explanatory covariates. Quadratic terms of soil properties were also included as it was expected that earthworms would not favour extremes (Edwards, Hendrix and Arancon, 1996; Curry, 2004). Field nested within Farm was fitted as the random explanatory factor to account for any dependency among observations of the same field and farm site. In order to determine if there were any differences in response to soil variables between earthworms species, separate models were fitted following the same procedure as above but with the earthworm ecological grouping (abundance) as the response variable. However, it was not possible to fit a model with the ecological grouping of anecic earthworms or at the species level due to too few observations.

2.3.5.3 Liming experiment: Earthworm summary community descriptions

To provide an overview of the earthworm community over the four years of the BACI liming experiment several summary descriptors were used, including the total number of individual earthworms recorded (abundance), earthworm species richness and earthworm species abundance. These were summarised at the year level.

2.3.5.4 Liming experiment: The relationship between lime treatment and soil properties

To determine whether liming via a wildlife sympathetic management of surface dressing with no till had the agriculturally desired impact on soil pH (hypothesis 2), the relationship between soil pH and whether or not lime had been applied was tested using a linear mixed effects model (LMM). The model included data from all four years of the liming experiment and included a baseline year (2014) which was prior to the application of lime treatment, followed by three consecutive years of annual monitoring (2015, 2016, 2017). Due to unforeseen circumstances, lime treatment was only applied to seven out of the ten farms originally surveyed in the baseline year and therefore the data excluded sites and fields which had not received any lime. In 2014 and 2015 this consisted of seven farm sites (n=15 fields), but was reduced in the final two years of field trials due to logistical constraints to six farm sites (n=13 fields) in 2016 and four farm sites (n=11 fields) in 2017 resulting in a total of 1,080 soil cores collected (Table 2.1 & Table 2.2). Although a total of 1,080 soil cores were taken over the four years, formal analysis was only conducted on 980 soil cores due to 100 cores not having full data on soil properties. This was caused by a technical issue with the penetrometer and soil moisture probe at one field (n = 20 soil cores), and by sampling differences conducted at the two limed fields in Northern Ireland in the baseline year and in the first year post lime application (n = 80 soil cores). Data collected from Northern Ireland in these two years was available for earthworm abundance at the soil core level, however, the soil samples were mixed at the field plot level post extraction of earthworms. Soil analysis was therefore at the plot level and was therefore not useable for these two years in formal analysis, although full data at the soil core level was available in Y2. Soil pH was used as the response variable with the unit of replication at the soil core level. Field nested within Farm was included as a random effect (grouping) within the model, and Year included as a fixed effect as well as an interaction term between Year and Treatment. The soil properties of soil moisture (%) and soil organic matter (%) were also fitted as continuous covariates as well as ground bumpiness, as it was thought that the bumpiness of the ground may create micro-habitats that could influence liming. To test if there was an effect of lime treatment or year on the other soil properties (moisture, organic matter and penetrability) and ground bumpiness, each was fitted as the response in a separate model with random effect of field nested within year, fixed effects of Treatment and Year as well as the interaction effect between Treatment and Year, and soil properties as covariates. The response variables of soil penetrability and ground bumpiness were log transformed, while moisture and organic matter were fitted as generalised models.

An additional LMM was run comparing field halves which had had two lime applications (i.e. had been subjected to two consecutive years of liming as the lime requirements were greater than the recommended maximum amount permitted in one dressing) to those that had received one lime application. The model was fitted with soil pH as the response variable and the following explanatory predictors as fixed effects: soil moisture (continuous), soil organic matter (continuous), number of lime applications (binary category of 1 or 2). A random effect (grouping) of Farm with Field nested was fitted to account for repeated measures, and a random effect of Year. This model only included data from soil cores which had received at least one application of lime and therefore data from the baseline year and from control were excluded resulting in a total of 370 soil cores from six farms (n = 13 fields) sampled in Year 1, six farms (n=13 fields) sampled in Year 2, and four farms (n = 11 fields) sampled in Year 3.

2.3.5.5 Liming experiment: The relationship between lime treatment and earthworm abundance

A GLMM was used to test the relationship between earthworm abundance and lime treatment, while controlling for known soil properties which influence earthworm abundance (hypothesis 3). Data for this model was from seven farm sites (n=15 fields) over four years and included a baseline year (2014) which was prior to the application of lime treatment, followed by three consecutive years of annual monitoring (2015, 2016, 2017) resulting in a dataset with 1,080 observations (Table 2.1 & Table 2.2). However, formal analysis was only conducted on 980 soil cores as described above due to soil properties not being available for 100 of the soil cores. Total earthworm count (abundance) per soil core was fitted as the response variable, with the soil properties of organic matter content (continuous) and soil moisture (continuous) as the potential explanatory covariates as well as the quadratic term for each of these. Soil pH was not included as a covariate due to its relationship with lime treatment (see Results). Field nested within Farm was fitted as the random explanatory factor to control for repeat visits to sites, and a main effect of Year (categorical: four levels) and Treatment (categorical: two levels) was fitted as fixed factors as well as the interaction term for Year x Treatment. The critical term in the model is the Year x Treatment interaction which measures whether earthworm abundance per soil core differs between Years which have subsequently been influenced by lime application (Treatment) and those that have not had any lime applications (Control).

In order to determine if there were differences in response to lime treatment between earthworms species, separate models were fitted following the same procedure as above but with the earthworm ecological grouping (abundance) as the response variable. However, it was not possible to fit a model with the ecological grouping of anecic earthworms, or separate models at the species level due to too few observations.

2.3.5.6 Preliminary analysis, model selection, validation and post-hoc tests

All statistical analyses were performed with R version 3.6.2 (R Development Core Team, 2019). All (G)LMMs were conducted using the lme4 package (Bates *et al.*, 2015) unless otherwise stated. Covariates were standardised by centring and then scaled prior to analysis to ease interpretation of model coefficients (Schielzeth, 2010). Explanatory variables were checked for collinearity (see Figure A2 in Appendix) prior to analysis with correlation < 0.7 considered acceptable (Dormann *et al.*, 2013). Collinearity was an issue between farm and the soil characteristics of soil type and soil texture, and therefore farm was used in models only as this variable was required as a random effect (Figure A3 and Figure A2 in Appendix).

All models where the response variable was a count were initially fit with a Poisson error structure and log link; however, when count models were over-dispersed (dispersion parameter ≥ 1.4; Payne *et al.*, 2018) additional models were explored including Negative Binomial models with a log link to combat overdispersion of Poisson errors (Thomas, 2017), as well as models with an observation level random effect (OLRE) and Poisson error structure (also known as a log-normal Poisson model) to account for this over-dispersion. In cases where models were zero-inflated these were fitted as zero-inflated negative binomial models with a log link and with a single zero-inflation parameter applying to all observations using the glmmTMB package (Brooks *et al.*, 2017). Where the response variable was a percentage (e.g. soil moisture) the model was built applying a beta regression with a logit link in the glmmTMB package because continuous proportions are more suited to a beta regression than a binomial regression model, which is more appropriate for proportions derived from counts (Douma and Weedon, 2019). Where the response variable was a proportion (e.g. soil organic matter) a binomial model was fitted using the before burning weight and the post burning weight. The minimal adequate models (MAMs) were obtained using a manual stepwise backwards selection tested by likelihood ratio tests (LRT) of nested models, retaining all variables that were significant at the 5% level and fixed factors that accounted for the study design. Model fit was assessed by calculating pseudo R^2 which provides a measure (%) of the response variable variation that is explained by the model, and was calculated using the rsquared() function in the piecewiseSEM package (Zuur et al., 2009; Nakagawa and Schielzeth, 2013; Leftcheck, 2016). The pseudo R^2 is more appropriate for mixed effects models than the standard R^2 and provides two types of R^2: the marginal and the conditional. The marginal R^2 provides a measure of the variance explained by the fixed effects only, while the conditional R^2 provides a measure of the variance explained by both the fixed and random effects (i.e. the variance explained by the whole model; Nakagawa and Schielzeth, 2013; Nakagawa *et al.,* 2017). Values are bounded by 0 and 100 where 0% indicates a model that explains none of the response data around its mean, and 100% indicates a model that explains all the variability of the response data around its mean. In general, a higher R^2 indicates a model that better fits the data and therefore has smaller error in the predictions (i.e. more precise predictions with smaller prediction intervals). Thus R^2 does not affect interpretation of the relationship of the coefficients, but represents confidence in the precision of the model predications. It is therefore difficult to provide a "rule of thumb" threshold regarding what is an acceptable R^2 value as it will depend on the objectives of the model (e.g. if it is wanted for predicting response or for interpreting relationships between the response with variables) and on the "true" variability in the response such as ecology or behavioural studies which have natural levels of high variability and therefore low R^2 values are commonly reported of less than 50% (Fuentes-Montemayor, Goulson and Park, 2011; Isaksson *et al.*, 2016; McCallum *et al.*, 2016, 2018; Spake *et al.*, 2016; Kelly *et al.*, 2021). The R^2 is therefore a helpful index for comparing competing models in the model selection phase, and in determining how much variation in the response is explained by the fixed effects (marginal R^2), but will be used in conjunction with other model testing to ensure that the model is suitable. Model residuals were checked graphically for normality and homogeneity of variance (Zuur, Ieno and Smith, 2007) using the DHARMa package (Hartig, 2020) for GLMMs, and compared by Akaike information criterion (AIC) to determine goodness of fit, where models with >2 AIC units lower were considered the significantly better model.

Post hoc tests were conducted to calculate the estimated marginal means (EMMs) for each level of the categorical variables in (G)LMMs using the emmeans package (Lenth, 2020) with EMMs computed by averaging together predictions on the linear-predictor scale and then back transforming to the response scale. A pairwise turkey method was used for comparing EMMs to test if these were statistically significantly different.

2.4 RESULTS

2.4.1 Observational study: Earthworm summary community descriptions

A total abundance of 1,484 earthworms were sampled across the nine farms (n = 18 fields), comprising of 14 species, although the majority of records (c.90%) were of juvenile/immature earthworms (Table 2.4). Earthworm species richness ranged between three to eight species identified per field with no two farms sharing the same species assemblage. Fields from the same farm were more similar to each other in terms of their species community composition than fields from other farms in terms of their location within multidimensional space visualised graphically as colours in the biplot (Figure 2.5). Likewise there was a large amount of overlap in terms of the similarity of species composition by geographical location of fields within North Scotland, West Scotland and East Scotland, compared with fields from Northern England or Northern Ireland visualised in the biplot by shape, with grey hulls superimposed to aid observations of the relationship among samples in ordination space.

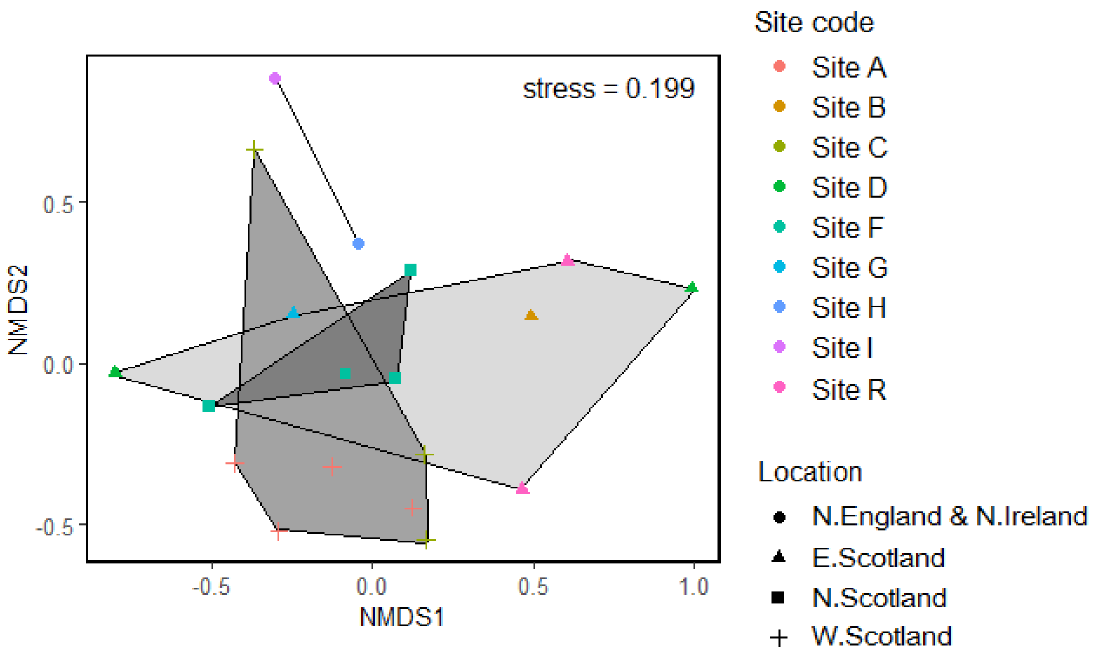


Figure 2.5. NMDS biplot of earthworm species composition at the field scale, where colour represents the farm site; and shape represents the geographic location within the United Kingdom. Each point represents a community at the field level. Where points are closer together, they are more similar in their species community composition.

Table 2.4. Summary table of the total number of individual earthworms recorded (abundance), earthworm species richness and Simpson's Diversity Index (D) and Simpson's measure of evenness of the community (E) for each field. Sample size was nine farms (n = 18 fields; 20 cored per field) in the baseline year (2014) resulting in 360 observations (soil cores).

Location	Farm Code	Field identity	Abundance	Species richness	Simpson's D	Simpson's E
North Scotland	F	F8	87	5	0.56	0.25
		F9	131	6	0.55	0.29
		F10	67	7	0.57	0.35
		F11	100	6	0.56	0.29
West Scotland	A	A1	117	8	0.59	0.23
		A2	57	6	0.61	0.31
		A3	77	6	0.56	0.29
		A4	261	7	0.56	0.25
	C	C13	40	5	0.27	0.26
		C6	128	9	0.60	0.20
		C7	109	6	0.56	0.29
East Scotland	B	B5	58	6	0.58	0.30
	D	D14	20	6	0.65	0.32
		D15	23	5	0.37	0.29
	G	G12	62	7	0.64	0.27
	R	R17	36	5	0.25	0.26
		R19	27	5	0.38	0.29
Northern England	H	H18	70	8	0.67	0.24
Northern Ireland	I	I16	14	3	0.44	0.52
Totals			1,484	14		

2.4.2 Observational study: The relationship between soil properties and earthworms

There was strong evidence of a positive relationship of soil pH with total earthworm abundance, and strong evidence of a decline in total earthworm abundance with increasing soil organic matter content, but there was no relationship with soil moisture (Table 2.5, Table 2.6, Figure 2.6). After controlling for other variables, earthworm abundance at the soil core level showed a linear and positive relationship with soil pH. Displaying the relationships of a single variable in a multi-variate model is difficult to achieve, and in the case of Figure 2.6, this linear relationship is not obvious at first glance. Quadratic relationships between earthworm abundance and soil properties were explored in the model building phase and were non-significant. The apparent curvilinear response in Figure 2.6 can be explained by the site level effect (random effect). The random effect of site explained a high proportion of the variance within the response (earthworm abundance). Using site as a random effect in the model allows for individual intercept points of the model regression, reflecting the relationship that earthworm abundance was correlated at the site level. The relationship between earthworm abundance and soil properties is linear at the site level but this is masked when data from all sites is shown and only the population level response is displayed as in Figure 2.6. Please see Figure A4 in Appendix for plots of earthworm response at the site level which shows the linear relationship more clearly. In addition, please see Figure A6 in Appendix for plot of earthworm abundance against pH as H^+ concentration rather than at the pH scale. After controlling for other variables, earthworm abundance was estimated at 1.9 (95% CI: 1.15 – 3.25) per soil core in acidic soils of pH 4.0, but this increased with increasing soil pH (more neutral soil) with over 7 earthworms predicted under pH values of 6.00 (estimated marginal mean of 7.15 earthworms per soil core with 95% CI: 3.95 – 12.96). For soil organic matter content, the estimated total earthworm abundance was predicted to be, on average, 3.56 (95% CI: 2.40 – 5.28) per soil core at values of 20% soil organic matter content. However, earthworm abundance was predicted to more than halve in more peaty soil of 80% organic matter content to 1.11 (95% CI: 0.48 – 2.56) earthworms per soil core. The pseudo R^2 values for this model were 0.105 for the marginal R^2 (R^2_M) and 0.484 for the conditional R^2 (R^2_C).

Epigeic earthworm species exhibited strong evidence of an increase in abundance with soil pH, but there was no relationship with soil moisture nor soil organic matter (Table 2.6, Figure 2.6). The estimated epigeic earthworm abundance was predicted to be, on average, 0.30 (95% CI: 0.14 – 0.87) per soil core with acidic soil pH of 4.0; however this was predicted to increase to 3.00 (95% CI: 1.15 – 7.87) epigeic earthworms per soil core in more neutral soil conditions of pH 6.0, a tenfold increase. The model does not explain variation in epigeic earthworm abundance well when only the fixed effects are considered (R^2_M = 0.051) but variation explained is greater when the full model (fixed effects taking into account random effect) is considered (R^2_C = 0.824).

Endogeic earthworm abundance was not related to soil pH or soil moisture but showed very weak evidence of a negative effect of increasing soil organic matter (Table 2.6, Figure 2.6). The model explains a low proportion of variation in endogeic earthworm abundance as seen by the low R^2 values for this model (R^2_M = 0.002; R^2_C = 0.003). Although the sample size of soil cores were high, the number of observations of endogeic earthworm species were low reducing the power to detect effects.

Table 2.5. Summary table of the final (G)LMMs fitted to test the association between earthworm abundance and soil properties. Note that SOM is used to abbreviate "soil organic matter" in the table.

Response variable	Type	Error structure	Link	Random effects	Fixed effects
Total earthworm abundance	Over dispersed count	Negative binomial	Log	Field Identity	pH + SOM
Epigeic earthworms	Over dispersed count	Negative binomial	Log	Field Identity	pH
Endogeic earthworms	Over dispersed zero inflated (ZI) count	ZI negative binomial	Log	Field Identity	SOM

Table 2.6. Summary table showing the minimal adequate GLMMs investigating the relationship between soil properties and earthworms at the soil core scale using data from across nine farms (n = 18 fields) from the baseline survey (2014) resulting in 360 observations. Model 1 tests soil properties on total earthworm abundance; Model 2 tests soil properties on epigeic earthworm abundance; Model 3 tests soil properties on endogeic earthworm abundance. Note that covariates were scaled and centred prior to analysis for ease of coefficient interpretation. Table shows results of the species minimum adequate models with significance levels denoted by *'s where: * ≤ 0.05; ** ≤ 0.01; *** ≤ 0.001.

Predictor	Type (df)	Parameter estimates ± SE (test statistic)		
		Model 1 – response: Total earthworms	Model 2 – response: Epigeic earthworms	Model 3 – response: Endogeic earthworms
Soil pH	Covariate (df =1)	0.28 ± 0.08 (z = 3.28**)	0.46 ± 0.10 (z = 4.73***)	-
SOM (%)	Covariate (df =1)	-0.30 ± 0.10 (z = -3.02**)	-	-0.27 ± 0.14 (z = -1.85)

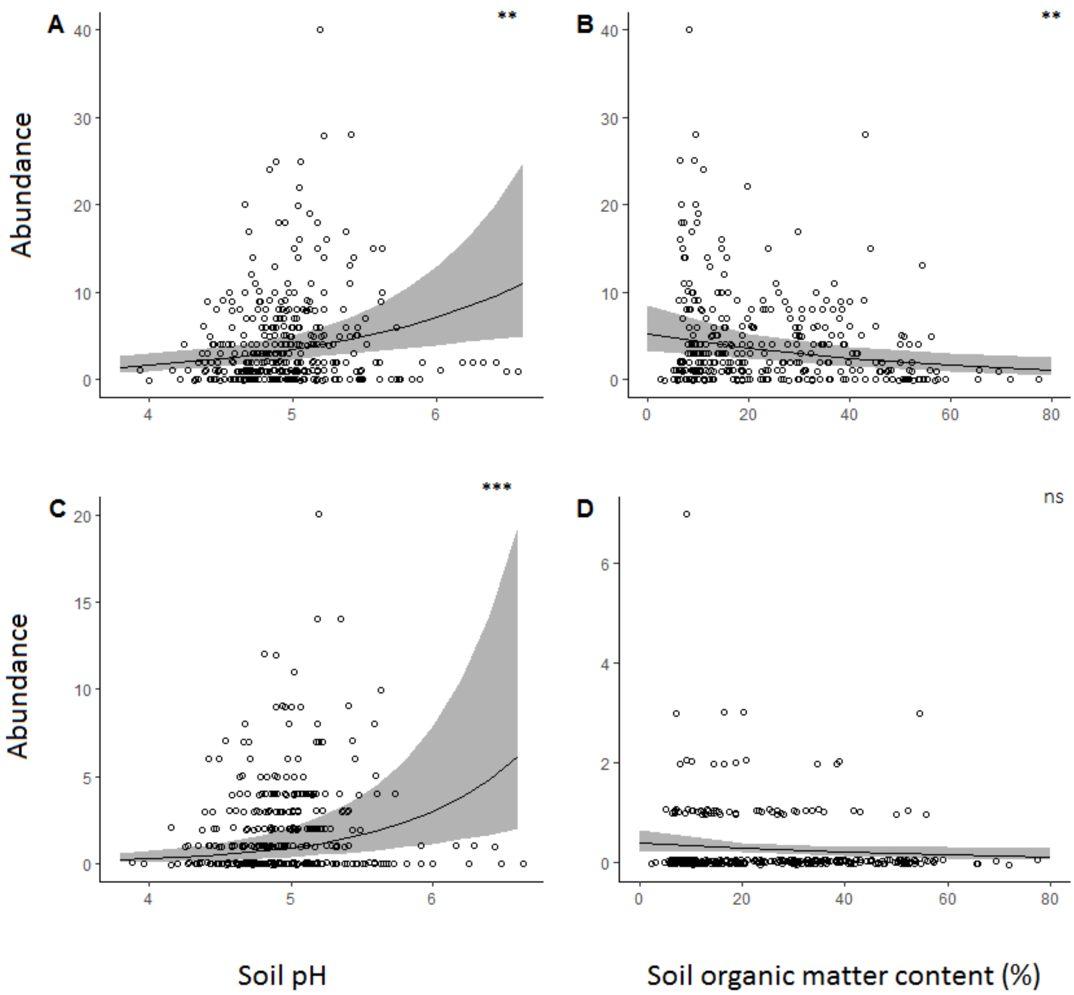


Figure 2.6. The relationship between earthworm abundance per soil core and soil properties. **A)** Total earthworm abundance against soil pH; **B)** total earthworm abundance against soil organic matter content %; **C)** epigeic earthworm abundance against soil pH; **D)** endogeic earthworm abundance against soil organic matter content %. Sample size is nine farms (n = 18 fields) from the baseline survey year (2014) resulting in 360 observations. Raw data is represented by open black circles, while model predictions shown as a solid black line with 95% confidence intervals shaded in grey. Please note that the y-axis of the plots are on different scales. Where the model showed that the coefficient estimates of the predictor was significant this is denoted by *'s where: ns > 0.05; * ≤ 0.05; ** ≤ 0.01; *** ≤ 0.001.

2.4.3 Liming experiment: The relationship between lime treatment and soil properties

Between individual soil cores (n = 980), soil pH ranged from highly acidic pH values of 3.4 to more neutral values of pH 6.6, with a median pH of 4.9 (see Figure A5 in Appendix), while soil organic matter ranged from 3.1% to 93.0% with a median value of 24.3%. On average, mean field soil pH was lower in the baseline sampling year than in the final year of sampling (Table A 0-6).

There was strong evidence of an interaction effect between Year and Treatment with soil pH, as well as strong evidence of an inverse relationship of soil pH with soil organic matter content (Table 2.7, Table 2.8, Figure 2.7). On average, soil pH in control and treatment plots were not significantly different in the baseline year of the study before any lime treatment was applied; however, there was significant difference in soil pH between control and treatment plots within all years post the application of lime with the lime treatment plot predicted to have a higher (more neutral) soil pH value (Table 2.9). There were also significant differences estimated for soil pH between consecutive years. On the lime plot, this was non-significant between the baseline year and the first year post lime application, but was significant between the second year after lime was applied and the third year after lime was applied with soil pH significantly more neutral in the final year of monitoring. On the control plot, soil pH was significantly different between each year which was estimated to decline (become more acidic) with each consecutive year, until the final year when soil pH was estimated to increase (become more neutral). Between the baseline year and the final year of the survey, soil pH was estimated to be significantly different with higher soil pH values in the final year compared with the baseline on both the control and lime treatment plots. For the relationship with soil organic matter content and pH, the model predicted soil with a mean soil organic matter content of 26% to have a soil pH of 4.89 with soil pH declining (more acidic) with increasingly peaty (high organic matter) soil. The model explained variation in soil pH well (R^2_M = 0.354; R^2_C = 0.558).

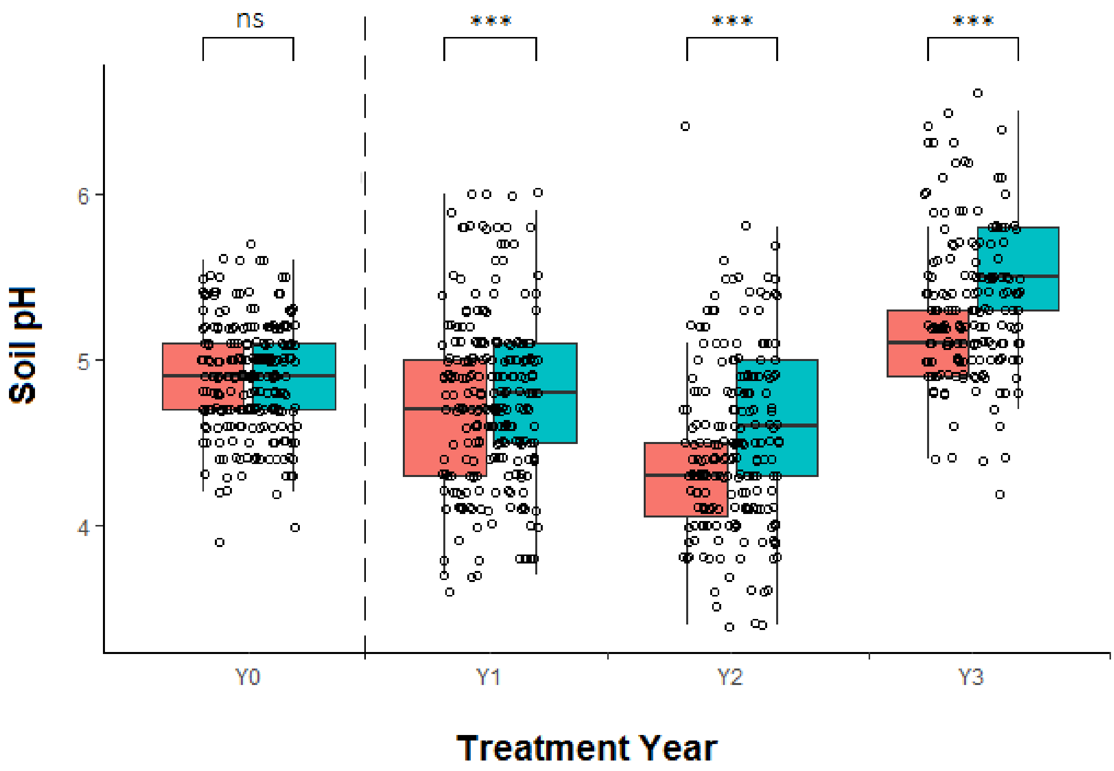


Figure 2.7. Boxplot graphically displaying the significant interaction effect between treatment (red = control; blue = lime treatment) and year (Y0 = baseline year (spring) before any lime treatment applied in the autumn; Y1 = first spring following the application of lime onto the lime treatment plots; Y2 = second spring following the application of lime onto the lime treatment plots; Y3 = third spring following the application of lime onto the lime treatment plots) on soil pH. Sample size is seven farms (n = 15 fields) and four years of data. Formal analysis was conducted on 980 observations. Raw data is represented by open black circles, while model predictions (conditional taking into account random effects) from the LMM are shown as a boxplot displaying the interquartile range (IQR), medium as a black line and whiskers representing values that fall no further than 1.5 times the IQR. Dashed black line represents the application of lime treatment onto the lime treatment plots. Significance of pairwise comparisons of the estimated marginal means (Table 2.13) within years denoted by *'s where: ns > 0.05; * ≤ 0.05; ** ≤ 0.01; *** ≤ 0.001.

There was no evidence for an effect of liming on any of the other soil properties (moisture, organic matter, penetrability) or on ground bumpiness. However, there was a main effect of year on soil moisture, organic matter and penetrability (Table 2.8). On average, soil in the baseline year (2014) had a higher SOM content (c.15% higher) than in 2015 and 2017 (Table 2.9). For soil moisture, soil in 2016 was significantly drier than all other years surveyed. There was also evidence of a positive relationship between soil organic matter content and soil moisture; soils with a higher soil moisture tended to have higher organic matter content (e.g. 80.0% soil moisture = 33% SOM with 95% CI of 23.0 – 46.0%; compared to 20% soil moisture = 18.0% SOM with 95% CI of 12.0 – 28.0%). Soil penetrability was inversely associated with soil moisture; wetter soil was softer to probe than dry soils. The proportion of variance in soil properties explained by these models are low when only the fixed effects are considered but variation explained is greater when the full model is considered (SOM: R^2_M = 0.065; R^2_C = 0.870; Moisture: R^2_M = 0.059; R^2_C = 0.809; Bumpiness: R^2_M = 0.016; R^2_C = 0.147).

There was weak evidence for an effect of the number of lime applications (either 1 or 2) on soil pH, as well as an inverse relationship with soil organic matter content, and a main effect of year (Table 2.8). However, when controlling for other variables, the estimated marginal means of soil pH either with 1 or 2 lime applications were non-significant (Table 2.9). Soil pH was not statistically different between Y1 and Y2, but was predicted to be significantly different between Y2 and Y3, with Y3 having a more neutral soil pH than Y2. The model explained variation in soil pH well (R^2_M = 0.403; R^2_C = 0.609).

Table 2.7. Summary table of the final (G)LMMs fitted to test the effect of lime treatment on soil properties. Note that SOM is used to abbreviate "soil organic matter" in the table.

Response variable	Type	Error structure	Link	Random effects	Fixed effects
Soil pH	Continuous	Gaussian	Identity	Farm/Field	Year*Treatment + SOM
Soil moisture	Continuous bounded (0–1) proportion	Beta	Logit	Farm/Field	Year + Soil moisture
SOM	Proportion	Binomial	Logit	Farm/Field	Year + Soil moisture
Log(Soil penetrability)	Continuous	Gaussian	Identity	Farm/Field	Year + Soil moisture
Log(Ground bumpiness)	Continuous	Gaussian	Identity	Farm/Field	SOM + Soil moisture
Soil pH	Continuous	Gaussian	Identity	Farm/Field	Number of lime applications + Year + SOM

Table 2.8. Summary table showing the minimal adequate (G)LMMs. Table shows the five models testing the effect of lime use on the response variables of soil pH, moisture, organic matter (SOM), penetrability and ground bumpiness as well as significant predictor covariates in the models. The final model is testing the effect of the number of lime applications (either 1 or 2) on the response variable of soil pH. Sample size was seven farms (n = 15 fields) across the four survey years (2014, 2015, 2016, 2017) for the first five models, while the sample size for the final model was seven farms (n = 15 fields) across three survey years only (2015, 2016, 2017). Note that covariates were scaled and centred prior to analysis for ease of coefficient interpretation. Table shows results of the species minimum adequate models with significance levels denoted by *'s where: * ≤ 0.05; ** ≤ 0.01; *** ≤ 0.001.

	Parameter estimates ± SE (test statistic with significance*) by predictor variable (type; df)					
Model (response)	**Treatment (factor; df = 1)**	**Year (factor; df = 3)**	**Year : Treatment (factor; df = 3)**	**SOM (covariate; df = 1)**	**Moisture (covariate; df = 1)**	**Lime applications (factor; df = 1)**
Soil pH	-	-	-	-0.11 ± 0.02	-	-
	(F = 69.66)	(F = 164.33***)	(F = 9.83***)	(t = -5.35***)		
Moisture	-	-	-	0.11 ± 0.03	-	-
				(z = 3.95***)		
SOM	-	-	-	-	0.01 ± 0.00	-
		(χ^2= 30.10*)			(z = 4.73***)	
Penetrability	-	-	-	-	-0.27 ± 0.03	-
		(F = 98.36***)			(t = -9.81***)	
Bumpiness	-	-	-	-0.08 ± 0.02	0.05 ± 0.02	-
				(t = -3.34***)	(t = 2.43*)	
Soil pH	-	-	-	-	-	-
						(F = 4.02*)

Table 2.9. Estimated marginal means (EMMs) for the (G)LMMs investigating the relationship between lime treatment and soil properties at the soil core scale. Lime Model 1 tests limed vs non-limed (control) and year on soil pH; Lime Model 2 tests limed vs non-limed (control) and year on soil moisture; Lime Model 3 tests limed vs non-limed (control) and year on soil organic matter (SOM); Lime Model 4 tests limed vs non-limed (control) and year on soil penetrability; Lime Model 6 tests fields receiving 1 or 2 applications of lime on soil pH. EMM values were computed by averaging together predictions on the linear-predictor scale and then back transforming to response scale with covariates set to mean values. A 95% confidence level was used. A pairwise tukey method was used for comparing EMMs with significant p values shown in bold. Please note table is continued on the following page.

	Marginal mean estimate ha^{-1}	Lower CI	Upper CI	Pairwise comparison	Test Statistic	p value
Lime Model 1 response: Soil pH						
Year*Treatment						
Y0:Control	4.94	4.69	5.18	Y0:Control – Y1:Control	t = 5.68	**<0.0001**
Y1:Control	4.65	4.41	4.90	Y0:Control – Y2:Control	t = 10.87	**<0.0001**
Y2:Control	4.35	4.10	4.60	Y0:Control – Y3:Control	t = -3.64	**0.0071**
Y3:Control	5.14	4.89	5.38	Y0:Control – Y0:Lime	t = -0.70	0.9969
Y0:Lime	4.97	4.73	5.22	Y0:Control – Y1:Lime	t = 1.39	0.8637
Y1: Lime	4.87	4.62	5.11	Y0:Control – Y2:Lime	t = 6.05	**<0.0001**
Y2: Lime	4.62	4.37	4.86	Y0:Control – Y3:Lime	t = -11.65	**<0.0001**
Y3: Lime	5.58	5.34	5.83	Y1:Control – Y2:Control	t = 5.57	**<0.0001**
				Y1:Control – Y3:Control	t = -8.88	**<0.0001**
				Y1:Control – Y0:Lime	t = -6.37	**<0.0001**
				Y1:Control – Y1:Lime	t = -4.37	**0.0004**
				Y1:Control – Y2:Lime	t = 0.66	0.9979
				Y1:Control – Y3:Lime	t = -16.85	**<0.0001**
				Y2:Control – Y3:Control	t = -13.79	**<0.0001**
				Y2:Control – Y0:Lime	t = -11.42	**<0.0001**
				Y2:Control – Y1:Lime	t = -9.59	**<0.0001**
				Y2:Control – Y2:Lime	t = -4.87	**<0.0001**
				Y2:Control – Y3:Lime	t = -21.42	**<0.0001**
				Y3:Control – Y0:Lime	t = 2.98	0.0592
				Y3:Control – Y1:Lime	t = 4.91	**<0.0001**
				Y3:Control – Y2:Lime	t = 9.21	**<0.0001**
				Y3:Control – Y3:Lime	t = - 7.69	**<0.0001**
				Y0:Lime – Y1:Lime	t = 2.10	0.4160
				Y0:Lime – Y2:Lime	t = 6.62	**<0.0001**

Table 2.8. Continued from previous page.

	Marginal mean estimate ha^{-1}	Lower CI	Upper CI	Pairwise comparison	Test Statistic	p value
				Y0:Lime – Y3:Lime	t = -10.97	<0.0001
				Y1:Lime – Y2:Lime	t = 4.744	0.0010
				Y1:Lime – Y3:Lime	t = -12.93	<0.0001
				Y2:Lime – Y3:Lime	t = -16.96	<0.0001
Lime Model 2 response: Soil moisture						
Year						
Y0	64.5%	60.7%	68.0%	Y0 – Y1	t = -2.33	0.0931
Y1	67.8%	64.2%	71.2%	Y0 – Y2	t = 15.70	<0.0001
Y2	40.3%	36.5%	44.1%	Y0 – Y3	t = 1.89	0.2342
Y3	61.7%	57.8%	65.4%	Y1 – Y2	t = 17.82	<0.0001
				Y1 – Y3	t = 4.20	0.0002
				Y2 – Y3	t = -13.75	<0.0001
Lime Model 3 response: Soil organic matter content						
Year						
Y0	26.6%	16.3%	34.5%	Y0 – Y1	t = 1.50	0.4207
Y1	25.1%	16.7%	35.8%	Y0 – Y2	t = 1.56	0.4002
Y2	27.8%	18.7%	39.1%	Y0 – Y3	t = 4.16	0.0002
Y3	24.7%	16.4%	25.5%	Y1 – Y2	t = -2.73	0.0330
				Y1 – Y3	t = 0. 34	0.9868
				Y2 – Y3	t = 2.94	0.0177
Lime Model 4 response: Soil penetrability						
Year						
Y0	3.3	3.10	3.41	Y0 – Y1	t = -2.72	0.0335
Y1	3.1	2.95	3.25	Y0 – Y2	t = -10.82	<0.0001
Y2	4.0	3.82	4.14	Y0 – Y3	t = -10.56	<0.0001
Y3	4.0	3.80	4.13	Y1 – Y2	t = -14.64	<0.0001
				Y1 – Y3	t = -14.35	<0.0001
				Y2 – Y3	t = 0.307	0.9899

Table 2.8. Continued from previous page.

	Marginal mean estimate ha^{-1}	Lower CI	Upper CI	Pairwise comparison	Test Statistic	p value
Lime Model 6 response: Soil pH						
Number of lime applications						
1 application	5.12	4.80	5.43	1 – 2 applications	t = 1.89	0.0612
2 application	4.92	4.59	5.24			
Year						
Y1	4.77	4.45	5.09	Y1 – Y2	t = 1.28	0.4060
Y2	4.66	4.35	4.98	Y1 – Y3	t = - 10.24	**<0.0001**
Y3	5.61	5.30	5.93	Y2 – Y3	t = - 14.24	**<0.0001**

2.4.4 Liming experiment: Earthworm community summary descriptions

Across the seven farms (n=15 fields) over the four years of the liming experiment a total of 6,272 earthworms were collected of which approximately 10% were adult earthworms consisting of 18 species (

Table 2.10). Please see Table A 0-5 in Appendix for full species inventory by site and treatment. The most abundant species recorded were *Aporrectodea caliginosa*, followed by *Aporrectodea limicola*, with the green coloured morph of *Allolobophora chlorotica* the third most abundant species recorded (Figure 2.8). Sample size of individuals were too low for formal species specific analysis and further complicated by the fact that species assemblages were different between each farm, with *A.caliginosa* being the only species identified at all seven farms, while *A.chlorotica* was identified at five of the farms and *A.limicola* at only three of the farms. However, the raw data shows that *A.limicola* and *Aporrectodea rosea* were recorded in smaller numbers on lime treatment soils, whereas *A.caliginosa* and *A.chlorotica* were recorded in higher numbers on the lime treatment soils compared to control soil.

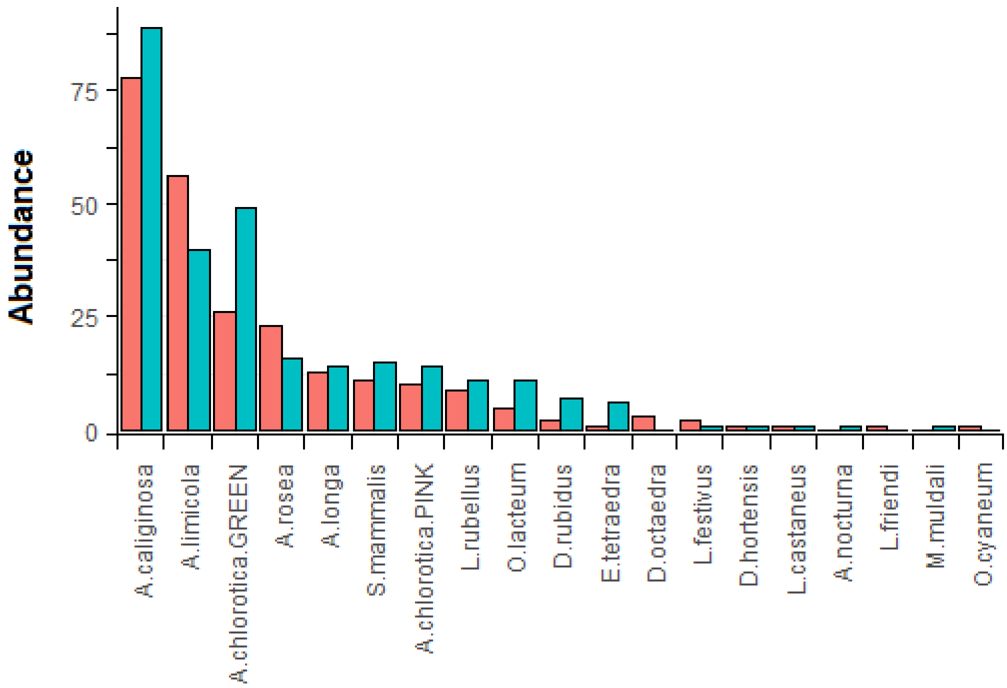


Figure 2.8. Bar plot showing the raw abundance data of adult earthworms identified to species level as part of the BACI lime experiment. Individuals recorded on control (non-limed) soil shown in red, and individuals recorded on lime treatment soil in green. Sample size was seven farms (n = 15 fields) across all four years of sampling.

Table 2.10. Summary table by year of the total number of individual earthworms recorded (abundance) by ecological grouping of adults (epigeic, endogeic, and anecic) and by mouth part for juveniles (tanylobic and epilobic), as well as the total earthworm abundance and species richness. Total sample size was seven farms (n = 15 fields) but varied between years and is shown in column N as the number of fields sampled for each year.

Year	N	Adult Epigeic	Adult Endogeic	Adult Anecic	Juvenile (tanylobic)	Juvenile (epilobic)	Total earthworm abundance	Species richness
Y0	13	12	76	14	621	593	1,338	12
Y1	13	23	115	5	835	717	1,774	14
Y2	12	21	92	3	732	674	1,530	14
Y3	11	14	136	7	726	713	1,630	10
Totals	15	2,984	419	29	2,914	2,697	6,272	18

2.4.5 Liming experiment: The relationship between lime treatment and earthworm abundance

There was strong evidence of an interaction effect between Year and Treatment with total earthworm abundance, as well as strong evidence of a decline in total earthworm abundance with increasing soil organic matter, but there was no relationship with soil moisture (

Table **2.11** and

Table **2.12**). Earthworm abundance per soil core was estimated, on average, to be 4.40 earthworms in both the control and lime treatment in the baseline year; however, abundance was estimated to be significantly different within Years between the control and lime treatment in Y2, with a higher abundance in the lime compared to the control (Figure 2.9A; Table 2.13). Between consecutive Years there was no evidence for a difference in earthworm abundance on the control plot with the exception of Y2 and Y3 which were significantly different with an abundance of earthworms higher in control soil of Y3 than control soil of Y2. For lime treatment soil there was a significant difference between the baseline year and all years post the application of lime onto the lime treatment plots, with earthworm abundance estimated to be lower in the baseline year. For soil organic matter content, earthworm abundance was predicted to be, on average, 4.8 (95% CI 3.52 – 6.65) earthworms per soil core with 20% soil organic matter, but was almost halved in very peaty soil of 80% organic matter with only 2.5 (95% CI 1.66 – 3.88) earthworms per soil core. The model explains only a small amount of variation in total earthworm abundance when only the fixed effects are considered (R^2_M = 0.064) but variation explained is greater when the full model is considered (R^2_C = 0.181).

Epigeic earthworm species also exhibited strong evidence of an interaction effect between Year and Treatment, as well as evidence of a decline in abundance with increasing soil organic matter, but again there was no relationship with soil moisture

Table 2.12). Epigeic earthworm abundance was predicted to have, on average, approximately 2.00 earthworms per soil core on both the control and lime treatment in the baseline Year (Table 2.13; Figure 2.9B). Within Years, there was a significant difference of epigeic earthworm abundance between control and lime treatment for Y2, with a higher epigeic abundance on the lime treatment than the control. Between Years, on the lime treatment plots there was a significant difference between the baseline Year and all post lime applications Years. For the control plots, there was only a significant difference between the baseline and the final post lime application Year (Y3). For soil organic matter content, epigeic earthworm abundance was estimated as 2.1 (95% CI 1.4 – 3.1) per soil core at 20% soil organic matter compared to 1.4 (95% CI 0.7 – 2.2) epigeic earthworms per soil core in peaty soil conditions of

80% organic matter. The model explains a low proportion of variation in epigeic earthworm abundance when only the fixed effects are considered (R^2_M) but explains a greater proportion of variation in epigeic earthworm abundance when both the fixed and random effects (R^2_C) are considered (R^2_M = 0.041; R^2_C = 0.223).

There was no evidence for an interaction effect between Year and Treatment with endogeic earthworm abundance, but there was strong evidence for a main effect of Year (

Table 2.12). Endogeic earthworm abundance was estimated to be significantly different between the baseline Year and the final Year (Y3), with abundance lower in the baseline Year compared with Y3 (Table 2.13; Figure 2.9C). Similarly, there was a significant difference between the Y2 and the final Year (Y3), with abundance higher in Y3 compared to Y2. There was no evidence to suggest an effect of soil properties (moisture and organic matter) on endogeic earthworms. The model explains a low proportion of variation in endogeic earthworm abundance when only the fixed effects are considered (R^2_M = 0.040) but explains a greater proportion of variation in endogeic earthworm abundance when the full model is considered i.e. both the fixed and random effects (R^2_C = 0.154).

Table 2.11. Summary table of the final (G)LMMs fitted to test the effect of lime treatment on earthworm abundance. Note that SOM is used to abbreviate "soil organic matter" in the table.

Response variable	Type	Error structure	Link	Random effects	Fixed effects
Total earthworms	Over dispersed count	Negative binomial	Log	Farm/Field	Year*Treatment + SOM
Epigeic earthworms	Over dispersed count	Negative binomial	Log	Farm/Field	Year*Treatment + SOM
Endogeic earthworms	Over dispersed ZI count	ZI negative binomial	Log	Farm/Field	Year*Treatment

Table 2.12. Summary table showing the (G)LMMs investigating the relationship between lime treatment and earthworms at the soil core scale while controlling for soil properties. Earthworm Model 1 tests limed vs non-limed (control) on total earthworm abundance; Earthworm Model 2 tests limed vs non-limed (control) on Epigeic earthworms; Earthworm Model 3 tests limed vs non-limed (control) on Endogeic earthworms. The sample size for all three Earthworm Models was seven farms (n = 15 fields) and four survey years (2014, 2015, 2016, 2017). Note that covariates were scaled and centred prior to analysis for ease of coefficient interpretation. Table shows results of the species minimum adequate models with significance levels denoted by *'s where: * ≤ 0.05; ** ≤ 0.01; *** ≤ 0.001.

Predictor	Type (df)	Parameter estimates ± SE (test statistic)		
		Earthworm Model 1 response: Total abundance	Earthworm Model 2 response: Epigeic abundance	Earthworm Model 3 response: Endogeic abundance

Treatment	Factor	-	-	-
	(df =1)	(F = 12.42)	(F = 16.20)	(χ^2 = 0.08)
Year	Factor	-	-	-
	(df =3)	(F = 8.73**)	(F = 7.05)	(χ^2 = 16.57***)
Year : Treatment	Factor	-	-	-
	(df =3)	(F = 4.54**)	(F = 4.85**)	
SOM (%)	Covariate	-0.19 ± 0.05	-0.11 ± 0.05	-
	(df =1)	(z = -3.56***)	(z = -2.17*)	

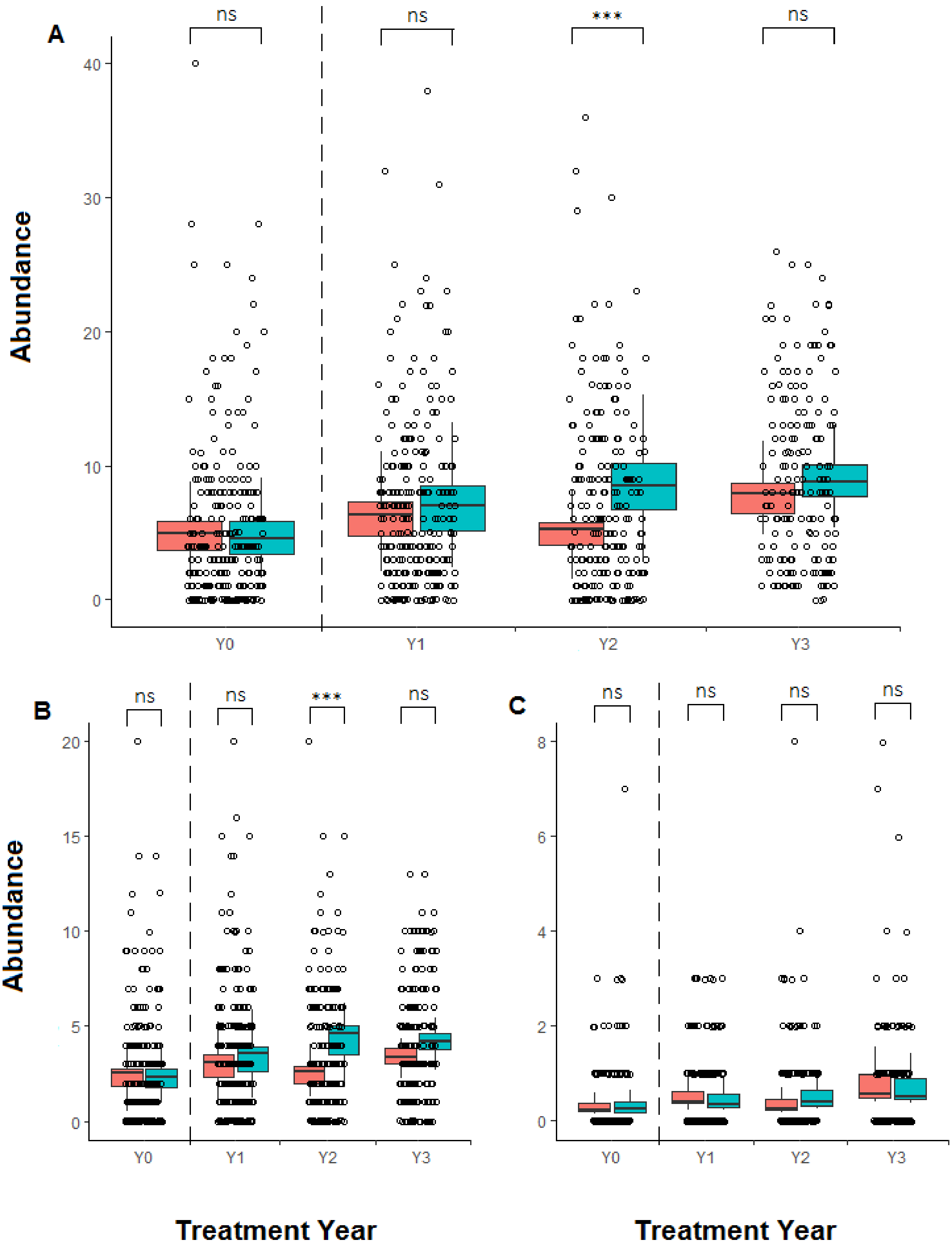


Figure 2.9. Boxplot graphically displaying the significant interaction effect between treatment (red = control; blue = lime treatment) and year (Y0 = baseline year (spring) before any lime treatment applied in the autumn; Y1 = first spring following the application of lime onto the lime treatment plots; Y2 = second spring following the application of lime onto the lime treatment plots; Y3 = third spring following the application of lime onto the lime treatment plots) on (A) total earthworm abundance; (B) epigeic abundance; and (C) endogeic abundance. Sample size is seven farms (n = 15 fields) and four years of data with formal analysis conducted on 980 observation. Raw data is represented by open black circles, while model predictions (conditional taking into account random effects) from the GLMM are shown as a boxplot displaying the interquartile range (IQR), medium as a black line and whiskers representing values that fall no further than 1.5 times the IQR. Dashed black line represents the application of lime treatment onto the lime treatment plots. Significance of pairwise comparisons of the estimated marginal means (Table 14) within years denoted by *'s where: ns > 0.05; * ≤ 0.05; ** ≤ 0.01; *** ≤ 0.001.

Table 2.13. Estimated marginal means (EMMs) for the (G)LMMs investigating the relationship between lime treatment and earthworms at the soil core scale while controlling for soil properties. Earthworm Model 1 tests limed vs non-limed (control) on total earthworm abundance; Earthworm Model 2 tests limed vs non-limed (control) on Epigeic earthworms; Earthworm Model 3 tests limed vs non-limed (control) on Endogeic earthworms. The sample size for all three Earthworm Models seven farms (n = 15 fields) and four years of data with formal analysis conducted on 980 observation. Values computed by averaging together predictions on the linear-predictor scale and then back transforming to response scale with covariates set to mean values. A 95% confidence level was used. A pairwise tukey method was used for comparing EMMs with significant p values shown in bold. Please note that table continues on following page.

	Marginal mean estimate ha^{-1}	Lower CI	Upper CI	Pairwise comparison	Test Statistic	p value
Earthworm Model 1 response: Total abundance						
Year : Treatment						
Y0:Control	4.49	3.28	6.16	Y0:Control – Y1:Control	z = -1.95	0.5173
Y1:Control	5.59	4.08	7.69	Y0:Control – Y2:Control	z = 0.45	0.9998
Y2:Control	4.26	3.07	5.90	Y0:Control – Y3:Control	z = -2.71	0.1196
Y3:Control	6.23	4.48	8.65	Y0:Control – Y0:Lime	z = 0.19	1.0000
Y0:Lime	4.40	3.20	6.04	Y0:Control – Y1:Lime	z = -3.14	**0.0363**
Y1: Lime	6.40	4.67	8.74	Y0:Control – Y2:Lime	z = -4.30	**0.0005**
Y2: Lime	7.51	5.46	10.33	Y0:Control – Y3:Lime	z = -3.99	**0.0017**
Y3: Lime	7.28	5.23	10.11	Y1:Control – Y2:Control	z = 2.29	0.2983
				Y1:Control – Y3:Control	z = -0.88	0.9878
				Y1:Control – Y0:Lime	z = 2.15	0.3849
				Y1:Control – Y1:Lime	z = -1.18	0.8721
				Y1:Control – Y2:Lime	z = -2.45	0.2153
				Y1:Control – Y3:Lime	z = -2.17	0.3696
				Y2:Control – Y3:Control	z = -3.07	**0.0422**
				Y2:Control – Y0:Lime	z = -0.27	1.0000
				Y2:Control – Y1:Lime	z = -3.39	**0.0163**
				Y2:Control – Y2:Lime	z = -4.59	**0.0001**
				Y2:Control – Y3:Lime	z = -4.31	**0.0004**
				Y3:Control – Y0:Lime	z = 2.91	**0.0702**
				Y3:Control – Y1:Lime	z = -0.21	1.0000
				Y3:Control – Y2:Lime	z = -1.53	0.7930
				Y3:Control – Y3:Lime	z = -1.83	0.5971
				Y0:Lime – Y1:Lime	z = -3.35	**0.0183**
				Y0:Lime – Y2:Lime	z = -4.51	**0.0002**

Table 2.11. Continued from previous page.

	Marginal mean estimate ha^{-1}	Lower CI	Upper CI	Pairwise comparison	Test Statistic	p value
				Y0:Lime – Y3:Lime	z = -4.20	**0.0007**
				Y1:Lime – Y2:Lime	z = -1.37	0.8711
				Y1:Lime – Y3:Lime	z = -1.09	0.9593
				Y2:Lime – Y3:Lime	z = 0.26	1.000
Earthworm Model 2 response: Epigeic abundance						
Year : Treatment						
Y0:Control	2.00	1.34	2.98	Y0:Control – Y1:Control	z = -1.87	0.5736
Y1:Control	2.48	1.67	3.69	Y0:Control – Y2:Control	z = 0.12	1.0000
Y2:Control	1.97	1.31	2.96	Y0:Control – Y3:Control	z = -1.86	0.5757
Y3:Control	2.52	1.68	3.79	Y0:Control – Y0:Lime	z = 0.19	1.000
Y0:Lime	1.95	1.31	2.91	Y0:Control – Y1:Lime	z = -3.08	**0.0433**
Y1: Lime	2.84	1.92	4.21	Y0:Control – Y2:Lime	z = -4.84	**<0.0001**
Y2: Lime	3.60	2.42	5.36	Y0:Control – Y3:Lime	z = -3.70	**0.0052**
Y3: Lime	3.15	2.10	4.73	Y1:Control – Y2:Control	z = 1.90	0.5522
				Y1:Control – Y3:Control	z = -0.12	1.0000
				Y1:Control – Y0:Lime	z = 2.07	0.4344
				Y1:Control – Y1:Lime	z = -1.23	0.9241
				Y1:Control – Y2:Lime	z = -3.15	**0.0347**
				Y1:Control – Y3:Lime	z = -2.00	0.4854
				Y2:Control – Y3:Control	z = -1.94	0.5260
				Y2:Control – Y0:Lime	z = 0.06	1.0000
				Y2:Control – Y1:Lime	z = -3.03	**0.0508**
				Y2:Control – Y2:Lime	z = -4.88	**<0.0001**
				Y2:Control – Y3:Lime	z = -3.72	**0.0049**
				Y3:Control – Y0:Lime	z = 2.06	0.4405
				Y3:Control – Y1:Lime	z = -1.01	0.9727
				Y3:Control – Y2:Lime	z = -2.89	0.6172
				Y3:Control – Y3:Lime	z = -1.80	0.6172
				Y0:Lime – Y1:Lime	z = -3.30	**0.0217**
				Y0:Lime – Y2:Lime	z = -5.06	**<0.0001**
				Y0:Lime – Y3:Lime	z = -3.92	**0.0023**

Table 2.11. Continued from previous page.

