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HARPER ADAMS UNIVERSITY

Conservation at the crop edge.

A long-term study of conservation headlands without fertiliser

Natural England 1

Submitted by Edward T Baxter to Harper Adams University

as a thesis for the degree of Doctor of Philosophy. July 2020.

I certify that all material in this thesis which is not my own work has been identified and that no material has previously been submitted and approved for the award of a degree by this or any other University.

E. B .

Signature………………………………………… ……………………………………………….

Abstract

The 2020 Agriculture Bill, which will shape government support for agriculture in the UK for the foreseeable future, has a focus on wider issues beyond direct support for agricultural production.The public has to a large extent identified habitat destruction, increased use of agrochemicals, and landscape-wide structural simplification as unwanted (Tscharntke *et al.,* 2012). The need to address these concerns is reflected in the Bill with an emphasis, although no longer stated on the face of the Bill, of public goods for public money.

Exactly how much farmland under positive environmental management is necessary to achieve the desired effect is a vexed question. Smith *et al*. (2020) considered the amount of uncropped land that should be allocated on farms for “biodiversity”, with figures from a range of studies; from Aebischer and Ewald (2004) advocating 6% for Grey partridge (*Perdix perdix* – afterwards referred to in this thesis as “partridge”), up to 10% for farmland birds in general (Henderson *et al*., 2012). Smith *et al*. (2020) themselves suggested that the value of Ecological Focus areas (EFAs) and similar agri-environment measures could be greatly enhanced by managing desirable arable weeds within crops to achieve a 10% covering.

This thesis looked at managing desirable arable weeds in the cropped area at the edge of cereal fields. The overarching research question was “Do wild headlands offer a viable option for arable farmers aiming to integrate biodiversity with production?”

The thesis begins with an oversight of current discussion surrounding farmland intensification and considered the solutions introduced under Agri-Environment Schemes (AES) and the rationale behind them conservation headlands and their derivative, wild headlands, are discussed in some detail. It includes a detailed description of wild headlands, their origins and their practice, as well as a general introduction to the study site in Chapter 2.

The thesis then looked at answering three “sub questions” of the overarching research question. The first, “What impact do wild headlands have on above ground biodiversity in the crop edge?” was considered in Chapter 3, which looked at a population over the long term of partridges, an indicator species of ecosystem health on farmland. The population trend was examined and tested for any relationship between arable weeds in wild headlands, the host plants for phytophagous invertebrates needed by partridge chicks, and availability of those invertebrates. It was found that brood production

remained steady on farms with wild headlands but declined on farms without. Chick Survival Rate on farms without wild headlands was below the minimum 30% needed to maintain a partridge population and that this was reflected in the availability of host plants for invertebrates and the invertebrates themselves. The second question “What was the economic cost of implementing wild headlands?” was considered in Chapter 4, which compared crop yield and gross margin in headlands of 82 fields with and without wild headlands in four cereal crops over two years. Yields of cereal crops in wild headlands were about 60% of field yield, but with variation between crops. The savings in fertiliser and sprays on wild headlands made up for the shortfall in some crops in some years, and costs of wild headlands were strongly influenced by output prices and fertiliser costs for individual crops. The third question, “How do wild headlands influence arable seedbanks?” was answered in Chapter 5 by examining seedbanks in 25 fields, some of which have had wild headlands intermittently for 20 years. It was found that after allowing for soil characteristics, wild headlands drove species assemblages with greater species richness, evenness and abundance in fields which had had wild headlands. Also found was that wild headlands could restore seedbank populations of weeds to levels seen in the 1970s, but that herbicides in intervening years had maintained seedbanks within reasonable limits. Finally, in Chapter 6 the conclusions of each chapter were brought together and the costs and environmental benefits of wild headlands and any implications for farming in general, opportunities for further research and future agricultural policy that arose from the study discussed.

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Introduction

This PhD has had a long gestation. Like many of the present author's generation who farm and shoot, Dick Potts, who became a good friend, has been an inspiration. We'd adopted his conservation headlands on my family farm in 1989 to integrate wildlife into our commercial farming and restore populations of partridge to levels last seen here in the 1960s. The changes we'd made to our farming, including the developments we introduced in the mid-1990s to try and solve problems found with conservation headlands, won recognition with a prize in the inaugural Purdey awards in 1999 and national press coverage for "the Gilston variant" of conservation headlands. Wild headlands (Matt Ridley's name for them) have now been part of the way we farm for almost 30 years.

In 2014, with the encouragement of Prof. Tony Trewavas and Dr Keith Dawson, who had been my agronomist since 1988, I approached Prof. Geoff Squire whose team at the James Hutton Institute had been sampling and visiting my farm for many years. It was my intention to examine wild headlands and their impact within a formal experimental structure and he kindly agreed to supervise a part-time PhD on the subject. He subsequently introduced me to Harper Adams University, where Dr Nicola Randall has been an unwavering support over the past six years.

In the course of the analysis of my seedbank data I approached Prof. Anne Magurran at St Andrews University for advice. Her insights into measuring Biological diversity are renowned and it is only with her invaluable support that I've been able to describe the impacts which were found. My debt to her, my wife Cath and my children, who have indulged my passion, is immense.

E.T.Baxter . Gilston July 2020

Declaration

Thesis chapters

The three data chapters within this thesis were originally written in the format of manuscripts for publication and were intended to stand alone. Consequently, information may be repeated across chapters although to avoid excessive repetition the general introduction to the study sites has subsequently been extracted from each chapter and can be found in the Methods chapter.

I (ETB) am the primary contributor to all chapters, but support was given by (in alphabetical order): Haley Arnold (HA), Jane Askey (JA), Dr Matt Bell (MB), Brian Birrell (BB), Dr Ian Brown (IB), Dr Ed Harris (EH), Dr Cathy Hawes (CH), Donald Hay (DH). Jèrèmy Lesiourd (JL), Prof Joah Madden (JM), Prof Anne Magurran (AM), Faye Moyes (FM), Dr Nicola Randall (NPR), Neil Robson (NR), Prof Geoff Squire (GRS) and Dr Andy Wilcox (AW). See below for detail of their involvement with each chapter.

Chapter Three: Wild headlands, partridges and invertebrates

ETB and CH devised and designed the study of emerged weeds and invertebrates. ETB added partridges to the study and collected weed data and invertebrate data, NR and BB supported collection of invertebrate data in 2014 and 2017 respectively. JL collected partridge data from 2014 – 2019. CH helped with weed ID, NPR and CH with invertebrate ID. AM gave broad statistical guidance on the models to use with the analyses of weeds and invertebrates. FM helped with R code and the preparation of heat maps. JM and MB advised on modelling the partridge population. JM, NPR, GRS and AW provided comments on the manuscript.

Chapter Four: Yields and gross margins at the crop edge.

ETB designed the study and collected the data. Support with the modelling in R was provided by HA, EH. CH contributed initial advice on modelling the data in Genstat. Comments on the manuscript were made by NPR, GRS and AW.

Chapter Five: Arable plant communities after 20 years of modified conservation headlands.

ETB and CH devised and designed the study. ETB, NR and IB collected soil samples which were processed in the greenhouse by ETB. JA and NR helped ETB water the seedlings and CH and DH contributed to weed ID. Model selection and statistical guidance was given by AM and CH. Practical support with R code was given by FM. Comments on the manuscript were made by AM, NPR and AW with GRS, CH and two anonymous reviewers commenting on an earlier version.

1 Background to wild headlands and literature synthesis

1.1 Context and scope of literature review

There is significant evidence that the intensification of agriculture has reduced populations of partridges (Potts, 1980; Potts, 1986; Potts, 2012; Warren *et al.,* 2017). Potts (2002), in his review of options for game and wild life for the Royal Agricultural Society of England, observed that the increase in wheat yields had been mirrored by a simultaneous decline in partridge populations. As the partridge depends on suitable nesting habitat and the ready availability of phytophagous invertebrates to feed its young, both reduced by farmland intensification, the partridge is a very useful indicator of farmland health. The mechanisms advocated to restore a supply of suitable invertebrates and partridge habitat within the arable landscape, for example, conservation headlands (Sotheron, 1991) and beetle banks (Thomas and Marshall, 1999; Thomas *et al*., 2002) can make a substantial contribution to the promotion of wider biodiversity.

In this review, the intensification of agriculture with particular relevance to the drivers of partridge decline is discussed and one of the measures, conservation headlands, developed in the mid-1980s to mitigate the impact of agricultural intensification reviewed. The review also considers the issues and shortcomings associated with conservation headlands. It was problems with the implementation of conservation headlands on a farm in East Scotland arising from these shortcomings which led to refinements of the original technique in the mid-1990s, a wild headland (Baxter, 2000), and subsequent use of wild headlands for the last 20 years.

It is the evaluation after this long period of the efficacy of wild headlands and the impact it has had on a partridge population, emerged weeds, invertebrates, implementation costs and soil seedbanks which is the subject of this thesis. The conclusions have implications for the development of AES within the UK post-Brexit and more generally in achieving sustainable intensification of industrialised cereal-growing agriculture.

1.2 Introduction to partridges

The partridge is a game bird with its origins in the Asian steppes. It lays the largest clutch of any species of bird, averaging 15 – 17 eggs, so is potentially highly prolific. Partridges nest on the ground concealed amongst dead grasses, chosen to match their plumage. Chicks hatch in mid-late June and at once the “brood” leaves the nest and spends its time concealed in cereals or other long grasses where they are looked after by both parents. The chicks, who feed themselves but are guided by their parents, rely on phytophagous invertebrates for feather growth, so their food supply has been indirectly reduced by the use of herbicides (Potts, 1986). Unlike other farmland birds where provisioning at the nest is carried out by the parents, the partridge covey (parents and brood) make their own way through crops in search of invertebrates, which makes the species particularly sensitive to changes in habitat.

1.3 Farmland intensification

The increase of output in global agriculture in the 20th Century, growing 2.85 times the amount of food on the same acreage in 2010 compared to 1960 (Ridley, 2020), has resulted in the loss of ecological heterogeneity at multiple spatial and temporal scales (Benton *et al*., 2003). In the UK the intensification of agriculture necessary for such growth in output post-war has been in agrochemical usage, cultivation practices, simplified rotations, inorganic fertiliser and homogenisation of cropping systems, of non-crop habitats and landscapes (Benton *et al*., 2003; Storkey and Neave, 2018). Forecasts for global population growth to 9.7 billion people at a time when agriculture is already occupying 40% of ice-free land (including grazing lands) raise questions over how further output can be achieved sustainably (Landis, 2017 and the references therein). Impacts of intensification are already widespread but sustainable intensification is an important component of farming policy (Sustainable intensification research platform. DEFRA anon). Sustainable intensification is an objective of those wishing to reconcile the environment with productive agriculture (see www.leafuk.org for examples).

1.3.1 Landscape and heterogeneity

Benton *et al*. (2003) wrote that reversing declines in farmland biodiversity would require enhancing heterogeneity of farmland from within individual fields to whole landscapes. Frison *et al*. (2011) argued that wider deployment of agricultural biodiversity was an essential component in the delivery of a sustainable food supply. Resilience of agricultural systems and their ability to recover from perturbations caused by disease, drought and climate change had been undermined by the focus on production traits at the expense of general agricultural biodiversity (Frison *et al*., 2011). However, estimation of the value of agricultural biodiversity *per se* is difficult. Jackson *et al*. (2007) posited that heterogeneous composition of ecosystems in agricultural landscapes provide insurance value that is not detected by the local-scale experiments that are typical of most agricultural research. This “localism” was highlighted by Hawes *et al*. (2010), who when comparing organic farms and farms practicing Integrated Farm Management (IFM) in their 100 field survey across Eastern Scotland, found despite species richness at the field level being highest on organic farms, IFM farms tended to have even higher species richness at farm and landscape scales due to greater variation of crop types and of cropping practices between fields. Re-introducing heterogeneity into the arable landscape on the Sussex downs was a key driver in the restoration of the partridge population on the Norfolk Estate (Potts, 2012). Landscape impacts of agricultural intensification affect a range of taxa, beside partridges. Dornelas *et al*. (2009) found creating environmental heterogeneity by varying management treatments across the landscape can be an effective way of promoting biodiversity and decreasing the abundance of problematic species. It was particularly relevant to modified landscapes such as agro-ecosystems, where intensive management created highly homogeneous landscapes which often led to loss of rare taxa and dominance by a few aggressive species (Dornelas *et al*., 2009). Approaching the issue from a slightly different perspective, Holland *et al*. (2012) suggested that more robust biological control may be expected in complex landscapes as a consequence of species complementarity and niche separation.

1.3.2 Homogenisation and impact on weeds and invertebrates

Homogenisation of habitat has selected for fewer dominant species with similar resource requirements to the crop, which has been correlated with the wider loss of cropping system resilience (Storkey and Neave, 2018). The dominance of a few, well adapted, species in arable systems has produced shifts in weed assemblages over time, with increasing numbers of grass weeds and fewer dicots in the seedbank of UK and European agricultural soils (Sutcliffe and Key, 2000; Storkey *et al*., 2012; Squire, 2017). This has negatively impacted the host plants for the invertebrates on which partridge depend. In an analysis of functional traits within arable weeds, Pinke and Gunton (2014) found that rare arable species of cereal fields tended to combine low nitrogen requirements, germination in late winter or early summer, and short flowering periods. The shift to winter cropping which selects against these species traits has caused a decline in species richness with weed assemblages dominated by a few ruderal species (Hawes *et al*., 2010). In parallel with the intensification of agriculture has been a decline in invertebrate numbers (Benton *et al*., 2002), although not at a constant rate and not the same at all spatial scales (Bell *et al*., 2020). van Klink *et al*. (2020), in their global meta-analysis of invertebrate studies, found terrestrial invertebrates had declined by ~9% per decade with variation over different time periods. Cole *et al*. (2017) earlier maintained that agricultural intensification and associated loss of high-quality habitats were key drivers of insect pollinator declines (Cole *et al*., 2017). In a study linking hirundine populations to invertebrate counts on car windscreens, Bowler *et al*. (2019) found that at the time that insect declines were being reported in many European countries, insectivorous bird populations were declining at both a European scale and at a national scale in Denmark. They posed the question whether bird declines were related to changes in insect populations brought about by change in agriculture, particularly in grassland systems (Bowler *et al*., 2019).

1.4 FIELD MARGINS

Marshall and Moonen (2002) observed at the time that in the UK the majority of semi-natural habitats in agricultural landscapes were in field margins, defined by them to include the permanent barrier (usually a hedge, wall or fence), the grass/habitat (managed or otherwise beside the crop) and the crop edge itself which may include a conservation headland. Field margins are an important contributor to landscape heterogeneity, which may a have a mediating effect on bird decline (Redlich *et al*., 2018). Fig 1.1 shows the components of a field margin updated from the original drawing to show an Ecological Focus Area (EFA) margin as a component of the field margin strip, with a wild headland instead of a conservation headland within the crop edge.

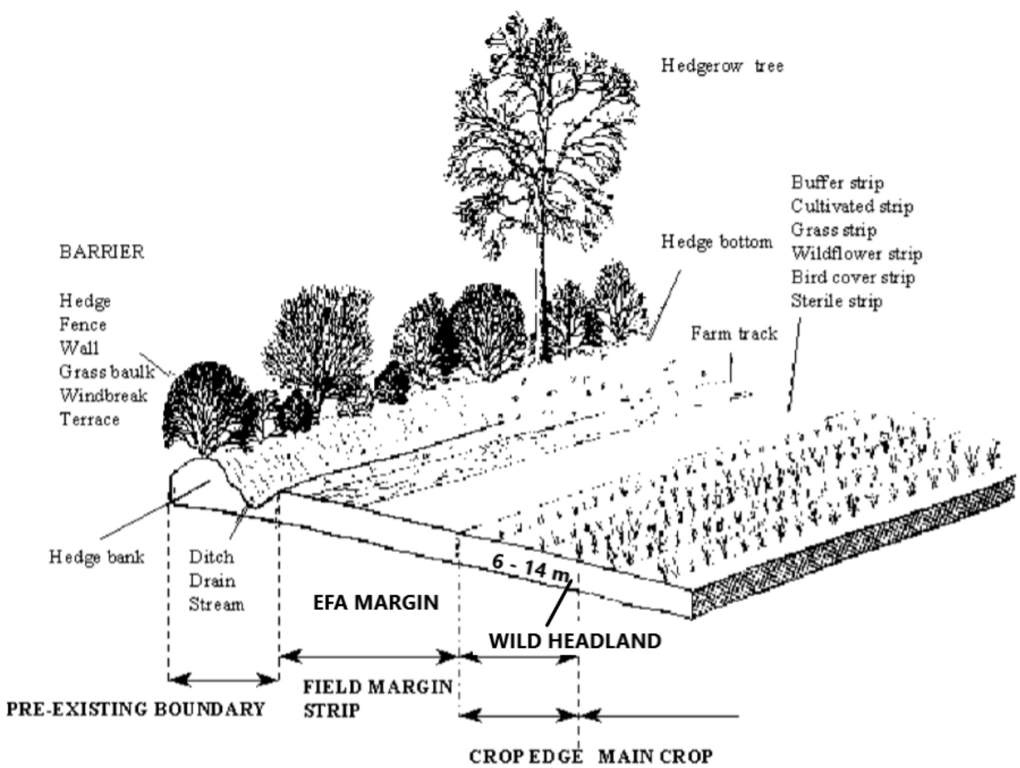

Fig. 1.1 The field margin with components adapted from the original in Marshall and Moonen (2002) to show the location of EFA margins and a wild headland within the crop edge. It is not drawn to scale as the width of individual components will vary. Note wild headlands are included in the wider field margin, not the field margin strip.

Field margins have long been the focus of AES (Winspear *et al*., 2010). Sympathetically managed field margins can provide a range of plant and invertebrate food resources for birds both in summer and winter (Vickery *et al*., 2002). Buffer strips (usually grass margins) can also be used to reduce surface movement of water and entrained sediment into watercourses and prevent some leaching of nutrients and agrochemicals (Marshall and Moonen, 2002) and as habitat for carabids (Woodcock *et al*., 2012). Because of the relative simplicity and low establishment costs of grass margins under AES, their use has been widespread (Westbury *et al*., 2017), with an impact across a range of taxa. Holland *et al*. (2012) found conclusive evidence of impact of two predatory guilds, epigeal and aerial natural enemies, on levels of cereal aphid control in winter wheat in farms with contrasting proportions of grass margins in the vicinity. Brickle (2000) recorded that Corn bunting (*Emberiza calandra*) provisioning nestlings foraged in grassy margins more than any other habitat relative to their availability within the maximum foraging range. Leaving a grassy margin at the foot of hedges benefited Dunnock (*Prunella modularis*), Whitethroat (*Sylvia communis)* and Yellowhammer (*Emberiza citronella*) as foraging and nesting sites (Moreby and Stoate, 2001). Westbury *et al*. (2017) however, were concerned that resources (habitat and food) wouldn't be available to foraging birds because of structural homogeneity in sown grass margins and recommended remedial work through scarification. Vickery *et al*. (2002) concluded their review of habitat provision for farmland birds to say "In general, the best winter food supplies (mainly seeds) will be provided by game cover crops. The most abundant summer food supplies (invertebrates and seeds) will be provided by a diverse sward; grass/wildflower strips, uncropped wildlife strips and naturally regenerated rotational strips followed by conservation headlands."

1.5 Conservation headlands – a key component of the field margin.

1.5.1 Introduction

Conservation headlands were pioneered by the Game Conservancy Trust (now Game and Wildlife Conservation Trust - GWCT) 30 years ago to enhance the availability of resources for farmland birds. They are selectively-sprayed headlands *within* cereal crops where pesticide applications are modified to maintain a population of broadleaved weeds as host plants for phytophagous chick-food invertebrates (Fig 1.1).The outer boom section of the sprayer (usually 6 - 7 m) is switched off by the operator when broadleaved herbicides are being applied to the rest of the field and insecticides are not applied after 15th March (Sotherton, 1991). The concept was developed progressively in the 1980s (*cf*. Rands, 1985; Boatman *et al*., 1999) as the technique depended on the *management* of the outer 6 - 14 m of the crop. New pesticides were screened for their effect on insect fauna and novel uses and combinations of those already available quantified (Dover, 1991). Solutions were proposed to agronomic problems as they occurred (Boatman *et al*., 1999) and some of these are discussed below.

1.5.2 History of conservation headlands

Populations of partridges, a nidifugous ground nesting bird closely associated with cereal crops, declined in the United Kingdom by 80% from the 1940s to 1980 (Sotherton *et al*., 1989) and work in the 1970s had identified that a scarcity of the invertebrates in the cereal ecosystem that made up the diet of partridge chicks was the probable cause (Potts, 1980; Potts, 1986; Southwood and Cross, 2002). Vickerman (1974) demonstrated that spraying off *Poa trivialis* in plots reduced the availability of chick-food items to partridge chicks. Potts (1986) concluded that the observed declines in partridge numbers were caused by herbicides used to remove the host plants of phytophagous chick-food invertebrates, many of which were broadleaved plants, and not the direct action of insecticides. In Germany in the 1970s herbicide use in the edge of cereal crops had been modified to protect rare arable weeds (so called herbizidfrei Ackerrandstreifen – herbicide free crop headlands) (Schumacher, 1980; Schumacher, 1987). The GWCT adapted the technique and named it a conservation headland (Oliver-Bellasis and Sotherton, 1986).

1.5.3 Conservation headlands and farmland birds

1.5.3.1 Effect on partridges

Between 1983 and 1986 a field experiment was carried out on Manydown Estate, Basingstoke, Hants, to test the effect of conservation headlands on partridge chick production. Across 3 separate beats over 2 years brood sizes in areas with unsprayed headlands were almost double the sprayed areas and differences in partridge chick food prey items were also statistically significant (Rands, 1985). The experiment was repeated in Sweden on 10 paired farms with larger broods found on the experimental farms (Chiverton, 1993).

1.5.3.2 Effect on other farmland birds

Between 1995 and 1997 a study of Corn buntings on the South Downs showed that abundance of chick-food invertebrates close to the nest was positively correlated with the weight of nestlings. Corn bunting foraged more frequently on un-sprayed cereal margins compared to margins sprayed with herbicide (Brickle *et al*., 2000). Vickery *et al*. (2002) identified an important role for conservation headlands in resource provisioning for a wide range of seed and invertebrate feeders, either directly through resources in the conservation headland (before and after harvest) or their role in shielding valuable nectar and berry resources in adjacent field margins from herbicide drift. Wood mice, a prey item for owls and kestrels, were more common in conservation headlands. Radio-tracked wood mice selected conservation headlands over other parts of the field and were therefore adjacent to hedgerows and fields margins where avian predators hunt (Tew *et al*., 1992).

1.5.4 Conservation headlands and arable plants

The inclusion of 20 arable plants in the UK Biodiversity Action Plan is a recognition that arable weed populations are vulnerable to modern farming methods (Storkey and Westbury, 2007). The edges and corners of arable fields tend to have the greatest botanical diversity (Wilson and Aebischer 1995; Marshall, 1989) so the use of conservation headlands, with the original aim of increasing populations of dicot weeds to increase the abundance of insects, has also turned out to be crucial to the conservation of many plant species (Potts, 2012). Chiverton and Sotherton (1991) comparing sprayed and unsprayed plots in a Spring wheat field on Manydown Estate, Hants, found statistically significant differences in species number between unsprayed headlands and sprayed headlands. In sprayed plots *Poa annua* made up 93% of weed plant numbers and 89% of weed biomass compared to 61% and 31% on unsprayed plots, while there was significantly greater percentage of weed cover on the unsprayed. *Matricaria* spp, *Polygonum aviculare, Veronica* spp and *Stellaria media* were all more abundant in the unsprayed plots (Chiverton and Sotherton, 1991). In a study over two years (1986 and 1987) in south-east Scotland similar patterns emerged (Fisher *et al*., 1988). In a study on four farms in Hampshire over two years the numbers of seedlings of 13 species in a spring barley crop (and 11 species in the seedbank) decreased significantly as distance from the crop edge increased (Wilson and Aebischer, 1995). In the autumn survey of winter wheat carried out as part of the same study, the number of seedlings of 15 species decreased in relation to distance from the crop edge, including species known to be adapted to arable conditions and to form persistent seedbanks in soil. Farming operations are less intense at the crop edge and crop yields are lower (Wilcox *et al*., 2000). It may be that as a result, opportunities are created for species present in the seedbank to flower and seed in proximity to the field edge. Uncommon species found in the surveys were all within 4m of the crop edge and sympathetic management of the field margin is required if these populations are to survive (Wilson and Aebischer, 1995). Additionally, some studies indicate that the use of conservation headlands can enable rapid restoration of the pre-herbicide era flora. For example, an examination of the long-term changes in the flora of the cereal ecosystem on the Norfolk Estate on the Sussex Downs, where no-fertiliser conservation headlands are in place, found 92 dicotyledonous species with no significant overall change in occurrence between 1968 and 2005 (Potts *et al*., 2010).

1.5.5 Conservation headlands and invertebrates

The favoured invertebrates of partridge chicks: Heteroptera, Coleoptera [particularly *Curculionodae, Chrysomelidae, Carabidae*], Lepidoptera and Hymenoptera [esp. *Tenthredinidae*] are often found in conservation headlands (Hughes *et al*., 1999). In a meta-analysis of 23 studies on the effect of reduced pesticide input (exclusion of herbicides or both herbicides and pesticides) in arable field edges, chick-food insects showed the largest response with an increase in abundance of almost three times compared to contemporaneous controls. (Frampton and Dorne, 2007). Chiverton and Sotherton (1991) in a within-field experiment in a Spring wheat field found that unsprayed headland plots supported significantly higher densities of non-target arthropods compared to sprayed plots, especially the non-pest species which are important in the diet of insect-eating gamebird chicks. These plots also contained higher densities of predatory arthropod groups, especially the polyphagous species and their alternative prey (Chiverton and Sotherton, 1991). In a two-year study in Montana USA in 1998 and 1999, Taylor *et al*. (2006) demonstrated that weedy plots supported significantly more chick food insects and beneficial arthropods than monoculture plots. Hassall *et al*. (1992) in a comparison of two types of headland management in the Brecklands ESA, showed that Carabids and Heteroptera were significantly more abundant in conservation headlands than in sprayed headlands, but less so than uncropped headlands. Observations of the pierid species *P.brassicae, P.napi and P.rapae* butterfly species by Dover (1997) showed that their behavior in cereal field margins was strongly influenced by the management of the cropped headland and that this was because they were detecting and exploiting the resources present in the conservation headlands not present in fully sprayed ones.

1.5.6 Conservation headlands within the farming system

1.5.6.1 Conservation headlands per GWCT

The extensive work carried out by the GWCT in the 1980s on conservation headlands was centered on trying to maximize yield whilst retaining benefits to wildlife (Potts, 2012). However, later studies highlighted potential problems with extended use. Hughes *et al*. (1999) demonstrated a build-up of *Poa annua* to critical yield thresholds where conservation headlands were retained continuously on the same plots for three years. Preliminary results by Chiverton (1993) showed no significant differences in yields of spring sown cereals between sprayed and unsprayed headlands, but significant yield reductions in winter wheat. An experiment testing conservation headlands in wheat, potatoes and sugar beet on fertile marine clays discontinued in sugar beet following excessive weed growth and major harvest losses (30%) (de Snoo, 1997). Wilcox *et al*. (2000) in a three-year field experiment on two sites found conservation headland management in winter cereals resulted in lower yield but only on one site in the 3rd year. In that study factors other than weed infestation affected yield response across the headland (Wilcox *et al*., 2000). On the Allerton research farm at Loddington however, using amidosulfuron to control *Gallium aparine,* Boatman *et al*. (1999) demonstrated that conservation headlands are a viable management option on heavy soils without serious crop loss. Costs were moderate, but did include two additional passes with the sprayer (Boatman *et al*., 1999).

Weed problems of fertilised conservation headlands can extend beyond harvest losses. If fertiliser is used in conservation headlands the resulting thick growth of weeds can reduce survival of chicks in wet weather, which negates their value to partridge chicks. Attempts to control thick weed growth through herbicide use can have other consequences beside the additional chemical and operational cost. On the Norfolk Estate for example Potts (2012) described how amidosulfuron used to control *Galium aperine* caused reductions of 61% in the broadleaved weed index, 68% decline in the number of broadleaved weeds, 50% reduction in the abundance of the main partridge chick foods and a 10 - 20% points reduction in chick survival rates. As a result, the chemical is no longer used on conservation headlands on the Norfolk Estate (Potts, 2012).

