Wading birds in the UK uplands: threats and conservation interventions

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Abstract

Breeding populations of European waders are declining, particularly across north-west Europe. In the UK uplands, populations breeding in habitats such as agricultural grasslands and moorlands are subjected to a range of threats which are contributing to wader declines. A literature review for this thesis found some threats to be well studied such as intensification of livestock farming and predation, whereas other potentially important threats and associated conservation interventions have received less attention. Evidence gaps include the threat of reduced invertebrate resources and the conservation intervention of rush *Juncus spp.* management (available within agri-environment scheme (AES) prescriptions) which this thesis subsequently addresses. The primary study region for this research was the south-west of the Peak District National Park, England (South West Peak, hereafter "SWP"), which is representative of UK upland habitats, land management, and land-use. I first consider reduced invertebrate resources, which has the potential to limit wader populations, by investigating the environmental conditions influencing invertebrate abundance and in turn, the potential for invertebrate prey to influence upland wader abundance. Results revealed important environmental conditions for key invertebrate prey for waders including vegetation height, soil moisture, and rush presence. Such findings could advise habitat management to retain sufficient invertebrate abundance to bolster wader populations where food availability is limiting. I then assess rush management from ecological and social science perspectives. By comparing fields with and without rush management advocated by AES prescriptions, breeding wader surveys show, with a degree of uncertainty, that rush management has the potential to increase Common Snipe *Gallinago gallinago* and Northern Lapwing *Vanellus vanellus* densities, but not Eurasian Curlew *Numenius arquata*. The artificial nest experiment in these fields revealed that daily nest predation rates were two times higher in fields with rush management, highlighting the importance of considering impacts on nest success when designing rush management AES prescriptions, particularly for species which may be attracted to these fields such as Snipe. When interviewing upland farmers in the primary SWP study region, the effectiveness of rush management, both within and outside of AES, at benefiting upland waders was called into question by some farmers. Increasing financial payment for farmers and implementing farmer-endorsed improvements such as herbicide application and improving the flexibility of management dates could help to increase the efficacy and uptake of rush management within AES prescriptions. If rush management is effective at improving breeding wader habitat, managing the environment to ensure removal of other potentially limiting factors is essential. Yet, for upland wader populations in the SWP, this thesis found that invertebrate biomass was not a crucial driving factor, lending greater weight to rush management and the balance between foraging and predation. Overall, this thesis' findings support the landscape scale provision of a mosaic of upland

habitats and vegetation structures to benefit foraging, nesting, and chick-rearing waders as well as their invertebrate prey. Recommendations for future research include extending the research to other upland UK regions over more years with a larger number of fields and interviewees. Specifically for rush management, research priorities involve assessing whether wader species select cut or uncut rush for nesting using real wader nests, and identifying the optimal spatial configuration of cut and uncut rush patches and overall sward structure for different species. In addition, more research effort is required for other conservation interventions within AES prescriptions that lack empirical evaluation, and for other upland habitat such as moorlands to identify the best vegetation management for waders.

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Author’s declaration

I, the author, confirm that the thesis is my own work. I am aware of the University’s Guidance on the Use of Unfair Means (www.sheffield.ac.uk/ssid/unfair-means). This work has not previously been presented for an award at this, or any other, university.

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The manuscript is replicated in its entirety in this thesis, with minor adjustments to the formatting.

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Chapter 1

General Introduction

Wader footprints in a South West Peak bog

1.1 European wader declines

Loss of biodiversity is a global issue of which birds receive a considerable proportion of the attention (Butchart *et al*., 2010; BirdLife International, 2018). Bird populations are well-documented, with >99% of species having sufficient data to provide an IUCN Red List category. Of all extant bird species, nearly one in four are globally threatened or near threatened with extinction (BirdLife International, 2018). A once-common group which is becoming increasingly scarce is shorebirds or wading birds (hereafter waders; Zöckler *et al*., 2003; Pearce-Higgins *et al*., 2017), with much research focused on European populations.

Breeding populations of European waders are exhibiting widespread declines, especially in north-west Europe (BirdLife International, 2017). Populations of upland breeding waders have often received less attention than lowland ones, but are declining rapidly, including in the UK (Table 1.1; Scridel, 2014; Siriwardena *et al*., 2017). Three such species, the Eurasian curlew *Numenius arquata* (hereafter curlew), northern lapwing *Vanellus vanellus* (hereafter lapwing), and Eurasian oystercatcher *Haematopus ostralegus* (hereafter oystercatcher) are now listed as globally Near Threatened and Vulnerable in Europe (BirdLife International, 2015; BirdLife International, 2017; IUCN, 2020). Conditions on the passage or wintering grounds can influence European wader populations (Duriez *et al*., 2012) but low breeding productivity is typically the primary demographic cause of the declines (Roodbergen *et al*., 2012). Management at wader breeding habitats is thus crucial for stabilising and ultimately reversing population declines (O'Brien and Wilson, 2011; Fisher and Walker, 2015; Buchanan *et al*., 2017).

This novel review focuses on evaluating all major causes of upland breeding wader population declines, and their relative importance, in north-west Europe, using information from across the region but focusing on the UK where much of the current research has taken place. We start by assembling the ten wader species breeding in UK uplands into four groups based on their shared ecological traits and habitat requirements. We then take each threat in turn, assessing which species are likely to be affected and of those, which have been studied, thus highlighting knowledge gaps in the process. Subsequently, we discuss the impacts of each threat on those studied species. We then review current conservation interventions for upland waders, based on the present limited understanding of many threats, emphasising the need for further research.

1.1.1 Literature search

The primary source of literature during the literature search process was Web of Science (https://app.webofknowledge.com). Initial literature searches were conducted between October

2017 and January 2018 using various combinations of search terms: (i) wader species common names (Eurasian curlew; northern lapwing; common snipe; Eurasian oystercatcher; Eurasian/European golden plover; common sandpiper; common redshank; common greenshank; dunlin; dotterel) or wader*, (ii) geographic location (such as UK; Britain; upland), (iii) threats (such wind farm; disease OR parasite*; (recreational) disturbance; peat extraction), and (iv) conservation interventions (such as agri-environment scheme). Additional references were obtained at later dates via the references cited in the papers found.

1.2 Upland waders

Upland waders are defined as those that breed in areas which, due to their climatic and edaphic constraints, impose disadvantages to farming systems (otherwise known as Less Favoured Areas; Bonn *et al*., 2009). This land is typically encountered at higher altitudes, but in the north and west of the UK and in parts of Scandinavia, for example, such areas are found down to sea level (European Environment Agency, 2004; Sim *et al*., 2005; Bonn *et al*., 2009). A range of upland habitats, both enclosed and unenclosed, are used by breeding waders including grassland, moorland, and watercourse edges. Due to differences in ecological traits and breeding habitat requirements (Pearce-Higgins and Grant, 2006), the ten UK upland wader species can be separated into groups based on ecological similarities (Table 1.2). The broad breeding habitats (such as moorland) and specific breeding habitat requirements (such as vegetation structure) for each species enabled the formation of groups. Although we focus on the broad similarities between species, these groups could be divided more finely using each species' own precise requirements.

1.3 Threats

The uplands are often experiencing multiple environmental change drivers simultaneously, which hinders diagnosis of the causes of wader declines, but evidence is emerging for which forms of environmental change are most important (Sim *et al*., 2005; Douglas *et al*., 2014; Buchanan *et al*., 2017). Nevertheless, many of these threats are likely to interact, often synergistically (Sim *et al*., 2005; Buchanan *et al*., 2017), with the magnitude and likelihood of each threat impacting the upland wader groups differently. These aspects are highlighted below, along with knowledge gaps when insufficient evidence is available.