	Marginal mean estimate ha^{-1}	Lower CI	Upper CI	Pairwise comparison	Test Statistic	p value
				Y1:Lime – Y2:Lime	z = -2.02	0.4706
				Y1:Lime – Y3:Lime	z = -0.87	0.9889
				Y2:Lime – Y3:Lime	z = 1.09	0.9600
Earthworm Model 3 response: Endogeic abundance						
Year						
Y0	0.29	0.19	0.45	Y0 – Y1	t = -2.40	0.0782
Y1	0.44	0.29	0.67	Y0 – Y2	t = -0.97	0.7646
Y2	0.35	0.23	0.55	Y0 – Y3	t = -4.22	**0.0002**
Y3	0.61	0.40	0.93	Y1 – Y2	t = 1.34	0.5409
				Y1 – Y3	t = -2.05	0.1719
				Y2 – Y3	t = -3.26	**0.0064**

2.5 DISCUSSION

The aim of this study was to examine if there were any associations between earthworm abundance and soil properties, and to experimentally test any effect of lime use (when applied as a surface dressing with no till) on soil pH and earthworms on upland grasslands. By amending soil pH, lime use could be a novel and simple tool for improving upland grassland fields for the benefit of farmland breeding waders and other farmland birds which rely on earthworms as a key food resource. Here, I show that earthworm abundance was correlated with several soil properties, which differed between the earthworm ecological groupings. Additionally, I demonstrate that lime use can influence both soil pH and earthworm abundance. The results here could therefore have important conservation implications for declining farmland birds in these habitats.

2.5.1 The relationship between soil properties and earthworms

In support of hypothesis 1, I found that total earthworm abundance increased with increasing soil pH and that earthworm abundance was very low in soils below pH 4.5 which accounted for 12% of samples, but had a tenfold increase in neutral soil conditions of pH 6.0, after controlling for soil organic matter. Total earthworm abundance was strongly related to soil organic matter with very few earthworms in soils with high organic matter content. In the environments sampled in this study, these sort of high organic soils would be capturing very peaty soils. Peaty soils typically have a high soil moisture content, low soil pH and a poor quality of organic matter which combine to make these soils generally unsuitable for earthworms (Edwards, Hendrix and Arancon, 1996).

Although total earthworm abundance was modelled to have a linear relationship with soil pH, it is likely that this relationship would not hold true indefinitely with increasing soil pH, and that with a wider soil pH range in the samples a second-order (e.g. curvilinear) effect may have been detected. This looks particularly the case in the raw data which appears to show a unimodal pattern peaking around pH 5.2 (Figure 2.5A). Similar studies on agricultural grasslands in Scotland have also found peak earthworm abundance occurring at around pH 5.2 (e.g. McCallum *et al.*, 2016). Additionally, it is likely that different earthworm species will have different preferences for soil conditions and may not all follow the positive association with soil pH observed with total earthworms in this study. The number of individuals recorded to species level was too low for formal analysis to test this hypothesis; however, of the earthworm ecological groups tested, epigeic earthworm abundance exhibited a strong and positive relationship with soil pH, but soil organic matter content was unimportant. Conversely, endogeic earthworm abundance exhibited a strong relationship with soil organic matter content, but soil pH was unimportant. In all instances, there was no association with soil moisture. This suggests that overall, soil pH is playing an important role on earthworm abundance.

Agricultural grasslands in upland landscapes are unlikely to have very alkaline soils, as high rainfall and low buffer capacity tend to lead to leaching and become acidified over time. Indeed, the majority of samples recorded in this study fell below pH 5.0 (n = 56% of soil cores) which is below the optimum for agricultural grassland. The combination of low soil pH in these habitats and the marked additive effect of soil pH on both total earthworm abundance and epigeic earthworm abundance in the model supports the hypothesis that increasing soil pH in upland grassland areas through lime use may have benefit for farmland breeding waders by increasing earthworm abundance.

2.5.2 The relationship between lime treatment and soil properties

In support of hypothesis 2, liming carried out as part of the BACI field experiment resulted in higher average soil pH on the lime treatment compared to the control that had not been limed in all years following the application of lime, with the largest effect size in year 2 (average of 0.52 pH units higher on lime treatment). There was no difference in soil pH, or any of the other soil properties (moisture, organic matter, penetrability) or ground bumpiness in the baseline survey Year (Y0) between treatments. There was no effect of liming on any of the other soil properties or ground bumpiness, suggesting that the differences in soil pH which were detected post liming were likely due to liming rather than intrinsic differences in the nature of the soil.

Differences in soil pH were also detected between Years on both the lime treatment and control soils. As anticipated there was a lag period on the lime treatment soils before an observable increase in soil pH from the baseline year was detected. Lime treatment was applied in autumn 2014, with the first year of sampling post lime application (Y1) in spring 2015 resulting in a period of around six months between sampling. Lime can take between 6 - 12 months for an observable effect on soil pH to be detected, although this will vary depending on the neutralising value (NV) of the lime product, lime particle size, soil type, soil texture, fertiliser use and the method in which lime was applied (Goulding and Annis, 1998; SRUC, 2014). Soil pH in other studies have been shown to increase more quickly when tilled in addition to receiving lime (e.g. McCallum, 2012). Tillage enables lime to be incorporated into the soil faster than when lime is applied directly onto the grass surface, thus speeding up the time between initial application and an increase in soil pH (Conyers *et al.*, 2003). However, conventional tilling practices have been shown to be detrimental to earthworm species (Briones and Schmidt, 2017). In this study lime was applied as a surface dressing only in order to mitigate the negative impacts on earthworms and other soil invertebrates, as well as to conserve permanent grassland and to provide a simple and quick management option which may potentially benefit farmland breeding waders. In the second spring of sampling post lime application (Y2) a period of 18 months had passed since lime was applied onto the lime treatment plots, and an observable effect would have been expected in soil pH. Although there was a significant difference in soil pH within Year between the control (non-limed) and lime treatment soils, the soil pH was not greater than the baseline year on the lime treatment. This could have been caused by the fact that eight out of the 14 fields sampled in Y2 had required a second application of lime in autumn six months previously (in autumn 2015). Furthermore, the weather conditions of this survey year (2016) may have influenced the soil pH results observed. The 2016 winter months of January and February were uncharacteristically wet with several storms occurring resulting in periods of heavy rain and local flooding (Kendon *et al.*, 2016; McCarthy *et al.*, 2016). Excessive rainfall events and flooding can mobilise ions in the soil solution resulting in loss of these ions and thus an alteration in soil acidity (Holland *et al.*, 2018). This may have naturally lowered soil pH resulting in the more acidic soil pH values observed than would be expected in this year. In the third sample year (Y3) after the first application of lime there was a significant difference detected in soil pH on the lime treatment compared to lime treatment in the baseline year which showed an average increase of 0.6pH units, after controlling for other variables, raising the estimated marginal mean soil pH from 5.0 to 5.6 on the lime treatment. In Scotland, mineral soils below pH 5.6 have soluble aluminium which can inhibit root growth of some crops, but is less of a concern in organic soils which contain less aluminium. In this study, if we use a threshold of 20% soil organic matter content to represent mineral soils, approximately 22% of our samples were mineral soil and would therefore have benefitted from the application of lime applied in this study. It is unlikely that if soils had been sampled for an additional year further increases in soil pH would have been detected on the lime treatment soils. Fine lime particles would have reacted to change soil pH in the first instances, but if any larger (i.e. coarse) particles of lime were remaining at the end of the study these are likely to dissolve more slowly and maintain the soil pH, rather than result in a further substantial increase. Indeed, previous studies have reported soil pH to decline by around 0.07 pH units with every year since liming on agricultural grasslands (McCallum *et al.*, 2016), and it is recommended that soils require additional lime to maintain soil pH every 5 – 10 years, although soil analysis prior to lime application is always recommended.

Soil pH of the un-limed control also increased during the three years, although this increase was smaller than for the limed treatments, with an average increase of 0.2pH units in Y3 compared to Y0. It is likely that the increase in pH on control arose from contamination of lime, as has been noted in previous trials involving liming (Bishop *et al.*, 2008; McCallum *et al.*, 2016). There are a number of ways that contamination may have occurred including; difficulties constraining lime application to the 2ha plot using machinery that was designed to spread lime over a large area, or by transfer of lime between treatments by livestock or leaching of lime between plots following heavy rain despite the buffer strip between the plots. The study also observed a general decrease in soil pH in the first (Y1) and second (Y2) survey year post the application of lime treatment, although the decline in soil pH was greater on the control soils. This could have arisen by a couple of reasons: 1) gradual acidification of soils occurs naturally over time, particularly in upland systems where the underlying geology has poor buffer capacity and experiences heavy rainfall which leaches the soils. This process is amplified under agricultural management by livestock excretion. All the fields in the study were grazed with livestock free to move between plots. So some level of pH decline is expected to have occurred. 2) Soil pH varies significantly within fields especially if there is a range in soil textures and types within the field so natural variation in mean soil pH can occur. As sample points for soil cores were randomised each year, the observed overall decline in soil pH may have occurred by chance alone.

Overall, liming carried out within the BACI lime experiment of either one or two applications of 4 t ha^{-1} of dust lime applied as a surface dressing (no-till) onto agricultural grassland in upland areas resulted in an increase in soil pH of, on average, 0.60 units resulting in soil pH considerably closer to that recommended for growing grass (pH 5.3-5.5 for organic soils; and pH > 5.6 for mineral soils; SAC 2014). The effect of liming on soil pH in this study is not that dissimilar to those previously reported of lime applied as a surface dressing, particularly when we consider the greater application rate of lime in other studies; approximately 1.4 pH units following an application of 17.3 t ha^{-1} (Bolton, 1977), approximately 1 pH unit following an application of 12 t ha^{-1} (Stevens and Laughlin, 1996), approximately 0.9 pH units following an application of around 8 t ha^{-1} (Bailey, 1997), and a pH increase of around 1.3 pH units following three applications of 4 t ha^{-1} (McCallum, 2012). For established grassland with no soil cultivation, no more than 7.5 t ha^{-1} should be applied in one application. Therefore the results shown here support the hypothesis that spreading lime as a surface dressing onto established grassland in upland areas which had sub-optimal starting pH amend acidic soil to more neutral values.

2.5.3 The relationship between lime treatment and earthworms

My results mainly support hypothesis 3, where liming carried out as part of the BACI field experiment resulted in higher total earthworm abundance and epigeic earthworm abundance on the lime treatment compared to the control (not limed) in Y2 only, but not within the other two years of post-lime application sampling (Y1 and Y3). For between Years, total earthworm abundance and epigeic earthworm abundance on the lime treatment plots increased significantly from the baseline year (Y0) in the first sampling year post lime application (Y1), with abundance remaining higher than the baseline year in all post lime sampling years (Y2 and Y3). On the non-limed control there was no significant difference in total earthworm, epigeic nor endogeic abundance from the baseline year with any post lime sampling years.

For endogeic earthworms, there was no evidence to suggest an effect of lime; however there was an additive main effect of Year, with higher earthworm abundance estimated, on average, with every consecutive Year. The variance explained by the model for this ecological grouping of earthworms was low, likely as a result of small numbers and a high number of zeros in the dataset which would reduce the power of analysis to detect an effect of lime. However, the results for endogeic earthworms in this study appear to reflect other studies of earthworms and liming which have found either none, or only small increases in endogeic earthworm species from liming (Bishop, 2003; Potthoff *et al.*, 2008).

The similarity of the shape of earthworm response in control and limed plots (Figure 2.9) is interesting and has the potential to reduce confidence in the measurement of differences. Firstly, it may reduce confidence in whether or not an effect of treatment is a true effect which differs from the control, and secondly it may mask the true effect size of treatment on the response. To add robustness to the liming experiment and confidence in the results I used a split plot design at the field level following a before-after-control-impact (BACI) study. It is important that the same field was used for both control and lime treatment in order to reduce variability that cannot be controlled for in studies conducted in agricultural fields such as weather or management history. In this way the response observed from control and treatment can be compared to establish a cause-and-effect relationship as both control and treatment will have experienced the same conditions, the difference in the response can therefore be attributed to the treatment variable. Furthermore, the nature of the BACI experiment provides additional validity to the results of the study as it allows for comparisons pre-intervention as well as post-intervention. Although there was a high proportion of variance in the data this is common for ecological studies conducted in the field, as opposed to controlled laboratory studies. This high variance can reduce the power to detect an effect; however, the multiple years of fieldwork and number of farms included in the study strengthens confidence in the results that there is a true effect of lime treatment, despite the similarity in the shape of response. Larger sample sizes, multiple field seasons, and replication of the experiment reduce the likelihood of an effect of treatment to have occurred by chance alone (i.e. a false positive effect of treatment). Thus the large sample sizes here can give confidence in the results of the study to provide a true positive response of treatment on earthworm populations. However, confidence in the precision or accuracy of the effect size of lime treatment may be low. Annual variation in wild populations will occur naturally, and earthworms are highly mobile species. There were no physical barriers in the field between control and lime treatment, meaning that earthworms could move freely between treatments. It is plausible that this movement could affect the results of the study as a potential increase in reproduction (as a result of improved soil condition on lime treatment) could have led to spill over between plots reducing overall effect sizes. The ability to detect a response may also have been reduced due to the pH difference between control and treatment being relatively small, and thus the effect of lime on earthworm response may vary if larger differences in soil pH is achieved at the individual field level. This will be entirely influenced by the starting and end point pH of a field, and the species of earthworm that are present in the field at the start of lime treatment.

In this chapter, the low numbers of endogeic earthworms is in part due to the high proportion of juveniles within the dataset making identification of ecological grouping difficult, but may also be an artefact of the sampling design as the soil core was only taken to a depth of 10cm, and in general, endogeic earthworms (which build complex horizontal burrow systems within the top soil) can be found to a depth of 30cm, and may have therefore been missed. Furthermore, soil cores suffer from avoidance behaviour of earthworms, with potential lateral escape of near-surface dwelling species and vertical escape of deeper burrowing species of the anecic ecological grouping such as *Lumbricus terrestris* which will be almost entirely missed by sampling with a 10cm soil core. This could have been mitigated somewhat by digging larger and deeper soil pits as recommended by the Earthworm Society of Britain, but this would have reduced the number of samples and farms covered in this study due to the logistical constraints of processing and extracting earthworms. Alternative earthworm sampling methods include the use of a chemical treatment such as mustard solution which is relatively time efficient (Gunn, 1992; Lawrence and Bowers, 2002). However, chemical extraction was considered unsuitable for this study because the penetration of the chemical irritant depends upon soil porosity and water-logging (Nordström and Rundgren, 1972) and may therefore not be consistent between sampling points. In addition, the earthworm’s escape response to chemical irritants is age- and species-specific which can cause bias to the data, which is thought to be due to differences in diapause patterns and the direction and stability of burrow systems (Nordström and Rundgren, 1972; Bouche and Gardner, 1984). Hand-sorting of the soil cores to extract earthworms may also have led to some human error as the efficiency of hand-sorting can vary between species and soil types, with smaller and darker species tend to be missed, particularly in heavier soils (Raw, 1960). Nevertheless, for the comparative purposes of this study, and because I was interested primarily in investigating the possibilities of higher earthworm abundances as food resource for farmland breeding waders, soil cores were considered a sufficient method. It is also unlikely that lime applied as a surface dressing would have penetrated and neutralised soil acidity deeper than 7cm, so it is plausible to presume that earthworms missed by the soil core method to a depth of 10 cm would not have been affected by lime treatment.

Although the sampling method used in this study may therefore have reduced the power to detect an effect of lime on earthworms on endogeic or anecic ecological groupings, endogeic earthworms rarely come to the soil surface, and instead spend their life in the soil where they feed on mineral soil and decayed organic matter. It is therefore unlikely they contribute as much to wader diet. Indeed, Curlew and Lapwing diet of earthworms has been found to be primarily made up of the smaller epigeic earthworm species (Gillings and Sutherland, 2007; Navedo *et al.*, 2020) which were shown in this chapter to increase with lime treatment. The BACI lime treatment in this chapter also showed that lime treated soils had higher soil pH, coupled with the results of the observational study which showed a positive correlation of abundance with pH, suggests that the mechanism behind the increase in earthworm abundance on the lime treatment plots was from the change in soil pH. Increased cocoon production and hatching success has been reported from the application of lime to acidic soils of forests, resulting in increased earthworm abundance at high pH (Robinson *et al.*, 1992). Although we did not explicitly test the mechanism on how earthworm abundance increased in this study, it is likely to have been through increased reproduction rather than colonization. The increased number of earthworms may have also spilled over into the control plots and could explain why a difference between treatment plots within Year was not detectable in Y3 of the study, particularly as soil pH also increased on the control soil in that sampling year.

At an earthworm species level, I was unable to test whether there were species level effects caused by lime use, although it would be expected that species will have different tolerances to soil pH and other soil properties. Of the three most abundant species recorded in this study, *Aporrectodea caliginosa* and the green morph of *Allolobophora chlorotica* are considered to be common and abundant species typical of agricultural grasslands, whereas *Aporrectodea limicola* is considered a rare but locally abundant species of grasslands and moist habitats (Sherlock, 2018). Similarly a recent national survey of earthworms in England commissioned by Natural England also found that *A.chlorotica* was highly abundant, making up 34% of all earthworm specimens identified (Jones and Eggleton, 2014). *A.caliginosa* and *A.chlorotica* were identified in higher numbers in lime treated soil than in control in this chapter suggesting that they may be positively influenced by amending soil pH. Although *A.chlorotica* has a wide tolerance band to soil pH ranging from pH 4.5 to pH 8.2 (Sims and Gerard, 1985) this species is typically considered one of neutral and alkaline soils (Kalmus, Satchell and Bowen, 1955) and has previously been shown to respond positively to liming (Bishop, 2003; McCallum *et al.*, 2016). Although *A.chlorotica* is an endogeic earthworm species, it is generally found in the upper 5 cm of soil amongst the grass roots (Gerard, 1967), which is considered a depth that Lapwing chicks can forage within to find food resources (Beintema *et al.*, 1991) and radio tagged Lapwing chicks have been observed to forage in areas with higher densities of *A.chlorotica* than randomly selected locations within the field more often than by chance (McCallum *et al.*, 2016). The small size of *A.chlorotica*, its abundance in agricultural fields, and its depth within the soil profile make it a likely prey item for waders. The most abundant earthworm species in this study, *A. caliginosa,* is also an endogeic earthworm which is typically found lower in the soil profile within the top 10 cm (Gerard, 1967) and may therefore not be as suitable as a prey item for chicks, although it should still be accessible to adult waders with longer bill lengths. The third most abundant species, *A.limicoloa*, was identified in higher numbers on control soil. This earthworm species is considered an acid loving species (Sherlock, 2018), and therefore it is not surprising that lime treatment would have altered soil pH negatively for this earthworm. Although I could not test for this formally, the raw data highlights that lime use may not be beneficial in all instances, and that the earthworm species of the field should be considered prior to the use of lime to benefit waders.

2.5.4 Conclusions, conservation implications and further research

The results of this chapter show that total earthworm abundance is positively correlated with soil pH, and that the use of lime as a surface dressing on fields with sub-optimal pH of agricultural grasslands in marginal upland areas can increase soil pH, and increase total earthworm abundance. Earthworms have been identified as important food items for wader body condition during the pre-breeding season (Högstedt, 1974; Galbraith, 1989; Berg, 1993) as well as for chick development and condition, the latter being positively associated with the number of earthworm chaetae within their faeces (Beintema *et al.*, 1991; Sheldon, 2002). Reduced foraging quality can, therefore, lead to poor breeding success (e.g.

Kentie *et al.*, 2013) contributing to population declines. These results are therefore of significant conservation importance when we consider the decline in lime use in the UK by more than 70%, and that declining soil pH associated with a reduction in lime is likely to have led to declines in total earthworm abundance and reduced the quality of foraging habitat for waders. This will especially be the case where underlying geology has poor buffering capacity and high rainfall levels which gradually reduce soil pH through leaching, which are typical of marginal upland agricultural grasslands

There are negative aspects of lime use which needs to be considered carefully if it is be to used as a conservation tool, notably how it influences carbon sequestration and the emission of greenhouse gases from soil such as the release of carbon dioxide through the neutralising process (West and McBride, 2005; De Klein, 2006; Fornara *et al.*, 2010; Shaaban *et al.*, 2017; Holland *et al.*, 2018). Although there is some evidence to suggest that lime use may decrease nitrous oxide release from soils (Hénault *et al.*, 2019), and potentially increase carbon stocks and phosphorus uptake in soils (Paradelo, Virto and Chenu, 2015; Barrow, 2017) the overall net climate change impact is likely to be negative (Gibbons *et al.*, 2014). Additionally, lime use may have detrimental effects on wider biodiversity, such as other soil invertebrates not discussed here (e.g. tipulid larvae), as well as above ground invertebrates such as spiders and ground beetles which also contribute to the diet of many farmland birds, or botany of conservation interest, which could be negatively impacted by changes in the sward structure and soil chemistry from lime use. The effect on earthworms may also be varied, particularly if a site has a species composition of “acid-loving” earthworms, which would be negatively affected by lime. Furthermore leaching of lime can occur which should be considered carefully when near waterbodies. Therefore the use of agricultural liming as a bird conservation intervention in upland areas would need to be carefully targeted, and further research on the impact of lime on other species of agricultural grasslands would be beneficial, as well as studies on the foraging and breeding success of waders on limed fields, as although the abundance of potential earthworm prey may be increased, if accessibility to prey is limited (e.g. due to dense sward) overall foraging condition may not be improved.

To conclude, lime use can increase total earthworm abundance on agricultural grasslands and may be suitable as a conservation tool to improve foraging habitat for farmland breeding waders and other birds which rely on earthworms, while also allowing for productive farming. A strategy for liming in the uplands would be needed to exploit its potential conservation benefit by careful targeting of agricultural grasslands of low biodiversity interest that are suitable for farmland breeding waders and where soil pH has fallen below that recommended for agricultural grass production, to create a patchwork of limed and non-limed fields within the landscape.

CHAPTER 3 | EFFECTS OF LIME MANAGEMENT ON ARTHROPODS OF UPLAND AGRICULTURAL GRASSLANDS: IMPLICATIONS FOR FARMLAND BREEDING WADERS.

3.1 ABSTRACT

The expansion and intensification of agriculture is generally regarded as one of the major contributors to the observed declines in farmland biodiversity globally. Agri-environment schemes are designed to halt and reverse this loss by encouraging the uptake of beneficial management for farmland biodiversity. Recently, the use of lime has been proposed as a novel option for farmland birds to manipulate soil condition which may have become locally acidified due to anthropogenic activity. The mechanism behind the benefit to birds being an increase in earthworm abundance, an important food resource for the Northern Lapwing *Vanellus vanellus* and Eurasian Curlew *Numenius arquata*. However, change in soil pH may also alter vegetation structure and thereby impact other non-target arthropods such as beetles (Coleoptera) and spiders (Aranea), which are also important prey for many farmland birds. Furthermore, changes in sward can affect the accessibility of prey items to foraging birds, which may be influenced by the use of lime. In this study, the effects of lime use on vegetation (e.g. height and heterogeneity) and on spiders and beetles was tested over two consecutive years (2015 and 2016) and analysed by season (early and late). I show that there was no effect of lime treatment on either vegetation height or vegetation heterogeneity. There was also no effect of lime on beetle abundance. However, there was an observed positive effect of lime use on spider abundance in the late season which varied by year; spider abundance was similar on limed and control in both years, but there was a marginally non-significant increase of 38% in abundance on limed plots between 2015 and 2016. The results from this study suggest that there are no detrimental impacts on spiders or beetles from lime treatment, and that lime treatment is unlikely to produce significant changes in sward height and heterogeneity on grazed marginal grassland. Overall, this provides additional support to the hypothesis that lime use may be suitable as a highly targeted conservation tool for farmland birds in marginal upland areas where localised acidification may have occurred over time.

3.2 INTRODUCTION

The expansion and intensification of agricultural landscapes have been characterized by an increase in management intensity, and a consequent reduction and fragmentation of natural and semi-natural habitats, and is considered a major driver of worldwide biodiversity loss (Benton *et al.*, 2002; Robinson and Sutherland, 2002; Kleijn and Sutherland, 2003; Mattison and Norris, 2005; Tscharntke *et al.*, 2005; SCBD, 2014a). Agri-environment schemes (AES), are incentive-based mechanisms aimed at counteracting the adverse effects of modern agriculture on biodiversity and involve payments to farmers in exchange for environmental goods and services (Burrell, 2012). While the success of agri-environment schemes in conserving and restoring farmland biodiversity is difficult to assess because they are poorly monitored (Kleijn and Sutherland, 2003), several studies have found biodiversity improvements in response to changed agricultural practices under AES programs (e.g. Knop *et al.*, 2006; MacDonald *et al.*, 2012) with some of the best examples of AES driving population recovery of the target group at regional and/or national levels in farmland birds (e.g. Peach *et al.*, 2001; O'Brien, Green and Wilson, 2006; Bretagnolle *et al.*, 2011; Perkins *et al.*, 2011). Conversely, other studies have shown mixed or limited benefits (e.g. Kleijn *et al.*, 2004; Feehan, Gillmor and Culleton, 2005; Verhulst, Kleijn and Berendse, 2006) and even negative biodiversity outcomes (e.g. Fuentes-Montemayor, Goulson and Park, 2011; Besnard and Secondi, 2014). Despite the mixed results of AES, uptake has been high within Europe with approximately 25% of the EU's utilised agricultural area under some form of AES contract with farmers, including organic farming (Science for Environment Policy, 2017b). The financial investment of AES is also large, with the European budget for the period 2014 – 2020 at c. €363 billion (Navarro and López-Bao, 2018), representing a significant contribution to conservation effort. As a result, AES are considered an important policy instrument to preserve and restore biodiversity in European agricultural landscapes and have the potential to be an important tool to address biodiversity declines in farmland (Vickery *et al.*, 2004; Kleijn *et al.*, 2006; Batáry *et al.*, 2015).

The mixed success of AES may be due to the scale (temporal and spatial) of the implementation (O'Brien and Wilson, 2011), poor design of measures that vary in their effectiveness (Smart *et al.*, 2013) or that the current options are missing specific habitat requirements of the target species (Redpath *et al.*, 2010; Fuentes-Montemayor, Goulson and Park, 2011). One area of sustainable agriculture which has received relatively little attention in the context of farmland wildlife is soil degradation, in particular pH, which has shown to have an additive effect and improve model fit of some farmland bird species (McCallum *et al.*, 2015; Johannesdottir *et al.*, 2018). Soil pH can influence vegetation structure and growth, as well as the diversity and abundance of soil invertebrates associated with the soil (Curry, 2004; Cole *et al.*, 2006), such as earthworm species which are more abundant in temperate agricultural soils with a more neutral soil pH (Edwards, Hendrix and Arancon, 1996). Soil invertebrates are important food resources for a wide range of farmland-feeding birds including waders, thrushes, starlings and corvids (e.g. Tucker, 1992; Gruar, Peach and Taylor, 2003) and the accessibility and abundance of food resources are likely to influence field use by birds (Robinson and Sutherland, 1999; Brickle *et al.*, 2000; Atkinson, Buckingham and Morris, 2004). It is therefore surprising that soil pH has not received much consideration in studies on farmland birds, although there is good evidence on the influence of soil moisture on soil invertebrates for farmland birds (e.g. Devereux *et al.*, 2004; Eglington *et al.*, 2010) and consequentially options are available within AES for manipulating water levels for the benefit of birds.

Soil pH is reduced (becomes more acidified) naturally over time through the leaching of base cations, but this process can become intensified under agricultural management through the use of nitrogenous fertilisers, cropping, and livestock excretion as well as acid deposition from industrial pollution (Rowell and Wild, 1985). The effects of soil acidification can be largely counteracted through agricultural liming ($CaCO_3$) to raise soil pH (MAFF, 1969; Wilkinson, 1998; Spaey, Traon and Joas, 2012); however, both agricultural lime production and sales have declined in the United Kingdom (UK) and Europe (European Comission, 2018; Holland *et al.*, 2018; EuLA, 2020). In the UK lime sales have declined by over 70%, following a peak in the 1960s linked to Government lime subsides which were abolished in the 1970s (Church, 1985; Skinner and Todd, 1998; Wilkinson, 1998; McCallum *et al.*, 2016). Additionally, the production of lime products for agricultural uses (e.g. limestone, dolomite and chalk) have declined in Great Britain since the 1980s, with annual production of lime over the last 10 years around 2,000,000 tonnes (Idoine *et al.*, 2016), around half of the average estimated annual lime loss for the UK (> 4,000,000 tonnes of $CaCO_3$; Goulding and Annis, 1998). Overall, liming is becoming less common in the UK (Goulding, 2016).

Historical declines in soil pH have also been reported in the UK between the 1970s and 2001, particularly in the west and north of the UK in areas that are dominated by livestock farms in higher altitude and higher rainfall environments (Baxter, Oliver and Archer, 2006), where the underlying geology (i.e. acidic bedrock) has poor buffer capacity to leaching, making them more prone to localised acidification (White, 2006). Although there is some evidence to suggest that the trend in declining soil pH is now being reversed (Baxter, Oliver and Archer, 2006; Emmett *et al.*, 2010; Kirk, Bellamy and Lark, 2010), recent reports of UK soil samples from the Professional Agricultural Analysis Group (PAAG, 2021) show that approximately 15% of samples sent for soil testing from arable soils in the year 2019/2020 had a pH of

<6.0, and for grasslands samples 15% were below pH 5.5. Although PAAG does not represent a rigorous sampling across the UK, the collation of 125,000 soil analyses (pH, P, K, Mg) can help to provide some broad trends. Based on these results, there seem to be significant areas of both arable land and grassland where lime application would be recommended based upon good agricultural practice (GAP) in England and Wales (DEFRA, 2010) and in Scotland (Sinclair, Crooks and Coull, 2014).

Localized soil acidification in the UK, potentially related to a reduction in lime use, may have reduced the quality of farmland-feeding bird foraging habitat due to lower abundances of soil invertebrates associated with more neutral soil conditions. Mitigating these impacts via the use of lime management has been suggested as a potential conservation tool for agricultural grasslands in marginal upland areas for the benefit of farmland breeding waders, notably the Eurasian Curlew, *Numenius arquata,* and Northern Lapwing, *Vanellus vanellus* (McCallum *et al.*, 2016). These waders are dependent on farmland for breeding and foraging habitat during the summer, particularly agricultural grasslands (Piersma, 1986). However, several agricultural intensification practices are detrimental to waders on grassland such as the drainage of wet grasslands, increased livestock densities, and changes in the timings and frequency of agricultural operations (Vickery *et al.*, 2001; Wilson, Vickery and Pendlebury, 2007), which have reduced both the availability and security of suitable nesting habitat, as well as the availability of large, soft-bodied soil arthropod prey which these birds depend upon (Newton, 2004; Wilson, Evans and Grice, 2009). The association between Lapwing distribution in Scotland and higher altitude areas with relatively high (more neutral) soil pH also supports the hypothesis that waders may benefit from lime use in the uplands (McCallum *et al.*, 2015). Lime use can increase soil pH for several years with limed fields holding higher earthworm and Lapwing densities than non-limed fields (McCallum *et al.*, 2016). Therefore lime use could provide long-term benefits of improved soil conditions for crop (grass) growth as well as maintaining quality habitat for foraging where earthworms play an important role in the diet (see Chapter 2). This would have significant conservation implications, as farmland breeding waders such as the Curlew, are considered one of the most pressing conservation priorities in the UK (Brown *et al.*, 2015; Harris *et al.*, 2020) and options that benefit these species are therefore vital. However, to date, a lack of field-scale trials of liming, under controlled conditions, limits our knowledge of how targeted amending of soil pH might affect wider farmland biodiversity if used as a conservation intervention.

Lime amendment is likely to alter the sward height, composition and livestock carrying capacity as soil pH influences the availability of nutrients to plants e.g.(Yu, Fraser and Evans, 2011; Goulding, 2016;

Holland *et al.*, 2018). Changes to the sward can influence the accessibility of invertebrates to birds which forage for prey visually on and within the soil surface (Whittingham and Markland, 2002; Butler and Gillings, 2004; Devereux *et al.*, 2004). The visibility of invertebrates is likely to be easier in short vegetation with patches of open ground, and dense swards will be harder to penetrate to the soil. Furthermore, the abundance and diversity of invertebrates within the field such as spiders (Araneae) and ground beetles (Carabidae) will also be influenced by changes to the sward structure and field management (Rushton, Topping and Eyre, 1987; Uetz, 1991; Morris, 2000; Perkins *et al.*, 2000; Bell, Wheater and Cullen, 2001; McCracken and Tallowin, 2004; Tichit, Durant and Kernéïs, 2005). For example, web spinning spiders require vegetation of varying heights to help construct and anchor the web (Alderweireldt, 1994; Bell, Wheater and Cullen, 2001; Diehl *et al.*, 2013), whereas other ground hunting spider species or beetles that rely on sight to pursue prey are associated with areas that have patches of bare and short vegetation (Maelfait and De Keer, 1990; Bell, Wheater and Cullen, 2001). Furthermore, the prey of some predatory beetles and spiders consist of soil invertebrates such as springtails (collembola) and earthworms, which may be influenced by changes in the soil brought about by liming (Marcussen, Axelsen and Toft, 1999; Chagnon, Paré and Hébert, 2000; Agustí *et al.*, 2003). Consequently, the use of lime may prove detrimental to farmland birds if they reduce accessibility of prey items, or negatively influence surface-dwelling arthropods, counteracting any positive effects of lime use through the increase in earthworm abundance.

Besides providing a food source for birds, arthropods are also of conservation importance in their own right due to recent declines (Asher *et al.*, 2001; Losey and Vaughan, 2006). Declines in the distribution and abundance of invertebrate groups have been widely reported throughout Europe for arachnids (Aebischer, 1991; Nyffeler and Bonte, 2020), and Coleoptera (Hallmann *et al.*, 2020), as well as other groups such as European grassland butterflies (Habel *et al.*, 2019; van Strien *et al.*, 2019), bees (Goulson, Lye and Darvill, 2008; Potts *et al.*, 2010; Woodcock *et al.*, 2016) and moths (Groenendijk and Ellis, 2011; Randle *et al.*, 2019). This is worrying as invertebrates contribute to a large proportion of the biomass within habitats. For example, beetles are one of the most diverse taxa of insects within grasslands, representing a variety of functional groups and high overall abundance (Woodcock *et al.*, 2008). Arthropods also contribute to nutrient cycling (Yang and Gratton, 2014) and the control of invertebrate pests (spiders: Marc, Canard and Ysnel, 1999; Schmidt *et al.*, 2005; beetles: Sunderland *et al.*, 1987; Kromp, 1999). Spiders, for example, consume between 400–800 million metric tons of invertebrates per annum (Nyffeler and Birkhofer, 2017), while the larvae of ladybirds and hoverflies are major aphid predators, one of the most important pests of cultivated crops in temperate regions (Van Emden and Harrington, 2007). The ecosystem services provided by insects have been estimated at $57 billion annually for the USA alone (Losey and Vaughan, 2006), thus highlighting the importance of understanding how they may be impacted by both current and future management. It is therefore essential that novel AES studies take into account other biodiversity, rather than just focussing on the impact of a single groups of organisms, which could lead to management practices adversely affecting another group, and ultimately the option failing to deliver its biodiversity benefit.

In this chapter, I test whether the use of lime on agricultural grassland influences the sward (and thus accessibility of prey items to foraging birds), and the abundance and diversity of other (non-target) organisms, namely beetles (Coleoptera) and spiders (Araneae). Specifically, I will test 1) if there is an effect of lime treatment on the sward height and structure (e.g. vegetation height and heterogeneity). Here I hypothesise that both vegetation height and heterogeneity will be positively influenced by lime use as compared to control; and 2) if there is an effect of lime treatment on the total abundance of beetles and spiders. Here I hypothesise that abundance will increase on lime treatment. I also hypothesise that 3) the habitat variables of vegetation height and vegetation heterogeneity will be positively associated with total spider abundance. I used spiders and beetles in this study because they are highly abundant and mobile species of grasslands which are likely to respond to local management change affecting soil conditions and vegetation structure, and because they are relatively easy to sample for, as well as being important prey items of many farmland insectivorous birds. The results of this chapter will help to provide empirical evidence on how targeted amending of soil pH might affect wider farmland biodiversity if used as a conservation intervention.

3.3 METHODS

3.3.1 Study area and BACI liming experiment

The study was conducted on a subset of the farms (n = 3; 9 fields) included in the before-after-control-impact (BACI) lime experiment of agricultural grasslands in marginal upland areas described in Chapter 2. For more detailed methods of the BACI lime experiment, please refer to Chapter 2, but briefly this involved a split plot experimental design using 2ha plots where grassland fields of greater than 4ha were divided in half, with one field half receiving a surface dressing of lime ($CaCO_3$; lime treatment) and one field half not receiving any lime (control treatment). Fields were monitored for soil properties (pH, organic matter, moisture) and vegetation (height, heterogeneity) in spring of the baseline year (2014) prior to the application of lime onto treatment field halves. Lime was applied onto the lime treatment plots at an application rate of 4 tonnes ha^{-1} for two consecutive years, once in autumn 2014, and again in autumn 2015. Farms were monitored annually in spring for three years (2015, 2016, and 2017). Normal management of the field was allowed to continue as usual, and livestock were able to move freely between the field plots and graze the entire field allowing for direct comparison between the control and treatment. There were climatic, grazing management, and habitat differences between the three farms with Farm H consisting of a longer and more diverse sward structure with "bumpier" ground than Farm A which had a more uniform and short grass structure (Table 3.1; Figure 3.1D & Figure 3.1E). Farm F had higher mean soil organic matter content and wetter soils than the other two farms (Table 3.1; Figure 3.1C). The three farms in this study were RSPB reserves with tenant graziers.

Table 3.1. Summary table showing site characteristic of the three agricultural grassland farms involved in the BACI lime experiment and arthropod study. Note that Control stands for control plots (not limed); and Treatment for treatment plots (limed). Please note that table is continued on the following page.

Variable	Farm A	Farm F	Farm H
Elevation (m)	188.0	145.0	219.0
Annual Rainfall (mm)[1]	1372.3	1158.9	1254.7
Mean Air Temperature (°C)	7.6	7.6	9.5
Soil Class	Mineral	Organic	Mineral
Soil Type[2]	Brown earth	Peaty Podzols/Gleys	Peaty Gleys
Soil Texture[2]	Sandy loam to silty loam	Sand to sandy loam	Clayey loam to silty loam
Grazing Management[3]	Ewes and lambs in spring, followed by ewes. No winter grazing.	Ewes and lambs in spring, followed by yearling cattle. No winter grazing.	Ewes and lambs in spring, followed by ewes. No winter grazing.
Mean soil $pH_{(Ca2Cl)}$ [4]	Control = 4.8 ± 0.06;	Control = 5.1 ± 0.04;	Control = 4.7 ± 0.08;
	Treatment = 5.0 ± 0.04	Treatment = 5.1 ± 0.04	Treatment = 4.5 ± 0.09
Mean soil organic matter (%)[4]	Control = 10.0 ± 1.02;	Control = 35.8 ± 1.32;	Control = 13.3 ± 0.76;
	Treatment = 8.6 ± 0.51	Treatment = 35.1 ± 1.26	Treatment = 15.0 ± 3.40
Mean soil moisture (%)[4]	Control = 43.1 ± 5.70;	Control = 55.3 ± 1.51;	Control = 46.8 ± 3.53;
	Treatment = 31.9 ± 3.90	Treatment = 61.2 ± 1.62	Treatment = 60.8 ± 5.76
Mean vegetation height (cm)[4]	Control = 7.0 ± 0.51;	Control = 10.6 ± 0.46;	Control = 13.3 ± 0.77;
	Treatment = 7.0 ± 0.31	Treatment = 11.0 ± 0.44	Treatment = 12.3 ± 0.94
Mean vegetation heterogeneity (SD)[4]	Control = 1.8 ± 0.18;	Control = 2.7 ± 0.22;	Control = 3.2 ± 0.42;
	Treatment = 1.9 ± 0.11	Treatment = 3.0 ± 0.28	Treatment = 2.9 ± 0.44

Table 3.1. Continued from previous page.

Mean ground bumpiness (SD)[4]	Control = 43.1 ± 5.70; Treatment = 31.9 ± 3.90	Control = 55.3 ± 1.51; Treatment = 61.2 ± 1.62	Control = 46.8 ± 3.53; Treatment = 60.8 ± 5.76
Lime application autumn 2014	4 tonnes ha $^{-2}$	4 tonnes ha $^{-2}$	4 tonnes ha $^{-2}$
Lime application autumn 2015	4 tonnes ha $^{-2}$	4 tonnes ha $^{-2}$	4 tonnes ha $^{-2}$
Number fields	4	4	1
Number weeks pitfall traps open year^{-1}	8	8	4 (early season only)

[1] Mean annual rainfall and temperature calculated from the three year mean values (2014, 2015 and 2016) to capture the time period of the pitfall trap study from local weather stations accessed from *Scottish Environment Protection Agency (SEPA) Open Source Rainfall data for Scotland* (https://apps.sepa.org.uk/rainfall) and *Met Office MIDAS Open: UK Land Surface Stations Data (1853-current)* (https://catalogue.ceda.ac.uk/uuid/dbd451271eb04662beade68da43546e1) respectively; [2]Soil details accessed from UK Soil Observatory (http://www.ukso.org/); [3]Grazing management from personal communication with the landowner; [4]mean value of a soil core with standard error recorded in spring of the baseline year (2014) of the BACI lime experiment prior to any application of agricultural dust lime onto treatment plots.

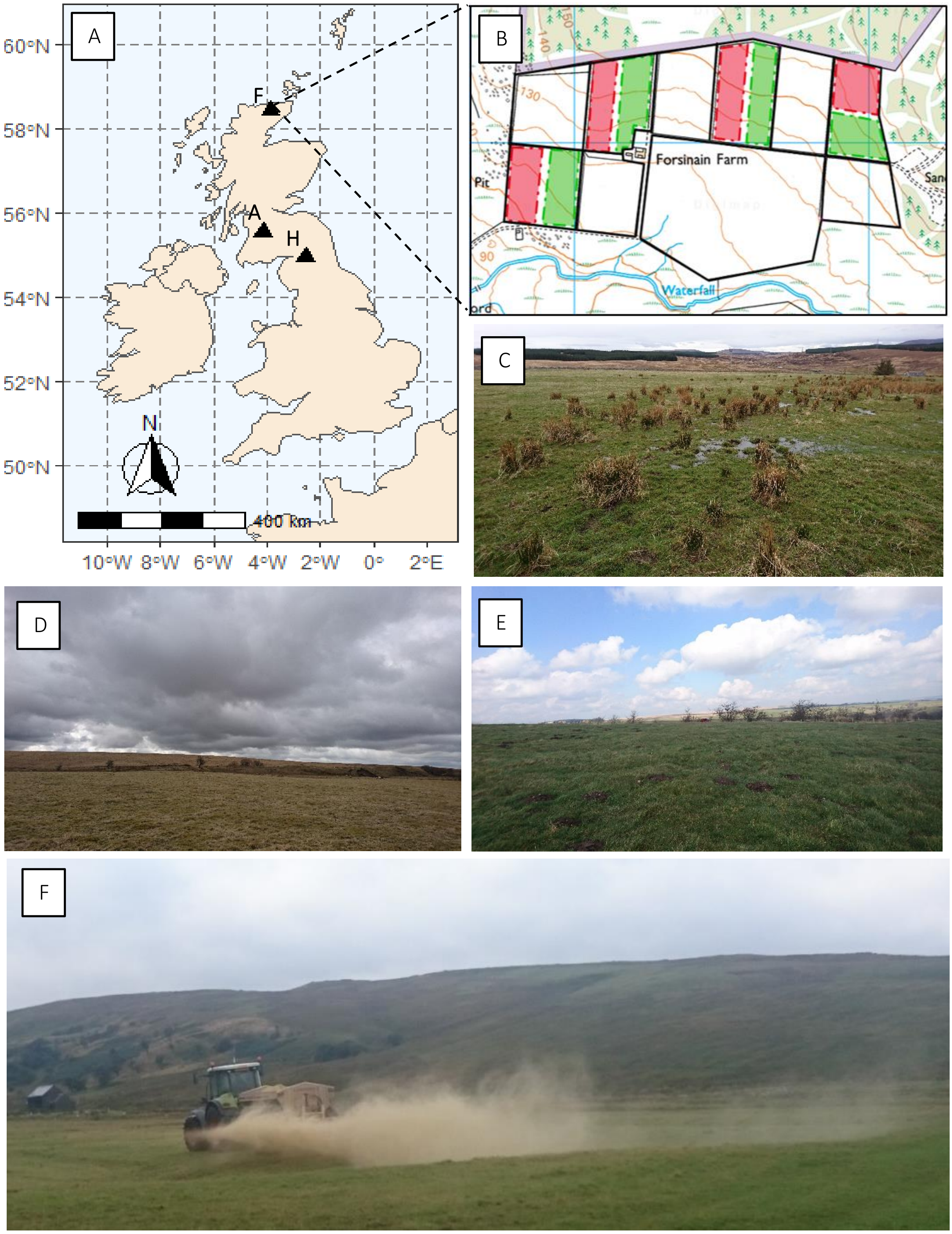


Figure 3.1. **(A)** Map of the United Kingdom (UK) showing the locations of the three upland grassland farms as black triangles where pitfall trapping of above ground active invertebrates were sampled as part of the BACI lime experiment. Farm codes shown in black text on the upper left of the symbol. **(B)** Ordnance Survey map of the study site, Forsinain Farm (Farm F), showing the split-plot experimental design of the lime field trials at four of the in-bye grassland fields. Lime was applied to one half of the field (treatment: green), and the other half of the field received no lime (control: red). **(C - E)** Photos of a typical field at Farm F, Airds Moss (Farm A) and Geltsdale (Farm H) respectively. **(F)** Photo of agricultural dust lime being spread directly onto the field surface (no till) at Farm H.

3.3.2 Arthropod sampling

Surface-active invertebrates were sampled using pitfall traps. These consisted of plastic pint cups containing two inches of a 50% solution of propylene glycol (non-toxic to vertebrates) and water. Three drops of odourless washing up liquid were also added to reduce the surface tension of the solution so invertebrate specimens would sink and be preserved. Two small holes were drilled 2cm from the top of the cup to allow for drainage of rainwater. A plastic tube was also used to create extra stability to the hole that the cup was placed into. Pitfall traps were buried up to their rims in the soil and had a labelled metal "hat" pushed to 1cm from the soil surface to help reduce rainfall into the trap, reduce predation from birds and exclude small mammals (Figure 3.2A & Figure 3.2C). The location of pitfall traps followed a stratified, random design generated using a sampling design tool in ArcGIS version 10.2.2 (Buja and Menza, 2013), with ten traps installed in each of the 2ha treatment plots (control and lime treatment) and the location recorded by GPS (Garmin Etrex Vista HCx).

Pitfall trapping was conducted for eight weeks on two consecutive years (2015 and 2016). The eight weeks of sampling was separated into two four week periods each year, with pitfall traps running for four weeks between March and April ("early season"), and for a further four weeks in May and June ("late season"). Pitfall trapping at Farm H was only conducted for the first four weeks (early season). Pitfall traps were checked and emptied into sealable pots every two weeks to ensure that traps did not flood out with rainwater and to clean the traps and re-fill with solution, resulting in two collections per season. However, in formal analysis data was pooled for each pitfall trap at the season level resulting in a sample size of 180 pitfall traps per year in the early season (i.e. 360 pitfall traps over the two years of sampling), and 160 pitfall traps per year in the late season (i.e. 320 pitfall traps over the two years).

The invertebrates caught were initially identified to order level, with beetle and spiders further identified to species level by two taxonomic experts (beetles: Llyod Garvey; spiders: Dave Beaumont) and preserved in 70% ethanol. Only full specimens with a head were counted to prevent double counting. Beetles were identified to species using Joy (1932) and Forsythe (2000). For beetle specimens, larvae were not identified to species level but were included in total abundance of beetles. The carabids *Pterostichus nigrata* and *Pterostichus rhaeticus* were recognised as separate species in the 1980s (Koch, 1986). However, they can be difficult to separate requiring dissection so were recorded as a *Pterostichus nigrata/rhaeticus* species complex in the dataset. Spiders were identified to species using Roberts (1987) and the sex (male, female) of individuals was also recorded. The separation of individuals into either *Dicymbium nigrum* or *Dicymbium brevisetosum* is based on male characteristics so for females it is not possible to distinguish between species. As a result, the species complex of *Dicymbium nigrum/brevisetosum* was used in the dataset. Unidentifiable spider specimens were included in total spider abundance and were predominantly of immature spiders.

The use of pitfalls traps was considered by the Ethics committee at the University of Stirling and approved prior to the start of fieldwork.

3.3.3 Soil properties and habitat variables

The same sampling location as the pitfall traps were used to record soil properties and habitat variables. Variables were sampled over the four years of the BACI lime experiment (2014, 2015, 2016 and 2017). The soil properties of pH and soil organic matter content (SOM) were sampled and analysed as described in chapter 2, with the hole created from the soil corer (10.5cm diameter to a depth of 10cm; Figure 3.2B) used for the pitfall trap. Measurements of pH and SOM were therefore only taken once each year in March/April ("early season"). For all other habitat variables, measurements were recorded within a quadrat (50cm x 50cm). For the years 2014 and 2017 measurement were taken twice, once in the early season and once in the late season. In years 2015 and 2016 when pitfall trapping was being conducted, measurements were taken on every visit resulting in three recordings per season (e.g. once when pitfall traps installed, once when pitfall traps checked after two weeks, and once when pitfall traps closed for the season/removed). The mean value per season (e.g. early and late) was used in analysis at the pitfall trap level.

Three readings of soil moisture were taken using a soil moisture meter probe (HH2 moisture meter, SM200 moisture sensor, Delta-T Devices, Cambridge, England) and the mean value recorded. Vegetation height was measured using a ruler to the nearest cm at five locations within the quadrat (four corners and centre) and the mean value taken similar to the 'sward stick method' (Barthram, 1986) where the height of the tallest piece of green non-flowering vegetation was recorded. The standard deviation (SD) of vegetation height was used as an index of vegetation heterogeneity where a higher value indicated vegetation height which varied more from the mean, while low values closer to zero indicated a uniform vegetation height. The percentage of bare ground within the quadrat was estimated by eye to the nearest 5%. The ground bumpiness (i.e. how smooth or bumpy the ground was) was measured using a handmade 'bumpimeter' which consisted of a 50cm length of wood with six equally spaced holes and two wooden legs that supported the device 10cm above ground level (Figure 3.2D). A peg marked with 1cm gradations was pushed through each hole in turn until it reached the ground and the distance between the ground surface and wood platform was recorded to the nearest 1cm (Sheldon, 2002). The standard deviation of the six distances measured was used as an index of variability in bumpiness (i.e. ground micro-topography).

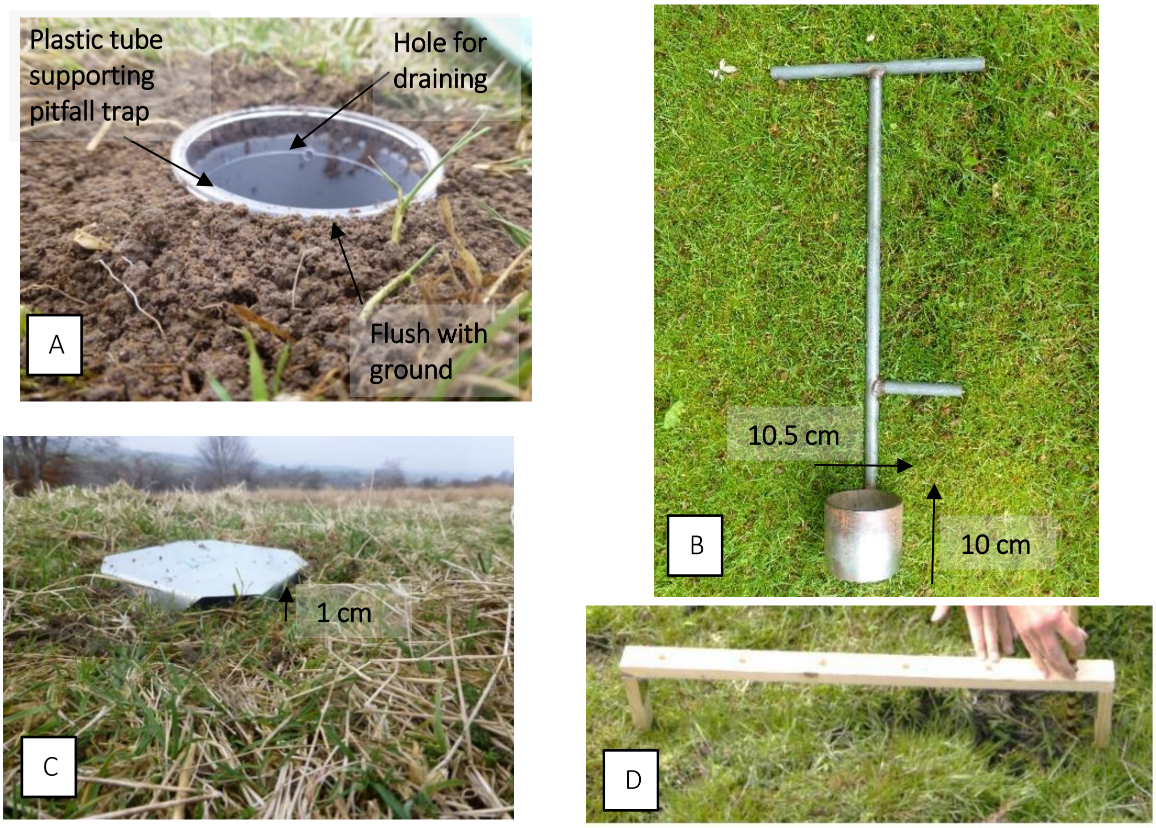


Figure 3.2. (A) Photo of a pitfall trap cup flush with the ground surface, with plastic tube supporting the hole that the trap sits into visible. **(B)** Photo of the soil corer (10cm depth and 10.5cm diameter) used to extract soil for pH and organic matter content and to create the hole for the pitfall trap. **(C)** Photo showing one of the pitfall traps installed at Farm H with its metal "hat" placed over the trap to reduce vegetation debris, rain water and small mammals from getting into the trap. **(D)** Photo of the handmade '*bumpimeter*' used to measure variation in ground bumpiness (i.e. ground micro-topography).

Table 3.2. Summary table of **(A)** response, and **(B)** predictor variables used to investigate the relationship of total beetle and spider abundance collected from pitfall traps with soil properties, vegetation, and lime treatment in linear mixed models (LMMs) at the pitfall trap scale.

Variable name	Unit of replication	Method of data collection
(A) Response:		
Beetle abundance	Pitfall trap	Pitfall trapping for four weeks per season (early and late) with identification by use of light microscope and keys (range: 0 – 62, mean: 7.2)
Spider abundance	Pitfall trap	Pitfall trapping for four weeks per season (early and late) with identification by use of light microscope and keys (range: 0 – 38, mean: 4.8)
(B) Predictor:		
Vegetation height	Pitfall trap	Measured to nearest cm with a ruler from five locations within a quadrat (range: 1.0 – 28.0 cm, mean: 5.8 cm)
Vegetation heterogeneity	Pitfall trap	Standard deviation of the mean vegetation height from the five measurements recorded from the pitfall trap (range: 0 – 8.5, mean: 1.8)
Ground bumpiness (i.e. ground micro-topography)	Pitfall trap	Measured using a "bumpimeter" with six values recorded of the distance from a fixed point to the ground, and the standard deviation of the mean used as an index for how flat/bumpy the ground was (range: 0.0 – 3.0, mean: 0.7)
Bare ground	Pitfall trap	Estimated as a proportion be eye within a quadrat (range: 0 – 5%; mean: 0)
Soil pH	Pitfall trap	Measured in the laboratory using 0.01M $CaCl_2$ annually (range: 3.4 – 5.6, mean: 4.5)
SOM	Pitfall trap	Measured in the laboratory from loss of ignition (LOI) annually (range: 2.5 – 93.0%, mean: 25.2%)
Soil moisture	Pitfall trap	Recorded in the field using soil moisture probe (range: 15.4 – 90.0%, mean: 66.0%)
Treatment	Pitfall trap	Recorded from the lime field trials (two levels: Lime treatment; Control (non-limed).

3.3.4 Statistical analysis

3.3.4.1 Effects of lime treatment on vegetation height and heterogeneity

To determine whether lime use had an effect on the sward (hypothesis 1), a Linear Mixed Model approach was used. Separate models were fitted by season (early or late) and for each of the following response variables: vegetation height and vegetation heterogeneity (Table 3.1 & Table 3.3). The response was log transformed + 1 prior to analysis and fitted with a Gaussian error structure and identity link with the unit of replication at the quadrat scale. Sample size for the early season models was three farms (n = 9 fields) and for the late season models was two farms (n = 8 fields) over the four years of the BACI lime experiment (2014, 2015, 2016 and 2017) resulting in 720 and 640 observations for the early and late models respectively. Field identity was included as a random effect (grouping) within the model, and Year included as a fixed effect as well as the interaction term between Year and Treatment. Ground bumpiness and the soil properties of moisture (%) and organic matter (%) were fitted as continuous covariates. Soil pH was not fitted as a predictor variable in these model as lime treatment was shown to have a significant effect on pH (chapter 2) and therefore pH could not be included as an independent covariate in the LMMs testing the effects of lime treatment on the response.

3.3.4.2 Arthropod summary community descriptors and model-based ordination on the effects of lime treatment on species community composition

Several summary descriptors and indices were used to provide an overview of the beetle and spider community separately. These comprised of the total number of individuals recorded (abundance), species richness and Simpson's Diversity Index (D) and Simpson's Measure of Evenness (E) of the community (Magurran, 2004) calculated at the treatment plot scale (summed data from across the ten pitfall traps per treatment plot) and by season (summed data over the four weeks of sampling in both the early and late season).