1.5.6.2 Conservation headlands without fertiliser

Grundy *et al*. (1996) tested reduced fertiliser rates as part of the suite of experiments on Manydown Estate, Hants, leading to the development of conservation headlands, but were discounted. The drop in yield at a time before area-based subsidy contributed to farm profitability dissuaded the GWCT from recommending the option (Sotherton, 1991). There were implications to the full fertiliser approach advocated by the GWCT. Mahn (1988) observed:

- Increasing doses of nitrogen caused simultaneous changes: a general decline in the number of individuals because of increased competitive ability of the crop.
- An increase in weed biomass, especially those species and individuals that survived the initial phase of severe competition with the crop.

This observation of Mahn (1988) had further implications for weed assemblages. Storkey *et al*. (2010) in an analysis of survey data from the Broadbalk long-term experiment, found that as N inputs increased, the abundance of the two functional groups that contained only common species remained stable or increased, whereas the groups dominated by rare or threatened species declined. Light appears to be the important factor. Pysek and Leps (1991) and Goldberg *et al*. (1990) reported that nitrogen fertilisers have a significant effect on the composition of the weed community through reduction of light levels. Klein and van der Voort (1997) found light penetration in conservation headlands was directly related to weed performance limiting species richness and plant growth of both the total weed vegetation and individual species. Light penetration too proved to be the most important correlate of plant growth for 5 rare weeds grown in the greenhouse and transplanted to a Spring barley field in 1994 (Klein and van der Voort, 1997), while in AES fields in Germany without herbicide or fertiliser increased light transmissivity explained species richness and community composition (Seifert *et al*., 2014)

Walker *et al*. (2007) surveyed options under AES in England in 2005 for the schemes' impact on the conservation of arable plants from 39 randomly selected 20 km^2. Fertilised conservation headland margins were the least diverse option and similar to crop controls. No-fertiliser conservation headlands were the most diverse cropped option with almost twice as many dicotyledons and 3 times as many rare species compared to the fertilised conservation headland sites (Walker *et al*., 2007).

1.5.7 Conservation headlands - alternatives

In recent work Wagner *et al*. (2017) manipulated seed and fertiliser rates in conservation headlands in a bid to overcome the effect of light exclusion in dense crops. They reduced seed rates in winter wheat and Spring barley by 75% from standard application rates and added rare arable weed seeds, but with inconclusive results. Costs were not reported in their study and given the likely cost and management input required it is unlikely to be a technique adopted at a farm scale.

Interreg PARTRIDGE (Interreg North Sea region, 2020) are funding a network of 10 demonstration sites across Northern Europe, based around developing permanent multi-functional habitat for partridges on farms. However, Bro *et al*. (2004) in a multi-site study on intensively cultivated ground in central France showed that permanent wildlife strips were not effective in increasing partridge numbers. The likely explanation is that the strips, mostly maize or kale-based mixtures, concentrated the surviving partridges and became a predator trap during the winter months (Bro *et al*., 2004). Funding for PARTRIDGE nevertheless is secure until 2023, which demonstrates the continuing interest across the EU in the fate of this iconic farmland bird.

1.5.8 Conservation headlands without fertiliser – rotated.

Incorporating conservation measures for the partridge within the farming system can have wider implications for conservation (Potts, 2012). A further refinement of the GWCT conservation headland technique, in addition to avoiding fertiliser, which may contribute to the sustainability of the technique, is to introduce rotation of the headland around the field in succeeding cereal crops (Baxter, 2000). Headlands in the same place give rise to an unsupportable weed burden (Hughes *et al*., 1999; Chiverton, 1993; de Snoo, 1997) which is avoided by herbicide use in intervening years as wild headlands (so named) are rotated around the field when in cereals. Wild headlands rotated annually each have the same aspect in any year so there is a two-fold advantage: First, their location will more easily be remembered by operators and second, there is likely to be increased resilience in the agro-ecosystem through the even distribution of headlands across the farm.

1.5.9 Conservation headlands and agri-environment policy

The biodiversity declines associated with increased post-war agricultural production have prompted the use of AES (Krebbs *et al*., 1999) in an attempt to mitigate some of the effects, notably for farmland birds. These included conservation headlands and cereal field margins were one of the first 14 key biodiversity habitats prioritised under the UK Biodiversity action plan (Vickery *et al*., 2009). Working on the Potts model of partridge population dynamics (Potts, 1980; Potts, 1986) and extrapolating the Sussex data, Aebischer and Ewald (2004) predicted that 6% of UK arable area allocated to insect-rich brood-rearing habitat would give a chick survival rate for partridge broods of 0.44 (defined in Potts, 1980), close to the levels Potts (1986) recorded in the pre-herbicide era and thus reverse the decline in partridge populations.

The evidence prompted governments in the UK, and elsewhere in Europe (Albrecht *et al*., 2016), to support conservation headlands but with the introduction of set-aside in 1992, initially as a production control measure, the GWCT attempted to replicate conservation headlands on set-aside (Sotherton, 1998). With the demise of set-aside in 2007, focus shifted back to encouraging support for conservation headlands under AES. There had been an option in Countryside Stewardship from 1996 (Potts, 2012) and subsequently in Arable Stewardship (later Environmental Stewardship) in England and Rural Stewardship (RSS) in Scotland. In assessments of the Arable Stewardship Scheme, Critchley *et al*. (2004) found dicotyledonous species widespread in conservation headlands, although species richness was over a third higher in no-fertiliser sites. Storkey and Westbury (2007) reviewing the management of arable weeds for biodiversity identified that in-crop solutions increased the weeds of biodiversity value when compared to a naturally regenerated margin. However, they went on to say that farmers preferred to establish vegetation on uncropped areas as they were "viscerally opposed to managing weeds in crops". It was probably for this reason that conservation headlands weren't widely adopted (Potts, 2012) and in the latest iteration of AES in England, Countryside Stewardship, there is only support under Mid-tier for "unharvested cereal headlands" rather than conservation headlands *per se*. As there is a requirement to establish them between February and April and leave them in place until February in the following year, the option will only apply to spring crops (Natural England, 2016). In Scotland support under Agri-Environment Climate Scheme (AECS) is for "Unharvested conservation headlands for wildlife" (in any cereal and oilseed crop), which again must be left in place until 1st March the following year. (Scottish government, 2016). It appears therefore that the era of direct UK government support for funded conservation headlands as such is at an end.

1.6 THE DEVELOPMENT AND INTEGRATION OF AGRI-ENVIRONMENT POLICY

The link between changes in agricultural management and farmland birds has been made for some time. For example, in parallel with changes in agricultural management from 1970 to 2000, Chamberlain *et al*. (2000) suggested that the associated intensification had been accompanied by population declines among farmland bird species. They analysed trends in agricultural management alongside changes in the farmland bird community and concluded that large shifts in agricultural management were a plausible explanation for the declines in farmland bird populations (Chamberlain *et al*., 2000). The trend has continued with declines in the index of common farmland birds in Europe from 52% of 1980 figures in 2000 to 43% in 2017 (EBCC, 2020). These declines have been a powerful motive for investment by governments into AES where payments are made to farmers in exchange for environmental goods and services such as biodiversity conservation (Vickery *et al*., 2004; Ansell *et al*., 2016). Substantial sums have been invested in pursuit of this goal, but AES need to be carefully designed and targeted to be effective (Batáry *et al*., 2015). There are however, many cases where policy has been effective, although seldom the subject of a cost benefit analysis (Ansell *et al*., 2016). Success came at many scales, from Cirl bunting (*Emberiza cirlus*) in south west England (Macdonald *et al*., 2012), to effects over larger scales. Geiger *et al*. (2010) found negative effects caused by agricultural intensity on the abundance and species richness of wintering farmland birds was mitigated when management practice was changed through provision of winter bird food under AES. Baker *et al*. (2012) found strong evidence for the positive effects of management that provided winter food resources under AES on population growth rates across multiple granivorous species, at three landscape scales. Winspear *et al*. (2010), in a review of farmland bird packages under AES, posited that farmland bird populations are likely to be increased if farmland bird measures are adopted on at least 7% of arable farmland, but that Higher Level Stewardship (HLS) could only fund this level of investment on tightly targeted areas. Intervention to promote biodiversity on farmland however, is based on individual choice. Ewald *et al*. (2010), reviewing the effectiveness and take up of measures designed to support partridge, observed that the most appropriate measures were rare outside farms where the farmers were especially motivated and that economic drivers usually determined the choices made. Jackson *et al*. (2007) observed that adoption of biodiversity-based practices for agriculture is only partially based on the provision of ecosystem goods and services, since individual farmers typically react to the private use value of biodiversity, not the 'external' benefits of conservation that accrue to wider society.

Elsewhere, Macdonald and Johnstone (2000) investigated farmer's motives for adopting positive environmental measures on their farms. While economic reasons were predominant in motivating farmers to remove hedgerows and other habitats in the 1970s, a large proportion of farmers then also professed positive attitudes to wildlife and stated that they would be willing to co-operate with schemes for habitat restoration if subsidies were available. In the 1990s subsidies did become available, and many of the 1990s respondents had made use of the various schemes recently in place to encourage habitat restoration and preservation (Macdonald and Johnstone, 2000).

1.7 Conservation headlands – moving forward.

Conservation headlands originated 40 years ago with work done by the GWCT and the literature is dominated by the research they did at that time. As conservation headlands became widespread and international so others picked up the thread and initiated further research. More recent studies reviewing the impact of conservation headlands (Walker *et al*., 2007; Storkey and Westbury, 2007; Potts, 2012) have highlighted the evolution to the GWCT technique which has occurred since the original research. The later work has explained the mechanisms which underpin the success of the no-fertiliser technique (Seifert *et al*., 2014). The evidence together suggests that fully fertilised conservation headlands have unfortunate drawbacks which limit their usefulness for partridges, as do the semi-permanent solutions advocated as their replacement.

This thesis tested the hypothesis that wild headlands, un-fertilised conservation headlands, rotated annually around cereal fields will limit the build-up in dominant weeds in the seedbank while fostering an abundant supply of phytophagous chick-food invertebrates for partridge chicks and other wildlife.

The evidence on partridge populations and arable weed seedbanks within this study has implications for arable farmers willing to integrate biodiversity and production. Given that is an objective of UK agricultural policy, incorporation of wild headlands into the flagship Environmental Land Managers Scheme (ELMS Defra, 2020) would seem a logical proposition.

2 METHODS

2.1 WILD HEADLANDS

2.1.1 Introduction

The subject of this study is wild headlands. A wild headland is the name given to a modification of the GWCT conservation headland developed in the 1980s. The conservation headland is an area, usually the width of a sprayer boom (~6m), where within the outer edge of cereal crops (the edge defined in Marshall and Moonen, 2002. See Fig 1.1) certain broadleaved herbicides are eschewed and insecticides are not used after the 15th March in any year (Sotherton, 1991). Modifications of the conservation headland took place in the mid-1990s. The first, introduced by Robert Cameron, a tractor driver working for Sandstone Farming Ltd, was that no nitrogen fertiliser was applied to conservation headlands. The second, that these fertiliser-free conservation headlands were then rotated annually around the headlands of fields when in cereal crops, was added later (Baxter, 2000). Each cereal field across the farm had a wild headland on the same side (north, south, east or west) in any year, so had a wild headland (defined in this study as an "intervention" and the term used throughout) a maximum of once every four years. As fields were not in cereals every year, but in rotation with other crops, the interval between wild headlands was often greater.

2.1.2 Detailed practice

2.1.2.1 Establishment and maintenance

In autumn, when winter cereals are sown, wild headlands are marked out on a plan. They are in the cropped edge of arable fields (as defined in Marshall and Moonen, 2002) and sown with the rest of the field at a uniform seed rate. The location of wild headlands is determined by the annual rotation of wild headlands around fields when in cereal crops. They are on the same (changing) side of cereal fields in any year so are distributed evenly across the landscape and their location easily remembered by operators. Wild headlands on the study farms are usually 7m wide, the boom section of a sprayer, but vary depending on sprayers.

Residual herbicides applied to winter crops in the autumn are not applied to wild headlands. When nitrogen fertiliser is applied in the spring, buffer management on the fertiliser spreaders is engaged to avoid distributing granular or liquid fertiliser onto the wild headland. The same protocol is used for spring cereal crops; seed rates are maintained across the fields, fertiliser is excluded from the wild headlands and no broadleaved herbicides are applied to wild headlands. Insecticides, if required, are not applied after the 15th March in any year to the outer tramline (28m) on the field side where there are wild headlands. Although fungicides and growth regulators are permitted in conservation headlands to keep crops standing and control disease, lack of nitrogen in the crops mean they are seldom required. The photographs in Fig 2.1 illustrate a wild headland and the crop development through the year showing the light green border in the winter barley where there was no nitrogen and *Matricaria* spp. flowering in summer.

Fig. 2.1 Photographs of a wild headland in winter barley on one of the study farms taken in April, May and July 2015. Note the pale colour of the barley without fertiliser (visible from April) and the *Matricaria* spp flowering in wild headlands without herbicide.

2.1.2.2 Harvest

Wild headlands are harvested with the remainder of the field and at the same time. The volume of crop and weeds in a wild headland is a small proportion of total field volume, so not usually kept separate. Combining wild headlands simultaneously with the field avoids operational complications and simplifies management at a busy time of the year. Fig 2.2 shows a combine with a 7m table combining a wild headland. Note the *Matricaria* spp flowering in the crop. This was a spring barley study field in 2014.

Fig. 2.2 A New Holland combine with a 7m table combining a wild headland in spring barley 2014. Note the flowering weeds in the cereal crop.

2.1.3 Introduction to the study site

The study included fields on five commercial farms covering 3,500 Ha in the East Neuk of Fife in the maritime farming area on the East coast of Scotland. (Latitude: 58º N, Longitude 2.50º W). Four of these were adjacent and have been farmed together under a single management, the fifth was contiguous. Their location is in Fig 2.3

Fig. 2.3 Location of the study farms (shown to scale) in East Fife. Dundee is the North, Edinburgh to the South. Farmworks (a crop recording package developed by Trimble Agriculture) 2020.

2.1.4 General background

The East Neuk of Fife is in the maritime farming area of the Central Scottish Lowlands. Although Scotland is well known for livestock production, the central lowlands of Scotland and the coast fringe contain arable soils suitable for a wide range of crops. Mixed farming, the integration of livestock and arable cropping used to be widespread, but increasingly livestock farming is concentrated in the north and west of Scotland, while eastern areas are predominantly arable.

Farms in the study area are as a consequence predominantly arable. The study farms are within the humid hemiboreal or fairly humid northern temperate Euoceanic subsectors defined by Birse (1971) in his assessment of climatic conditions in Scotland. The typical range in July temperatures at Leuchars weather station (12km) was from 10.5º C to 19º C, mean winter temperature was 4º C. Mean annual rainfall is from 650 – 700 mm. Fig 2.4 shows the bioclimatic sub-regions with location of the study farms highlighted.

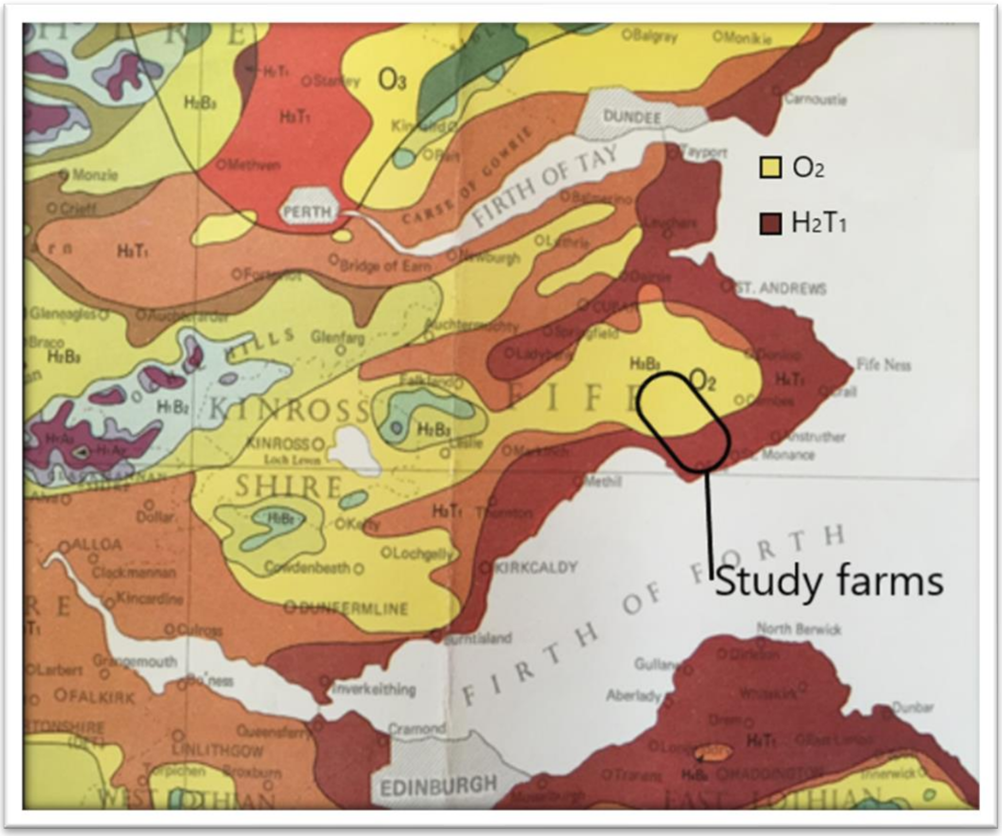

Fig. 2.4 Climatic Zones for Central Eastern Scotland showing the main cities (Edinburgh and Dundee) and the two bioclimatic sub regions covering the study farms: O_2 (humid hemiboreal) in buff and H_2T_1 (fairly humid northern temperate) in dark brown of the Euroceanic subsector. All other colour keys are available on request. (Birse, 1971). Approximate location of the study farms is outlined in black.

2.1.5 The soils

The study farms are either comprised of soils from the Dreghorn series (raised beach sands and gravels derived from carboniferous rocks with some old red sandstone material) on the lower, coastal areas, or from the Rowanhill/Giffnock/Winton series (drifts derived from Carboniferous sandstones, shales and limestones) inland (Walker *et al*.,1982). The Rowanhill/Giffnock/Winton soils underlying the majority of the Fife study farms are brown forest soils and normally have a profile comprising a dark grey-brown loam to silty clay loam topsoil or a grey or light brownish grey sandy clay loam. Texture is the main physical limitation affecting land use, although soils are capable of producing a moderate range of crops and on the coastal lowlands of Fife high yields are obtainable (Walker *et al*.,1982). Fig 2.5 shows the distribution of the soils across the study farms.

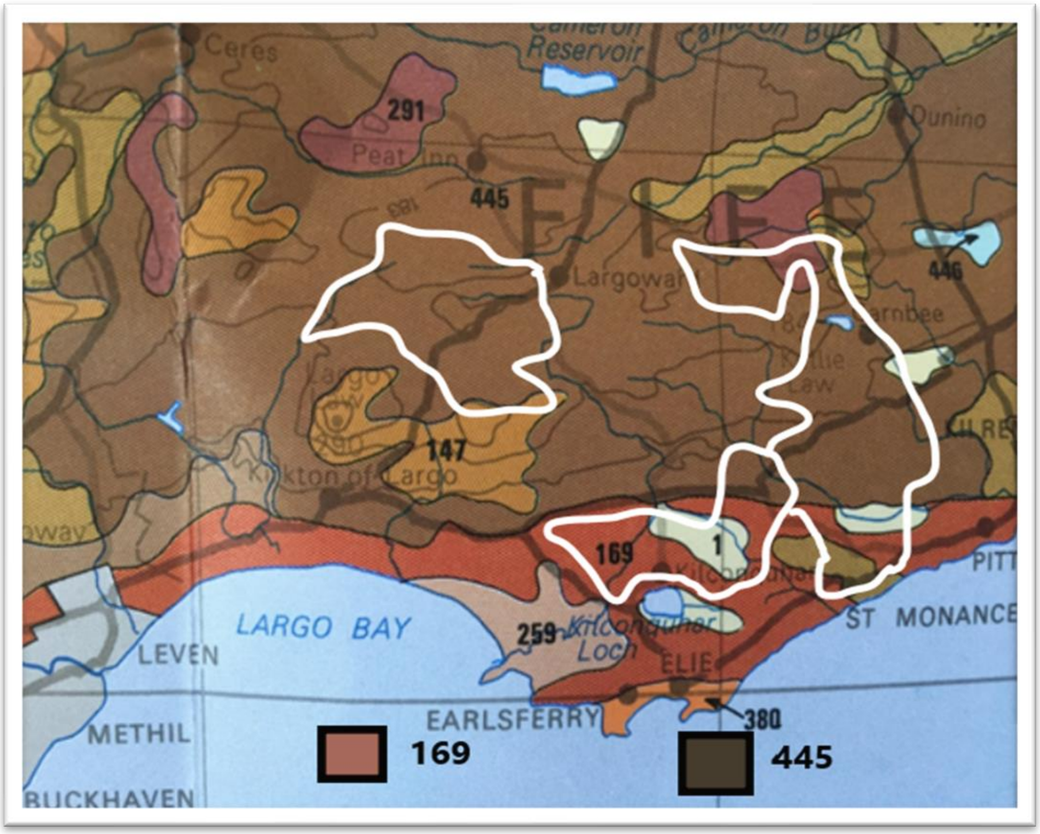

Fig. 2.5 Predominant soil types underlying the Fife study farms showing the division between Dreghorn (No.169 on the map), coloured reddish-brown and Winton/Giffnock/Rowanhill (No.445) coloured dark brown. Approximate outline of study farms in white. (Walker *et al*.,1982)

In the maps showing the Land capability for agriculture derived from the soil survey of Scotland (Walker *et al.,*1982) the study farms include 3 classes: class 2 is land capable of producing a wide range of crops [although there are limitations within this class these are always minor in their effects as land in this class is highly productive]; class 3.1 is land capable of producing consistently high yields of a narrow range of crops (principally cereals and grass); class 3.2 is land capable of average production but high yields of barley, oats and grass (Walker *et al.,*1982). Fields are from 4 – 16 ha, while permanent field boundaries across all farms are primarily hedges (*Crataegus monogyna* or *Fagus sylvatica)* bounded by 1 – 2m perennial grass strips, dominated by *Dactylis glomerate*. Field boundaries also include wire fences, stone walls and some mixed woodlands. It is an open landscape with few trees apart from amenity planting around Estate parks. Fig 2.6 shows the study farms with the relevant yield classes.

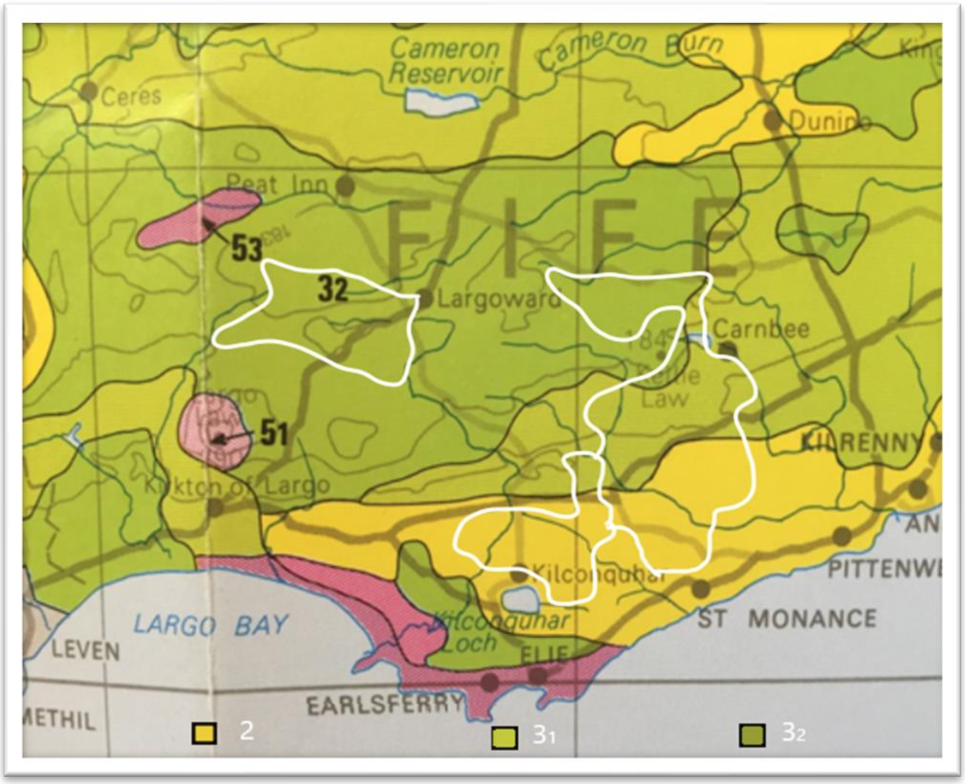

Fig. 2.6 Study farms in Fife showing their classification for Agriculture. Class 2 = yellow, class 3_1 = light green and Class 3_2 = dark green. Approximate outline of the study farms shown in white (Walker *et al*., 1982).

2.1.6 The farming system

The soils and climate, alongside subsidy and market opportunities, have determined cropping patterns on the study farms over the 19-year period of this study. The Sandstone Farming Co, on behalf of the principals of 4 of the farms, operated a mostly ploughed based cropping system with consistent agronomy across all farms. The farming system has been influenced by the principles of Integrated Farm Management espoused by Linking Environment and Farming (LEAF). Gilston Mains, one of the study farms, has been a LEAF demonstration farm since 2000 and Balcaskie, another of the study farms also operating a plough-based system, has been a member of LEAF since 2009. Information on LEAF and integrated Farm Management is available from LEAF (www.leafuk.org)

Crops are normally grown in sequence, one per year, and in separate fields. Crops are rotated annually with winter wheat normally following winter oilseed rape or winter oats. Spring oats, spring barley or a second winter wheat crop follow winter wheat. Winter barley often follows spring barley and almost invariably precedes winter oilseed rape. Arable crops cover most of the farmed area and where there is grass it is usually permanent pasture and not in rotation with arable crops. The division into winter sown and spring sown crops is designed to spread labour and machinery requirement throughout the year and the pattern is further influenced by soil type and marketing opportunities. Vegetable and root crops (broccoli, carrots and parsnips) are occasionally grown on the Dreghorn series soils and over the past 19 years potatoes have been grown in most arable fields. The principal crops grown in recent years with mean yields t ha^{-1} for the Fife farms is given in Table 2.1 (4 years' data)

Crop	2014	2015	2016	2017
Winter oats	8.25	8.39	8.33	8.02
Spring oats	7.83	7.69	7.61	7.05
Winter wheat	9.38	10.2	9.78	9.40
Winter barley	8.90	10.03	8.36	8.65
Spring barley	7.05	7.16	6.06	6.07
Winter oil seed rape	4.70	4.55	3.40	4.36

Table 2.1 Showing mean yields in tonnes ha^{-1} at 85% dry matter (91% in the case of oilseed rape) for 4 years of the principal crops on farms comprising the study sites in the East Neuk of Fife.

2.1.7 The study farms

The location of the study farms in relation to the sea and each other is shown in Fig 2.7. Farming at Gilston Mains (shown in green), has been conducted according to the principles of Integrated Farm Management (IFM) since 1989. Lathallan, (red), has been farmed by Sandstone Farming under a tenancy or as contractors since 1992, Kilconquhar, (yellow), has been under a share farming agreement from 1990 and Easter Pitcorthie, (blue) under contract from 2004. Four of the five farms therefore have had the same agronomy (supplied by Dr Keith Dawson) and management for much of the last 25 years, while Easter Pitcorthie has been managed similarly since 2004. Balcaskie, (grey), although under different ownership, management and agronomy has the same range of altitude, climatic sub zones and cropping as the four farms farmed by Sandstone Farming Ltd.

Gilston Mains and Lathallan have had a prolonged history of intervention through use of wild headlands; Kilconquhar and Pitcorthie to a lesser extent and Balcaskie not at all (until the onset of this study). All farms contributed fields to the study which were chosen to give a sufficient sample size and to represent the range of available soil types and intervention histories.