Table 1.1. Global assessments, European regional assessments, UK Birds of Conservation Concern 4 (BoCC4) categories, European population trends, UK breeding population trends, and breeding range changes for upland wading birds in Great Britain and Ireland. Global and European regional assessment key: LC = Least Concern; NT = Near Threatened; VU = Vulnerable (www.iucnredlist.org). BOCC4 categories: green = least concern; amber = moderate concern; red = high concern (Eaton *et al*., 2015). European population trends are provided by the Pan-European Common Bird Monitoring Scheme (n/a = no trend available). UK Breeding population trends are provided by the Breeding Bird Survey where available, otherwise alternative national surveys are used (n/a = no trend available). Sources: [a] IUCN Red List global category (IUCN, 2020); [b] IUCN Red List European regional assessment (BirdLife International, 2015); [c] Birds of Conservation Concern 4 (Eaton *et al*., 2015); [d] PECBMS (1980-2016; PECBMS, 2020); [e] For the period 1980-2016; [f] For the period 1981-2016; [g] For the period 1998-2016; [h] Breeding Bird Survey (1995-2018; Harris *et al*., 2020); [i] Statutory Conservation Agency and RSPB Annual Breeding Bird Scheme (1987/88-2011; Hayhow *et al*., 2017); [j] Bird Atlas 2007-2011 (Balmer *et al*., 2013); [k] For the period 1968/72-2008/11; [l] For the period 1988/91-2008/11

Species	**Global assessment[a]**	**European regional assessment[b]**	**UK BoCC4[c]**	**European population trend %[d]**	**UK breeding population trend % (95% CI)**	**Breeding range change %[j]**	
						Great Britain	**Ireland**
Eurasian oystercatcher *Haematopus ostralegus*	NT	VU	Amber	-6[e]	-24 (-34 to -14)[h]	+28[k]	-5[k]
Eurasian golden plover *Pluvialis apricaria*	LC	LC	Green	-13[f]	-5 (-32 to 27)[h]	-20[k]	-50[k]
Northern lapwing *Vanellus vanellus*	NT	VU	Red	-55[e]	-43 (-51 to -36)[h]	-17[k]	-53[k]
Eurasian curlew *Numenius arquata*	NT	VU	Red	-36[e]	-48 (-55 to -41)[h]	-17[k]	-78[k]
Common sandpiper *Actitis hypoleucos*	LC	LC	Amber	-36[e]	-26 (-42 to -6)[h]	-14[k]	-41[k]
Common redshank *Tringa totanus*	LC	LC	Amber	-54[e]	-42 (-61 to -7)[h]	-43[k]	-55[k]
Common snipe *Gallinago gallinago*	LC	LC	Amber	-48[e]	+26 (4 to 55)[h]	-31[k]	-34[k]
Dunlin *Calidris alpina*	LC	LC	Amber	n/a	n/a	-11[k]	-69[k]
Eurasian dotterel *Charadrius morinellus*	LC	LC	Red	n/a	-57[i]	-17[l]	n/a
Common greenshank *Tringa nebularia*	LC	LC	Amber	+1[g]	n/a	+2[k]	-100[k]

Table 1.2. Separation of the ten UK upland wader species into groups based on (i) broad breeding habitats and (ii) specific breeding habitat requirements. Conditions for breeding habitats principally obtained from Balmer *et al*. (2013), and Snow and Perrins (1998). Any additional data sources are cited in the table (iii).

Group	Species	Conditions
1a	Eurasian curlew *Numenius arquata*	i) Grassland and moorland ii) Heterogeneous vegetation height with tussocks and high soil moisture content iii) Durant *et al*., (2008); Buchanan *et al*., (2017); Douglas *et al*., (2017)
	Common snipe *Gallinago gallinago*	i) Grassland and moorland ii) Heterogeneous vegetation height with tussocks and high soil moisture content iii) Hoodless *et al*., (2007); Durant *et al*., (2008); Buchanan *et al*., (2017); Douglas *et al*., (2017)
	Common redshank *Tringa totanus*	i) Grassland ii) Heterogeneous vegetation height with tussocks, high soil moisture content, and shallow pools iii) Smart *et al*., (2006); Durant *et al*., (2008)
1b	Northern lapwing *Vanellus vanellus*	i) Grassland, moorland, and arable ii) Short vegetation with shallow pools iii) Taylor and Grant, (2004); Durant *et al*., (2008); Eglington *et al*., (2008); Smart *et al*., (2013); Buchanan *et al*., (2017)
2a	Eurasian golden plover *Pluvialis apricaria*	i) High-altitude blanket bog, moorland, and peatland, and agricultural grasslands ii) Short vegetation and damp conditions iii) Pearce-Higgins and Yalden, (2003); Buchanan *et al*., (2017)
	Dunlin *Calidris alpina*	i) High-altitude blanket bog, moorland, and peatland ii) Short vegetation with clusters of pools iii) Lavers and Haines-Young, (1996)
2b	Common greenshank *Tringa nebularia*	i) High-altitude blanket bog, moorland, and peatland; down to sea-level in northern Scotland ii) Short vegetation, often in areas with abundant standing or flowing water; accepts scattered trees and shrubs iii) Thompson and Thompson, (1991)
3	Eurasian dotterel *Charadrius morinellus*	i) Montane alpine plateaux ii) *Racomitrium lanuginosum*-*Carex bigelowii* heath and *Juncus trifidus* heath

		iii) Thompson and Brown, 1992; Galbraith *et al*., 1993; Thompson and Whitfield, 1993; Hayhow *et al*., 2015
4	Common sandpiper *Actitis hypoleucos*	i) Inland watercourses, such as rivers, lakes, and reservoirs ii) Grassy banks and shingle islands/shores; typically nests close to water and conceals nest in vegetation iii) Dougall *et al*., (2010)
	Eurasian oystercatcher *Haematopus ostralegus*	i) Grassland, arable, and inland watercourses, such as rivers and lakes ii) Open, bare ground and short vegetation; grassy banks and rocky shores iii) O'Brien, (2001); Mandema *et al*., (2013); van de Pol *et al*., (2014)

1.3.1 Land-use

Here we will discuss the impacts of five major upland land-uses on breeding waders: intensification of livestock farming, game management, afforestation, wind farms, and peat extraction. Indirect additional mechanisms of land-use change, such as agricultural machinery and drainage, are covered in the subsequent corresponding sections.

1.3.1.1 Intensification of livestock farming

Upland farming is predominantly pastoral and typically occurs on in-bye grassland and adjacent moorland (Fuller and Gough, 1999). Since the 1800s and particularly the 1950s, the UK has experienced a shift from mixed livestock farming to exclusive rearing of sheep and a significant increase in stocking densities from 8 million to 35.8 million (1860s-2002), with a peak of 44 million in 1993 (Anderson and Yalden, 1981; Fuller and Gough, 1999; Pearce-Higgins, Grant, *et al*., 2009). All four UK countries have since experienced moderate declines in upland sheep densities, ranging from -10.3% (England) to -31.0% (Northern Ireland; 2000-2010; Silcock *et al*., 2012). Despite recent declines, current stocking densities remain considerably higher than historic levels. These changes in livestock farming have been enabled by additional agricultural changes to increase productivity from the land. Drainage of grassland and moorland, and increased inorganic and organic fertiliser application, liming, rolling and chain-harrowing, and ploughing and re-seeding (Baines, 1988; Baines, 1990; Vickery *et al*., 2001; Fuller *et al*., 2002; Newton, 2004; Devereux *et al*., 2004; McCracken and Tallowin, 2004) have been used to improve grazing quality for livestock and enable silage production (Fuller and Gough, 1999; Fuller *et al*., 2002; Newton, 2004).

Direct impacts on wader breeding success transpire from these livestock farming trends. Higher livestock densities affect nest and chick mortality rates through consumption (a possible major evidence gap) and trampling (Beintema and Müskens, 1987; Pennington, 1992; Vickery *et al*., 2001; Newton, 2004), and silage production (and other associated agricultural activities such as rolling, chain-harrowing, and rush cutting during the breeding season) causes mechanical destruction of nests and chicks (Baines, 1990; Vickery *et al*., 2001; Newton, 2004). Intensive sheep grazing can result in conversion from heather-dominated moorland to grassland (Anderson and Yalden, 1981; Fuller and Gough, 1999; Welch *et al*., 2005; Hartley and Mitchell, 2005), which will likely adversely impact moorland-specific waders from group 2 and heath-favouring Eurasian dotterel *Charadrius morinellus* (hereafter dotterel; group 3; Brown and Stillman, 1993; Galbraith *et al*., 1993; van der Wal *et al*., 2003). Waders that preferentially breed in short grass such as lapwing from group 1b and oystercatcher from group 4, however, could benefit from this conversion. Yet, for waders less dependent on moorland