Unconstrained model-based ordination was used to visualise the beetle and spider community level response to lime treatment and to farm site by using *non-metric multidimensional scaling* (NMDS) with a Wisconsin double standardization applied to square root transformed data in order to reduce influence of large counts (Bray and Curtis, 1957). A separate NMDS analysis was run for beetles and spiders, using pitfall trap data at the treatment plot scale and by season in a species matrix. The NMDS goodness of fit was determined by its Stress value using a threshold of 0.2 as values above this are considered poor and potentially uninterpretable biologically (Kruska, 1964; Clarke, 1993; Dexter, Rollwagen-Bollens and Bollens, 2018), alongside other complementary criteria to evaluate the ordination fit such as the Shepherd plot as the Stress value can be influenced by the number of samples, species and dimensions in the ordination (Clarke, 1993; Vanderklift, Ward and Jacoby, 1996; Zuur, Ieno and Smith, 2007; Dexter, Rollwagen-Bollens and Bollens, 2018). To identify intrinsic species responsible for driving the location within multivariate ordination space Monte Carlo permutation tests (1000 permutations) were used to determine the significance of each species. Where species had a significance of <0.05 their positions in multivariate ordination space were superimposed onto the bi-plot as arrows representing the direction and strength of that species.

3.3.4.3 Effects of lime treatment on total spider and beetle abundance

To determine whether lime use affected the total abundance of spiders and beetles (hypothesis 2), LMMs were used with a Gaussian error structure and identity link on log transformed + 1 abundance data. Separate models were run for beetle and spider data and by early and late season resulting in a total of four models. The unit of replication was at the pitfall trap level, with data summed over the four weeks of sampling by season. Sample size for the early season models was three farms (n = 9 fields) over two years (2015, 2016) and for the late season models was two farms (n = 8 fields) over the two years (2015, 2016) resulting in 360 and 320 observations for the early and late models respectively. Field identity was included as a random effect (grouping) within the model, and Year included as a fixed effect as well as the interaction term between Year and Treatment. The soil properties of moisture (%) and organic matter (%) were fitted as continuous covariates, as were vegetation height, vegetation heterogeneity and ground bumpiness. Soil pH was not fitted as a predictor variable in these model as lime treatment was shown to have a significant effect on pH (chapter 2). These models also allowed for the testing of hypothesis 3) that vegetation height and heterogeneity would be positively associated with beetle and spider abundance.

3.3.4.4 Model selection, validation and post-hoc tests

All statistical analysis was conducted in R version 3.5.1 (R Core Team, 2019) with data exploration carried out following the protocol described in Zuur, Ieno and Smith (2007). Summary community descriptors were calculated using the Vegan package in (Oksanen *et al.*, 2020) and ordination analysis implemented through the MVAbund package (Wang *et al.*, 2012). Multicollinearity of numerical predictors was assessed using Pearson correlation coefficients and the correlation between variables was considered acceptable if the coefficient was less than 0.7 (Dormann *et al.*, 2013). In this study, correlation coefficients were $\leq$0.6 with the exception of vegetation height and vegetation heterogeneity which had a value of 0.7 (Figure A7 in Appendix). Models were therefore fitted twice with either vegetation height or vegetation heterogeneity in the full model, but not both, and the minimal adequate model (MAM) found as described below. Linear mixed effects models (LMMs) were implemented using the lme4 package (Bates *et al.*, 2015). Model validation was assessed using graphical visualisation of the model residuals to check for normality and homogeneity of variance (Zuur, Ieno and Smith, 2007; Zuur *et al.*, 2009; Hartig, 2020). The marginal R^2 and conditional R^2 values were calculated following Nakagawa and

Schielzeth (2013) using the *rsquared* function in the piecewiseSEM package (Leftcheck, 2016) which quantifies the proportion of variance explained by the fixed effects without considering the random effects (marginal R^2) as well as the proportion of variance explained by the full model while also considering the random effects (conditional R^2). Thus R^2 values act as a measure of model goodness of fit and were used when comparing models in the model building phase. Covariates were standardised and centered with a mean of zero prior to analysis (Schielzeth, 2010). Estimated marginal means (also known as least-squares means) were calculated for factors using the emmeans package where appropriate (Lenth, 2020). The minimal adequate models (MAM) were obtained using a manual stepwise backwards selection tested by likelihood ratio tests (LRT) of nested models, retaining all variables that were significant at the 5% level.

3.4 RESULTS

3.4.1 Effects of lime treatment on vegetation height and heterogeneity

There was no evidence for an effect of treatment, nor of an interaction effect of treatment with year on vegetation height, nor on vegetation heterogeneity in either the early or late season models (Table 3.3, Table 3.4). However, vegetation height varied between years in both the early and late season models. Vegetation heterogeneity also varied between years in the late season model (Table 3.5). There was also a positive relationship between vegetation height and soil organic matter, and an inverse relationship with soil moisture in both the early and late season models. For vegetation heterogeneity there was evidence of a positive relationship with ground bumpiness in both of the season models. The models explained variation in vegetation properties well (Height early: $R^2_M = 0.460$; $R^2_C = 0.589$; Height late: $R^2_M = 0.335$; $R^2_C = 0.452$; Heterogeneity early: $R^2_M = 0.536$; $R^2_C = 0.549$; Heterogeneity late: $R^2_M = 0.388$; $R^2_C = 0.415$).

Table 3.3 Summary table showing the final Linear Mixed Models used in analysis to test the effect of lime treatment on vegetation.

Model response variable	Type	Unit replication	Link	Error Structure	Random Effect	Fixed Effects
Log(Height) – early season	Continuous	Quadrat	Identity	Gaussian	Field identity	Year + SOM + Moisture + Heterogeneity
Log(Height) – late season	Continuous	Quadrat	Identity	Gaussian	Field identity	Year + SOM + Moisture + Heterogeneity
Log(Heterogeneity +1) – early season	Continuous	Quadrat	Identity	Gaussian	Field identity	Year + Height + Bumpiness
Log(Heterogeneity +1) – late season	Continuous	Quadrat	Identity	Gaussian	Field identity	Height + Bumpiness

Table 3.4. Summary table showing the minimal adequate models for the LMMs investigating the relationship between lime treatment and vegetation (height and heterogeneity) at the quadrat scale by season (early and late). The sample size for early season models was three farms (n = 9 fields) and for the late season models was two farms (n = 8 fields) across the four survey years (2014, 2015, 2016, 2017) resulting in a total of 720 and 640 observations respectively. Note that covariates were scaled and centred prior to analysis for ease of coefficient interpretation. Table shows results of the species minimum adequate models with significance levels denoted by *'s where: * ≤ 0.05; ** ≤ 0.01; *** ≤ 0.001.

	Parameter estimates SE (test statistic with significance*) by predictor variable (type; df)					
Model	**Year (factor; df = 3)**	**SOM (covariate; df = 1)**	**Moisture (covariate; df = 1)**	**Height (covariate; df = 1)**	**Heterogeneity (covariate; df = 1)**	**Bumpiness (covariate; df = 1)**
Height – early season	- (F = 11.12***)	0.11 ± 0.02 (z = 4.57***)	-0.11 ± 0.02 (t = -5.51***)	-	0.26 ± 0.02 (t = 15.11***)	-
Height – late season	- (F = 3.62*)	0.08 ± 0.03 (z = 3.01**)	-0.08 ± 0.03 (t = -3.32***)	-	0.22 ± 0.02 (t = 13.74***)	-
Heterogeneity – early season	- (F = 3.48*)	-	-	0.25 ± 0.01 (t = 18.27***)	-	0.04 ± 0.01 (t = 2.94**)
Heterogeneity – late season	-	-	-	0.21 ± 0.01 (t = 16.48***)	-	0.03 ± 0.01 (t = 2.62**)

Table 3.5. Estimated marginal means (EMMs) for significant predictor factors in the LMMs investigating the relationship between lime treatment and vegetation (height and heterogeneity). Values were computed with covariates set to mean values, and by averaging together predictions on the linear-predictor scale and then back transforming and taking away 1 from the log +1 transformed response used in analysis. A 95% confidence level was used. A pairwise tukey method was used for comparing EMMs with significant p values shown in bold.

	Marginal mean estimate ha^{-1}	Lower CI	Upper CI	Pairwise comparison	Test Statistic	p value
Vegetation height – early season						
Year						
2014	5.05	4.35	5.87	2014 – 2015	t =3.40	**0.0040**
2015	4.39	3.78	5.10	2014 – 2016	t = 1.49	0.4422
2016	4.71	4.06	5.47	2014 – 2017	t = -2.45	0.0690
2017	4.58	4.81	6.49	2015 – 2016	t = -1.75	0.3006
				2015 – 2017	t = 5.70	**<0.0001**
				2016 – 2017	t = -3.62	**0.0019**
Vegetation height – late season						
Year						
2014	5.31	4.62	6.11	2014 – 2015	t = 0.41	0.9762
2015	5.21	4.53	6,05	2014 – 2016	t = -0.67	0.9089
2016	5.53	4.76	6.42	2014 – 2017	t = -2.66	**0.0401**
2017	5.98	5.21	6.89	2015 – 2016	t = -0.94	0.7841
				2015 – 2017	t = -3.02	**0.0140**
				2016 – 2017	t = -1.51	0.4349
Vegetation heterogeneity – early season						
Year						
2014	3.13	2.83	3.49	2014 – 2015	t = 8.94	**<0.0001**
2015	2.16	1.97	2.39	2014 – 2016	t = 7.99	**<0.0001**
2016	2.20	1.99	2.44	2014 – 2017	t = 11.43	**<0.0001**
2017	1.88	1.70	2.05	2015 – 2016	t = -0.45	0.9692
				2015 – 2017	t = 3.97	**0.0005**
				2016 – 2017	t = 4.49	**0.0001**

3.4.2 Arthropod summary community descriptors and model based ordination on the effects of lime treatment on community composition

In total, 5,168 beetles were recorded across the three farms (n = 9 fields), from ten families consisting of 97 species. A total of 440 adults were not possible to identify to species level and were instead separated into 17 different species complexes (e.g. to subfamily or genus) with the majority of these complexes (n = 14) consisting of only 1 - 3 individuals. However, the remaining three groups are of note; 17 individuals recorded as Lathrobium species (a genus of the family Staphylinidae; rove beetles); 292 individuals recorded as Aleocharinae species (a subfamily of the family of Staphylinidae; rove beetles); and 110 individuals recorded as *Pterostichus nigrata/rhaeticus* (a species complex of the family Carabidae; ground beetle). In general, total beetle abundance was higher in the late season (late = 4,090; early = 1,078), as was the abundance of beetle larvae (late = 927; early = 365). In terms of species community composition, Simpson's diversity indexes were high and similar between treatment, season, and farms. However, the measure of species evenness was low (albeit higher in the late season) with a few species dominating the community primarily from the Staphylinidae (Table 3.6 & Table 3.7; Figure 3.3). Species richness also varied between farms with 68 species recorded in Farm A, compared to only 25 at Farm H (although this farm was only surveyed in the early season). There was a clear pattern in multidimensional space for fields within a farm to be more similar to each other than fields from other farms regardless of the season sampled (shown by colour in the NMDS biplot; Figure 3.4A & Figure 3.4B); however there did not appear to be any effect of treatment on beetle communities as there was a large amount of overlap between the cloud of points between treatments.

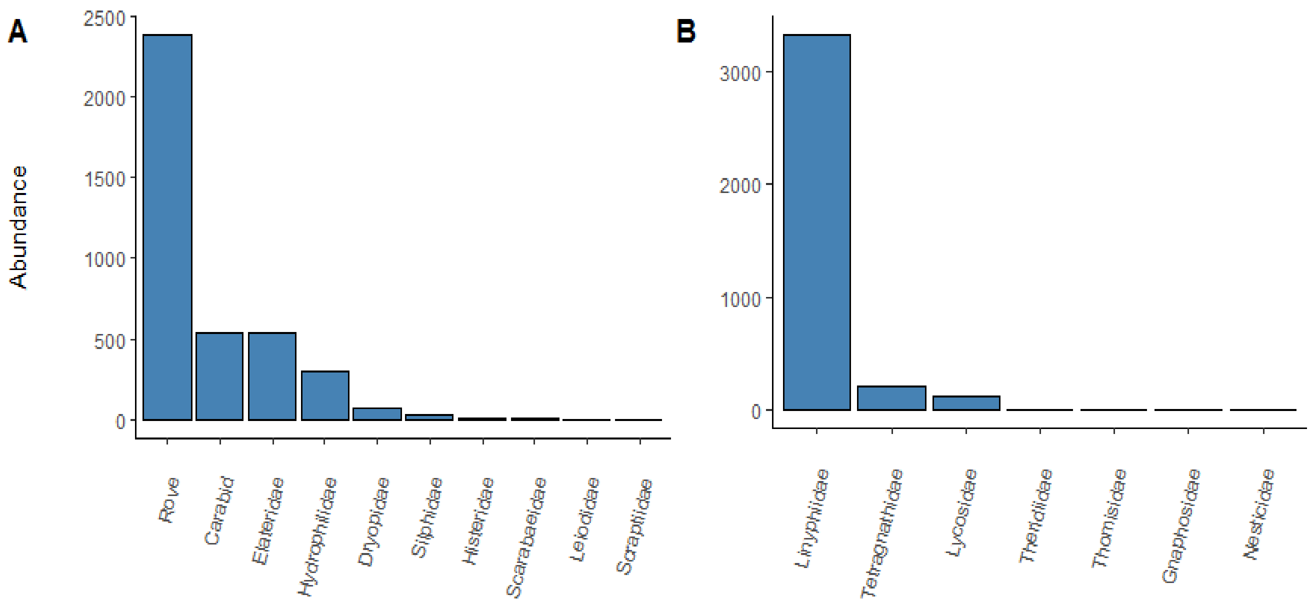


Figure 3.3. Bar plot of the raw data of the total abundance of **A)** beetles and **B)** spiders by family.

A total of 4,262 spiders were recorded, comprising of 69 species across seven families of which 1,839 were female, 1,808 male and 616 immature (Table 3.5). Approximately 14% of spiders were unidentifiable and consisted almost entirely of immature individuals. In general, total spider abundance was greater in the late season, compared to the early season (late = 2,692; early = 1,570). In terms of species community composition, the data was dominated by a few species with high abundances, primarily from the Linyphiidae family (money spiders; Table 3.6; Figure 3.3). Spider species richness ranged between 21 and 56 species per farm, although Farm H was only sampled in the early season. There was no effect of treatment on either Simpson's diversity index or on Simpson's measure of evenness, and these measures of community composition were also similar between farms and season (Table 3.7). There was a clear pattern in multidimensional space for fields within a farm to be more similar to each other than fields from other farms regardless of the season sampled (shown by colour in the NMDS biplot; Figure 3.4C & Figure 3.4D); however there was no pattern observable with treatment.

Table 3.6. Summary table of the five most abundant species of beetles and spiders recorded over the two year sampling period of pitfall trapping by site. The total percentage/dominance of each species per site and within the dataset is shown italicised and in brackets. Please note that these values do not add up to 100% as many other species were identified which are not displayed here.

	Family	Farm A	Farm F	Farm H	Total
(A) Beetles					
Tachinus signatus	Staphylinidae	541 (*22.5%*)	1,086 (*41.7%*)	11 (*6.6%*)	1,638 (*31.7%*)
Hypnoidus riparius	Elateridae	303 (*12.6%*)	215 (*8.3%*)	6 (*3.6%*)	524 (*10.1%*)
Megasternum concinnum	Hydrophilidae	57 (*2.4%*)	234 (*9.0%*)	2 (*1.2%*)	293 (*5.7%*)
Aleocharinae Species	Staphylinidae	146 (*6.1%*)	128 (*4.9%*)	18 (*10.8%*)	292 (*5.7%*)
Philonthus laminatus	Staphylinidae	3 (*0.1%*)	119 (*4.6%*)	-	122 (*2.4%*)
Abundance of larvae	-	781 (*32.6%*)	473 (*18.2%*)	38 (*22.9%*)	1,292 (*25.0%*)
Total abundance (inc. larva)	-	2,399	2,603	166	5,168
Total species richness	-	68	56	25	97
Simpson diversity index (D)	-	0.82	0.77	0.88	0.82
Simpson evenness index (E)	-	0.03	0.04	0.09	0.02
(B) Spiders					
Oedothorax retusus	Linyphiidae	340 (*21.8%*)	248 (*9.8%*)	3 (*1.6%*)	591 (*13.9%*)
Oedothorax fuscus	Linyphiidae	87 (*5.4%*)	441 (*17.5%*)	-	528 (*12.4%*)
Tiso vagans	Linyphiidae	332 (*21.3%*)	141 (*5.6%*)	2 (*1.1%*)	475 (*11.1%*)
Erigone dentipalpis	Linyphiidae	161 (*10.3%*)	268 (*10.6%*)	6 (*3.2%*)	435 (*10.2%*)
Dicymbium nigrum/brevisetosum	Linyphiidae	21 (*1.3%*)	187 (*7.4%*)	29 (*15.7%*)	237 (*5.6%*)
Unidentifiable specimens	-	52 (*3.3%*)	529 (21.0*%*)	19 (*10.3%*)	600 (*14.1%*)
Total abundance (inc. unidentifiable)	-	1,559	2,518	185	4,262
Total species richness	-	56	52	21	69
Simpson diversity index (D)	-	0.88	0.89	0.88	-
Simpson evenness index (E)	-	0.04	0.05	0.11	-

Table 3.7. Summary table of **(A)** beetles; and **(B)** spiders recorded (abundance), species richness and Simpson's Diversity Index (D) and Simpson's measure of evenness of the community (E) for each field by treatment with breakdown by season in brackets (early; late). Sample size was three farms (n = 9 fields) across two years (2015 and 2016) resulting in 180 observations for the early season; and two farms (n = 8 fields) across two years (2015 and 2016) resulting in 160 observations for the late season. Please note that table is continued on the following page.

Location	Farm Code	Field ID	Abundance		Species richness		Simpson's D		Simpson's E	
			Lime treatment	Control (not limed)	Lime treatment	Control (not limed)	Lime treatment	Control (not limed)	Lime treatment	Control (not limed)
(A) Beetles										
North Scotland	F	F8	302 (99; 203)	302 (68; 234)	23 (11; 21)	22 (10; 18)	0.96 (0.67; 0.74)	0.96 (0.64; 0.76)	0.11 (0.16; 0.10)	0.11 (0.17; 0.11)
		F9	413 (79; 334)	304 (64; 240)	15 (10; 19)	21 (10; 16)	0.96 (0.61; 0.64)	0.94 (0.71; 0.67)	0.11 (0.17; 0.09)	0.14 (0.19; 0.11)
		F10	189 (42; 147)	392 (63; 329)	23 (15; 19)	17 (14; 13)	0.96 (0.85; 0.84)	0.94 (0.89; 0.67)	0.11 (0.15; 0.12)	0.14 (0.16; 0.14)
		F11	303 (55; 248)	398 (92; 306)	21 (11; 17)	18 (10; 16)	0.96 (0.62; 0.69)	0.95 (0.69; 0.68)	0.12 (0.15; 0.11)	0.14 (0.18; 0.12)
West Scotland	A	A1	272 (46; 226)	237 (27; 210)	20 (*9; 15*)	27 (10; 24)	0.95 (0.80; 0.78)	0.96 (0.86; 0.81)	0.12 (0.22; 0.14)	0.09 (0.22; 0.09)
		A2	353 (72; 281)	315 (43; 272)	25 (14; 22)	22 (*8; 21*)	0.96 (0.85; 0.80)	0.96 (0.84; 0.77)	0.10 (0.16; 0.10)	0.11 (0.26; 0.10)
		A3	211 (25; 186)	123(17; 106)	22 (*8; 19*)	23 (12; 16)	0.96 (0.82; 0.75)	0.96 (0.91; 0.79)	0.11 (0.25; 0.11)	0.11 (0.19; 0.13)
		A4	486 (79; 407)	402 (41; 361)	31 (18; 28)	26 (14; 23)	0.97 (0.82; 0.78)	0.96 (0.86; 0.77)	0.08 (0.12; 0.08)	0.10 (0.16; 0.09)
North England	H	H18	67 (*67; 0*)	99 (99; 0)	15 (*15; 0*)	21 (*21; 0*)	0.83 (0.83; 0.00)	0.90 (0.90; 0.00)	0.16 (0.14; 0.00)	0.12 (0.11; 0.00)
Totals			2,596	2,572	75	75	0.82	0.82	0.03	0.03
			(*564; 2,032*)	(*514; 2,058*)	(*43; 62*)	(*47; 60*)	(*0.83; 0.80*)	(*0.87; 0.78*)	(*0.05; 0.04*)	(*0.05; 0.04*)

Table 3.7. Continued from previous page.

Location	Farm Code	Field ID	Abundance		Species richness		Simpson's D		Simpson's E	
			Lime treatment	Control (not limed)	Lime treatment	Control (not limed)	Lime treatment	Control (not limed)	Lime treatment	Control (not limed)
(B) Spiders										
North Scotland	F	F8	142 (*62; 80*)	136 (*59; 77*)	21 (59; 77)	19 (59; 77)	0.90 (0*.88; 0.87*)	0.84 (0*.75; 0.87*)	0.11 (0*.15; 0.15*)	0.11 (0*.21; 0.15*)
		F9	143 (*52; 91*)	164 (64; 100)	23 (59; 77)	26 (59; 77)	0.86 (0*.85; 0.84*)	0.89 (0*.84; 0.86*)	0.10 (0*.16; 0.13*)	0.09 (0*.14; 0.11*)
		F10	720 (288; 432)	697 (232; 465)	24 (17; 20)	33 (22; 26)	0.86 (0*.85; 0.78*)	0.88 (0*.86; 0.84*)	0.09 (0*.12; 0.10*)	0.07 (0*.10; 0.08*)
		F11	277 (75; 202)	239 (88; 151)	24 (15; 20)	23 (13; 19)	0.83 (0*.81; 0.76*)	0.86 (0*.86; 0.83*)	0.09 (0*.13; 0.10*)	0.09 (0*.17; 0.11*)
West Scotland	A	A1	303 (125; 178)	150 (*64; 86*)	27 (17; 21)	20 (13; 14)	0.84 (0*.79; 0.85*)	0.84 (0*.82; 0.83*)	0.08 (0*.12; 0.10*)	0.11 (0*.16; 0.15*)
		A2	184 (46; 138)	137 (31; 106)	25 (12; 22)	30 (11; 25)	0.86 (0*.85; 0.83*)	0.90 (0*.80; 0.90*)	0.09 (0*.17; 0.10*)	0.08 (0*.19; 0.09*)
		A3	106 (*27; 79*)	143 (36; 107)	20 (11; 13)	17 (*9; 13*)	0.86 (0*.86; 0.83*)	0.83 (0*.86; 0.80*)	0.11 (0*.20; 0.16*)	0.12 (0*.22; 0.15*)
		A4	262 (73; 189)	275 (69; 206)	32 (19; 26)	30 (17; 21)	0.86 (0*.84; 0.85*)	0.83 (0*.88; 0.78*)	0.07 (0*.12; 0.09*)	0.07 (0*.13; 0.09*)
North England	H	H18	78 (*73; 0*)	106 (*106; 0*)	15 (15; 15)	17 (10; 14)	0.87 (0*.87; 0.00*)	0.87 (0*.87; 0.00*)	0.14 (0*.14; 0.00*)	0.13 (0*.13; 0.00*)
Totals			2,215	2,047	54	61	0.91	0.91	0.04	0.04
			(821; 1,394)	(749; 1,298)	(37; 45)	(39; 50)	(0.90; 0.88)	(0.91; 0.88)	(0.05; 0.05)	(0.06; 0.06)

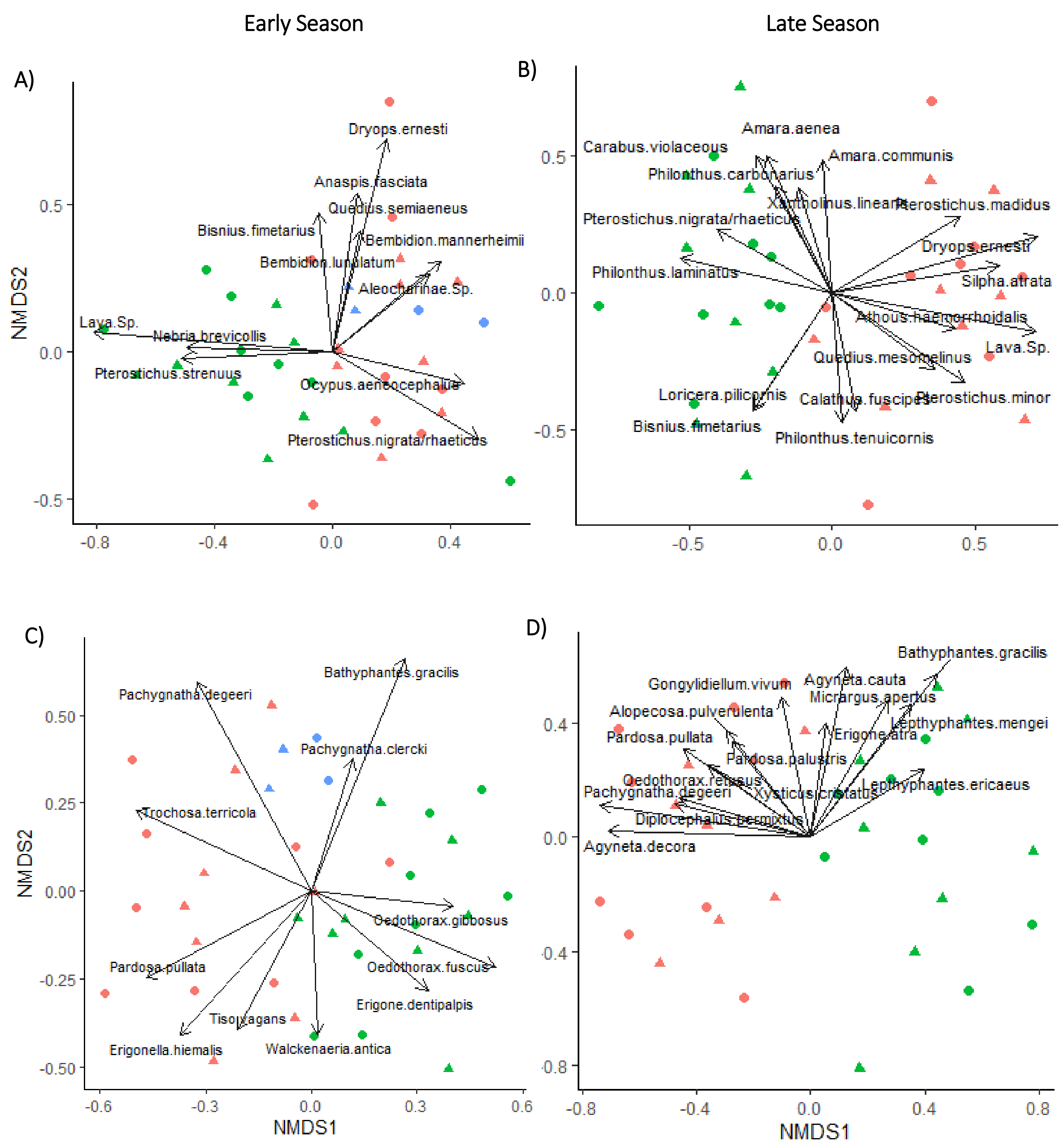


Figure 3.4. NMDS biplot representing graphically the community composition of beetles (A and B) and spiders (C and D) in multidimensional space by season sampled. Each data point represents pitfall trap data from a four week sampling period (e.g. early or late) pooled at the treatment level. Data points closer together on the biplot are more similar to each other in terms of species community composition than data points further apart. Treatment is represented by shape where; circles = control (non-limed); and triangles = lime treatment. Location of the plot is represented by farm as a colour where; red = Farm A; green = Farm F; and blue = Farm H. Intrinsic species significant at the 0.05 level driving the position in space shown as arrows where direction and length represent direction in space and strength respectively. Sample size for early season is three farms (n = 9 fields) across two years (2015 and 2016) resulting in 36 observations. Sample size for late season is two farms (n = 8 fields) across two years (2015 and 2016) resulting in 32 observations. Stress values for the NMDS were < 2.0.

3.4.3 Effects of lime treatment on total spider and beetle abundance

There was no effect of liming or of year on total spider abundance in the early season (Table 3.8). However, there was evidence of an inverse relationship of spider abundance in the early season with soil organic matter content (SOM): soils with low SOM of 20% had, on average, 5.75 spiders per pitfall trap (95% CI: 4.44 – 7.46), whereas in very peaty soils with 80% SOM the estimated marginal mean was only 3.94 spiders per pitfall trap (95% CI: 2.64 – 5.87). In the late season, there was weak evidence of an effect of year which varied by treatment on spider abundance: within year abundance was similar on the limed and unlimed control plots, but between years there was a marginally non-significant increase in abundance of spiders on the limed plots in 2016 compared with 2015 ($t = 2.21$; $p = 0.1$; Table 3.10). There was also evidence of an inverse relationship of spider abundance in the late season with soil moisture (%) and evidence of a positive association with vegetation height. With a soil moisture of 20%, spider abundance was estimated at 12.3 (95% CI: 7.54 – 33.78), whereas in very wet soils with a soil moisture of 80% the abundance of spiders per pitfall trap declined by more than 40% to 7.03 (95% CI: 4.62 – 10.59). Pitfall traps in short vegetation with a height of 4cm had an estimated average spider abundance of 8.12 per pitfall trap (95% CI: 5.42 – 12.30), whereas in longer vegetation with a height of 12cm spider abundance per pitfall trap was higher with 11.94 (95% CI: 7.32 – 19.49). The proportion of variance in spider abundance explained by these models are lower when only the fixed effects are

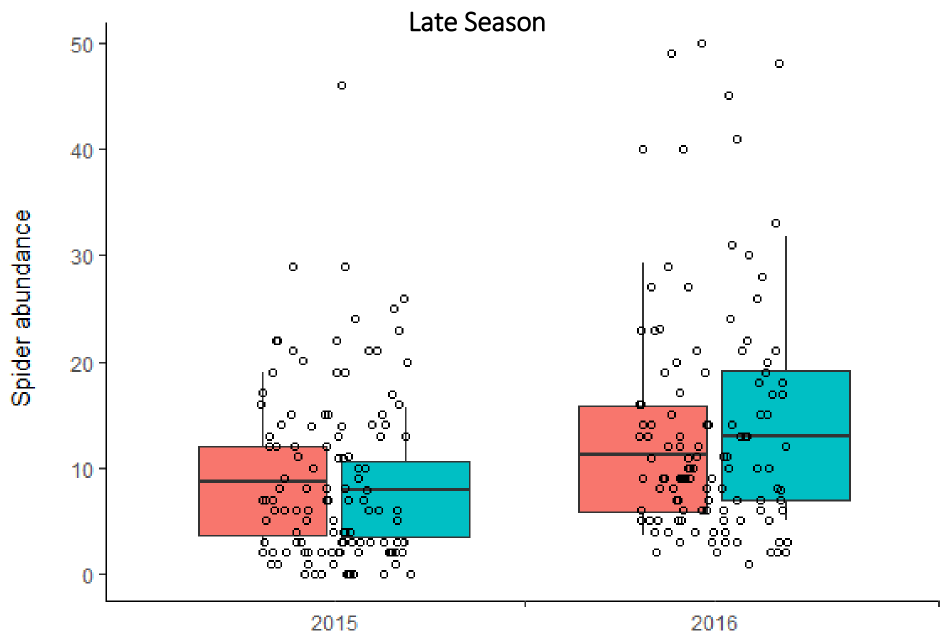


Figure 3.5. Boxplot graphically representing the interaction effect between Treatment (red = control; blue = lime treatment) and the two sampling Years (2015, 2016) from the pitfall trap data for total spider abundance in the late season. Sample size is two farms (n = 8 fields) resulting in 320 observations. Raw data is represented by open black circles, while model predictions (conditional taking into account random effects) from the LMM are shown as a boxplot displaying the interquartile range (IQR), medium as a black line and whiskers representing values 1.5*IQR.

considered (R^2_M) but variation explained is greater when the full model (fixed effect and random effects; R^2_C) is considered (early season: R^2_M = 0.032; R^2_C = 0.368; late season: R^2_M = 0.140; R^2_C = 0.549).

Liming had no effect on beetle abundance in either the early or late season. However, there was a strong effect of year on abundance with significantly fewer beetles in 2016 than 2015. Pitfall traps in the early season contained, on average, 24% fewer beetles in 2016 than in 2015 (Table 3.10). The effect size of year was even greater in the late season with on average, 63% fewer beetles per pitfall trap in 2016 compared with 2015. There was also evidence of an inverse relationship between beetle abundance and soil moisture (%) in both the early and late season. In the early season, abundance per pitfall trap was 8.08 beetles (95% CI: 5.99 – 10.81) with soil moisture of 20%, whereas in very wet soils of 80% moisture abundance declined by c.40% to 4.76 beetles per pitfall trap (95% CI: 4.26 – 5.31). In the late season, beetle abundance was c.45% lower in wetter soils (80%) compared to drier soil with a soil moisture of 20% (80% soil moisture: 18.36 beetles with 95% CI: 13.60 – 24.78; compared with 20% soil moisture: 33.78 beetles with 95% CI: 23.1 – 49.4 compared). There was also evidence of a positive association with vegetation heterogeneity and beetle abundance in the late season: a uniform and even vegetation structure of 1.0 SD from the mean vegetation height was associated with 22.65 beetles per pitfall trap (95% CI: 17.12 – 27.93), whereas with a diverse vegetation structure of 5.0 SD abundance was 32.46 beetles per pitfall trap (95% CI: 21.98 – 47.47). The pseudo r^2 values for the early season model were R^2_M = 0.114 and R^2_C = 0.124; and for the late season model were R^2_M = 0.265 and R^2_C = 0.462.

Table 3.8. Summary table showing the final Linear Mixed Models used in analysis to test the effect of lime treatment on total beetle abundance and on total spider abundance.

Model response variable	Type	Unit replication	Link	Error Structure	Random Effect	Fixed Effects
Log(Beetle abundance +1) – early season	Count	Pitfall trap	Identity	Gaussian	Field identity	Year + Moisture
Log(Beetle abundance +1) – late season	Count	Pitfall trap	Identity	Gaussian	Field identity	Year + Moisture + Heterogeneity
Log(Spider abundance +1) – early season	Count	Pitfall trap	Identity	Gaussian	Field identity	SOM
Log(Spider abundance +1) – late season	Count	Pitfall trap	Identity	Gaussian	Field identity	Year*Treatment + Moisture + Height

Table 3.9. Summary table showing the LMMs investigating the relationship between lime treatment and arthropods at the pitfall trap scale by season (early and late). Sample size was three farms (n = 9 fields) in the early season models and two farms (n = 8 fields) in late season across two survey years (2015, 2016) resulting in 360 and 320 observations respectively. Note that covariates were scaled and centred prior to analysis for ease of coefficient interpretation. Table shows results of the minimum adequate models with significance levels denoted by *'s where: * ≤ 0.05; ** ≤ 0.01; *** ≤ 0.001. Please note that the interaction between Year and Treatment in the spider abundance (late season) was non-significant but was retained in the model as this was only marginal (p = 0.055).

Predictor	Type (df)	Parameter estimates ± SE (test statistic)			
		Beetles (early season)	Beetles (late season)	Spiders (early season)	Spiders (late season)
Year	Factor (df =1)	- (F = 16.13**)	- (F = 93.97**)	-	- (F = 1.98)
Treatment	Factor (df =1)	-	-	-	- (F = 1.15)
Year : Treatment	Factor (df =1)	-	-	-	- (F = 3.67$^{.}$)
Moisture	Covariate (df =1)	-0.11 ± 0.04 (t = -3.06***)	-0.18 ± 0.06 (t = -3.12***)	-	-0.12 ± 0.05 (t = -2.36**)
SOM	Covariate (df =1)	-	-	-0.12 ± 0.05 (t = -2.36*)	-
Vegetation height	Covariate (df =1)	-	-	-	0.10 ± 0.04 (t = 2.44*)
Vegetation heterogeneity	Covariate (df =1)	-	0.09 ± 0.04 (t = 2.27***)	-	-

Table 3.10. Estimated marginal means (EMMs) for the LMMs investigating the relationship between lime treatment and arthropods at the pitfall trap scale while controlling for other variables of importance set to their mean values. Sample size was three farms (n = 9 fields) in the early season models and two farms (n = 8 fields) in late season model across two survey years (2015, 2016) resulting in 360 and 320 observations respectively. Values were computed by averaging together predictions and then back transforming from the transformed response of log(Abundance + 1). A 95% confidence level was used. Pairwise tukey method was used for comparing EMMs with significant p values shown in bold.

	Marginal mean estimate ha^{-1}	Lower CI	Upper CI	Pairwise comparison	Test Statistic	p value
Beetles (early season)						
Year						
2015	5.05	4.57	5.64	2015 – 2016	t = 4.00	**0.0001**
2016	3.86	3.49	4.31			
Beetles (late season)						
Year						
2015	24.05	17.46	33.11	2015 – 2016	t = 9.64	**<0.0001**
2016	8.85	6.36	12.30			
Spiders (late season)						
Year : Treatment						
2015:Control	8.85	5.53	14.30	2015:Control – 2016:Control	t = -0.25	0.9942
2016:Control	9.21	5.70	14.88	2015:Control – 2015:Lime	t = 0.59	0.9360
2015:Lime	8.33	5.16	13.46	2015:Control – 2016:Lime	t = -1.86	0.2488
2016:Lime	11.47	7.10	18.54	2016:Control – 2015:Lime	t = 0.68	0.9062
				2016:Control – 2016:Lime	t = -2.12	0.1509
				2015:Lime – 2016:Lime	t = -2.21	0.1228

3.5 DISCUSSION

Liming has recently been proposed as a way of improving foraging habitat for farmland breeding waders on grasslands in upland areas which are prone to acidification due to rain water-leaching, slope of the landscape and low buffering capacity, which can result in a reduction in earthworms which are pH-sensitive (McCallum *et al.*, 2016). In chapter 2 I demonstrated that the use of agricultural dust lime ($CaCO_3$) applied as a surface dressing in autumn 2014 resulted in an average pH difference between lime treatment and control plots of 0.22 pH units in 2015; 0.52 pH units in 2016 and 0.44 pH units in 2017; and an average difference in the number of earthworms per soil core between lime treatment and control plots of 0.81 earthworms in 2015; 3.25 earthworms in 2016; and 1.05 earthworms in 2017. The aim of this chapter, therefore, was to test whether the use of lime influences other non-target organisms, namely beetles (Coleoptera) and spiders (Araneae) which also contribute to the diet of Lapwing and Curlew, and are important prey items of other grassland birds such as Northern Wheatear (*Oenanthe oenanthe*), Meadow Pipit (*Anthus pratensis*), starlings (Sturnidae) and thrushes (Turdidae). In this study, I found that there was no effect of lime treatment on beetle abundance, although there may be a weakly positive effect of lime use on spider abundance in the late season (38% increase in abundance on limed plots between 2015 and 2016 significant at the 0.1 level). Additionally, spiders in the late season were more abundant in longer vegetation, and beetles in the late season were more abundant in highly heterogeneous vegetation. There was no effect of lime treatment on either vegetation height or vegetation heterogeneity. The results from this study therefore, have implications for the conservation of farmland birds in these habitats, as it provides evidence that lime use does not negatively impact upon other invertebrates or produce significant changes in sward height and heterogeneity of upland grazed grassland, and thus lime may be an appropriate conservation tool to improve foraging habitat through increases in earthworm abundance (at least in the short term) for waders where localised acidification has occurred.

3.5.1 The effects of lime treatment on vegetation height and heterogeneity

I found no support for hypothesis 1. There was no evidence to suggest an effect of lime use on either mean vegetation height, or on the vegetation heterogeneity index. These results suggest that it is unlikely that lime use will negatively affect utilization of the field by foraging birds, which can be affected by changes in the sward structure. However, there was a main effect of year which estimated that there was a significant difference in height between the baseline year (2014) and final year of the study in 2017. In the early season, there was an average decrease in height of 0.5 cm between 2014 and 2017, and in the late season an average increase in height of 0.9 cm between 2014 and 2017. Vegetation height was similar in the years 2015 and 2016 when the pitfall trap study occurred so is not likely to have influenced the invertebrate data. The nine study fields were actively grazed by livestock (predominantly sheep) which could move freely between the control and lime treatment plots. Livestock grazing could have reduced the power to detect an effect of liming on vegetation than would have been observed from liming non-grazed grasslands or meadow fields. Additionally, changes in grazing intensity over the period of the study could account for the relatively small change (<1cm) detected in vegetation height between 2014 and 2017. Although no data on livestock density were collected in this study to test this hypothesis, there has been a large body of research conducted on the influence of livestock grazing on the sward, with evidence that higher numbers of sheep in particular results in shorter vegetation (Rook *et al.*, 2004). A change in the stocking density or number of days grazing could therefore have influenced the yearly difference in sward height, as could annual weather fluctuations. A decrease in vegetation height in the early season could also have been influenced by weather conditions, which may have affected the growing season that year. It is also possible that the methods used in this study were not sensitive enough to detect an effect of lime use on vegetation between the plots. For example, measuring height with a ruler similar to the sward stick method could be affected by human error making it more difficult to detect small changes. An alternative method to determine sward height is the "drop disk" methods which is less sensitive to human error (Stewart, Bourn and Thomas, 2001). The use of the pin drop method to quantify the plant species abundance and community composition could have also provided more detailed information on changes within the sward as a response of liming. However, established agricultural grasslands, particularly those that would be targeted for soil amendment for the benefit of farmland breeding waders, are likely to have low botanical diversity and to have a relatively short sward height from grazing, and therefore the methods implored in this study were considered appropriate.

3.5.2 The effects of lime treatment on arthropods

There was little evidence to support hypothesis 2, as I found that there was no effect of lime treatment on beetle abundance, and only weak evidence of an effect of lime use on spider abundance in the late season which varied by year. Spider abundance in the late season was 38% higher in 2016 than 2015, of an average of 3 more spiders in 2016 per pitfall trap, as well as 25% higher spider abundance in the lime treatment compared to the control in 2016 with an average of 2 more spiders per pitfall trap in limed plots (albeit this was a marginally non-significant difference). The increase in total spider abundance in year 2016 on lime treatment, coincides with the year that the largest effect size in pH was detected between lime treatment and control plots. Spiders are highly mobile species with the spiderlings of several species capable of undergoing ballooning, a process of air travel which can be a few metres to hundreds of kilometres after hatching (Richter, 1970; Suter, 1992) giving them the ability to colonise habitats relatively quickly compared to some other groups of invertebrates, meaning that they can respond to environmental change over a relatively short period of time. Indeed, spiders have been shown to perform better than ground beetles as indicators of the effects of farming practices on arthropods in Scotland (Cole *et al.*, 2005). It is therefore likely that we would detect a response in spiders before beetles, which may be slower to respond, particularly as the pitfall trap study was only conducted over a short period of time. Additionally, the use of pitfall traps may not have been sensitive enough to capture an effect due to selective pressures of pitfall trapping.

As I did not find any evidence for an effect of lime use on vegetation properties, any differences in beetle and spider abundance from lime use would likely be due to indirect responses through changes in foraging composition within the soil fauna of their prey items (e.g. earthworms, collembola and mites), rather than changes to vegetation from lime use. Although spiders are polyphagous predators and can prey on a broad spectrum of species, higher densities of both Collembola and earthworms have been found on more neutral soils (e.g. earthworms: see chapter 2; Collembola: Chagnon, Paré and Hébert, 2000) and other studies investigating the response of soil fauna to liming in acidified forestry have reported increases in epigeic collembola, mites and earthworms within the soil of acidified forests in central Europe (Deleporte and Tillier, 1999; Kula and Matějková, 2007) which would support the hypothesis of spiders responding to improved prey abundance from lime use, although the evidence to support a positive response to liming on micro arthropod numbers in temperate grasslands of upland sites is mixed to date (e.g. Cole, Buckland and Bardgett, 2005; Fountain *et al.*, 2008).

In this study, although spiders and beetles were identified to species level, formal analysis on the effect of lime use was on total abundance only as the NMDS did not show any obvious effects of treatment on the community. It is likely that analysis conducted at family level or by functional guilds such as hunting strategy, would provide richer insight into the response of lime use and may detect specific trends on the effect of lime use which are not sensitive enough to detect from total abundance. Indeed, Buckton and Ormerod (1997) noted higher abundances of the spider families of linyphiids (money spiders) and tetragnathids (long-jawed orb weaver spiders), and of the beetle family hydrophilids (water scavenger beetles) at three limed upland stream catchment sites (mires) in Wales four years after liming had occurred, but higher abundances of lycosids (wolf spiders) and carabids (ground beetles) at the 10 unlimed reference sites. This highlights the importance of considering the community response and not just the total abundance. In this chapter, ground beetles and water scavenger beetles did not make up a high proportion of the species identified. Instead, this was dominated by staphylinids (rove bettles) which are a large group of beetles, many of which, to date, do not have keys for identification to species level. For spiders, the Linyphiidae were dominate in the data and are a large family which account for over 40% of the spider species found in Britain (n = 270 species). Money spiders are typically very small (<5mm long) and make sheet webs to catch their prey, which consists of horizontal sheets of silk with guide threads both above and below the web that helps to direct their prey into the centre of the web where the spider is (Bee, Oxford and Smith, 2017). The money spider species of *Tiso vagans* and *Oedothorax retusus* were the most abundant spider species recorded in the study, both of which are common and abundant species of grassland. It is therefore possible that the increase in total spider abundance observed here is due to a response in money spiders to lime use. Conversely, the number of spiders in the Lycosidae family (wolf spiders) were lower than expected for grassland habitats (n = 136 or 2.6% of individuals recorded). Wolf spiders are visual hunters which actively hunt prey on the ground and do not build webs. Therefore they are associated with diverse habitats with patches of more open ground, as well as areas of denser patches to hide in. The low trap catch of wolf spiders could be an effect of lime use, although it is more likely to be an artefact of the sampling design as the style of the pitfall trap used here included a square sheet of metal as a “hat” which could have added a selective pressure on the pitfall trap catch, excluding some of these larger wolf spiders which avoid hunting in dark enclosed spaces. Pitfall traps are a widely used sampling method to catch active ground dwelling invertebrates in grasslands and heathlands (Gardner *et al.*, 1997; Haysom *et al.*, 2004; Lyons *et al.*, 2017) and estimates relative invertebrate activity rather than an absolute measure of density (Greenslade, 1964). Consequently, they are less likely to capture sedentary species or those occupying higher levels of vegetation, which would also explain the high proportion of money spiders in the dataset, and why there was only weak evidence of a positive association between spider abundance and vegetation properties found in this chapter. Although the use of the metal trap lid may have influenced catch, it was selected for this study to prevent traps being trampled or interfered with by livestock.

At the community level, the NMDS bi-plot showed no obvious spatial pattern in terms of treatment and therefore this was not explored further in this study. This may have been because the study was conducted on too short a time scale to detect this. However, use of Canonical Correspondence analysis (CCA) or Redundancy analysis (RDA) which allows for the inclusion of explanatory variables may have produced more detailed ordination which might have identified trends not detected in the NMDS. There was also no baseline data in this study, although the experimental set-up of the BACI liming study should not have led itself to differences in arthropods between control and lime treatment plots prior to lime applications, it would have been a useful comparison for between years.

I also found evidence of a significant decline in total beetle abundance between the years 2015 and 2016 of 24% for the early season and 64% in the late season. The life cycle of beetles is very varied. The larval stages have very different strategies depending on species, which can vary from feeding on the roots of plants, to being predatory. Exceptional weather conditions can influence beetle development, activity and population dynamics, possibly at multiple time spans and with variable time lags (Jonason, Franzen and Ranius, 2014). The meteorological winter (December – February) of 2015/2016 broke several climate records for the UK and was the wettest winter on record for Scotland since 1910 (Kendon *et al.*, 2016; McCarthy *et al.*, 2016). Several storms including *Abigail*, *Desmond*, and *Gertrude* resulted in high levels of rainfall, strong winds and localised flooding. This was followed by a drier summer than average. The drier summer conditions is reflected in the soil data presented in this thesis with the year 2016 significantly drier than in 2015 (average soil moisture 67.8% in 2015 versus 40.3% in 2016; $t = 17.82$; $p<0.0001$). Thus the wetter winter and drier summer may have negatively impacted beetles, and driven the results seen in the year 2016 of low beetle abundance.

Both total spider and beetle abundance were higher in the late season than the early season. This is to be expected as there is, in general, a peak in abundance of most invertebrate species between late spring and early autumn. The higher pitfall catch in the late season also coincides with the timings of chick rearing for farmland breeding waders. The greater availability of these ground active invertebrates in the late season will therefore be of benefit to growing chicks, particularly with the increase in spider abundance observed on the lime treatment plots in this chapter.

Additionally, spiders in the late season were more abundant in longer vegetation, which is likely to be beneficial to spiders due to an increase in anchor points for web building. Beetles in the late season were more abundant in highly heterogeneous vegetation. This more diverse sward may be easier for beetles to move around and provides micro-habitats. Beetles were also negatively associated with soil moisture in both the early and late season. Water-logged soils may make it more difficult for beetles to move around in the field. It is unlikely that the relationship in lower abundances of beetles in wetter soils is due to pitfall traps being flooded out and specimens lost, as this relationship with moisture was not observed with spider abundance.

Finally, *Porrhomma campbelli*, a small spider species of the Linyphiidae family, is a rarity, and was recorded in this study providing the eighth record of this species for Scotland since 1992 (British Spiders, 2021). Spiders, and invertebrates in general, are typically less well studied and monitored than other taxa, such as birds. Studies such as this are therefore important, not just because of the research aims of the study in itself, but also for the species records that they can provide.

3.5.3 Conclusions, conservation implications and further research

Agricultural grasslands offer great potential for biodiversity conservation, subject to being managed in a suitable way (Hopkins and Holz, 2006). However, there are ongoing threats of grassland abandonment or habitat loss to woodland in many marginal and upland areas in regions of Europe (Pakeman *et al.*, 2019) which could lead to the loss of important habitat for many birds and other wildlife associated with these habitats, including farmland breeding waders. Balancing environmental and agricultural objectives is a high priority and there is a need for simple and effective grassland management options which enable the integration of production with biodiversity objectives in agricultural systems (Isselstein, Jeangros and Pavlu, 2005). The widespread application of fertilisers have contributed to biodiversity decline (Hopkins and Wilkins, 2006); however, here I suggest that, in specific circumstances, lime use could be carefully targeted to areas that have become locally acidified which may have positive effects on biodiversity (e.g. spiders).

The marginally non-significant increase in spider abundance following liming provides some evidence that lime use may be beneficial for this group, or at least is not detrimental. Although earthworms are an important component of wader diet, many birds (including waders) also eat spiders and beetles, suggesting that lime use may also benefit other birds which consume arthropods other than earthworms. There was also no evidence of an effect of lime use on vegetation, which suggests that lime use will not alter the accessibility of prey items to grassland birds, although studies on the foraging behaviour and field use of birds within limed fields are needed to provide additional evidence to support this. Furthermore, evidence for an effect of lime use on temperate grasslands on micro-arthropods such as collembola would be of interest to identify the potential mechanism behind the increase in spiders observed here, as would more detailed study on the effect of lime use at the family or functional group level for spiders and beetles on upland agricultural grasslands.

CHAPTER 4 | THE EFFECTS OF LIME MANAGEMENT ON DENSITY AND FORAGING BEHAVIOUR OF FARMLAND BREEDING WADERS.

4.1 ABSTRACT

Agricultural intensification is one of the major contributors in the observed declines in biodiversity within Europe with the evidence for the negative effects on farmland birds being particularly well documented. Changes in food supply as a result of the changes in agricultural management have been proposed as an important factor in these declines for many farmland bird species. Lime use to amend acidic soil pH has declined in the UK, and may be one factor contributing to lower soil pH widely observed in the UK. Soil pH influences soil fauna which are an important component of the diet of many invertebrate-feeding farmland birds. Gradual acidification could be contributing to a decline in foraging conditions through reduced soil invertebrates in lower pH soils. The addition of soil pH amendment (via lime use) as a tool for conservation management may therefore be beneficial for farmland birds to mitigate the historical change in management of lime. Here I test experimentally, at the field-scale, the efficacy of lime management on marginal enclosed agricultural grasslands as a conservation tool to 1) improve field density of farmland breeding waders, a group of birds of high conservation concern which feed on soil invertebrates; and 2) improve the foraging habitat by investigating three feeding metrics (peck rate minute^{-1}, intake rate minute^{-1} and peck success rate) at the species level. There was evidence of a significant and positive main effect of lime treatment on total wader density with limed plots having, on average, a modelled wader density of 50% higher than that of control plots which had received no lime. There was no evidence of an effect of lime treatment on wader peck rate minute^{-1} but peck rate declined with increasing vegetation height. There was an effect of lime treatment which varied by year on both intake rate minute^{-1} and peck success rate (proportion) which did not vary by species. The largest effect size was observed in year 2017 with an average bird on lime treatment having an intake rate double that of control. There was also an effect of species on intake rate minute^{-1} and peck success rate which varied by year. The year 2016, on average, had a higher intake rate minute^{-1} for Curlew, and a higher proportion of successful pecks for Curlew and Golden plover, compared with the other years of monitoring. Furthermore, birds observed with chicks took significantly fewer prey items, although the feeding of chicks themselves was not recorded. These results suggest that, for breeding waders in marginal grassland landscapes, the use of lime management alongside the current agri-environment recommendations would likely benefit farmland breeding waders, although this should be based on a site by site evaluation of the initial soil pH.

4.2 INTRODUCTION

Farmland breeding waders have declined globally, associated predominantly with the degradation and loss of more than 50% of the world's wetlands (Dugan, 1993). However within Europe, which holds a high proportion of the global breeding population of waders (chapter 1), long-term population declines have occurred in several key breeding populations on agricultural land (Piersma, 1986; Tucker and Heart, 1994; Norris *et al.*, 1998; Thorup, 2004; Wilson, Ausden and Milsom, 2004; Stroud *et al.*, 2006; Teunissen and Soldaat, 2006; Brown *et al.*, 2015; Birdlife International, 2017b; Harris *et al.*, 2020). These declines have been linked to changes in agricultural management and intensification (Vickery *et al.*, 2001; Newton, 2004; Wilson, Ausden and Milsom, 2004). The EU's Common Agricultural Policy (CAP) has been instrumental in driving agricultural intensification although the specific changes to farming management is variable between countries (Robson, 1997). In general, grassland intensification in Europe has included widespread drainage, reseeding with a more limited assemblage of more productive grass species supporting higher livestock densities and a change in cutting regimes from hay (one cut) to silage (several cuts). On arable systems, an increase in chemical input from fertilisers and pesticides has reduced arable weeds and invertebrates, and a switch in sowing dates has affected suitability of nesting sites. There has also been a loss of non-farmed habitat such as field margins, ponds and hedges, and in the UK a geographically polarised landscape of arable farming predominantly in the South and East, and livestock grazing towards the North and West. Consequently, intensification of the farmed landscape has seen an overall loss in small farms and mixed farming systems, and an increasingly simplified and uniform habitat with widespread wildlife losses (e.g. Chamberlain *et al.*, 2000; Robinson and Sutherland, 2002; Wilson, Evans and Grice, 2009). For farmland breeding waders, the intensification and improvement of grassland is likely to have reduced both the area and the suitability of habitat for nesting and foraging (Wilson, Vickery and Browne, 2001; Newton, 2004; Wilson, Evans and Grice, 2009).

A large proportion of the European population of farmland breeding wader assemblages are in the UK and the Netherlands e.g.(Burfield and Bommel, 2004; Brown *et al.*, 2015) and in these countries highly targeted conservation interventions in the form of agri-environment schemes (AES) have been implemented based on well researched species ecology and knowledge on the negative effects of agricultural management e.g.(Shrubb, 2007). AES measures have been included within the CAP since the 1990s and involve financial payment in return for wildlife friendly farming practices. For farmland breeding waders on grassland these are: 1) the manipulation of livestock management to reduce nest trampling and produce a desirable sward structure for nesting and foraging birds e.g.(Beintema and Muskens, 1987; Teunissen and Soldaat, 2006; O'Brien and Wilson, 2011); and 2) the manipulation of the water table to create wet features which improves foraging habitat e.g.(Green, 1988; Ausden, Sutherland and James, 2001; Eglington *et al.*, 2010; Fisher *et al.*, 2011; Baker *et al.*, 2012). Although there is good evidence of the local benefit of these AES options for waders at the farm and field scale, such as increases in nesting success and abundance e.g.(Sheldon, Chaney and Tyler, 2007; Rickenbach *et al.*, 2011), to date they have not been sufficient to halt or reverse population declines of waders on agricultural land at the national scale (Ausden and Hirons, 2002; Kleijn *et al.*, 2004; O'Brien and Wilson, 2011; Smart *et al.*, 2013) and that wader response to AES is variable between the species (O'Brien and Wilson, 2011). Indeed, declines of breeding wader populations are striking with 24% for Oystercatcher, 43% for Lapwing, 48% for Curlew and 42% for Redshank recorded in the latest Breeding Bird Survey for the UK between 1995 and 2018 (Harris *et al.*, 2020). These declines have been observed across most of the UK, particularly on enclosed lowland farmland in England and Wales, and more recently on upland sites in England and Scotland (Baines, 1990; O'Brien, Tharme and Jackson, 2002; O'Brien and White, 2003; Sheldon *et al.*, 2004; Sim *et al.*, 2005; Wilson *et al.*, 2005; Forrester *et al.*, 2007; Shrubb, 2007; Smart *et al.*, 2008). This is particularly concerning when we consider that both Curlew and Lapwing are globally Near Threatened and are Red Listed species of priority conservation concern in the UK (Brown *et al.*, 2015; Birdlife International, 2017b, 2017a).