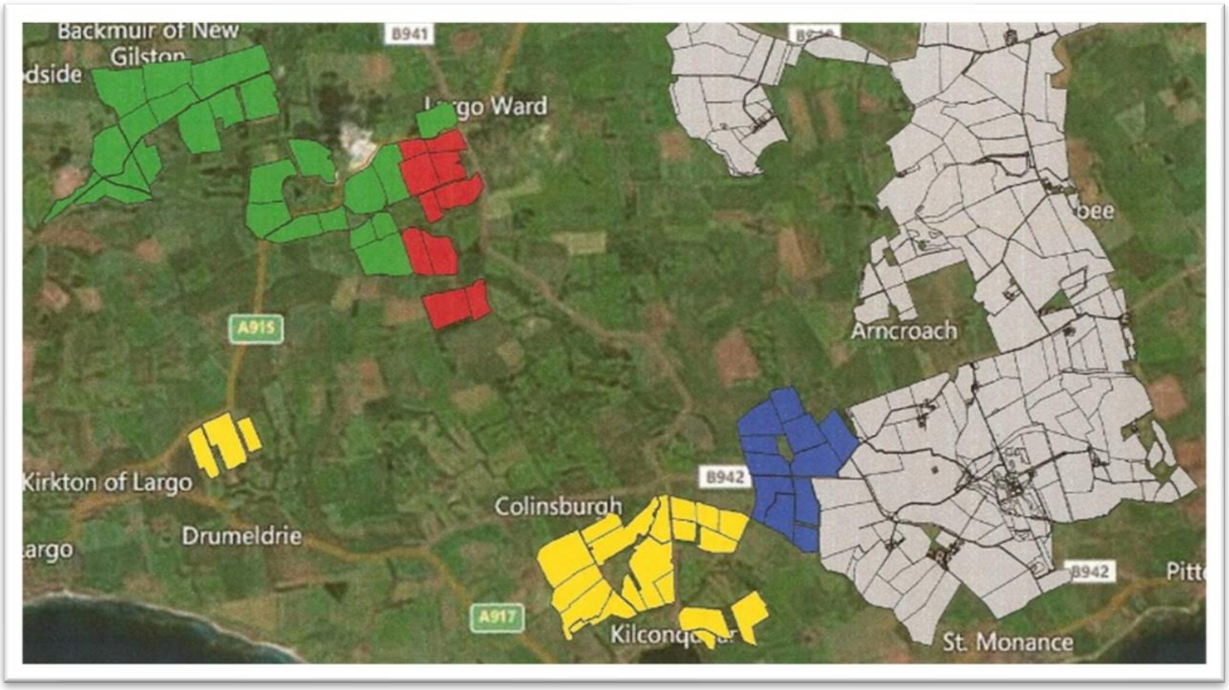

Fig. 2.7 Plan showing the Fife study farms super-imposed on a satellite image with villages named in grey type. The Firth of Forth is at the base of the image. Farms are coloured; Gilston = green, Lathallan = red, Easter Pitcorthie = blue, Kilconquhar = yellow and Balcaskie = grey (Farmworks 2014).

2.1.8 The study fields

Fields used in the different experiments were drawn from a subset of the fields on the 4 farms. Fig 2.8 shows all the fields used in the experiments. As is apparent from a comparison with Fig 2.7, the experiments included almost all fields. A detailed map of the fields used in the respective experiments is given in each chapter.

Fig. 2.8 Fields (pink) on the respective study farms used for experiments from 2014–2019. Individual subsets of fields used in the respective experiments are highlighted in each chapter. A north point and scale are given. Farmworks 2020

2.2 THE EXPERIMENTS

In the course of the study five experiments were carried out over four years; two connected to seedbank analysis over two years, a yield experiment looking at field and headland yield over two years, a field study of emerged weeds and invertebrate sampling of headlands over two years and a count of partridges over six years.

The timing of the experiments is given in the Gant chart Fig 2.9:

Experiment	2014	2015	2016	2017	2018	2019
Seed bank studies (Green house)						
Yield Experiment (Field)						
Within – Field Arable Weeds						
Invertebrate studies						
Partridge Counts						

Fig. 2.9 Gant chart showing the timing of the experiments in this study. Shaded area are years when experiments were conducted.

Detailed methodology for each experiment is given in appropriate chapters in this thesis. There are additional materials and further illustrations in appendices: 2.1, 2.2, 2.3 & 2.4

3 WILD HEADLANDS, PARTRIDGES AND INVERTEBRATES

Abstract

Conservation headlands, developed by the GWCT in the mid-1980s were a novel solution to the conflicts caused by the intensification of agriculture at the expense of other ecosystem services and farmland birds. However, the problems they caused farmers with weeds meant they were never widely adopted within the UK or continental Europe. In this study we tested how a modified version of conservation headlands (wild headlands) affected the reproductive output of a key indicator farmland bird, the partridge, over a 5-year period and explored mechanisms that might explain these effects via the emerged weeds and associated epigeal invertebrates. We found that mean chick survival rate (CSR) over 5 years was 37% on farms with wild headlands, while farms without had a CSR of 28%, below the minimum threshold of 30% required for the population to survive. Partridge brood size remained constant on farms with wild headlands over the period of the study but declined on farms without. An exp(*H*_Shannon) measure of emerged weeds and invertebrates on wild headlands showed significantly greater species richness and abundance compared to controls, while a hierarchical cluster dendrogram using Bray-Curtis dissimilarity indices showed distinctive weed and invertebrate assemblages in wild headlands. As the wild headlands occupied ~2% of the arable area of the farms that supported them, this research has implications for developing AES in Europe and for the promotion of sustainable intensification.

3.1 INTRODUCTION

Declines in populations of farmland birds which depend on phytophagous invertebrates to feed their young have been severe in recent decades with a 90% decline observed from 1970 – 2015 in Tree sparrow (*Passer montanus)*, a 56% decline in Yellowhammer, an 89% decline in Corn bunting and a 92% decline in Partridge (Holland *et al*., 2006; Perkins *et al*., 2011; Potts, 2012; Hayhow *et al*., 2017). Increased cereal yields in the UK post-war can be partly attributed to the efficient use of herbicides (Potts, 2002). These have reduced abundance of groups of common broadleaved weeds through dicotyledon-specific herbicides (Ewald and Aebischer, 2000), causing a change in weed species (Sutcliffe and Kay, 2000) and a shift in weed communities away from broadleaved in favour of grass weeds (Squire, 2017). Many of the invertebrate taxa which depend on broadleaved weeds are an important component of the diet of a wide range of bird species. Wilson *et al*. (1999) examined the abundance and diversity of invertebrate foods of 26 granivorous (farmland) bird species of northern Europe. In their diets they found spiders (Araneae) and beetles (Coleoptera), particularly ground beetles (Carabidae). Diets also included: weevils (Curculionidae); grasshoppers, crickets, bush crickets (Orthoptera); flies (Diptera), especially leatherjackets (Tipulidae); bugs (Hemiptera), primarily aphids (Aphididae); ants, bees, wasps and sawflies (Hymenoptera), particularly ants (Formicidae); and butterflies, moths and their larvae (Lepidoptera).

In order to enhance the availability of resources for these phytophagous chick-food invertebrates, the GWCT developed conservation headlands in the mid-1980s. Their philosophy was to reduce the impact of modern farming systems on a wild [but harvested] species in a manner acceptable to the farming community (Dover, 1991). Conservation headlands are an in-field measure based on the principle of land sharing (Fischer *et al*., 2014). They are selectively sprayed headlands within cereal crops where pesticide applications are modified to maintain a population of broadleaved weeds as host plants for phytophagous invertebrates. The concept was developed and applied progressively within the UK and is still in use today. (*cf*. Rands, 1985; Boatman *et al*., 1999; Ewald *et al*., 2012). The technique depends on the management of the outer 6 -10 m of the crop while maintaining fertiliser applications and crop yield. The outer boom section of the sprayer (usually 6 - 7 m) is switched off by the operator when certain broadleaved herbicides are being applied to the rest of the field and insecticides are not applied after 15th March in any year (Sotherton, 1991). Conservation headlands have been included in AES across Europe with funding to farmers based on the opportunity cost of forgone yield (Walker *et al*., 2007; Albrecht *et al*., 2016). This has provided conservation benefits by improving brood size through better chick survival in partridges (Rands, 1985; Chiverton, 1993). However, despite these clear benefits, conservation headlands have never been widely taken up in the UK (Clothier, 2013) or in Germany (Albrecht *et al*., 2016), attributed in part to farmers' dislike of the weeds which flourished in arable crops tended with full fertiliser and no herbicide (Storkey and Westbury, 2007).

To address the weed burden associated with conservation headlands, in the early 1990s an alternative approach, called a wild headland, was developed at Gilston in Eastern Scotland (Latitude: 58º N, Longitude 2.50º W). Firstly, nitrogen fertiliser was restricted, which limited the vigour of nitrophilous weeds and crop growth. Secondly, these nitrogen-free conservation headlands were rotated annually around fields when in cereal crops (Baxter, 2000). (For a full account and description of a wild headland, please refer to the general methods chapter.)

Later chapters in this thesis examine the positive and negative effects, including economic cost, of wild headlands. However, it is critical to demonstrate that wild headlands enhance biodiversity on farmland, specifically partridge populations. Partridges serve as a useful indicator species, with their productivity in terms of brood production and survival being intrinsically related to the supply of prolific invertebrate prey and nesting cover (Potts, 1986). A healthy farmland bird population infers an underlying healthy ecosystem, with a suitable quantity and diversity of invertebrates to provide chick food and a flora sufficient to support these invertebrates. The mechanism proposed was that the implementation of wild headlands permitted a greater abundance and diversity of emerged weeds [although inhibited through herbicide over the next three years], which in turn led to a greater abundance and diversity of associated epigeal invertebrates, specifically food items for partridge chicks. This led to larger broods and hence higher local productivity, supporting population increases. Partridge were used as an indicator species because measuring their productivity via counting birds in broods is feasible at an entire farm scale via well-developed partridge count methods (Ewald *et al*., 2010). Partridge diet is well understood (Southwood and Cross, 2002; Browne *et al*., 2005; Potts 2012) and partridges have already been the focus of research for conservation headlands (Rands, 1985; Chiverton, 1993), making judgements on the efficacy of wild headlands practical.

First, it was asked if broods from farms with wild headlands were larger than those from control farms and whether the differences in productivity were likely to have population consequences [i.e. were they at levels above those necessary for population growth]. It was then asked if these differences could be explained by differences in abundance and diversity of invertebrates in wild headlands, specifically focusing on taxa previously reported to constitute an important component of partridge chick food: e.g. Hymenoptera, Hemiptera/Aphidoidea, Homoptera, Coleoptera/carabidae and Collembola/Sminthuridae (Potts, 2012). Finally, it was asked whether these important invertebrates might be available because the underlying flora was sufficiently diverse and prolific to support them. Parasitoids, sap feeders, leaf chewers and their predators have a strong association with dicot weed species (Marshall *et al*., 2003; Hawes *et al*., 2009; Smith *et al*., 2020), so in turn it was tested to see if differences in invertebrates could be explained by the abundance and diversity of their host plants in wild headlands.

3.2 METHODS

3.2.1 Study sites

Autumn brood counts were conducted across a contiguous set of 136 fields, 81 fields (μ = 12.83 ha) with wild headlands when in cereals (from 2015) and 56 fields (μ = 10 ha) within a block which had not included wild headlands within the rotation (Fig 3.1). The 136 fields are under the same ownership, although under different management. Within the overall farm boundary there was identical legal predator control protecting nesting partridges, primarily from ground predators; foxes (*Vulpes vulpes*), stoats (*Mustela erminea*), weasels (*Mustela*) and rats (*Rattus norvegicus)* who predated them, their eggs and their young and from aerial predators; Crows (*Corvus corone*) and magpies (*Pica pica*) who predated their eggs. There was a supplementary feeding regime conducted across the farms with wheat supplied in small hoppers along hedgerows. Over 2014 and 2017 weeds were sampled and invertebrates from 18 fields, of which 8 were a subset of these fields, 3 with wild headlands and 5 without (Table 3.1). For logistical reasons (and so that all fields we were investigating were sown with the same crop, winter barley), some of these fields were located on adjacent farms not counted for partridges. Fields sampled were divided into Farm 1 with a long history of past wild headlands from 1995 to 2013, Farm 2 where wild headlands from 1995 – 2013 had been absent or less frequent and Farm 3 where partridges were counted and which, after an experimental year, adopted wild headlands in 2015. As well as sites with different wild headland histories, the sites covered different bio-climatic zones and different soil types. In 2014, 4 fields with wild headlands were sampled (all on Farm 1) and 5 without (2 on Farm 2, 3 on Farm 3). In 2017 4 fields with wild headlands were sampled (1 on Farm 1, 3 on Farm 3) and 4 without (2 on Farm 2, 2 on Farm 3).

Fig. 3.1 Satellite map showing the count area from 2014 - 2019. Fields (green) had wild headlands from 2015, fields (orange) did not. Winter barley study fields used in 2014 and 2017 marked in pink with the ~ border of the bioclimatic zone (which is also the approximate soil type boundary) marked in blue. Farms are numbered: Gilston & Lathallan = 1, Easter Pitcorthie = 2, Balcaskie = 3. Farmworks 2020

3.2.2 Autumn partridge count

The post-harvest stubble count of partridge was carried out each year from 2014 to 2019 on the green block (1040 ha) (Fig 3.1) and the green and orange blocks from 2015 – 2019 (1,600 ha). The counts were carried out by the resident keeper/wildlife ranger over a number of days each autumn following methodology developed by the GWCT and adopted for their partridge count scheme (Ewald *et al*., 2010). In surveys carried out for 2 hours after dawn and 2 hours before dusk [from a motorised vehicle acting as a temporary hide and using 10x magnification binoculars] location, sex of adults and number of young from individual coveys were recorded for all partridges seen when driving around stubble fields. The keeper was careful to avoid duplication of coveys which can usually be identified by brood size, age and location. Around 200 ha per day can be counted using this technique. Given the long experience of the keeper/wildlife ranger and his familiarity with the ground it is probable that all partridges in the area were counted.

3.2.3 Invertebrate and weed sampling and experimental design

To identify the impact of wild headlands on emerged weed flora and their associated invertebrate populations, nine headlands on four farms in fields growing winter barley were selected for sampling in June/July 2014 shown in Fig 3.1. The headlands were on the north side of fields. A further eight headlands of winter barley fields were sampled in June/July 2017 and per the practice of rotating wild headlands annually, in 2017 these were the west sides of fields. The sites chosen for the study were determined by where winter barley was being grown in those years. The distribution of fields by farm and treatment is given in Table 3.1 below, with numbers showing the distribution of fields in each category.

	2014			*2017*		
Farm	**Fields (n)**	**Wild Headlands**	**No Wild Headlands**	**Fields (n)**	**Wild Headlands**	**No Wild Headlands**
Farm 1	4	4	0	1	1	0
Farm 2	2	0	2	2	0	2
Farm 3	3	0	3	5	3	2
Total	9	4	5	8	4	4

Table 3.1. Winter barley fields sampled for emerged weeds and invertebrates on each farm in 2014 and 2017 with detail of treatment that year: wild headland (no herbicide and insecticide in the outer 7m) (WH) or conventional treatment (no WH) that year. The 8 fields on Farm 3 are a subset of the fields we counted for partridges.

3.2.4 Invertebrate sampling

Epigeal arthropods were sampled from headlands used for emerged weed sampling in July 2014 and July 2017 using a Vortis suction sampler (Arnold, 1994). This suction-sampling technique is comparable with the conventional D-vac suction sampler and has been used widely in similar entomological field studies (e.g. Moreby et al., 1997). Although extraction efficiency is always less than 100%, suction samples represent a constant proportion of the population density, thus allowing valid statistical comparisons to be made between treatments for the same habitat (Haughton *et al*., 2003). Samples consisted of 5 x 10 second Vortis 'sucks' taken 1.5 m apart at 3 m from the crop edge at each of the same six sampling points used for emerged weed flora sampling. The Vortis machine was placed carefully over the growing crop and suspended 1 - 2 cm above the ground surface for each 10 second "suck". This gave an area of 0.09 m^2 over which each set of bulked samples was taken. Samples were taken when both soil and vegetation were dry to the touch, and sampling was completed for each headland within 1 hour on each occasion.

Arthropod samples collected were placed in a labelled a Ziploc polythene bag and placed in a cool box containing frozen blocks during transit from the field, and thereafter stored in a freezer at -18º C. The frozen contents were placed on a plastic sample tray under a strong light and by careful examination arthropods separated from other organic matter and soil particles. A hand-held magnifying glass was used as an aid (Equipment used and an illustration of the invertebrates found is in appendix 2.2 - 2.4). Arthropod samples in 2014 were placed in labelled test tubes containing EtOH (Ethyl alcohol) at 72% prior to being counted and identified to the appropriate taxonomic level under a 45x zoom binocular microscope. Samples from 2017 were counted and identified without first being stored in alcohol.

Total counts of epigeal arthropods were analysed for the following taxonomic groups: Araneae (order and selected size); Collembola (Super-family and family); Hymenoptera (suborder and selected family); Coleoptera (order and selected family); Hemiptera (suborders Heteroptera and Homoptera [incl. super family Aphidoiderae]; Diptera (order and size); Thysamoptera (order); Dermaptera (order); Thyranura (order) and Trichoptera (order). Potts (2012) quoted the extent to which the above orders appeared in partridge diets between 1948 and 2011 based on gizzard and crop analysis. Aphidoiderae and Sminthuridae were the most frequent items found in partridge chick diets. Coleoptera (especially Chrysomelidae), Hymenoptera and Araneae were important. Thysamoptera, Collembola (excepting Sminthuridae – Lucerne fleas) and Diptera (excepting crane flies) seldom appeared in their diet. Thysampotera and most Collembola were likely unimportant as they were too small and Diptera as they were usually unavailable.

3.2.5 Emerged weed sampling

The James Hutton Institute protocol for Biodiversity Indicator Monitoring (Firbank *et al*., 2006; Perry *et al*., 2003 with later adaptations) was followed to sample emerged weeds in the headlands of winter barley fields. Sampling was conducted in mid-June/July to capture the effect of herbicide applications (which had taken place over winter in the control sites) when compared to the wild headlands that had received no herbicide. Of the eight fields sampled for invertebrates in 2017, three had later been sprayed with a pre-harvest application of glysophate making weed ID for these fields impractical. For the fields sampled in 2014, standardised methodology for evaluating weed flora (Hawes *et al*., 2010) was used as follows: six samples were taken in each headland 3 m into the crop and at 20 m intervals about the middle of either the north (2014) or west (2017) side of each field. Boundary width and boundary vegetation were noted but not analysed. A 50 cm^2 quadrat was placed at each sampling point, a visual evaluation of % weed cover was made and all emerged weeds counted and identified to species. As some weeds were still at the seedling stage and therefore difficult to identify they were amalgamated into 4 groups: *Poa spp,* grasses other than *Poa spp*, *Matricaria spp*. and *Epilobium spp*. To identify arable weeds important as host plants to phytophagous invertebrates, the number of interactions derived from information in the Database of Insects and their Food plants (DBIF) between individual weed species and key chick food invertebrates (and their families) was extracted. Weed species recorded in the emerged weed sampling were divided into 4 groups based on the number of interactions for each taxon. The data base records only interactions between invertebrates and plants and doesn’t include abundances or relative importance. However, it is a useful guide to the relative importance of weed species to invertebrates (Marshall *et al*., 2003).

Group	**Number of weed species in group**	**Number of interactions recorded**
A	8	0 – 4
B	8	5 - 14
C	6	16 - 17
D	6	20 - 71

Table 3.2 Number of weed species in our study and interactions with invertebrate taxa and their families important in chick food diets recorded from information in the DBIF.

3.3 STATISTICAL ANALYSES

3.3.1 Partridges

Two analyses were carried out on the partridge data from our count area. The first was a calculation of Chick Survival Rate (CSR) on farms with and without wild headlands. (Potts,1986). Potts found that the number of young hatched per successful nest was consistent between year and study [yr:1907 – 1984, 27 studies, mean 13.84 s.e. ± 0.1]. Using these data he estimated CSR up to the age of six weeks using the power-curve equation, where the geometric brood size = *x*

$$CSR = 3.665x^{1.293} \qquad \text{(Potts, 1986)}$$

The geometric mean for young from all coveys (so not including pairs with no young as the lack of young was probably caused by egg predation at the nest) was calculated for each year from 2015 – 2019. The Potts formula was used to calculate CSR for broods on each site each year. The sample size was too small for further analysis and the raw data was presented.

For the second analysis, data was recorded on sizes of all broods each year, including pairs with no young, from 2014 – 2019. A Generalised Linear mixed model was run with counts of young per covey the dependent variable and as numbers of partridge were count data, a poisson distribution was assumed. Year was added as a random term accounting for the fact that all observations within a year are correlated (poor or good weather at peak hatching for example) and it was assumed that both sites (farms with and without wild headlands) would be similarly affected. Furthermore, the random term made no assumption about a systematic relationship between young and year, it merely measured variation accounted for by year. Also included as a fixed effect was year to allow testing to see whether brood sizes in general changed over the study period. Site (farm) was included as a fixed effect to allow testing to determine whether wild headlands had an effect on brood size. Finally, interaction between year and site was included to enable testing to see whether changes in brood size over the study period differed between sites with and without wild headlands. The analysis is available in an R Markdown file for this and all other analyses on request.

3.3.2 Invertebrates

To compute dissimilarities between cropped headlands with and without wild headlands, a pair-wise comparison between headlands was made using Bray-Curtis dissimilarity indices measuring relative abundance of each insect order in each headland in 2014 and 2017. Bray-Curtis:

$$BC_{ij} = 1 - \frac{2C_{ij}}{S_i + S_j}$$

where $C_{i,j}$ is the sum of the lower of the two abundances of all specimens for only those orders in common between headlands in each pair of headlands and S_i and S_j are the total number of specimens (of orders) counted at each headland *i* or *j*. (Magurran, 2004).

To identify groupings of sites in terms of their composition each field was analysed against every other and the result used in a hierarchical cluster analysis using general agglomerative hierarchical clustering (Ward "D2").

In order to analyse invertebrate abundance and richness in headlands with and without wild headlands alpha biodiversity (α) was measured in several ways: S or species richness - the number of different species (or in this case orders) seen at a point in space or time, N or abundance - the total number of individuals counted (across all orders) at a point in space or time and exp(*H*_Shannon) - or Hill number 1. (Jost *et al*., 2010). exp(*H*_Shannon) takes into account both richness and (numbers of species/orders as well as abundances of species/orders) and is commonly used with data of this type as invertebrate assemblages well fit the prerequisite of the function calculation (infinite population, sampled randomly).

For further analysis, heat maps were prepared using a hierarchical clustering (Ward D2) and Euclidean distances to compute dissimilarities between headland treatments.

3.3.3 Emerged weeds

The analysis used for measuring weed beta (β) diversity from data of emerged weeds collected in 2014 (insufficient data had been collected for analysis in 2017) was, as for invertebrate analysis, a Bray-Curtis measure of dissimilarity. Wild and conventional headland assemblages were expressed in a hierarchical cluster. For measuring alpha (α) diversity, S, a measure of species richness and exp(*H*_Shannon) were used to compare wild and conventional headlands. Shannon is often used to examine seedbank and emerged weed data as it combines species richness and abundance (Hawes et al., 2010). Results comparing α diversity between fields with and without wild headlands were tested for statistical significance. R statistical package 4.02 (R core team 2020) for the α analysis and R package Vegan (Oakensen et al., 2019) for the Bray-Curtis analysis and heat maps.

3.4 RESULTS

3.4.1 Partridges

For the first analysis 885 young partridges in 159 coveys were counted over 5 years. Chick Survival Rate for partridge was lower on farms which hadn't adopted wild headlands compared to farms where they were used in cereal crops in four of the five years (Fig 3.2). Farms without wild headlands had a CSR below the CSR required to maintain partridge populations of 30% (Aebischer, 1997) in 2017, 2018 and 2019. On farmland with wild headlands, CSR was >30% in all five years, reaching a maximum of 42% in 2019.

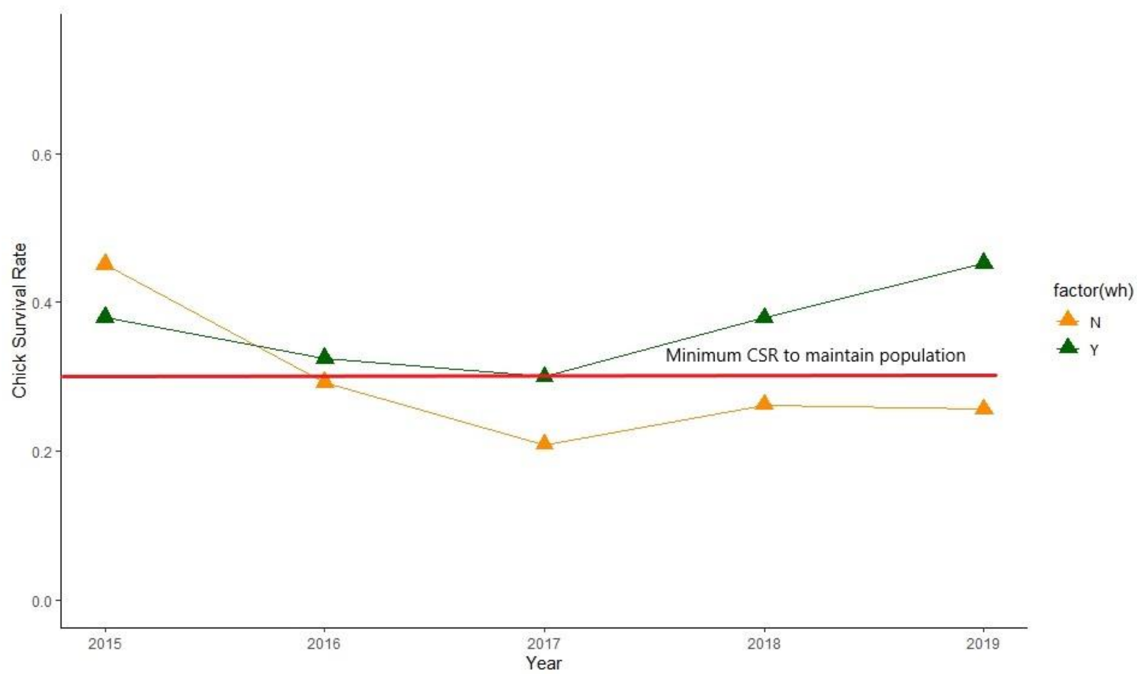

Fig. 3.2 CSR in partridge broods on farms with (green) and without (orange) wild headlands from 2015 to 2019. CSR = 0.3, the minimum level to maintain a population marked in red. Factor (N), without wild headlands, (Y) with wild headlands.

In the detailed analysis of brood number, 183 pairs and coveys were counted over the study period and a significant interaction was found between year and site, confirming that farms with wild headlands have larger broods and which remained constant over the years partridges were counted, while broods on farms without wild headlands declined. The median brood sizes on land without wild headlands changed across the study period at a different rate to those on land with wild headlands (Site*Year: effect size (+/-SE) = -0.166+/-0.054, z = -3.07, P = 0.002; Fig 3.3). Brood sizes on land without wild headlands declined while those on land with wild headlands remained constant (Fig 3.3). Over the 5 years partridges were counted on both sites median brood size on farmland without wild headlands declined by 40% whereas those on farmland with wild headlands remained stable. In 2015 both treatments had a median brood size of ~6 chicks; by 2019 median brood size on farms without wild headlands was ~4. The number of coveys on farms without wild headlands increased over the course of the study, while remaining constant on farms with wild headlands.