and heath, vegetation structure may be more critical than vegetation type (Pearce-Higgins and Grant, 2006; Amar *et al*., 2011). Extreme, uniform vegetation structures are created by intensive sheep grazing (short vegetation) and silage production (tall vegetation), both of which affect invertebrate resources and predation pressure (see sections 1.3.2. and 1.3.3; Baines, 1988; Vickery *et al*., 2001; Newton, 2004; Whittingham and Evans, 2004; Dennis *et al*., 2008; Calladine *et al*., 2014; Evans *et al*., 2015). Reseeding to boost these agricultural practices (livestock grazing and fodder crop production) initially increases breeding lapwing abundance, but this is short-lived with subsequent declines in the long-term due to lower breeding densities and hatching success (although abundance can remain higher than in the surrounding landscape; Düttmann *et al*., 2018; McCallum *et al*., 2018). Despite these adverse impacts, short vegetation aligns with Eurasian golden plover *Pluvialis apricaria* (hereafter golden plover) and lapwing habitat preferences for nesting sites (Whittingham *et al*., 2006; Pearce-Higgins and Grant, 2006; Smart *et al*., 2013; Douglas and Pearce-Higgins, 2014), and waders such as curlew are known to nest in the taller vegetation of silage meadows (Jefferson, 2005; Byrkjedal *et al*., 2012) though selection of such fields and onset of nesting will often occur prior to the rapid increase in silage vegetation growth. Nevertheless, neither of these agriculturally intensified vegetation structures are ideal for breeding upland waders as heterogeneous vegetation structures, to varying extents for each species, are most beneficial for both nesting and chick rearing phases (see sections 1.3.2 and 1.3.3; Whittingham *et al*., 2001; Pearce-Higgins and Grant, 2006; Bracken *et al*., 2008). Silage production and nitrogen deposition, derived from silage-associated fertiliser application and livestock dung (as well as anthropogenic air pollution), result in reduced plant diversity and subsequent invertebrate diversity, leading to lower invertebrate abundance (Baines, 1988; Vickery *et al*., 2001; Newton, 2004; Calladine *et al*., 2014; Ewing *et al*., 2020). Moreover, lowered soil moisture due to agricultural drainage can reduce moisture-loving invertebrate availability and abundance, such as cranefly (Diptera: Tipulidae) larvae (Baines, 1988; Pearce-Higgins and Yalden, 2003; Newton, 2004), and thus produce shorter breeding seasons leading to less opportunities for re-nesting and reduced breeding success (Green, 1988; Green *et al*., 1990; Green and Robins, 1993; Whittingham and Evans, 2004).

To conclude, as in lowland settings, pastoral farming practices in the uplands can cause changes in habitat quality, reductions in invertebrate accessibility and abundance, and increased egg and chick mortality rates. As this is a key issue affecting upland waders, the impacts of livestock farming on all four wader groups have been studied, although groups 2, 3, and 4 have received less attention.

1.3.1.2 Game management

Shooting of red deer *Cervus elaphus* and red grouse *Lagopus lagopus scoticus* for sport has been synonymous with the UK uplands for nearing two centuries (Sotherton *et al.*, 2009). Land managed for deer hunting artificially inflate the number of hinds, and thus stags, to provide more game and greater revenue (Bullock *et al.*, 1998). As with livestock grazing, high deer densities result in overgrazing, altering plant communities and reducing the proportion of moorland habitat (see section 1.3.1.1; Pearce-Higgins *et al.*, 2009). Many estates also fertilise land and burn heathland vegetation to improve grazing (Trenkel *et al.*, 1998; Davies *et al.*, 2010) producing the same consequences of nitrogen deposition and heather burning as those mentioned in section 1.3.1.1 and the current section 1.3.1.2, respectively.

Management for grouse, on the other hand, is a more complex and prevalent issue than that for red deer. Between 6 and 15% of the UK uplands (56% of the English uplands) comprise managed red grouse moorland (Sotherton *et al.*, 2009; Grant *et al.*, 2012). Grouse moor management, particularly moors managed for “driven” grouse shooting, is a contentious issue due to raptor persecution (Melling *et al.*, 2018) and potential adverse environmental effects including reduced water quality and carbon storage (Glaves *et al.*, 2013), but these are not discussed further here. Driven grouse moors are managed using rotational heather burning and predator control, as well as vegetation cutting and livestock grazing, yet management intensity varies (Tharme *et al.*, 2001; Grant *et al.*, 2012; Newey *et al.*, 2016; Ludwig *et al.*, 2019). Difficulties arise during research synthesis as studies do not quantify burning intensities, and knowledge of temporal impacts are limited due to the exclusive study of short-term burning effects. All group 1 and 2 species will breed on moorland so could be affected by this management, although the nature and magnitude of impacts vary. Common sandpiper *Actitis hypoleucos* from group 4 could also possibly be affected if the water bodies they breed at are within areas managed for grouse moors.

Few studies of the impacts of grouse moor management on waders have directly and successfully teased apart the effects of predator control and heather burning, despite attempts by Daplyn and Ewald (2006), and Fletcher *et al.* (2010) (Harper *et al.*, 2018). Yet, Littlewood *et al.* (2019) and Ludwig *et al.* (2019) concluded that waders associated far more strongly with predator control and burning, and Franks *et al.* (2017) revealed a negative association between burning and curlew breeding densities. Burning could adversely affect invertebrate prey populations through direct mortality and increased habitat aridity (Grant *et al.*, 2012; Harper *et al.*, 2018) but could increase accessibility of the remaining populations through shorter vegetation (Pearce-Higgins and Yalden, 2004). Similarly, opposing effects on wader nesting habitats arise as short vegetation derived from burning, cutting, and livestock grazing will benefit lapwing and golden plover (Pearce-Higgins and Yalden, 2004; Pearce-Higgins and Grant, 2006; Grant *et al*., 2012; Douglas *et al*., 2017) at the expense of more concealed nesters such as common snipe *Gallinago gallinago* (hereafter snipe; Snow and Perrins, 1998; Laidlaw *et al*., 2020). Such vegetation management techniques, however, could provide the mosaic of different vegetation heights and compositions required for all nesting moorland waders, as well as invertebrate prey diversity and abundance, depending on the size and spatial configuration of managed areas (Usher, 1992; McFerran *et al*., 1995; Eyre *et al*., 2003). Smaller patches of managed and unmanaged vegetation could create a heterogeneous vegetation structure at the size of individual breeding pairs' territories, thus providing suitable nesting, chick-rearing, and foraging areas. Depending on the timing of heather burning, however, there is the possibility that burning could destroy early wader nests as burning is permitted into April (England and Wales: 15th April; Scotland: 30th April; Moss *et al*., 2005; Grant *et al*., 2012), yet the likelihood of destruction is cited as only 1-2% (Glaves *et al*., 2005). Meanwhile, predator control has beneficial impacts through increased breeding success and wader abundance (Tharme *et al*., 2001; Fletcher *et al*., 2010; Amar *et al*., 2011; Littlewood *et al*., 2019; Ludwig *et al*., 2019). Although, it is possible that higher grouse numbers could inadvertently increase predator populations, similar to the release of common pheasants *Phasianus colchicus* and red-legged partridges *Alectoris rufa* in the lowlands (Newton, 2004; Pringle *et al*., 2019). The greater supply of red grouse as prey could boost predator populations where food availability is a limiting factor.

When combining the overall effects of grouse moor management, beneficial impacts are shown for lapwing (with no impact in one case; Ludwig *et al*., 2019), common redshank *Tringa totanus* (hereafter redshank), and common sandpiper (Table 1.3). However, overall impacts of such management on curlew, golden plover, snipe, dunlin *Calidris alpina*, and common greenshank *Tringa nebularia* (hereafter greenshank) are neither entirely positive nor negative (Table 1.3). This amalgamation of multiple studies (Table 1.3), although giving an idea of the overall effect of grouse moor management on different wader species, will not provide an entirely true representation. Only single studies have researched redshank, greenshank, and common sandpiper. Future research must investigate the independent effects of vegetation management (heather burning, cutting, and grazing) and predator control on all moorland-breeding waders, rather than speculating the drivers of population change. Moreover, the long-term impact of vegetation management such as burning (including the quantification of burning intensities) needs to be incorporated. As management associated with grouse moors will vary between sites, the magnitude of such management must be assessed for each wader species individually due to differing habitat requirements (Table 1.2), as well as impacts on other aspects such as invertebrate prey.

Table 1.3. The impacts of grouse moor management practices (rotational heather burning and predator control) on eight upland wader species known to nest on moorland. Overall moorland management relates to instances where the separate impacts of heather burning, and predator control are not distinguished. The total number of studies (including both peer-reviewed and grey literature) finding positive, negative, and no impacts are shown for each wader species. Some studies show differing directions of effect on the same species depending on the management practice considered.