Failure of AES to halt population declines may result from poor implementation and design of measures that vary in their effectiveness (Smart *et al.*, 2013), high predation rates suppressing populations (Roos *et al.*, 2018) or scale of the implementation (O'Brien and Wilson, 2011). Furthermore, it may be that some aspects of the habitat requirements of breeding waders in agricultural landscapes are not accounted for, such as the condition of the soil e.g.(McCallum *et al.*, 2015, 2016; Johannesdottir *et al.*, 2018). Waders are predominantly invertebrate feeders with a high proportion of their diet consisting of carabids, millipedes, spiders, tipulid larva and earthworms (Cramp and Simmons, 1983; Gillings and Sutherland, 2007; Hoodless, Ewald and Baines, 2007; Navedo *et al.*, 2020). Earthworms in particular are important in the diet during the breeding season for both adults and chicks, likely due to the high calorific and protein content of earthworms and the high energetic demands during this period (Barnard and Thompson, 1985; Galbraith, 1989; Baines, 1990; Beintema *et al.*, 1991; Paoletti *et al.*, 2002). Soil properties (moisture, organic matter, and pH) are well known to influence the diversity and abundance of invertebrates associated with the soil (Edwards, Hendrix and Arancon, 1996; Curry, 2004; Cole *et al.*, 2006). For example, earthworms are more abundant under neutral soil conditions with fewer earthworm species and abundances in acidic soils below pH 4.3 (Edwards, Hendrix and Arancon, 1996). However, the existing AES measures for waders are focussed on manipulating vegetation, stocking rates and hydrology, but not the soil itself, despite wader diet consisting of soil invertebrates. Specifically, there has been little consideration of manipulating the soil (with exception of soil moisture) to improve the effectiveness of AES despite the relationship between agricultural processes, soil properties, vegetation and invertebrate communities (Webb *et al.*, 2001; Bardgett *et al.*, 2005). Thus a key aspect of wader ecology may be largely missing from the current toolkit of measures.

Soil pH acidifies naturally over time which can be caused by a number of factors including heavy rainfall resulting in the loss of Calcium ions. Soil pH is also influenced by agricultural processes such as the addition of ammonium-based fertilisers, and livestock excretion (Rowell and Wild, 1985; Goulding, 2016). The use of agricultural lime ($CaCO_3$) can be used to amend acidic soil pH, with improvements to soil condition that can persist for several years (DEFRA, 2010). Maintaining more neutral soil conditions is beneficial for grass growth with the use of lime recommended for agricultural grasslands every 5-10 years to maintain optimum conditions (DEFRA, 2010). However, lime use in the UK has decreased by over 70% in the last 50 years, following a peak in sales in the 1950s/60s after the establishment of Government subsidies of lime which were then later abolished in the 1970s (MAFF, 1979; McCallum et al., 2016). A reduction in lime use may be contributing to lower soil pH widely observed by Baxter et al., (2006) between 1970 and 2001 particularly in western and northern areas of the UK that are dominated by livestock farms in higher altitude, higher rainfall environments. These areas are typical of marginal in-bye land (enclosed farmland below the moorland) that are important for supporting the UK's population of farmland breeding waders (Whittingham, Percival and Brown, 2000; Siriwardena *et al.*, 2016). Acidification of soils may have been further exacerbated in some areas by atmospheric deposition (sulphur and nitrogen) and by the nitrification of ammonium salts (Van Breemen, Driscoll and Mulder, 1984; Rowell and Wild, 1985). There has been a decline in atmospheric deposition from industrial deposition in the UK since The Clean Air Act of 1956, suggesting that pressure on acidification more recently may have been reduced as well as some evidence to suggest that soil pH by 2007 in the UK is no longer continuing to decline (Emmett *et al.*, 2010; Fowler *et al.*, 2012). However, agricultural soils require applications of lime periodically to neutralize the acidity caused by crop and livestock production, as well as from leaching effects in areas of high rainfall, slope and non-buffering geology and it has been estimated that much less lime is being applied in the UK than is required to offset this acidification, with many arable and grassland soils likely below optimum pH (Goulding, 2016).

Changes in food supply have been proposed as an important factor in the declines of many farmland bird species (Benton *et al.*, 2002; Robinson and Sutherland, 2002; Newton, 2004). For lapwing, reductions in reproductive success rather than adult mortality as a result of changes to farming management is considered the driving mechanism behind their declines, alongside nest loss, increased predation rates of eggs/chicks, and poor chick growth from reduced habitat quality contributing to this e.g. (Peach, Thompson and Coulson, 1994; Vickery *et al.*, 2001; Wilson, Vickery and Browne, 2001). It is therefore possible that changes to upland grasslands in the UK, such as gradual acidification, have contributed to the decline in farmland breeding waders that use these fields, as a result of a decline in foraging conditions through reduced soil invertebrates in lower pH soils. The addition of soil pH amendment (via lime use) as a tool for wader conservation management may therefore be beneficial for waders to mitigate the historical change in management of lime use in the UK. This is likely to benefit other farmland feeding birds which consume soil invertebrates as part of their diet including thrushes, starlings and corvids (e.g. Tucker, 1992; Gruar, Peach and Taylor, 2003). However, the conservation value of lime use will depend on whether the management practice is attractive to waders and results in improved feeding success, as opposed to just increasing abundance of food resources which may not be accessible. There is some evidence to support the hypothesis that lime use is attractive to waders. For example, field use of Lapwing and Black-tailed Godwit have been observed to increase following lime application, which was coupled with an observed increase in earthworm abundance (Brandsma, 2004). Additionally, Mccallum *et al.*, (2018) observed that the density of breeding Lapwings was 52% higher on fields which had been limed as part of re-seeding grass leys following fodder crop management than fields which had not undergone this process at a single farm within Scotland. The positive effects on field soil pH and higher Lapwing densities than control fields persisted for up to seven years. Overall, these studies suggest that lime use may be attractive to waders. Soil pH also influences vegetation structure and growth, via increases in the nutrient availability to plants which could also affect wader foraging, as prey detection is primarily based on vision and therefore accessibility of prey items, as well as abundance, will be affected by the sward (Whittingham and Markland, 2002; Butler and Gillings, 2004; Devereux *et al.*, 2004, 2006; Smart *et al.*, 2006). Additionally, soil penetrability and moisture is important for the accessibility of prey items from probing (Green, Hirons and Cresswell, 1990). Both the accessibility and the abundance of food resources are likely to influence field use by birds (Robinson and Sutherland, 1999; Brickle *et al.*, 2000; Atkinson, Buckingham and Morris, 2004).

In this chapter, I experimentally test the effect of lime use on grasslands in marginal upland areas on wader density and foraging behaviour. Specifically, I test 1) if there is an effect of lime treatment on wader density. Here I hypothesise that lime treatment will have higher wader densities than control due to the improved field conditions as brought about by liming; and 2) if there is an effect of lime treatment on wader foraging behaviour on three metrics of foraging success. Here I hypothesise that wader peck rate (number of pecks taken per minute), wader intake rate (number of prey items taken per minute) and peck success rate (proportion of pecks which resulted in prey taken) are higher on lime treatment than control, due to improved foraging conditions. The choice of foraging metrics were drawn from classical foraging behavioural studies of waders on agricultural grassland during the breeding season (Galbraith, 1989). The number of pecks and steps have also been used in foraging studies of waders on intertidal areas as indicators of habitat quality in determining conservation management (Lourenço, Granadeiro and Palmeirim, 2005; Mander, Marie-Orleach and Elliott, 2013). The three metrics were also chosen for practical reasons, balancing fieldwork constraints (time and effort) and the usefulness of the metric to capture the biological information required to address the impact of liming on foraging behaviour. I also hypothesise that 3) the habitat variables of vegetation height and ground penetrability will be negatively associated with the three feeding metrics, and that soil moisture will be positively associated with the three feeding metrics as these variables may influence accessibility of prey items to waders regardless of treatment. Combined, these hypotheses will examine the effectiveness of lime use applied as a surface dressing for the benefit of waders as a potential conservation management intervention. Studying the foraging success of birds is preferred to direct measurements of food abundance as a method of assessing habitat suitability because it reflects the resources directly available to foragers (Whitehead, Wright and Cotton, 1996). However, some indication of changes in food abundance following lime use was recorded in previous chapters (2 and 3) and will be used to help discuss the results of this chapter.

4.3 METHODS

4.3.1 Study area and BACI lime experiment

The study was conducted on a subset of the farms (n = 5; 12 fields) included in the before-after-control-impact (BACI) lime experiment of agricultural grasslands in marginal upland areas described in Chapter 2. For more detailed methods of the BACI lime experiment, please refer to Chapter 2, but briefly this involved a split plot experimental design using 2ha plots where grassland fields of greater than 4ha were divided in half, with one field half receiving an application of agricultural dust lime ($CaCO_3$; lime treatment) as a surface dressing (no till) and one field half not receiving any lime (control treatment) with a buffer strip of 0.5 m between the control and lime treatment. In this chapter, this results in a total study area size of 60.6 ha. Fields were monitored for soil properties (pH, organic matter, and moisture) and vegetation (height, heterogeneity) annually including in spring of the baseline year (2014) prior to the application of lime onto treatment field halves and for three years after lime application (2015, 2016 and 2017). Lime was applied onto the lime treatment plots at an application rate of 4 tonnes ha^{-1} for either one or two consecutive years in autumn of 2014 and 2015 where required depending on starting pH and soil type (Table 4.1). Bird data was collected for the years 2015, 2016 and 2017 but due to time constraints it was not possible to conduct surveys in 2014 (Table 4.2). Normal management of the field was allowed to continue as usual, and livestock were able to move freely between the field plots and graze the entire field allowing for direct comparison between the control and treatment. Two of the farms in this study were RSPB reserves with tenant graziers, while three were commercial farms owned and grazed by the farmer. All farms comprised of extensive livestock grazing of black faced sheep, with the exception of three farms which also grazed beef cattle for part of the year, and one farm which also grazed a small herd of llamas. All farms conducted some form of management of predator control as determined by personal communication with the landowner, although no quantitative data was available on this.

Due to the large geographical distances between farms, there were some climatic and edaphic differences (Table 4.1; Figure 4.1). For example, Farm B had the wettest soil conditions, while Farm A had the driest soils and a high soil organic matter content. Fields were a mix of semi/improved acid and neutral grasslands. National Vegetation Classification (NVC) codes were available for some fields and comprised predominantly of MG6 (classical habitat of improved grassland), MG10 (grasslands of low botanical nature conservation value with high proportion rush cover) and U4 (community of upland (sub-montane) areas of north and western Britain associated with a range of acidic soils on lime-poor substrates (Rodwell, 1991; Averis *et al.*, 2004)

Table 4.1. Site details for the five upland farms where wader observations occurred as part of the BACI lime use experiment. Soil properties and habitat variables calculated at the plot scale showing mean ± SE for the baseline year only. Please note table continues on following page.

Variable	Farm A	Farm F	Farm C	Farm B	Farm G
Elevation (m)	188.0	145.0	282.0	200.0	230.0
Annual Rainfall (mm)[1]	1391.8	1205.7	1882.1	919.4	875.4
Mean Air Temperature (°C)	7.7	7.7	7.7	7.9	7.9
Soil Class	Mineral	Organic	Mineral	Mineral	Mineral
Soil Type[2]	Brown earth	Peaty Podzols/Gleys	Alluvial soils	Alluvial soils	Alluvial soils
Soil Texture[2]	Sandy loam to silty loam	Sand to sandy loam	Sandy loam to silty loam	Clay to sandy loam	Clay to sandy loam
Grazing Management[3]	Ewes and lambs in spring, followed by ewes. No winter grazing.	Ewes and lambs in spring, followed by yearling cattle. No winter grazing.	Ewes and lambs in spring, followed by ewes and llamas. No winter grazing.	Cattle grazing only. No winter grazing.	Mixed cattle and sheep grazing. No winter grazing.
Mean soil $pH_{(Ca2Cl)}$ [4]	Control = 4.8 ± 0.06;	Control = 5.1 ± 0.04;	Control = 4.9 ± 0.06;	Control = 5.1 ± 0.03;	Control = 4.7 ± 0.08;
	Treatment = 5.0 ± 0.04	Treatment = 5.1 ± 0.04	Treatment = 4.9 ± 0.08	Treatment = 5.2 ± 0.06	Treatment = 4.7 ± 0.04
Mean soil organic matter (%)[4]	Control = 10.0 ± 1.02;	Control = 35.8 ± 1.32;	Control = 21.7 ± 2.60:	Control = 21.3 ± 1.57;	Control = 39.2 ± 4.96;
	Treatment = 8.6 ± 0.51	Treatment = 35.1 ± 1.26	Treatment = 22.6 ± 2.63	Treatment = 21.9 ± 1.13	Treatment = 42.7 ± 2.62
Mean soil moisture (%)[4]	Control = 43.1 ± 5.70;	Control = 55.3 ± 1.51;	Control = 66.1 ± 1.60;	Control = 67.4 ± 3.88;	Control = 49.0 ± 4.53;
	Treatment = 31.9 ± 3.90	Treatment = 61.2 ± 1.62	Treatment = 68.2 ± 0.88	Treatment = 75.6 ± 1.45	Treatment = 52.6 ± 3.53
Mean soil penetrability (Newtons)[4]	Control = 3.7 ± 0.12;	Control = 3.2 ± 0.07;	Control = 3.4 ± 0.06;	Control =2.5 ± 0.22;	Control = 3.9 ± 0.17;
	Treatment = 3.8 ± 0.13	Treatment = 3.1 ± 0.06	Treatment = 3.3 ± 0.04	Treatment = 2.3 ± 0.21	Treatment = 4.2 ± 0.11

Table 4.1. Continued from previous page.

Variable	Farm A	Farm F	Farm C	Farm B	Farm G
Mean vegetation height (cm)[4]	Control = 7.0 ± 0.51; Treatment = 7.0 ± 0.31	Control = 10.6 ± 0.46; Treatment = 11.0 ± 0.44	Control = 3.5 ± 0.17; Treatment = 3.3 ± 0.12	Control = 3.6 ± 0.13; Treatment = 3.6 ± 0.29	Control = 3.6 ± 0.46; Treatment = 3.6 ± 0.44
Mean vegetation heterogeneity (SD)[4]	Control = 1.8 ± 0.18; Treatment = 1.9 ± 0.11	Control = 2.7 ± 0.22; Treatment = 3.0 ± 0.28	Control = 1.0 ± 0.08; Treatment = 1.0 ± 0.07	Control = 1.2 ± 0.10; Treatment = 1.0 ± 0.20	Control = 1.0 ± 0.03; Treatment = 1.1 ± 0.02
Mean bumpiness (SD)[4]	Control = 0.8 ± 0.08; Treatment = 0.7 ± 0.03	Control = 0.6 ± 0.01; Treatment = 0.6 ± 0.02	Control = 0.7 ± 0.06; Treatment = 0.7 ± 0.01	Control = 1.2 ± 0.02; Treatment = 1.3 ± 0.03	Control = 0.9 ± 0.05; Treatment = 0.7 ± 0.00
Mean bare ground (%)[4]	Control = 0.0 ± 0.00; Treatment = 0.0 ± 0.00	Control = 0.2 ± 0.01; Treatment = 0.2 ± 0.01	Control = 0.0 ± 0.01; Treatment = 0.0 ± 0.01	Control = 0.0 ± 0.01; Treatment = 0.0 ± 0.01	Control = 0.3 ± 0.00; Treatment = 0.3 ± 0.00
Mean rush cover (%)[4]	Control = 0.1 ± 0.01; Treatment = 0.1 ± 0.01	Control = 0.2 ± 0.01; Treatment = 0.2 ± 0.01	Control = 0.2 ± 0.03; Treatment = 0.2 ± 0.03	Control = 0.1 ± 0.00; Treatment = 0.1 ± 0.00	Control = 0.1 ± 0.00; Treatment = 0.1 ± 0.00
Mean standing water (%)[4]	Control = 0.0 ± 0.00; Treatment = 0.0 ± 0.00	Control = 0.2 ± 0.01; Treatment = 0.2 ± 0.01	Control = 0.2 ± 0.02; Treatment = 0.2 ± 0.02	Control = 0.1 ± 0.00; Treatment = 0.1 ± 0.00	Control = 0.1 ± 0.00; Treatment = 0.1 ± 0.00
Lime application 2014	4 tonnes ha $^{-2}$	4 tonnes ha $^{-2}$	4 tonnes ha $^{-2}$	4 tonnes ha $^{-2}$	NA
Lime application 2015	4 tonnes ha $^{-2}$	4 tonnes ha $^{-2}$	4 tonnes ha $^{-2}$	NA	4 tonnes ha $^{-2}$

[1] Mean annual rainfall and temperature calculated from the four years (2014, 2015, 2016 and 2017) of the BACI study from the nearest local weather stations accessed from *Scottish Environment Protection Agency (SEPA) Open Source Rainfall data for Scotland* (https://apps.sepa.org.uk/rainfall) and *Met Office MIDAS Open: UK Land Surface Stations Data (1853-current)* (https://catalogue.ceda.ac.uk/uuid/dbd451271eb04662beade68da43546e1) respectively; [2]Soil details accessed from UK Soil Observatory (http://www.ukso.org/); [3]Grazing management from personal communication with the landowner; [4] calculated from soil core/quadrat with standard error recorded in spring of the baseline year (2014) of the BACI lime experiment prior to any application of agricultural dust lime onto treatment plots.

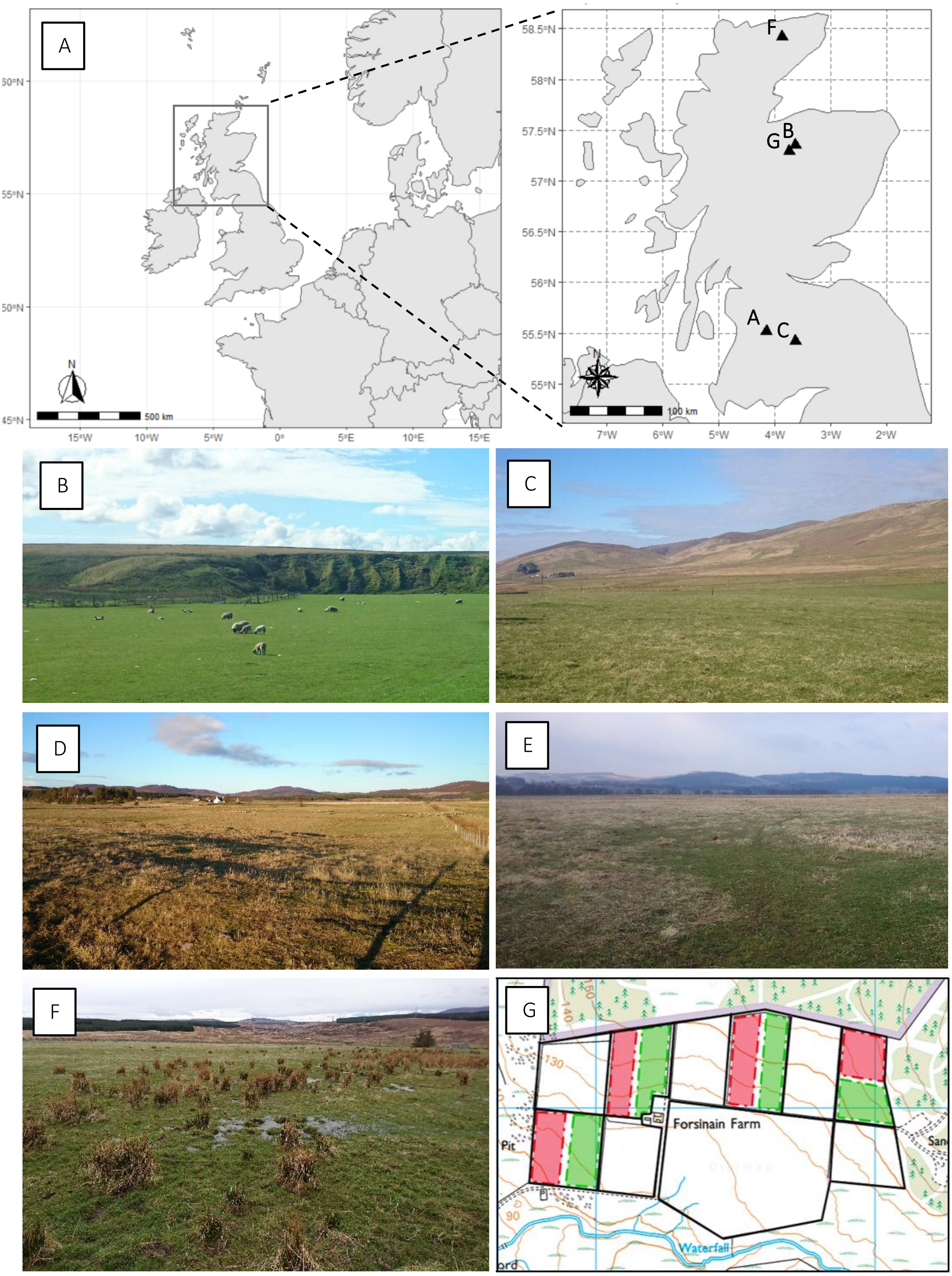


Figure 4.1. (A) Map of the United Kingdom (UK) with insert of Scotland showing the location of the five upland grassland farms as black triangles where bird observations were conducted as part of the lime experiment. Farm codes shown in black text on the upper left of the symbol. **(B-F)** Photos of a typical field at Farm A; Farm C; Farm B; Farm G; and Farm F respectively. **(G)** Ordnance Survey map of Forsinain Farm showing the split-plot experimental design of the lime field trials. Fields selected were in-bye grazed grassland fields (n = 4). Each field consisted of two plots (control/lime). Location within the field of each plot is shown by red for control, and green for lime treatment.

4.3.2 Bird count data

Bird observations were conducted from within a camouflaged and mobile bird hide (Stealth Gear Two Man Wildlife Photography Chair Hide; Figure 4.2) at a suitable vantage point in an adjacent field so as not to disturb the normal behaviour of the birds within the target field. Each field was visited twice between May and June with a minimum of two weeks between visits, and spot counts of all bird within the field and their location (control/lime treatment) recorded in the morning on an hourly basis between 6am and 11am using 8 x 42 binoculars (RSPB BG.PC Binocular) and a telescope (RSPB 82 HD scope). Counts were only conducted on good weather days (no rain or excessive wind) and observers were trained and tested in identification prior to the start of the survey period. In Scotland, lapwings and other waders typically begin nesting at the end of March, with most first clutches laid during April. Therefore, where possible, the sex (male/female) and age (adult/chick/fledgling) of birds were recorded. Only birds within the field were recorded, while those perched on fences were not included in analysis as it was not possible to determine which of the adjacent fields the birds were using. Due to logistics and time constraints caused by initial set up of the project, none of the fields in the baseline year 2014 have observational bird data, and only three of the farms were surveyed in the final year of the study due to unforeseen circumstances unfortunately reducing the number of sites that were possible to visit that year (Table 4.2).

Table 4.2. Number of farms and fields that were surveyed for waders in each year of the study.

Study Year	Number of Farms	Number of Fields
2015	5	12
2016	5	12
2017	3	10

Figure 4.2. **(A)** Photograph of the two person mobile hide used in wader observations; **(B)** photograph of a view looking out of the hide with sheep visible and birds flying; **(C)** photograph of the hide in a field (on left).

4.3.3 Bird foraging behaviour

Focal observations (Galbraith, 1989) of waders were conducted at each of the fields on the same morning as the spot counts between 6 - 11am resulting in a total of two days of observations per field annually. Focal observations were conducted until the bird was obscured or lost from view for more than three seconds, or for up to a maximum of ten minutes using a 20 x 60 telescope (RSPB 82 HD scope) from an elevated and suitable vantage point and the birds location within the field recorded (control/lime treatment) onto a paper map. Whether or not the bird was solitary, in a pair, or in a same/mixed species flock was noted as feeding behaviour can be altered by the presence of others (Barnard and Thompson, 1985) as well as the presence or absence of chicks. Dunlin (*Calidris alpina*) and wader chicks were considered too small to observe accurately in the field and were excluded from the foraging observations. Due to the low abundance of waders in a field at any one time it was possible to identify individuals from the same morning by their pattern, markings and position within the field. Where the same bird was observed on more than one observation in the same morning, a minimum period of 10 minutes was left between each repeat observation and a unique bird identification was recorded.

Waders detect prey visually and use a behaviour known as pause/travel feeding, consisting of periods of steps, pausing, prey searching/scanning, crouching and pecking (Shrubb, 2007). For lapwings, crouching prior to a peck is often associated with targeting larger prey items in the soil such as earthworms (Barnard and Thompson, 1985). As a result of their characteristic pause/travel feeding, their behaviour is relatively easy to monitor. Observations on the feeding behaviour of individuals included: the duration of each observation (minutes and seconds recorded by stop watch); the total number of pecks made during the observation (recorded by hand-held clicker); and the number of pecks that resulted in prey capture (recorded by hand-held clicker). A single peck was classified as an attempt to capture food items and was initiated from the contact of the bird's beak with the ground/foliage and its subsequent removal from the ground/foliage and did not discriminate between the type of peck (e.g. surface peck or deep probing in the soil). A peck was classified as a "success" i.e. prey captured, when either a prey item was observed in the beak and ingested, or from the observation of swallowing (Lourenço, Granadeiro and Palmeirim, 2005). Where possible, prey items were identified. Larger items, such as earthworms (Annelida: Lumbricidae) were occasionally pulled out of the soil and were easily visible. However, from observation distance the consumption of small and medium prey such as insects, larvae and small earthworms was likely to be cryptic and difficult to record accurately as such prey can be ingested during continuous probing. Thus, under recording of successful pecks is likely to have occurred and should be interpreted as a conservative index of the intake of large prey taken and not of total prey consumption. In addition, too few observations of Redshank and Snipe were recorded (less than 3) and therefore these two species were also not included in analysis.

The following variables were chosen for analysis as proxies of the relative foraging habitat quality based on (Galbraith, 1989): peck rate (number of pecks per minute of observed foraging behaviour); intake rate (number of pecks per minute where an item of food was observed/swallowed); and peck success rate (proportion of successful pecks that resulted in food being swallowed).

4.3.4 Soil properties and habitat variables

For full methods on soil properties and habitat variables please see chapters 2 and 3 respectively. Briefly, soil cores were collected (n = 20 per field) annually and pH and organic matter content analysed in the laboratory from dried and processed soil. All other variables were collected in the field and recorded on each visit (three to six times annually depending on whether pitfall traps were installed at that farm) and the mean value recorded per data point (i.e. soil core) used in analysis (Table 4.3).

Table 4.3. Summary table of **(A)** response, and **(B)** predictor variables used to investigate the relationship of waders with soil properties, habitat, and lime treatment in Generalised linear mixed models (GLMMs). Range and mean shown at the treatment plot level over three years that bird surveys took place (2015, 2016, and 2017). Please note table continues on following page.

Variable name	Unit of replication	Method of data collection
(A) Response:		
Wader density ha^{-1}	Plot	Hourly spot counts conducted between 6am – 11am on two morning visits per year and the maximum count (summed across all species) used in analysis (range: 0 – 12 waders; mean: 2 waders)
Peck rate $minute^{-1}$	Bird	Focal observations of individual birds over a maximum of ten minutes with the total number of pecks recorded by hand-held clicker and divided by the duration observed in minutes (range: 0 – 96.7 $minute^{-1}$; mean: 5.9 $minute^{-1}$)
Intake rate $minute^{-1}$	Bird	Focal observations of individual birds over a maximum of ten minutes with the number of successful pecks recorded by hand-held clicker and divided by the duration observed in minutes (range: 0 – 7.7 $minute^{-1}$; mean: 1.3 $minute^{-1}$)
Success rate (%)	Bird	Calculated as the proportion of successful pecks (prey items taken) to the number of total pecks (range: 0 – 100%; mean: 28.9%)
(B) Predictor:		
Soil pH	Soil core	Measured in the laboratory using 0.01M $CaCl_2$ annually (range: 3.9 – 5.8; mean: 4.9)
Soil organic matter	Quadrat	Measured in the laboratory from loss of ignition (LOI) annually (range: 7.2 – 62.0%; mean: 28.5%)
Soil moisture	Soil core	Recorded in the field using soil moisture probe (range: 27.3 – 87.5%; mean: 65.4%)
Vegetation height	Quadrat	Measured to nearest cm with a ruler from five locations within a quadrat (range: 2.2 – 18.2 cm; mean: 6.3 cm)
Vegetation heterogeneity index (SD)	Quadrat	Standard deviation of the mean vegetation height from the five measurements recorded from the pitfall trap (range: 0.6 – 5.4; mean: 1.8)
Ground bumpiness (i.e. micro-topography) index (SD)	Quadrat	Measured using a "bumpimeter" with six values recorded of the distance from a fixed point to the ground, and the standard deviation of the mean used as an index for how flat/bumpy the ground was (range: 0.5 – 1.4; mean: 0.7)
Bare ground (%)	Treatment	Measured as a proportion estimated by eye viewing field from vantage point (range: 0 – 5%; mean: 1%)

Table 4.3. Continued from previous page.

Variable name	Unit of replication	Method of data collection
Surface level water (%)	Treatment	Measured as a proportion estimated by eye viewing field from vantage point (range: 0 – 25%; mean: 13%)
Rush cover (%)	Treatment	Measured as a proportion estimated by eye viewing field from vantage point (range: 0 – 30%; mean: 13%)
Treatment	Treatment	Recorded from the BACI lime use experiment (two levels: Lime treatment; Control (non-limed))
Year	Year	Year surveyed (four levels: 2014; 2015; 2016; 2017)
Plot size	Treatment	Measured from Ordnance Survey Digital Terrain map using 50-m grid (four levels: 2014; 2015; 2016; 2017)
Farm identity	Farm	Assigned arbitrary code (five levels: Farm A; Farm B; Farm C; Farm G; Farm F)
Field identity	Field	Assigned arbitrary code (twelve levels: F1 – F12)
Bird identity	Bird	Assigned arbitrary code (one hundred and sixty seven levels: B1 – B166)
Species	Bird	Recorded bird species (Curlew; Lapwing; Oystercatcher; Golden plover; Dunlin; Redshank; Snipe)
Bird social context	Bird	Recorded bird position during feeding observation (four levels: Solitary; Pair; Flock; Mixed species flock)
Chick presence	Bird	Recorded bird with a chick(s) or not during feeding observation (two levels: 0;1)

4.3.5 Statistical Analysis

4.3.5.1 Wader summary community descriptors

Several summary descriptors and indices were used to provide an overview of the wader community. These comprised of the total number of individuals recorded (abundance), species richness and Simpson's Diversity Index (D) and Simpson's Measure of Evenness (E) of the community (Magurran, 2004) calculated at the farm scale.

4.3.5.2 Effect of lime treatment on wader density

To determine whether lime use had an effect on wader density (hypothesis 1), a GLMM model approach was taken. As the number of waders counted were low, the maximum count of waders (summed across all species) over the two visits was taken as the response variable with the unit of replication at the treatment plot scale, resulting in a total of 68 data points from five farms (n = 12 fields) over the three year period (2015, 2016 and 2017). The model was fitted with a Poisson error structure and log link and with an offset of the natural log of plot size (ha). The Poisson model was checked for over-dispersed and was acceptable (values ≤ 1.4; Payne *et al.*, 2018). Treatment (factor: control/lime treatment), year (factor: 2015/2016/2017) and an interaction effect between the two were fitted as fixed effects. Farm (factor: A; B; C; F; G) was also included as a fixed effect to account for the sampling design as too few levels were used for a random effect. The random effect of field identity (factor: F1-F12) was included to account for repeated measures from the same field. Soil properties and habitat variables were also fitted in the model as predictor covariates of wader density (Table 4.3). Please note that pH was not included as a covariate in the model as this soil property is affected by lime treatment (chapter 2).

At the species level, values were very low and there was a high proportion of zeros within the dataset, resulting in insufficient data for separate species abundance models testing the relationship between density and lime treatment. Presence-absence (1,0) GLMMs were explored with a binomial error structure and logit link; however model fit was very poor and these were therefore removed from analysis.

4.3.5.3 Effect of lime treatment on wader feeding metrics

To test whether feeding behaviour was influenced by lime treatment (hypothesis 2), three metrics were analysed 1) peck rate (number of pecks per minute of observed foraging behaviour); 2) intake rate (number of pecks per minute where an item of food was observed/swallowed); and 3) success rate (proportion of pecks that resulted in food being swallowed). These were fitted as three separate GLMMs with the unit of replication the individual bird. Models with the response of peck rate and intake rate were initially fitted with a Poisson error structure and log link with an offset of the natural log of time (in minutes); however, both models were over-dispersed and observed to be zero-inflated and were therefore fitted as zero-inflated negative binomial models. The success rate model was fitted as a binomial model with logit link using the number of successes and failures as the response. In all instances, both field identity and bird identity was fitted as a random effect. Farm was fitted as a fixed effect to account for the sampling design. The fixed effects of wader species (factor: Lapwing/Curlew/Oystercatcher/Golden plover), treatment, year and the three way interaction between species, treatment and year were included. Fitting species and the interaction effects in the model controls for how different structurally wader species are from each other, and how they may therefore behaviourally respond differently in terms of feeding metrics to treatment. In addition, the binary presence/absence of chicks with the observed bird was included as a factor, along with the birds social context (factor: mixed-species flock/same species flock/solitary/pair) which can alter feeding behaviour, as for example, adults with chicks may be primarily vigilant rather than foraging. Soil properties (with the exception of soil pH) and habitat covariates were also included as fixed effects in the model to control for variables of importance while testing for an effect of treatment, and also allowed for the testing of hypothesis 3) that sward height and soil penetrability would be negatively associated with the three feeding metrics, while soil moisture would be positively associated.

4.3.5.4 Model selection, validation and post-hoc tests

All statistical analysis was conducted in R version 3.5.1 (R Core Team, 2019) with data exploration carried out following the protocol described in Zuur, Ieno and Smith (2007). The summary community descriptors were calculated using the Vegan package (Oksanen *et al.*, 2020). Multicollinearity of numerical predictors was assessed using Pearson correlation coefficients and the correlation between variables was considered acceptable if the coefficient was less than 0.7 (Dormann *et al.*, 2013). Strong correlations between two or more predictor variables can cause instability in parameter estimation in regression-type models, and are therefore typically recommended to be removed although there is some debate on the threshold and whether variables should be removed at all (see Graham, 2003) . In this study, the variable pairs of vegetation height with vegetation heterogeneity were above the threshold considered acceptable ($r \geq 0.7$), as was the proportion of rush cover with surface level water, and the pairs of proportion of soil moisture with penetrability (Figure A8 in Appendix). These pair combinations are likely capturing very similar habitat characteristics, for example, rush favours damp, wet and boggy soil patches. Despite the fact that highly correlated variables may supply redundant information in regression models, vegetation height, heterogeneity, rush cover, surface level water and soil moisture have been shown to be highly important predictors of waders in previous studies and were therefore retained in this study. Therefore, in order to combat collinearity, multiple starting models were fitted with either one or the other of the pairs of correlated variables in the starting full model, but not both, and the minimal adequate model (MAM) found as described below (Graham, 2003).

Generalised linear mixed effects models (GLMMs) were implemented using the lme4 package of R (Bates *et al.*, 2015), with the exception of models fitted with a zero-inflation parameter which were implemented using the glmmTMB package (Brooks *et al.*, 2017). Model validation was achieved from graphical visualisation of the model residuals to check for normality and homogeneity of variance (Zuur, Ieno and Smith, 2007; Zuur *et al.*, 2009; Hartig, 2020). Pseudo R^2 values were calculated following Nakagawa and Schielzeth (2013) to indicate how well the model predicted the response (a measure of the explained variance/goodness of fit) without the random effects (marginal R^2) and when accounting for the random effects (conditional R^2) using the *rsquared* function in the piecewiseSEM package of R (Leftcheck, 2016) and using the performance package of R (Lüdecke, Makowski and Waggoner, 2020) for models built using glmmTMB. Covariates were standardised and centered with a mean of zero prior to analysis so that the model intercept held more meaningful interpretation (Schielzeth, 2010). Estimated marginal means (also known as least-squares means) were calculated for factors using the emmeans package in R where appropriates (Lenth, 2020). The minimal adequate models (MAM) were obtained using a manual stepwise backwards selection tested by likelihood ratio tests (LRT) of nested models, retaining all variables that were significant at the 5% level.

4.4 RESULTS

4.4.1 Wader summary community descriptors

In total, 217 waders were recorded across the five farms (n = 12 fields) consisting of 7 species from the wader maximum spot counts (Table 4.4). At four of the farms Lapwing chicks/fledglings were recorded, and at one farm Curlew chicks/fledglings were recorded. Species richness ranged between three waders and five waders per farm. Farm F was the only farm to have golden plover and dunlin recorded. Golden plover were the most abundant wader species recorded, with Lapwing second most abundant. Farm G had the highest Simpson's diversity and Farm A the lowest, while Farm C had the highest Simpson's evenness.

For the focal observations, a total of 243 observations were recorded across four wader species (Curlew, Lapwing, Golden plover, Oystercatcher) equating to approximately 32 hours of recordings (n= 1,943 minutes). Of this, 35 observations were recorded of adult birds with chicks. Some of these observations were of the same bird, thus the unique number of adult birds observed with at least one chick was 24 adults of which 14 were Lapwing, 9 were Curlew and 1 Oystercatcher. Eleven of the birds with chicks were observed on control, while thirteen of the birds with chicks on lime treatment. Overall, Lapwing observations made up the highest number of observation minutes, with Oystercatcher the fewest.

Table 4.4. Summary table of the maximum count of waders recorded across the two annual visits, summed over the three years (2015, 2016, and 2017) and shown at the site level. Where both adults and chicks were recorded the total abundance is shown for that species with brackets showing the number of adults and chicks respectively.

Species & community metrics	Farm A	Farm B	Farm C	Farm G	Farm F	Total
Lapwing	3	13 (7; 6)	9 (7; 2)	5 (3; 2)	15 (6; 9)	45 (26; 19)
Curlew	3	8	23 (21; 2)	3	4	41 (39; 2)
Oystercatcher	25	1	11	5	0	42
Golden plover	0	0	0	0	62	62
Snipe	0	1	0	2	1	4
Redshank	0	1	0	0	0	1
Dunlin	0	0	0	0	22	22
Total	31	24 (18; 6)	43 (39; 4)	15 (13; 2)	104 (95; 9)	217 (196; 21)
Species richness	3	5	3	4	5	7
Simpson diversity index (D)	0.33	0.59	0.60	0.72	0.58	-
Simpson evenness index (E)	0.46	0.36	0.61	0.51	0.36	-

4.4.2 Effect of lime treatment on wader density

After accounting for variation between sites, there was strong evidence for a main effect of treatment on wader density, but no evidence for an effect of year on wader density (Table 4.5, Table 4.6; Figure 4.3). Wader density was, on average, 50% higher on lime treatment than on control, representing a main effect of 0.7 waders ha^{-1} (control: 1.4 waders ha^{-1} with 95% CI: 0.96 – 2.04; versus lime treatment: 2.1 waders ha^{-1} with 95% CI: 1.44 – 2.94; z = -2.79; p value = 0.005). Wader density was also positively associated with the proportion of rush cover, with a higher rush cover associated with a higher wader density (5% rush cover = density of 0.38 waders ha^{-1} with 95% CI: 0.19 – 0.74; compared with 30% rush cover and a density of 1.50 waders ha^{-1} with 95% CI: 0.74 – 3.90). The model explains a high proportion of variance in wader density ha^{-1} (R^2_M = 0.337; R^2_C = 0.517).

Table 4.5. Summary table showing the final (Generalised) Linear Mixed Models selected for analysis on the effects of lime treatment on wader density. Sample size was 68 observation per treatment from five farms (n = 12 fields) over three years, although the number of farms and fields varied by year resulting in a total of 68 observations per treatment.

Model response variable	Type	Unit replication	Link	Error Structure	Random Effect	Fixed Effects
Wader max count	Count + offset of log(Area)	Treatment plot	Log	Poisson	Field Identity	Farm + Treatment + Rush cover

Table 4.6. Summary table of the GLMMs investigating the relationship between lime treatment and wader count at the treatment plot scale while controlling for other variables of importance. Sample size was from five farms (n = 12 fields) over three years, although the number of farms and fields varied by year resulting in a total of 68 observations per treatment. Table shows results of the minimum adequate models with significance levels denoted by *'s where: * ≤ 0.05; ** ≤ 0.01; *** ≤ 0.001.

Predictor	Type (df)	Parameter estimates ± SE (test statistic) of waders
Farm	Factor (df =4)	- (F = 3.97**)
Treatment (displayed as Lime compared to Control)	Factor (df =1)	0.38 ± 0.14 (z = 2.79**)
Rush cover (%)	Covariate (df =1)	0.52 ± 0.20 (z = 2.68*)

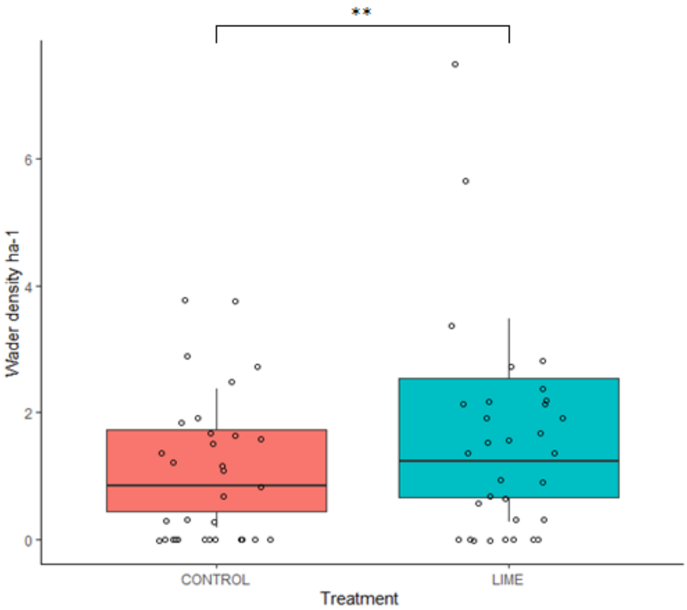


Figure 4.3. Boxplot showing the effect of Treatment (red = control; blue = lime treatment) on wader density ha^{-1}. Sample size was from five farms (n = 12 fields) over three years, although the number of farms and fields varied by year resulting in a total of 68 observations per treatment. Raw data is represented by open black circles, while model predictions (conditional taking into account random effects) from the GLMM shown as a boxplot with median as a black line and whiskers representing values 1.5*IQR. Significance of pairwise comparisons of the estimated marginal means denoted by *'s where: * ≤ 0.05; ** ≤ 0.01; *** ≤ 0.001.

4.4.3 Effect of lime treatment on wader feeding metrics

After controlling for variables of importance, there was no evidence for an effect of treatment on wader peck rate (Table 4.7, Table 4.8). However, there was significant difference in wader peck rate between species which varied by year (Figure 4.4; see Table A0-7 in Appendix for the EMMs). Within species, the peck rate was similar between years. However, between species the peck rate was significantly different in year 2016 with: Curlew peck rate higher than that of Golden plover (t = 4.58; p = 0.0005) and Lapwing (t = 4.57; p = 0.0007); and Oystercatcher peck rate higher than that of Golden plover (t = 4.20; p = 0.0022). There was no difference in peck rate between species in the other years surveyed. A relationship between peck rate and vegetation height was identified with shorter vegetation having a higher peck rate than longer vegetation (Figure 4.5). Peck rate was also positively associated with the ground bumpiness index; bumpier ground had a higher peck rate than more uniform flatter ground. Furthermore, the presence or absence of chicks had a large effect size on peck rate with birds foraging without chicks having almost a quadruple peck rate than birds foraging with chicks likely due to parental vigilance (chicks present: 1.5 pecks min^{-1} with 95% CI: 0.97 – 1.37; versus chicks absent: 5.6 pecks min^{-1} with 95% CI: 4.72 – 6.64; t = 5.94; p value = <0.0001). The marginal (R^2_M) and conditional (R^2_C) values for this model were 0.358 and 0.378 respectively.

For intake rate, there was evidence for an effect of treatment which varied by year (Table 4.8; Figure 4.5; EMMS in Table A0-8 in Appendix). Within years, the pairwise comparisons between control and lime treatment were non-significant in 2015 and 2016; however, by 2017 an average bird was estimated to have double the intake rate on lime treatment than on control (control: 0.32 min^{-1} with 95% CI: 0.20 – 0.53; versus lime treatment: 0.64 min^{-1} with 95% CI: 0.40 – 1.02; t = 3.25; p = 0.0168). Between years there was evidence for intake rate on control plots to be different with 2016 having a higher intake rate than 2017. There was also an effect of species on intake rate which varied by year. Within species, intake rate was similar between years for three of the waders, but Curlew intake rate was higher in 2016 than in the other two years surveyed. Within years, intake rate was higher for Curlew than Lapwing in 2016. Intake rate was also positively associated with soil organic matter content (Figure 4.7). The presence or absence of chicks had a large effect size on intake rate, with waders observed without chick(s) having an intake rate of eight times more per minute than waders observed with chicks (chicks present: 0.2 min^{-1} with 95% CI: 0.14 – 0.42; versus chicks absent: 1.6 min^{-1} with 95% CI: 1.32 – 1.92; t = 6.71; p value = <0.0001). The marginal (R^2_M) and conditional (R^2_C) values for this model were 0.420 and 0.432 respectively.

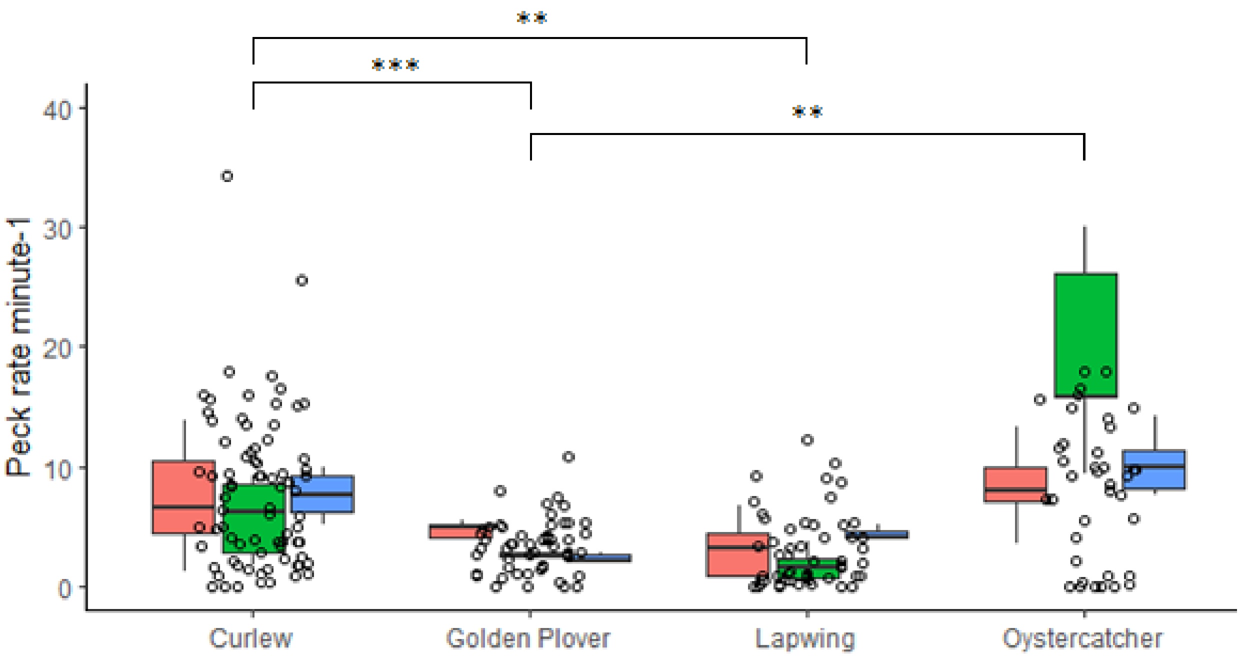


Figure 4.4. Boxplot graphically representing the interaction effect between wader species and year (red = 2015; green = 2016; blue = 2017) on peck rate $minute^{-1}$. Raw data is represented by open black circles (n = 243 observations), while model predictions (conditional) are shown as boxplots displaying the IQR, medium as a black line and whiskers for 1.5*IQR. Significant pairwise comparisons of the estimated marginal means are shown where: * ≤ 0.05; ** ≤ 0.01; *** ≤ 0.001. Please note that three raw data points were removed from graphic (Oystercatcher) to ease comparison between species.

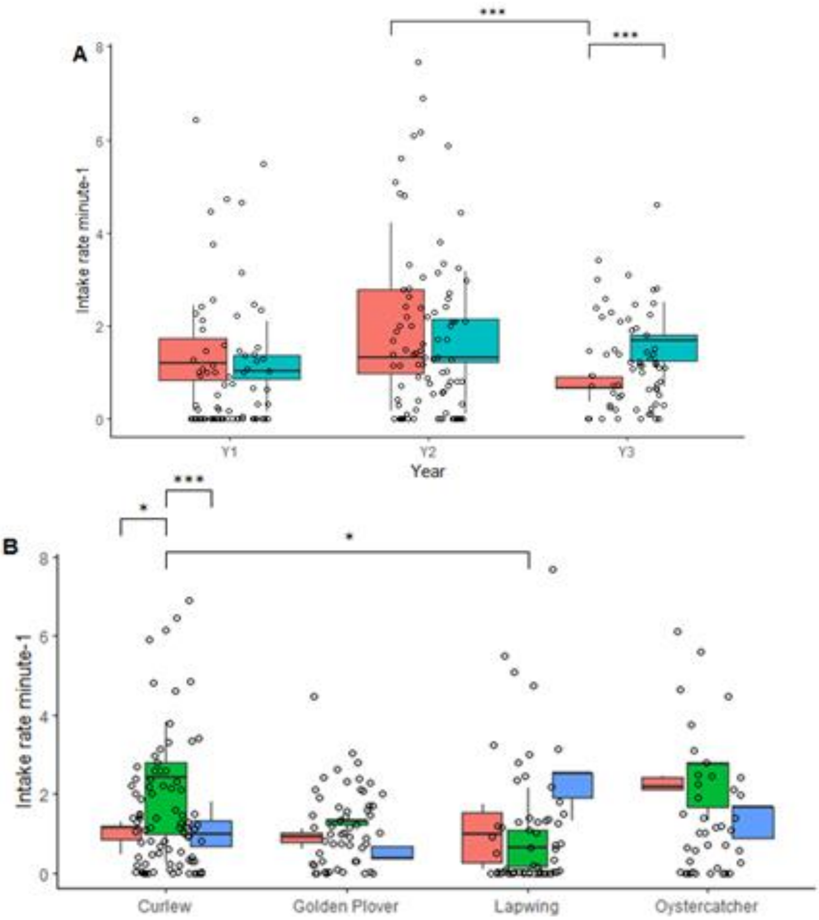


Figure 4.5. Boxplot graphically representing intake rate minute^{-1} and (A) the interaction effect between treatment and year (Y1 = 2015; Y2 = 2016; Y3 = 2017) and (B) the interaction effect between wader species and year (red = 2015; green = 2016; blue = 2017). Raw data is represented by open black circles (n = 243 observations), while model predictions (conditional) are shown as boxplots displaying the IQR, medium as a black line and whiskers for 1.5*IQR. Significant pairwise comparisons of the estimated marginal means are shown where: *≤ 0.05; **≤ 0.01; ***≤ 0.001.

For peck success rate, after controlling for other variables of importance, there was evidence for an interaction effect of Year and Treatment (Table 4.8; Figure 4.6; Appendix Table A0-9). There was no difference in peck success rate by treatment within years, but this varied between years with, on average, birds on control plots in 2016 having a greater peck success rate than in 2015 (control 2015: 18.4% peck success with 95% CI: 13.2 – 25.0; versus control 2016: 40.4% peck success with 95% CI: 32.0 – 49.0%; t = 4.25; p = 0.0003). There was also evidence for an effect of species on peck success rate which varied by year. Within species: Curlew had a higher peck success rate in year 2016 than in year 2017 (t = 7.44; p = 0.0001); Golden plover had a higher peck success rate in year 2016 than in year 2017 (t = 4.21; p = 0.0015); and Golden plover had a higher peck success rate in year 2016 than in year 2015 (t = 6.17; p = <0.0001). Between species: in year 2017 Lapwing had a higher peck success rate than Curlew (t = 3.46; p = 0.0267) and Oystercatcher (t = 3.36; p = 0.0380); and Golden plover had a higher peck success rate than Oystercatcher (t = 3.45; p = 0.0276). The peck success rate was also positively associated with soil organic matter content, and inversely associated with the ground bumpiness (Figure 4.7). The marginal (R^2_M) and conditional (R^2_C) values for this model were 0.107 and 0.185 respectively.

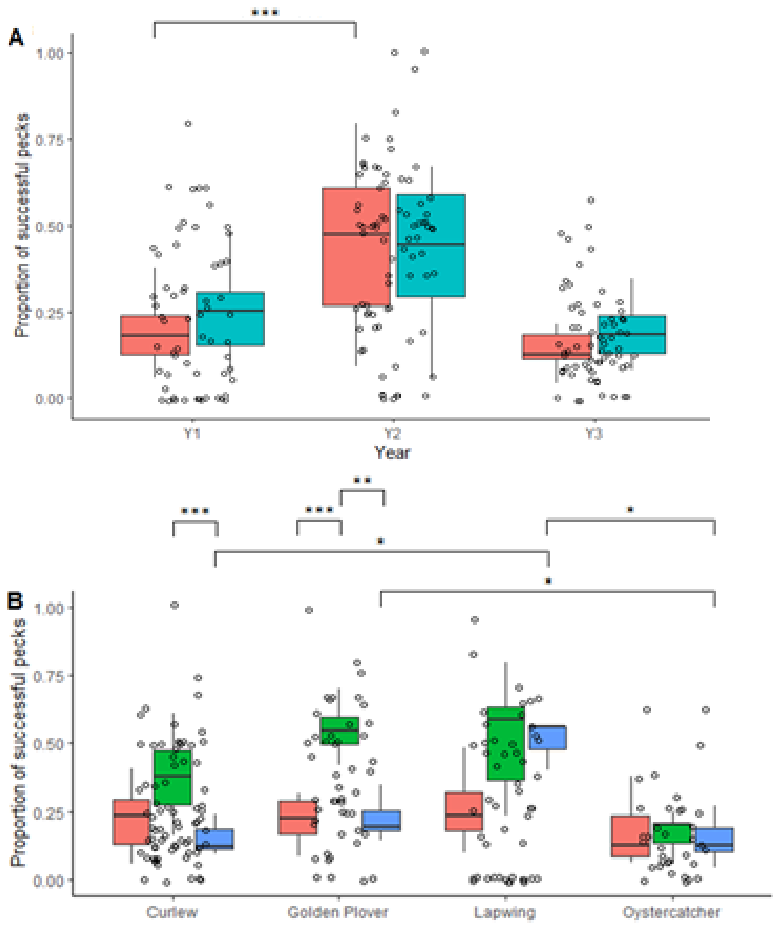


Figure 4.6. Boxplot graphically representing intake rate minute^{-1} and **(A)** the interaction effect between treatment and year (Y1= 2015; Y2 = 2016; Y3 = 2017) and **(B)** the interaction effect between wader species and year (red = 2015; green = 2016; blue = 2017). Raw data is represented by open black circles (n = 243 observations), while model predictions (conditional) are shown as boxplots displaying the IQR, medium as a black line and whiskers for 1.5*IQR. Significant pairwise comparisons of the estimated marginal means are shown where: * ≤ 0.05; ** ≤ 0.01; *** ≤ 0.001.

Table 4.7. Summary table showing the final (Generalised) Linear Mixed Models selected for analysis on the effects of lime treatment on wader foraging metrics. Sample size is 243 bird observations from five farms (n = 12 fields) and three years of monitoring (2015, 2016, 2017) although the number of farms survey in the final year was only 3 farms.

Model response variable	Type	Unit replication	Link	Error Structure	Random Effect	Fixed Effects
Peck rate	Count + offset of log(Time)	Individual bird	Log	Zero-inflated negative binomial	Field Identity + Bird identity	Farm + Species + Year + Year:Species + vegetation height + ground bumpiness + chicks
Intake rate	Count + offset of log(Time)	Individual bird	Log	Zero-inflated negative binomial	Field Identity + Bird identity	Farm + Species + Year + Year:Species + Treatment + Treatment:Year + chicks + soil organic matter
Peck success rate	Proportion (Successes, Failures)	Individual bird	Logit	Binomial	Field Identity + Bird identity	Farm + Year + Species + Year:Species + Treatment + Treatment:Year + soil organic matter + ground bumpiness

Table 4.8. Summary table showing the GLMMs investigating the relationship between lime treatment and wader feeding metrics at the individual bird scale while controlling for other variables of importance. Sample size is 243 bird observations from five farms (n = 12 fields) and three years of monitoring (2015, 2016, 2017) although the number of farms survey in the final year was only 3 farms. Table shows results of the minimum adequate models with significance levels denoted by *'s where: * ≤ 0.05; ** ≤ 0.01; *** ≤ 0.001.