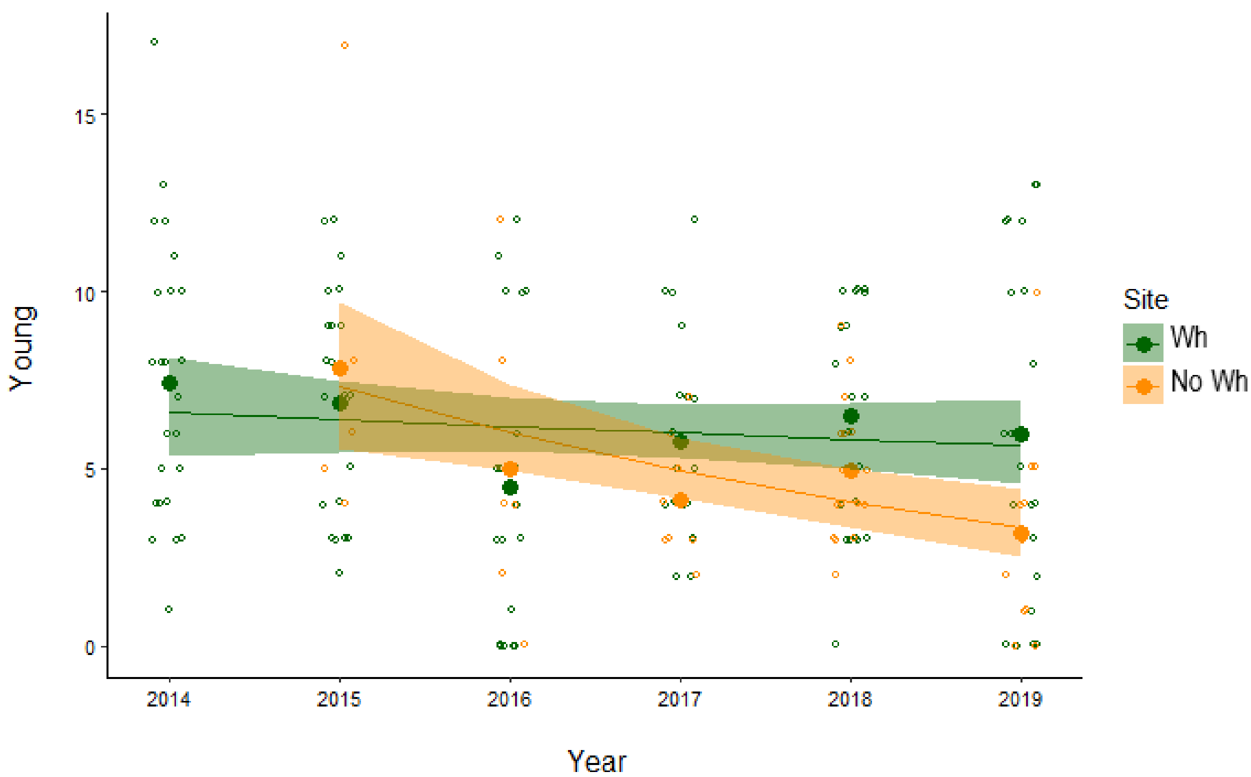

Fig. 3.3. Median brood sizes (closed circles) from coveys counted on farms with (green) and without (orange) wild headlands. Raw data is show as open circles ("jittered" for clarity). Solid line; estimates from the model; shaded areas, 95% confidence interval.

3.4.2 Invertebrates

Populations of invertebrates in wild headlands were similar to each other than to populations of invertebrates in fields with conventional headlands in 2014, but not in 2017 (Fig 3.4 & Fig 3.5). Fields with wild headlands were distributed more randomly on the cluster dendrogram in 2017 compared to 2014.

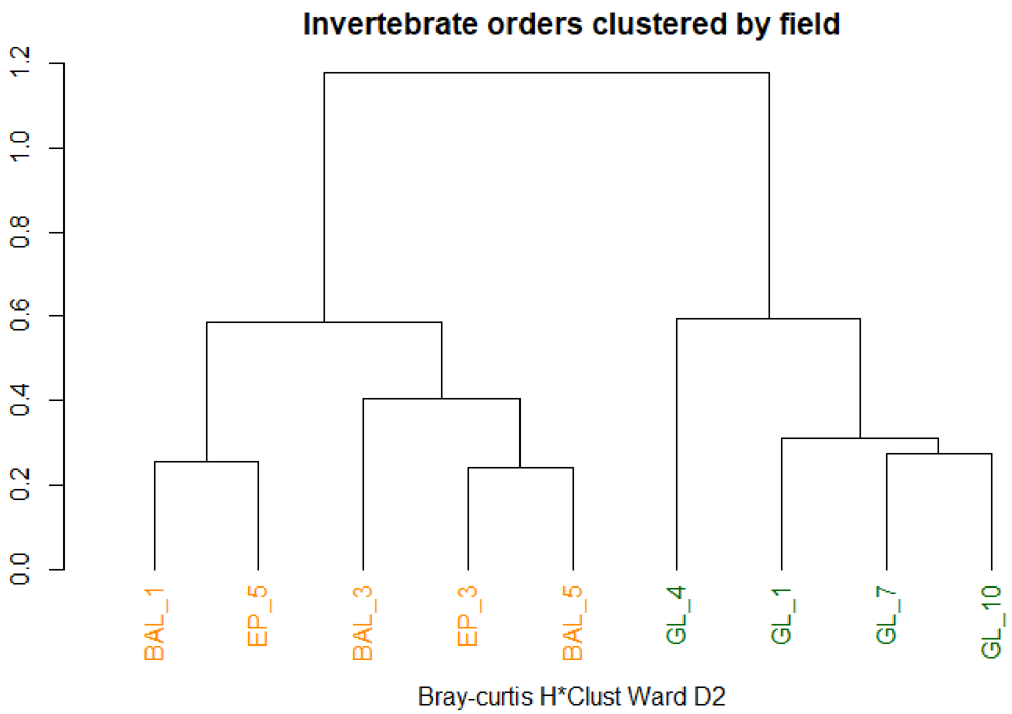

Fig. 3.4 Hierarchical cluster dendrogram using ward D2 clustering and Bray-Curtis dissimilarity measures of order in invertebrate samples of invertebrates collected in 2014. Branches are labelled with field codes. Fields with (green) and without (orange) wild headlands.

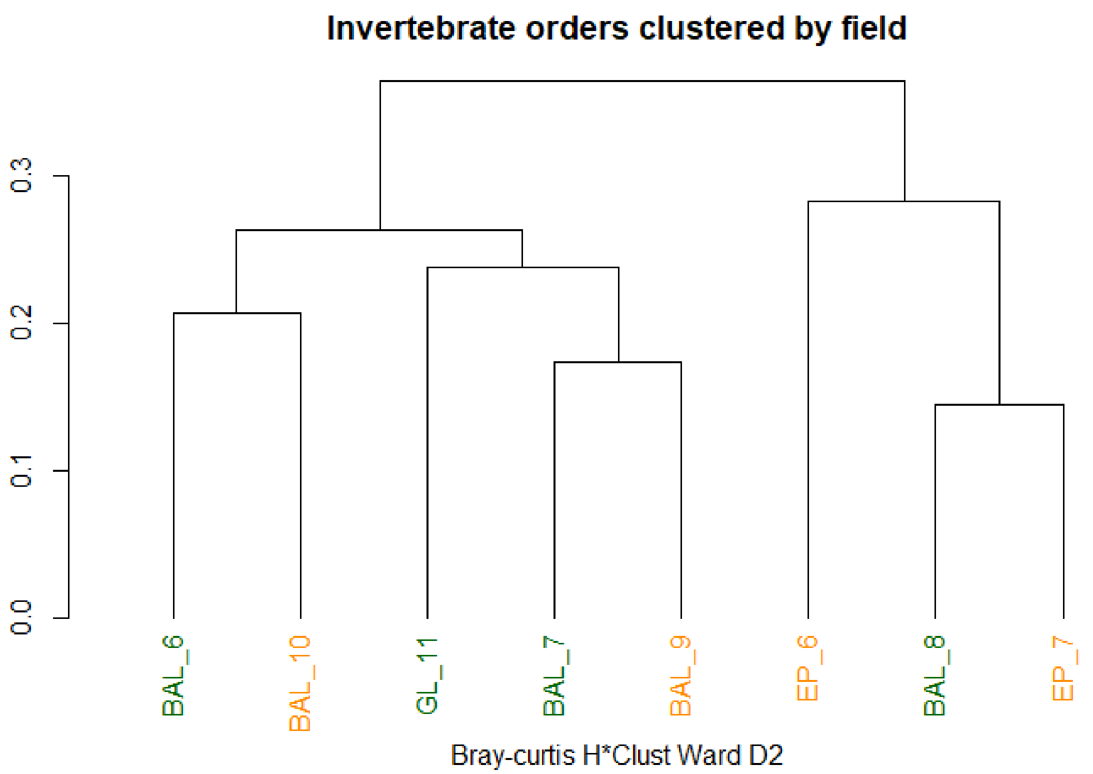

Fig. 3.5 Hierarchical cluster dendrogram using ward D2 clustering and Bray-Curtis dissimilarity measures of order in invertebrate samples of invertebrates collected in 2017. Branches are labelled with field codes. Fields with (green) and without (orange) wild headlands.

The pattern of differences in invertebrate populations between wild and control headlands in 2014 was equally apparent in the α diversity metrics. In 2014 exp(*H*_Shannon), the measure combining species (order) richness and abundance ($F_{7,1} = 6.899$; $p = 0.034$; Fig 3.6), and species richness(order), S ($F_{7,1} = 8.24$; $p = 0.02397$; Fig 3.7) differed in sites with wild headlands, but differently in each case. In 2017 neither measure differed between sites with or without wild headlands. exp(*H*_Shannon) ($F_{6,1} = 0.08$, $p = 0.78$), S ($F_{6,1} = < 0.0001$, $p = 1$).

The Boxplot showing exp(*H*_Shannon) measure of species (in this case Order) richness, evenness and abundance in headlands in winter barley crops in 2014 is given in Fig 3.6. S (species (order)) is in Fig 3.7.

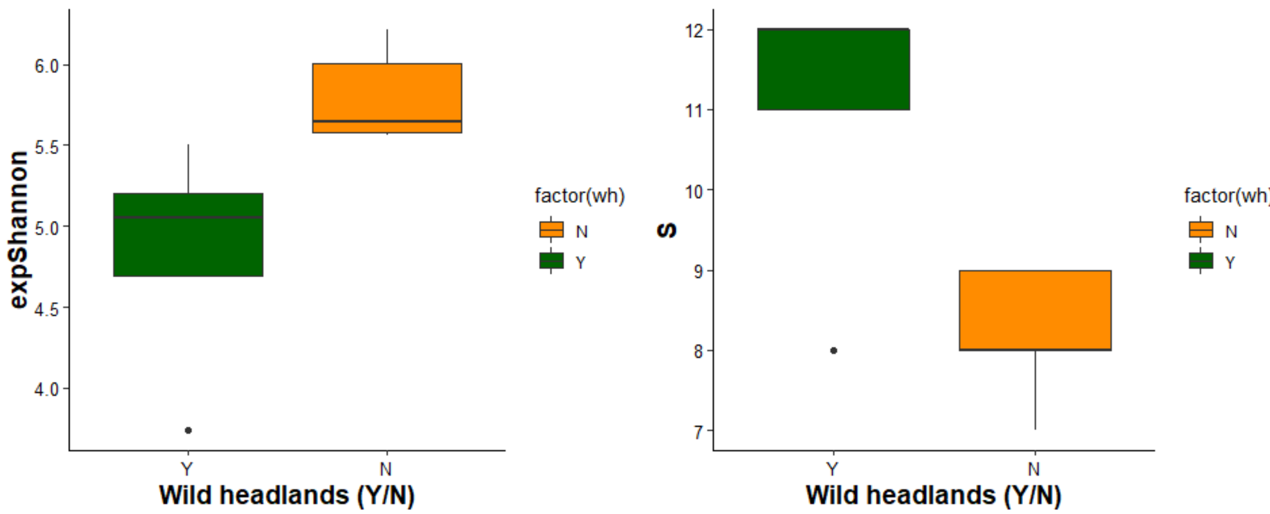

Fig. 3.6 and Fig. 3.7 Median measures of exp(*H*_Shannon) diversity and abundance of invertebrates and S found in crop headlands with (green) and without (orange) wild headlands in 2014. Boxes denote upper and lower quartiles. Whiskers = 1.5 x interquartile range beyond the top and bottom of the boxes. Factor (Y) headlands with wild headlands, (N) without.

The two "heat maps" in Figs 3.8 & 3.9 illustrate the number and order of invertebrates collected in 2014 and 2017. Not all the orders recorded in 2014 were found in the 2017 samples and vice-versa. In Fig 3.8 (2014), note Aphidoiderae and Hymenoptera in fields with wild headlands (key partridge chick food insects) and the lower overall numbers of invertebrates in fields without wild headlands. In Fig 3.9 (2017) fields are not clustered by wild headland but observe the numbers of Collembola in all fields.

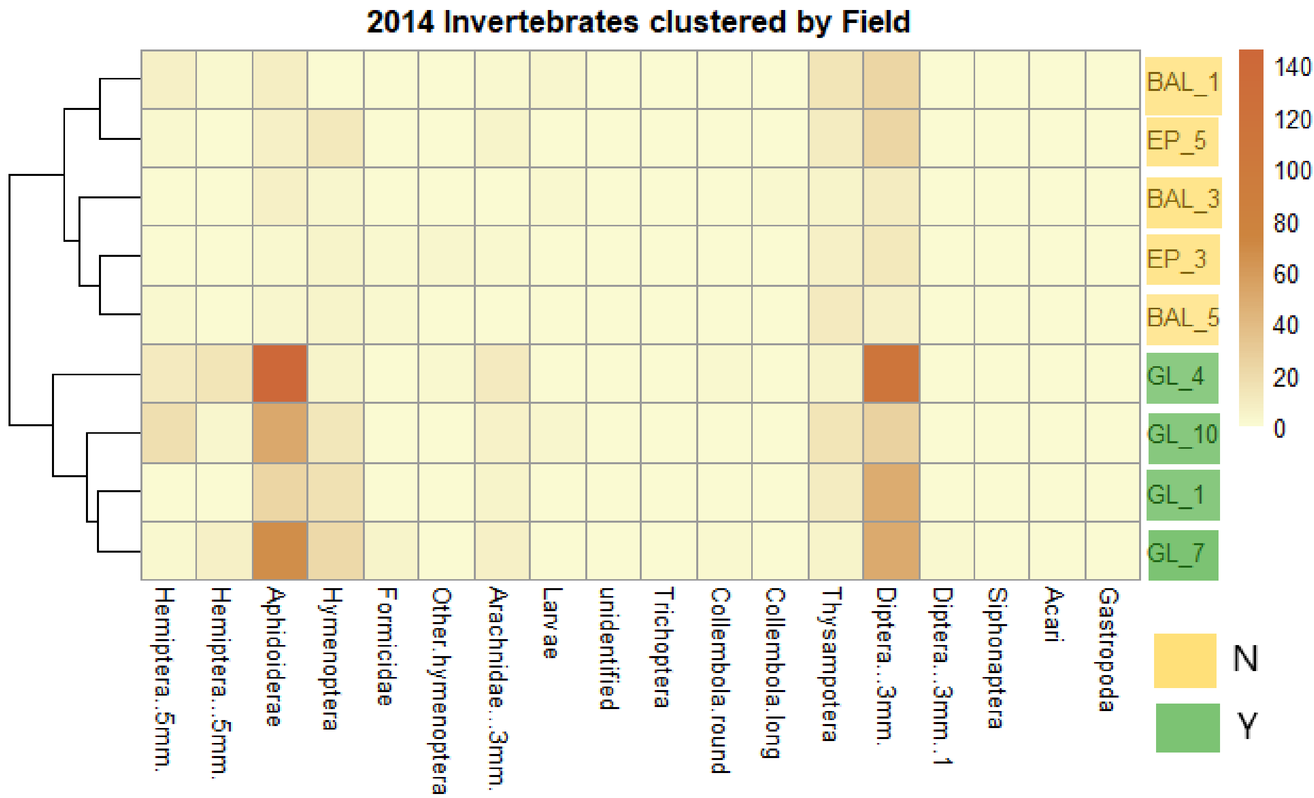

Fig. 3.8 Heat map showing numbers of invertebrate by order (subdivided by size) clustered by field in 2014. Darker colours indicate higher numbers (scaled on the right 0:140). Fields are listed on the Y axis and colured; with wild headlands = green, without = orange.

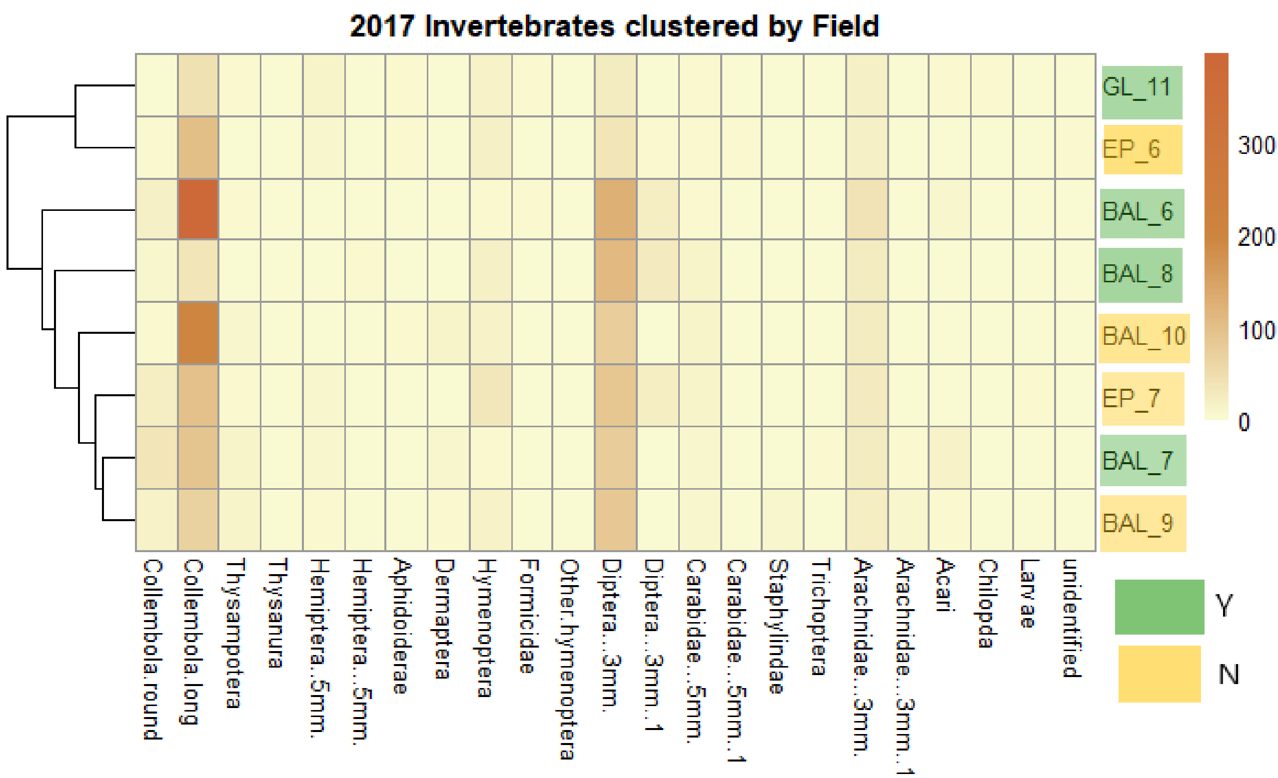

Fig. 3.9 Heat map of populations of invertebrate orders clustered by field in 2017. Darker colours indicate higher numbers of invertebrate orders (Scaled 0:400). Fields are listed on the Y axis, fields with wild headlands = green, without = orange.

3.4.3 Emerged weeds.

Weed species assemblages in 2014 followed the same pattern as invertebrate assemblages. Within wild headlands, assemblages were more similar to each other than to fields without wild headlands (Fig 3.10). The same comparison was not made in 2017 because too few samples were collected.

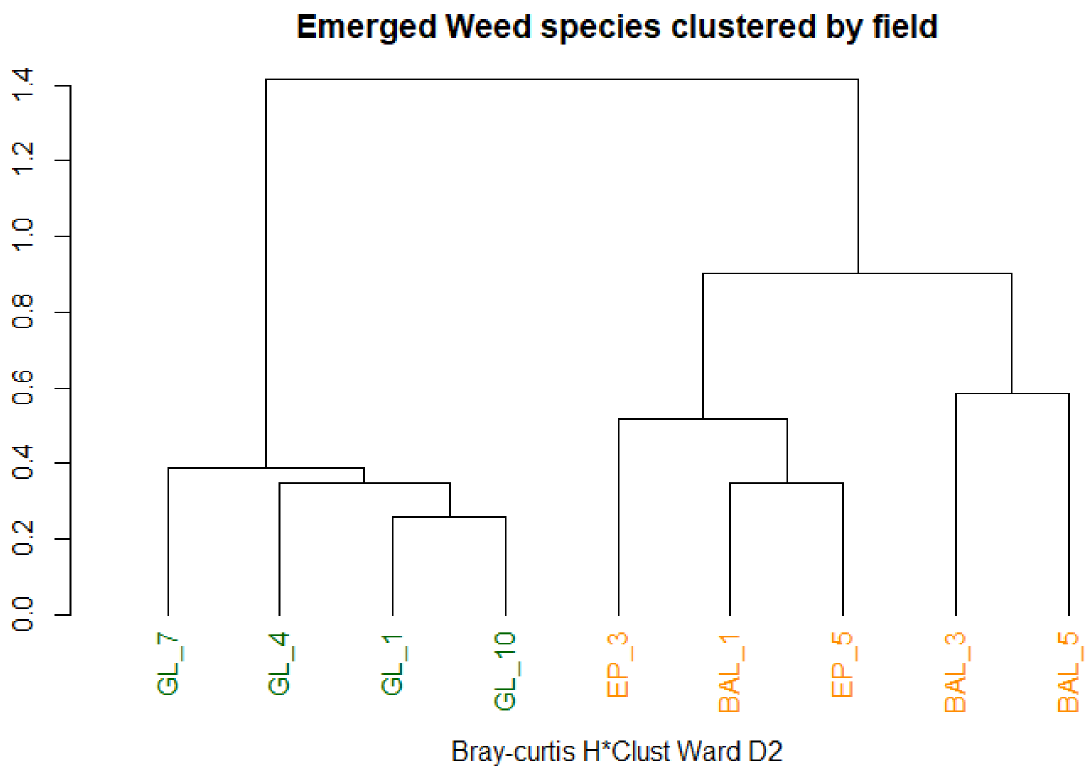

Fig. 3.10 Hierarchical Cluster dendrogram using ward D2 clustering and Bray-Curtis dissimilarity measures of species in samples of emerged weeds in 2014. Fields with (green) and without (orange) wild headlands.

Differences in species richness of emerged weeds between fields with and without wild headlands were considerable. α diversity in 2014; S (species richness) was higher in sites with wild headlands than those without. S: ($F_{7,1}$ = 58.16; p = 0.0001236; Fig 3.11), while exp(*H*_Shannon) was not significant: ($F_{7,1}$ = 2.887; p = 0.13).

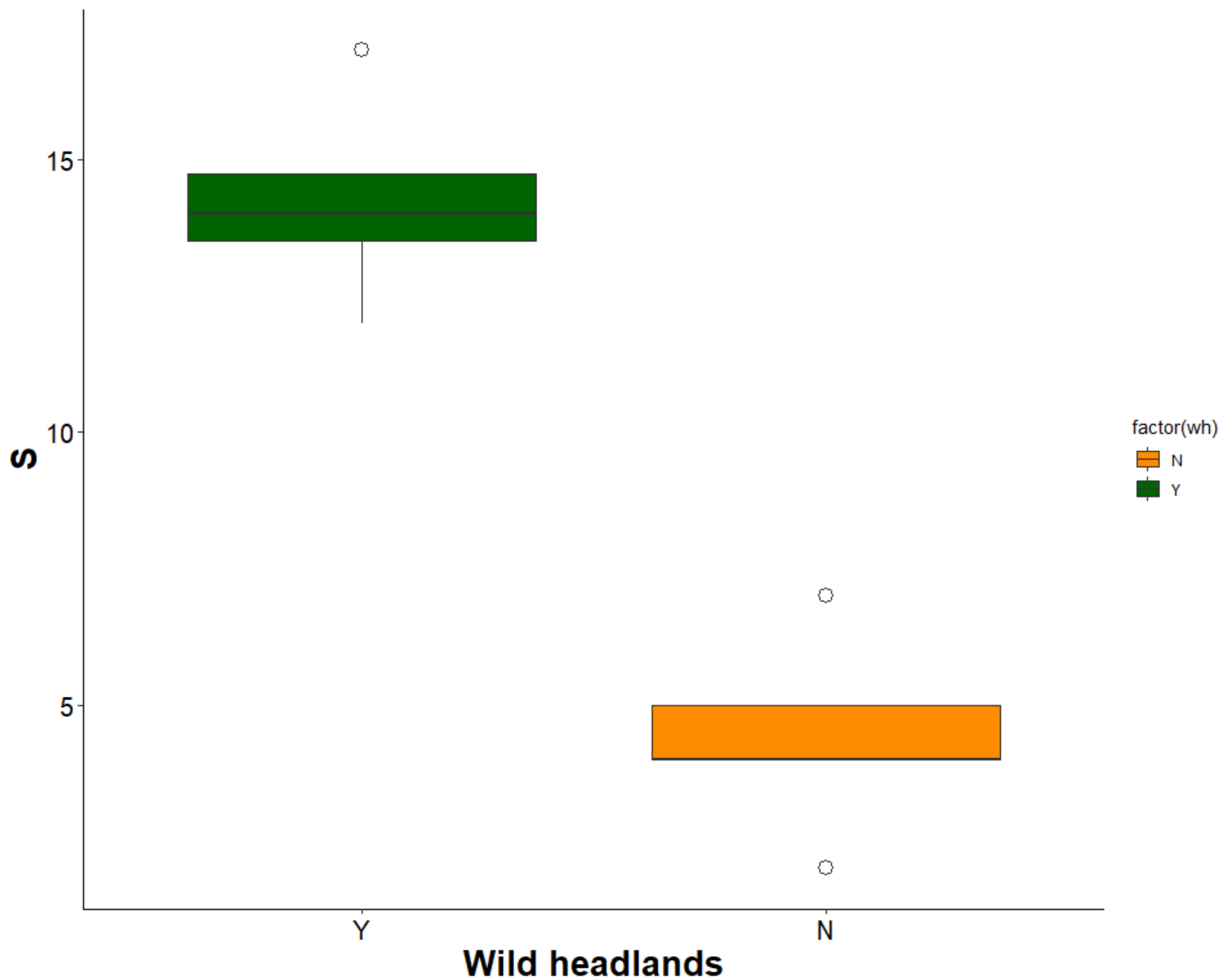

Fig. 3.11 Median measures of S of emerged weeds found in crop headlands with (green) and without (orange) wild headlands in 2014. Boxes denote upper and lower quartiles. Whiskers = 1.5 x interquartile range beyond the top and bottom of the boxes. Outliers are > 1,5 x interquartile range. Factor (Y) headlands with wild headlands, (N) without.

Weed species were linked to invertebrate orders in the DBIF arranged fields by wild headland and grouped weed species by the number of chick-food invertebrate interactions. The heat map in Fig 3.12 illustrates the result. Wild headlands contain many more weed species that are suitable hosts for a wide range of invertebrate families compared to fields which were sprayed. Weed assemblages in this study bore a very strong similarity to work done on conservation headlands 40 years ago. Of the weed species occurring most frequently in wild headlands in this study: *Matricaria* spp*, Polygonum aviculare, Veronica spp. Stellaria media, Myostotis arvensis, and Poa* spp.; *Matricaria* spp, *Polygonum aviculare,* *Veronica* spp and *Stellaria media* were all more abundant in unsprayed plots than controls in a spring wheat field in Hampshire. (Chiverton and Sotherton, 1991).

When those weed species identified by DBIF as being important hosts for invertebrates were considered, it was observed that those in group D (most suitable for inverts) were only found in sites with wild headlands, while those in group C were generally more prolific in sites with wild headlands compared to those without wild headlands.