Wader species	**Impact of grouse moor management**	**Overall moorland management**	**Heather burning**	**Predator control**	**Studies (total)**
Eurasian curlew *Numenius arquata*	Positive	– Higher breeding densities on managed moors (Tharme *et al*., 2001) – Lower abundance when management ceased and higher abundance when management restored (Ludwig *et al*., 2019) – Higher breeding densities positively associated with red grouse abundance (Franks *et al*., 2017)	– Greater proportion of habitat burned > higher breeding density (Daplyn and Ewald, 2006) – Greater proportion of habitat burned > higher abundance (Newey *et al*., 2016)	– Reduce predation pressure > increase occupancy (Haworth and Thompson, 1990) – Reduced red fox and carrion crow abundance > increase breeding success and breeding abundance (Fletcher *et al*., 2010) – Positive association between predator control and abundance (Littlewood *et al*., 2019)	8
	Negative		– Increase burning intensity > decrease density (Dallimer *et al*., 2012) – Greater area of rotational strip burning > lower breeding densities (Franks *et al*., 2017)		2
	None		– Little support for effect of burning on abundance (Littlewood *et al*., 2019)		1
Common snipe *Gallinago gallinago*	Positive			– Positive association between predator control and abundance (Littlewood *et al*., 2019)	1
	Negative		– Reduce burning intensity > increase density (Dallimer *et al*., 2012)		1
	None	– Significant positive and negative associations between management and abundance depending on	– Poor support for effect of burning on abundance (Littlewood *et al*., 2019)		3

		ordination analysis method (Newey *et al*., 2016) – Higher abundance when management ceased and when management restored (Ludwig *et al*., 2019)			
Common redshank *Tringa totanus*	Positive			– Reduce predation pressure > increase occupancy (Haworth and Thompson, 1990)	1
	Negative				0
	None				0
Northern lapwing *Vanellus vanellus*	Positive	– Higher breeding densities on managed moors (Tharme *et al*., 2001) – Lower abundance when management ceased (Ludwig *et al*., 2019) – More intensive management associated with lower declines (Amar *et al*., 2011)	– Greater proportion of habitat burned > higher breeding densities (Daplyn and Ewald, 2006)	– Reduced red fox and carrion crow abundance > increase breeding success and breeding abundance (Fletcher *et al*., 2010)	5
	Negative				0
	None	– No change in abundance when management restored (Ludwig *et al*., 2019)			1
Eurasian golden Plover *Pluvialis apricaria*	Positive	– Higher breeding density on managed moors (Tharme *et al*., 2001; Daplyn and Ewald, 2006) – Lower abundance when management ceased and higher abundance when management restored (Ludwig *et al*., 2019)	– Higher likelihood of prevalence with greater proportion of habitat burned (Newey *et al*., 2016) – Greater area of habitat burned > higher abundance in the initial post-burning period (Douglas *et al*., 2017)	– Reduce predation pressure > increase occupancy (Haworth and Thompson, 1990) – Associated with areas of gamekeeper activity (Daplyn and Ewald, 2006) – Reduced red fox and carrion crow abundance > increase breeding success and breeding abundance (Fletcher *et al*., 2010)	7

				– Positive association between predator control and abundance (Littlewood *et al*., 2019)	
	Negative	– More intensive management associated with greater declines (Amar *et al*., 2011)			1
	None		– Poor support for effect of burning on abundance (Littlewood *et al*., 2019)		1
Dunlin *Calidris alpina*	Positive	– Higher breeding density, and lower rate of decrease in density on managed moors (Daplyn and Ewald, 2006)	– Greater proportion of habitat burned > higher abundance (Newey *et al*., 2016)		2
	Negative		– Greater proportion of habitat burned > decrease in breeding density (Daplyn and Ewald, 2006)		1
	None				0
Common greenshank *Tringa nebularia*	Positive				0
	Negative				0
	None	– Significant positive and negative associations between management and abundance depending on ordination analysis method (Newey *et al*., 2016)			1
Common sandpiper *Actitis hypoleucos*	Positive		– Greater proportion of habitat burned > higher abundance (Newey *et al*., 2016)		1
	Negative				0
	None				0

1.3.1.3 Afforestation

Establishment of forestry plantations, along with associated practices such as drainage, are becoming an increasing problem for breeding waders in open upland areas (Lavers and Haines-Young, 1997; Worrall and Evans, 2009; Amar *et al*., 2011; Wilson *et al*., 2014; Fraixedas *et al*., 2017). Afforestation, particularly on moorland, has increased significantly since the 1940s (Scottish Government., 2009; Bunce *et al*., 2014) with over 12% of the UK now forested predominantly with exotic conifers, although afforestation with native broadleaved trees is now increasing (Mason, 2007; Bunce *et al*., 2014). These changes in afforestation practice to continued species-rich woodland expansion derive from associated ecosystem services, such as flood and climate change mitigation, and conservation (Scottish Government., 2009; Wilson *et al*., 2014). However, renewed ambition in recent years to afforest the UK to increase carbon sequestration is leading to concerns that ambitious targets to plant 30,000 ha of woodland per year (Burke *et al*., 2021) could drive non-native plantations on ecologically sensitive land, particularly in the uplands.

Proximity to plantations consistently negatively impacts upland waders (Stroud *et al*., 1990; Avery and Haines-Young, 1990; Hancock and Avery, 1998; Finney *et al*., 2005; Hancock *et al*., 2009; Amar *et al*., 2011; Franks *et al*., 2017), although one study by Avery (1989) does not find evidence of this. Only dotterel (group 3) and oystercatcher (group 4) have not been studied as generally neither species breeds on open moorland where upland tree planting often occurs (Table 1.2). The initial effect of plantation establishment involves the direct displacement of waders from the plantation site (Amar *et al*., 2011). Subsequently, the surrounding 400-700 m can dissuade waders due to edge effects (Stroud *et al*., 1990; Chadwick *et al*., 1997; Amar *et al*., 2011; Douglas *et al*., 2014; Wilson *et al*., 2014), with these adverse impacts intensifying as forests become established (Stroud *et al*., 1990; Franks *et al*., 2017). Waders may avoid areas surrounding woodland because of perceived or actual predation risk (Amar *et al*., 2011; Wilson *et al*., 2014; Hancock *et al*., 2020). Indeed, plantations provide breeding sites for predators such as red fox *Vulpes vulpes* and carrion crow *Corvus corone* (Amar *et al*., 2011; Douglas *et al*., 2014; Hancock *et al*., 2020), and the reduction or cessation of grazing on land surrounding plantations can increase alternative prey abundance such as voles (Chadwick *et al*., 1997; Evans *et al*., 2006) which could boost such predator populations. Moreover, habitat quality of the surrounding land can be diminished; drainage ditches installed for plantations and transpiration by the trees lowers the water table (Shotbolt *et al*., 1998; Worrall and Evans, 2009), which could reduce invertebrate prey abundance (see section 1.3.3). Importantly, Douglas *et al*. (2014) noted reduced wader breeding success in areas close to forests. Whether this is due to increased predation pressure, reduced habitat quality, or a combination of both is open to debate. Lower breeding success, however, could lead to abandonment of such areas in future years (Stroud *et al*., 1990).

All the afforestation research discussed has examined the impacts of commercial conifer plantations, with native broadleaf woodland also located in some study areas. The direct displacement and indirect edge effects of woodland creation on waders are likely to be similar, regardless of woodland type.

1.3.1.4 Wind farms

Installation of wind farms in upland areas has become increasingly prevalent since the early 2000s with over 200 wind farms built (Houses of Parliment, 2019; Newton, 2020), leading to concerns such as collision mortality and disturbance displacement in breeding birds (Drewitt and Langston, 2006). The limited, high-quality research concerning wind farm impacts on upland waders, of which additional research is required, suggests that collision mortality is not a key factor (Pearce-Higgins *et al*., 2012; Sansom *et al*., 2016), and generates contradictory findings regarding the effect of the two phases of wind farm development: construction and operational. Nevertheless, irrespective of the phase, the impacts on specific species are generally agreed upon (Pearce-Higgins, Stephen, *et al*., 2009; Douglas *et al*., 2011; Pearce-Higgins *et al*., 2012; Sansom *et al*., 2016). Wind farms negatively impact breeding abundance and distribution of curlew (density decline: 42.4% within 500 m (Pearce-Higgins, Stephen, *et al*., 2009); 40.0% within 620 m (Pearce-Higgins *et al*., 2012)) and snipe (density decline: 47.5% within 500 m (Pearce-Higgins, Stephen, *et al*., 2009); 53% within 620 m (Pearce-Higgins *et al*., 2012)), whereas lapwing and dunlin do not appear to be adversely affected (Pearce-Higgins, Stephen, *et al*., 2009; Pearce-Higgins *et al*., 2012). The effect on golden plover is less conclusive though negative impacts are probable (Pearce-Higgins *et al*., 2008; Pearce-Higgins, Stephen, *et al*., 2009; Douglas *et al*., 2011; Pearce-Higgins *et al*., 2012; Sansom *et al*., 2016). Habitat change could explain this difference in effect between wader species. Damage to the ground vegetation during construction creates a more open vegetation structure: beneficial for nesting lapwing, dunlin, and potentially golden plover, but potentially detrimental for nesting curlew and snipe (Pearce-Higgins and Grant, 2006; Hancock *et al*., 2009; Pearce-Higgins *et al*., 2012; Smart *et al*., 2013). Curlew and snipe may also be more sensitive to disturbance (see section 1.3.5). Behavioural avoidance of human activity during the construction phase or the turbines themselves could lead to displacement and reductions in breeding abundance and distribution (Pearce-Higgins, Stephen, *et al*., 2009; Pearce-Higgins *et al*., 2012; Sansom *et al*., 2016). This could result in abandonment of wind farm sites as breeding territories in future years, depending on breeding success at the new displacement sites (Berg, 1994; Pearce-Higgins *et al*., 2012). Recent research has also shown that earthworms, a key dietary component for several wader species (Buchanan *et al*., 2006; Pearce-Higgins, 2010), decrease in abundance closer to wind turbines due to vibratory noise (Velilla *et al*., 2021); this decrease therefore has the potential to reduce adult and chick fitness, and consequently breeding success (see section 1.3.3). Whilst wind farm impacts have not been studied for waders in groups 3 and 4, such impacts are likely to be limited as these species occur at low densities in the moorland, bog, and rough grassland sites on which wind farms tend to be built (Pearce-Higgins, Stephen, *et al*., 2009; Douglas *et al*., 2011; Pearce-Higgins *et al*., 2012).