Predictor	Type (df)	Parameter estimates ± SE (test statistic)		
		Peck rate $minute^{-1}$	Intake rate $minute^{-1}$	Success rate
Farm	Factor (df =4)	- (χ^2 = 17.07*)	- (χ^2 = 13.45**)	- (F = 4.79)
Species	Factor (df =3)	- (χ^2 = 39.97***)	- (χ^2 = 22.53**)	- (F = 3.62*)
Year	Factor (df =2)	- (χ^2 = 18.20*)	- (χ^2 = 50.32***)	- (F = 53.47***)
Year : Species	Factor (df =6)	- (χ^2 = 11.78*)	- (χ^2 = 14.44*)	- (F = 3.83***)
Treatment	Factor (df =1)	-	- (χ^2 = 15.26**)	- (F = 2.02)
Year : Treatment	Factor (df = 2)	-	- (χ^2 = 15.19***)	- (F = 2.42*)
Vegetation height	Covariate (df =1)	-0.20 ± 0.08 (z = -2.48*)	-	-
Ground bumpiness	Covariate (df =1)	0.32 ± 0.15 (z = 2.10*)	-	-0.53 ± 0.16 (z = -3.29***)
Chicks present/absent	Covariate (df =1)	- (χ^2 = 29.19***)	- (χ^2 = 33.94***)	-
Soil organic matter content	Covariate (df =1)	-	0.33 ± 0.10 (z = 3.11*)	0.33 ± 0.11 (z = 2.90**)

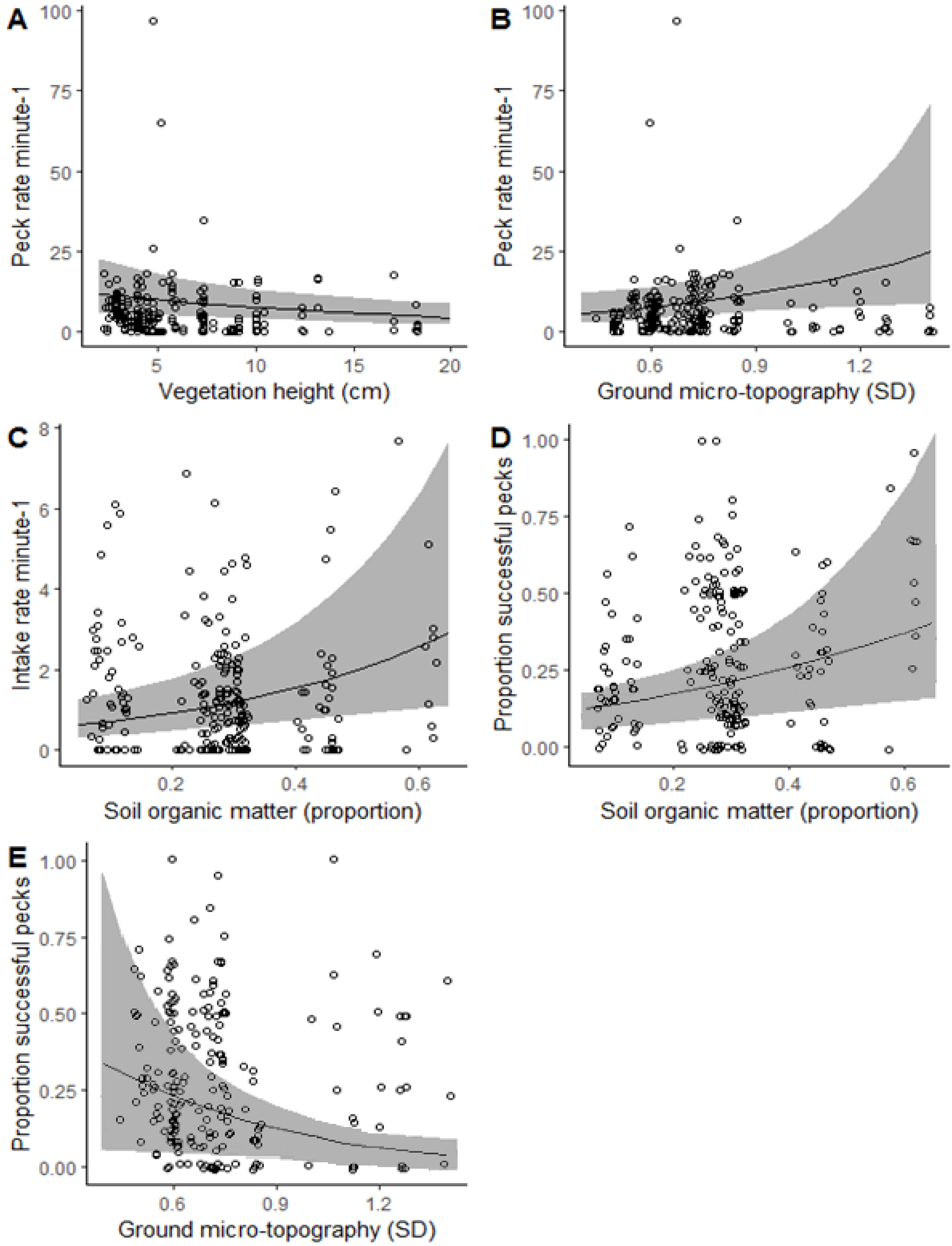


Figure 4.7. Scatterplot graphically representing the effect of significant covariates on wader feeding metrics with the marginal mean estimate shown as a solid black line with 95% confidence intervals shown in grey for **(A)** vegetation height (cm) on peck rate minute^{-1}; **(B)** ground micro-topography (i.e. ground bumpiness) on peck rate minute^{-1}; **(C)** soil organic matter content (%) on intake rate minute-1; **(D)** soil organic matter content (%) on the proportion of successful pecks; and **(E)** ground micro-topography (i.e. ground bumpiness) on the proportion of successful pecks. Sample size is five farms (n = 12 fields) over three years from 243 bird observations.

4.5 DISCUSSION

The addition of soil pH amendment (via lime use) has been suggested as a potential tool to be included in the options available in conservation management interventions for the benefit of farmland breeding waders (McCallum *et al.*, 2016). Existing AES measures for waders are focussed on manipulating vegetation, stocking rates and hydrology, but the soil itself is not specifically targeted (with the exception of soil moisture) despite wader diet consisting of soil invertebrates which are pH-sensitive (e.g. Curry, 2004). The aim of this study was to test the effect of lime use on grasslands in marginal upland areas on wader density and on foraging behaviour through the use of three feeding metrics (peck rate minute^{-1}, intake rate minute^{-1} and peck success rate) at the species level. Here I showed that total wader density was 50% greater on lime treatment than control (non-limed) from spot counts. Focal observations of birds showed an effect of lime treatment on wader intake rate minute^{-1} and peck success rate which varied by year but not by species suggesting that the response was consistent across the four wader species despite differences in feeding ecology. Within years, intake rate minute^{-1} of an average bird was double on lime treatment than that of control for year 2017; but there was no difference in the previous years of monitoring. I also demonstrated that there was an effect of species on intake rate minute^{-1} and peck success rate which varied by year. The year 2016, on average, had a higher intake rate minute^{-1} for Curlew, and a higher proportion of successful pecks for Curlew and Golden plover, compared with the other years of monitoring. There was no effect of treatment on peck rate but there was an effect of species which varied by year. In terms of habitat characteristics both intake rate and peck success rate were higher on soil which had a higher organic matter content. Peck success rate was also higher on less bumpy ground. Furthermore, peck rate minute^{-1} was greater in shorter vegetation.

The results from this study provide evidence to support the hypothesis that lime use could be a targeted conservation tool for the benefit of farmland breeding waders in marginal upland areas by increasing field densities of these birds and improving foraging conditions by increasing intake rate of prey items and by increasing the proportion of successful pecks taken. The effect of lime on wader density is immediate (occurs the year following lime use), while that of feeding metrics show a time lag of a couple of years. Lime use would thus be important in areas where localised acidification may have occurred over time, resulting in a decline in abundance of soil invertebrates. Managing fields for vegetation structure (e.g. short grass) is also likely to be beneficial for waders which is likely to improve accessibility of prey items. Overall, including soil properties into studies on waders can provide insight into an overlooked aspect of their ecology.

4.5.1 The effects of lime treatment on wader density

In support of hypothesis 1, I found that there was strong evidence for a main effect of lime treatment on wader density of 0.7 waders ha^{-1}, which equated to an average mean estimate of 50% more waders on lime treatment than control. Although data was not available for year 2014 (before lime was applied to treatment), after controlling for other variables of importance, lime treatment held higher wader densities than control in all years surveyed (2015, 2016, and 2017). It is unlikely that the difference between lime treatment and control was not because of lime use. Soil properties (pH, moisture, organic matter content, penetrability) and habitat variables (e.g. vegetation height) were shown to be the same between lime treatment and control in year 2014 (see chapters 2 and 3). Furthermore, there was no effect detected of lime treatment on any of these variables in subsequent years after the application of lime in the previous chapters.

In Figure 4.3 there are two individual observations (raw data) on wader density ha^{-1} that appear to depart from the remainder of the data set for lime treatment. Outliers can influence linear regression model output, with the potential for extreme outliers to dominate the outcome and partially (or fully) obscure the main trend. Thus the identification of outliers is a necessary step in data exploration and model building in order to obtain appropriate model results. In this instance, model residuals were explored to identify the presence of problematic outliers. Model residuals did not show an extreme event, however, these are not always the best approach for checking for outliers and is something that will need to be considered more fully in future analyses for journal publication.

Studies investigating field use of birds following lime use is rare e.g.(Brandsma, 2004), and often combined with other grassland field improvements such as inorganic fertiliser input and drainage e.g.(Baines, 1988; Vickery *et al.*, 2001) or other agricultural practises such as fodder crop management and reseeding e.g.(McCallum *et al.*, 2018). The addition of inorganic fertilizers on invertebrates tends to decrease the numbers and diversity of grassland invertebrates (Fenner and Palmer, 1998; Vickery *et al.*, 2001) and at high levels earthworms are also negatively affected e.g.(Edwards, Butler and Lofty, 1975; Edwards and Lofty, 1982); while fodder crop management leads to other habitat changes such as increase in bare ground which is attractive to nesting Lapwing. This therefore makes it difficult to tease apart the effects of lime use on its own. However, field use by Lapwing has been reported to increase in one study following lime use on upland pasture, along with an increase in earthworm abundance (Brandsma, 2004). Additionally, reseeded grass ley fields were observed to hold higher densities of

Lapwing above levels of control fields for approximately seven years after they were reseeded, and pH of these fields remained higher than control, suggesting that Lapwing may have been responding to the use of lime (McCallum *et al.*, 2016, 2018). The results presented in this chapter therefore provide empirical evidence that field use by waders on marginal enclosed grassland is higher for at least three years following the application of lime. This effect is not expected to continue indefinitely, as soil pH will gradually acidify with future applications of lime required to maintain more neutral soil pH.

It is likely that field use by waders increases following lime use due to the increase in earthworm abundance (chapter 2) as a result of the less acidic soil. However, it is unclear how this is being detected by waders, particularly as there was no evidence in previous chapters that lime treatment influenced habitat variables.

Wader density was also positively associated with rush cover. This is consistent with established relationships between waders and habitat with previous studies also reporting wader density to be positively correlated with rush cover for Curlew e.g.(Baines, 1988), and for Lapwing e.g. (O'Brien, 2001) although when a threshold level of rush cover is exceeded (between 15 and 40%) then Lapwing decline. In this chapter, the range in rush cover recorded was between zero and 30%, with a mean of 13%, and therefore the range falls below the threshold rush level, above which a field becomes less suitable for breeding Lapwing.

4.5.2 The effects of lime treatment on wader feeding metrics

In support of hypothesis 2, I found some evidence of an effect of lime treatment on wader foraging behaviour as determined by the three metrics of foraging success. There was no evidence to suggest that wader peck rate minute^{-1} was influenced by treatment. However, there was strong evidence of an effect of species which varied by year, although the peck rate within individual species was similar between the years. The differences between species is expected as waders are morphologically quite different from each other and have different methods of feeding. The Curlew bill length is the longest of the waders recorded in this study and its length varies by sex; males have an average bill length of 115.5 mm, while females have a longer bill of 145.0 mm (Summers *et al.*, 2013). The length of the Oystercatcher bill has a mean which ranges between 70 – 80 mm depending on age, sex, and diet specialization in the non-breeding period (Swennen *et al.*, 1983; Van De Poll *et al.*, 2009). Lapwing and Golden plover, on the other hand, have shorter bills of 22 – 26 mm and 21 – 26 mm respectively (Cramp and Simmons, 1983; Shrubb, 2007). As a result of these differences in bill length, Curlew and Oystercatcher can probe deep within the soil in search of prey. While Lapwing and Golden plover are restricted to the soil surface and just below the ground. Oystercatcher, and to a lesser extent, Curlew, have a drilling behaviour to their soil probing which can result in many pecks within a short time period. Lapwing and Golden plover rely more on visual cues as they hunt for prey on the surface and take fewer pecks with longer periods of time spent searching and pausing between pecks. This difference is reflected in the results I presented here, which is most marked in the year 2016 with Oystercatcher and Curlew having a significantly higher peck rate than Lapwing and Golden plover. The year 2016 was characterised by having a very wet winter with several storms, followed by a drier summer (Kendon *et al.,* 2016; McCarthy *et al.,* 2016). These weather patterns could have influenced the invertebrates which waders prey upon resulting in the particularly large difference in peck rate observed between species in 2016. For example, tipulid larvae and earthworm abundance and distribution can be influenced by soil moisture (McCracken, Foster and Kelly, 1995; Curry, 2004; Carrol *et al.*, 2011; Carroll, 2012; Martay and Pearce-Higgins, 2018), and are important food items for waders. Some earthworm species will move deeper into the soil when conditions are drier and warmer to avoid drying out (Edwards, Hendrix and Arancon, 1996; Perreault and Whalen, 2006). In these conditions, long billed waders such as Curlew and Oystercatcher will be at an advantage as they can access food deeper in the soil profile and may therefore take more pecks than a species restricted to foraging on the soil surface only.

There was evidence for an effect of lime treatment on both wader intake rate minute^{-1} and on wader peck success rate which varied by year but not by species. For peck success rate, the pairwise comparisons of the estimated marginal means showed no significant difference between control and lime treatment within years. Instead the significant pairwise comparisons were primarily for between years, notably that peck success rate in 2016 was more than doubled from its rate in 2015 on the control plots, and then was estimated to be similar to 2015 again in 2017. A similar observation of higher peck success rate was also estimated for 2016 on the lime treatment plots, but this difference between years was non-significant. For intake rate, an average bird was estimated to take double the number of prey items on lime treatment than on control plot in 2017, but there was no difference in the previous years of monitoring. This suggests that by 2017, all four of the wader species had greater consumption of large prey items when foraging on lime treatment compared with control. It is possible that it takes several years following lime use for an observable difference between treatments to be detected in bird intake rate, as invertebrate prey may need time to respond to the change in soil pH. Additionally, the feeding metric of intake rate is likely to be influenced by the size of invertebrates consumed, and therefore differences between plots in terms of intake rate may not have been visible to the surveyor in earlier years if small items were primarily consumed. Earthworm response to lime use varies between the ecotypes with increases reported primarily of epigeic species (small worms on the soil surface) as a surface dressing of lime takes time for the neutralisation of $CaCO_3$ to penetrate deeper into the soil and thus manipulate pH and influence larger earthworms deeper in the soil profile e.g.(Deleporte and Tillier, 1999; Bishop *et al.*, 2008; McCallum, 2012). It is therefore possible that the power to identify a response to lime use, due to an increase in food items (e.g. earthworm) is low from observational studies of foraging birds in the first few years following lime use if this increase is predominantly of small earthworms.

There was also evidence of an effect of species on both wader intake rate and peck success rate which varied by year. This is likely due to the morphology and feeding behavioural differences between the species as described above. However, it should be noted that the error for intake rate (and thus peck success rate) may be higher for Curlew and Oystercatcher, as the consumption of small and medium prey items can be achieved while the bill is probing within the soil, making food intake cryptic and masking the full foraging success of these birds. Indeed, pairwise comparisons of bird species within years showed that Golden plover and Lapwing had a significantly higher success rate than Curlew and Oystercatcher in 2016 and 2017, which could be due to this difference in feeding. However, it could also be due to increases in the epigeic earthworms following lime use which would be more accessible to Golden plover and Lapwing which predominantly take prey items from the soil surface. There were also significant difference between years for individual species identified, with Curlew estimated to have a higher peck success rate in 2016 than in the other years, and for Golden plover to have a higher peck success rate in 2016 than other years. Peck success rate for Lapwing was, on average, higher with every consecutive year surveyed; however this was marginally non-significant between the years (Appendix item Table A0-8). As discussed earlier, the peck rate of Curlew and Golden Plover were also high in 2016. These two feeding metrics are likely related to some extent. The high number of pecks taken and high proportion of successful pecks suggest that this was a particularly favourable foraging year for Curlew and Golden Plover, possibly influenced by the weather anomaly of the year 2016.

There was evidence to suggest that habitat variables influenced the three feeding metrics. After controlling for other variables of importance, peck rate was higher in shorter vegetation. Many studies have demonstrated habitat associations of waders, including a preference for short swards for some species such as Lapwing e.g. (Perkins *et al.*, 2000; Milsom *et al.*, 2002; Devereux *et al.*, 2004; Durant *et al.*, 2007). Short swards are likely to facilitate surface prey detection for visual predators, improve mobility within the field (particularly of chicks with short legs; Devereux et al., 2004) and alter vigilance patterns as it is easier for adult waders to scan for predators in short vegetation. This may result in more time spent actively pecking. There was also a relationship between the bumpiness of the ground and wader peck rate, with bumpier ground resulting in more pecks being taken (although there was a high confidence interval on this estimate). The micro-topography of the ground may be more favourable for foraging habitat due to micro-habitats created for invertebrate prey, compared with a uniform and flat ground structure. For both intake rate and peck success rate, soil organic matter (SOM) was positively associated with the number of prey items consumed per minute/success rate of pecks. In this study, the mean SOM was around 30%, but ranged from low (SOM: 7.2%) to very peaty soil (SOM: 62%). Soil fauna play an important role in nutrient cycling, with detritivores such as earthworms breaking up organic matter and returning it into the soil (Lavelle *et al.*, 2006). However, at very high levels of SOM a negative relationship with earthworms is observed. It is therefore likely that SOM had a positive relationship in this study on intake rate and success rate as the threshold value upon which SOM becomes unfavourable to earthworms were not crossed in this chapter on the subset of farms included in the wader feeding study, or that other soil invertebrate prey where still abundant and accessible at high SOM levels. A relationship between the peck success rate and ground bumpiness was also identified in this study, and had a negative effect on peck success. This was initially surprising (and counter intuitive) given that the number of pecks taken was positively associated with ground bumpiness. However, habitat structure is likely to influence abundance of invertebrate prey as well as the accessibility. It is plausible that abundance was greater in the more structurally diverse habitat leading to higher peck rates, but less accessible in the bumpy ground as invertebrates can hide and escape capture more easily resulting in a decrease in the peck success rate.

The foraging metrics used in this study will have varied in robustness. For example, the number of prey items consumed will be related to the number of pecks taken; with more prey eaten when more pecks are taken. The two foraging metrics of intake rate min^{-1} and peck success rate is therefore likely to be influenced directly by factors determining the overall peck rate min^{-1}. Furthermore, the number of prey items taken will be influenced by human error as smaller prey items consumed may not be visible to the field surveyor. This human error may also vary by species being observed and should be considered when interpreting the results of this study. Thus intake rate min^{-1} and success rate of pecks are likely to be underestimated, and should be interpreted as a conservative indicator of the successful consumption of large prey items. Therefore, peck rate min^{-1} could be considered as a more robust foraging metric as it will likely be less affected by human error, and should still capture factors also influencing foraging success. However, not all pecks will be successful and therefore feeding success is still an important variable which can provide useful biological information. For example, Goss-Custard (1970) found that peck rate was independent of prey density in intertidal zones for Redshank, but that there was a relationship between peck success rate and prey density in some of the zones with feeding success significant higher in area of high prey density than low density. This sort of information is important in understanding habitat management for the conservation of species. Indeed, peck success rate has been shown to be an important variable in feeding behaviour when determining the success of habitat restoration projects for waders (Mander, Marie-Orleach and Elliott, 2013). The three foraging metrics used in this chapter were therefore considered important to the lime use study which was manipulating the soil and potentially influencing prey abundance.

There was no effect of soil moisture, nor of soil penetrability on the feeding metrics. I had predicted that these variables would influence foraging as waders are highly associated with wet features, and the soil invertebrates they feed on are influenced by soil moisture. However, the moisture levels recorded in this study were relatively high, and it may be that soil moisture was not a limiting factor here.

4.5.3 Conclusions, conservation implication, and further research

The decline of waders in lowland grasslands has enhanced the relative importance of upland areas as remaining, valuable strongholds for breeding waders in the UK. However, these marginal areas of enclosed farmland lying below the moorland, may be prone to acidification due to leaching effects in areas of high rainfall common for the Atlantic climate of Scotland, slope of the landscape and non-buffering geology, resulting in a decline in pH-sensitive soil invertebrates over time which contribute a high proportion of wader diet. In this chapter, I showed that the lime treatment plots had 50% higher wader densities than control plots, and that wader prey intake rate of large prey items was significantly higher on the lime treatment than control plots several years after the application of lime. This is likely to be a conservative estimate of food intake as smaller items will be consumed cryptically. In chapter 2 of this thesis, I demonstrated that soil pH was amended by lime use, and that total earthworm abundance was positively associated with soil pH, as well as showing that an increase in total earthworm abundance and of total epigeic earthworm abundance was influenced by lime. Epigeic earthworms are small and easily accessible on the soil surface to waders, which can make it difficult to successfully interpret waders feeding on these prey items from observational studies. It is plausible that the main effect of lime treatment on wader density in this study is in response to pH change following lime use, and thus improved foraging success from earthworms, even though this is not detected until year 2017 from feeding observations. Furthermore, in chapter 3, I showed that there was no detrimental effect of lime use on other arthropods which are food items of waders (spiders: Arachnida; and beetles: Coleoptera) which provides further evidence that the response of waders is likely due to differences in earthworm abundance brought about by change in soil pH. Further research from faecal analysis of foraging waders on lime treatment versus control would provide important evidence to support this hypothesis.

Habitat structure of fields is likely to influence the accessibility of invertebrate prey and thus intake rates of foraging birds (Whittingham and Markland, 2002). Changes in grassland management, such as the use of lime could increase sward growth and density, which could decrease the foraging success of grassland birds that forage on surface and soil invertebrates as prey is likely to be easier to detect and capture in shorter and more open vegetation. Indeed in this study I detected a significant inverse relationship between wader peck rate and vegetation height, with pecking decreasing with increasing vegetation height. The fields used in this study were actively grazed, and during the course of the four years of monitoring there was no evidence to suggest an effect of lime treatment on vegetation height or vegetation heterogeneity (chapter 2). It is likely that in these systems, the positive impact of lime on grass productivity is out-weighed by that of livestock grazing – although changes in grazing management of fields following lime use would affect this. In this study I did not measure vegetation density or plant species composition which can be influenced by the change in availability of nutrients in the soil to plants following liming e.g.(Kirkham *et al.*, 2008; Yu, Fraser and Evans, 2011; Goulding, 2016) and thus may have impacts on abundance and accessibility of invertebrates. In general, marginal enclosed grasslands are not botanically rich, but it would be an area of research to consider if lime use was to be recommended as a conservation tool.

In summary, changes in food supply as a result of changes in farming practices have been proposed as an important factor in the declines of many farmland bird species (Benton *et al.*, 2002; Robinson and Sutherland, 2002; Newton, 2004). The wader communities of Scotland on marginal upland grasslands are important strongholds for supporting the UK breeding populations of waders, where they are still relatively widespread and common. However, upland areas receive high levels of rainfall and are characterised by typically peaty soils with a low buffer capacity and a high organic matter content that results in these areas being particularly prone to acidification over time (White, 2006; Aitkenhead *et al.*, 2012). This may have important implications for the conservation of farmland breeding waders as localized acidification, potentially related to a reduction in lime use of upland grassland systems

(Kuylenstierna and Chadwick, 1991; Baxter, Oliver and Archer, 2006; Goulding, 2016) may have reduced the quality of foraging habitat due to lower abundances of earthworms in less neutral soil conditions (Edwards, Hendrix and Arancon, 1996). Here I have shown that wader densities are higher on lime treatment than control, likely due to an increase in prey intake, as a result of pH amendment and thus an increase in pH-sensitive invertebrates. Lime use, therefore, could be an important tool for the benefit of farmland breeding waders, alongside other well established habitat managements for these birds such as livestock grazing to maintain a suitable sward.

CHAPTER 5 | CASE STUDY OF THE EFFECTIVENESS OF AGRI-ENVIRONMENT SCHEMES FOR THE BENEFIT OF FARMLAND BREEDING WADERS: THE IMPLICATIONS OF WOODLAND CREATION.

5.1 ABSTRACT

Within Europe, agri-environment schemes (AES) are the key delivery mechanism for biodiversity conservation on farmland. Schemes have a range of land management options designed to deliver outcomes for target habitats or species. Breeding waders are one group of farmland birds which have suffered dramatic and long-term population declines both within Europe and at the global scale, and are considered to be birds of the wider farmed landscape. In the UK, which supports internationally important numbers of these breeding birds, multiple land management options within AES have been specifically designed to benefit waders on agricultural grasslands. In this chapter, I assess whether habitat variables often associated with these schemes, such as rush cover and water management, are associated with higher densities of four waders species of international conservation concern in the Badenoch and Strathspey region, in the Highlands of Scotland, which is considered a stronghold for these birds. Wader density was significantly different between years, with the density of Lapwing and Redshank 50% lower in 2015 compared with 2000 at the farm scale, and for Curlew were 20% lower in 2015 than 2000 although this was non-significant. At the field scale, taking into account other habitat variables of importance, there was a positive association between wader numbers at the field scale with some, but not all, of the habitat variables that are manipulated by AES with the intention of benefitting waders, although this may have been affected by the crude nature of measures used. Fields with a higher proportion of rush cover were predicted to hold higher abundances of Curlew and Snipe, and fields with larger areas of standing water to hold higher abundances of Redshank and Snipe. Additionally, there was no interaction effect between habitat variables and survey year, indicating that the effect was consistent between years. There was an additive and adverse effect of proximity to woodland edge for all four species with effects apparent over several hundreds of metres. This scale of woodland edge effect is broadly consistent with other wader studies in open-ground habitats which have investigated the effect of conifer plantations, whereas in this study the majority of woodland edge was from broadleaved, suggesting that buffer zones of several hundreds of metres for woodland planting would help to mitigate a negative effect for these species, particularly given the current interests in woodland planting as a climate change mitigation measure.

5.2 INTRODUCTION

Agricultural land accounts for over 40% of the world's ice-free land area, and c.50% of the surface area of the European Union (Clay, 2004; FAOSTAT, 2013; EEA, 2017). Agricultural management and policy, therefore, has the potential to affect large areas of land and impact the rural communities who depend upon it, and the ecosystem services that it provides. The implementation of the Common Agricultural Policy (CAP) in 1957 has played a significant role in driving agricultural intensification and land use change within Europe. These changes varied among countries, but in general agricultural intensification of semi-natural grasslands included drainage, increased use of chemicals, and re-sowing with more productive leys (e.g. Fuller, 1987), whereas on arable crop land there was a shift to autumn sown cereals, and a loss of non-cropped areas such as the removal of hedges to produce larger fields (e.g. Deckers *et al.*, 2005). In contrast, land use change was characterised by the abandonment of farmland and tree planting particularly in Mediterranean regions (e.g. Debussche, Lepart and Dervieux, 1999). The drive for more productive and intensive agriculture has resulted in less intensive traditional practices becoming no longer profitable, and overall this led to a loss of mixed farming (Robinson and Sutherland, 2002) and temporal and spatial homogenization of habitats (Benton, Vickery and Wilson, 2003).

The land use changes and agricultural intensification associated with the CAP have been linked to the decline and range contraction of European farmland wildlife across a broad range of taxa (Krebs *et al.*, 1999; Donald, Green and Heath, 2001; Benton *et al.*, 2002; Robinson and Sutherland, 2002; Newton, 2004; Foley *et al.*, 2005; Smith, Jennings and Harris, 2005; Donald *et al.*, 2006; Butler *et al.*, 2010). However, when extensively managed, agricultural land can be an important resource for supporting local wildlife (Bignal and McCracken, 2000; Tscharntke *et al.*, 2005), with many species of conservation concern in Europe dependent on extensively managed farmland for all or at least part of their life cycle (Halada *et al.*, 2011; Feniuk, Balmford and Green, 2019). Typically these are species of open landscapes which require some level of disturbance in the habitat to prevent habitat succession (Thomas, 1991). Traditional agricultural practices such as extensive grazing and rotational mowing which have occurred for thousands of years within Europe can therefore be valuable management practices for conservation (Sutherland, 2004). Environmental measures, such as Agri-Environment Schemes (AES), introduced into the CAP through the Agricultural Structures Regulation of 1985 (European Regulation 797/85), have been partly designed to address this and are now a major source of nature conservation funding within the European Union. Since 1992 AES has been compulsory for all EU Member States (EU Regulation 2078/92) and are one aspect of the Rural Development pillar of the CAP. The measures available under

AES are voluntary and are designed to mitigate or reverse biodiversity loss, with payments available to support land managers to conduct less intensive agricultural practices as a financial incentive and to offset potential loss of yield (Batáry *et al.*, 2015). Each Member State (in addition to non-EU states such as Switzerland and Norway) designs its own AES, resulting in a wide range of measures which can be targeted at local and national conservation priorities. Uptake of schemes has been high, with approximately 20% of EU farmland within some form of AES (Kleijn and Sutherland, 2003; Donald and Evans, 2006); however, the effect of these schemes on biodiversity conservation has been mixed (e.g. Kleijn *et al.*, 2001; Peach *et al.*, 2001; Breeuwer *et al.*, 2009; Santana *et al.*, 2014; Batáry *et al.*, 2015; Sanderson *et al.*, 2016; Heldbjerg, Sunde and Fox, 2018; Leyrer *et al.*, 2018) and is poorly monitored (Kleijn and Sutherland, 2003). Nevertheless, when appropriately designed and targeted (Evans, Armstrong-Brown and Grice, 2002), AES have the potential to deliver large-scale benefits to wildlife populations on agricultural land (e.g. Aebischer *et al.*, 2000; Peach *et al.*, 2001; Bradbury *et al.*, 2004; Fuller, Hinsley and Swetnam, 2004; Kleijn *et al.*, 2004; Vickery *et al.*, 2004; Perkins *et al.*, 2011). The cost of AES is expensive, with the European budget for the period 2014 - 2020 approximately €363 (Navarro and López-Bao, 2018). Given that financial resources for AES payments are finite, in competition with other demands, and that many priority conservation species are reliant on farmland, ensuring the cost-effectiveness of schemes to deliver their biodiversity targets remains a key priority within Europe (Kleijn and Sutherland, 2003; Sutherland *et al.*, 2006).

Birds have been widely used as indicator species of the health of agricultural environments as they are conspicuous, highly mobile, and have well established and long-term monitoring programmes both at a national (e.g. BTO/RSPB/JNCC Breeding Bird Survey) and European level (e.g. Pan-European Common Bird Monitoring Scheme) which means that changes in bird abundance, range and timings of arrivals can provide useful indication of broad environmental change (IUCN, 2010; Schmeller *et al.*, 2012). Severe declines in many farmland bird populations in Europe have been observed and associated with increases in agricultural intensity (e.g. Donald, Green and Heath, 2001; Foley *et al.*, 2005; Donald *et al.*, 2006; Butler *et al.*, 2010). In particular, one group of farmland birds commonly referred to as farmland breeding waders, notably; the Northern Lapwing (*Vanellus vanellus*), Eurasian Curlew (*Numenius arquata*), Eurasian Oystercatcher (*Haematopus ostralegus*), Common Snipe (*Gallinago gallinago*), Black-tailed Godwit (*Limosa limosa*) and Common Redshank (*Tringa tetanus*) hereafter referred to as Lapwing, Curlew, Oystercatcher, Snipe, and Redshank respectively, have suffered dramatic and long-term population declines in several key breeding populations on agricultural land within Europe, as well as in many wetland habitats globally (Piersma, 1986; Tucker and Heart, 1994; Norris *et al.*, 1998; Thorup, 2004; Wilson, Ausden and Milsom, 2004; Stroud *et al.*, 2006; Teunissen and Soldaat, 2006; Brown *et al.*, 2015; Birdlife International, 2017b; Harris *et al.*, 2020). For example, the Netherlands holds c.40% of the breeding population of Godwits in Europe (Burfield, van Bommel and Gallo-Orsi, 2005) with approximately 85% of these breeding on agricultural grasslands (Kleijn *et al.*, 2010), and has recorded estimated declines of almost 75% from the first national survey conducted in 1967 to 2015 (Mulder, 1972; Kentie *et al.*, 2016). Many wader species, including Lapwing, Curlew and Godwits are therefore classified as Vulnerable to extinction at the European scale due to these on-going declines (Birdlife International, 2015).

Given the severe decline of farmland breeding waders at both a global and European scale, many species are of high conservation priority and subsequently their ecology and habitat requirements have been particularly well studied on agricultural land across a wide range of systems (e.g. Galbraith, 1988; Baines, 1990; Sheldon, 2002; Birdlife International, 2017). Although these waders are often considered shorebirds, some winter on farmland, and they are dependent on farmland for nesting and foraging during the breeding season, particularly agricultural grasslands, although Lapwing are also able to breed on arable land if close to suitable chick habitat such as grasslands and wet features (Galbraith, 1988b; Berg, 1993; Sheldon *et al.*, 2004). Waders are ground nesting birds which select nest-sites with open views often in relatively flat, large fields, and the birds tend to avoid areas with perches (e.g. trees) and field boundaries that restrict their ability to scan for predators (Elliot, 1985; Milsom *et al.*, 2000; Wallander, Isaksson and Lenberg, 2006; Shrubb, 2007). For both coastal and inland grasslands within Europe, livestock grazing is a key driver of habitat quality for waders with Lapwing preferring a short sward whereas species like Curlew and Redshank are more often associated with longer swards with a preference for a higher proportion of tussocks/rush cover within fields than Lapwing (Baines, 1988; Treweek *et al.*, 1997; Valkama, Robertson and Currie, 1998; Tichit, Durant and Kernéïs, 2005; Smart *et al.*, 2006; Durant *et al.*, 2007; Shrubb, 2007). Tall swards may increase the likelihood of predators locating nests, chicks or adults by reducing adults' ability to spot approaching predators, while uniform swards may increase the likelihood of predation by reducing cover and camouflage (Vickery *et al.*, 2001). Small areas of rush patches within fields can, therefore, be beneficial by providing cover from predators for chicks, shelter, and important foraging areas of invertebrate prey (e.g. Baines, 1988). It is generally thought that grazing by cattle results in a more varied vegetation structure due to the mechanism of grazing (e.g. cows tearing off vegetation with the tongue compared to sheep biting/nibbling with the teeth) and behavioural selectiveness of grazing which includes the avoidance of foraging around dung patches and a "less fussy" palate compared with sheep that results in a more diverse grassland habitat including patches of tall and short vegetation suitable for waders (Rook *et al.*, 2004; Tichit, Durant and Kernéïs, 2005; Smart *et al.*, 2006). Waders are also highly associated with wet features (Smart *et al.*, 2006; Rhymer *et al.*, 2010; Schmidt *et al.*, 2017). A higher water table helps to move soil invertebrates closer to the surface and patches of bare wet muddy features created by standing water can provide a mosaic of nesting habitat and important foraging areas (Ausden and Hirons, 2002; Milsom *et al.*, 2002; Ausden *et al.*, 2003; Eglington *et al.*, 2010). As waders are opportunistic foragers which hunt visually for invertebrate prey on vegetation, and both on and within the soil, sward density and soil penetrability are important as they can reduce the accessibility of prey items (Butler and Gillings, 2004; Devereux *et al.*, 2004; Smart *et al.*, 2006).

The recent intensification of agricultural land use has, in general, made conditions less suitable for breeding waders and reduced invertebrate food availability (Shrubb, 2007; Wilson, Vickery and Pendlebury, 2007), as well as potentially increasing the density of predators which can limit breeding success and site occupancy (Grant *et al.*, 1999; Evans, 2004; Macdonald and Bolton, 2008; Teunissen *et al.*, 2008; Roodbergen, Werf and Hötker, 2012; Douglas *et al.*, 2014; Manton *et al.*, 2016; Roos *et al.*, 2018). Targeted conservation measures in the form of AES for the benefit of farmland breeding waders within Europe has been built on the extensive research into their ecology and responses to agricultural intensification (Sheldon, 2002) and have predominantly focussed on two broad prescriptions for grassland management; 1) the manipulation and timings of grazing and cutting to reduce chick mortality and nest loss from livestock trampling or mechanical operation (Teunissen and Soldaat, 2006) and to create a desired sward structure (Beintema and Muskens, 1987; Vickery *et al.*, 2001), and 2) the manipulation of the water table and creation of non-linear wet features (i.e. scrapes) to improve foraging habitat by increasing invertebrate prey (Eglington *et al.*, 2010) and to attract breeding waders to fields (Smart *et al.*, 2006; Eglington, 2008; Fisher *et al.*, 2011). In both cases, farmers typically enter into management agreements on a field-by-field basis. Although farmland birds offer some of the best examples of AES driving population recovery of the target group at regional and/or national levels (Aebischer *et al.*, 2000; Peach *et al.*, 2001; O'Brien, Green and Wilson, 2006; Bretagnolle *et al.*, 2011; Perkins *et al.*, 2011), studies of wader AES across Europe have shown that they often do no better than slow the rate of population declines (Ausden and Hirons, 2002; Kleijn and van Zuijlen, 2004; Kleijn *et al.*, 2004; Ottvall and Smith, 2006) and that wader response to AES is variable between the species (O'Brien and Wilson, 2011).

The effectiveness of AES for farmland breeding waders is particularly important in the United Kingdom (UK), which is a key breeding area for these birds within Europe. However, declines have also been recorded in the UK of 24% for Oystercatcher, 43% for Lapwing, 48% for Curlew and 42% for Redshank between 1995 and 2018 (Harris *et al.*, 2020). These severe declines have been observed across most of the UK, particularly on enclosed lowland farmland in England and Wales, although until recently the uplands of Scotland have been considered to have largely escaped these losses (Baines, 1990; O'Brien, Tharme and Jackson, 2002; O'Brien and White, 2003; Sheldon *et al.*, 2004; Sim *et al.*, 2005; Wilson *et al.*, 2005; Forrester *et al.*, 2007; Shrubb, 2007; Smart *et al.*, 2008) there is now evidence showing declines within Scotland too (Balmer *et al.*, 2013; Bell and Calladine, 2017). Therefore, the farmland breeding wader community of Scotland on marginal upland agricultural grasslands are critically important strongholds in supporting the UKs breeding population. For example, Scotland holds most of the UK's breeding Curlew, and is responsible for almost 30% of the global population of this globally Near Threatened species (O'Brien, 2004; Brown *et al.*, 2015; Birdlife International, 2017a). However, in recent years, the evidence has been increasing for stark declines in wader populations also within Scotland (Taylor and Grant, 2004; Risely *et al.*, 2011; Bell and Calladine, 2017; Harris *et al.*, 2020). Agricultural policy in the UK is a devolved issue, and within Scotland has undergone numerous reforms since it was introduced in 1988 with options for the benefit of farmland breeding waders available from 1993/4 (Table 5.1). The current seven year programme to deliver Pillar 2 of the EU's CAP, known as the Scottish Rural Development Programme (SRDP), has a total budget over 1.3 billion euros, of which 4.5 million euros was allocated for the delivery of AES, and 2.4 million euros for Forest Grant Schemes (FGS; Scottish EU Funding portal, 2016). The SRDP is spatially targeted, and aims to increase the cost-benefit effectiveness of the options available by targeting effort in key areas for maximum benefit. For full methods of the Scottish Governments implementation of the spatially targeted areas see Pakeman and McKeen (2019). One of the spatially targeted areas for farmland breeding wader options is situated in the Badenoch and Strathspey region within the Cairngorms National Park in the Scottish Highlands. The Cairngorms National Park claims to support more farmland breeding waders than Wales, has a quarter of the rare and endangered species in the UK, and contains 24% of Scotland's native forests with almost 50% of the park recognised as being of international importance for nature (Cairngorms National Park Authority, 2020). The Badenoch & Strathspey region, in particular, is important for farmland breeding waders due to the uniquely expansive semi-natural floodplain of the Spey, and the low-intensity mixed farming systems in the area. However, farmland breeding wader surveys conducted in the Badenoch and Strathspey region between 2000 and 2010 showed a decline of 42% in the total breeding population of waders, with a decline of 60% for Lapwing and 55% for Redshank (RSPB, 2012). In recognition of the importance of this area for farmland breeding waders, in 2008 the Strathspey Wetlands and Wader Initiative (SWWI), a local partnership between the RSPB, Cairngorms National Park Authority, and several local agricultural agencies, was initiated with the aim of assisting landowners with the maintenance of good wader habitat and to improve wader habitat in the Strathspey catchment. The SWWI provides site specific conservation management advice and aids in the delivery of AES options targeted at farmland breeding waders. Additionally, the SWWI now conducts farmland breeding wader surveys in the area on a five-year basis, which were initially started by the RSPB in 2000. As a result, targeted conservation effort and monitoring of waders has been high in the Strathspey (RSPB, 2012).

A primary driver of current land use policy is the planting of trees to sequestrate carbon dioxide, CO_2, and thus reach key climate change (net zero) targets as committed to by national Governments, including Scotland (Climate Change Committee (CCC), 2020; Scottish Forestry, 2020b). Woodland creation of both commercial conifer plantations and native broadleaved forestry has also occurred within the Badenoch and Strathspey supported by a range of government grants for creating new woodland and managing existing woodland (Scottish Forestry, 2019). The Woodland Grant Scheme (WGS) was first introduced in 1988 following the phasing out of tax reliefs for commercial conifer planting which resulted in significant tree planting, particularly in the uplands where land values were lower, but landscape and conservation interests were often greater (Avery and Leslie, 1990). The rapid expansion of woodland within the UK between 1940s – 1980s had some well-publicised negative effects in Scotland on biodiversity and conservation priority species, with notable examples being the adverse impacts of conifer planting in the peatlands of Sutherland and Caithness ('The Flow Country'), which resulted in huge direct losses of habitat on the planted ground with estimates that habitat of 5,000 pairs of Curlew were lost (Ratcliffe, 2007). Since the late 1980s, woodland grants have had a greater emphasis on biodiversity targets and rural development in Scotland and have undergone numerous reforms including incentives for native broad-leafed planting (Scottish Forestry, 2020c). Reforestation with native trees is expected to benefit woodland biodiversity in Scotland, including some taxa of conservation importance such as capercaillie (*Tetrao urogallus*) and black grouse (*Lyrurus terix*) (e.g. Quine and Humphrey, 2010; Scridel, Groom and Douglas, 2017). However, there are also likely to be negative impacts on ground-nesting birds as open ground and marginal farmland is often proposed for tree planting. Therefore, despite the move to native woodland planting, newly created woodland will lead to direct habitat loss of long-established wader habitat if planting occurs on suitable sites used for nesting or feeding, although there may be a lag in the effect of woodland planting while the woodland matures (e.g. Cayford, 1993; Pearce-Higgins *et al.*, 2007; Hancock, Grant and Wilson, 2009; White, Warren and Baines, 2013; Wilson *et al.*, 2014; Franks *et al.*, 2017).

There is increasing evidence of "woodland edge effects" on waders where the displacement of breeding waders occurs due to behavioural avoidance of nesting in enclosed fields or near woodland patches (e.g. Wallander, Isaksson and Lenberg, 2006) resulting in previously suitable areas becoming unsuitable for waders. Indeed, several studies have reported indirect effects of conifer plantations on the wader species Golden Plover (*Pluvialis apricaria*), Dunlin (*Calidris alpina*), Greenshank (*Tringa nebularia*), Lapwing, and Curlew from woodland edge effects in the Flow Country (e.g. Stroud *et al.*, 1987; Lavers and Haines-Young, 1997; Sim *et al.*, 2005; Hancock, Grant and Wilson, 2009; Wilson *et al.*, 2014), as well as lower abundances of waders when adjacent to commercial conifer plantations on unenclosed moorland (e.g. Stroud, Reed and Harding, 1990; Pearce-Higgins *et al.*, 2009), and lower abundance of Lapwing nests in lowland wet grasslands than expected by chance within 500 m of woodland patches (Bertholdt *et al.*, 2017). Woodland edge effects also include changes in predator density as mature woodlands may harbour predators resulting in predator activity being greater on open ground adjacent to woodlands which has been reported to have an effect of up to 1km (Valkama and Currie, 1999; Amar *et al.*, 2011; Douglas *et al.*, 2014; Wilson *et al.*, 2014). New woodland causes fragmentation of the remaining open landscape (Ratcliffe, 2007). Nest predation of waders by predators (primarily avian corvid species and mammalian red fox) have been shown to be significantly higher in agricultural landscape fragmented by woodland than that compared with continuous open agricultural landscapes (Valkama and Currie, 1999; Douglas *et al.*, 2014), and reduced breeding success and wader population changes have been shown to be positively associated with the absolute proportional coverage of woodland (broadleaf and coniferous) surrounding breeding sites (Douglas *et al.*, 2014; Wilson *et al.*, 2014; Franks *et al.*, 2017).

With Scotland's pledge to reach 21% woodland cover by 2032 (Scottish Forestry, 2019), tree planting and natural regeneration, could pose a significant threat for waders in key areas, such as the Badenoch and Strathspey. Woodland edge effects could be contributing to local population declines either by avoidance of woodlands by these species or lower survival of individuals breeding close to woodlands, thus limiting the capacity for species recovery. The current evidence base on the effects of woodland expansion in the UK is dominated by research on non-native conifer plantations (for review see Burton *et al.*, 2018), and for waders in particular, have focussed on unenclosed moorland. However, both conifer and native woodland planting is taking place on farmland and the lack of research on the effects of broadleaved woodland on waders represents a significant gap in the literature on the effects of woodland creation on biodiversity. Regardless of the woodland type, planting trees will result in a large shift in habitat type, with the replacement of large areas of open habitat with fragmented woodland. Understanding the ecological effects of woodland expansion on waders is therefore crucial for informing how tree planting should be delivered in sensitive areas particularly in the context of designing future government policy and AES in Scotland post Brexit. Currently, eligibility for wader AES options in

Scotland suggest avoiding selecting fields which are closer than 30m from wooded areas, and restrict planting new trees or hedges within 30m from the field for the duration of the five year agreement. The distance of 30m is relatively short, considering the woodland edge effect observed on waders in unenclosed moorland and peatland which have been found up to 1km (e.g. Douglas *et al.*, 2014; Wilson *et al.*, 2014). Conversely, woodland grant eligibility do not stipulate that they cannot be planted within a set buffer distance from wader habitat (as far as I am aware). This could result in potential conflict between mitigating climate change land use policy and priority species conservation on the same space of land, with fields which rely on AES wader funding becoming no longer eligible if planting occurs within the 30m distance of previously suitable fields. Furthermore, the initial planting grants and annual five year maintenance grants for tree planting, plus the option to sell credits to the UK government for the CO_2 which the forest absorbs, can make trees more profitable than traditional farming in some marginal areas and may result in substantial land use change in these areas (O'Neill *et al.*, 2020).

In this chapter, I will analyse farmland breeding wader survey data collected over a fifteen year time period between 2000 and 2015 in the Badenoch and Strathspey, building on the preliminary population level wader analysis reported for 2000 - 2010 conducted by the SWWI. Specifically, I will test 1) at the farm scale, whether the density of four wader species of international conservation concern: Curlew, Lapwing, Redshank and Snipe, have changed over time. Here I hypothesize that wader density of each species will have declined between each of the consecutive survey periods. 2) I will test at the field scale, the associations between habitat variables characteristically used in AES management options (e.g. the presence of standing water during the breeding season) with the density of each of the four wader species to test the a priori hypothesises that there are positive associations and to determine if this response is variable between species while controlling for other known habitat constraints of importance (e.g. field slope and enclosure score). Furthermore, I will test 3) the association of distance to woodland edge and woodland type (conifer plantation vs native woodland) on farmland breeding wader density, filling a gap in the current wader literature which has predominantly looked at distance to conifer plantation only or not discriminated between conifer vs broadleaf woodland in the surrounding area of breeding sites. Here I hypothesize that there will be a large woodland edge effect on all four of the wader species. Although this chapter is focussed on one regional area in the Badenoch and Strathspey, I expect the results to be of use more widely elsewhere across Scotland where woodland creation is occurring, as well as in upland sites across the UK. I will discuss the results of this chapter in the context of agricultural policy and conservation measures for farmland breeding waders with the aim of identifying broad scale relationships to provide an evidence-based framework from which to develop policy-level solutions and specific conservation measures to help reverse the decline of farmland breeding waders.

Table 5.1. Summary table and brief description of the various implementations of Agri-Environment Schemes (AES) within Scotland, the available targeted options for the benefit of farmland breeding waders, and the years that the schemes were available, as a result of several major reforms to the European Union (EU) Common Agricultural Policy (CAP) in 1992, 2003, and 2013. Information summarised and compiled from legislation.gov.uk; gov.scot; and hutton.ac.uk.

Scheme Name	Key Description	Wader Options	Years Available
Environmentally Sensitive Areas (ESA)	Geographically restricted to ten high conservation value areas within Scotland. Agreements ran for ten years with the option to close after five. Included options to sustain (tier 1 – basic payments to prevent decline and standardised across all ESAs) or adopt and enhance (tier 2 – higher paying management options which differed between ESAs) environmentally friendly farming practices. Options for waders were available from 1993/4.	• Water margin grazing control • Wetland grazing control	1988 – 2000
Countryside Premium Scheme (CPS)	Options now widely available throughout Scotland. Agreements ran for five years and were competitive (i.e. not all applicants accepted and based on a point score). Farmers entering the scheme had to conform to General Environmental Conditions (equivalent to ESA Tier 1 but with no payments) as well as selecting the environmentally friendly farming practices they would manage in exchange for payment.	• Flood plain management • 'Grassland for birds' management • Wetland creation and management	1997 – 2000
Rural Stewardship Scheme (RSS)	Replaced both ESA and CPS with a single scheme delivered through the Scottish Rural Development Plan (SRDP). Agreements ran for five years, were widely available and were based on a competitive point system. Decoupling of payments was introduced through the formation of the Single Farm Payment (SFS) which was dependent on cross-compliance measures such as maintaining land in good agricultural and environmental condition (GAEC) in 2003. Guaranteed entry into RSS was awarded when previous ESA or CPS agreements expired.	• Flood plain management • Grazed grassland for birds • Mown grassland for birds • Wet grassland for waders • Wetland creation and management	2001 – 2006
Agri-Environment Payments (AEP)	Delivered through the Rural Priorities (RP) element of SRDP. Five year agreements awarded based on a competitive point score system which were widely available throughout Scotland. Forestry grants were now also incorporated into the RP scheme.	• Open grazed/wet grassland for wildlife • Maintenance of rough grazing for birds	2007 – 2013
Agri-Environment and Climate schemes (AECS)	Delivered through the RP element of SRDP. Five year agreements awarded based on a competitive point score with options now spatially targeted. SFPs reformed to a new direct payment scheme made up of a basic payment and a greening payment where 30% of the overall payment was linked to 3 greening requirements: crop diversification, maintenance of permanent grasslands, and the need to establish Ecological Focus Areas on at least 5% of arable land.	• Wader and wildlife mown grassland • Wader grazed grassland • Predator control (Sites of Special Scientific Interest and Natura sites only)	2014 – 2020

5.3 METHODS

5.3.1 Study Area

The study was conducted in the northeast of Scotland in the Badenoch and Strathspey region of the Cairngorms National Park which, for the purposes of this study, delimits an area of c.23,000 ha of the main Spey valley from approximately Kinloch Laggan in the west to Advie in the east, including the surrounding hill sides of the Monadhliath Mountains up to the river Dulnain watersheds in the North, and the lower hills that fringe the strath together with the mouth of Glen Truim and Glen Tromie in the south to approximately Dalwhinnie (Figure 5.1). The River Spey meanders through this region, with the floor of the Spey valley being c.150m wide at its narrowest just upstream of Aviemore, where it is constricted by the fringing mountains, and c.1,500m at its largest between Kingussie and Loch Insh. The elevation of the valley has a gentle slope from c.250m to 200m, while the surrounding hill area consists of steeper land and higher altitudes up to 800m. The complexity of the Strathspey landscape was formed by the movement of glacier ice and subsequent meltwater (Young, 1978) and the principal rocks of the area consists of folded schists and gneisses of the Moinian Series and their associated intrusions

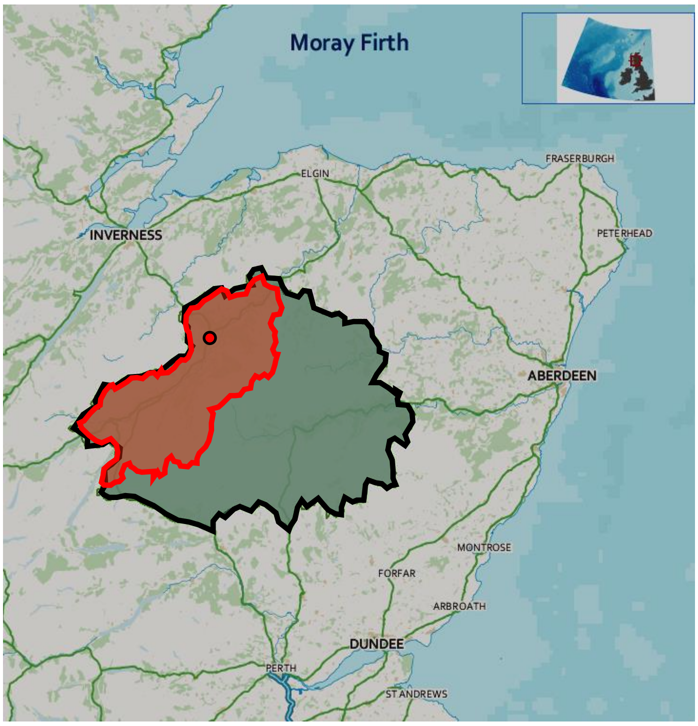


Figure 5.1. Map of the study area showing the Badenoch & Strathspey region in red, and the Cairngorms National Park in green with an insert of the United Kingdom in the top right corner. The town of Aviemore is shown as a red circle with a black outline where the weather data was obtained for the exploratory data phase of this study. Major roads are shown by dark green lines, and major rivers by a blue line. Figure adapted from the Scottish Government https://data.gov.uk/dataset/national-scenic-areas and accessed from https://commons.wikimedia.org/w/index.php?curid=70688184.

(Barrow, Hinxman and Craig, 1913; Hinxman and Anderson, 1915). The soils of the Strathspey are primarily Humus-iron podzols in the valley with a wide alluvial plain composed of silts, sand and waterborne pebbles, while the surrounding hill sides and mountains a combination of peat, peaty gleys, and peaty podzols (Soil Survey of Scotland, 1981). The median topsoil pH in the Strathspey ranges from 5.0-5.9 in the valley, and between 4.0-4.9 in the surrounding hillside with some patchy areas below pH 4.0 based on the 1:250,000 National soil map of Scotland. Land adjacent to the river Spey is periodically flooded resulting in areas of wetland and rich soil suitable for arable cropping creating a complex landscape of mixed low-intensity farming. The Strathspey sits in the temperate zone and has a climate which is typically cool with rainfall throughout the year, categorised by the Köppen-Geiger climate classification as oceanic (Cfb) bordering on a subpolar oceanic climate (Cfc; Rubel and Kottek, 2010).

5.3.2 Bird Data

Field scale data on four farmland breeding wader species (Lapwing, Curlew, Redshank and Snipe) were collected as part of the SWWI farmland breeding wader surveys conducted on a five-yearly basis between 2000 and 2015 covering a total of 53 farms. Wader species were counted on a field by field basis by RSPB volunteer surveyors following O'Brien and Smith (1992) which consisted of three survey visits within three hours of either dawn or dusk between 15th April and 21st June with at least one week between visits. Surveys were conducted on foot, walking to within 100m of all points in each field and scanning 200 – 400m ahead with binoculars to note the numbers and behaviour of all breeding waders. The maximum count recorded across the three visits was used as the response variable in data analysis, excluding birds in flocks of 10 or more and juvenile birds to reduce inflation of abundance.

Although a total of 53 farms were surveyed at least once between 2000 and 2015, the number of farms surveyed each year differed (Table 5.2). At the field scale, field names were not consistent between survey periods and there were also instances where changes in field boundaries occurred and/or field management differed between survey periods such as the conversion from grassland to woodland. To address differences in survey effort between years at the farm scale, a subset of 41 farms surveyed in each of the four survey periods were used in analysis at the farm level. To address inconsistencies at the field scale "consistent" fields (i.e. fields which had not been split into smaller fields, merged into larger fields, or converted into woodland) were identified for the analysis of the association between maximum wader count and habitat variables. These were selected by identifying field parcels which occupied the same space (i.e. location) in each of the four survey periods using the field central grid reference (latitude, longitude) to four decimal places and giving them a unique identifier which was consistent between survey periods. The second stage of identifying fields to use in analysis was selecting only the "consistent" fields that had full data on habitat variables (see section 5.3.3) in all survey periods. This resulted in a data set with a total of 20 farms and 123 fields.

Table 5.2. Summary table of the data structure showing the number of farms, fields and total area (ha) surveyed for farmland breeding waders in the Strathspey & Badenoch region by the SWWI in each of the four survey periods; **(A)** full dataset; **(B)** subset of data used in analysis at the farm level; **(C)** subset of data used in analysis of habitat associations with waders at the field level; **(D)** subset of data used in analysis investigating the association of distance to woodland with waders at the field level when controlling for other known variables of importance.

Survey Year	Number of Farms	Number of Fields	Total Area Surveyed (ha)
(A)			
2000	45	471	7,847.47
2005	43	381	7,893.74
2010	44	430	8,214.79
2015	50	506	9,351.67
(B)			
2000	41	571	6,779.39
2005	41	525	6,779.39
2010	41	582	6,779.39
2015	41	565	6,779.39
(C)			
2000	20	123	843.23
2005	20	123	843.23
2010	20	123	843.23
2015	20	123	843.23
(D)			
2015	33	341	4,034.32

5.3.3 Habitat Variables

Explanatory habitat variables known to influence wader distribution were recorded at the field level by surveyors as part of the SWWI farmland breeding wader surveys and included measures of field vegetation height (Milsom *et al.*, 2000; Vickery *et al.*, 2001; Shrubb, 2007), type and number of livestock grazing (Beintema and Muskens, 1987; Milsom *et al.*, 2000; Hart *et al.*, 2002), % of soft rush and % standing water/flooding within the field measured by eye (Milsom *et al.*, 2000; O'Brien, 2001; Rhymer *et al.*, 2010). As livestock are regularly moved between fields, I note that the measure of type and number of livestock grazing in a field is a weak index of long-term grazing pressure on a field, even within one season. However, no other quantitative data was available on grazing at the time. In addition, a category for field wetness was also recorded by surveyors but was only available for the survey years 2010 and 2015 so this variable was only included in the analysis on woodland edge effects (hypothesis 3) which only used data from 2015. However, soft rush and standing water/flooding are strongly associated with field wetness so these two variables are likely to capture the habitat characteristic of field wetness which has been shown to be linked to wader distribution (Ausden, Sutherland and James, 2001; Milsom *et al.*, 2002; Smart *et al.*, 2006; Eglington *et al.*, 2007, 2010) and should therefore be appropriate to answer hypothesis 2. Categories of land use were also recorded by surveyors and were standardised prior to analysis resulting in a total of six categories capturing arable land, and grassland habitats (Table 5.7). Fields were spot checked using google maps street view to confirm the land use category used was appropriate when the field was adjacent to the road and where street view images were available for the year surveyed. This was particularly required where surveyors used combinations of codes for the same field parcel, and to quantify the extent of fields lost through time to newly planted woodland, natural woodland regeneration (e.g. tree/scrub build up), new buildings or conversion to recreational use such as football pitches or picnic areas.