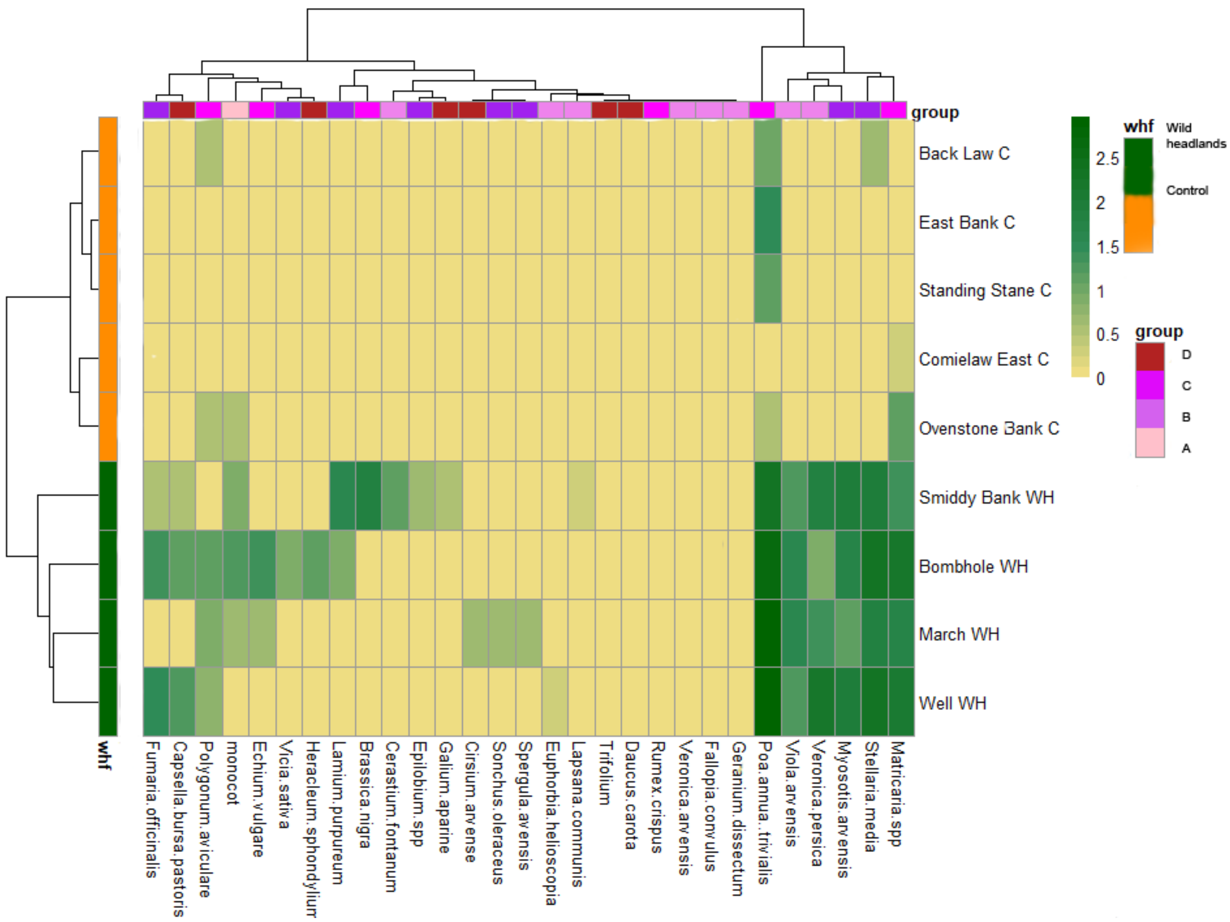

Fig. 3.12 Heat map showing weed species in 2014 clustered by wild headland and suitability as host plants for invertebrates. The colour shading on the top row (groups A:D) represents the frequency that weed species are linked to partridge chick food invertebrates in the DBIF (Table 2). D (dark pink) is most frequent. The Y axis (dark green) = fields with wild headlands; dark orange = fields without wild headlands. Within the plot the darker the green the higher the number of weed specimens as per the green-yellow scale (0-3) shown.

3.5 DISCUSSION

On farms with wild headlands, partridge produced larger broods and exceeded the CSR required to sustain the population, while in control areas without wild headlands brood size declined over the study period. In several years CSR on the control areas fell below critical values, which was likely - in the absence of immigration - to lead to population declines. Differences in brood sizes can be (at least partially) explained by differences in the availability of important groups of chick-food items (Rands, 1985; Potts, 2012), which in turn are linked to pesticide practice. Chiverton (1993) found significantly higher mean densities of chick-food items in unsprayed headlands. In Scotland, Hughes (1999) reported that conservation headlands were responsible for increased abundancies in all insect groups. Frampton and Dorne (2007), in a meta-analysis bringing these and other studies together, found abundance of Heteropteran invertebrates up to 12.9 times higher where pesticide was restricted. In our study, the relationship between weed species richness in wild headlands and invertebrate communities was clear in 2014. In both cluster dendrograms (Fig 3.4 & Fig 3.10) and the diversity analyses in 2014 (Figs. 3.6 & 3.7 and 3.11) of weed and invertebrate communities were defined by wild headlands, although the results from the Shannon diversity analyses were counter-intuitive given the raw data. In 2017 there was no obvious pattern in the cluster dendrogram (Fig 3.5) and wild headlands did not explain species richness or abundance. It may be because aspect and field boundary explain local variation [although pesticide use is more important in explaining trends] (Ewald *et al*., 2015). In 2017 wild headlands were all placed on the west side of fields (so open to the east) and consequently conditions were necessarily different to 2014 when they were on the north boundary (and so open to the south). Field boundary types were more varied in 2017 than 2014 with one (conventional) headland beside a wood including many Collembola. Regardless of these inconsistent differences in weed and invertebrate diversity between treatments, the differences in partridge brood size and productivity between wild headlands and control sites remained and indeed increased over the study period.

The relationship between chick productivity and invert abundance and diversity may arise because many of the sampled invertebrates are a source of crude protein, key for feather growth and resistance to chilling in birds (Southwood and Cross, 2002). Figures given in Southwood and Cross for crude protein in some taxa that we encountered in our samples are as follows: 58% for Collembola [dominant in 2017 per the heat map Fig 3.9], 58% for Diptera and ~50% for Heteroptera, Hymenoptera and Orthoptera. Although Collembola, with the exception of Sminthuridae, are seldom found in the diet of partridges they are an important food source for ground beetles (which are eaten by the chicks) while Aphididae were the most abundant items in partridge chick diet in the GWCT Sussex study (Potts, 2012). Differences in invertebrate abundance and diversity between sites may be attributable to the presence and abundance of a small set of weeds that are especially palatable to invertebrates (those found in Groups C & D of the DBIF classification), which although not abundant themselves, were more prolific in sites with wild headlands. Further detailed study to explore the plant-animal relationships in this context is desirable, with preferably higher taxonomic resolution to tease out effects. The differences in invertebrate population abundances and composition between years are perhaps not surprising. Short-term variations in invertebrate populations are not uncommon. Between-year variation was found in two recent large-scale meta-analyses by Bell *et al*. (2020) looking at aphid and moth abundancies across Great Britain and in a recently published global review of invertebrates (van Klink *et al*., 2020). Both found population fluctuations over different time periods while van Klink *et al*. (2020) also reported a decline in terrestrial insect abundance by ~9% per decade. It would be interesting to explore whether long-term patterns of invertebrate abundance and diversity were higher and more stable in sites with wild headlands, including other crops, although this would require established long-term monitoring. Meanwhile, it is likely that in the absence of intervention invertebrate populations on farmland will continue to decline (Harvey *et al*., 2020).

Explanations, other than invertebrate availability, for the differences in brood sizes across the count area can likely be ruled out, although these metrics would be difficult to quantify without an extensive survey. Legal predator control, protecting nesting partridges and their eggs, was conducted uniformly across the count area, so too the provision of feed hoppers containing grain. Nesting cover in km^{-1} was consistent too with field sizes similar on farms with and without wild headlands. Indeed, farms without wild headlands in the count area were mostly on light Dreghorn series soils in an open landscape with few trees, and therefore considered ideal ground for partridges (Potts, 2002. Pers comm. Dr Potts counted partridges on the study site in 2002). Climatic and soil conditions in our count area were therefore *more favourable* across farms without wild headlands. Despite these more favourable conditions on control sites and a high level of game management (predator control and feeding), we still found a marked difference in chick productivity that was associated with the presence of wild headlands. This indicates that, as suggested previously, partridge populations may be highly dependent on farmland management that supplies invertebrate chick food and the weeds that support them (Aebischer and Potts, 1998; Aebischer and Ewald, 2004). The provision of wild headlands appears to fulfil these demands and hence ensure higher partridge chick productivity.

Wild headlands differ from conservation headlands as without nitrogen fertiliser there is a reduced biomass of both crop and weed (Blackshaw *et al*., 2003). Given that sites with high vegetation densities provide more food for polyphagous predators as they attract more herbivorous invertebrate prey

(Hassal *et al*., 1992), there *may* be a negative impact on invertebrate provision in wild headlands through reduced biomass. However, in mitigation, conservation headlands without fertiliser (a wild headland) have a more open structure and so have greater species richness than fully fertilised conservation headlands, probably as a result of increased light penetration below the crop canopy (Kleijn and van der Voort 1997, Walker *et al*., 2007, Seifert *et al*., 2014). We found higher weed species richness in wild headlands in fields that were sampled from a farm with a long history of intervention. This is consequential as Auchenorrhyncha, Heteroptera and Araneae, all important invertebrates in bird diets (Wilson *et al*., 1999) are significantly correlated with plant species richness (Asteraki *et al*., 2004; Haddad *et al*., 2009; Storkey *et al*., 2013; Smith *et al*., 2020). An open structure also allows access to resources (plant and invertebrate) for the benefit of farmland birds which can otherwise be difficult in sown margins (Vickery *et al*., 2009) although relieved through scarification (Westbury *et al*., 2017). Aebischer and Ewald (2004) used the Potts model (Potts, 1986) to calculate that 6% of arable area was needed in insect rich habitat to give a chick survival rate of 0.44, without predation control. Wild headlands, at 6m wide and on only one side of the field, are likely to cover no more than 2% of farms in our count area which had wild headlands. Nevertheless, CSR was up to 0.45 (Fig 3.2), which suggests that keeping 6% of the arable area in conservation headlands may not necessarily be required to maintain partridge populations, although these figures would be different in the absence of predator control (Potts, 1986).

This study, asking how wild headlands influenced the productivity of an indicator farmland bird, the partridge, demonstrates that this novel management option has the potential to increase brood sizes, critical to consistently achieve levels above those required to sustain the population. Modifications to farmland management will only be adopted if practical. It is encouraging that wild headlands were adopted across Farm 3 in 2015 and continue to date. The simplicity and apparent sustainability (sustainability is explored in later chapters) of the approach appealed to the farmer (Anstruther, 2018. *Pers comm*. Toby Anstruther owns Balcaskie). The effect was achieved by *not* doing something, in this case *not* applying fertiliser and *not* applying herbicide, which makes operations easier in a large integrated business. Their continued use has been justified by the steady population of partridges in the face of declines elsewhere, perhaps even at a very local level as was demonstrated with lower CSR and smaller broods in neighbouring fields lacking wild headlands adjacent to the sites. Unfortunately, despite brood production in the partridge population on farms with wild headlands remaining constant, the population hasn't increased over the course of this study. This may be because the local area is saturated and so farms with wild headlands are acting as sources for the population that are then lost in nearby, less favourable sink sites. It is suspected that this is not the case given historic records of partridge population levels being 10 times greater than currently observed. Alternatively, wild headlands are only part of the solution and additional measures; field division and “umbrella cover” to protect against raptors for example, will be necessary if an increase in the population is the desired objective. A key strategy will be mitigating the substantial winter losses, such as having 1% of the farm in winter holding cover for partridges (Nickerson, 1989) and thus helping avoid migration to adjacent areas which are acting as population sinks. The successful restoration of a partridge population in Sussex at Peppering from a population of 3 pairs to over 350 in eight years required the dedication of 10% of the arable area to partridges and a farming system designed around their requirements, of which provision of chick food formed an essential part (Potts, 2012).

3.6 Conclusion

This thesis set out to investigate whether wild headlands enhanced biodiversity on arable farms and used the partridge to test the hypothesis that they did. The result, that provision of chick food invertebrates alone in wild headlands positively influenced the productivity of an indicator farmland bird, is confirmation that they do. This conclusion has implications for adopting wild headlands as part of a suite of measures to promote sustainable intensification in the UK countryside.

3.7 Acknowledgements

The authors are grateful to Prof. Anne Magurran and Faye Moyes of St Andrews University for their invaluable advice and help with statistical analysis and figure presentation of plant and invertebrates, to Prof Joah Madden of Exeter University and Dr Matt Bell of Edinburgh University for their guidance on modelling the partridge population and to Dr Cathy Hawes and Prof. G.R. Squire of the James Hutton Institute for further advice and guidance. The authors are grateful to the landowners of the East Neuk Estates http://www.eastneukestates.co.uk/ for access to their farms, particularly Balcaskie Estate for permission to use their partridge data. We also thank the staff at Gilston for their support with invertebrate collection. This research did not otherwise receive any specific grant from funding agencies in the public, commercial, or not-for-profit sectors.

4 YIELDS AND GROSS MARGINS AT THE EDGE OF CEREAL CROPS WITH RESTRICTED FERTILISER AND PESTICIDE.

Abstract

The GWCT developed their conservation headland in the mid-1980s in a bid to restore a food source for the phytophagous invertebrates critical for partridge chicks by retaining key host plants within crop headlands. Although successful and supported under AES across Europe, they were never widely adopted, principally through farmers' dislike of weeds growing in their crops. At an arable farm in East Fife (Latitude: 58º N, Longitude 2.50º W), the original prescription was modified in the early 1990s to exclude nitrogen fertiliser, thus cutting down weed growth. Headlands were rotated annually when in cereal crops, which limited the build-up of dominant weeds in the seedbank through conventional herbicide applications in the four (or more) intervening years. This study looks at yields and gross margins in wild headlands compared to field yields in ~70 fields after 20 years intermittent use of wild headlands. We found crop yield in wild headlands to be ~ 50% of field yield, while gross margin was little affected in low value cereal crops where area subsidy was a large proportion of gross output. The study has implications for government agencies designing AES schemes in the post-Brexit environment.

4.1 INTRODUCTION

Following the ending of WW2 there was a significant increase in the intensity of agriculture in the UK. Scientific developments: improvements in plant genetics, the widespread introduction of herbicides, fungicides and mineral fertiliser (particularly nitrogen fertiliser) resulted in an increase in production that has plateaued since 1990 (Storkey and Westbury, 2007). Per Food and Agriculture Organisation (FAO) data, crop yield in industrialised countries grew at c 1.05% per annum from 1961 to 2014 and at 0.88% p.a. from 1991 to 2014. From 2001 to 2014 this had slowed to 0.75% p.a. implying a considerable slowing of growth in output from 1991. Cropping intensity meanwhile had increased sharply (Fuglie, 2018).

The pattern in crop yield in Scotland, an industrialised country, has followed this trend. While yield growth has stalled in Scotland pesticide and nitrogen usage have continued to rise per unit of output (Squire, 2015).

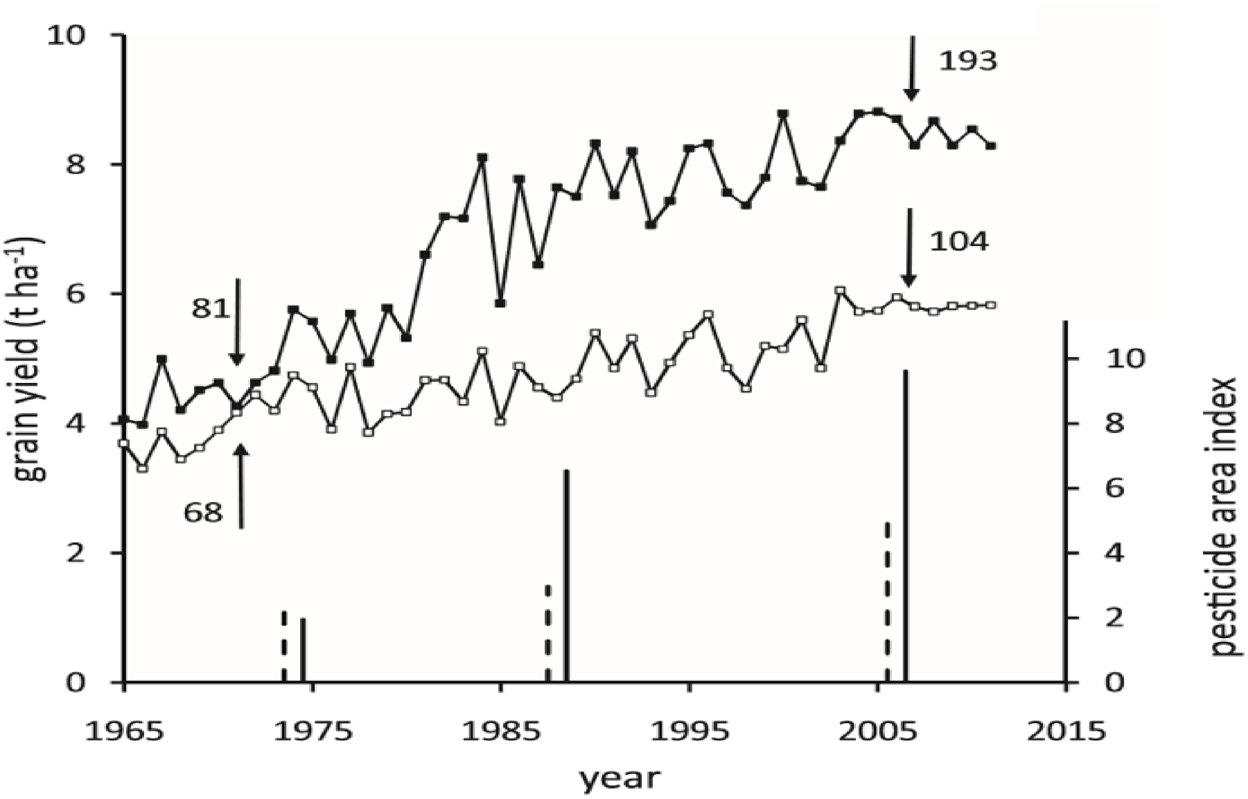

Fig. 4.1 Yield in t ha^{-1} for winter wheat (top) and spring barley (lower). The vertical arrows are for kg N ha^{-1} applied at the mid-point in two ranges: 1969-1972 and 2002-2007, with pesticide use for wheat (solid bars) and spring barley (dotted) (Squire *et al*., 2015).

Commensurate with the increased agricultural output has been a corresponding decline in habitat heterogeneity (Benton *et al*., 2003, Richards *et al*., 2018), a loss of resource (habitat and food) for farmland birds (Wilson *et al*., 1999, Westbury *et al*., 2017) and consequent drop in farmland bird population (State of Nature, 2020). There has been a particularly steep decline in one farmland bird species, the partridge with numbers reducing by 82% in the UK from 1970 to 2010. (Potts, 2012). Potts in his Royal Agricultural Society of England (RASE) paper on the options for game and wildlife (Potts, 2002) linked this decline to increasing wheat yields. The factors which had enabled an increase in wheat yield: short strawed varieties responsive to additional nitrogen, herbicides, growth regulators and fungicides gave homogenous wheat crops with little available resource for wildlife (Potts, 2002).

Potts (1980) identified that one such key resource for partridges were phytophagous invertebrates whose availability had been reduced by increased intensity, particularly the use of broad leaf herbicides which controlled their host plants. From his work the GWCT developed conservation headlands, a technique eschewing certain broadleaf herbicides in the outer edge of cereal crops. This enhanced the availability of arable weeds and hence invertebrates for farmland birds (Sotherton,1991).

The technique was researched and applied progressively in the 1980s within the UK (*cf*. Rands, 1985; Boatman *et al*., 1999) and introduced into AES schemes at that time. As designed by the GWCT it depended on the management of the outer 6 - 10 m of the crop while maintaining fertiliser applications and crop yield. The idea was that the arable field margin, defined to include the area between the field boundary and the first tramline (Marshall and Moonen, 2002), is particularly important for resource provision for farmland birds in intensively managed arable landscapes (Vickery *et al*., 2009). The seedbank is often more diverse at the crop edge (Marshall, 1989; de Snoo, 1997) with consequent potential for fostering a diverse field margin (Asteraki *et al*., 2004) and promoting dicotyledonous arable weeds in cereal crops (Wilson and Aebischer, 1995).

Yields at the crop edge are often lower than mid-field (Chaney *et al*., 1999), so management intervention at the crop edge also has a lower opportunity cost in terms of forgone crop than whole field measures. Farmland bird packages developed under AES measures as a result have largely been concentrated at the field edge (Winspear *et al*., 2010). Walker *et al*. (2007) found one such measure, fertiliser-free conservation headlands, particularly beneficial for arable plants. This study looks at the economic impact of fertiliser-free conservation headlands on field headlands over the very long term.

The study took place on four adjacent arable farms in East of Scotland (Latitude: 58º N, Longitude 2.50º W) and considered the implications of adopting wild headlands, the modified version of the GWCT conservation headland. For a full description of a conservation headland and a wild headland, please see the general methods chapter, which includes background information on the study farms.

This study examined headland yield in 4 principal cereal crops grown in Fife to establish the impact of wild headlands on crop yield, to test the effect of headland compaction on crop yield and to determine if after 20 years periodic use of wild headlands (interventions) there was a legacy as well as in-year effect. All these factors, including crop input and cereal prices, were included in a financial model. Payment rates within AES are calculated on profit forgone (Keenleyside *et al*., 2011) so this study will compare the cost of wild headlands to the farmer to current payment rates under Countryside Stewardship, the most recent AES support in England. The information on costs of wild headlands is valuable for agencies designing future agri-environment schemes.

4.2 METHODS

4.2.1 Study sites

The study sites comprised 67 fields across 4 farms covering ca. 1,500 ha of our wider study site. The location of the fields for this study is shown in Fig. 4.2 (for back ground to the study sites please refer to the general methods chapter).

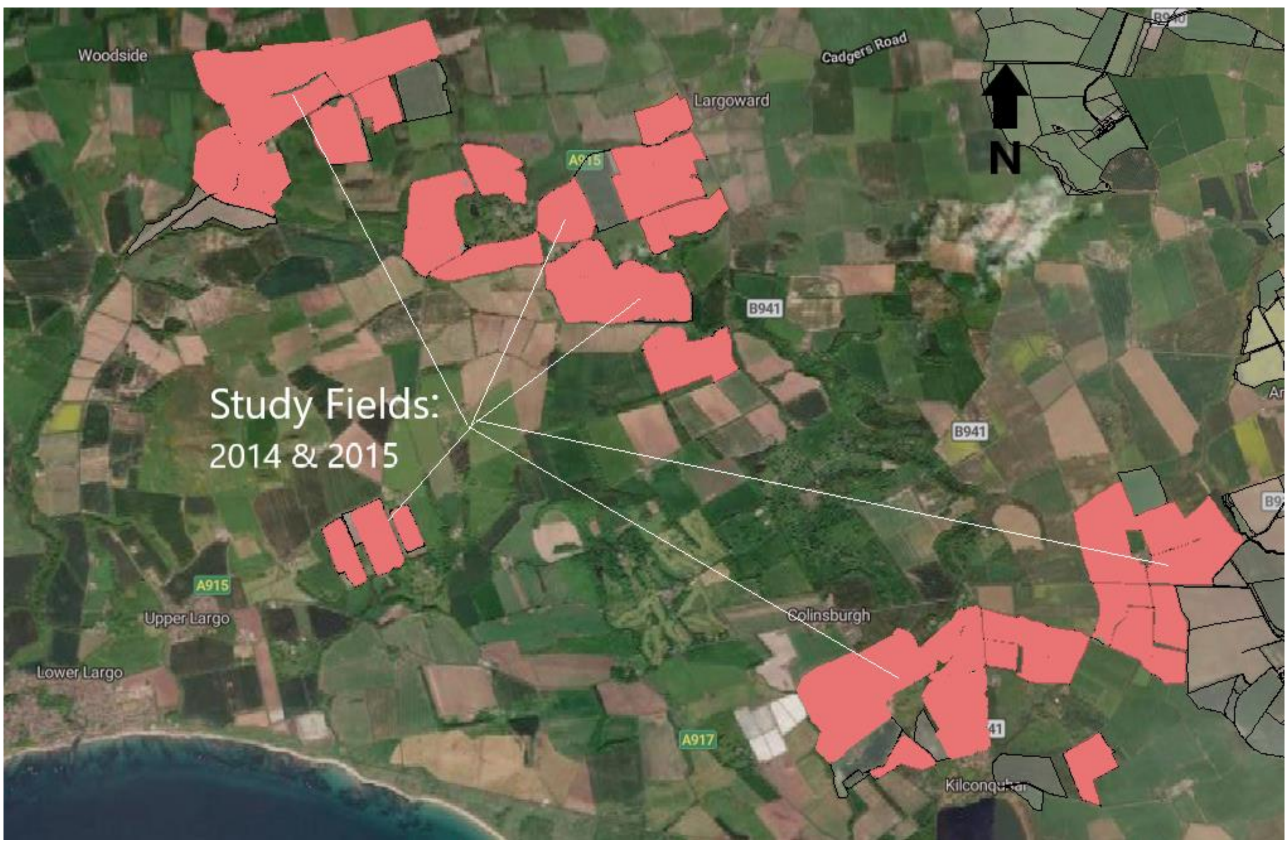

Fig. 4.2 Study fields in 2014 & 2015 shown in pink. Some fields had yield data recorded in both 2014 & 2015 including headlands on the North in 2014 and East in 2015. Detail in Table 4.1. Farmworks 2020.

4.2.2 Sampling methods:

Wild headlands 7m wide (a boom section of a Knight 28m sprayer) were established on the northern 7m of 21 headlands in four cereal crops for the 2014 cropping year with 21 headlands as controls. For 2015, per the practice of rotating wild headlands, wild headlands were emplaced on the eastern headland of 20 fields with a further 20 eastern headlands as controls (Table 4.1).

Year	**Fields**	**Turn y**	**Turn n**	**Cat. h**	**Cat. l**	**SB wh**	**SB c**	**SO wh**	**SO c**	**WB wh**	**WB n**	**WW wh**	**WW n**
2014	42	21	21	20	22	7	4	8	9	3	4	4	3
2015	40	20	20	21	19	5	7	8	7	1	3	5	4
	82	41	41	41	41	12	11	16	16	4	7	9	7

Table 4.1 Fields in 2014 and 2015 with numbers of fields in the study. SB = spring barley, SO = spring oats, WB = winter barley, WW = winter wheat. wh = wild headland c= conventional. Turn (y/n) is if the headlands were used for turning by machinery and is a proxy for compaction. “Cat”, category, is a record of past intervention where (h = >2, l = ≤ 2) (see text for detail). wh = with, c = conventional headland.

Twenty-five fields were included in both 2014 and 2015 years giving a total of 82 combinations of fields and headlands in the study. As cropping changed between years and different headlands were measured each year, we didn't test for field effect on the 25 fields occurring in both years. No wild headlands had herbicide or insecticide applications but had a full fungicide and growth regulator program, with a consistent seed rate across the field with no reduction in the 7m margin. Spring crops in the study received an NPK fertiliser (Compound fertiliser including nitrogen, phosphate and potash) when sown and a top dressing of nitrogen at Zadocks GS20, but both base fertiliser and top dressing were omitted (2 x 4m drill widths) on wild headlands. Using a Kuhn broadcast spinner with headland management no wild headlands in winter cereals received nitrogen fertiliser on the outer 7m of the crop, while the remaining crop received fertiliser applications based on recommended practice codified in RB209 (Agricultural and Horticultural Development Board Nutrient Management Guide. Anon 2020). Full details of applications and input costs per field: seed, fertiliser, pesticides, lime and dung are available on request. Recorded history for each headland of fields in the study over the previous 19 years is known and divided into 2 categories as follows: No wild headlands or one occasion, (n= 41) and 2-3 occasions or more (n=41). Crop yield for headlands and fields was recorded in 2014 and 2015 using the on-board yield meter on a New Holland NH 9070 combine, equipped with a 7m table. The meter measures clean grain volume in the elevator to calculate yield values. These are measured every second, adjusted for moisture and the location recorded through GPS as the combine advances through the crop. Field tests have shown this system to be 97% accurate (Burks *et al*., 2003) and in this study results were referenced against field yields reconciled over a weighbridge. Measurements were recorded over a 100m distance around the mid-point of headlands (except for one 94 m headland where fewer samples were recorded) and the mean calculated. Mean field yield (excluding the 7m headland) and mean headland yield was calculated for each field in each crop in t ha^{-1} at 85% dry matter. The ratio of mean headland yield to mean field yield was calculated for each field. Crop inputs; fertiliser (nitrogen and base fertiliser), seed and pesticides were recorded for each field and sale prices for harvested crops recorded in each year. Where lime and Farm Yard Manure (FYM) were applied to fields (fields limed n = 15, FYM applied n = 9), this was over the whole field with no distinction made for headlands.

4.3 Analyses

Data was analysed in two tranches using R Version 4.02 (R core team 2020) by fitting a Linear mixed model to the data. The first analysis considered headland yield as a proportion of field yield as the dependent variable and the second used headland gross margin, including all input costs, as a proportion of field gross margin as dependent variable. We added year as a random term accounting for the fact that all observations within a year are correlated (poor or good weather at harvest for example) and assumed that all crops and fields would be similarly affected. Furthermore, the random term made no assumption about a systematic relationship between the proportion of headland yield and year; it merely measured variation accounted for by year. Fixed terms we used were; the interaction between crop and wild headlands (the interaction plot is included in appendix 4.2), year, crop, wild headland, category and turning (a proxy for compaction) (Håkansson, 2005). We included category to see if the past history of intervention explained any variance and likewise, if turning did as well. Terms were subsequently dropped until the minimum adequate models (lowest Akaike's Information Criterion) contained only factors whose elimination would reduce the explanatory power of the models. The models were tested using a Wilcoxon signed rank test with an examination of QQ plots and residuals to check assumptions over distribution. Full financial information for each field is available on request.