1.3.1.5 Peat extraction

Of the approximately 3 million ha of peatland cover in the UK, large-scale degradation and loss has occurred through multiple land-uses including peat extraction (Joint Nature Conservation Committee, 2011; Marsden and Ebmeier, 2012; Evans *et al*., 2017). The demand for peat extraction derives from its use in the generation of energy and gardening (Vasander *et al*., 2003), with peat extraction for such purposes accounting for 4.9% of UK peatlands (energy = 4.6%; horticulture = 0.3%; Evans *et al*., 2017). Similarly in Europe, approximately 6% of the 96 million ha of peatland cover has undergone extraction (Vasander *et al*., 2003). Nevertheless, extraction rates waned between 2000 and 2010 in England (Marsden and Ebmeier, 2012), and the UK government intends to end the use of peat for horticultural purposes by 2030 (Department for Environment Food & Rural Affairs, 2018); as extraction for fuel encompasses 15 times more peatland than extraction for horticulture, however, the UK government should focus on ceasing the use of peat for energy too. In addition to concerns regarding carbon release and climate change (Limpens *et al*., 2008), there are concerns over the impact on waders. Little research has directly examined the impacts of peat extraction on waders, yet adverse effects can be construed via the influence on peatland habitat (Lavers and Haines-Young, 1996; Henderson *et al*., 2002; Fraixedas *et al*., 2017).

Peat extraction has a direct effect on habitat quality, leading to degradation, fragmentation, and loss of wader habitat (Henderson *et al*., 2002), particularly for groups 1 and 2 which breed on peatland. However, drainage may be the most significant widespread consequence of peat extraction as it degrades peatland habitat for foraging waders which rely on wet substrates (Douglas *et al*., 2014; Fraixedas *et al*., 2017). The footprint of peat extraction is indirectly extended via drainage as water level reductions range between 5 m and 200 m from drainage systems depending on scale and arrangement of ditches (Landry and Rochefort, 2012). Drainage enables the initial extraction of the top peat layer; yet drainage subsequently leads to compaction and lowering of the remaining substrate. Such large-scale drainage, not only for peat extraction but also for livestock farming, is a considerable problem in the UK where the majority of peatland is consequently no longer intact

(Holden *et al*., 2004; Fraixedas *et al*., 2017). Furthermore, disturbance caused by human activity during peat extraction could indirectly affect breeding waders (Hockin *et al*., 1992).

The threat that peat extraction poses for waders in the UK uplands may not be critical due to limited spatial extent of extraction. Nevertheless, where local breeding wader populations and peat extraction sites overlap, this practice has the potential to considerably degrade the available habitat, potentially causing reduced breeding success or abandonment of the area as a breeding site.

1.3.2 Predation

Ground-nesting birds, such as waders, are particularly susceptible to predation (Gibbons *et al*., 2007; Roos *et al*., 2018) with the degree of vulnerability differing between species due to variations in defensive behaviour exhibited by adults, breeding habitat selection, and detectability of chicks. Nevertheless, they are long-lived species and do not need to produce multiple fledglings annually (Ausden *et al*., 2009). Wader nests and chicks are preyed upon by a wide variety of mammalian and avian predators, both diurnal and nocturnal (Teunissen *et al*., 2008; see for a non-exhaustive list of wader predator species). Despite this natural inter-specific interaction, some predators have increased in Britain, such as carrion crow, raven *Corvus corax*, and common buzzard *Buteo buteo* (Amar *et al*., 2010; Balmer *et al*., 2013; Harris *et al*., 2020). Underlying causes of these predator population increases will likely include multiple drivers: increased food resources from human refuse, artificially inflated native gamebird numbers and increasing large-scale annual releases of non-native gamebirds, sheep carcasses, and livestock dung-dwelling invertebrates; recovery of avian predators following cessation of organochlorine pesticide use; and declines in killing by humans (Fuller and Gough, 1999; Gibbons *et al*., 2007; Pringle *et al*., 2019; Newton, 2020). Due to the resultant ecological imbalance, with non-predation associated reductions in wader numbers exacerbating this imbalance, these high predator to prey ratios are increasingly impacting wader breeding success through direct mortality and effects on behaviour such as decreased nest attendance (which reduces incubating adult predation risk but increases nest predation risk; Grant *et al*., 1999; Schekkerman *et al*., 2009; Cervencl *et al*., 2011; Rickenbach *et al*., 2011). Although, compared to red fox and corvids, other predators such as raptors are less likely to limit prey populations (Roos *et al*., 2018). Consequently, predation has developed into a major threat to breeding waders with the resultant low breeding productivity cited as a primary cause of wader declines (Grant *et al*., 1999; MacDonald and Bolton, 2008b; MacDonald and Bolton, 2008a; Roodbergen *et al*., 2012; Franks *et al*., 2017; Mason *et al*., 2018; Cook *et al*., 2021).

Changes in vegetation structure through land-use change can further increase the risk of nest or chick predation. Short, homogenous vegetation decreases nest crypsis, removes cover for chicks, and reduces small mammal abundance (an alternative prey source to waders, low abundance of which can thus increase wader predation risk; Baines, 1990; Vickery *et al*., 2001; Whittingham and Evans, 2004; Laidlaw *et al*., 2013; Evans *et al*., 2015). Conversely, expanses of tall vegetation associated with silage production (Newton, 2004; Calladine *et al*., 2014), encroachment of rush and other rank vegetation (Robson and Allcorn, 2006; Fisher and Walker, 2015; Douglas *et al*., 2017), and land abandonment (Kamp *et al*., 2018) reduces the detectability of predators for foraging and open-nesting waders (Whittingham and Evans, 2004). Regarding livestock farming, higher stocking densities can cause more frequent disturbance of incubating or brooding waders and thus attract predators to nests and broods (Jönsson, 1991; Fuller and Gough, 1999; Hart *et al*., 2002; Newton, 2004).

Comprehensive research has been done on all four wader groups but more so on species such as curlew, redshank, and lapwing from group 1. Predation is a key issue, driving wader population declines through high nest and chick predation (Roodbergen *et al*., 2012; Roos *et al*., 2018; Cook *et al*., 2021). Red fox and carrion crow are often cited as the primary mammalian and avian culprits (MacDonald and Bolton, 2008b; Roos *et al*., 2018), but variability in the suite and abundance of predator species between areas means that the magnitude of this threat varies on a site-by-site basis (Bolton *et al*., 2007; Teunissen *et al*., 2008). For example, in some study areas, corvids such as carrion crow exert less pressure on waders than expected (Madden *et al*., 2015).