In addition to the explanatory habitat variables obtained from the SWWI surveys, topographical data on field characteristics likely to influence the suitability of a field for breeding waders were obtained for field area (ha), slope (°) and elevation (m) from the UK Ordnance Survey Digital Terrain model, and a measure of field enclosure score. Both slope and elevation were calculated as the mean of all points within a field (50-m grid). Enclosure scores were calculated by measuring the length of field boundaries consisting of trees, hedges, buildings or scrub using Google EARTH PRO and dividing this by the total length of the field perimeter. All geographical information system (GIS) manipulations were conducted with ARCGIS 10.2.2 (ESRI, 2011).

Woodland data was accessed from the Forestry Commission Open Data hub which holds annual National Forest Inventory (NFI) data from 2010 onwards as digital maps based on spring/summer aerial photography, satellite imagery and field surveys on all woodland with an area of 0.5 ha or more and with a minimum of 20% canopy cover (or the potential to achieve it) and a minimum width of 20 metres

(Forest Research, 2020). Prior to 2010, the 1995 – 1999 National Inventory of Woodland and Trees (NIWT) of Great Britain was conducted but at a coarser resolution producing digital maps on all woodland with an area of 2 ha or more with a 50% canopy cover, although some woodlands with an area of 0.1 – 2.0 ha were also included in the dataset based on ground surveys which were conducted on a sample basis. Due to differences in resolution between the earlier NIWT survey and the more recent NFI surveys, as well as the fact that there is a mis-match between the SWWI farmland breeding wader survey periods and the NIWT woodland surveys for the years 2000 and 2005, the analysis on the effect of woodland on waders was conducted for a single survey year only (2015). Woodland variables were therefore extracted using the NFI 2016 data. The distance of fields to woodlands was measured in ARCGIS (in metres) using the available GIS layer for the NFI 2016 Map for Great Britain from the field centroid point to the edge of its nearest woodland feature in terms of a straight line. The type of woodland was extracted from the woodland feature attributes table of *Interpreted Forest Types (IFT)* and simplified into two categories of woodland types: assumed conifer and assumed broadleaved.

Weather data was obtained from the Met Office Integrated Data Archive System (MIDAS) Open dataset held at CEDA (https://catalogue.ceda.ac.uk/uuid/dbd451271eb04662beade68da43546e1 accessed 12th November 2018). Hourly rainfall data (precipitation in mm) and air temperature (minimum and maximum in °C) was extracted from the Aviemore weather station for the years 1995 to 2015 with time series plots used in the exploratory data phase to identify if extreme weather events may have influenced the survey years of interest to this study. No obvious trend in the weather data was observable, although there were seasonal trends in the data with the month of June having higher air temperatures than the month of April, and a general trend for the month of June to have less total precipitation, although the total amount of monthly rainfall was variable between survey years (see Appendix item Figure A10 & Figure A11). Weather variables were therefore not included in analysis, particularly as wader counts were only conducted during "good" conditions avoiding cold, wet and windy days which can influence the detectability of birds (Connor and Hicks, 1980).

Table 5.3. Summary table of the variables used to explain wader counts collected at the field scale. **(A)** Field data collected by surveyors conducting the SWWI farmland breeding wader surveys; **(B)** field data extracted using Geographical Information System (GIS). Please note table is continued on following page.

Variable	Type	Method and Data Range
(A)		
Year	Categorical	Recorded year of survey visit (4 levels: 2000; 2005; 2010; 2015)
Vegetation height	Categorical	Estimated while walking the field and defined into categories based on the dominant type i.e. 75% of the field fits within that category (5 levels: Bare (no vegetation); Short (vegetation less than ankle height); Long (vegetation greater than ankle height); Medium (vegetation at ankle height); and Mixed (combination sward of mixed heights with no dominant type).
Rush cover (%)	Continuous	Percentage estimated by eye across each field (field range: 0 – 80%; field mean: 8.0%)
Field Wetness	Categorical	Only recorded in survey years 2010 and 2015 and based on the soil condition (5 levels: Dry (ground dry and hard which would be difficult to probe with a spade); Soft (ground soft which would be easy to probe); Wet (ground wet which squelches when walked upon); Standing water (water on surface of the ground); Combination (mixed soil wetness).)
Standing water (%)	Continuous	Percentage estimated by eye across each field (field range: 0 – 50%; field mean: 5.0%)
Livestock type	Categorical	Type of livestock present in field (5 levels: No livestock; Cattle; Sheep; Horse; Mixed grazing)
Grazing pressure	Categorical	Livestock units (LU) ha^{-1} calculated as LU = cattle*0.7 + calves*0.2 + sheep*0.08 + horses*1.0 divided by the field area with the grazing pressure defined as 4 categories based on recommendations by Chesterton (2006) (4 levels: None (0 LU); Light (<0.75 LU); Moderate (0.76–1.5 LU); Intensive (>1.5 LU)).
Land use	Categorical	Land use of field (6 levels: Arable (land used for cropping); Managed Grassland (typically enclosed land which is managed by grazing/cutting and may have had some chemical input); Marsh/Wetland (fields which have standing water for a large proportion of the year); Unmanaged Rank Grassland (land which has infrequent grazing/cutting and receives no chemical input); Rough grazing/Heath/Moorland (open moorland often extensively grazed); Rush pasture (land which has become dominated by rush cover probably due to previous heavy grazing, with the rushes usually managed by mowing or spraying and some level of grazing).

Table 5.3. Please note table continues from previous page.

Variable	Type	Method and Data Range
(B)		
Area (ha)	Continuous	Extracted from UK Ordnance Survey Digital Data layers and calculated in hectares (field range: 0.37 – 135.29ha; field mean: 6.75ha)
Slope (°)	Continuous	Extracted from UK Ordnance Survey Digital Data layers and calculated as the mean of all points within a field (50-m grid) in degrees (field range: 0.08 – 7.54°; field mean: 1.57°)
Elevation (m)	Continuous	Extracted from UK Ordnance Survey Digital Data layers calculated as the mean of all points within a field (50-m grid) in metres (field range: 178.86 – 318.84m; field mean: 219.26m)
Enclosure score	Continuous	Proportion of field boundary consisting of trees, hedges, scrub or buildings measured in Google EARTH PRO (field range: 0.00 – 0.91; field mean: 0.22).
Distance to woodland (m)	Continuous	Shortest linear distance to nearest woodland edge in metres (m) from the field centroid point – assessed using the NFI data layers for 2016 for fields recorded in year 2015 only. (field range: 0 – 768.45m; field mean: 232.95m)
Woodland type	Categorical	Woodland classification as recorded by the National Forest Inventory *Intepreted Forest Type (IFT)* and simplified into two levels (2 levels: Assumed conifer, Assumed broadleaved).

5.3.4 Statistical Analysis

5.3.4.1 The relationship between wader density and year at the farm level

To test whether the density of four wader species (Curlew, Lapwing, Redshank and Snipe), have changed between years in the Badenoch and Strathspey at the farm level (hypothesis 1), a generalised linear mixed model (GLMM) was used, with farm specified as a random effect. In this way, the model takes into account the fact that wader counts taken from the same farm over multiple years will not be independent. The response variable used in these models was the maximum number recorded at the site across the three farm visits. This was considered to be a better representation of the population of waders than the mean count or number of breeding pairs, because some breeding birds are likely to remain undetected during at least some of the visits reducing the overall abundance when the mean or breeding counts are used. Each of the four wader species were fitted as a separate model specifying a log link and Poisson error distribution. The predictor variable in these models was the categorical fixed effect of survey year (2000, 2005, 2010 and 2015). In addition, an offset was fitted of the natural log (Log_e) of farm area allowing counts to be converted into estimates of density ha^{-1}. Where models showed over dispersion, a negative binomial error structure was fitted instead, and if the models showed zero-inflation then a zero-inflated negative binomial model was fitted (ZINB).

5.3.4.2 Association between wader density and habitat variables at the field scale

GLMMs were used to measure the association between habitat variables and the density of the four wader species at the field scale. The following model was fitted for each wader species, specifying a log link and Poisson error distribution with an offset of the Log_e of field area, and field identity nested in farm as a random effect to account for spatial auto-correlation and to control for repeat measures from the same field. Where models showed over dispersion, a negative binomial error structure was fitted instead:

count = year + habitat variables + habitat variables^2 + year:habitat variables + field slope + field elevation + field enclosure score + grazing pressure*livestock type + (1/farm|field identity) + offset(natural log of field area)

Where *count* is the maximum number recorded of the species at the field scale, and *year* the categorical fixed effect of survey year (2000, 2005, 2010 and 2015). Habitat variables were a mix of both categorical and continuous predictors (see Table 5.3); and slope, elevation and enclosure score as continuous fixed effects. Polynomial terms for the habitat variables of *% rush cover* and *% water cover* were also included to account for potential non-linear effects. In all instances, the full model was fitted using habitat variables known to influence wader distribution on agricultural land (see above), following recommendations of Burnham and Anderson (2002) in defining a biologically informed model set as the starting point. In order to answer hypothesis 2, I used the habitat variables of *% rush cover*, *% standing water, grazing pressure* and *vegetation height* as proxy measures of AES management for waders to determine the effect on wader density, as formal data on AES agreements/options at the field scale were not available. The a priori hypotheses were that 1) wader density of all four species would be positively associated with % standing water and with less intensive grazing pressure; 2) Curlew, Redshank and Snipe would be strongly positively associated with % rush cover, while Lapwing would be weakly positively or negatively associated, reflecting the differences in ecology between the species; and 3) Lapwing would be positively associated with short vegetation, whereas Curlew, Redshank and Snipe would be associated with longer vegetation categories.

5.3.4.3 Investigating the effect of woodland on wader density at the field scale in survey year 2015

To investigate whether the density of the four wader species was associated with woodland variables (hypothesis 3), a GLMM was used specifying the maximum count of each wader species as the response variable at the field scale, a log link and Poisson error, random effect of field identity nested in site and fitting Log_e of field area as an offset. The main predictor variables of interest in this model were distance to woodland (continuous), woodland type (categorical), and the interaction term between these to test if the effect of woodland edge (i.e. distance) differed between conifer plantations and native broadleaved woodland. Distance to woodland edge was log_{10}-transformed for analysis as small changes in distance when close to woodland was expected to have a greater effect than at large distances from woodland. To ensure that the effect of transformation did not alter the relationship, analyses were also re-run with the untransformed woodland variable and a quadratic term was also explored. Topographical variables known to influence waders were also fitted in the model as explanatory predictor variables as well as habitat variables. Where models showed over dispersion, a negative binomial error structure was fitted instead.

To explore if there was a threshold effect of distance to woodland, variants on the final models in which distance to woodland edge was a continuous covariate were fitted using distance reduced to a binary variable of “below” and “above” threshold values set at 50 m increments ranging from 50 m to 750 m. The binary threshold models were compared to the continuous model using Akaike information criterion (AIC) to determine goodness of fit, where models with >2 AIC units lower were considered the significantly better model.

As only one year of data was required for this analysis, the use of “consistent” fields was not required, thus allowing a larger sample size. Only fields which did not have small patches of woodland within the field parcel and had full data on habitat variables were selected for analysis resulting in a dataset of 33 farms and 341 fields (Table 5.5). Because woodland edge contributed to the enclosure score of some fields (n = 134) a “de-coupling” of the enclosure score was undertaken for the woodland analysis only. To achieve this, where woodland edge contributed to field boundary, the length of this feature was removed in the calculation of enclosure score.

5.3.4.4 Model selection, validation and post-hoc tests

All statistical analysis was conducted in R version 3.6.2 (R Core Team, 2019). Data were initially explored and visualised graphically prior to formal analysis and model fitting following methods as recommended by Zuur, Ieno and Smith (2007). Prior to model fitting all covariates were standardized (scaled to z-values) and centred on a mean of zero to aid interpretation of regression coefficients (Schielzeth, 2010). To check for collinearity which could distort model estimates, pairwise correlations were conducted on pairs of numerical predictor variables, which showed that these were acceptable (<0.7; Figure A9 in Appendix; Dormann *et al.*, 2013). For categorical predictor variables, collinearity was explored using chi-square test of independence. The pairs identified as not independent (p<0.05) were plotted as mosaic plots to explore this relationship further and identify where the correlation may be occurring and then further explored using Cramer’s V which helps to determine the *strength* (i.e. effect size) of the association identified by the chi-square test (Table A0-10 in Appendix). Where highly correlated categorical predictors variables were identified as problematic in the model via Cramer V scores of >0.35 (e.g. *land use* and *vegetation height*), alternative species models were fitted, with analyses including only one of the predictor variables, and the alternate model with the other correlated predictor variable, and followed through to the minimal adequate model (MAM) as described below. Where this process resulted in two final models for the species one with each of the correlated variables, the model with the best fit was selected as the final model using Akaike information criterion (AIC).

GLMMs were fitted using the lme4 package (Bates *et al.*, 2015) with the exception of zero-inflated models which were constructed using the glmmTMB package (Brooks *et al.*, 2017). The minimal adequate models (MAM's) were obtained using Wald Chi Square test and likelihood ratio tests (LRT) of nested models (Everitt, 2002) to evaluate the contribution of each variable to the model at each stage of a backwards model construction. A nominal 5% significance level was used as the criterion to include or reject terms, until the MAM was found in which all variables were significant at the 5% level (Green, Osborne and Sears, 1994; Parish, Lakhani and Sparks, 1994; Milsom *et al.*, 2000; Murtaugh, 2009). Although stepwise modelling has been criticized in favour of approaches using information criteria and model averaging (e.g. Whittingham *et al.*, 2006), stepwise deletion for model selection has been shown to perform as well as other methods of producing models, particularly when the initial full model is based on existing knowledge of the species (Murtaugh, 2009) and was therefore deemed suitable for this study. Model validation was conducted by graphical visualisation of the model residuals to check that model assumptions of normality and homogeneity were met (Zuur, Ieno and Smith, 2007) using the DHARMa package (Hartig, 2020) and multicollinearity was checked in the fitted model by variance inflation factors where values <3 were considered acceptable. R^2 values were calculated to quantify the amount of variation in the data explained by the model using the piecewiseSEM package (Leftcheck, 2016), based on Nakagawa and Schielzeth (2013) of the fixed effects (marginal R^2) as well as the fixed effects while accounting for the random effects (i.e. full model; conditional R^2) which will be referred to from now on as R^2_M and R^2_C. The effect sizes of categorical variables were calculated using the emmeans package (Lenth, 2020; formally known as lsmeans; Lenth, 2016) when applicable. Where factors had three or more levels (i.e. survey year) post-hoc pairwise comparisons were made using a test of Tukey's Honest Significant Difference (HSD) in the emmeans package. MAM's were checked for spatial autocorrelation using x and y co-ordinates of the centre of each field in a Moran's I test, and temporal autocorrelation with a Durbin-Watson test using the DHARMa package and showed that neither were present in the residuals of the final models (Table 5.4).

5.4 RESULTS

5.4.1 The relationship between wader density and survey year at the farm level

Forty-one farms were surveyed between 2000 and 2015, covering an area of c.6,780 ha of farmland in the Badenoch and Strathspey. There were highly significant differences in density between survey years for all wader species (Lapwing: R^2_M = 11.2%, R^2_C = 59.1%, $LRT_{(3)}$ = 52.95, p = <0.0001; Redshank: R^2_M = 2.8%, R^2_C = 84.2%, $LRT_{(3)}$ = 24.36, p = <0.0001; Snipe: R^2_M = 3.0%, R^2_C = 56.6%, $LRT_{(3)}$ = 12.03, p = 0.0070), with the exception of Curlew (R^2_M = 1.4%, R^2_C = 70.6%, $LRT_{(3)}$ = 6.91, p = 0.0750). Densities were lower for all species in the year 2005 compared to 2000, although this was only significant for Lapwing and Redshank (Figure 5.2; Table 5.4, Table 5.5). After the survey year 2005, Redshank density did not differ significantly between the following consecutive five year surveys. Lapwing densities were significantly lower in 2010 than in 2005, but were not significantly different between 2010 and 2015. For both Lapwing and Redshank, density was significantly lower in 2015 than 2000. For Snipe, there was a significant increase in density between the years 2005 and 2015. Overall, the estimated marginal means of wader density showed a significant decline in wader density between 2000 and 2015 which was highly significant for Lapwing and Redshank (Table 5.10).

Table 5.4. Summary table showing the final models selected testing the relationship between wader density and year at the farm level for each of the four wader species.

Model	Unit replication	Link	Error Structure	Random Effect	Fixed Effects
Curlew	Site	Log	Negative binomial	Site	Year
Lapwing	Site	Log	Zero-inflated negative binomial	Site	Year
Redshank	Site	Log	Negative binomial	Site	Year
Snipe	Site	Log	Negative binomial	Site	Year

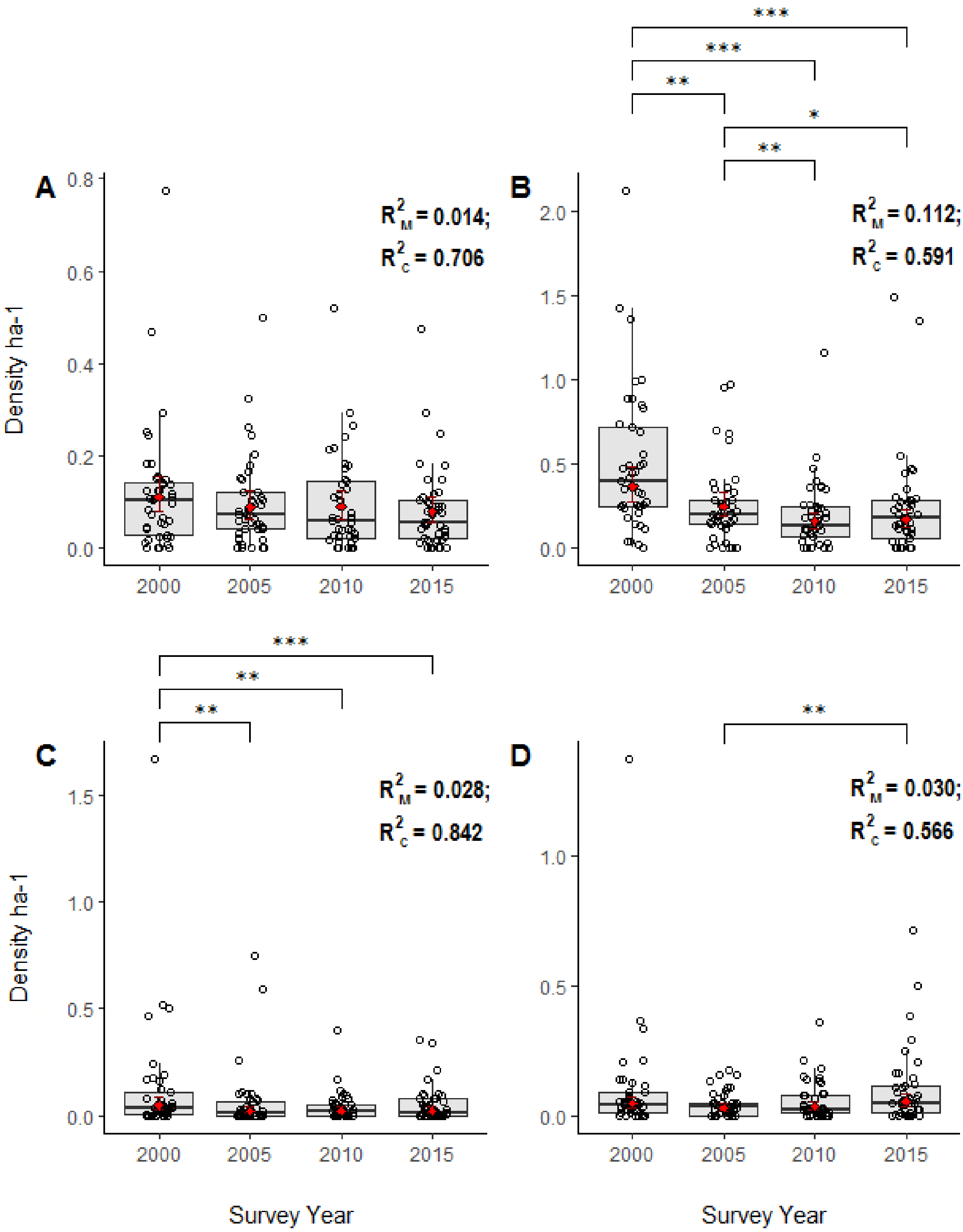


Figure 5.2. The relationship between wader density ha^{-1} and survey year at the farm scale for **(A)** Curlew; **(B)** Lapwing; **(C)** Redshank; and **(D)** Snipe. Sample size is 41 farms over four survey years (n= 164 observations). Raw data is represented by open black circles, while model predictions are shown as boxplots which represent the distribution in the estimates: median as a black solid line within the grey boxplot; the interquartile range (IQR) by the top and bottom of the boxplot; and two whiskers which represent values no further than 1.5*IQR from the boxplot. Please note that the y-axis of the four wader species are on different scales. A pairwise tukey method was used post hoc for comparing the model estimated marginal means (see Table 8) between levels of survey year with significance levels denoted by *'s where: * ≤ 0.05; ** ≤ 0.01; *** ≤ 0.001. The species GLMMs goodness of fit is represented by the R^2_M and R^2_C values of the marginal and conditional R^2 values respectively.

Table 5.5. Estimated marginal means (EMMs) for the GLMMs testing the relationship between wader density and survey year. Values computed by averaging together predictions on the linear-predictor scale and then back transforming to the response scale with covariates (i.e. site area) set to their mean value. Values were then divided by the mean site area to get the mean estimates of wader density ha^{-1}. A 95% confidence level was used. A pairwise tukey method was used for comparing EMMs between survey years with significant p values shown in bold.

	Marginal mean estimate ha^{-1}	Lower CI	Upper CI	Pairwise comparison	Test Statistic	p value
Curlew						
2000	0.11	0.08	0.15	2000 – 2005	z = 1.67	0.3388
2005	0.09	0.06	0.12	2000 – 2010	z = 1.59	0.3873
2010	0.09	0.06	0.12	2000 – 2015	z = 2.56	0.0515
2015	0.08	0.05	0.11	2005 – 2010	z = -0.09	0.9997
				2005 – 2015	z = 0.88	0.8135
				2010 – 2015	z = 0.99	0.7574
Lapwing						
2000	0.36	0.28	0.48	2000 – 2005	t = 4.04	**0.0005**
2005	0.25	0.18	0.33	2000 – 2010	t = 7.84	**<0.0001**
2010	0.16	0.11	0.21	2000 – 2015	t = 7.51	**<0.0001**
2015	0.17	0.12	0.22	2005 – 2010	t = 3.92	**0.0008**
				2005 – 2015	t = 3.53	**0.0031**
				2010 – 2015	t = -0.47	0.9657
Redshank						
2000	0.05	0.03	0.09	2000 – 2005	z = 4.04	**0.0003**
2005	0.02	0.01	0.04	2000 – 2010	z = 3.67	**0.0014**
2010	0.02	0.01	0.04	2000 – 2015	z = 3.85	**0.0007**
2015	0.02	0.01	0.04	2005 – 2010	z = -0.28	0.9927
				2005 – 2015	z = -0.13	0.9992
				2010 – 2015	z = 0.15	0.9989
Snipe						
2000	0.05	0.03	0.07	2000 – 2005	z = 2.25	0.1092
2005	0.03	0.02	0.05	2000 – 2010	z = 1.42	0.4892
2010	0.04	0.02	0.05	2000 – 2015	z = -0.89	0.8102
2015	0.06	0.04	0.08	2005 – 2010	z = -0.82	0.8435
				2005 – 2015	z = -3.13	**0.0093**
				2010 – 2015	z = -2.33	0.0908

5.4.2 Association between wader density and habitat variables at the field scale

Twenty farms (n = 123 fields), covering approximately 840ha of farmland in the Badenoch and Strathspey were included in this analysis over the four survey periods. There was strong evidence for a negative relationship of increasing enclosure score on wader density for all four species after controlling for other variables of importance; fields with a smaller enclosure score (i.e. more open fields) held higher wader densities than fields with a higher enclosure score (Table 5.7). The raw data also reflects this relationship of a decline in wader density with increasing enclosure score, albeit with higher variance at low scores, with more open fields ranging from zero to densities greater than 1.0 ha^{-1} while at higher enclosure scores only low or zero values of wader density are observed with no observations greater than 0.5 waders ha^{-1} (Figure 5.3).

Table 5.6. Summary table showing the final models selected for testing the association between wader density and habitat variables at the field scale for each of the four wader species.

Model	Unit replication	Link	Error Structure	Random Effect	Fixed Effects
Curlew	Field	Log	Negative binomial	Site	Year + Enclosure score + % Rush cover
Lapwing	Field	Log	Negative binomial	Site	Year + Enclosure score + Slope
Redshank	Field	Log	Negative binomial	Site	Year + Enclosure score + % Water cover + Elevation
Snipe	Field	Log	Negative binomial	Site	% Rush cover + % Water cover + Enclosure score

Redshank densities declined sharply with elevation, while Lapwing density was higher on flatter ground. The habitat variables of % rush cover and % water cover showed a positive relationship with some wader species (Figure 5.4). A high proportion of zero densities were recorded at low values of both % rush and % water cover in all four wader species, and a large amount of variation is observable in the raw data. However, both Curlew and Snipe density were predicted to increase with increasing rush cover albeit with larger confidence intervals at the higher % values possibly reflecting the fewer data points of fields almost completely covered by rush. There was also large variance in the data with % water cover, with the models showing a significant and positive association with increasing % water for Redshank and Snipe. Habitat models were also explored using quadratic terms for % rush and % water but these were non-significant.

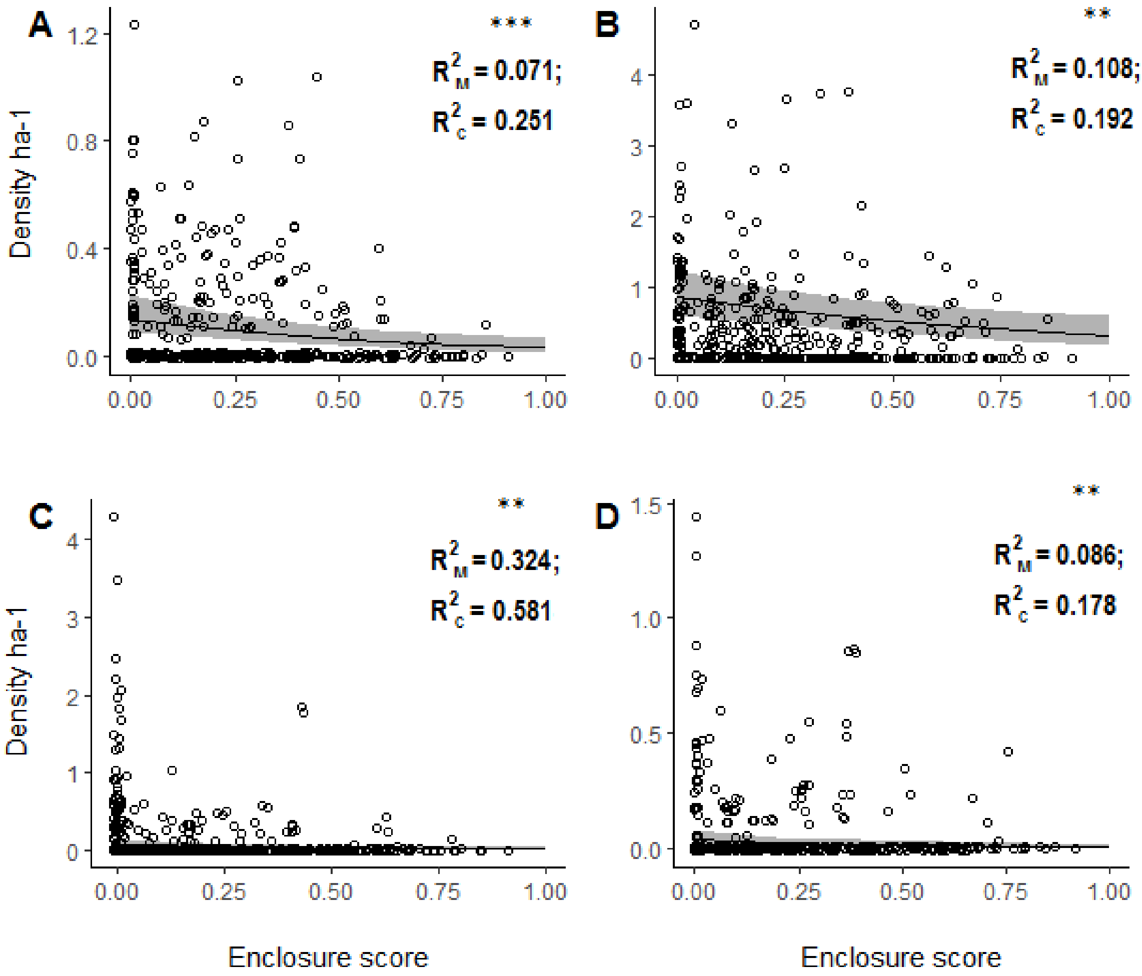


Figure 5.3. The relationship between wader density ha^{-1} and enclosure score at the field scale for **(A)** Curlew; **(B)** Lapwing; **(C)** Redshank; and **(D)** Snipe. Sample size is 20 farms (n = 123 fields) over the four survey years (2000, 2005, 2010, 2015) resulting in 492 observation. Raw data is represented by open black circles, while model predictions shown as a solid black line with 95% confidence intervals shaded in grey. Please note that the y-axis of the four wader species are on different scales. Where the model showed that the coefficient estimates of the predictor of enclosure score was significant this is denoted by *'s where: ns > 0.05; * ≤ 0.05; ** ≤ 0.01; *** ≤ 0.001. The species GLMMs goodness of fit is represented by the R^2_M and R^2_C values of the marginal and conditional R^2 values respectively.

There was no main effect of vegetation height, grazing pressure, livestock type nor land use category on wader density, and no significant interaction between survey year and any of the habitat variables. However, there were significant differences in wader density between survey years for Curlew, Lapwing and Redshank, but not for Snipe, after controlling for other variables of importance (Table 5.7; Figure 5.5). With the exception of Snipe, all wader species had significantly lower densities in 2015 than in 2000.

Table 5.7. Summary table of GLMMs investigating the relationship between wader density with survey year, habitat variables and topographical variables at the field scale using data from across 20 farms (n = 123 fields) over the four survey years (2000, 2005, 2010, 2015) resulting in 492 observation. Note that covariates were scaled and centred prior to analysis for ease of coefficient interpretation. Table shows results of the species minimum adequate models with significance levels denoted by *'s where: * ≤ 0.05; ** ≤ 0.01; *** ≤ 0.001.

Predictor	Type (df)	Wader species parameter estimates ± SE and test statistic			
		Curlew	Lapwing	Redshank	Snipe
Year	Factor (df = 3)	- (LRT = 9.36*)	- (LRT = 51.26***)	- (LRT = 8.67*)	-
% rush cover	Covariate (df =1)	0.24 ± 0.07 (z = 3.50***)	-	-	0.68 ± 0.09 (z = 7.40***)
% water cover	Covariate (df =1)	-	-	0.30 ± 0.10 (z = 2.88**)	0.20 ± 0.08 (z = 2.53*)
Enclosure score	Covariate (df =1)	-0.33 ± 0.09 (z= -3.67***)	-0.21 ± 0.07 (z = -3.04**)	-0.38 ± 0.15 (z = -2.62**)	-0.44 ± 0.13 (z = -3.28**)
Elevation	Covariate (df =1)	-	-	-1.17 ± 0.32 (z = -3.64***)	-
Slope	Covariate (df =1)	-	-0.34 ± 0.09 (z = -3.70**)	-	-

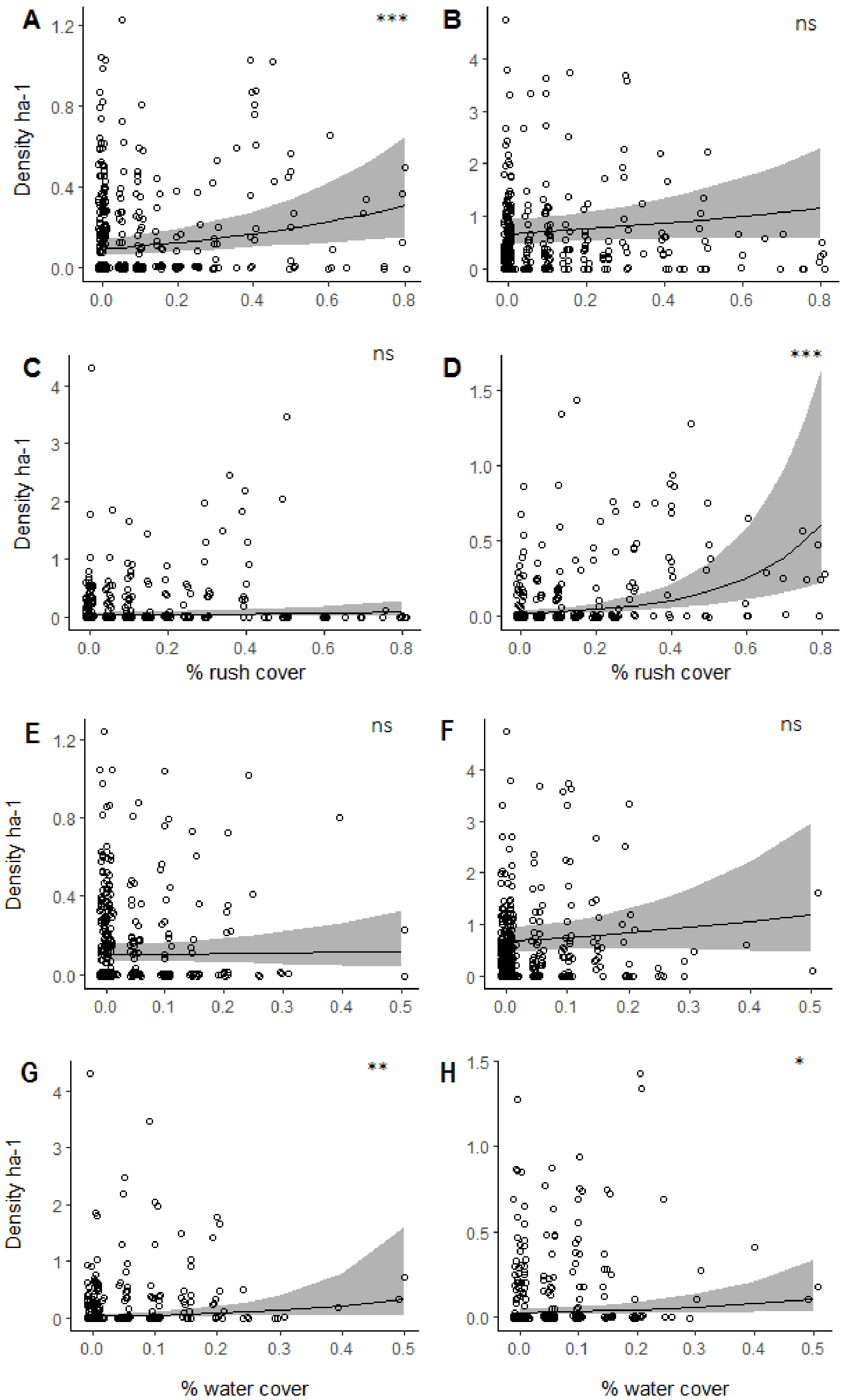


Figure 5.4. The relationship between wader density ha^{-1} with % rush cover at the field scale for **(A)** Curlew; **(B)** Lapwing; **(C)** Redshank; **(D)** Snipe; and with % water cover for **(E)** Curlew; **(F)** Lapwing; **(G)** Redshank; and **(H)** Snipe. Sample size is 20 farms (n = 123 fields) over the four survey years (2000, 2005, 2010, 2015) resulting in 492 observation. Raw data is represented by open black circles, while model predictions shown as a solid black line with 95% confidence intervals shaded in grey. Please note that the y-axis of the four wader species are on different scales. Where the model showed that the coefficient estimates of the predictor of enclosure score was significant this is denoted by *'s where: ns > 0.05; * ≤ 0.05; ** ≤ 0.01; *** ≤ 0.001.

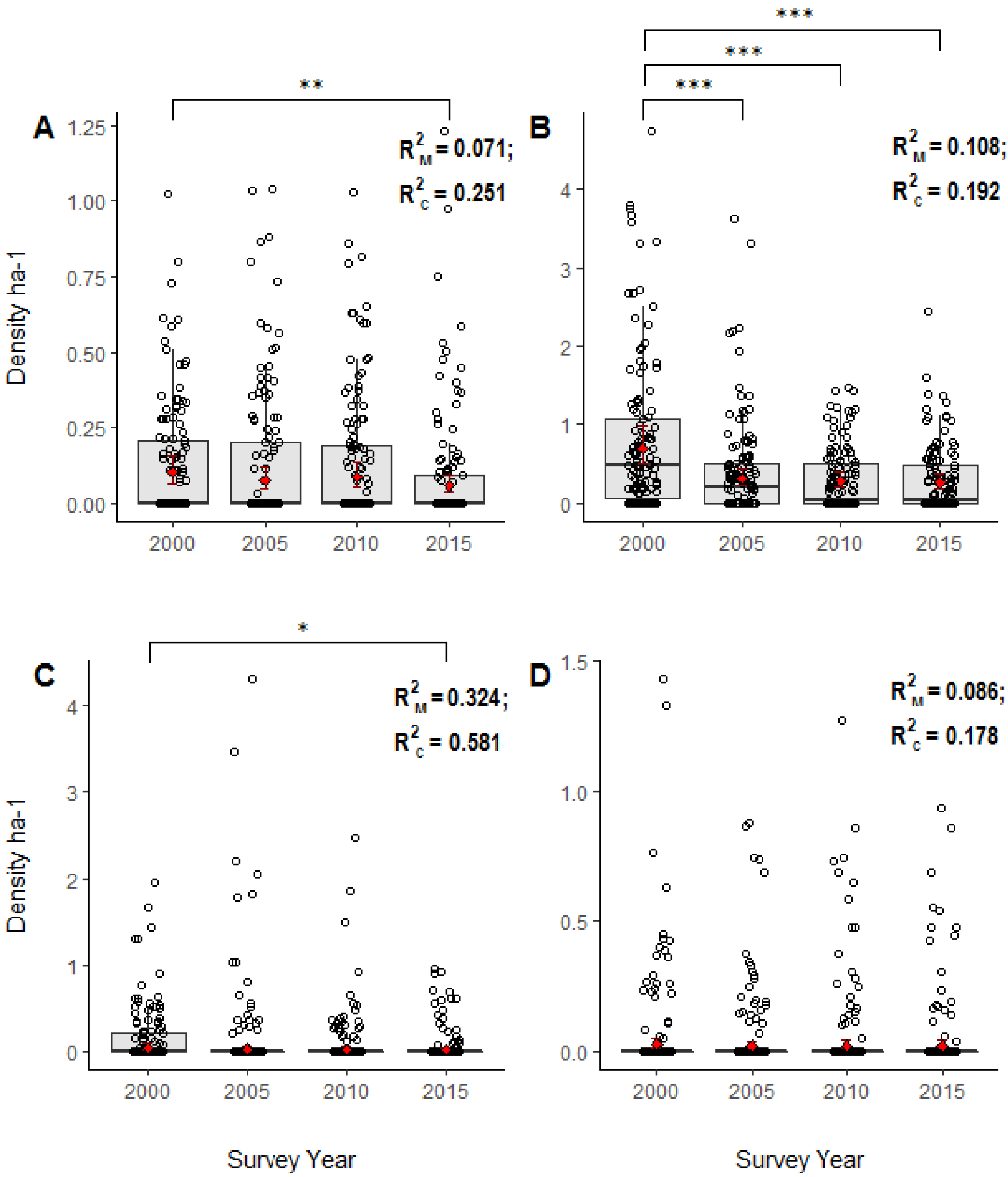


Figure 5.5. Estimated marginal means (EMMs) shown as filled black circles with 95% confidence intervals as whisker for the GLMMs at the field scale for **(A)** Curlew; **(B)** Lapwing; **(C)** Redshank and; **(D)** Snipe against survey year. Data used is across 20 farms (n = 123 fields) over the four survey years (2000, 2005, 2010, 2015) at the field scale resulting in 504 observation. Values computed by averaging together predictions on the linear-predictor scale and then back transforming to the response scale with covariates of importance in the model set to their mean value (e.g. elevation). Values were then divided by the mean site area to get the estimates of wader density ha^{-1}. A pairwise tukey method was used for comparing EMMs between levels of survey year with significance levels denoted by *'s where: * ≤ 0.05; ** ≤ 0.01; *** ≤ 0.001. Please note that the y-axis of the four wader species are on different scales. The species GLMMs goodness of fit is represented by the R^2_M and R^2_C values of the marginal and conditional R^2 values respectively. The species GLMMs goodness of fit is represented by the R^2_M and R^2_C values of the marginal and conditional R^2 values respectively.

5.4.3 Investigating the effect of woodland on wader density at the field scale

Thirty-three farms (n = 341 fields) were surveyed in 2015 covering an area of c.4,000 ha of farmland in the Badenoch and Strathspey holding, on average, wader densities of 0.06 Curlew ha^{-1}, 0.18 Lapwing ha^{-1}, 0.04 Redshank ha^{-1}, and 0.05 Snipe ha^{-1}. Wader density increased with distance from woodland for all four species (Figure 5.6). At distances further away from woodland, the estimated marginal means for distance to woodland and waders in an average field, showed that Curlew density was approximately six times greater at a distance of 700m (0.17 Curlew ha^{-1} with 95% CI 0.09 – 0.31ha^{-1}) than at a distance of 30m (0.03 Curlew ha^{-1} with 95% CI 0.02 – 0.07 ha^{-1}) and 17 times greater for Lapwing (30m distance: 0.03ha^{-1} with 95% CI 0.02 – 0.06 ha^{-1} vs 700m distance: 0.51ha^{-1} with 95% CI 95% CI 0.24 – 1.08 ha^{-1}) representing a reduction of 80% and 90% respectively when closer to woodland. Re-running the analysis using the untransformed variable for distance to woodland made little difference to the overall effect of the distance to woodland variable, but produced models with a higher AIC than the transformed variable. Variants on the final species models in which distance to woodland edge was reduced to a binary variable of "below" and "above" threshold values set at 50m increments resulted in poorer goodness-of-fit as determined by AIC for the best threshold model for Lapwing and Snipe of 200m and 300m respectively (Lapwing = 4.7 and Snipe = 3.0 AIC units greater than the continuous effect model), while the models for Curlew and Redshank were equally good or "better" than the continuous effect model with thresholds of 400m and 300m from woodland respectively (Curlew = 1.7 and Redshank = 2.2 AIC units lower than the continuous effect model). In all instances of the best threshold model, the binary threshold variable was a highly significant predictor in the model (Curlew: R^2_M = 11.2%; R^2_C = 59.1%, $LRT_{(3)}$ = 13.98, p = 0.0002; Lapwing: R^2_M = 11.2%; R^2_C = 59.1%, $LRT_{(3)}$ = 27.47, p = <0.0001; Redshank: R^2_M = 2.8%; R^2_C = 84.2%, $LRT_{(3)}$ = 21.04, p = <0.0001; Snipe: R^2_M = 3.0%, R^2_C = 56.6%; $LRT_{(3)}$ = 11.94, p = 0.0005) with a large and positive parameter estimate for the binary variable "above" compared to the "below" defined distance to woodland threshold which was highly significant (Table 5.9, Figure 5.7).

There was no main effect of woodland type nor of the interaction between distance to nearest woodland edge and woodland type on wader density in any of the models (i.e. the effect of distance to woodland edge on wader density was the same across both conifer plantations and native broadleaved woodland).

Similar to the previous field scale analysis across all four survey years without woodland variables described above, there was evidence for an association of several habitat and topographical variables with waders. Curlew and Lapwing counts were lower at higher enclosure scores even with the decoupling of woodland edge from this variable; Redshank and Snipe were more abundant with fields with a higher % rush cover (Table 5.9). Both Lapwing and Snipe had significant differences in wader density between vegetation height categories, with Lapwing more abundant on fields with short vegetation than long or medium vegetation heights; and Snipe more abundant on fields with long vegetation than fields with short, medium or mixed vegetation heights (Table 5.11). There were significant differences in Curlew density between livestock type categories, with fields grazed by mixed livestock holding higher numbers of Curlew than fields grazed by just sheep. There was no statistical evidence for an association between field wetness categories or field water % on waders, nor of grazing pressure. The topographical variable of elevation was negatively associated with Curlew with fewer Curlew predicted at higher elevations, but there was no evidence for an effect of slope nor of field land use on any of the four wader species.

Table 5.8. Summary table showing the final models selected for investigating the effect of woodland on wader density at the field scale in survey year 2015 for each of the four wader species.

Model	Unit replication	Link	Error Structure	Random Effect	Fixed Effects
Curlew	Field	Log	Negative binomial	Site	Enclosure score + Elevation + Livestock Type + Distance to woodland edge
Lapwing	Field	Log	Negative binomial	Site	Enclosure score + Vegetation height + Distance to woodland edge
Redshank	Field	Log	Negative binomial	Site	% Rush cover + Distance to woodland edge
Snipe	Field	Log	Negative binomial	Site	% Rush cover + Vegetation height + Distance to woodland edge

Table 5.9. Summary table of GLMMs investigating the relationship between wader density and habitat, topographical and woodland variables at the field scale using data from across 33 farms (n = 341 fields) from the survey year 2015 only (n = 354 observation). Note that covariates were scaled and centred prior to analysis for ease of coefficient interpretation. Table shows results of the minimum adequate models only with significance levels denoted by * where: * = 0.05; ** = 0.01; *** = 0.001.

Predictor	Type (df)	Wader species parameter estimates ± SE (test statistic and *s)			
		Curlew	Lapwing	Redshank	Snipe
Enclosure score	Covariate (df = 1)	-0.38 ± 0.14 (z = -2.79**)	-0.23 ± 0.11 (z = -1.98*)	-	-
Elevation	Covariate (df = 1)	-0.28 ± 0.13 (z = -2.22*)	-	-	-
Log_{10}(distance to woodland)	Covariate (df = 1)	0.39 ± 0.12 (z = 3.30***)	0.72 ± 0.13 (z = 5.40***)	0.72 ± 0.18 (z = 4.05***)	0.49 ± 0.13 (z = 3.79***)
Rush %	Covariate (df = 1)	-	-	0.39 ± 0.17 (z = 2.32*)	0.54 ± 0.12 (z = 4.61***)
Vegetation height	Factor (df = 4)	-	- (LRT = 16.57**)	-	- (LRT = 20.78**)
Livestock type	Factor (df = 4)	- (LRT = 12.44*)			

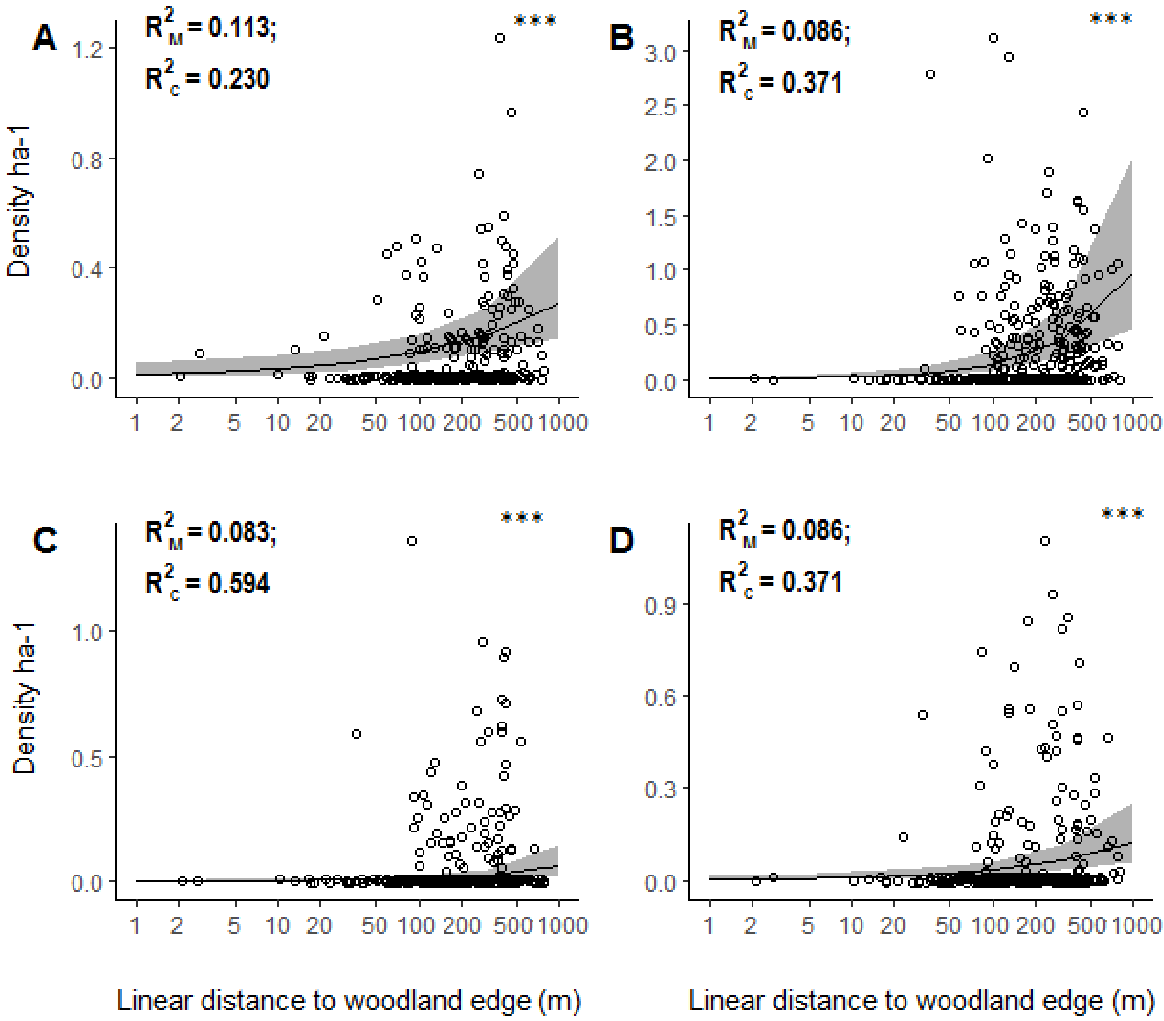


Figure 5.6. The relationship between wader density ha^{-1} and distance in metres from the field parcels centroid point to its nearest linear woodland edge for (A) Curlew; (B) Lapwing; (C) Redshank; and (D) Snipe. Sample size is 33 farms (n= 341 fields) over one survey year only (2015). Raw data is represented by open black circles, while model predictions of the conditional effects (using the mean of covariates and conditioned on reference levels for factors of vegetation height as "Mixed sward height" and for livestock type as "Mixed grazing") are shown as a solid black line with 95% confidence intervals shown in grey shading. Please note that the y-axis of the four wader species are on different scales and that the x axis is shown on the log scale. Where the model showed that the coefficient estimates of the predictor of linear distance to woodland edge was significant this is denoted by *'s where: ns > 0.05; * ≤ 0.05; ** ≤ 0.01; *** ≤ 0.001. The species GLMMs goodness of fit is represented by the R^2_M and R^2_C values of the marginal and conditional R^2 values respectively.

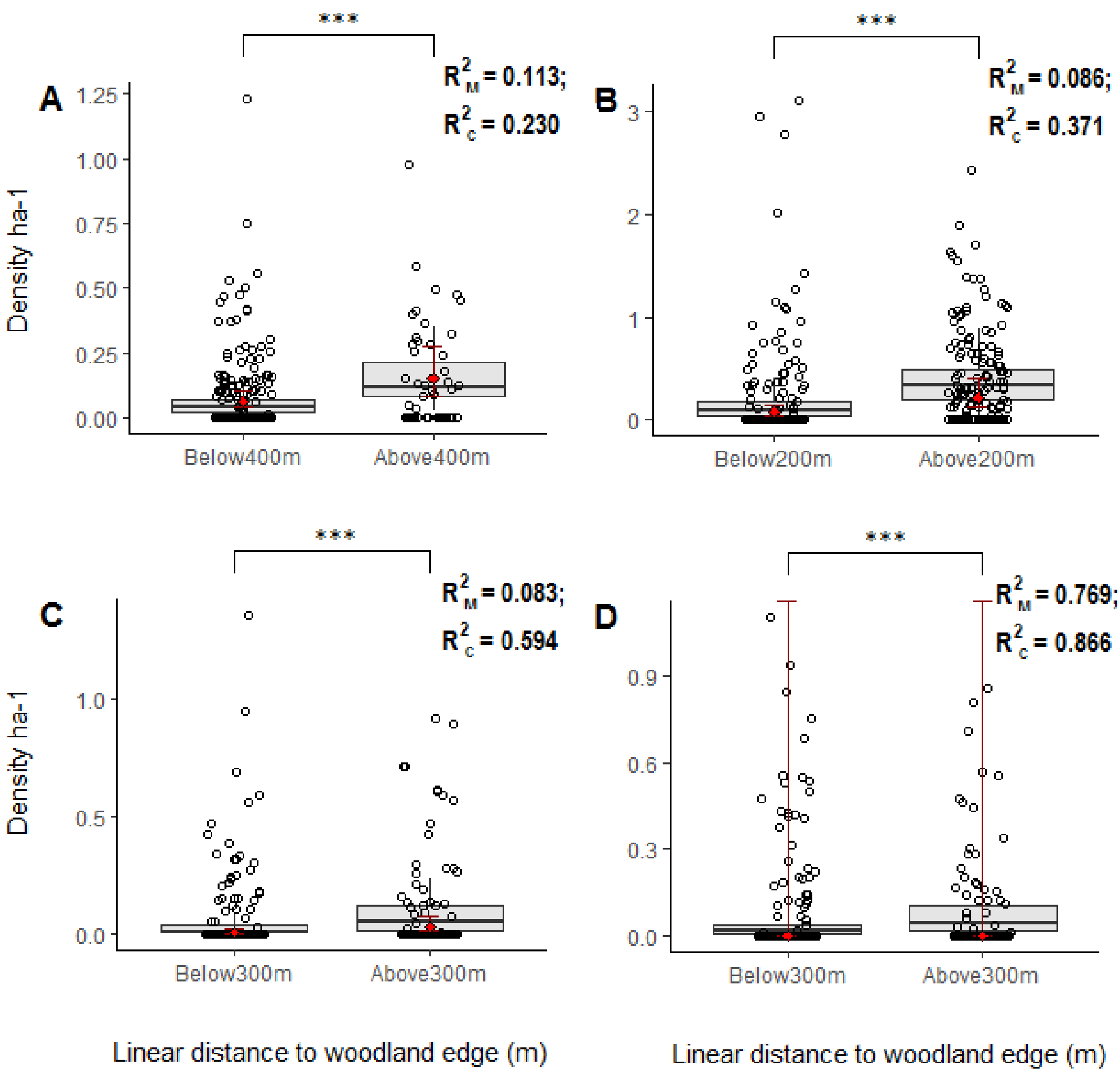


Figure 5.7. The relationship between wader density ha^{-1} and distance to woodland thresholds for **(A)** Curlew; **(B)** Lapwing; **(C)** Redshank; and **(D)** Snipe. Sample size is 33 farms (n= 341 fields) over one survey year only (2015). Raw data is represented by open black circles, while model predictions are shown as boxplots which represent the distribution in the estimates: median as a black solid line within the grey boxplot; the interquartile range (IQR) by the top and bottom of the boxplot; and two whiskers which represent values no further than 1.5*IQR from the boxplot. Please note that the y-axis of the four wader species are on different scales. A pairwise tukey method was used for comparing estimated marginal means (EMMs) shown as a filled red diamond with 95% confidence intervals shown as red whiskers between the two levels of below and above the distance to woodland threshold with significance levels denoted by *'s where: * ≤ 0.05; ** ≤ 0.01; *** ≤ 0.001. The species GLMMs goodness of fit is represented by the R^2_M and R^2_C values of the marginal and conditional R^2 values respectively.

Table 5.10. Summary table investigating wader density and distance thresholds to woodland GLMMs at the field scale using data from across 33 farms (n = 341 fields) from the survey year 2015 only (n = 354 observation). Note that covariates were scaled and centred prior to analysis for ease of coefficient interpretation. The covariates of the threshold factors are shown as the comparison between the levels showing the difference between the further away as compared to the closer distance to woodland. Table shows results of the minimum adequate models only with significance levels denoted by * where: * = 0.05; ** = 0.01; *** = 0.001.

Predictor	Type (df)	Wader species parameter estimates ± SE (test statistic and *s)			
		Curlew	Lapwing	Redshank	Snipe
Enclosure score	Covariate (df = 1)	-0.33 ± 0.14 (z = -2.38*)	-0.20 ± 0.12 (z = -1.71)	-	-
Elevation	Covariate (df = 1)	-0.24 ± 0.13 (z = -1.83)	-	-	-
>200m to woodland compared to <200m	Factor (df = 2)	-	0.75 ± 0.14 (z = 5.22***)	-	-
>300m to woodland compared to <300m	Factor (df = 2)	-	-	0.82 ± 0.18 (z = 4.61***)	0.56 ± 0.16 (z = 3.51***)
>400m to woodland compared to <400m	Factor (df = 2)	0.60 ± 0.16 (z = 3.76***)	-	-	-
% rush cover	Covariate (df = 1)	-	-	0.38 ± 0.17 (z = 2.20*)	0.53 ± 0.12 (z = 4.56***)
Vegetation height	Factor (df = 4)	-	- (LRT = 15.30**)	-	- (LRT = 19.03***)
Livestock type	Factor (df = 4)	(LRT = 12.60*)	-	-	-

Table 5.11. Estimated marginal means (EMMs) for the GLMMs testing the relationship between wader density and habitat, topographical and woodland variables at the field scale for the survey year 2015. Values computed by averaging together predictions on the linear-predictor scale and then back transforming to the response scale with covariates of importance in the model set to their mean value and averaged across the levels of factors. Values were then divided by the mean of field area to get the estimates of wader density ha^{-1}. A 95% confidence level was used. A pairwise tukey method was used for comparing EMMs between levels with significant p values shown in bold. Please note table continues on following page.