4.4 Results

4.4.1 Yield impact of wild headlands

Yield in wild headlands was lower than in the rest of the field, but the degree varied by crop. Turning on headlands had an impact on yield in some crops, but not all, and headland yield tended to be lower than field yield with or without a wild headland. The full model (crop, wild headland, headland history (category) and the interaction between crop and wild headland) explained 64% of the variation in the data set. Year was not significant. Fig 4.3 shows the impact on the proportion of field yield in headlands for four crops, in fields with and without wild headlands. Median yields for wild headlands were 60% of field yield for spring barley, 70% for spring oats and ~ 50% for winter barley and winter wheat. Yields in conventional headlands were unaffected for winter crops and ~90% of field yield in spring crops. This compares to a 10% reduction in headland yield of winter wheat in a 5-year study of a farm in Oxfordshire (Pywell *et al.*, 2015). For clarity, the boxplot in Fig 4.3 used only crop and wild headland as fixed terms, while for the model we used the terms. Model: crop + WH + crop:WH + newcategory + (1 | year). ($F_{73,8} = 19.43$; $p < 0.001$; Fig 4.3).

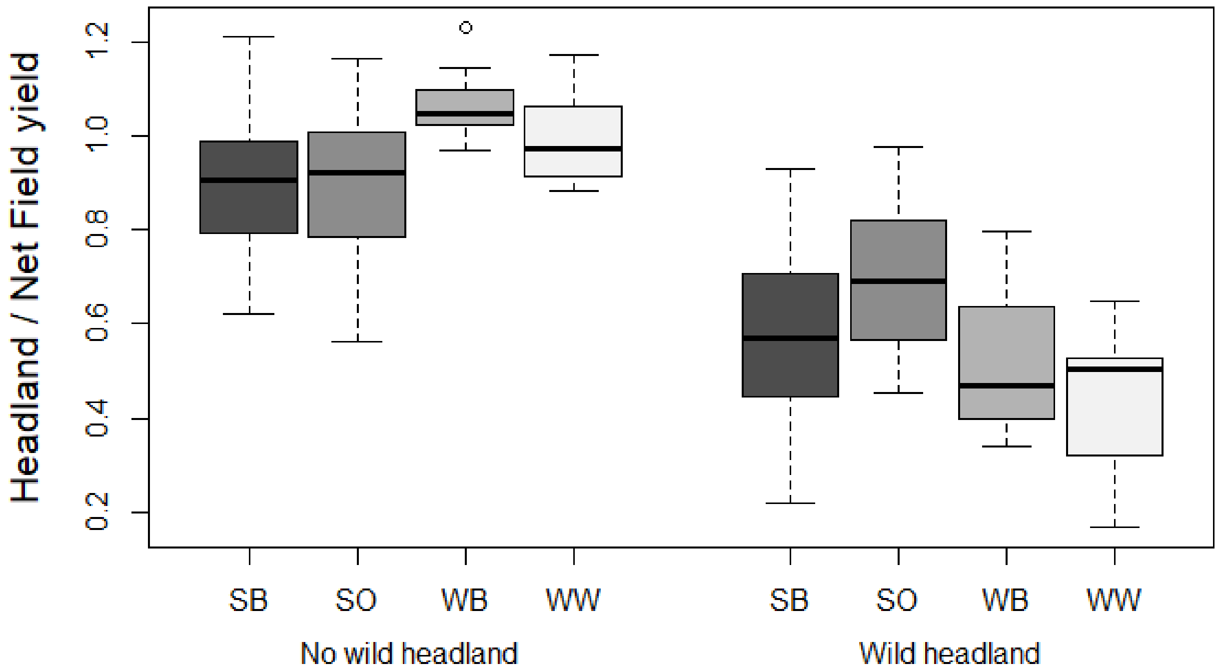

Fig. 4.3 Median headland yield as a fraction of net field yield for 4 crops combined over 2014 and 2015 without and with wild headlands where SB = spring barley, SO = spring oats, WB = winter barley, WW = winter wheat. The outlier "o" lies > 1.5 interquartile ranges beyond the top of the box. The Upper and Lower whiskers = 1.5 x interquartile range beyond the top and bottom of the boxes which delineate upper and lower quartiles. Not all model terms were included in the construction of the boxplot to maintain clarity. Summary field information per Table 4.1

Turning had a significant effect on headland yield ($p < 0.05$) but didn't increase the explanatory power of the final model. The past incidence of wild headlands, where there had been two or more wild headlands in the past 20 years, was significant ($p < 0.001$) and is included. Additionally, wild headland yield as a proportion of field yield for spring oats had a strong interaction ($p = 0.04$) and the interaction was included in the model.

4.4.2 Financial implications of wild headlands

Gross margin (GM) in headlands as a proportion of average field GM varied across crops and was highly influenced by savings in inputs on wild headlands. Winter crops showed similar differences in proportion of GM to the differences in yield but this was not repeated for spring crops. Box plots for the

4 crops without wild headlands are very similar to Fig 4.3 as expected, but with wild headlands there is greater variation as a result of the savings in inputs. For spring crops median values are similar, while there are substantial differences for winter wheat and winter barley. Significant terms in the financial model were: wild headland, year and the interaction between the past history of wild headlands and crop. The model explained 37% of the variation Model: ($F_{71,9}$ = 4.369; p<0.001; Fig 4.4).

Fig 4.4 illustrates the impact of wild headlands on GM in the headland in the 4 crops in the study.

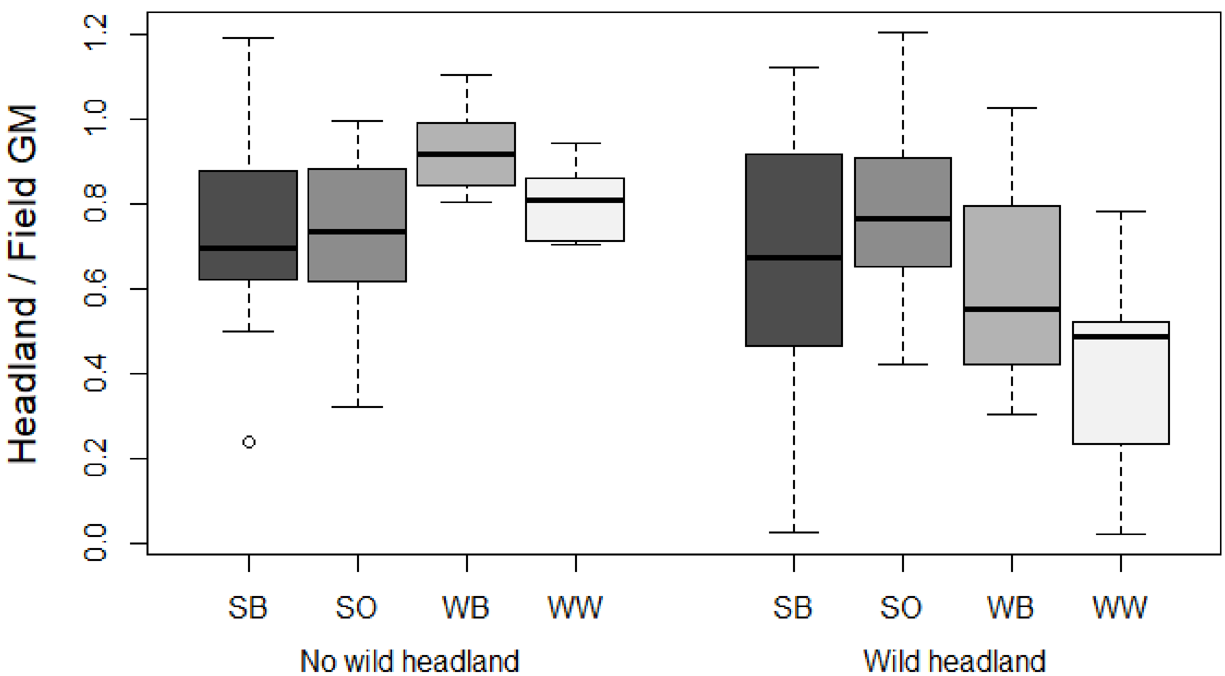

Fig. 4.4 Median headland GM as a proportion of field GM over two years for 4 crops without and with a wild headland. SB= spring barley, SO= spring oats, WB = winter barley and WW = winter wheat. The outlier “o” lies > 1.5 interquartile ranges beyond the bottom of the spring barley box. The Upper and Lower whiskers = 1.5 x interquartile range beyond the top and bottom of the boxes. The top and bottom of the boxes = upper and lower quartiles. For clarity other terms included in the model are not shown in the box plot. Cropping details per Table 4.1

The summary of the financial information for headlands of 82 fields with and without wild headlands combined over 2014 and 2015 is given below (Table 4.2). GM was calculated for fields and headlands by crop, with the figures for the differences in red. GM was lower for all crops in headlands compared to the field average, but after allowing for input costs, GM was similar in wild headlands for spring oats and spring barley. For winter cereals, the GM was lower in wild headlands compared to GM in conventional headlands.

Crop	Whole Field GM ha^{-1}	Wild headland difference in GM ha^{-1} to fields	Number of Fields in sample	Conventional headland difference in GM ha^{-1} to fields	Number of Fields in sample
Spring barley	£416.45	£131.02	12	£111.38	11
Spring oats	£831.11	£191.44	16	£231.80	16
Winter barley	£541.44	£203.23	4	£41.38	7
Winter wheat	£717.11	£366.55	9	£170.12	7

Table 4.2 Reduction in GM ha^{-1} in wild and conventional headlands from field GM ha^{-1} for 4 crops over 2014 and 2015 combined. Numbers of fields are given. Individual field details and GM calculations are available on request. *n* fields = 82.

Year was a significant term in the financial model largely as a consequence in variation of cereal prices and the changes in the cost of nitrogen fertiliser over the two years. The variation explains the poor explanatory power of the model compared to the model for yield alone. The financial figures are derived from a complex interaction of yield, price and cost savings which apply to each field separately for each crop depending on the inputs used, which were not standardised in this “real world” experiment. Table 4.3 gives relevant prices over two years for crop sales and fertiliser cost.

Crop	**2014** **£ per tonne**	**2015** **£ per tonne**	**Change** **£**	**Change**
Winter Barley	113	102	-9	-9.7%
Winter Wheat	142	133	-29	-20.42%
Spring Barley	114	121	+7	6.1%
Spring Oats	163	138	-25	-15.3%
Nitrogen Fertiliser	200	220	+20	

Table 4.3 Prices per tonne for 4 crops and nitrogen fertiliser in 2014 and 2015 with changes in price per tonne and % change from 2014 – 2015.

4.5 Discussion

From the evidence in this study, in the absence of nitrogen fertiliser, crop yield is substantially reduced. Fig 4.3 shows that in winter crops wild headland yield is ~50% of field yield. In spring oats, the ability of the crop to scavenge nutrients (Watson and More, 1956) has meant median wild headland yield for spring oats was 70% of field yield, but for spring barley, particularly where there was a combination of past intervention (high category), a wild headland that year and compaction, yields were much reduced. A single spring barley field for example had almost no crop and was excluded from the financial model so as to maintain the model integrity, although the decision had no impact on any conclusion formed from the analysis.

The results from the financial model, illustrated in the box plot in Fig 4.4, show a more nuanced picture than the yield model. Year was a significant factor in the financial model which it hadn't been in the yield model, explained in part by the considerable differences in cereal prices for individual crops between years illustrated in Table 4.3. Had the cost been only forgone output (yield x price) without savings in input cost, the cost to the farmer because of the reduced yield of wild headlands would have been much greater. There is consistency too in the size and range in the box plots in Fig 4.3 for yield for conventional headlands compared to box plots in Fig 4.4 for GM in conventional headlands – as expected – but for the box plots for crops with wild headlands in Fig 4.4 there is greater variance, particularly for spring barley. Here the impact of savings in inputs costs interacting with changing yield and price has given a greater range of outcomes for individual fields. The complex interaction between yield, price and variable costs (the summary shown in Table 4.2. Full detail available on request) explain the poor explanatory power of the model. Although there were 82 field/headland combinations in the study, field treatments were in response to agronomic drivers (the need for lime etc.) so were very varied. A randomised, replicated study where treatments, output prices and input costs could be standardised would possibly demonstrate a more conclusive outcome.

Table 4.2 gives the cost of wild headlands ha^{-1}. In England harvested conservation headlands *per se* are no longer funded under Countryside stewardship (Countryside stewardship. Gov.uk anon 2020) but Low input harvested cereals, a near equivalent, receive £266 ha^{-1} and unharvested spring sown cereal headlands receive £640 ha^{-1}. In both cases additional management is required, increasing complexity for farmers through alterations to crop rotations and additional field operations.

In the full data set (available on request) and the box plot in Fig 4.4, there are outliers where weed competition has been excessive and there is almost no crop. Careful evaluation by the farmer of the potential weed burden on each headland before committing to a wild headland is a sensible precaution if such losses are to be avoided. On balance, while average yields are lower, a thin crop canopy, a desired outcome of conservation headlands (Potts, 2012), is maintained through limiting the weed burden with herbicides and fertilising headlands in intervening years.

The opportunity cost to farmers of wild headlands has been reduced since 1992 by the change in subsidy regime. When originally conceived economic support for arable farmers was in the price of cereals, so research into conservation headlands was focused on maintaining yield through herbicide manipulation (Sotherton, 1991). Subsequently, the Macsharry reforms and arable area aid (Cunha and Swinbank, 2011) switched some of the support away from price support to a payment per ha for eligible crops. This reduced the opportunity cost of wild headlands as ~25% of gross output for the crop was unaffected by yield. Under the present system entitlements based on eligible land (which includes cropped ground) trigger Basic and Greening payments, so while not subject to the precisely the same rules (c.f Arable Area Aid and Single farm payment rules), the principle of payment for land area continues.

The study shows (per Table 4.2) that headland GM in conventional crops is lower than average field GM (though with variation between crops), consistent with the findings in Wilcox *et al*. (2000). This headland effect is known and Sparkes *et al*. (1998) posited that as a result permanent headland set-aside was cost effective instead of whole-field set aside. The widespread adoption of yield monitoring on combines in recent years has demonstrated to farmers differences in yield as a result of compaction, with consequent amelioration of farming practice (e.g. low ground pressure tyres, minimising cultivation passes, controlled traffic) and remediation through subsoiling (Hamza and Anderson, 2005). On the study farms the contractor is aware of compaction and equipped to manage it, headland compaction although not explaining variation in the model on crop yield, remained statistically significant. (Box plot in appendix 4.1)

4.6 Conclusion

Pywell *et al*. (2015) examined the net cost/benefit on a whole farm of adopting agri-environment prescriptions at the field edge under Higher Level Scheme (HLS) (Natural England, anon). They found that ecological intensification at the field margin increased farm profitability overall, particularly through the impact of pollinators on field beans. In an early study in winter wheat crops invertebrate density and diversity at the field edge was 20 times that in the middle (Potts, 1986), which encouraged the GWCT to develop conservation headlands at the field edge. On the farms in this study wild headlands emplaced at the edge of cereal fields are easily incorporated into the farming system at a modest cost. Wild headlands provide a cost-effective technique for agencies to deliver environmental objectives within an intensively managed arable landscape. They have the advantage over other AES prescriptions in that they were developed and practiced for many years in a commercial setting.

4.7 ACKNOWLEDGEMENTS

The authors are grateful to the farming clients of The Sandstone Farming Company for permission to use their farms for this study. This research did not otherwise receive any specific grant from funding agencies in the public, commercial, or not-for-profit sectors.

5 ARABLE PLANT COMMUNITIES AFTER 20 YEARS OF MODIFIED CONSERVATION HEADLANDS.

Abstract

In the 1980s The GWCT pioneered conservation headlands. Although supported by AES they were never widely adopted in Europe, principally through farmer's dislike of weeds in their crops. On a farm in Eastern Scotland, (Lat 58º N, Long 2.50º W), the GWCT prescription was modified in the mid-1990s to exclude fertiliser, which cut down weed problems and "wild headlands" were thereafter rotated intermittently around cereal fields for 20 years. This study looked at the seedbank of headlands which had been wild headlands and found that after 20 years seedbank weed populations had been restored to levels found in the 1970s. After allowing for % sand in fields, these headlands had a greater species richness and diversity than untreated headlands: exp(*H*_Shannon) and S were significantly positively correlated with wild headlands. The results have implications for restoring in-field biodiversity and delivering a wide range of ecosystem services on farms.

5.1 INTRODUCTION

Agricultural intensification in the UK post-war has resulted in improved yield in arable crops, but commensurate with the increased agricultural output has been a decline in habitat heterogeneity (Benton *et al*., 2003; Richards *et al*., 2018), a loss of resource (habitat and food) for farmland birds (Aebischer *et al*., 2003; Holland *et al*., 2006; Vickery *et al*., 2009), and consequent drop in UK farmland bird population Gov.UK (2018). In the case of those farmland birds which primarily depend on phytophagous invertebrates to feed their young; e.g. Turtle dove (*Streptopelia turtur)*, Corn bunting and Partridge, the decline has been particularly acute (Krebs *et al*., 1999; Perkins *et al*., 2011; Potts 2012), with one cause the reduction in arable weed flora through the widespread use of effective herbicides (Mayor and Dessaint, 1998; Marshall *et al*., 2003). The loss of this weed flora has wider impacts as these arable plants are an important source of biological diversity, contributing substantially to ecosystem function and critical to the effective functioning of food webs (Hawes *et al*., 2010). The decline in weed flora is mirrored in seedbank decline, with mean seedbank density in the U.K from 1960 to 1990 reducing from >10,000 m^{-2} to 3,000 – 5,000 m^{-2} (Hawes *et al*., 2005). Over this period there was also a shift in weed communities at the expense of broadleaf weeds in favour of grass species, with grass species increasing from 10% to > 50% as a proportion of the total seedbank (Squire, 2017).

In the 1980s the GWCT, building on German work from the 1970s designed to protect rare arable weeds (Schumacher, 1980), developed conservation headlands in order to enhance the availability of resources for farmland birds. Conservation headlands are selectively sprayed headlands within cereal crops where pesticide applications are modified to maintain a population of broadleaved weeds as host plants for phytophagous chick-food invertebrates. The concept was developed and applied progressively in the 1980s (*cf*. Rands 1985; Boatman *et al*., 1999) within the UK. The technique depends on the management of the outer 6 - 10 m of the crop while maintaining fertiliser applications and crop yield. The outer boom section of the sprayer (usually 6 - 7 m) is switched off by the operator when certain broadleaved herbicides are being applied to the rest of the field and insecticides are not applied after 15th March in any year (Sotherton, 1991). Conservation headlands have been included in AES across Europe with funding to farmers based on the opportunity cost of forgone yield (Walker *et al*., 2007; Albrecht *et al*., 2016), with notable success in improving brood size through better chick survival in partridge (Rands, 1985; Chiverton,1993). However, despite the evidence supporting them, conservation headlands have never been widely taken up in the UK (Clothier, 2013) or in Germany (Albrecht *et al*., 2016), attributed in part to farmers' dislike of the weeds which flourished in arable crops with full fertiliser and no herbicide (Storkey and Westbury, 2007).

To overcome the weed problems found with conservation headlands an alternative approach, called a wild headland, was developed on an arable farm in Eastern Scotland (Lat 58º N, Long 2.50º W) in the early 1990s. (For a full description of the wild headland please refer to the general methods chapter.) The location of the farm where wild headlands were developed and the study farms is shown in Fig 5.1

Fig. 5.1 Study farms and their position in the maritime farming area of East Scotland. Dundee is to the North and Edinburgh to the South. Farmworks 2020

Arable seedbanks record changes in the farming system over the long term and are affected by seed rain from year to year (Heard *et al*., 2003). Perhaps their greatest significance is their role in determining future vegetation, particularly after natural or deliberate perturbation (Roberts, 1981). The primary aim of this study was to examine the long-term effect on weeds in the arable seedbank where wild headlands had first been used and thereafter expanded across neighbouring farms over the last 20 years. We also investigated changes to seedbank population between years in the presence/absence of a wild headland to test for any between-year impact of wild headlands on seedbanks.

To achieve our primary aim, we first examined soil physical and chemical properties, field size, cropping history and margin type which might influence patterns of seedbank diversity. Thereafter, to understand how these environmental factors and wild headlands jointly shape alpha diversity (α – the number and relative abundance of species) we examined seedbank populations of arable weeds at the crop edge. Finally, to explore these influences on beta diversity (β – species composition) we ran a diversity analysis on the seedbank data collected. For our secondary aim, to investigate over-year changes in seedbank population, we examined seedbanks over two years before and after a wild headland treatment. The outcome of this work may have implications for using wild headlands to maintain and restore within-field arable biodiversity in the long term.

5.2 METHODS

5.2.1 Study sites

The study sites comprise the headlands of 25 fields between 4 and 16 ha across 4 farms from our study area. (For a general background to our study area, please refer to the general methods chapter).

The location of the study fields, numbered by farm, is given in Fig 5.2

Fig. 5.2 Study fields are highlighted in pink and the different farms are numbered: Gilston & Lathallan = 1, Kilconquhar = 2, Easter Pitcorthie = 3 and Balcaskie = 4. The blue line denotes the approximate boundary of the two bio-climatic zones. Farmworks 2020.

Wild headlands were first developed at Gilston (1), introduced shortly thereafter to Lathallan (also numbered 1), an adjacent farm in the same bio-climatic zone (zone A) and intermittently thereafter to two farms (2 & 3) on the coast (zone B). Balcaskie (4), introduced wild headlands in 2014.

5.2.2 Sampling and Experimental design.

In order to identify the long-term drivers of seedbank community and population 25 cropped headlands from the north side of arable fields were selected for sampling in February 2014 across the five farms. To test for changes in seedbank population after a farming year, the same field headlands were re-sampled in 2015. The sites chosen for the study were divided into Gilston and Lathallan (Farm 1) with a long history of past wild headlands (as described above) during the period from 1995 to 2013, the two adjacent coastal farms where wild headlands from 1995 – 2013 had been absent or less frequent (Farms 2 & 3) and Balcaskie (Farm 4) which only adopted wild headlands at the onset of this study. As well as sites with different wild headland histories, the sites covered different bio-climatic zones and different soil types. The distribution of fields by farm and environmental factors is given in Table 5.1 below, with numbers showing the distribution of fields in each category.

Farm	**Fields (n)**	**SaLo**	**SaSi Lo**	**Bio-zone A**	**Bio-zone B**	**1995 - 2013 WH 0**	**1995 - 2013 WH 1**	**1995 - 2013 WH 2-3**	**1995 - 2013 WH 4-6**
Farm 1:GL	10	4	6	10	0	0	0	3	7
Farm 2: KL	(4)	3	0	0	3	1	0	2	0
Farm 3: EP	5	4	1	0	5	1	3	1	0
Farm4: BAL	(6)	4	1	3	2	5	0	0	0
Total	(25)	15	7	13	10	7	3	6	7

Table 5.1 Fields by farm, soil type characterised by their % of sand, silt and clay; sandy loam (SaLo) and sandy silt loam (SaSiLo), bio-climatic zone, and incidence (*n* occasions) of wild headlands from 1995 - 2013. Numbers in brackets indicate where fields have been lost from the data set through sampling errors. Full details of fields and supplementary analysis in appendices; 5.1, 5.2, 5.3, 5.4

So as to allow for different crop phenology in the between – year seedbank experiment, the original sampling design dived these fields in 2014 into 10 growing winter barley; 5 wild headlands and 5 controls and 15 growing spring barley; 10 wild headlands and 5 controls. It meant field choice was determined by where the crops were being grown. Numbers of fields from Table 1 in each category are given in Table 2.

	Fields (n)	**WB (C)**	**WB (WH)**	**SB (C)**	**SB (WH)**
Farm 1:GL	10	0	5	0	5
Farm 2: KL	(4)	0	0	1	(3)
Farm 3: EP	5	2	0	2	1
Farm 4: BAL	(6)	3	0	(2)	1
Total	(25)	5	5	(5)	(10)

Table 5.2 2014 Crop growing in fields (or planned) when seedbank sampling was carried out; WB is winter barley; SB is spring barley. (C) Fields farmed with a full range of pesticides and fertilisers in 2014 (WH) wild headlands in 2014 with limited pesticides and no fertiliser. Fields in brackets indicate fields lost from the original data set through sampling errors.

5.2.3 Seedbank sampling

Seedbank sampling took place in February each year before spring crops were sown and five months after the sowing of winter crops but before any weed seed had been shed. Standardised methodology for evaluating seedbanks (Hawes *et al*., 2010) was used as follows: six samples were taken in each headland 3 m into the crop and at 20 m intervals about the middle of the north side of each field. Soil was dug to plough depth (20 cm) within a 50 cm quadrat, carefully mixed in a bucket and a representative c.2 L sample bagged up and transferred to an un-heated greenhouse. Soil samples were sieved though a 10 mm sieve and divided into subsamples: 25 cl retained for later analysis of soil chemistry and c.1.2 L placed in 21 x 15 x 4 mm plastic seed trays, levelled off and consolidated. The remainder was discarded. Insufficient soil had been collected from one field, so it was not included in the study. The remaining 144 seed trays were distributed randomly on slatted benches in an un-heated greenhouse and hand watered as necessary to maintain a moist seed bed. The quantity of soil was similar to that used in early studies (Roberts and Chancellor, 1986) and in other more recent studies at the James Hutton Institute (Heard *et al*., 2003; Hawes *et al*., 2010).

During the next 4 months emerging seedlings were identified as far as possible to species, removed and their number recorded. Four groups of plants were difficult to identify to species at the seedling stage and were amalgamated into groups: *Poa* spp*,* grasses other than *Poa* spp, *Matricaria* spp. and *Epilobium* spp. Species identified were allocated a functional group number based, amongst other variables, on seed size, germination timing and growth habit (Hawes *et al*., 2009). In early June, after a 2-week period with no further seedling emergence, the soil was allowed to dry out, re-sieved and the germination and recording procedure repeated for the second flush. This germination method, while it may not capture all seedlings which can germinate for up to two years with occasional disturbance, gives a reasonable estimate of seedbank population for comparison purposes (Heard *et al*., 2003).

For our investigation into changes in seedbank population over two years after the weed seed shedding period in summer and autumn 2014 and vernalisation over winter, 23 of the 25 fields sampled in 2014 sampling was repeated at the same sites the following year (February 2015). 10 of the headlands had received herbicide and fertiliser in 2014, while 15 of the headlands were wild headlands. Two fields sampled in 2014 were rejected through sampling error and a further withdrawn because of uncertainty over past cropping. In 2015 soil from 4 of the 6 sample points (Nos 1, 3, 4 and 6) in each headland was collected. The samples were processed as before, the seed trays again distributed randomly in the same un-heated greenhouse and the 2014 experimental protocol repeated.

5.2.3.1 Soil chemistry and habitat factors

2014 soil samples amalgamated by field were analysed by third party commercial laboratories for available Phosphate (P) using the Olsen method (Valentine *et al*., 2012), available Potassium (K) extracted with ammonium nitrate and the solution assessed with adsorption flame photometry, Soil Organic Matter from loss on ignition and pH in a 1:2.5 soil/water solution. Additionally, soil texture was measured by placing soil samples in solution and through laser analysis determining the constituent fraction of sand, silt and clay (Lancrop Laboratories). Environmental and habitat factors recorded for use as co-variants in the analyses, in addition to soil chemical and physical properties, were: incidence of past wild headlands, farm, field size, margin width (to boundary), margin composition and bioclimatic zone per Birse and Robertson (1970). Crops grown for the years 2009 to 2013 were recorded to give two intensity measures for use in later multivariate analyses. (Table 5.1 with detail in the appendicies 5.2 & 5.3)

5.3 STATISTICAL ANALYSES

5.3.1 Seedbank diversity from past use of wild headlands and habitat factors.

To compute dissimilarities in species assemblages between headlands, a pair-wise comparison between headlands was made using Bray-Curtis dissimilarity indices measuring relative abundance of each species in each headland:

$$BC_{ij} = 1 - \frac{2C_{ij}}{S_i + S_j}$$

where $C_{i,j}$ is the sum of the lower of the two abundances of all specimens for only those species in common between headlands in each pair of headlands and S_i and S_j are the total number of specimens of species counted at each headland *i* or *j*. (Magurran, 2004).