1.3.3 Reduced invertebrate resources

Food resources for waders are determined by abundance of specific invertebrate prey (primarily influenced by land management decisions and climate change) and accessibility of these prey items (primarily influenced by vegetation structure and soil penetrability). Invertebrates typically taken consist of surface and sub-surface invertebrates such as adult Coleoptera and earthworms, respectively (Buchanan *et al*., 2006; Pearce-Higgins, 2010). Worrying long-term declines in invertebrate populations (Hallmann *et al*., 2017; Fox *et al*., 2021) is a potential overlooked driver of low wader breeding success and an area of research which should be prioritised. Possible reduced food intake could diminish adult fitness (Smart and Gill, 2003) and increase egg and chick mortality (Pearce-Higgins and Yalden, 2002; Pearce-Higgins and Yalden, 2004; Eglington *et al*., 2009; Douglas and Pearce-Higgins, 2014). In black-tailed godwits *Limosa limosa* (a lowland-breeding UK wader species) and lapwing, lighter parents exhibit lower nest and brood survival; lighter parents incubate less, nest predation is higher, replacement clutch likelihood is lower for lighter females, and lighter males stop tending broods earlier as the chicks grow (Hegyi and Sasvári, 1998). Lighter female waders also produce smaller eggs resulting in smaller chicks with lower growth and survival rates (Galbraith, 1988a; Galbraith, 1988b; Thompson *et al*., 1990). It is also possible that smaller wader chicks, due to poor foraging conditions, have an increased risk of starvation and predation (Beintema and Visser, 1989; Baines, 1990; Pearce-Higgins and Yalden, 2004; Schekkerman *et al.*, 2009). Smaller adults, resulting from poor growth as chicks, are also more susceptible to extreme weather conditions (Clark, 2009; see section 1.3.4). Evidently, food intake is an intrinsic part of survival, yet the impacts of invertebrate abundance have only been studied directly in lapwing (Baines, 1990) and golden plover (Pearce-Higgins and Yalden, 2003; Pearce-Higgins and Yalden, 2004; Douglas and Pearce-Higgins, 2014).

Invertebrate abundance can be positively and negatively influenced by land management and climate. Certain land management practices and outcomes produce opposing impacts on invertebrates depending on the intensity of the land management, and sometimes on the invertebrate taxa. For example, inorganic fertiliser application generally reduces the diversity of invertebrate taxa but changes in abundance varies, particularly depending on the quantity of fertiliser applied (Vickery *et al.*, 2001). Moderate applications increase earthworm abundance, a key dietary component for many waders such as snipe (Vickery *et al.*, 2001; Hoodless *et al.*, 2007). However, nitrogen-containing fertilisers, as well as a reduction in lime application, atmospheric acid deposition, and cropping, can reduce soil pH and thus earthworm abundance (McCallum *et al.*, 2016). In comparison, organic fertiliser application such as farmyard manure, and dunging from livestock in general, is typically beneficial for invertebrates when applied in low to moderate quantities and derived from non-avermectin (pesticide) treated livestock (Vickery *et al.*, 2001; McCracken and Tallowin, 2004; Buchanan *et al.*, 2006).

Vegetation height and structural complexity, another factor influenced by agricultural practices, also impacts invertebrates on a continuum. Higher stocking densities, for example, lower plant biomass and simplify the sward structure, typically resulting in lower arthropod abundance, particularly foliar groups (Vickery *et al.*, 2001; Dennis *et al.*, 2005; Dennis *et al.*, 2008; Evans *et al.*, 2015). However, for soil-dwelling cranefly larvae, another key dietary component, abundance increases with stocking density (Dennis *et al.*, 2005; Buchanan *et al.*, 2006). Cranefly larvae are also prone to desiccation; thus, climate change, alongside management practices such as land drainage, can reduce larvae abundance by lowering soil moisture content (Pearce-Higgins *et al.*, 2010; Carroll *et al.*, 2011; Carroll *et al.*, 2015). Out of sheep and cattle, the typical livestock found in upland areas, sheep have the greatest impact on vegetation by creating shorter, more homogenous swards (Vickery *et al.*, 2001). Such cropped vegetation structures are also created by cutting regimes for hay and silage, with timing of cuts changing the severity depending on the peak emergence times and feeding behaviour (herbivores, pollinators, predators) of different invertebrate taxa (Vickery *et al.*, 2001).

Rotational heather burning, associated with game management (see section 1.3.1.2), can also influence vegetation structure by increasing heterogeneity via burnt and unburnt patches, with associated increases in invertebrate abundance and diversity (Usher, 1992; McFerran *et al*., 1995; Eyre *et al*., 2003). Yet, this management practice is not without consequence, with burning leading to loss of habitat (although recolonisation from unburnt areas can be rapid) and direct mortality of invertebrates (Grant *et al*., 2012; Harper *et al*., 2018). In addition, direct mortality through agricultural and grouse moorland management-associated pesticides is a controversial issue. Avermectin, as mentioned previously, is an anthelmintic pesticide used to treat parasites and insect pests of livestock, which enters the environment through dung (Halley *et al*., 1993; McCracken, 1993; Vickery *et al*., 2001; McCracken and Tallowin, 2004). Anthelmintic drugs used to treat red grouse, administered via medicated grit or direct treatment, can also potentially impact invertebrate growth, reproduction, and survival (Oh *et al*., 2006; Thompson *et al*., 2016). The true impact of anthelmintics on invertebrates, however, has yet to be determined (Thompson *et al*., 2016).

The importance of invertebrate abundance is, however, closely tied to invertebrate accessibility as waders need to be able to successfully find the invertebrates. As with abundance, vegetation structure affects availability with shorter vegetation (via grazing, cutting, and burning) increasing detectability and accessibility of invertebrates; decreasing vigilance rates for predators and thus increasing time available for foraging; and potentially improving wader mobility (Vickery *et al*., 2001; Butler and Gillings, 2004; Devereux *et al*., 2004; Newton, 2004; Whittingham and Evans, 2004; Douglas and Pearce-Higgins, 2014). Other exclusively negative changes include soil compaction from high stocking densities and agricultural machinery (Vickery *et al*., 2001; Chesterton, 2009), and lower soil moisture (via climate change and land drainage) leading to reduced soil-probing ability during foraging and shifting of earthworms deeper into the soil (Green *et al*., 1990; Smart and Gill, 2003).

1.3.4 Climate change

High emission climate change scenarios for the UK predict increasingly higher temperatures (0.7-4.2°C (winter) and 0.9-5.4°C (summer) increase by 2070; 10 and 90% probabilistic forecast, UKCP18), with higher winter and lower summer precipitation (Met Office, 2019). In recent decades (1961-2000), mean temperature and overall winter precipitation have already risen in the UK uplands (Burt and Holden, 2010). These changes in climatic conditions could alter breeding wader distributions (Huntley *et al*., 2008), and impact other key factors affecting wader population sizes and breeding success (which could in turn affect distribution): habitat quality, invertebrate resources, phenology, and predation (Smart and Gill, 2003; Renwick *et al*., 2012; Franks *et al*., 2017). Indirect effects of climate change on land use by humans (for example, conversion of grassland to bioenergy crops in the

Netherlands) poses an additional, overlooked consequence with the potential to exert a greater effect on waders than direct ecological impacts (van Dijk *et al*., 2015). As climate change is a large-scale threat, the impacts on all four wader groups have been investigated. According to Martay *et al*. (2017), climate change may already have contributed to overall snipe declines in Great Britain. Nevertheless, species-specific responses will differ due to variations in breeding range, distribution, and ecology (Smart and Gill, 2003; Renwick *et al*., 2012).

For habitat specialists (for example, dunlin; Lavers and Haines-Young, 1996; Lavers *et al*., 1996), and species at their southern and western range and altitudinal limits (for example, dotterel, greenshank, and golden plover; Thompson and Thompson, 1991; Pearce-Higgins *et al*., 2010; Hayhow *et al*., 2015), climate change will exert a stronger force (Pearce-Higgins, 2010; Chamberlain *et al*., 2012; Pearce-Higgins *et al*., 2015). Climate change-related habitat loss, together with other anthropogenic-related reductions in habitat quality, could result in distribution shifts for such species. For example, peat bog cover would decline by 20% and all montane habitat would be lost (breeding habitats used by wader groups 2 and 3, respectively) with a 3°C temperature increase, due to lowered water tables and conversion to drier soil-associated vegetation compositions including heath and woodland (Smart and Gill, 2003). Moreover, soil acidification from air pollution- and agriculture-associated nitrogen deposition, along with intensive sheep grazing, can further alter vegetation composition by homogenising and lowering diversity (van der Wal *et al*., 2003; Ross *et al*., 2012; Hayhow *et al*., 2015; Ewing *et al*., 2020), exacerbating habitat loss and thus breeding wader range changes.