	Marginal mean estimate ha^{-1}	Lower CI	Upper CI	Pairwise comparison	Test Statistic	p value
Curlew:						
Livestock type						
None	0.05	0.04	0.08	None – Sheep	z = 2.32	0.1372
Sheep	0.03	0.02	005	None – Cattle	z = -0.38	0.9956
Cattle	0.06	0.03	0.11	None – Horse	z = -1.06	0.8291
Horse	0.12	0.03	0.54	None – Mixed	z = -2.25	0.1604
Mixed grazing	0.20	0.06	0.66	Sheep – Cattle	z = 2.00	0.2673
				Sheep – Horse	z = -1.86	0.3369
				Sheep – Mixed	z = -3.14	**0.0145**
				Cattle – Horse	z = -0.85	0.9162
				Cattle - Mixed	z = -1.87	0.3337
				Horse - Mixed	z = -0.55	0.9818
Lapwing:						
Vegetation height						
Bare	0.21	0.07	0.65	Bare – Short	z = 0.40	0.9944
Short	0.17	0.10	0.30	Bare – Medium	z = 1.76	0.3962
Medium	0.08	0.04	0.15	Bare – Mix	z = 1.31	0.6870
Mix	0.11	0.06	0.20	Bare – Long	z = 2.20	0.1815
Long	0.06	0.02	0.14	Short – Medium	z = 2.96	**0.0256**
				Short – Mix	z = 1.89	0.3202
				Short – Long	z = 2.92	**0.0288**
				Medium – Mix	z = -0.84	0.9167
				Medium – Long	z = 0.85	0.9151
				Mix – Long	z = 1.48	0.5740

Table 5.9. Please note table continued from previous page.

	Marginal mean estimate ha^{-1}	Lower CI	Upper CI	Pairwise comparison	Test Statistic	p value
Snipe:						
Vegetation height						
Bare	0.00	0.00	Inf	Bare – Short	t = -0.00	1.0000
Short	0.02	0.01	0.03	Bare – Medium	t = -0.00	1.0000
Medium	0.04	0.02	0.08	Bare – Mix	t = -0.00	1.0000
Mix	0.02	0.03	0.05	Bare – Long	t = -0.00	1.0000
Long	0.07	0.03	0.14	Short – Medium	t = -2.99	**0.0247**
				Short – Mix	t = -1.11	0.8010
				Short – Long	t = -3.74	**0.0020**
				Medium – Mix	t = 1.71	0.4318
				Medium – Long	t = -1.25	0.7197
				Mix – Long	t = -2.91	0.0313

5.5 DISCUSSION

Farmland breeding waders have suffered long-term and dramatic population declines within the UK, particularly in the lowlands where agricultural intensification has been more pronounced. The wader assemblages of upland farmland are valuable strongholds for supporting the UK's breeding population, with the Badenoch and Strathspey a key area for waders in Scotland, which is characterised by extensive mixed farming and wetland habitat. This area has also received high levels of conservation effort delivered through the Strathspey Wetlands and Wader Initiative (SWWI). The aim of this study was to test if wader densities had changed between the five-yearly SWWI farmland breeding wader surveys, if there were associations between waders and habitat variables at the field scale, and to test for an effect of woodland edge. Here I showed that wader density was significantly different between survey years at the farm scale for three of the wader species (Lapwing, Redshank and Snipe) but not for Curlew. Several habitat variables were associated with wader densities at the field scale; however, the crude measure of the variables meant that some variables, such as grazing intensity, which is well established within the literature as influencing waders showed no significant effect. Furthermore, I showed that there was a highly significant woodland edge effect which did not vary between woodland types (conifer vs broadleaved) on wader density on all four species, with wader numbers low for several hundreds of metres from the edge of woodland.

The results from this study provide important evidence on woodland edge effects for waders, which will have important conservation implications for these birds if land use policy on tree planting to reach climate change targets by the Government are to be met.

5.5.1 Changes in farmland breeding wader density between 2000 and 2015

In support of hypothesis 1, I found that there were significant differences in wader density between survey years for three of the wader species at the farm scale, but not for Curlew. For Lapwing and Redshank, densities were 50% lower in 2015 compared with 2000. Curlew densities were c.30% lower in 2015 than 2000 but these results were non-significant. For Snipe there was a small increase in wader density between 2000 and 2015. These results are largely consistent with results from monitoring studies of breeding wader pairs on farmland in other upland areas considered to hold important assemblage of waders in Scotland (Bell and Calladine, 2017), as well as in the wider landscape of the UK as reported by the latest BBS (Harris *et al.*, 2020). However, the declines reported in this chapter primarily occurred in the early stages of monitoring, with Redshank density stabilising after 2005, and Lapwing density stabilising after 2010. This could be due to the population now being at such low levels, that declines were inevitably smaller thus reducing the power to detect statistically significant differences between mean density ha^{-1}. Alternatively, it may be that conservation management interventions in the area have had positive effects ameliorating declines. The Badenoch and Strathspey is an important stronghold for waders and has received high levels of on-the-ground conservation targeting from advisors through the SWWI programme, which was initiated in 2008, as well as SRDP funding for wader sympathetic management through agri-environment schemes (AES). Conservation effort by the SWWI from 2008 onwards, predominantly of AES habitat management advice on rush management and scrape creation, may have influenced wader density in the following survey periods of 2010 and 2015. For example, Smart *et al.*, (2013) found that in England, targeted AES with the addition of advice performed better for Lapwing than targeted AES with no advice and non-targeted AES with no advice. This suggests that direct advisory support from conservationists may provide direct habitat benefits. Indeed, we also found that measures of rush cover and standing water had a positive effect on wader density at the individual field scale for three of the four species (see section 5.4.2). It would have been interesting to have explicitly tested for an effect of SWWI advice on wader density in the Badenoch and Strathspey as understanding the impact of conservation interventions on target populations and the extent to which those interventions mitigate or reverse population declines is an important question to answer in order to influence policy and management advice (Hoffmann *et al.*, 2010, 2015); however, data at the farm (and field) level was not available for this. An approach similar to Jellesmark *et al.,* (2021) of comparing breeding trends on nature reserves (e.g. areas of high targeted conservation management) with matched counterfactual trends from BBS would be a novel method of comparing how trends may have developed in the absence of conservation interventions; however, this was not within the scope of this study.

5.5.2 The association between wader density and habitat variables

There was some evidence to support hypothesis 2, with positive associations observed between several habitat measures commonly used in AES agreements on wader density, which varied by species. However, the explanatory variables used to capture favourable AES habitat for waders were crude, which will had reduced power to detect effects.

The a-priori hypothesis that all four wader species would be positively associated with an increasing proportion of standing water % at the field level was met for Redshank and Snipe only. Both Redshank and Snipe are common species of wet grasslands and marshy habitats, with the presence of both these birds having been shown in previous studies to be closely associated with wet habitats likely due to their feeding ecology of probing soil for earthworms and tipulid larvae (Green, 1988; Smart *et al.*, 2006; Hoodless, Ewald and Baines, 2007). However, there was no association between Lapwing and water despite several previous studies showing that Lapwing numbers are highly associated with wet features e.g.(Smart *et al.*, 2006; Rhymer *et al.*, 2010; Schmidt *et al.*, 2017), with standing water thought to be particularly important for lapwing chick foraging in lowland grassland habitat (Eglington *et al.*, 2010). Indeed, Lapwing have been shown to walk long distances with their chicks from suitable nesting habitat to find suitable chick foraging habitat and wet features in dry arable landscapes in Southern England (Shrubb, 2007). However, field wetness is likely not a limiting factor for Lapwing in the Badenoch and Strathspey as the land adjacent to the river Spey is periodically flooded resulting in large areas of wetland habitat. Conversely, land drainage has been widespread and prevalent in lowland England and is a major limiting factor for lapwing breeding there. The characteristic "wetness" of the Badenoch and Strathspey area coupled with targeted AES management for wetland habitats for waders probably makes this area very attractive to lapwing.

The a-priori hypothesis that all four wader species would be positively associated with increasing rush cover was met for Curlew and Snipe in GLMMs using data from across all four survey years, and was also identified in the GLMMs using survey data from one year only (2015) for Redshank and Snipe: densities were higher in fields with higher rush cover. There was no evidence for an effect of field rush cover on Lapwing abundance. The range of rush cover in this study was broad (0% to 80%); however the majority of fields had a very low cover of rush with an average field having only 8.0% rush cover. Additionally the type of rush (e.g. dense, scattered or regrowth) is likely to influence the suitability of fields for waders, as scattered rush even in large amounts would provide a range of open habitat and nest concealment, while the same amount of rush in dense growth is likely to be more detrimental. This level of detail was not sufficiently available at the field scale to investigate this, although I note that this would have been beneficial for exploring the effectiveness of AES rush management on waders. Rush management in an important component of UK upland AES prescriptions, as rush encroachment on upland grasslands has increased in recent decades (Silcock, Brunyee and Pring, 2012; Ashby *et al.*, 2020). This is partly due to high livestock densities and a switch from mixed grazing to sheep grazing changing selective grazing pressure on more palatable vegetation (Tweel and Bohlen, 2008) and trampling creating patches of bare ground that enables rush seeds to germinate and establish (Agnew, 1961).

Conversely a reduction of grazing or abandonment of wetter fields and soil compactions, can lead to rush dominated, tall and dense swards. Encroachment of rush results in unsuitable fields for waders. AES rush management for waders recommends the cutting of dense rush (Juncus) on rotation in the field, with the general aim to have < 30% rush cover to support a variety of breeding waders and open the sward (Natural England, 2012; RSPB, 2018; Scottish Government, 2018a). Wader abundance has been shown to increase following targeted rush management (Holton and Allcorn, 2006; Robson and Allcorn, 2006). However, there are differences between wader species' ecology. Lapwings are often observed in fields with short swards with patches of scattered tussocks to conceal their nests and chicks, whereas Snipe and Curlew prefer to be in taller, more rush dominant swards (Baines, 1988; Milsom *et al.*, 2000). For lapwing, a threshold level of rush has been observed between 15 and 40% above which a field generally becomes less suitable for breeding lapwing (O'Brien, 2001). Furthermore, recent research on daily wader nest predation rates using artificial nests were shown to be twice as high in fields which manage rush according to AES prescriptions than unmanaged (non-AES) fields (Kelly *et al.*, 2021). Predation rates are a key driver of wader declines, and it's possible that converting dense swards to open ones reduces nest concealment and thus increases predation risk. Further research into the management of rush on upland grasslands is therefore needed, to ensure that AES delivery is providing benefit to all aspects of wader ecology.

Vegetation height was also a significant predictor of wader species, but only for Lapwing and Snipe and only in the field-level analyses from 2015 looking at the effect of woodland edge on waders. The variable used in this study to investigate vegetation height was crude and is likely to have reduced power to detect an effect. Direct measurements of the sward providing a continuous measure, rather than a factor would have provided more useful information. However, on average, Lapwing were more abundant in fields with short vegetation than in long or medium vegetation heights, while Snipe were more abundant on fields with long vegetation height compared to fields with short, medium or mixed vegetation heights. Vegetation height is influenced by grazing pressure and livestock type. However, livestock type was only a significant predictor of Curlew abundance in the field-level analyses from 2015. Again, it should be noted that the variable used in this analysis to investigate grazing pressure was crude. Cattle or mixed grazing systems are considered optimal for breeding waders because they produce a varied sward height and structure and have higher abundance and availability of invertebrates (Vickery et al. 2001). Results from this chapter complement these findings as fields grazed by mixed livestock held higher numbers of Curlew than fields grazed by sheep alone. Most fields in this study had no grazing livestock during the bird breeding season (n = 64%) or were recorded as having sheep with lambs within the fields (n = 24%). Similarly, the grazing pressure recorded across the study area was either low or absent (n = 89%). Information on grazing management out with the survey dates were unavailable. Although sheep grazing is prevalent in the uplands (Fuller and Gough, 1999), careful stock management, timing and intensity of grazing can limit negative impact on vegetation and may provide suitable habitat for waders, reducing any particularly strong associations in analyses with this variable. The negative relationship between Curlew and sheep grazing could also be a proxy for other habitat characteristics not clearly captured in our habitat variables, such as a relationship between sheep grazing and smaller and more enclosed fields closer to farm buildings, as Curlew are often associated with more open moorland grazed habitats. The limited evidence for associations of grazing and vegetation height on waders in this study, may be because other land uses and factors, such as predation, are exerting stronger pressures which were not accounted for in this research.

Overall, the results from the field scale analysis between wader species and habitat variables showed a positive association between wader abundance and some habitat variables used in AES management options for the benefit of farmland breeding waders, such as rush and water cover; with fields with higher proportions of rush associated with higher abundances of Curlew, Redshank and Snipe; and fields with higher proportions of water cover associated with higher abundances of Redshank and Snipe. Further research would be useful to test at the field scale whether these variables tend to reduce any negative effect of year on wader density, which would require an interaction term in the models used in this study with year as a covariate, rather than as a factor. Indeed, previous studies on AES and farmland breeding waders have found differences in breeding wader abundance on sites that had come under AES management than on sites which remained outside AES showing a modest population increase for Redshank, and a reduction in the magnitude of decline for Lapwing in Scotland (O'Brien and Wilson, 2011).

5.5.3 The association between wader density and woodland edge

In support of hypothesis 3, this study identified strong evidence for an association between woodland distance and the abundance of all four wader species in the Badenoch & Strathspey, with fields further from woodland holding higher abundances of waders. After controlling for other variables, distance to woodland edge had a strong effect size on wader density, with the strongest effects in Lapwing and Redshank. Previous studies investigating the spatial relationship between waders and woodland have focussed predominantly on non-native conifer plantations and unenclosed moorland or peatland habitat. For example, Curlew nesting success and population change have been shown to be inversely associated with the exposure of sites to woodland (as determined by the percentage cover of a 1km radius around each site that comprised of woodland), providing quantitative evidence of indirect effects of tree planting on unenclosed moorland, with the suggested mechanism being edge effects on adjacent open ground (Douglas *et al.*, 2014). Similarly, national-scale analysis of the potential drivers of Curlew population change in Britain between 1995–99 and 2007–11 from Breeding Bird Survey data showed a negative association with woodland cover and Curlew abundance and population declines (Franks *et al.*, 2017). In lowland wet grasslands, lower abundances of Lapwing nests were observed than expected by chance within 500m of small woodland patches in the UK (Bertholdt *et al.*, 2017) as well as evidence for waders, particularly Dunlin of wet grasslands in Estonia to also nest further away from woodland than expected (Kaasiku, Rannap and Kaart, 2019). Similarly, distance to mature conifer plantations on peatland waders found that the breeding distribution of the two wader species, European Golden Plover and Dunlin, were associated with distance to woodland, with an adverse effect of proximity especially within 700m of woodland edge (Wilson *et al.*, 2014). In this chapter, the use of a binary threshold value was used to determine the distance over which woodland edge effects were exerted, suggesting 200m for Lapwing, 300m for Redshank and Snipe, and 400m for Curlew. These distances are somewhat shorter than those reported by Wilson *et al*., (2014) of 700m; however, this could be related to the type of woodland which was predominantly conifer in Wilson *et al.,* (2014) compared with predominantly broadleaved in this chapter, and due to the landscape which was open peatland in Wilson *et al.,* and mixed farmland here. Additionally, European Golden Plover and Dunlin may be more sensitive to woodland edge effects than the wader species explored in this chapter. Furthermore, the results in this chapter included a small subset of "young" forests (c.6%) as identified by the Forestry Commission through satellite imagery, and it is possible that if only mature woodland was used in analysis a stronger edge effect may have been detected as there can be a lag effect of woodland edge as the trees mature and the canopy becomes more dense. For Lapwing, a shorter distance of 200m from woodland edge was detected in this study than the 500m reported by Bertholdt *et al.,* (2017). This could have arisen due to the differences in the scale of woodland sizes used between the two studies, and there may be different pressures being exerted in lowland wet grasslands in England, compared with our site in the highlands of Scotland. Additionally, in this study the field centroid point was used to measure distance to woodland edge rather than nest location and it is therefore plausible that birds in the field could have been further away than 200m highlighting the fact that the threshold values identified in this chapter are only rough approximates of the distance to woodland edge effect. On average, fields had an area of 6.9 ha, but this ranged from 0.4 ha to 135.3 ha.

The effects of woodland on waders have been suggested to arise from either two mechanisms; the direct loss of habitat from replacement of open ground with woodland (e.g. Ratcliffe, 2007), and edge effects, predominantly caused by increased predator pressure and behavioural avoidance of nesting near enclosed areas. Both mammalian predators and Carrion crow are linked to woodlands, as they breed in woodland and forage over the surrounding open ground (Stroud *et al.*, 1987; Ratcliffe, 2007). Fragmented farmed landscapes with woodland hold higher abundance of foxes and crows than similar un-fragmented open farmland without woodland (Valkama and Currie, 1999). Nest predation rates of Curlew are higher nearer woodland edge, and other breeding waders have also shown negative relationships between nest predation rates and measures of productivity or population change woodland edge effects (Finney, Pearce-Higgins and Yalden, 2005; Amar *et al.*, 2011). These results are also consistent with a wider body of evidence, linking edge effects in fragmented landscapes to nest predation across a range of avian taxa, not just waders (Kurki *et al.*, 2000; Batáry and Báldi, 2004; Zurita *et al.*, 2012; Douglas *et al.*, 2014; Hancock, Klein and Cowie, 2020). Woodland may also harbour deer whose grazing in adjacent habitats may impact vegetation structure and composition, as well as increase tick densities close to plantations which can affect wader chicks (Newborn *et al.*, 2009; Gilbert, 2013).

In this chapter, it was not possible to test the underlying mechanisms associated with the relationship between distance to woodland edge and waders. However, it is likely that the impacts observed of woodland on waders probably arose via edge effects on adjacent open ground. It is also possible that direct loss of habitat has occurred recently in the Badenoch & Strathspey due to tree planting as suggested by "young trees" in the 2015 NFI data. Additionally, of the 41 farms surveyed between 2000 and 2015, ten farms were recorded to have woodland on at least 1 field parcel (c.240ha in total) in 2015, equivalent to a loss of 3.5% of the total area surveyed, further suggesting that the Badenoch and Strathspey may be being impacted by tree planting.

There was no evidence of an interaction effect between distance to woodland and woodland type, suggesting that the effect of distance was consistent on waders, regardless of whether it was predominantly coniferous plantation, or predominantly broadleaved native woodland. The National Forestry Inventory classifies woodland into several groups based on the assumed forest community from satellite images, and for this study we reduced the number of types into two categories only. It is possible that simplification of the categories into predominantly conifer vs predominantly broadleaved masked some of the variation between possible woodland types in this study such as differences in the diversity of the woodland in terms of stand height, or understory vegetation which could potential influence its ability to harbour predators. Overall though, the fact that an effect of woodland type was not detected in this study, and the strong effect of distance to woodland edge, is a key finding, particularly given the current political support for woodland expansion in order to help mitigate the impacts of climate change (Read *et al.*, 2009; Quine *et al.*, 2011; Scottish Forestry, 2020b) combined with the increasing concern for declining wader populations (e.g. Newton 2004). In Scotland, government policy aspires to increase woodland cover to 21% of its total land mass by 2032 (Scottish Forestry, 2019). Limited space, and the uncertainty of support for farmers post-Brexit make this a timely discussion of the location of woodland planting and the effects on breeding waders should be taken into consideration.

5.5.4 Conclusions, conservation implications and further research

This chapter provides landscape scale correlational evidence for detrimental effects of woodland proximity on important breeding bird species of adjacent open farmland in the Badenoch and Strathspey. Given the on-going national declines of farmland breeding waders, these findings are of concern and suggest that buffer zones of several hundred of metres (e.g. minimum 200m) should be considered, in addition to the effect of direct habitat loss, in the locations of woodland planting away from existing breeding bird strongholds. This may also have an effect on the delivery of AES for waders which recommends a distance of 30m. The results from this study therefore suggest that the detrimental effects of distance to woodland is currently underestimated in the delivery of conservation for these species. Further evidence on the impacts of woodland expansion, and of the quantitative effect of advisory support from SWWI staff, would assist in the development of a clear programme of action in the Badenoch and Strathspey to support farmland breeding waders. For example, predation pressure and low productive have been shown to drive wader declines on upland sites e.g. (Smart *et al.*, 2013). Wader nest predation rates would therefore provide additional evidence of woodland edge effects. Although this study was restricted to a relatively small study area, the implications from this chapter will be applicable throughout Britain, in particular in strong holds of farmland breeding waders where woodland planting and wader conservation may be in conflict. Determining if targeted conservation advice tailored for individual farms from groups such as the SWWI would also be useful to ensure that conservation effort and cost is being delivered in maximal ways to ensure the largest benefit for the target species.

CHAPTER 6 | GENERAL DISCUSSION

6.1 SUMMARY OF RESULTS

Farmland breeding waders have suffered long-term and dramatic population declines within Europe as a result of agricultural change (Wilson, Evans and Grice, 2009). The wader communities of Scotland on marginal upland grasslands are critically important strongholds for the UK's breeding populations of waders, and were previously thought to be a refuge for these species which had seen large declines in the lowlands of the UK (e.g. Sim *et al.*, 2005; Forrester *et al.*, 2007; Smart *et al.*, 2008). However, declines have also been recorded in Scotland more recently with the Breeding Bird Survey (BBS) reporting a decline of c.60% and c.50% for Curlew and Lapwing respectively between 1995 and 2017 (Harris *et al.*, 2020). This is particularly worrying as the UK supports 10% of the world's breeding population of Curlew (Brown *et al.*, 2015), and c.27% of the European breeding population (chapter 1). Waders have been extensively studied, with their species requirements well known and they have received highly targeted conservation intervention in the form of AES. These have predominantly focussed on manipulation of the sward and water table. However, these have not been sufficient to reverse population declines at the national or regional level, and it is possible that a lack of understanding of how soil condition and biota influence wader ecology is currently being missed from the suite of prescriptions. The research presented in this thesis has sought to evaluate the potential of lime use to amend acidic soil pH for the benefit of waders, by increasing earthworm populations. The research here also highlights the potential threat of tree planting in upland areas on farmland breeding waders, which is likely to pose a growing risk to waders as tree planting becomes an increasingly important government land-use policy to mitigate climate change.

In chapter 2 I demonstrated that the use of agricultural dust lime ($CaCO_3$) when applied as a surface dressing could amend acidic soil pH, although the effect of lime varied with time. I also showed that total earthworm abundance was positively associated with soil pH, and that earthworm numbers increased following the use of lime which varied with time. There was an average pH difference between lime treatment and control plots of 0.22 pH units in 2015; 0.52 pH units in 2016 and 0.44 pH units in 2017; and an average difference in the number of earthworms per soil core between lime treatment and control plots of 0.81 earthworms in 2015; 3.25 earthworms in 2016; and 1.05 earthworms in 2017. The potential contamination of lime between lime treatment and control (non-limed) may have contributed to a reduction in the overall effect sizes of the BACI study, which is likely to have underestimated the differences in pH and thus earthworms following lime use. The use of laboratory or mesocosm studies would be interesting further research which would allow for more control of the manipulation of soil conditions, and to test the mechanism behind the increase in earthworm numbers (e.g. increased cocoon production) and could help with idenitfying species level effects. The high proportion of immature earthworms recorded in chapter 2 is common in earthworm field studies (Lee, 1985) despite the fact that the proportion of adult earthworms in a population peaks in spring and autumn (Lee, 1985; Schmidt, 2001; Bartlett, Harris, *et al.*, 2008; Bartlett, James, *et al.*, 2008).

In chapter 3 I showed that vegetation variables were not influenced by lime treatment, and that there was no effect of lime treatment on beetles; however there was an observed non-significant increase in spiders following lime use. The lack of baseline data on beetles and spiders, is unlikely to have affected the results of this study, although the additional data would have made for a nice comparison. Further analysis of the data by family, functional group or considering the body size/biomass of beetles and spiders would have been useful information in this study on how lime use could affect wader food resources. The results from this chapter are important as they show no negative effect, either on beetle and spider abundance, or on habitat variables (height and heterogeneity) which could influence the accessibility of prey items to foraging birds.

In chapter 4 I demonstrated that wader density was 50% higher on limed treatment compared with control, and that intake rate of waders was greater on lime treatment than on control plots in 2017. Monitoring of the fields over a longer time period could have helped identify how long the positive effects of lime use on waders persists as it is likely that over time acidification of the field would occur again requiring lime to be reapplied. The positive effects of lime use on lapwings, in terms of abundance, has persisted for up to seven years compared to control fields at a farm site in Stirlingshire (McCallum *et al.*, 2016). Therefore applications of lime between 5 – 10 years is likely to be beneficial to waders. Further research which includes faecal studies of foraging waders would provide useful information on what the birds are feeding on, and help support the hypothesis that improved foraging conditions are arising from increased earthworms. Overall, the results from chapter 2, 3 and 4 provide evidence to support the hypothesis that lime use on marginal upland grasslands is likely to benefit farmland breeding waders by amending soil pH, and thus increasing earthworms, an important prey item for these birds.

In chapter 5 I showed that several habitat variables showed a positive association between wader densities which varied by species, including rush and water cover; with fields with higher proportions of rush associated with higher abundances of Curlew, Redshank and Snipe; and fields with higher proportions of water cover associated with higher abundances of Redshank and Snipe. Analysis of the habitat variables testing for an interaction effect with year as a covariate could help to identify if any of these variables are also associated with a reduced rate of decline over time. Understanding the effect of habitat variables which are used in AES management options for the benefit of farmland breeding waders are important to ensure that these are delivering on the ground conservation benefits, such as improved breeding success and reduced rate or halting or decline. Coupled with direct information on whether fields are actively being managed as part of an AES agreement would be useful further research. In this study I identified a strong woodland edge effect on waders. Woodland data going back through the other four years of surveys would have enabled analysis on trends to test if fields closer to woodland had a faster rate of decline. Management recommendations for tree planting should include buffer zones of several hundreds of metres from wader breeding habitat to mitigate negative effects on waders.

Finally, in this thesis I have discussed the use of lime as a conservation tool for the benefit of farmland breeding waders, further research required would be on how widespread and common lime use is in marginal uplands, and identification of the potential barriers to lime use. This research could be achieved by a widespread survey questionnaire. The Scottish Government's Economic Report on Scottish Agriculture 2020, shows that the amount of lime purchased in Scotland has decreased between 2003 and 2019 (Figure 6.1A), and that lime prices have increased over the same time period (Figure 6.1) suggesting that cost could be a barrier. If cost is a significant barrier to the use of lime in these areas than financial incentives would be required to encourage uptake of lime and would need to be explored. For example, AES agreements typically provide annual payment over five years on a field by field basis in return for a prescription of management over that five year period. However, lime use is usually a single purchase which is applied for one or more years based on the required amounts. A system of determining payments of lime use would have to capture this one off purchase, which would vary considerably between fields based on the starting and target pH. As agricultural lime use is viewed as an agricultural improvement, the AES prescription would also have to include the restriction of further field improvements (e.g. reseeding, grazing management). Furthermore, the targeting of this management would have to be very carefully applied as lime use is only likely to be of benefit to waders where pH has fallen below a threshold in which earthworm numbers have been significantly reduced. Widespread lime use in marginal areas or of moorland habitat is not recommended as a mosaic of habitat within the landscape is important to provide the best outcome for biodiversity.

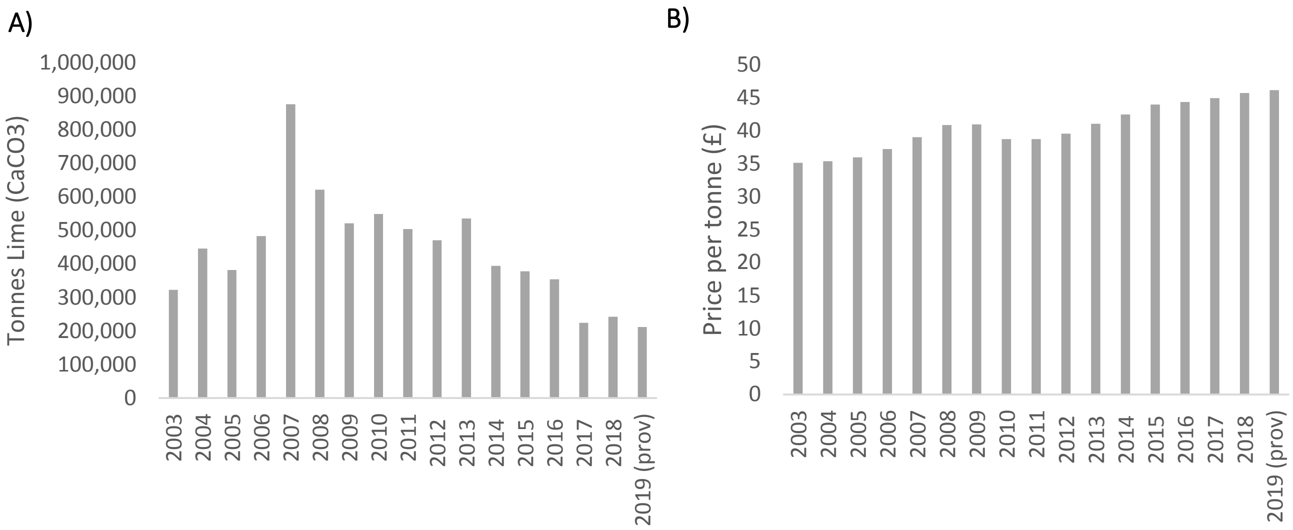


Figure 6.1. Bar charts displaying data extracted from The Scottish Government's Economic Report on Scottish Agriculture 2020 for 2003 to 2019 accessed from https://www.gov.scot/collections/economic-report-on-scottish-agriculture/ **(A)** Tonnes of lime ($CaCO_3$) purchased for agricultural use in Scotland; and **(B)** price of lime per tonne (£) in Scotland.

6.2 MANAGEMENT/POLICY RECOMMENDATIONS

The results highlighted in this thesis suggest that liming of previously improved grassland fields could benefit farmland breeding waders on marginal grassland fields by increasing food resources. Although there was a large amount of variation within the data, earthworm abundance was greater on lime treatment than control plots in year 2016. This suggests that there is a lag period in the ability to detect an effect of lime use on earthworms in the year 2015. Wader intake rate was also higher in year 2016 (although this did not vary significantly by treatment), and was significantly higher on lime treatment than control in year 2017. Wader density was significantly higher on lime treatment plots than control in all years surveyed. Additionally, earthworm abundance was positively associated with soil pH, and soil which had received lime was less acidic (higher soil pH) than control (non-limed) soil. Raw data of earthworm abundance against pH at the soil core scale peaked at pH 5.2, therefore fields that are currently below pH 5.0 but are in other ways suitable for breeding waders (e.g. suitable wet areas or features, unenclosed boundaries, and suitable vegetation/grazing management) could have biodiversity benefits from lime use.

Wader density is lower when closer to woodland regardless of whether it is conifer or broadleaved, and therefore a minimum threshold distance of 200m is recommended for planting in areas important to farmland breeding waders to mitigate woodland edge effect.

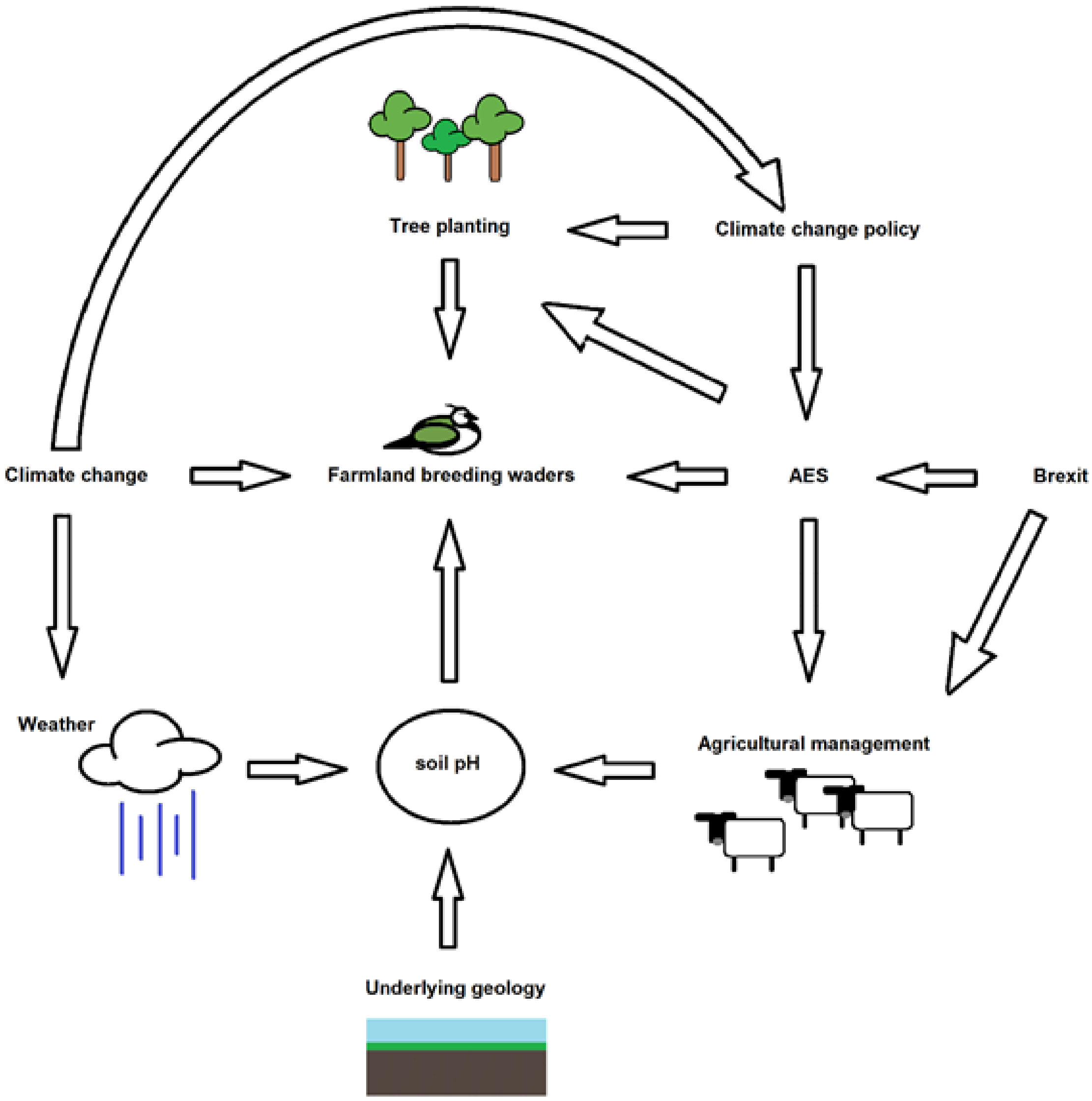


Figure 6.2. Schematic diagram displaying some of the influences affecting soil pH and farmland breeding waders of marginal upland agricultural grasslands in the UK.

6.3 FUTURE CHANGES IN FARMING IN THE UK THAT MIGHT IMPACT ON WADERS

6.3.1 Withdrawal of the United Kingdom from the European Union

On the 31st January 2020 the United Kingdom (UK) withdrew from the European Union (EU) in a process often referred to as "Brexit", following 47 years of being a member state. A transition period until the 31st December 2020 allowed the UK to participate in the EU Customs Union, European Single Market, the Common Agricultural Policy (CAP) and the Common Fisheries Policy for a time (Walker, 2019). Although the European Union (Withdrawal) Act 2018 enables much of the relevant EU law to be converted into domestic law albeit with amendments, it does not include some policies such as the EU agricultural law (e.g. the CAP). The CAP provides funding to member states who develop their own Rural Development Programmes for spending which is split into two pillars; Pillar I which is paid directly to farmers (Basic Payment Scheme allocated on a per hectare basis) and comprises of 75% of the UK CAP funding (Pe'er *et al.*, 2014); and Pillar II which funds agri-environment schemes (AES) from the remaining 25% (Science for Environment Policy, 2017b). In some cases the CAP funding provides nearly half of farm income (Lang, Millstone and Marsden, 2017), although reliance on the CAP varies by sector with dairy, poultry, pig farming and horticulture sectors less reliant on public funding (Scottish Government, 2020). Thus the loss of the CAP following Brexit could result in some farming communities' facing substantial economic hardship in the absence of any public funding (Colston, 2018). This will be particularly the case in less profitable areas, in which economic viability is often marginal but is also often associated with high value for nature, and will be further exacerbated by changes in trade and tariffs between the UK and the EU which have, to date of writing, not yet been resolved. In the long term this is likely to lead to a loss of smallholdings, abandonment, or the selling of land to other land uses such as building developments or tree planting particularly in areas where farming is no longer economically viable.

The UK Government has committed to honour subsidies and AES agreements already in place for the duration of this parliament, likely till March 2022, and has announced its intentions to formulate a novel agricultural policy based on "public funding for public goods" and the phasing out of the basic farm payment (Downing and Coe, 2018). This will be a radical shift from the general support of agriculture towards one more focussed on public goods. In particular, public goods will have an emphasis on environmental improvement such as water quality, climate change mitigation through the reduction and storage of greenhouse gas emissions, conservation and enhancement of biodiversity. Overall, the new agricultural bill aims to help the Government reach the objectives of its new 25 Year Environment Plan (H.M. Government, 2018), such as delivering on agricultural carbon emission reduction targets of net zero by 2050 (Lamb *et al.*, 2016). In England, this will take the form of Environmental Land Management schemes (ELMs) which will have three tiers (T1 to T3), with the roll-out planned by the end of 2024, and pilot schemes running 2021 - 2024 (see DEFRA's, (2018) Health and Harmony consultation timeline). At present, it has been proposed that T1 will incentivise environmentally sustainable farming and forestry such as tree or hedge planting and river management to mitigate flooding; T2 will be for locally targeted environmental outcomes; and T3 for landscape-scale action such as restoration of peat bogs. ELMs will therefore be a powerful tool in shaping the future of farming. However, there are concerns that biodiversity conservation will become a marginal aim, as agriculture increasingly becomes more concentrated on the mitigation of climate change focused outcomes (Wilson, Evans and Grice, 2009). Planting trees to sequester CO_2 is one example where conflict between bird species of conservation concern and climate change focussed outcomes may arise for waders (chapter 5).

As agriculture is a devolved area in Scotland, it will be able to adopt its own farming policy, which will include some funding from the UK treasury, although the amount is yet to be confirmed. Around 80% of Scotland's total land area consists of agriculture with the majority of this under livestock grazing (Scottish Government, 2018b), and 17% of the Scottish population live in rural areas (Scottish Government, 2018c). This therefore poses both a significant challenge, and a great opportunity, for policy makers to develop an agricultural policy that is designed for the priorities of Scotland. Some of these priorities have been identified as: maintaining a healthy and productive use of land; protecting animal welfare; advancing environmental protection; ensuring financial assistance is based on greatest need; and raising the profile of the sector through education and increasing public awareness (Diffley *et al.*, 2019). The Agriculture (Retained EU Law and Data) Scotland Bill became an Act in 2020 which makes provision for an interim approach to agricultural policy for Scotland based on simplifying and improving retained EU law. Furthermore, Scotland's Agri-Environment Climate Scheme (AECS) have been extended (albeit on a restricted basis) to 2024 to allow for continued support of organic farming and land management practices which support farmland waders, corn buntings and corncrakes, along with actions to improve public access and slurry storage to name a few (Scottish Government, 2021). Thus this should provide support to farms which rely on the CAP funding for a time, although a future policy of AES, to date of writing, has not yet been confirmed for Scotland.

A large proportion of agriculture in Scotland is under less productive (or of "less-favoured area", LFA) and therefore dominated by extensive livestock grazing (Scottish Government, 2018b). A reform of agricultural policy could therefore allow these areas to receive more attention and support than they have historically been given and a shift in emphasis from subsidy support being based on compensating agricultural disadvantage to providing active support for the maintenance of environmentally beneficial farming systems and traditional practices (Swales and Moxey, 2008). For waders, a significant amount of research effort has focused on lowland grazed and wet-grazed grasslands in England, where populations have declined in these habitats. As a result, the current AES prescriptions for waders have been predominantly designed around mitigating the negative effects of intensification on lowland habitat and compensating farmers for loss of economic value (e.g. by decreasing stocking densities, and manipulating water levels which reduces land suitable for grazing). However, in-bye land (enclosed farmland below the moorland) of upland agriculture is critically important for breeding waders, even in England where it has been reported that over 50% of England's breeding Curlew and Lapwing are now found on upland farms (Siriwardena *et al.*, 2016), and those that nest on moorland (e.g. Golden Plover) use these areas for foraging (e.g. Whittingham, Percival and Brown, (2000) and Chapter 4). The challenges facing waders on lowland grazed and wet-grassland or on moorland will likely be very different from those of marginal upland grassland. Prescriptions within AES may therefore be missing an important part of the picture in these areas, where waders are now starting to decline, even in important breeding locations (see chapter 5). As Scotland holds a high proportion of the UK's breeding population of waders e.g.(Brown *et al.*, 2015) and a large proportion of agriculture in Scotland is in these marginal areas, research on wader prescriptions targeted for upland farming and an improved understanding of the resources offered in these marginal areas are very timely. Development of a new AES for UK could provide significant benefits to species of open low-intensity agriculture. For example, changes to upland grasslands in the UK, such as gradual acidification, may have contributed to the decline in farmland breeding waders like the Curlew and Lapwing that use these fields, as a result of a decline in foraging conditions through reduced earthworm numbers in lower pH soils. Earthworms are likely to be an important food resource to waders in other systems, such as for Lapwing on arable land, where a range of agricultural operations may reduce earthworm numbers (e.g. tilling; for review see Briones and Schmidt, 2017) understanding mechanisms in upland areas could help conservation in other landscapes.

Cattle grazing in Scotland has decreased, while the numbers of sheep have increased (Scottish Government, 2018b). The change in the cattle to sheep ratio will influence the way vegetation is grazed and thus changes to the structure and composition of the vegetation may occur as well as differences in dunging which can influences invertebrates. Without identifying the support required for cattle grazing (e.g. social, economic) this trend in decline could continue which can have important nature conservation impact, as vegetation structure from grazing is important in producing the nesting and foraging habitat for birds e.g. (Fuller and Gough, 1999). Some options new to Scotland which are being explored to increase sustainability of livestock grazing are agro-forestry and mob grazing which could help farms adapt to a changing climate (Scottish Government). These would have to be carefully researched to ensure that they do not have negative impacts on waders (e.g. trampling).

6.3.2 Outcome vs prescription driven approaches to agri-environment schemes

Outcome driven (sometimes known as results driven) agri-environment schemes, as opposed to prescription driven (also known as action-orientated) agri-environment schemes, have recently been growing in interest with trials across Europe including the UK and Ireland (Herzon *et al.*, 2018; RBP, 2019). Although both schemes provide payment in support of wildlife friendly farming, results-based schemes are linked to evidence of outcomes in the habitat such as a diverse sward length, rather than evidence of completing prescribed management actions such as mowing a particular frequency or at specific times of the year as a means of achieving the desired results. In this way, farmers have the flexibility to manage land to achieve positive environmental results that are site specific and may differ between farms and fields depending on the geography and weather conditions experienced. An added benefit to these types of schemes is that regular monitoring is required to evidence that the targeted habitat results are being achieved, allowing for review of the success of management. This is very different to prescription based schemes where an agreement on annual or one off payments is made based on specific management actions that will be undertaken for a period usually of 5 years, with little monitoring of the outcome. The lack of flexibility and low priority on results have been identified as potential reasons for some of the limited success seen from prescription based AES (Batáry *et al.,* 2015) which is currently the dominant form of AES within Europe (Burton and Schwarz, 2013). Therefore, it is anticipated that results based schemes may provide better value for wildlife as payments are directly linked to the outcomes achieved on the land; the better the land is for wildlife, the higher the payment. However, determining what "good" looks like and the indicators to include in outcome based agri-environment schemes is a huge challenge (Kaiser, Reutter and Matzdorf, 2019; Ruas *et al.,* 2021). For these schemes to be a success, it is paramount that they are evidence based, regularly monitored and reviewed.

The results from this thesis could help inform trials of outcome driven AES targeted at marginal grasslands for the benefit of farmland breeding waders. One such trial is the NatureScot Piloting Outcome Based Approaches to Agri-environment Schemes (POBAS) project in Scotland (www.nature.scot/doc/piloting-outcomes-based-approach-scotland-pobas-project), which is testing this approach in five pilot areas, plus two additional pilot areas in partnership with the RSPB and the European Forum on Nature Conservation and Pastoralism, each with a specific habitat or target species. The POBAS trial in the Strathspey is focussed on grassland management for waders, and the results presented here could provide useful indicators to consider. For example, soil condition (pH) may be a useful indicator of potential food availability (e.g. earthworms) for breeding and foraging waders. Where soil pH is identified as being sub-optimal, lime management could be a useful tool in amending soil pH, thus providing suitable habitat for waders. Soil pH is relatively easy to monitor, and unlike the presence of birds or earthworms is not influenced by the weather on that day and does not require expert identification skills. Regular monitoring of soil condition through outcome driven AES, coupled with bird data could also provide powerful data to evaluate effectiveness of this variable as a predictor of outcome, and allow for flexible and adaptive management to environmental change (e.g. climate change).

6.3.3 Land sparing vs land sharing

The concept of "land sparing" involves taking land out of agricultural production for example, natural regeneration of habitat (e.g. woodland) or for nature reserves, while more intensive agriculture is conducted on other parcels of land and is focussed on high-yield as opposed to "land sharing" (also known as wildlife-friendly farming) in which food production and conservation are coupled (Green *et al.*, 2005; Phalan *et al.*, 2011; Luskin *et al.*, 2018; Lamb *et al.*, 2019). This, coupled with a reduction in consumption of meat and food waste, has been proposed could reduce net emissions from UK agriculture by over 80% (Lamb *et al.*, 2016). This would be a highly attractive option for the UK in enabling it to reach its target of net zero emissions by 2040 from the agricultural sector in England and Wales (NFU, 2019) contributing to the Governments UK wide net zero target by 2050 (H.M. Government, 2020). However, the UK has a long history of agriculture and has little "spared" land remaining and thus land sparing may be less beneficial in the UK as a conservation option as natural habitat specialists have already been lost (Lamb *et al.*, 2019). For example, broadleaved woodland, and fen and marsh habitat only account for 6% and 2% of the UK land area (Carey *et al.*, 2008). Land sparing is therefore likely to benefit woodland and wetland specialists, but will negatively affect farmland bird specialists (Lamb *et al.*, 2019). This will particularly be the case for waders which would lose suitable nesting habitat if intensification of farmland was to occur, and would be exacerbated if land sparing includes large scale tree planting and fragmentation of the open-farmed landscape by both increasing predation rate and the displacement of these birds. Tree planting and regeneration of lost natural habitat such as broadleaved woodland is likely to be a considerable area of conflict between land use in marginal areas (see chapter 5 and section 1.2.3).

6.3.4 Climate change

For Scotland, across all emission scenarios presented in the UK climate projections (UKCP18) there is a general consensus for several key long-term trends including: an increase in intense rainfall events; sea level rise; warmer and drier summers; and milder and wetter winters (MetOffice, 2019; Adaptation Scotland, 2020). Climate change therefore poses a risk to soil, natural carbon stores, agriculture, wildlife and coastal habitats. For waders in the summer, the manipulation of the water table and of wet features within fields such as man-made scrapes will therefore become increasingly important to mitigate dry soils. In winter months, rise of sea level may threaten coastal habitats used by waders as important feeding areas, with some evidence already suggesting the overwintering distribution of migratory waders have shifted. Action will also be required to protect soil from the impacts of climate change. Annual soil erosion is associated with rainfall patterns, and can therefore be expected to increase. For example, Scotland's annual average rainfall between 2009 and 2018 was 15% wetter than that of the 1961 to 1990 average, and with winters 25% wetter. Soil erosion is also impacted by land use and is likely to be greater on arable than grassland due to practices such as deep ploughing, although compaction of waterlogged soils by heavy machinery or high livestock numbers can also effect soil erosion through changes in soil structure. Changes in temperate and rainfall patterns are also likely to affect the abundance and activity of soil microflora (e.g. bacteria, fungi and protozoans) as well as soil macrofauna (e.g. earthworms), with implications for decomposition of organic matter and carbon storage, nutrient cycling and fertility-related ecosystem services. Increased rainfall could also intensify leaching and acidification of soil, with further implications on the soil fauna. Lime use, particularly on enclosed in-bye fields could therefore be an important tool in helping to mitigate some of the impacts of climate change on acidification and maintain populations of earthworms, and thus food resources for waders.

Carbon is naturally stored in soils and is considered to hold some of the largest terrestrial carbon stocks, particularly in organic (carbon-rich) soils such as peat. Vegetation growth helps to sequester CO_2 from the atmosphere into plant tissues which is then transferred to soil carbon through litter and humus. Soil conservation can therefore be important in climate change policy. However, soils can also be a source of CO_2 emissions through natural decomposition and respiration, which in some instances is also accompanied by losses of methane (mainly from wetlands) and nitrous oxide (mainly from artificial fertilisers). These gases are powerful GHG and degradation of soil condition can lead to them becoming sources, rather than sinks for carbon. Additionally, higher temperatures and changes in soil moisture could also increase carbon losses from soil due to enhanced soil respiration. The use of lime to amend acidic soil pH leads to an initial release of CO_2 emission (Equation 6.1) during the neutralising process (West and McBride, 2005; De Klein, 2006; Fornara *et al.*, 2010; Shaaban *et al.*, 2017; Holland *et al.*, 2018; Biasi *et al.*, 2008). The extraction and transport of lime also increases its carbon footprint. However, in the longer term, there is some evidence to suggest that lime use can increase carbon stocks and phosphorus uptake in soils likely due to the change in pH (Paradelo, Virto and Chenu, 2015; Barrow, 2017; Grover *et al.*, 2017) as well as reducing nitrous oxide released from soils (Hénault *et al.*, 2019). Results from research examining changes in the respiration of mircrofauna in the soil following lime use is mixed, making it difficult to determine if the overall effect in these systems is positive or negative in the context of CO_2 emissions (e.g. Holland *et al.*, 2018). Reducing GHG emissions from agriculture will play a vital role in combating climate change and is an important target of the government for net zero emissions in Scotland by 2045. Land management which potentially increases CO_2 emissions will therefore have to be considered carefully. Overall, long-term monitoring and research of soil health, in terms of soil carbon, erosion, acidification, lime and soil biota will be needed so that we can address knowledge gaps and access the magnitude of this risk, how this is linked to different land management strategies and to geographical variations in soil.

Equation 6.1. Chemical equation representing the neutralisation of acid soil with agricultural lime, $CaCO_3$ (aka reducing the ratio of free hydrogen ions in the soil solution).

$$2H^+ + CaCO_3 = Ca^- + H_20 + CO_2$$

Increasing the area of forestry in order to reduce current concentrations of CO_2 in the atmosphere is an ambitious global target which will help mitigate the impacts of climate change, as well as providing benefits in terms of natural flood management and ecotourism. Forestry accounts for 15% of land cover in Scotland equating to over 1.2 million ha and has been predicted to have sequestered 8.25 million tonnes of CO_2 in 2014. Forestry is also economically important and contributes £1 billion per year to Scottish economy and supports more than 25,000 jobs (Scottish Government, 2019). As a result, forestry is an important component of Scotland’s target to become carbon neutral. However, forestry will be affected by changes in climate including both heavy rainfall and drought periods, as well as extreme weather events and threats from pests and diseases. The Scottish Government’s 50 year vision for forests and woodlands includes restoring and managing current forestry, as well as large woodland creation targets with the aim to increase woodland cover to 21% by 2032 as committed to in the Scottish Governments Climate Change Plan. In the last year, Scotland exceeded its tree planting targets of 10,000 ha annually with more than 11,000 hectares planted. There are aims to increase this to 15,000 ha annually by 2024/5 which will require the rapid sighting of locations for planting (Scottish Forestry, 2019). This is an ambitious target which will drastically change land use and the landscape of Scotland, and will be a challenge given the limited amount of space. Deforestation in the UK has been extensive (Scottish Forestry), and new tree planting will lead to direct habitat loss of long-established wader habitat if planting occurs on suitable sites used for nesting or feeding, although there may be a lag in the effect of this while the woodland matures (e.g. Cayford, 1993; Pearce-Higgins *et al.*, 2007; Hancock, Grant and Wilson, 2009; White, Warren and Baines, 2013; Wilson *et al.*, 2014; Franks *et al.*, 2017). There is also increasing evidence of "woodland edge effects" on waders by conifer plantation in moorland and peatland habitat which causes displacement of waders which behaviourally avoid nesting close to enclosed spaces or trees (e.g. Wilson *et al.*, 2014), as well as evidence of an increase in predation rates reducing breeding success (e.g. Douglas *et al.*, 2014; Roos *et al.*, 2018). It has been proposed that the negative effects of increased planting of conifer plantations near moorland wader habitat can be mitigated with an increase in predator control (Douglas *et al.*, 2014). However, it was not known whether the same woodland edge effects held true for broadleaf woodland. The Scottish Government has pledged to increase the area of native broadleaved planting as well as of conifer plantations, which is expected to have biodiversity benefits for Scotland's native woodland species (e.g. Quine and Humphrey, 2010; Scridel, Groom and Douglas, 2017). However, the research presented in this thesis showed a strong negative effect of woodland edge on four species of wader of marginal farmland, which consisted of a large proportion of broadleaved woodland. Although this study was restricted to a relatively small study area, the implications of woodland edge (regardless of whether it is conifer plantation or native broadleaved) on waders is likely to be applicable in other similar landscapes. The threat of tree planting to combat climate change and reach net zero emissions is therefore high for waders, and careful placement of new woodland should be taken to avoid conflict between mitigating climate change and the conservation of globally Near Threatened species such as the Curlew, which Scotland supports critical numbers of and is considered one of the most pressing birds of conservation priority in the UK (Brown *et al.*, 2015).

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APPENDICES AND SUPPLEMENTARY MATERIAL

CHAPTER ONE

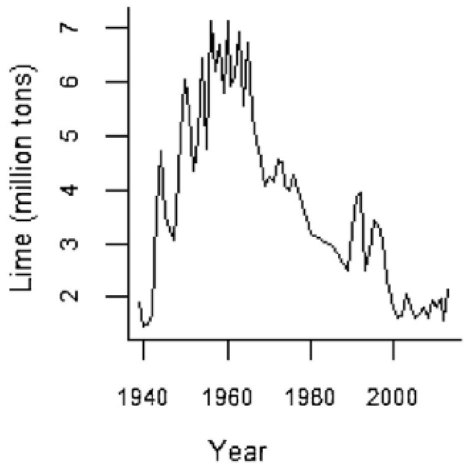


Figure A1. The quantity of lime sold in Great Britain for agricultural purposes, annually since 1939: figure from McCallum et al., (2016) with data sources: 1939–1976, Agricultural Lime Producers' Council (1977), 1980–1989, Wilkinson (1998), 1990–2000, Hillier et al. (2003), 2001–2013, Bide et al. (2015).

CHAPTER TWO

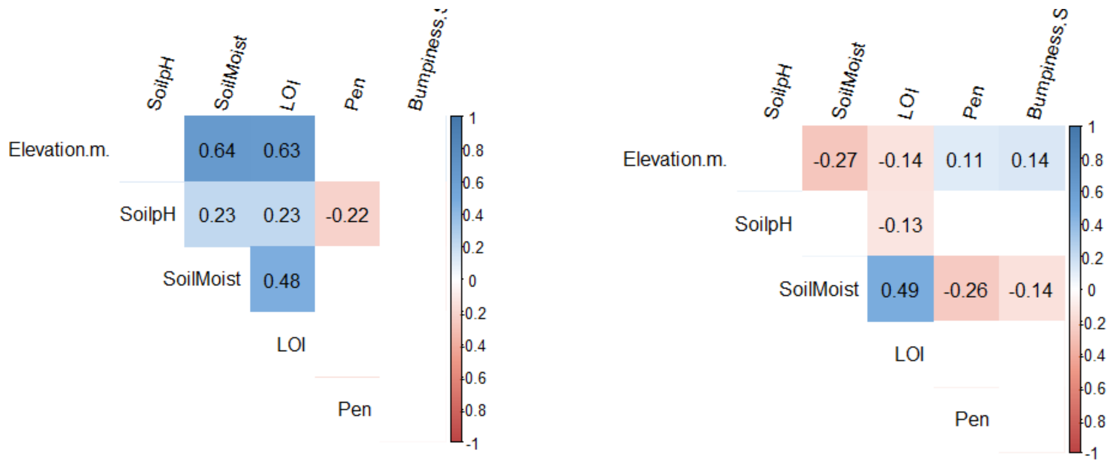


Figure A2. Correlogram representing the pearson's correlations for all pairs of numerical variables used in analysis of chapter 2 where correlogram to the left is from the observation study and correlogram on the right if from the BACI liming experiment to determine the effect of lime treatment on earthworm abundance and soil properties. Positive correlations are displayed in blue and negative correlations in red. The intensity of the colour is proportional to the correlation coefficient so the stronger the correlation (i.e., the closer to -1 or 1), the darker the boxes. The colour legend on the right hand side of the correlogram shows the correlation coefficients and the corresponding colours. A white box in the correlogram indicates that the correlation is not significantly different from 0 at the specified significance level (<0.05) for the couple of variables and is based on a correlation test. A correlation not significantly different from 0 means that there is **no linear** relationship between the two variables considered.

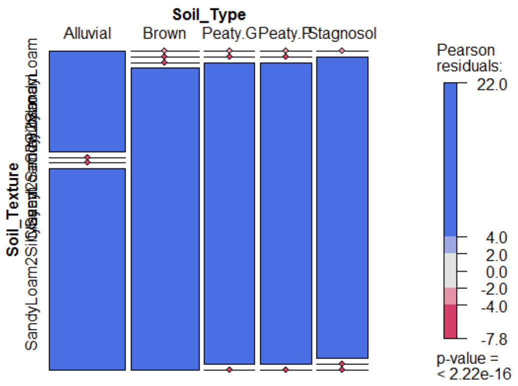


Figure A3. Chi-square test of independence tests whether there is a relationship between categorical variables displayed as a mosaic plot for the two variables of Soil Type and Soil Texture. A mosaic plot is similar to a barplot but the p-value of the Chi-square test is also displayed at the bottom right. The colours in the mosaic plot show where the observed frequencies deviates from the expected frequencies if the variables were independent. The red cases means that the observed frequencies are smaller than the expected frequencies, whereas the blue cases means that the observed frequencies are larger than the expected frequencies. In this case, soil type and soil texture are highly correlated.