To identify groupings of sites in terms of their species composition the solution from each pair-wise combination for all 23 fields in the 2014 data set (with each analysed against every other) was used in two analyses; first, a hierarchical cluster analysis using a general agglomerative hierarchical clustering (Ward “D2”) and second, in multidimensional scaling to examine the placement of these sites relative to one another. A global non-parametric multidimensional analysis, metaMDS (Minchin, 1987) (so particularly useful for species abundance data (Magurran, 2017 *pers comm*)) was constructed fitting environmental factors as co-variants to the 2-dimensional NMDS plot (Kruskal 1964). We first used Bray-Curtis for a distance measure to separate species assemblages, which given abundances > 50 were transformed using Wisconsin double square root (Faith *et al*., 1987). These were included in an ordination of multi-dimensional space, which was then arranged in an iterative way (…trial and error) to maximise the rank-order (i.e non-metric) correlation between real-world distance and Euclidean distance (i.e. straight-line distance) in ordination space. We used 3 dimensions (k=3) for our analysis as stress (goodness of fit) was greater than 0.2 with 2 dimensions. The analysis was repeated 9999 times to eliminate random errors. Environmental variables were then fitted to the plot, which enabled us to assess the environmental variables that provided the best explanation of the patterns of sites (fields) shown in the plot. (R Package Vegan, Oksanen *et al*., 2019).

In order to analyse weed seed abundance in headlands with and without wild headlands we measured alpha biodiversity (α) in several ways: S or species richness - the number of different species seen at a point in space or time, N or abundance - the total number of individuals counted (across all species) at a point in space or time and exp(*H*_Shannon) - or Hill number 1.,which takes into account both richness and abundance (numbers of species as well as abundances of species) (Jost *et al*., 2010) and is often used in analyses of emerged weeds and seedbanks (Hawes *et al*., 2010). We also looked at species dominance, particularly *Poa* spp. Per the importance of soil characteristics from the results of the NMDS analysis, we performed an ordinary least squares regression and plotted the diversity indices against a Z score of the proportion of sand in each sample [$z = (x-\mu)/\sigma$, where x is the raw score, μ is the population mean, and σ is the population standard deviation]. We subset the data into fields that have had wild headlands and fields that did not and examined model parameters. We used R statistical package 4.02 (R core team 2020) for the analysis.

5.3.2 Seedbank population from past use of wild headlands.

The impact on seedbank population of periodic use of wild headlands from 1995 – 2013 was assessed by an analysis of variance with past wild headlands the factor and seedbank population the dependent variable. (R Core Team 2020). Seedbank data were log transformed to give a normal distribution and the model run with the dependent variables: monocots and dicots combined, monocots, or dicots alone. The models were tested using a Wilcoxon signed rank test.

5.3.3 Determining seedbank replenishment between years.

To determine the impact of treatments (wild headland or conventional headlands with herbicide applied) in 2014 on recruitment to the seedbank in 2015 an analysis of variance of seedbank population was used with wild headland the previous year as the factor (R Core Team 2020). Three analyses were undertaken with dependent variables; numbers of monocots, numbers of dicots and total seedlings m^{-2}, amalgamated from 4 plots from each field in 2014 and the same 4 plots in 2015, log transformed to conform to a normal distribution. Terms in the models were: wild headland in the previous year, year, crop and any significant interaction observed with year as a random factor. Models were compared using Akaike's Information Criterion and the successful model tested using a Wilcoxon signed rank test.

5.4 RESULTS

5.4.1 Seedbank replenishment

Seed rain in the absence of herbicide had been considerable giving rise to an increase in seedbank populations after wild headlands, while seedbank populations had declined in the conventional headlands. Crop the previous year (either in 2013 or 2014) was not significant in the assessment of seedbank change, only the presence or absence of a wild headland in the previous year. Table 5.3 shows the totals of monocots and dicots grouped by headland from the four plots in each field used in the 2015 study and from the same four plots in 2014. Mean headland seedbank populations in the 22 fields in the study increased from 4810 m^{-2} to 6405 m^{-2} after a wild headland in 2014 and declined from 2575 m^{-2} to 2053 m^{-2} in conventional fields which had received herbicide.

	Conventional fields before herbicide **n=12**	**Conventional fields after herbicide** **n=12**	**Fields before a wild headland** **n = 8**	**Fields after a wild headland** **n=8**
Sampling Year	2014	2015	2014	2015
Monocots per field m^{-2}	989	580	3184	3492
Dicots per field m^{-2}	1586	1473	1626	2913
Total m^{-2}	2575	2053	4810	6405

Table 5.3. Mean seedbank densities m^{-2} for monocots and dicots in 8 conventional headlands and 14 wild headlands sampled in February 2014 and again in February 2015. Note the high monocot population in fields before a wild headland in 2014. See text for comment.

The effect of wild headlands the previous year on numbers of monocots was significant ($F_{37,6}$ = 4.968; p < 0.001) and monocots and dicots combined ($F_{42,1}$ = 5.418; p = 0.025; Fig 5.3). Dicots alone showed no statistically significant effect of a wild headland in 2014. Fig 5.3 Shows weed seed abundance with wild headland the previous year as factor.

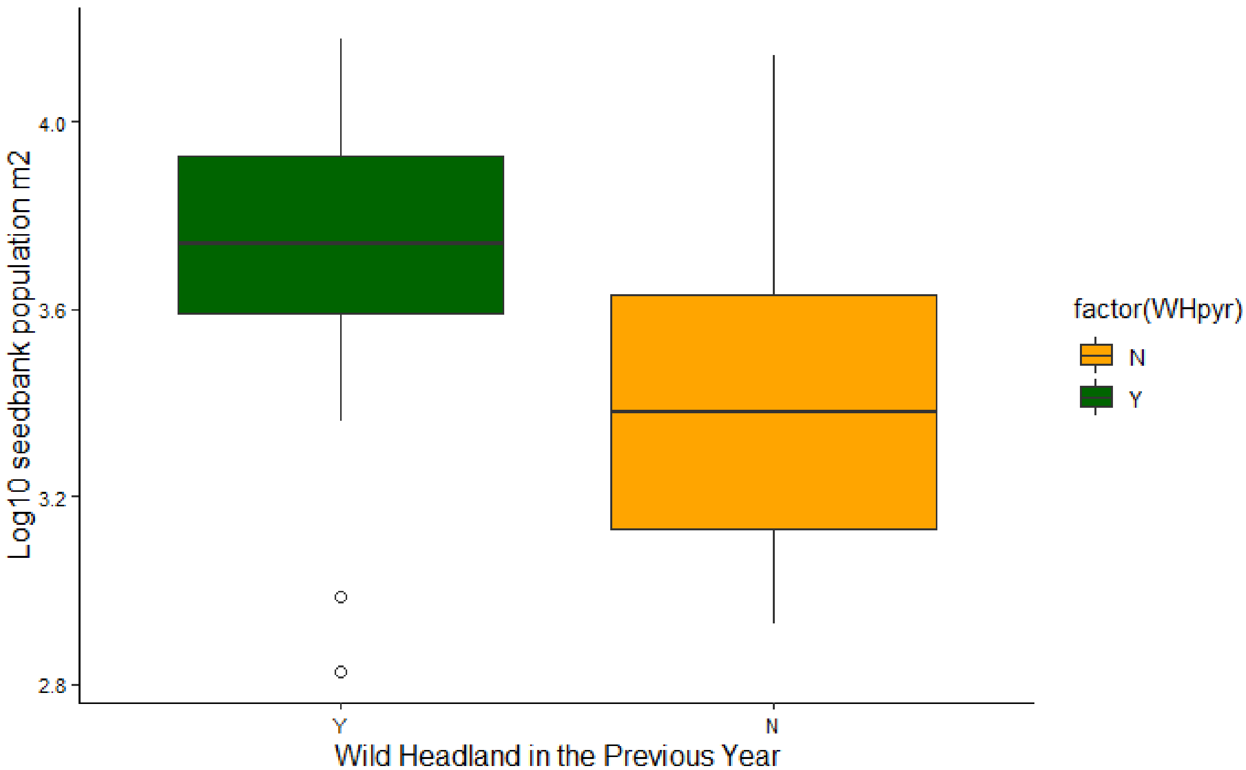

Fig. 5.3 Median values of weed seedbank abundance (monocots and dicots combined) with wild headland the previous year as factor from 22 fields: 14 wild headlands and 8 conventional.. The outliers "o" lie > 1.5 interquartile ranges beyond the bottom of the box. The upper and lower whiskers = 1.5 x interquartile range beyond the top and bottom of the boxes which delineate upper and lower quartiles. Factor (Y), fields which have had wild headlands the previous year, (N) fields which have not.

5.4.2 Seedbank population after 19 years.

Intermittent use of wild headlands over the past 19 years had impacted seedbank populations which in our study were high compared to recent studies. In the analysis of the 2014 seedbank data the factorial ANOVA grouped by fields with no previous wild headlands (n=7) and any wild headlands from 1995 – 2013 (n=16) monocot number ($F_{21,1}$ = 5.398; p = 0.03) and total seedling number ($F_{21,1}$ = 5.68; p =

0.026; Fig 5.4) were both above seedbank populations in recent studies carried out at the James Hutton Institute. A boxplot showing the seedbank population after 19 years for fields which have had past wild headlands and for fields with no history of wild headlands is shown in Fig 5.4 Median seedbank populations with no history of wild headlands over the period were 1800 m^{-2} and seedbanks with a history of wild headlands between 1995 and 2013 had a median of 4600 m^{-2}.

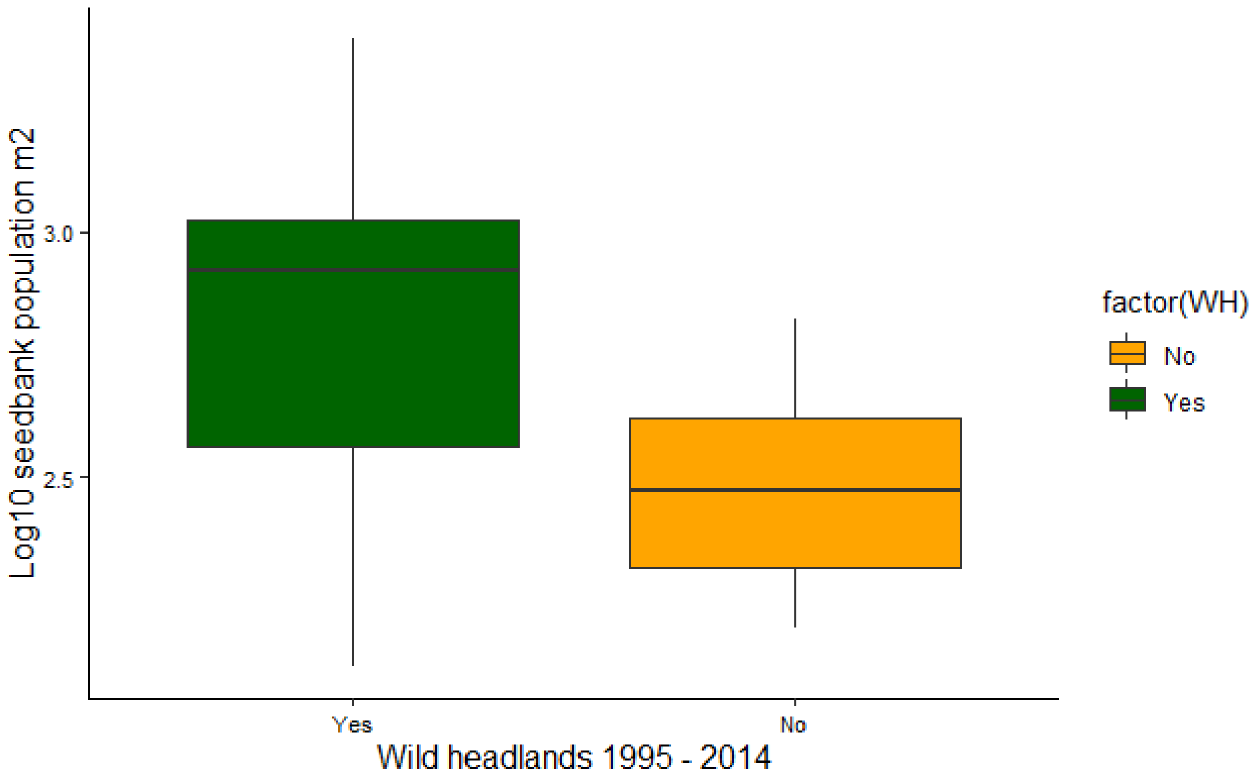

Fig. 5.4 Median weed seedbank abundance in headlands (monocots and dicots combined) after 19 years for headlands with and without wild headlands in the past. The Upper and Lower whiskers = 1.5 x interquartile range beyond the top and bottom of the boxes which delineate upper and lower quartiles. Factor (Y), fields which have had wild headlands, (N) fields which have not. Note Log scale.

5.4.3 Seedbank species assemblages after 19 years.

The analysis of species assemblage by field showed no clear pattern on first inspection, but detailed analysis and further interpretation showed a clear effect on contemporary seedbank weed assemblages on the history of past intervention. The hierarchical clustering shows in Fig 5.5 shows which field species assemblages are most closely allied with each other. Fields in the cluster dendrogram are coloured by intervention history: Past wild headlands (green) and no history (orange) and coded by farm. The fields are spread across farm, bioclimatic zone, soil and intervention history and no pattern is obvious. There are some fields from farms on the same branch (identifiable by their coding), but there appears to be no connection between the location of fields on the dendrogram and field history. There are two pairings, BAL_2 & BAL_4 and BAL_3 and BAL_5 adjacent on the ground and adjacent on the dendrogram, which are discussed later.

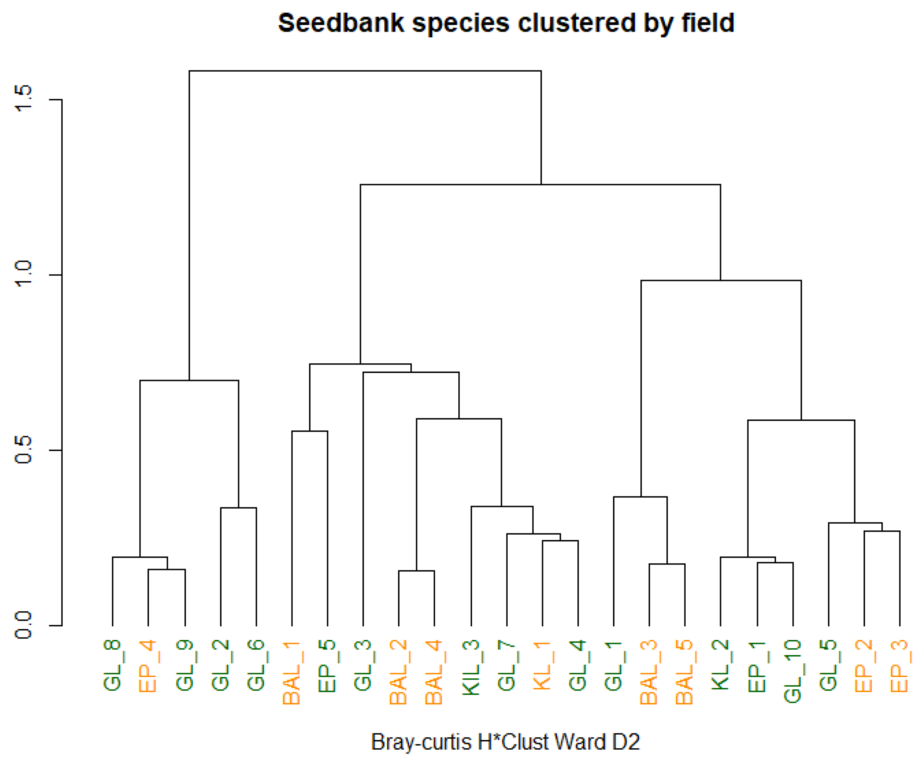

Fig. 5.5 Hierarchical cluster dendrogram using ward D2 clustering and Bray-Curtis dissimilarity measures of species in the 2014 seedbank. Fields are coded by farm. (GL Lathallan; EP, Easter Pitcorthie; KL, Kilconquhar and BAL, Balcaskie). Colouring indicates wild headland history: with (green) without (orange). Fields are numbered and full information is in the S.I. Note the pairings BAL_2 & BAL_4 and BAL_3 and BAL_5 which are referred to in the later discussion.

The environmental variables detailed in appendix 5.2 and listed in Table 5.3 were used in the NMDS (Fig 5.6). Our analysis using 3 dimensions (k=3) gave a reasonable goodness of fit between observed dissimilarity and ordination distance. Stress, the measure of goodness of fit, was 0.15, which is considered acceptable. We have supplied the stress plot (a shepherd plot) in appendix 5.5.

Levels above 0.2 indicate that it is difficult to make predictions from the distribution of sites (in this case, fields) on the plot (Oakensen *et al*., 2019).

Output from the NMDS (appendix 5.6) showed that sand was significant and explained 40% of weed species composition. Only P, K, Mg, % silt, % clay and pc texture (an agglomeration of sand, silt and clay) were also significant. Other environmental factors, including farm and indicators of cropping intensity, were not significant (Table 5.4).

Environmental Factor	***Significance (p values)***
Farm	NS
Past Cropping (pc measure of cropping and intensity	NS
Wild headlands	NS
Bioclimatic zone	NS
Margin width	NS
Field size	NS
Ph	NS
K	0.02
Mg	0.03
P	0.04
Organic matter	NS
% silt	0.02
% Sand	0.006
% Clay	0.001
Soil texture (PC score: sand:silt:clay)	0.002

Table 5.4 Environmental factors included in the NMDS analysis (Fig 5.6) with significance given. Environmental factors in appendix 5.2 Calculation of intensity scores is in appendix 5.3 and an example of the soil analysis is in appendix 5.4. The output from the NMDS giving the p values is in appendix 5.6

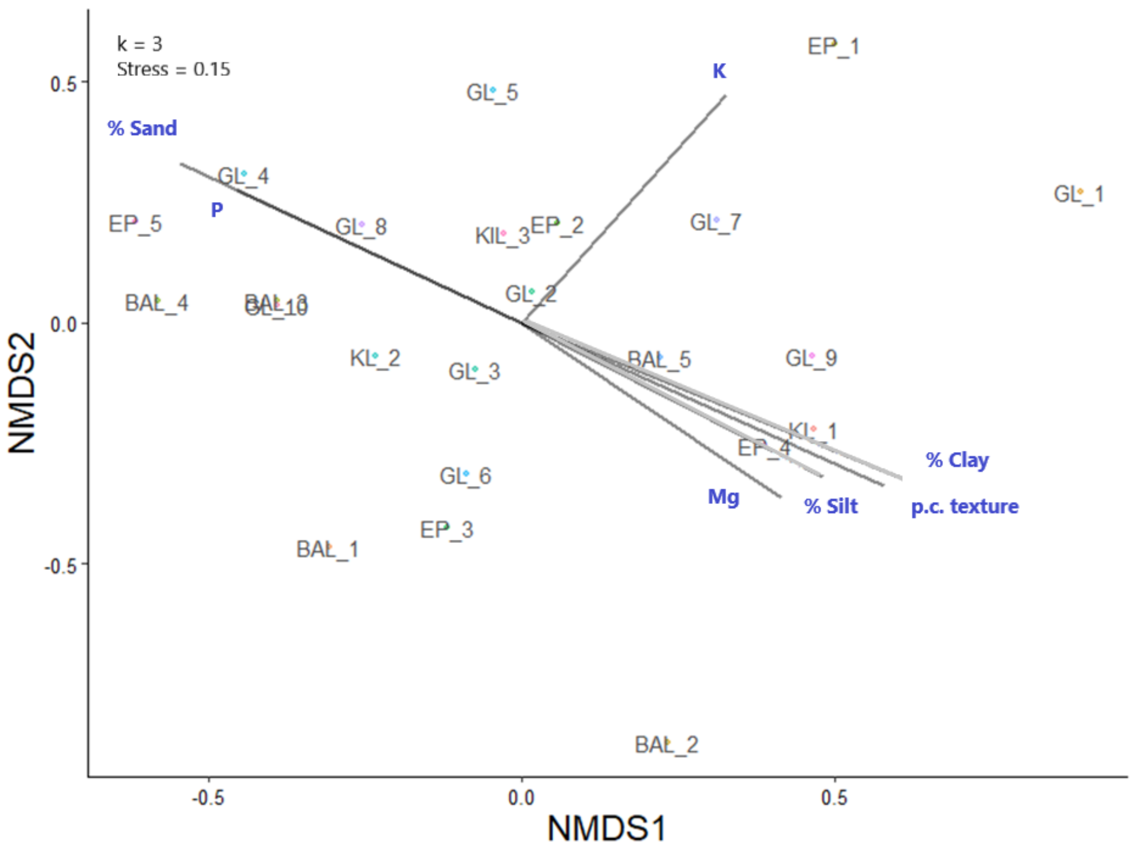

Fig. 5.6 3-dimensional output from the NMDS analysis displayed as a 2-dimensional plot showing environmental variables fitted to the output from the pair-wise Bray-Curtis analysis of species assemblages in seedbanks for 23 fields from the 2014 seedbank data set. Sites are numbered field codes: BAL; Balcaskie, GL; Gilston & Lathallan, KL; Kilconquhar and EP, Easter Pitcorthie. See Table 5.3 for detail of vectors (labelled blue) used in the analysis. k = 3, Stress = 0.15 (Oakensen *et al*., 2019)

In light of the very strong effect of soil shown in the NMDS we looked at diversity analyses of species richness, abundance and evenness in the weed assemblages allowing for sand. We examined the distribution of fields with and without a history of wild headlands based on the 2014 analysis of soil constituents (sand, silt & clay). A ternary plot (Hamilton and Ferry, 2018) showing the distribution of fields is given in appendix 5.7. There is no bias in the distribution of fields in the plot between those with and without a history of wild headlands.

Once we had accounted for sand, the degree to which wild headlands were responsible for differences in weed assemblages could be teased out. We found that wild headlands are having a significant effect on weed assemblages, both in species richness alone, species dominance and in species richness, evenness and abundance. We found S (species richness) and exp(*H*_Shannon) increased significantly when looking at the presence/absence of wild headlands when plotted against the Z score for sand. For S, ($F_{11.1}$ = 11.71*;* p = 0.005702; Fig 5.7) and exp(*H*_Shannon) ($F_{11, 1}$ = 11.08; p = 0.00672; Fig 5.8). After allowing for sand, dominance of *Poa* spp in wild headlands declined when compared to fields without intervention ($F_{11,1}$ = 9.816; p = 0.009527; Fig 5.9). Fields without a history of wild headlands were not significant in any of the analyses confirming the role wild headlands have in shaping weed assemblages in the seedbank.

Species richness is plotted in Fig 5.7 and exp(*H*_Shannon) analysis of 23 fields from the 2014 seedbank is given in Fig 5.8 The plot showing the declining dominance of *Poa* spp is shown in Fig 5.9 Fields marked in green with green triangles are fields with a history of past wild headlands (Table 5.1). Fields with no wild headland history are marked with orange circles.

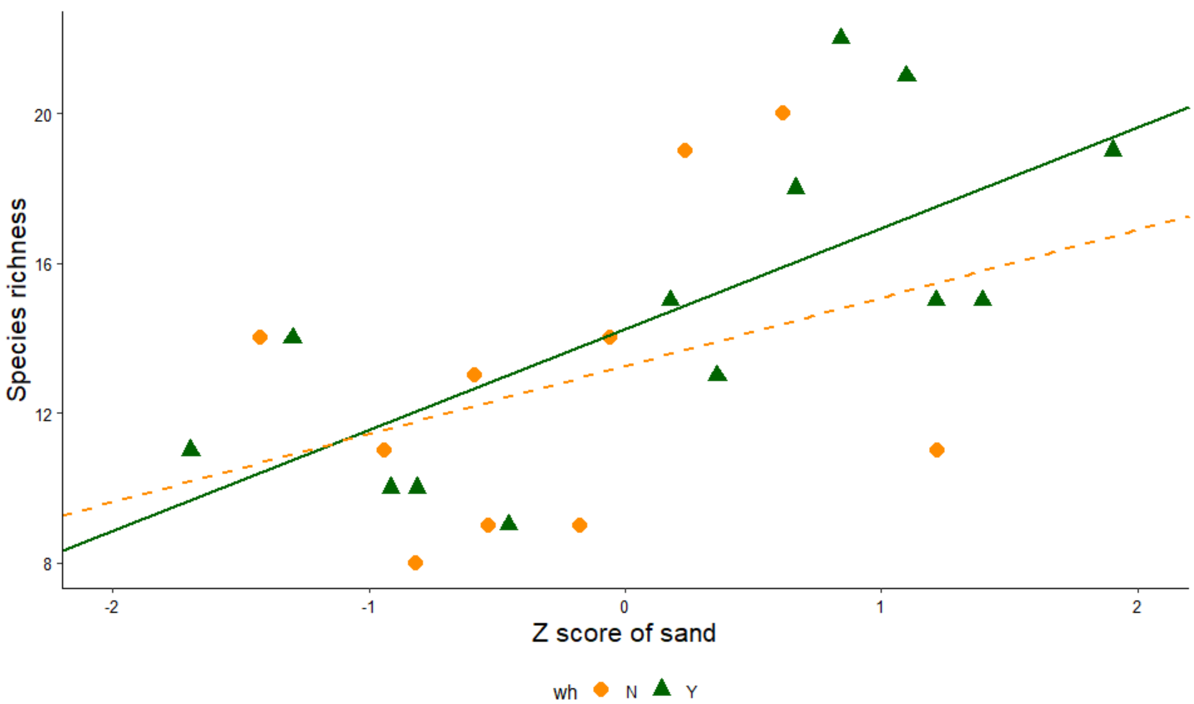

Fig. 5.7 Plot of species richness of seedbanks (S) against a Z score for sand for fields in the 2014 data set. Here the regression lines (solid and dashed respectively) and points are coloured according to wild headland or none (dark green & triangles and dark orange & circles respectively). Wild headlands in fields with a higher Z score are richer. The dotted line (fields without wild headlands) was NS.

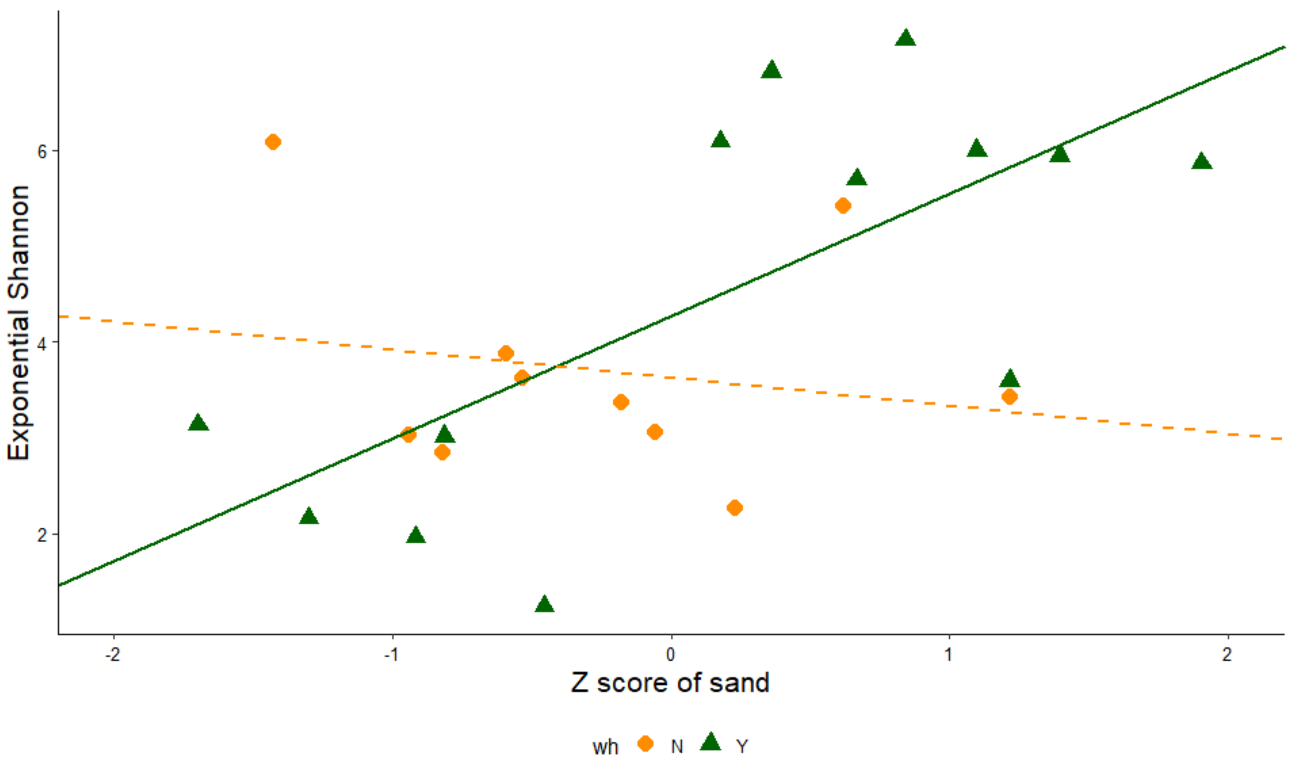

Fig. 5.8 Regression plot of exp(*H*_Shannon) diversity of seedbanks for fields with and without wild headlands over the previous 19 years plotted against a Z score of sand. The regression lines (solid and dashed respectively) and points are coloured according to wild headland (dark green & triangles) or none (dark orange & circles). Wild headlands in fields with a higher Z score have a higher score. The regression line for fields without wild headlands was NS.