For other less geographically restricted species (wader groups 1 and 4), habitat degradation rather than loss is more likely (Smart and Gill, 2003). Of key importance is climate change-related impacts on invertebrate prey resources. Whilst the activity and abundance of some invertebrates may increase in warmer temperatures (Smart and Gill, 2003), adverse impacts on some invertebrate taxa are already evident (Martay *et al*., 2017) with further declines expected as climate change progresses. For example, lower water tables and resultant drier soils decrease the abundance and availability of important moisture-associated invertebrates such as cranefly larvae and earthworms (Pearce-Higgins *et al*., 2005; Pearce-Higgins *et al*., 2010; Pearce-Higgins, 2010; Carroll *et al*., 2015). Such climate-associated substrate aridity will exacerbate the effects of land drainage (Pearce-Higgins, 2010). Warmer, drier conditions can also increase wildfire prevalence, particularly on moorland, further reducing invertebrate abundance, destroying wader breeding habitat, and potentially killing wader nests and young chicks (McMorrow *et al*., 2009; Albertson *et al*., 2010; Grant *et al*., 2012; for discussion of impacts associated with prescribed moorland burning, see section 1.3.1.2). Climate change-related changes in vegetation composition and structure (as discussed above) could also influence invertebrate communities and their availability for foraging waders (see section 1.3.3).

Not only could changes in climatic conditions influence invertebrate prey through altered habitat conditions, warmer temperatures could also trigger earlier emergence and increased development rates in invertebrates, leading to reduced availability during the chick rearing period (Smart and Gill, 2003; Pearce-Higgins and Yalden, 2004). Such phenological mismatch, the extent of which depends on the advancement rates of nesting by waders, has the potential to impact all upland waders. Although, changes in phenology are of potentially greater importance to long-distance migrants (for example, common sandpiper; Dougall *et al*., 2010) as they cannot predict weather conditions and thus habitat conditions or invertebrate prey availability at the breeding sites (Holland and Yalden, 1991; Stenseth and Mysterud, 2002; Smart and Gill, 2003). At present, however, evidence supporting phenological mismatch impacts is lacking (Pearce-Higgins, Yalden, *et al*., 2009; Chamberlain and Pearce-Higgins, 2013) with impacts on invertebrate abundance more important than phenology (Pearce-Higgins *et al*., 2010). Waders do appear capable of responding to warmer springs by exhibiting earlier laying-dates (Kruk *et al*., 1996; Both *et al*., 2005), but the rate of advance is critically important; slower advancement of farming activities compared to nesting by lapwing and curlew can result in the destruction of nests by agricultural machinery (Santangeli *et al*., 2018). Earlier laying-dates due to climate change could also enable longer breeding seasons resulting in additional opportunities for replacement clutches following clutch or brood loss, as long as soil conditions remain suitably wet for species such as snipe which primarily forage by probing the soil (Green, 1988; Smart and Gill, 2003). Nevertheless, there is potential for phenological mismatch to become more prevalent under future climate change scenarios (Pearce-Higgins *et al*., 2005; Chamberlain and Pearce-Higgins, 2013).

On the other side of the trophic chain to invertebrates, predation pressure upon waders could increase in response to climate change-related impacts on vegetation structure (see above). In addition to habitat changes and impacts on invertebrate prey, altered vegetation structure could influence nest and chick detection by predators, and the availability of alternative prey for predators (see section 1.3.2).

1.3.5 Human disturbance

Rural commercial activities (most likely from agriculture) and recreational disturbance, by humans and dogs, has the potential to impact upland waders in all breeding habitats (Watson, 1988; Watson *et al*., 1988; Yalden and Yalden, 1990; Haworth and Thompson, 1990; Hockin *et al*., 1992; Yalden, 1992; Finney *et al*., 2005; Pearce-Higgins *et al*., 2007) with increased public open access land granted in upland areas in 2000 (UK Parliament, 2000; Finney *et al*., 2005). The impacts of human disturbance on all four wader groups have been studied. Disturbance can affect breeding success, nest-site choice, territorial disputes, foraging behaviour, population density, and community structure either directly through displacement or indirectly through behavioural changes (reduced incubation, abandonment of nests, reduced parental care of chicks) which can lead to increased predation rates (van der Zande *et al*., 1980; Iversen, 1984; Yalden and Yalden, 1989; Yalden and Yalden, 1990; Haworth and Thompson, 1990; Hockin *et al*., 1992; Verhulst *et al*., 2001).

Impacts may differ due to the type and intensity of disturbance. High daily numbers of walkers can displace waders from breeding sites; although, installation or re-surfacing of footpaths can reduce displacement distances as fewer walkers stray from the path, despite increased visitor numbers (Pearce-Higgins and Yalden, 1997; golden plover: Finney *et al*., 2005; Pearce-Higgins *et al*., 2007; dunlin: Pearce-Higgins *et al*., 2007). In addition, anglers can also disturb upland waders as common sandpipers were flushed 29% more than in angler-free areas, resulting in reduced breeding densities (Yalden, 1992). Nevertheless, an effect is not always observed as Scottish breeding dotterel were not affected by increased human traffic in spring and summer at ski slope developments (Watson, 1988). In addition, frequent nest visits (every four days) by researchers was not found to decrease lapwing nest survival (Fletcher *et al*., 2005).

1.3.6 Disease and parasitism

The impact of disease and parasitism on upland waders is a neglected area of research. Only two UK studies have specifically investigated the impact of an ectoparasite, sheep tick *Ixodes rinicus,* and its associated disease, Louping-ill virus (LIV) on upland wader chick survival (Newborn *et al*., 2009; Douglas and Pearce-Higgins, 2019). Infestations of sheep ticks and LIV infection significantly increase grouse chick mortality rate by 78% (Gilbert, 2016). Wader chicks also carry sheep ticks. Newborn *et al*. (2009) found, on moorland with acaricide-treated sheep, that curlew chicks exhibit a high susceptibility to infestation with 91% of chicks infested compared to 47% for golden plover and 6% for lapwing. Despite different infestation rates, chicks of all three species carried relatively few ticks, potentially explaining the lack of effect on body condition and 0% LIV sero-positivity (Newborn *et al*., 2009). On an acaricide-free moorland, however, Douglas and Pearce-Higgins (2019) found that 90% of golden plover chicks (c.f. 47% in Newborn *et al*. (2009)) were infested and had a considerably higher mean tick load with lower chick survival, suggesting that tick management might reduce overall sheep tick abundance (Newborn and Baines, 2012). Because Newborn *et al*. (2009) found that curlew had high susceptibility to tick infestations compared to golden plover and lapwing, despite tick management, tick-related chick mortality could be an important factor contributing towards poor breeding success for curlew in acaricide-free areas. Thus, further research in untreated areas is required for wader groups 1 and 2, such as curlew chicks. Groups 3 and 4 may be less prone to the impacts of sheep ticks as montane zones and watercourse edges are less likely to be used for sheep farming or grouse shooting.

To fully understand the impact of sheep ticks on waders, the process by which ticks increase the mortality of chicks and the potential for tick-related chick mortality to limit wader populations must be researched. Other factors which could also be considered include upland sheep farming and grouse moor extent; alternative host species abundance (red grouse, sheep *Ovis aries*, red deer *Cervus elaphus*, and roe deer *Capreolus capreolus,* lagomorphs) and its impact on tick prevalence; and extent of acaricide treatment in sheep and red grouse (Fuller and Gough, 1999; Ward, 2005; Scharlemann *et al*., 2008; Jeffries *et al*., 2014; Baines and Taylor, 2016; Sotherton *et al*., 2017; Fletcher and Baines, 2018).

1.4 Conservation interventions

Current measures aimed at the conservation of waders primarily focus on threats associated with agriculture and predation. Interventions for some threats are likely to alleviate the impacts of others through shared adverse impacts, such as livestock farming and food resources. However, as discussed previously, a comprehensive understanding of many threats is limited. The conservation interventions listed below for each threat are therefore unlikely to be entirely effective.

1.4.1 Land-use

1.4.1.1 Intensification of livestock farming

The principal conservation intervention dealing with adverse implications of livestock farming is agri-environment schemes (AES). Whilst research is lacking regarding the effects of upland AES on waders (Batáry *et al*., 2015), AES have been implemented across 90% of upland grazing livestock land in England (mostly broad Entry Level options; Natural England, 2009) and waders are still declining signifying that current prescriptions are ineffective (for example, Smart *et al*., 2013). Potentially inaccurate perceptions of some practices, such as drainage ditch blocking reducing sheep grazing conditions (Wilson *et al*., 2011; Reed *et al*., 2017), could deter some prescription uptake. When successfully applied however, AES can ameliorate local population declines when targeted prescriptions reduce grazing, agrochemical application, and agricultural activities such as mowing during the breeding season (O'Brien and Wilson, 2011; Franks *et al*., 2018) and are applied over a sufficiently large proportion of the landscape (Whittingham, 2007; Dallimer *et al*., 2010; Franks *et al*., 2018). It seems to be the case, however, that despite generally positive effects, the extent of the impact is not enough to stabilise overall wader population declines (Franks *et al*., 2018).