Table A 0-1. Fisher's exact test performed with simulated p-value based on 2000 replicates to test for relationship between categorical variables which have small sample sizes (< 5 counts for a category level) instead of chi-square test of independence. In this case, soil type and soil texture were highly significantly related to both the farm and field ID.

	Farm	Field ID
Soil Type	<0.001	<0.001
Soil Texture	<0.001	<0.001

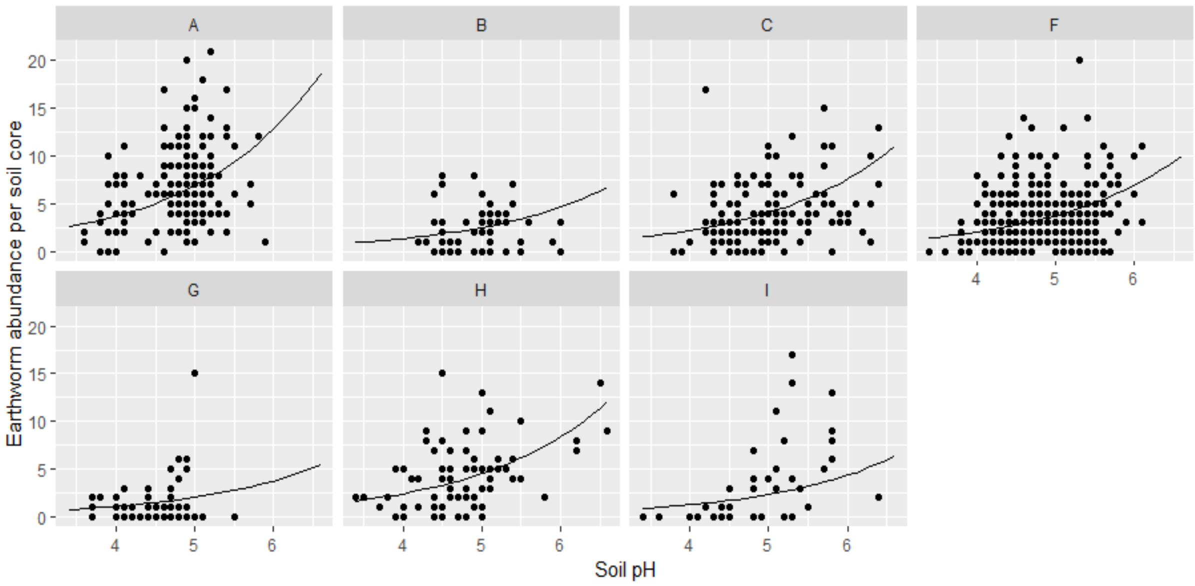


Figure A4. Scatter plots of earthworm abundance against soil core pH at the farm scale. Note that filled black circles represent the raw data, and the solid black line is the model predictions.

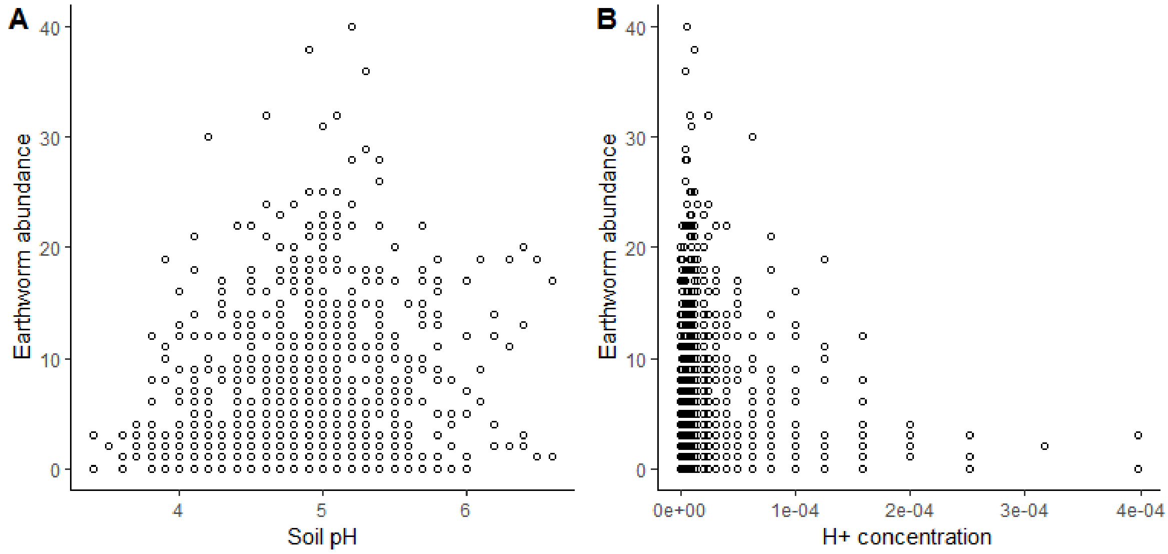


Figure A6. Scatter plot showing the raw data of earthworm abundance by soil pH on **(A)** the negative logarithmic soil pH scale; and **(B)** as the H+ concentration. Each data point represents a single soil core.

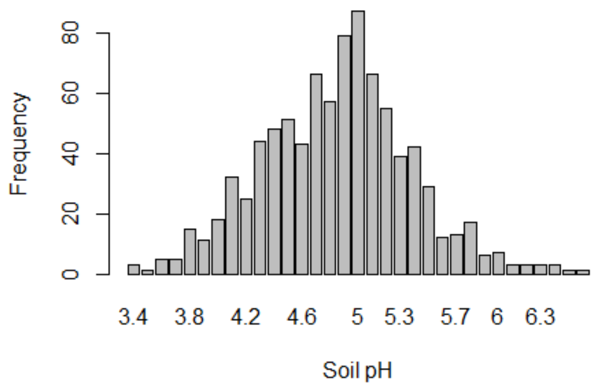


Figure A5. Bar plot showing the frequency of raw data of the soil pH values recorded in this study.

Table A 0-2. Comparison of soil $pH_{(CaCl2)}$ analyses from the mixed field sample (n =20 cores) between SAC Consulting and the University of Stirling (UoS) soils laboratory.

	A1	A2	A3	A4	B5	C6	C7	F10	F11	F8	F9	G12	H18
SAC Consulting	5.1	5.2	5.4	5.0	5.3	5.4	5.0	-	-	-	-	4.9	-
UoS	5.1	5.1	5.4	4.9	5.4	5.4	5.2	5.0	5.3	5.3	5.4	5.0	4.9

Table A 0-3. Table provided by Agronomists at SAC Consulting on the recommended lime requirements as t/ha of CaO required to get a field into agricultural optimum based on the baseline soil pH, land use (arable vs permanent grassland) and soil type.

LIME REQUIREMENT VALUES (t/ha)

Soil pH	Sands (S, LS)		Light Soils (SL, SZL, ZL)		Medium Soils (SCL, CL, ZCL)		Heavy Soils (SC, C, ZC)		Organic Soils (OL)		Peaty Soils (PL)		Peat Soils (LP)		Peats (P)	
	AC/TG	PG	AC/TG	PG	AC/TG	PG	AC/TG	PG	AC/TG	PG	AC/TG	PG	AC/TG	PG	AC/TG	PG
6.4	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
6.3	–	–	–	–	2	–	2	–	–	–	–	–	–	–	–	–
6.2	–	–	–	–	2	–	3	–	–	–	–	–	–	–	–	–
6.1	–	–	2	–	3	–	4	–	–	–	–	–	–	–	–	–
6.0	2	–	3	–	4	–	5	–	2	–	–	–	–	–	–	–
5.9	2	–	4	–	5	–	5	–	3	–	–	–	–	–	–	–
5.8	3	–	4	–	6	2	6	2	4	–	2	–	–	–	–	–
5.7	4	2	5	2	6	2	7	3	5	–	3	–	–	–	–	–
5.6	4	2	6	3	7	3	8	4	6	–	4	–	2	–	–	–
5.5	5	3	6	4	8	4	9	5	7	2	6	–	4	–	–	–
5.4	5	4	7	4	9	5	10	5	8	3	7	–	5	–	–	–
5.3	6	4	8	5	10	6	11	6	9	4	8	2	6	–	3	–
5.2	7	5	8	6	10	6	12	7	10	5	9	3	7	–	4	–
5.1	7	5	9	6	11	7	13	8	11	6	10	4	8	2	6	–
5.0	8	6	10	7	12	8	14	9	12	7	11	6	10	4	7	–
4.9	8	7	11	8	13	9	14	10	13	8	12	7	11	5	8	–
4.8	9	7	11	8	14	10	15	11	14	9	13	8	12	6	10	3
4.7	10	8	12	9	14	10	16	12	15	10	14	9	13	7	11	4
4.6	10	8	13	10	15	11	17	13	16	11	15	10	14	8	13	6
4.5	11	9	13	11	16	12	18	14	17	12	17	11	16	10	14	7
4.4	11	10	14	11	17	13	19	14	18	13	18	12	17	11	15	8
4.3	12	10	15	12	18	14	20	15	19	14	19	13	18	12	17	10

AC/TG = Arable Crops/ Temporary Grass
PG = Permanent Grassland.

Table A 0-4. Technical data on lime purchased for use in the BACI liming experiment. Personal details shaded out.

LEITHS
Built on Quality

TECHNICAL DATA SHEET

COVE
ABERDEEN
AB12 3LR
T: 01224 876333
W: leiths-group.co.uk

MATERIAL: CRUSHED AGRICULTURAL LIMESTONE
SOURCE: PARKMORE QUARRY, DUFFTOWN.

TYPICAL VALUES

CONSTITUENT	CRITERIA	VALUE
CARBONATE (AS CO_2)	TYPICAL VALUE	38%
CARBONATE (AS $CaCO_3$)	TYPICAL VALUE	86%
MAGNESIUM (AS MgO)	TYPICAL VALUE	1%

AVERAGE TNV (10 Results)	45
Specified Limit:	42 minimum.
Standard Deviation of TNV (10 Results)	1.8

Sieve Size (mm)	% Passing (Average 10 results)
6,3	100
5	100
4	97
1	56
0,250	35
0,125	27
0,063	18

UKAS TESTING
2239

Lime Report

SAC
Commercial Limited

Client:

Your reference:	Leadmore Lime

Lab sample no:	12500012
Batch no:	G40333
Date received:	26/03/2012
Date reported:	03/04/2012

Determination	Result	Units
Dry matter	97.0	%
Total potassium	0.10	%DM
Total phosphorus	0.008	%DM
Neutralising Value	53.3	% CaO
Aqua regia calcium	17.6	%DM
Aqua regia magnesium	12.1	%DM
Aqua regia sodium	0.06	%DM
Aqua regia zinc	< 15.2	mg/kgDM
Aqua regia copper	< 3.34	mg/kgDM
Aqua regia manganese	72.8	mg/kgDM
Aqua regia sulphur	< 0.01	mg/kgDM

Contact: Ricky Marwick
Inverness FBS

Authorised by June Gay (Client Manager): June Gay

Page 1 of 1

Analytical Services Department, Central Analytical Laboratory, SAC Bush Estate, Penicuik, Midlothian, EH26 0QE
No other party may rely on the report and if they do so, then they rely upon it at their own risk.
All work undertaken is in accordance with our written Standard Terms and Conditions of Supply and Service.

Table A 0-5. Species inventory by site and treatment for the liming experiment (n = 7 farms), where C = control plot and T = lime treatment. Data is pooled from across all years of the project. Number of soil cores is the number of samples which had at least one adult species identified. Species ecological groupings included; Epigeic = smallish earthworms which live in the litter and on the top of the soil surface. These species do not make burrows and consume rotting organic matter such as leaves. Endogeic = species which live in the top soil. These earthworms are medium sized, eat soil and make horizontal burrows. Anecic = large, long lived species which make permanent vertical burrows. Feed on vegetation on the soil surface which they drag into their burrows.

Species	Ecological Grouping	Site A		Site B		Site C		Site F		Site G		Site H		Site I	
		C	T	C	T	C	T	C	T	C	T	C	T	C	T
A. chlorotica	Endogeic	√	√	-	-	√	√	-	√ - 1 only	-	-	√	√	√	√
A. caliginosa	Endogeic	√	√	√	√	√	√	√	√	√	√	√	√	√	√
A. limicola	Endogeic	√	√	-	-	√	√ - 1 only	-	√	-	-	-	-	-	-
A. longa	Anecic	√	√	-	-	√	√	√ - 1 only	-	-	-	-	-	-	-
A. nocturna	Anecic	-	√ - 1 only	-	-	-	-	-	-	-	-	-	-	-	-
A. rosea	Endogeic	√	√	-	√ - 1 only	√	√	√	√ - 1 only	-	-	√	-	√ - 1 only	√ - 1 only
D. hortensis	Epigeic	√ - 1 only	-	-	-	-	-	-	√ - 1 only	-	-	-	-	-	-
D. octaedra	Epigeic	-	-	√ - 1 only	-	-	-	√	-	-	-	-	-	-	-
D. rubidus	Epigeic	-	-	-	-	-	√ - 1 only	√	√	-	√ - 1 only	-	√	-	√ - 1 only
E. tetraedra	Epigeic	-	√	-	-	-	√ - 1 only	√ - 1 only	-	-	-	-	-	-	√
L castaneus	Epigeic	√ - 1 only	-	-	-	-	-	√ - 1 only	-	-	-	-	-	-	-
L. festivus	Epigeic	√ - 1 only	-	√ - 1 only	-	-	√ - 1 only	-	-	-	-	-	-	-	-
L. friend	Epigeic	-	-	-	-	√ - 1 only	-	-	-	-	-	-	-	-	-
L. rubellus	Epigeic	-	-	-	-	-	√	-	√	-	-	-	√ - 1 only	-	-
M. muldali	Endogeic	-	-	-	-	-	-	-	-	-	-	-	√ - 1 only	-	-
O. cyaneum	Endogeic	-	-	-	-	√ - 1 only	-	-	-	-	-	-	-	-	-
O. lacteum	Endogeic	-	-	√	√	-	-	√ - 1 only	√	-	√ - 1 only	√ - 1 only	-	-	√ - 1 only
S. mammalis	Epigeic	√	√	-	-	-	√	-	-	-	-	√	√ - 1 only	-	-
Number soil cores		80	80	30	30	80	80	160	160	36	24	52	28	40	40
Species Richness		9	8	4	3	7	10	8	8	1	3	5	6	3	6
		11		5		12		12		3		8		6	

Table A 0-6. Summary table of the BACI lime experiment with the mean ± standard deviation (SD) per soil core of soil pH with mean ± SD per soil core of earthworm abundance shown in brackets at the field treatment half and by year. Please note that fields I16 and I17 from Northern Ireland had soil analysis conducted at the field half rather than at the soil core level in Year 0 and Year 1 and therefore only a pH value is shown with no SD for these fields. Year0 = baseline data prior to any application of lime treatment. Year1 = data one year post the application of lime onto treatment plots; Year2 = data two years post the application of lime onto treatment plots; Year3 = data three years post the application of lime onto treatment plots; C = control; T = lime treatment.

Field	Year 0 (Baseline)		Year 1		Year 2		Year 3	
	C	*T*	*C*	*T*	*C*	*T*	*C*	*T*
A1	4.7 ± 0.39	5.0 + 0.17	4.9 + 0.11	4.9 + 0.04	4.1 + 0.13	4.4 + 0.40	5.1 + 0.16	5.3 + 0.29
	(9.6 + 7.30)	(13.8 ± 3.35)	(8.8 ± 2.59)	(17.4 ± 13.35)	(8.2 ± 3.63)	(9.2 ± 4.15)	(11.4 ± 8.47)	(13.0 ± 8.72)
A2	4.9 + 0.18	4.9 + 0.17	4.9 + 0.19	4.9 + 0.21	4.1 + 0.30	4.1 + 0.29	5.0 + 0.11	5.1 + 0.26
	(5.4 ± 4.51)	(6.0 ± 4.00)	(9.8 ± 5.07)	(10.6 ± 7.54)	(7.4 ± 7.70)	(9.0 ± 3.74)	(14.6 ± 4.82)	(17.0 ± 6.67)
A3	4.9 + 0.21	5.1 + 0.10	4.7 + 0.42	5.4 + 0.10	3.8 + 0.25	4.4 + 0.56	4.9 + 0.12	5.4 + 0.25
	(14.3 ± 8.96)	(11.3 ± 3.51)	(13.7 ± 10.21)	(12.0 ± 6.93)	(3.3 ± 4.93)	(13.0 ± 5.57)	(6.7 ± 4.93)	(15.7 ± 6.51)
A4	4.8 + 0.18	4.9 + 0.19	5.0 + 0.20	5.0 + 0.08	4.1 + 0.24	4.2 + 0.46	5.2 + 0.44	5.2 + 0.19
	(16.6 ± 11.04)	(20.7 ± 6.68)	(15.6 ± 9.43)	(18.4 ± 7.23)	(16.4 ± 9.05)	(11.3 ± 6.97)	(13.1 ± 8.03)	(13.8 ± 4.43)
B5	5.1 + 0.42	5.2 + 0.20	5.1 + 0.26	5.3 + 0.33	-	-	-	-
	(4.3 ± 3.90)	(4.6 ± 3.47)	(5.0 ± 3.72)	(5.0 ± 3.43)				
C6	5.0 + 0.15	5.1 + 0.26	4.9 + 0.27	4.8 + 0.19	4.4 + 0.09	5.0 + 0.28	5.1 + 0.14	5.8 + 0.49
	(7.2 ± 4.34)	(5.6 ± 3.57)	(7.9 ± 3.81)	(6.9 ± 5.28)	(5.7 ± 3.89)	(6.0 ± 4.69)	(9.3 ± 6.24)	(10.0 ± 7.43)
C7	4.8 + 0.32	4.7 + 0.30	4.4 + 0.13	5.7 + 0.17	4.2 + 0.26	4.8 + 0.40	5.0 + 0.31	5.8 + 0.52
	(5.1 ± 6.37)	(5.8 ± 3.08)	(3.5 ± 3.37)	(9.6 ± 4.60)	(3.2 ± 3.52)	(13.0 ± 7.38)	(6.3 ± 5.68)	(8.8 ± 6.05)
F10	4.8 + 0.12	4.8 + 0.20	4.0 + 0.16	4.1 + 0.23	4.0 + 0.41	4.7 + 0.40	5.0 + 0.35	5.7 + 0.30
	(3.4 ± 2.67)	(3.3 ± 0.47)	(4.4 ± 3.50)	(3.7 ± 2.83)	(3.1 ± 3.92)	(9.5 ± 10.56)	(7.5 ± 8.00)	(11.9 ± 5.0)
F11	5.2 + 0.13	5.3 +0.22	4.3 + 0.13	4.6 + 0.20	4.5 + 0.28	4.8 + 0.43	5.3 + 0.20	5.5 + 0.18
	(2.7 ± 2.58)	(7.3 ± 7.57)	(5.0 ± 4.74)	(7.3 ± 7.06)	(8.2 ± 4.54)	(10.2 ± 6.34)	(7.5 ± 4.95)	(7.9 ± 5.22)
F8	5.1 + 0.20	5.0 + 0.12	4.6 + 0.21	4.7 + 0.10	4.6 + 0.26	4.8 + 0.44	5.1 + 0.18	5.6 + 0.23
	(3.9 ± 3.98)	(4.8 ± 2.86)	(6.6 ± 6.80)	(8.8 ± 4.29)	(6.7 ± 4.64)	(10.7 ± 6.33)	(5.2 ± 4.85)	(6.1 ± 4.09)
F9	5.4 + 0.22	5.3 + 0.17	4.3 + 0.25	4.5 + 0.37	4.6 + 0.42	4.6 + 0.49	5.3 + 0.14	5.5 + 0.28
	(7.5 ± 5.76)	(5.6 ± 3.72)	(4.7 ± 4.11)	(5.1 ± 3.41)	(1.4 ± 2.07)	(4.4 ± 3.75)	(4.7 ± 3.23)	(2.9 ± 1.97)
G12	4.7 + 0.39	4.7 + 0.23	3.9 + 0.18	4.3 + 0.45	-	-	-	-
	(1.9 ± 2.73)	(0.4 ± 0.65)	(2.9 ± 1.77)	(3.7 ± 6.21)				
H18	4.7 + 0.26	4.5 + 0.28	4.8 + 0.19	4.7 + 0.19	4.2 + 0.23	4.6 + 0.85	4.9 + 0.29	5.8 + 0.81
	(3.5 ± 2.16)	(3.4 ± 3.36)	(9.7 ± 7.90)	(6.3 ± 3.99)	(7.7 ± 5.42)	(6.2 ± 4.85)	(9.7 ± 4.38)	(12.4 ± 6.55)
I16	5.3	5.3	5.5	5.8	4.8 + 0.63	5.3 + 0.70	-	-
	(5.8 ± 2.30)	(7.0 ± 8.4)	(5.7 ± 6.66)	(11.0 ± 2.83)	(0.8 ± 1.60)	(9.1 ± 9.26)		
I17	4.8	4.8	4.8	5.5	4.8 + 1.13	5.1 + 0.54	-	-
	(6.4 ± 1.06)	(8.8 ± 2.53)	(3.5 ± 1.71)	(4.3 ± 4.04)	(1.7 ± 1.53)	(2.9 ± 4.18)		

CHAPTER THREE

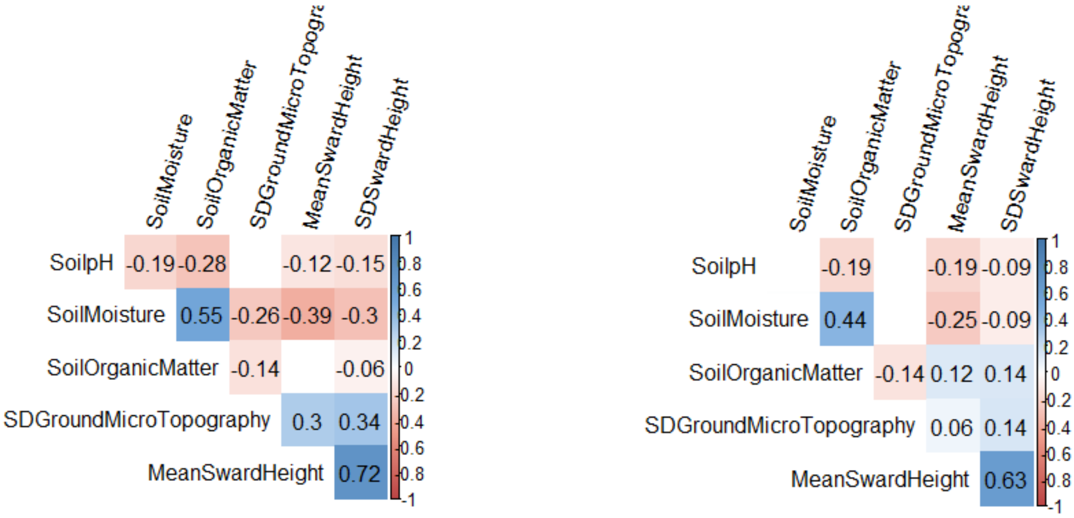


Figure A7. Correlogram representing the pearson's correlations for all pairs of numerical variables used in analysis of chapter 3 where correlogram to the left is from the early pitfall traps and correlogram on the right if from the late pitfall trap liming experiment to determine the effect of lime treatment on spider and beetle abundance and vegetation properties. The colour legend on the right hand side of the correlogram shows the correlation coefficients and the corresponding colours with positive correlations displayed in blue and negative correlations in red. A white box in the correlogram indicates that the correlation is not significantly different from 0 at the specified significance level (<0.05) for the couple of variables and is based on a correlation test.

CHAPTER FOUR

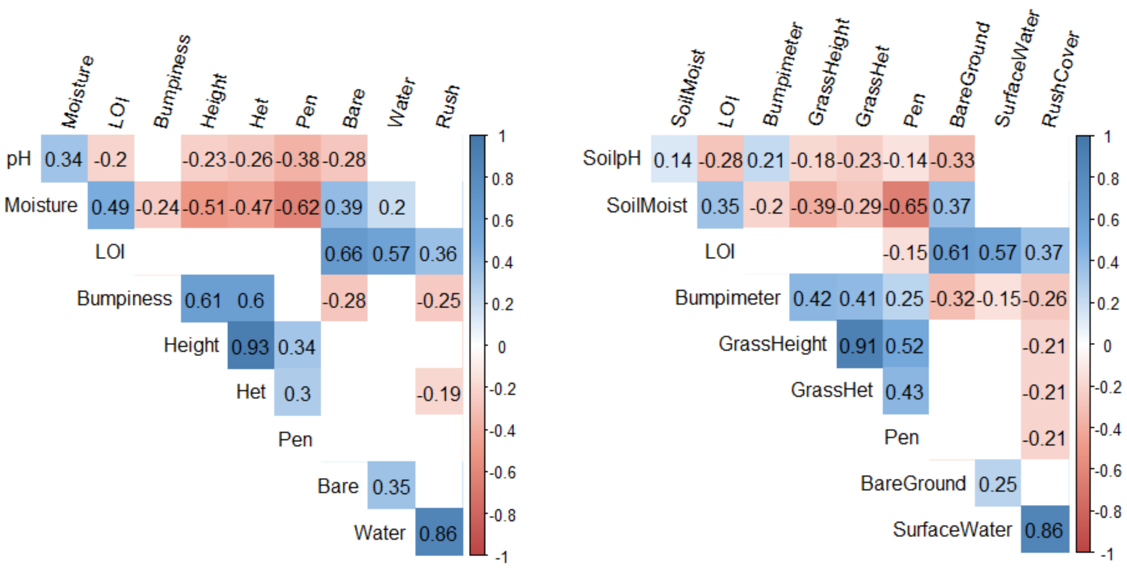


Figure A8. Correlogram representing the pearson's correlations for all pairs of numerical variables used in analysis of chapter 4 where correlogram to the left is from wader spot counts and correlogram on the right if from wader foraging observations investigating the effect of lime treatment on wader behaviour. The colour legend on the right hand side of the correlogram shows the correlation coefficients and the corresponding colours with positive correlations displayed in blue and negative correlations in red. A white box in the correlogram indicates that the correlation is not significantly different from 0 at the specified significance level (<0.05) for the couple of variables and is based on a correlation test.

Table A0-7. Estimated marginal means (EMMs) for the farmland breeding wader peck rate GLMM. Values computed by averaging together predictions and then back transforming from the transformed response with covariates set to mean values. A 95% confidence level was used. A pairwise tukey method was used for comparing EMMs with significant p values shown in bold.

	Marginal mean estimate ha^{-1}	*Lower CI*	*Upper CI*	*Pairwise comparison*	*Test Statistic*	*p value*
Peck rate – Year : Species						
2015 : Curlew	4.45	2.40	8.25	2015 : Cu – 2016 : Cu	t = - 0.845	0.9995
2016 : Curlew	5.85	4.09	8.37	2015 : Cu – 2017 : Cu	t = 0.497	1.0000
2017 : Curlew	3.81	2.61	5.55	2015 : Cu – 2015 : GP	t = 1.035	0.9967
2015 : Golden plover	3.01	1.78	5.11	2015 : Cu – 2016 : GP	t = 2.693	0.2369
2016 : Golden plover	1.74	1.10	2.77	2015 : Cu – 2017 : GP	t = 2.665	0.2513
2017 : Golden plover	1.62	0.94	2.84	2015 : Cu – 2015 : L.	t = 2.479	0.3587
2015 : Lapwing	2.05	1.46	2.90	2015 : Cu – 2016 : L.	t = 2.411	0.4035
2016 : Lapwing	1.86	1.21	2.87	2015 : Cu – 2017 : L.	t = 2.219	0.5374
2017 : Lapwing	1.27	0.49	3.29	2015 : Cu – 2015 : OC	t = 0.856	0.9994
2015 : Oystercatcher	3.31	2.03	5.40	2015 : Cu – 2016 : OC	t = - 1.618	0.9008
2016 : Oystercatcher	8.75	4.60	16.62	2015 : Cu – 2017 : OC	t = 0.931	0.9987
2017 : Oystercatcher	3.16	1.94	5.15	2016 : Cu – 2017 : Cu	t = 1.983	0.7042
				2016 : Cu – 2015 : GP	t = 2.241	0.5218
				2016 : Cu – 2016 : GP	t = 4.583	**0.0005**
				2016 : Cu – 2017: GP	t = 4.224	**0.0021**
				2016 : Cu – 2015 : L.	t = 4.493	**0.0007**
				2016 : Cu – 2016 : L.	t = 4.566	**0.0005**
				2016 : Cu – 2017 : L.	t = 2.991	0.1184
				2016 : Cu – 2015: OC	t = 2.048	0.6599
				2016 : Cu – 2016 : OC	t = - 1.154	0.9916
				2016 : Cu – 2017 : OC	t = 2.156	0.5831
				2017 : Cu – 2015 : GP	t = 0.910	0.9990
				2017 : Cu – 2016 : GP	t = 3.570	**0.0218**
				2017 : Cu – 2017 : GP	t = 3.199	0.0677
				2017 : Cu – 2015 : L.	t = 2.870	0.1594
				2017 : Cu – 2016 : L.	t = 2.917	0.1424
				2017 : Cu – 2017 : L.	t = 2.174	0.5705
				2017 : Cu – 2015 : OC	t = 0.534	1.0000
				2017 : Cu – 2016 : OC	t = -2.404	0.4081
				2017 : Cu – 2017 : OC	t = 0.689	0.9999
				2015 : GP – 2016 : GP	t = 2.078	0.6393

2015 : GP – 2017 : GP	t = 2.025	0.6763
2015 : GP – 2015 : L.	t = 1.348	0.9716
2015 : GP – 2016 : L.	t = 1.597	0.9085
2015 : GP – 2017 : L.	t = 1.55	0.9162
2015 : GP – 2015 : OC	t = - 0.276	1.0000
2015 : GP – 2016 : OC	t = - 2.610	0.2807
2015 : GP – 2017 : OC	t = - 0.137	1.0000
2016 : GP – 2017 : GP	t = 0.248	1.0000
2016 : GP – 2015 : L.	t = - 0.658	1.0000
2016 : GP – 2016 : L.	t = - 0.254	1.0000
2016 : GP – 2017 : L.	t = 0.591	1.0000
2016 : GP – 2015 : OC	t = - 2.064	0.6490
2016 : GP – 2016 : OC	t = - 4.202	**0.0022**
2016 : GP – 2017 : OC	t = - 1.859	0.7831
2017 : GP – 2015 : L.	t = - 0.795	0.9997
2017 : GP – 2016 : L.	t = - 0.441	1.0000
2017 : GP – 2017 : L.	t = 0.446	1.0000
2017 : GP – 2015 : OC	t = - 2.051	0.6576
2017 : GP – 2016 : OC	t = - 4.084	**0.0035**
2017 : GP – 2017 : OC	t = - 1.871	0.7755
2015 : L. – 2016 : L.	t = 0.376	1.0000
2015 : L. – 2017 : L.	t = 0.940	0.9986
2015 : L. – 2015 : OC	t = - 1.750	0.8424
2015 : L. – 2016 : OC	t = - 4.076	**0.0036**
2015 : L. – 2017 : OC	t = - 1.510	0.9365
2016 : L. – 2017 : L.	t = 0.720	0.9999
2016 : L. – 2015 : OC	t = - 1.857	0.7842
2016 : L. – 2016 : OC	t = - 4.049	**0.0040**
2016 : L. – 2017 : OC	t = - 1.643	0.8911
2017 : L. – 2015 : OC	t = - 1.864	0.7799
2017 : L. – 2016 : OC	t = - 3.709	**0.0137**
2017 : L. – 2017 : OC	t = - 1.903	0.7559
2015 : OC – 2016 : OC	t = - 2.658	0.2546
2015 : OC – 2017 : OC	t = 0.156	1.0000
2016 : OC – 2017 : OC	t = 3.051	0.1014

Table A0-8. Estimated marginal means (EMMs) for the farmland breeding wader intake rate GLMM. Values computed by averaging together predictions and then back transforming from the transformed response with covariates set to mean values. A 95% confidence level was used. A pairwise tukey method was used for comparing EMMs with significant p values shown in bold.

	Marginal mean estimate ha^{-1}	*Lower CI*	*Upper CI*	*Pairwise comparison*	*Test Statistic*	*p value*
Intake rate – Year : Treatment						
2015 : Control	0.58	0.37	0.91	2015 : Control – 2016 : Control	t = - 2.790	0.0629
2016 : Control	1.20	0.83	1.74	2015 : Control – 2017 : Control	t = 2.227	0.2295
2017 : Control	0.32	0.20	0.53	2015 : Control – 2015 : Lime	t = 0.532	0.9948
2015 : Lime treatment	0.51	0.33	0.79	2015 : Control – 2016 : Lime	t = - 1.120	0.8727
2016 : Lime treatment	0.74	0.50	1.09	2015 : Control – 2017 : Lime	t = - 0.363	0.9992
2017 : Lime treatment	0.64	0.40	1.02	2016 : Control – 2017 : Control	t = 4.769	**<0.0001**
				2016 : Control – 2015 : Lime	t = 3.317	**0.0135**
				2016 : Control – 2016 : Lime	t = 2.179	0.2519
				2016 : Control – 2017 : Lime	t = 2.305	0.1966
				2017 : Control – 2015 : Lime	t = - 1.733	0.5113
				2017 : Control – 2016 : Lime	t = - 3.699	**0.0037**
				2017 : Control – 2017 : Lime	t = - 3.247	**0.0168**
				2015 : Lime – 2016 : Lime	t = - 1.725	0.5171
				2015 : Lime – 2017 : Lime	t = - 0.824	0.9629
				2016 : Lime – 2017 : Lime	t = 0.681	0.9839
Intake rate – Year : Species						
2015 : Curlew	0.45	0.22	0.89	2015 : Cu – 2016 : Cu	t = - 3.687	**0.0148**
2016 : Curlew	1.82	1.30	2.55	2015 : Cu – 2017 : Cu	t = - 0.297	1.0000
2017 : Curlew	0.50	0.32	0.77	2015 : Cu – 2015 : GP	t = 0.353	1.0000
2015 : Golden plover	0.38	0.21	0.71	2015 : Cu – 2016 : GP	t = - 1.688	0.8718
2016 : Golden plover	0.87	0.51	1.47	2015 : Cu – 2017 : GP	t = 1.113	0.9938
2017 : Golden plover	0.27	0.14	0.54	2015 : Cu – 2015 : L.	t = - 0.865	0.9994
2015 : Lapwing	0.60	0.38	0.94	2015 : Cu – 2016 : L.	t = - 0.868	0.9993
2016 : Lapwing	0.64	0.38	1.08	2015 : Cu – 2017 : L.	t = - 0.713	0.9999
2017 : Lapwing	0.68	0.26	1.78	2015 : Cu – 2015 : OC	t = - 1.571	0.9177
2015 : Oystercatcher	0.85	0.46	1.49	2015 : Cu – 2016 : OC	t = - 1.239	0.9851
2016 : Oystercatcher	0.78	0.41	1.50	2015 : Cu – 2017 : OC	t = - 0.044	1.0000
2017 : Oystercatcher	0.45	0.26	0.80	2016 : Cu – 2017 : Cu	t = 5.304	**<0.0001**
				2016 : Cu – 2015 : GP	t = 4.721	**0.0003**
				2016 : Cu – 2016 : GP	t = 2.550	0.3152

2016 : Cu – 2017: GP	t = 5.168	**<0.0001**
2016 : Cu – 2015 : L.	t = 4.127	**0.0030**
2016 : Cu – 2016 : L.	t = 3.704	**0.0140**
2016 : Cu – 2017 : L.	t = 1.913	0.7500
2016 : Cu – 2015: OC	t = 2.579	0.2985
2016 : Cu – 2016 : OC	t = 2.374	0.4286
2016 : Cu – 2017 : OC	t = 4.320	**0.0014**
2017 : Cu – 2015 : GP	t = 0.911	0.9990
2017 : Cu – 2016 : GP	t = - 2.320	0.4655
2017 : Cu – 2017 : GP	t = 1.900	0.7582
2017 : Cu – 2015 : L.	t = - 0.796	0.9997
2017 : Cu – 2016 : L.	t = - 0.883	0.9992
2017 : Cu – 2017 : L.	t = - 0.622	1.0000
2017 : Cu – 2015 : OC	t = - 1.781	0.8266
2017 : Cu – 2016 : OC	t = - 1.312	0.9768
2017 : Cu – 2017 : OC	t = 0.281	1.0000
2015 : GP – 2016 : GP	t = - 2.733	0.2177
2015 : GP – 2017 : GP	t = 0.949	0.9985
2015 : GP – 2015 : L.	t = - 1.387	0.9649
2015 : GP – 2016 : L.	t = - 1.555	0.9227
2015 : GP – 2017 : L.	t = - 1.021	0.9971
2015 : GP – 2015 : OC	t = - 2.079	0.6382
2015 : GP – 2016 : OC	t = - 1.705	0.8640
2015 : GP – 2017 : OC	t = - 0.437	1.0000
2016 : GP – 2017 : GP	t = 3.602	**0.0197**
2016 : GP – 2015 : L.	t = 1.253	0.9838
2016 : GP – 2016 : L.	t = 0.960	0.9983
2016 : GP – 2017 : L.	t = 0.446	1.0000
2016 : GP – 2015 : OC	t = 0.062	1.0000
2016 : GP – 2016 : OC	t = 0.250	1.0000
2016 : GP – 2017 : OC	t = 1.742	0.8468
2017 : GP – 2015 : L.	t = - 2.219	0.5375
2017 : GP – 2016 : L.	t = - 2.179	0.5663
2017 : GP – 2017 : L.	t = - 1.558	0.9218
2017 : GP – 2015 : OC	t = - 2.750	0.2098
2017 : GP – 2016 : OC	t = - 2.320	0.4659
2017 : GP – 2017 : OC	t = - 1.208	0.9879
2015 : L. – 2016 : L.	t = - 0.202	1.0000

				2015 : L. – 2017 : L.	t = - 0.233	1.0000
				2015 : L. – 2015 : OC	t = - 1.069	0.9957
				2015 : L. – 2016 : OC	t = - 0.717	0.9999
				2015 : L. – 2017 : OC	t = 0.850	0.9995
				2016 : L. – 2017 : L.	t = - 0.104	1.0000
				2016 : L. – 2015 : OC	t = - 0.794	0.9997
				2016 : L. – 2016 : OC	t = - 0.498	1.0000
				2016 : L. – 2017 : OC	t = 0.931	0.9987
				2017 : L. – 2015 : OC	t = - 0.419	1.0000
				2017 : L. – 2016 : OC	t = - 0.280	1.0000
				2017 : L. – 2017 : OC	t = 0.852	0.9994
				2015 : OC – 2016 : OC	t = 0.211	1.0000
				2015 : OC – 2017 : OC	t = 1.786	0.8242
				2016 : OC – 2017 : OC	t = 1.616	0.9017

Table A0-9. Estimated marginal means (EMMs) for the farmland breeding wader peck success rate GLMM. Values computed by averaging together predictions and then back transforming from the transformed response with covariates set to mean values. A 95% confidence level was used. A pairwise tukey method was used for comparing EMMs with significant p values shown in bold.

	Marginal mean estimate ha^{-1}	*Lower CI*	*Upper CI*	*Pairwise comparison*	*Test Statistic*	*p value*
Peck success rate – Year : Treatment						
2015 : Control	18.4%	13.2%	25.0%	2015 : Control – 2016 : Control	z = - 4.249	**0.0003**
2016 : Control	40.4%	32.0%	49.4%	2015 : Control – 2017 : Control	z = - 1.056	0.8985
2017 : Control	23.1%	15.9%	32.3%	2015 : Control – 2015 : Lime	z = - 1.877	0.4163
2015 : Lime treatment	24.4%	19.1%	30.5%	2015 : Control – 2016 : Lime	z = - 3.738	**0.0026**
2016 : Lime treatment	35.1%	27.5%	43.4%	2015 : Control – 2017 : Lime	z = - 2.109	0.2824
2017 : Lime treatment	28.6%	20.8%	38.0%	2016 : Control – 2017 : Control	z = 2.878	**0.0461**
				2016 : Control – 2015 : Lime	z = 3.184	**0.0182**
				2016 : Control – 2016 : Lime	z = 1.433	0.7066
				2016 : Control – 2017 : Lime	z = 1.913	0.3942
				2017 : Control – 2015 : Lime	z = - 0.259	0.9998
				2017 : Control – 2016 : Lime	z = - 2.250	0.2151
				2017 : Control – 2017 : Lime	z = - 1.544	0.6356
				2015 : Lime – 2016 : Lime	z = - 2.337	0.1791
				2015 : Lime – 2017 : Lime	z = - 0.829	0.9623

				2016 : Lime – 2017 : Lime	z = 1.166	0.8530
Intake rate – Year : Species						
2015 : Curlew	22.3%	12.8%	35.9%	2015 : Cu – 2016 : Cu	z = - 2.101	0.6217
2016 : Curlew	39.6%	31.4%	48.4%	2015 : Cu – 2017 : Cu	z = 1.330	0.9753
2017 : Curlew	14.6%	10.5%	20.1%	2015 : Cu – 2015 : GP	z = 1.004	0.9977
2015 : Golden plover	15.5%	9.5%	24.5%	2015 : Cu – 2016 : GP	z = - 3.481	**0.0251**
2016 : Golden plover	55.7%	42.3%	68.3%	2015 : Cu – 2017 : GP	z = - 0.020	1.0000
2017 : Golden plover	22.5%	12.7%	36.7%	2015 : Cu – 2015 : L.	z = - 0.627	1.0000
2015 : Lapwing	26.7%	19.3%	35.8%	2015 : Cu – 2016 : L.	z = - 2.123	0.6062
2016 : Lapwing	43.8%	30.1%	58.5%	2015 : Cu – 2017 : L.	z = - 2.336	0.4505
2017 : Lapwing	59.2%	31.5%	82.1%	2015 : Cu – 2015 : OC	z = 0.101	1.0000
2015 : Oystercatcher	21.5%	14.1%	31.5%	2015 : Cu – 2016 : OC	z = 0.566	1.0000
2016 : Oystercatcher	17.3%	8.3%	32.7%	2015 : Cu – 2017 : OC	z = 0.773	0.9998
2017 : Oystercatcher	16.8%	10.5%	25.8%	2016 : Cu – 2017 : Cu	z = 7.435	<0.0001
				2016 : Cu – 2015 : GP	z = 3.885	0.0058
				2016 : Cu – 2016 : GP	z = - 2.085	0.6336
				2016 : Cu – 2017: GP	z = 2.103	0.6209
				2016 : Cu – 2015 : L.	z = 2.165	0.5751
				2016 : Cu – 2016 : L.	z = - 0.518	1.0000
				2016 : Cu – 2017 : L.	z = - 1.295	0.9800
				2016 : Cu – 2015: OC	z = 2.805	0.1778
				2016 : Cu – 2016 : OC	z = 2.509	0.3342
				2016 : Cu – 2017 : OC	z = 3.636	**0.0146**
				2017 : Cu – 2015 : GP	z = - 0.237	1.0000
				2017 : Cu – 2016 : GP	z = - 7.042	**<0.0001**
				2017 : Cu – 2017 : GP	z = - 1.485	0.9450
				2017 : Cu – 2015 : L.	z = - 2.966	0.1186
				2017 : Cu – 2016 : L.	z = - 4.401	0.0007
				2017 : Cu – 2017 : L.	z = - 3.463	**0.0267**
				2017 : Cu – 2015 : OC	z = - 1.497	0.9417
				2017 : Cu – 2016 : OC	z = - 0.433	1.0000
				2017 : Cu – 2017 : OC	z = - 0.500	1.0000
				2015 : GP – 2016 : GP	z = - 6.170	**<0.0001**
				2015 : GP – 2017 : GP	z = - 1.196	0.9894
				2015 : GP – 2015 : L.	z = - 2.111	0.6151
				2015 : GP – 2016 : L.	z = - 3.670	**0.0129**
				2015 : GP – 2017 : L.	z = - 3.113	0.0792
				2015 : GP – 2015 : OC	z = - 1.021	0.9973

2015 : GP – 2016 : OC	z = - 0.246	1.0000
2015 : GP – 2017 : OC	z = - 0.230	1.0000
2016 : GP – 2017 : GP	z = 4.213	**0.0015**
2016 : GP – 2015 : L.	z = 4.031	**0.0032**
2016 : GP – 2016 : L.	z = 1.247	0.9851
2016 : GP – 2017 : L.	z = - 0.218	1.0000
2016 : GP – 2015 : OC	z = 4.003	**0.0036**
2016 : GP – 2016 : OC	z = 3.453	**0.0276**
2016 : GP – 2017 : OC	z = 4.562	**0.0003**
2017 : GP – 2015 : L.	z = - 0.621	1.0000
2017 : GP – 2016 : L.	z = - 2.204	0.5469
2017 : GP – 2017 : L.	z = - 2.321	0.4613
2017 : GP – 2015 : OC	z = 0.121	1.0000
2017 : GP – 2016 : OC	z = 0.581	1.0000
2017 : GP – 2017 : OC	z = 0.790	0.9998
2015 : L. – 2016 : L.	z = - 2.195	0.5531
2015 : L. – 2017 : L.	z = - 2.194	0.5541
2015 : L. – 2015 : OC	z = 0.872	0.9994
2015 : L. – 2016 : OC	z = 1.168	0.9912
2015 : L. – 2017 : OC	z = 1.700	0.8688
2016 : L. – 2017 : L.	z = - 0.927	0.9989
2016 : L. – 2015 : OC	z = 2.567	0.2984
2016 : L. – 2016 : OC	z = 2.503	0.3375
2016 : L. – 2017 : OC	z = 3.194	0.0625
2017 : L. – 2015 : OC	z = 2.671	0.2409
2017 : L. – 2016 : OC	z = 2.749	0.2025
2017 : L. – 2017 : OC	z = 3.355	**0.0380**
2015 : OC – 2016 : OC	z = 0.557	1.0000
2015 : OC – 2017 : OC	z = 0.880	0.9993
2016 : OC – 2017 : OC	z = 0.072	1.0000

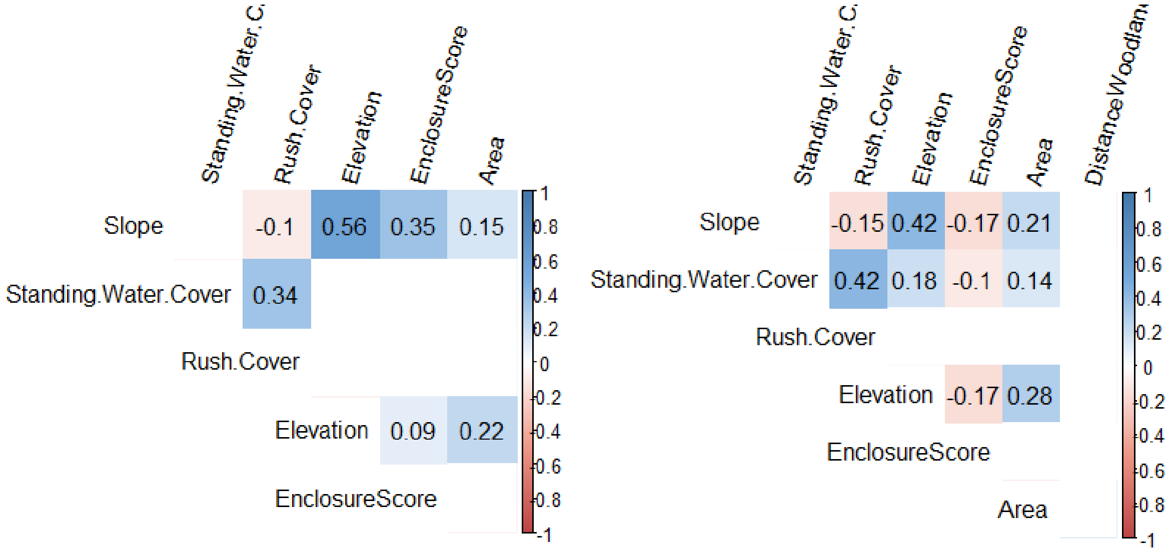
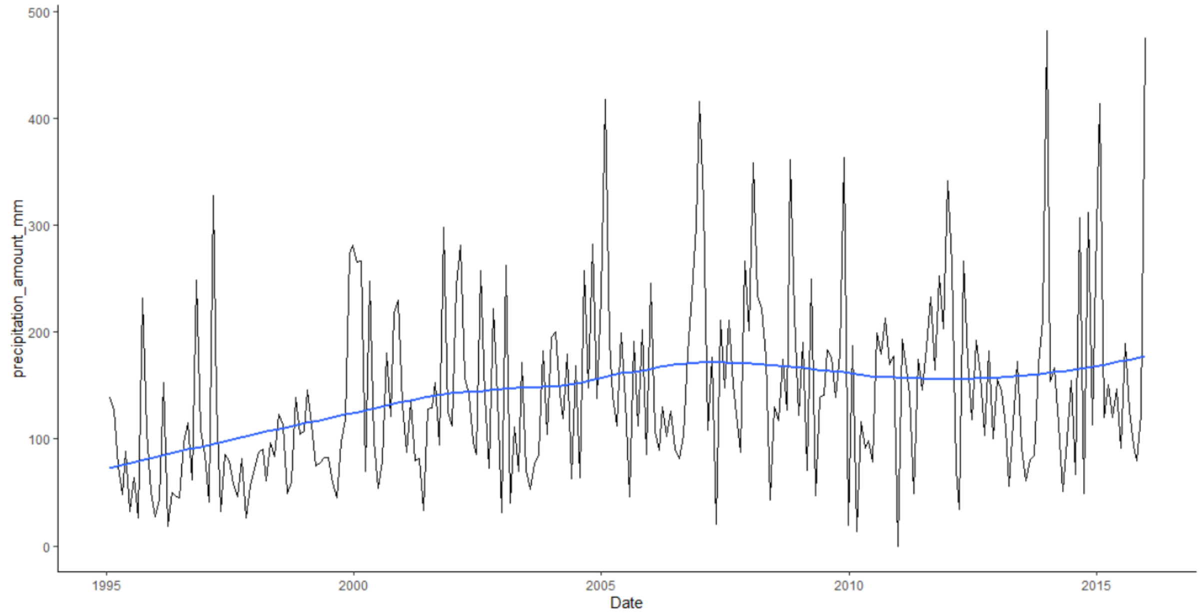


Figure A9. Correlogram representing the Pearson's correlations for all pairs of numerical variables used in analysis of chapter 5 where correlogram to the left is from field level data across all survey years and correlogram on the right is from field level data exploring effect of distance to woodland. The colour legend on the right hand side of the correlogram shows the correlation coefficients and the corresponding colours with positive correlations displayed in blue and negative correlations in red. A white box in the correlogram indicates that the correlation is not significantly different from 0 at the specified significance level (<0.05) for the couple of variables and is based on a correlation test.

Table A0-10. Summary table of Cramer's V score for pairs of categorical explanatory variables to test the strength of the association identified in the Chi-square test of independence. It is based on Pearson's chi-squared statistic and gives a value between 0 and 1. Values <0.25 suggest that the strength of the association in the two categorical predictor variables is weak, and therefore should not be an issue in modelling. Values above this but below 0.6 are moderate and of concern. Values > 0.6 suggest a strong effect of the association where the two categories are highly dependent and essentially capturing the same thing.

	Land Use	*Vegetation Height*	*Livestock Grazing*	*Livestock Type*	*Year*
Land Use	-	0.40	0.26	0.17	0.11
Vegetation Height	0.40	-	0.25	0.14	0.26
Livestock Type	0.17	0.14	1.0	-	0.11
Livestock Grazing	0.26	0.25	-	1.0	0.12
Year	0.11	0.26	0.12	0.11	-



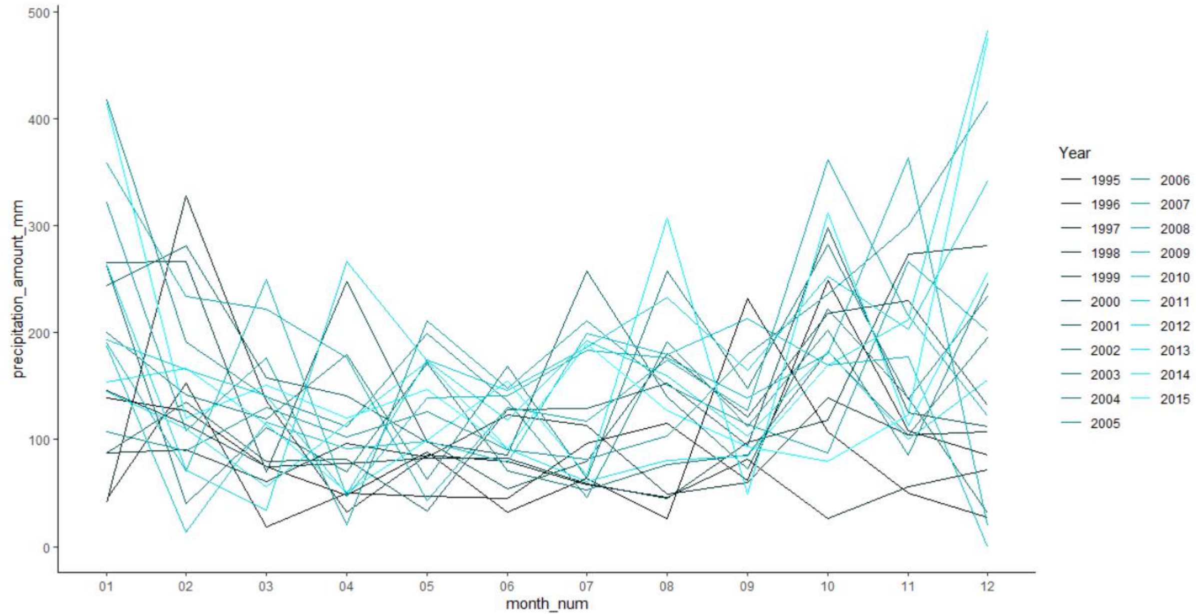


Figure A10. Daily rainfall (precipitation) in mm from Aviemore weather station between 1995 and 2015. Top figure shows a time series plot of day against precipitation with a loess smoother shown by a blue line. The bottom figure shows seasonal trend in the rainfall data with precipitation plotted by month and year shown by a black – blue colour palette.

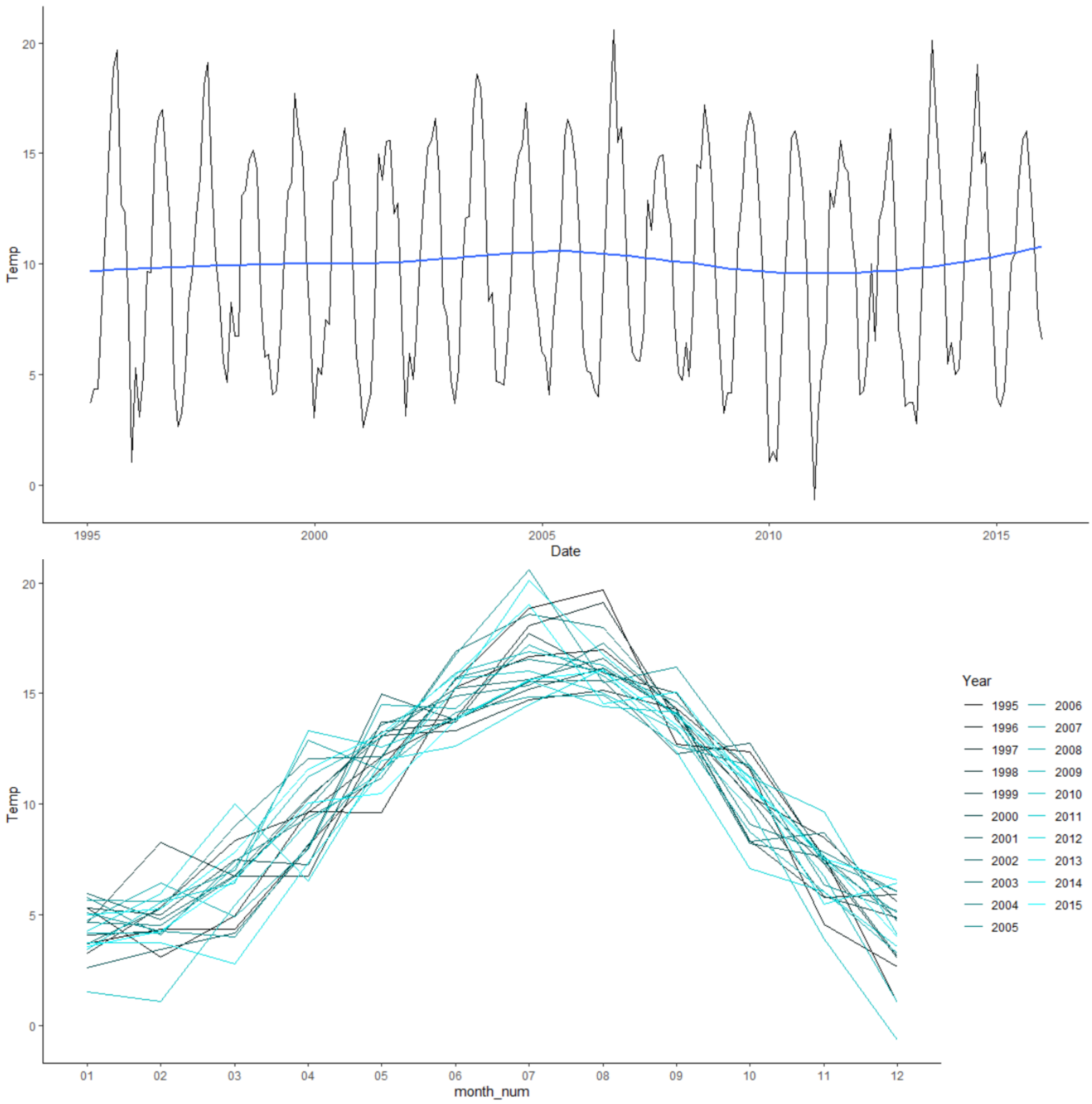


Figure A11. Daily maximum air temperature (°C) from Aviemore weather station between 1995 and 2015. Top figure shows a time series plot of day against temperature with a loess smoother shown by a blue line. The bottom figure shows seasonal trend in the data with temperature plotted by month and year shown by a black – blue colour palette.