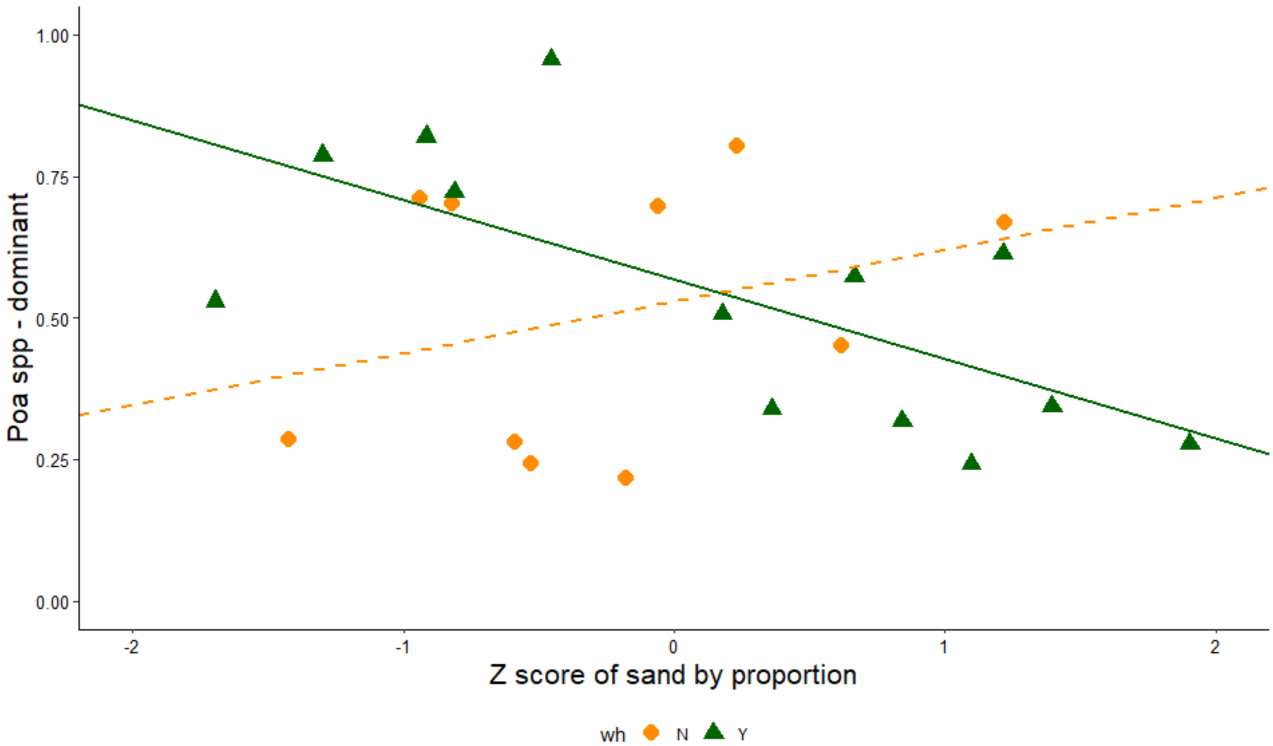

Fig. 5.9 Regression plot of *Poa* spp dominance for fields with and without wild headlands over the previous 19 years plotted against a Z score of sand. The regression lines (solid and dashed respectively) and points are coloured according to wild headland (dark green & triangles) or none (dark orange & circles). Wild headlands in fields with a higher Z score have a lower dominance of *Poa* spp in the seedbank. The regression line for fields without wild headlands was NS.

5.5 Discussion

The study was designed to test if wild headlands shifted species composition and altered species richness in the arable seedbank over time and if wild headlands were sustainable in the long term. The ANOVA looking at population in the seedbanks over the long term demonstrated that while seedbank populations were higher in fields after wild headlands, they were within reasonable limits. After allowing for the very strong signal of soil type in the analysis of β diversity (the Bray-Curtis analysis), this study has shown that wild headlands *have* changed seedbank composition over time. Not only is there greater species richness in fields with a history of wild headlands, but diversity analyses of community composition also showed increased species richness, evenness and abundance in fields with a history of wild headlands.

5.5.1 Seedbank species composition after wild headlands.

In Hawes *et al*. (2010), soil characteristics explained more variation in seedbank composition than past management, while available P and clay content were found by Andreasen and Skovgaard (2009) to be the major soil variables influencing weed communities. Fried *et al*. (2008), in their study of 700 fields in France noted species composition was influenced by soil texture with contrasts between basic clays and acidic sands. In our study, soil characteristics are equally important. The NMDS multi-variate analysis demonstrated very clearly that the chemistry and characteristics of soil were the most important drivers of seed assemblages in the seedbank. By plotting species richness and diversity against soil characteristics, we have demonstrated that with an increasing proportion of sand in soil samples, fields with a history of wild headlands have significantly increased species richness, abundance, evenness and diversity in seedbank assemblages compared to fields without that history (Figs.5.7 & 5.8). Fried *et al*. (2008) and Hawes *et al*. (2010) found that weeds assembled at the field scale as a result of crop rotation as well as soil chemistry and characteristics. Farms in this study have been subject to similar farming practices and rotation in the last 20 years, with the same balance of spring and winter cereals with predominantly winter oilseed rape as a break crop. Perhaps as a consequence, although we used two metrics, a pc score based on cropping over 5 years (Hawes *et al*., 2010) and another looking at intensity, neither demonstrated a similar effect of crop rotation to those found by Fried *et al*. (2008) or Hawes *et al*. (2010). Hawes *et al*. (2010) in their study also found farm type; defined by them as organic, IFM and “conventional”, drove species assemblages. In our study, all farms were IFM farms so the same farm type and “farm” we found had no effect in the analyses. Metcalfe *et al*. (2019), in an analysis of the well-known Field-scale evaluation (FSE) data set (Heard *et al*., 2003), found that constituents of the permanent field margin drove species assemblages in headlands. In this study however, field boundaries are similar across all farms. Marshall and Arnold (1995) in their study of a farm in Essex observed that boundary species were only occasionally found in the arable seedbank, as we did here such as *Epilobium* spp. in our study. Arable fields are a different ecotone to the field boundary and tend under climatic conditions in maritime Northern Europe to be dominated by ruderal species (Marshall and Moonen, 2002), as we found too.

Because fields in our study area shared very many environmental characteristics, we have been able to show that differences in weed assemblages were principally a function of individual soil characteristics and chemical properties of headlands, modified by the field history of wild headland intervention.

This is confirmed through two pairs of fields in the Bray-Curtis analysis shown in the cluster dendrogram (Fig 5.5). Where crop rotation, soil type, bioclimatic zone, farm and wild headland history are shared weed assemblages are very similar. Highlighted in the cluster dendrogram are pairs of fields with similar weed assemblages (pairings: BAL_2 & BAL_4 and BAL_3 and BAL_5). These particular paired fields adjacent in the cluster dendrogram are also adjacent on the ground. Through the tendency of farmers to block crop to facilitate spraying and harvesting (Long *et al*., 2014) these paired fields had been farmed together with identical management and none had any wild headland history. The two pairs have similar soil type and are the same (but different) bioclimatic zones, all of which have combined to give them very closely allied species assemblages. Where paired fields on the cluster dendrogram are not adjacent on the ground other factors are necessarily driving species assemblages. In our study, once we'd allowed for soil type, only wild headland history provided the explanation.

It may be that the wild headland technique mimics the effect found in Klein and van der Voort (1997) where unfertilised plots had a less dense crop canopy compared to fully fertilised plots due to the difference in nitrogen input. The associated increased light transmissivity from thinner crops enabled a wider spectrum of arable weed species to set seed (Klein and van der Voort 1997, Seifert *et al*., 2014). Increased light from reduced fertiliser and/or poor crop growth from compaction is typical too under normal circumstances 1 m – 2 m into a conventional crop (Wilcox *et al*., 2000). Often as a result the edge of cereal crops is the last refuge of rarer arable weeds (Marshall 1989; Wilson and Aebishcher, 1995). Wild headlands, covering up to 7 – 14 m of the headland, can over time maintain and extend a diverse seed bank (including occasional uncommon species such as *Spergula arvensis*) further from the outer crop edge into the field.

5.5.2 Sustainable continued use of wild headlands.

Dislike of weeds in arable crops has limited the take up of conservation headlands in Europe (Storkey and Westbury, 2007; Clothier, 2013). Without herbicide the build-up of weeds in a seedbank can be considerable. For example, Squire *et al*. (2000) showed that after 6 years of reduced herbicide in the TALISMAN project (Towards A Lower Input System Minimising Agrochemicals and Nitrogen), seedbank densities on some sites had increased up to 130,000 m^{-2}, dominated by a few ruderal species. In this study it's been demonstrated that herbicide use in the years between wild headlands has limited seedbank populations, even after 19 years intermittent use of wild headlands (Fig 5.4). In our comparison of seedbank population between 2014 and 2015, seedbank populations increased following relaxed weed management and decreased in the presence of herbicide (Table 5.2). It may be that the fields with wild headlands in 2014 are not true experimental controls as many had a high initial monocot (*Poa* spp) population at the onset of the experiment. With this caveat, the decline in the arable weed seedbanks in fields with herbicide applied between 2014 and 2015 are consistent with the findings in Lutman *et al*. (2002), Roberts and Neilson (1981) and Mayor and Dessaint (1998) where seedbanks declined under a modern herbicide regime. The seedbank decline is an important attribute of rotational wild headlands necessary to keep seedbanks in check. It would be interesting to follow the trajectory of these seedbank populations in future. Squire (2017) postulated that further decline in seedbank populations from current levels may reach a lower limit where ecosystem function is compromised. In our study, seedbank population in fields after 19 years of intermittent wild headlands is in the upper quartile of seedbank populations based on recent studies (e.g. Hawes *et al*., 2005), demonstrating that rotational deployment of wild headlands maintains seedbank densities at moderate levels: seedbanks neither declined to levels where ecosystem function might be compromised nor increased excessively to densities where crop yield would be limited.

Hawes *et al*. (2010) highlighted that the arable flora derived from seedbanks are an important source of biological diversity. Andreasen *et al*. (2018), revisiting seedbanks in 2014 that had been studied for over 50 years, found that while abundance had returned to 1964 levels from organic farming and fertiliser restrictions, seedbank diversity had not increased from their last survey in 1989. The enriched seedbank after wild headlands enhances function and promotes biodiversity, with consequent impact at higher trophic levels, particularly for the farmland birds that depend on invertebrates to feed their young (Potts 2012).

5.6 Conclusion

Wild headlands were developed to overcome excessive weeds in conservation headlands without the need for additional management intervention. Excessive weed had been a barrier to the take-up of the GWCT conservation headland across Europe, despite support under AES. There has been recognition of the ecological value of reduced fertiliser in conservation headlands (Walker *et al*., 2007) and increased funding for that option (e.g. Countryside stewardship anon Gov.uk 2020), but usually requiring specific additional management. Through their simplicity and effectiveness wild headlands have proved popular on the farms in this Fife study. Their continued use on commercial arable farms over the past 20 years suggests wild headlands can contribute to sustainable intensification in the future.

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6 DISCUSSION

6.1 ABSTRACT

The key question posed by this thesis was “Do wild headlands offer a viable option for arable farmers aiming to integrate biodiversity and production?”. This question is set in the context of agricultural intensification which has resulted in a decline in arable weeds, a loss of the invertebrates which depend upon them as host plants and the consequent impact that this has had at higher trophic levels, particularly on partridge chick survival.

A summary of the key findings of the thesis are presented followed by discussion of the implications and how these findings relate to existing and future policy developments.

6.2 SUMMARY OF FINDINGS

I began this thesis with an outline of the issues we face regarding agricultural intensification in the context of a growing population and a need for increased agricultural production, before giving a detailed evaluation of conservation headlands and an account of their evolution. The development of the wild headland was an attempt to overcome the problems that had limited the take up of the conservation headlands in UK and Europe, but it wasn’t known whether wild headlands would have the same positive impact on invertebrates as had been demonstrated experimentally with conservation headlands. It also wasn’t known if wild headlands had solved the problems in the long-term over weed infestation which characterised conservation headlands.

In Chapter 2 I gave a detailed (and illustrated) account of wild headlands, before a general introduction to my study site. The experimental work for this thesis was conducted on a series of large (by UK standards) commercial farms operating in a high-production environment on good soils. It is a study conducted “in the real world”, with all the limitations that implies on experimental design and opportunity for randomised and replicated trials. In Chapter 3 I compared invertebrate and emerged weed assemblages in fields with and without wild headlands and explored the impact on a partridge population over 5 years. The key finding was that there were differences between the richness and abundance of emerged weeds in wild headlands compared to conventional headlands, which followed through to differences in richness and abundance of invertebrate assemblages and ultimately to changes in the partridge population.

In Chapter 4 I was able to compare yields and gross margins in 82 fields, covering 4 major cereal crops and put a cost on implementing wild headlands. The yield data was compelling, with average yield reduction compared to mid-field across crops of ~40%, depending on crop. On average, the opportunity cost before subsidy in winter wheat was £366 ha^{-1}, less in spring barley and spring oats. Changes in cereal price and input costs over the two years of the experiment had a significant effect.

Chapter 5 answered the question "How do wild headlands influence seedbanks?" I looked at change in the seedbank population in the short term to elucidate the between-year effect of wild headlands and also looked at the long-term effect of wild headlands on the arable seedbank. This was a key component of my research as it tested to see if wild headlands were driving weed species assemblages in the seedbank, which they were, and whether wild headlands on a rotational basis could keep seedbank populations within limits, which they did. This conclusion could only be derived from the very long-term nature of my study and could not be replicated elsewhere in the short-term.

6.3 General discussion

The question "Do wild headlands offer a viable option for arable farmers wishing to incorporate biodiversity and production?" assumes that there will be farmers so inclined. Jackson (2007) observed that adoption of biodiversity-based practices for agriculture is only partially based on the provision of ecosystem goods and services, since individual farmers typically react to the private use value of biodiversity, not the 'external' benefits of conservation that accrue to the wider society. Macdonald and Johnson (2000) found that farmers would be willing to co-operate with schemes for habitat restoration if subsidies were available. Leon *et al.* (2016) in their study of semi-natural habitat on Dutch farms, found that attitudes to maintaining semi-natural habitat amongst traditional dairy farmers was characterised by nervousness over the interpretation of the rules by officials as it might lead to their entitlement to subsidy being compromised. Farmers' motive is an important consideration in assessing the likely take up of AES (McCraken *et al.,* 2015; van Dijk *et al*., 2016).

Within the UK, government has endeavoured to encourage actions and behaviour on farms which meet society's wider objectives through a combination of regulation and support. This twin approach is through Good Environmental and Agricultural Condition (GAEC) rules and regulation of fertilisers and pesticides on one hand, and support under AES on the other. Nevertheless, participants in schemes tend to follow a middle path between adopting measures which meet government objectives and those which least interfere with their farming operations. For example, Ewald *et al.* (2010) in their study on the take up of partridge friendly options under AES (of which they had identified 150) by farmers within their Partridge Count Scheme (PCS) found selection, even amongst PCS participants, appeared to have been either economic or perhaps based on a desire to select options that caused the least disruption to normal crop management practices. However, they did find that where differences appeared between PCS and non-PCS sites, options of major importance to grey partridges (and hence other farmland birds) were more common on PCS sites than non-PCS sites (Ewald *et al*., 2010). Even after it was clearly demonstrated that conservation headlands could help restore partridge populations (Rands, 1985) and foster increased species richness amongst arable plants (Walker, 2007), take up of conservation headlands across the UK and Europe was limited (Clothier, 2013; Albrecht *et al*., 2016). The benefits they accrued were perceived by farmers to be less than the problems they caused with weeds in crops (Storkey and Westbury, 2007). It was therefore essential that these problems were resolved if there is to be take-up of conservation headlands in future. A key output of this research has been to show that restricting fertiliser and rotating headlands has solved these problems.

6.4 INVERTEBRATES

The positive impact on invertebrates found in this study (Chapter 3), and demonstrated by the brood production in the partridge population, has implications for farmers wishing to pursue ecological intensification (EI). In a global review, Bonmarco *et al*. (2013) suggested that reducing numbers of approved pesticides, previously “regulators” of pests in farmland ecosystems, could be met by re-establishing ecosystem services generated in the soil and the surrounding landscape. This would include, for example, wild headlands. Increasing the natural “resilience” of the ecosystem through EI is possible as communities of natural enemies are often found to be more abundant and species rich in structurally complex landscapes (Bonmarco *et al*., 2013). One such pest is the Peach potato aphid, *Myzus persicae*, the most important vector of turnip yellows virus (TuYV) in the UK, which is capable of reducing oilseed rape yields by 30%. Resistance to organophosphates, carbamates and pyrethroids is widespread amongst Peach potato aphids, although Pymetrozine was an effective control. A ban on use of Pymetrozine however, was implemented in the UK in 2020 (Crop Protect, 2020). The grain aphid, *Sitobion avenae*, causes direct feeding damage to cereals and can cause significant yield losses and affect the quality of bread making wheat. It is becoming 30 - 40 % resistant to pyrethroid insecticides at field doses, and additionally pyrethroids may lose approval within 2 years (Anderson, 2020. *Pers comm*. Dr Anderson is Director of Scottish Agronomy). Holland *et al*. (2012), in exclusion trials in English wheat fields, found that although carabids seldom travelled further than 60m into the field from the field margin, aerial arthropods predatory Diptera and Linyphiidae (Araneae) achieved 87% control of cereal aphids (Holland *et al*., 2012). Diptera were the most common invertebrate order found in this study (Chapter 3). Increased taxonomic resolution would have identified key predatory aerial arthropods in our samples and increased resolution should be an important component of future research. Other studies have also found impacts from intensification. Rusch *et al*. (2016), in a review of multiple studies across the EU and North America looking at aphids, demonstrated that agricultural intensification through landscape simplification had negative effects on the level of natural pest control. Closer to home, the impact of EI was evaluated in a six-year study on a 900 ha arable farm in Oxfordshire. At various intensities of EI (up to 8% of cropped area), researchers were able to demonstrate that overall farm profitability was enhanced through EI at the field edge, principally through the action of pollinators on field beans. Positive effects had increased over the course of the study (Pywell *et al*., 2015).

Amongst the invertebrate community in wild headlands were parasitoids, supported by nectar in flowering plants (Hempel and Jervis, 2005). Although limited in their direct effect on aphids (Holland *et al*., 2012, but see Ramsden *et al*., 2017), parasitoids themselves and their larvae are a key partridge chick food. AES schemes aim to improve pollinator abundance and diversity on farmland by sowing wild flower seed mixtures (Nicols *et al*., 2019). In their study looking at bee visitations to a range of wild flowers, 14 wild flower species attracted 37 out of 40 bee species on their study farm and accounted for 99.7% of all visitations, but only 2 of those species were in current AES pollinator mixes (Nicols *et al*., 2019). The wildflower resource in wild headlands may have a wider role in supporting solitary bees not examined in this thesis and is an interesting avenue for further study. If sufficient, wild headlands are much less expensive than bespoke wild-flower mixes. The nectar resource in conservation headlands attracted *Pierid* butterfly species (Dover, 1997). While lepidoptera caterpillars appear in partridge chick diets (Potts, 2012), butterflies may have an intrinsic value. Randall and Smith (2020) identified a new stream of work identifying how exposure to semi-natural habitats can enhance the well-being of people that work, and play, in those areas. Biophilia, an innate love of nature (Wilson, 1984), has gained traction since it was first hypothesised and there are multiple studies identifying positive outcomes from time spent in nature (Terramai and references therein, 2020).

6.5 WEEDS

Still and Byfield (2007) wrote that "the ability of arable plants to lie dormant in the seedbank means, with correct management in the right place, species-rich assemblages can appear within the first year. With targeted action there is no reason why arable plants, the foundation of arable farmland biodiversity cannot return to the British countryside on a large scale." In their review of arable weeds in Sussex from 1968 – 2005 Potts *et al*., (2010) found that between 1968 and 1971 and 2004 - 2005 there was no overall change in occurrence. The weed seedbank remained sufficient to enable a rapid restoration of pre-herbicide flora (Potts *et al*., 2010). In this study, *Spergula arvensis*, not seen in Sussex post 1995 was found in seedbanks in 9 fields (a characteristic species of organic farms in their study. Hawes *et al*., 2010). In the study of emerged weeds (Chapter 3), the dominant dicot weeds included four which were also dominant in un-sprayed plots in a Hampshire wheat field 40 years earlier (Chiverton and Sotherton, 1991). Also occurring frequently in this study, both in the seedbank and in field sampling, was *Myostotis arvensis,* part of the vegetation classification OV12 *Poa annua* – *Myostotis arvensis* community (Rodwell, 2000). Smith *et al.* (2020) found *Myostotis arvensis* correlated with invertebrate richness in wheat fields. With wild headlands there was enhanced species richness compared to conventional headlands and difficult nitrophilous weeds, e.g. *Gallium aperine* was not an issue in wild headlands, which is important if resistance to including wild headlands in cereal fields is to be overcome. Increased species richness in the seedbank of fields with a long history of wild headlands, with abundance limited by herbicide use in intervening years, was observed at the level of sampling intensity used in this study. More intensive sampling would yield more species. Storkey and Neave (2018) looking at resilience in cropping systems at Rothamstead, hypothesised that a more diverse weed community would be less competitive, less prone to dominance by highly adapted, herbicide resistant species and that the diversity of the weed seedbank will be indicative of the overall sustainability of the cropping system (Storkey and Neave, 2018). An interesting area for further research based on this study would be to extend the between-year study of weeds in the seedbank after one year to look at seedbanks in the (minimum) 4 years between wild headlands. Could wild headlands be more frequent? It would increase the % of the farm in insect rich habitat with consequences for farmland birds, equally should wild headlands be wider? There would be an impact on the seedbank further into the field but if the restoration of the arable seedbank was a desired objective of government, this study has demonstrated that it is easily achieved through adopting wild headlands.

6.6 Wild headlands in the future

For farmers prepared to overcome, to quote Storkey and Westbury (2007), their "visceral dislike of managing weeds in their crops", wild headlands have a number of advantages. They have been tested in a commercial setting for 20 years. They are visible, effective, sustainable and easily incorporated into farming systems. At a time when Integrated Pest Management (IPM) is becoming a necessity on commercial farms through regulation of pesticides, they are a ready-made solution and integrate well with grass margins and Local Environmental Risk Assessment for Pesticides (LERAPS) (HSE, 2020). Auto shut-off on modern sprayers and reduced need for growth regulators and fungicides in the absence of fertilisers, has encouraged practitioners to shift from the "managed" pesticide regime developed by the GWCT to "unsprayed" headlands. Provided rotation is maintained (discussed in Chapter 5) there is no disadvantage to this approach and it makes operations even simpler. Although the partridge population was maintained on Farm 3 with 6m wild headlands (Chapter 3), increasing wild headland width to 14m (half a tramline width), doubles the amount of planned insect rich habitat within cereal crops. Potts (2012) introduced 20m no-fertiliser conservation headlands at Peppering to boost invertebrate numbers, but had them around all sides of the field with inevitable weed problems after a few years (Norfolk, 2020. *Pers comm*. The Duke of Norfolk owns Peppering).

This study has demonstrated the likely yield loss within wild headlands in 4 key cereal crops, and using real data from 2 years cropping for 41 fields each year, given an indication of the likely net forgone income for practitioners. Calculation of suitable compensation under AES is therefore possible, but the low opportunity cost to farmers of wild headlands militates against this. Under AES rules compensation is based in income forgone, which we demonstrated in Chapter 4 was £366 ha^{-1} in Winter wheat. As 1 km of wild headland at 7m is 0.7 ha, it equates to a cost of ~£250 km^{-1} in wheat. It is the case that when seeking compensation under AES (particularly where funding is limited or compulsory under greening rules) farmers will select "safe" options. This trend is unwittingly supported by governments. In a review of greening, anecdotal evidence suggested that the approach taken had been to include those elements that were most straightforward to implement, control and verify, not only to keep things as simple as possible in terms of implementation on the ground, but also to reduce the risk for national authorities of dis-allowance (Hart, 2015). The perhaps not unsurprising result was that in 2015, 66% of EFA in Europe was in nitrogen-fixing crops, catch crops or cover crops and 20% in EFA–fallow, with uncertain benefits for wider biodiversity (Underwood and Tucker, 2016). It could be possible to include wild headlands in AES if a flexible approach was adopted and their use was incentivised, i.e. payment rates reflected the value that accrues to society of the provision of public goods: flowers in the fields, increased song birds and reduced pollution rather than just to consider income forgone. A suitable measure under ELMS may be payment m^{-1} with a minimum width of wild headland, say £1 m^{-1} for a minimum width of 6 m. If increased to £5 m^{-1} for a minimum ~24 m width, it increases the % of insect rich habitat within cereal fields to ~8%, the levels required to sustain a recovery of the UK partridge population (Aebischer and Ewald, 2004). Under rules of Good Environmental and Agricultural Condition (GAEC) (Scottish Gov anon), photographs of grass fields are sufficient to demonstrate GAEC. It is apparent in Fig 2.1 that wild headlands are obvious in winter cereal crops from April to harvest. In spring cereals, the effects are less obvious although it's feasible to record weeds at harvest (Fig 2.2). Wild headlands as part of a suite of targeted measures to improve habitat for a particular taxon of concern under HLS, supported by conservation advice, will have wider benefits. MacDonald (2012), for example, when reviewing AES measures for cirl buntings wrote that "they have benefits for a range of taxa beyond the target species, and therefore, largely through reduction of management intensity and maintenance of land-use diversity, improve the overall biodiversity of the farmed landscape where they are present."

6.7 Opportunities for future research

The value of further research into the long-term trajectory of seedbanks in wild headlands has been highlighted. It's also been established that increased taxonomic resolution in invertebrate ID would be invaluable for determining the abundance of predatory Diptera in wild headlands, key for controlling aphids in the absence of pesticides. Diptera were abundant in fields with wild headlands. Additionally, aerial arthropods found in wild headlands, while not a food source for partridges, are a food source for insectivorous birds in decline in Europe (Bowler *et al.,* 2019). It has been established (Chapter 3) that wild headlands harbour the epigeal invertebrates needed by partridge chicks, and that birds have access to the resources within wild headlands, meeting a concern of Vickery *et al.* (2002). The study area falls within the range of one of the last strongholds in Scotland of corn bunting, which were found to have larger broods in areas with conservation headlands (Brickle, 2000). Corn buntings have increased on Balcaskie in recent years (RSPB, 2019) and there is opportunity for research into the interaction between corn buntings, partridge and their predators across a wide area. The ATLAS project (Advanced Tracking and Localization of Animals in real-life Systems. Minerva, 2020) has the capacity to record movements of many taxa simultaneously based on Time-of-arrival principles. The study area used for this thesis has the perfect topography for installing an ATLAS system (Madden, 2020. *Pers comm*. Prof. Madden is the UK partner in for the ATLAS system). It would then be possible to correlate habitat use and availability for a range of taxa illuminating, for example, interactions between predator and prey and the use of semi-natural habitats by avifauna and their predators across an arable landscape. The knowledge would be of immense value in aiding the design of robust and resilient arable landscapes.

6.8 CONCLUSION

Wild headlands were developed 20 years ago to overcome the problems encountered with the GWCT conservation headland. Their primary function was to ensure a supply of phytophagous invertebrates for partridges, but they have a wider role in maintaining ecosystem function at many levels. To paraphrase Hawes *et al.,* (2010), "the arable weeds [*that flourish in wild headlands and the populations of weed seeds they foster in seedbanks*] support a diverse array of herbivores, predators and parasitoids that depend on them for food and shelter. These in turn mediate essential biochemical processes [*above and below ground*] through the functioning of arable food webs" (Hawes *et al*., 2010 with additions in italics). Harvey *et al*. (2020) in a Nature paper published in January, urged no-regret solutions to global declines in insect populations, which they saw as a very real threat to society. Included were, inter-alia, replacing fertilisers and pesticides with agro-ecological measures and designing and validating insect-friendly techniques that are effective, locally-relevant and economically-sound in agriculture. For wild headlands, their time has come.

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