Based on current AES and the information collated for this review, the following measures could be implemented to ease, if not remove, agricultural pressures facing upland waders. Direct changes to livestock involve reducing stocking densities, switching to cattle or mixed grazing systems where feasible, and/or converting from spring to autumn grazing. This would prevent overgrazing, encourage more heterogenous vegetation structures and minimise soil compaction, creating conditions beneficial for a wider range of nesting upland waders, and improving invertebrate accessibility and abundance (Hope *et al*., 1996; Vickery *et al*., 2001; Newton, 2004; Whittingham and Evans, 2004; Dennis *et al*., 2008; Natural England, 2012; Evans *et al*., 2015). It should be noted, however, to not abandon grazing land as under-grazing could prove equally as detrimental as overgrazing to waders through resultant dominance of rank vegetation (Pollock *et al*., 2013; Johnstone *et al*., 2017; Kamp *et al*., 2018). Nest and chick survival would also benefit from these changes in livestock practices by reducing trampling, and reducing predation from livestock and other predators due to provision of appropriate nesting habitats and cover for chicks (Newton, 2004; Whittingham and Evans, 2004). Reducing stocking densities alone, however, would not reverse moorland to grassland conversion. A combination of the first two direct livestock changes (reducing stocking densities, and switching to cattle or mixed grazing systems), as well as re-seeding, are required (Hulme *et al*., 1999; Gardner *et al*., 2002; Gardner *et al*., 2009; Anderson *et al*., 2009). As previously discussed, however, vegetation composition may not be as important as vegetation structure.

Other agricultural changes which would benefit breeding waders include: blocking of drainage ditches to increase soil moisture (Newton, 2004); no inorganic and reduced organic fertiliser application to benefit invertebrate populations; conversion of silage to hay production to avoid nest and chick mortality (Vickery *et al*., 2001; Newton, 2004; Jefferson, 2005); creation or extension of wet features such as scrapes and pools for foraging (Natural England, 2012; Smart *et al*., 2013); sub-soiling and spiking to improve soil aeration for foraging (Vickery *et al*., 2001; Natural England, 2012); rush cutting to increase sward heterogeneity (Robson and Allcorn, 2006; Holton and Allcorn, 2006; Fisher and Walker, 2015); and scrub removal to remove avian predator perches (Natural England, 2012). To make these interventions feasible, however, government support for landowners is necessary (Gardner *et al*., 2009; Franks *et al*., 2018).

Despite this assortment of available interventions to minimise livestock farming impacts, several of which are available within UK AES prescriptions, evidence suggests lack of uptake of targeted prescriptions at the landscape scale could limit the magnitude of beneficial impacts for farmland birds such as waders (Perkins *et al*., 2011; Baker *et al*., 2012; Burgess *et al*., 2015; Franks *et al*., 2018). Moreover, reported benefits for some interventions are more anecdotal than evidence-based and thus require improved monitoring. One such intervention, rush management, lacks scientific evidence with only two studies detailing increased wader numbers in response to rush cutting in conjunction with other interventions such as scrape creation (Robson and Allcorn, 2006; Holton and Allcorn, 2006). Localised population increases do not necessarily indicate overall population increases or improved breeding success. Individuals may have moved in from surrounding areas, and breeding failure rates may still be high. Indeed, effects of rush management on breeding success are unknown. Chick survival is a key unheeded aspect of AES monitoring, which receives much less attention than nest survival (Franks *et al*., 2018). When aiming to benefit breeding success, AES evaluation must ensure it is monitored and both nest and chick survival are considered.

1.4.1.2 Game management (and disease and parasitism)

To combat overgrazing caused by elevated red deer populations, deer culls focusing on hinds should be instated due to lack of natural predators (Bullock *et al*., 1998). The common practices of heather burning, vegetation cutting, and livestock (primarily sheep) grazing should also be examined to understand the best method for managing moorland vegetation for different wader species (Trenkel *et al*., 1998; Tharme *et al*., 2001; Davies *et al*., 2010; Grant *et al*., 2012). Moreover, where burning is implemented, this should stop at the end of March prior to the start of the wader breeding season to prevent destruction of early nests (Harper *et al*., 2018). Yet, to eliminate the wider negative environmental impacts of burning whilst retaining the apparent beneficial shorter vegetation patches for some nesting waders such as lapwing and golden plover (Pearce-Higgins and Yalden, 2004; Pearce-Higgins and Grant, 2006; Grant *et al*., 2012), rotational burning should be substituted for cutting where possible (Douglas *et al*., 2017).

Leaving aside the issue of illegal raptor persecution associated with grouse moors, another aspect of game management that could adversely impact upland waders is the high host abundance for sheep ticks (see section 1.3.6). Using current evidence from the two studies investigating sheep ticks and wader chicks (Newborn *et al*., 2009; Douglas and Pearce-Higgins, 2019), lowering the artificially inflated densities of alternate hosts to wader chicks (i.e. red deer, red grouse, and sheep) could lessen any potential adverse effects of ticks. One of the main priorities, however, should be to investigate whether tick-related chick mortality can limit wader populations, and if so the mechanism by which ticks increase mortality rates, before calling for tick management via methods such as acaricide treatment of red grouse and sheep (Douglas and Pearce-Higgins, 2019).

1.4.1.3 Afforestation

Overall, research into the impacts of upland forestry provide a consensus of its negative effects on waders. It should be noted, however, that native woodland in upland regions, such as clough woodland, is an important habitat for species other than waders (Fuller *et al.*, 2007). The UK is among the least forested countries in Europe, with historical woodlands potentially lost anthropogenically and climatically from upland regions below the former tree line (for example, Smith and Cloutman, 1988; Fyfe *et al.*, 2003; Atkinson and Townsend, 2011). Increasing the proportion of woodland in the UK, ideally with native species, would benefit woodland species and the UK ecosystem (Fuller *et al.*, 2007). Managing the land for waders at the expense of other vulnerable species would be ill-advised. Therefore, expansive conifer plantation woodlands should be removed, and afforestation projects should be avoided in large flat areas with high densities of breeding waders (Hancock *et al.*, 2009; Douglas *et al.*, 2014; Hancock *et al.*, 2020), yet permitted in areas which would not reasonably be targeted for habitat restoration to encourage wader recolonisation. In addition, as woodlands harbour predators such as red fox, and foxes are known to travel on average 0.6 km and 1.3 km from their dens respectively during the spring and summer, increased predator control at wader breeding sites situated close to woodland could improve wader breeding success (Douglas *et al.*, 2014; Meisner *et al.*, 2014).

1.4.1.4 Wind farms

Although the quantity and type of research regarding wind farms and waders needs improvement, some general conservation interventions could be applied. Installation of wind farms could be avoided in key upland breeding areas, and construction could be restricted to the non-breeding season (Drewitt and Langston, 2006; Pearce-Higgins, Stephen, *et al.*, 2009; Pearce-Higgins *et al.*, 2012; Sansom *et al.*, 2016). Moreover, turbines could be installed in groups rather than lines to reduce the barrier effect and avoid flight paths (Drewitt and Langston, 2006). New automated camera technology that reduces turbine rotation when an avian collision risk is detected (McClure *et al.*, 2021), as well as pioneering bladeless turbines (Vortex Bladeless, 2015), could also be implemented, although as previously discussed (see section 1.3.1.4) evidence suggests collision risk is low for waders (Pearce-Higgins *et al.*, 2012; Sansom *et al.*, 2016). To ensure that these measures are effective, pre- and post-development monitoring, needs to be undertaken (Drewitt and Langston, 2006; Pearce-Higgins *et al.*, 2012).

1.4.1.5 Peat extraction

Evidenced by previously successful peatland protection and restoration projects in Europe, UK peatland management should follow suit (Vasander *et al.*, 2003; Marsden and Ebmeier, 2012; Fraixedas *et al.*, 2017). Prevention of new extraction projects and restoration of former extraction sites are required in areas with high breeding wader densities (Lavers and Haines-Young, 1996). In addition, to deal with degradation exacerbated by associated practices, drainage ditches should be blocked, and grazing and burning should be reduced (Vasander *et al*., 2003; Marsden and Ebmeier, 2012; Artz *et al*., 2018). Reseeding of bare peat could also be applied depending on local conditions (Artz *et al*., 2018). This will promote revegetation and pool formation, improving potential invertebrate resources for foraging waders (Joint Nature Conservation Committee, 2011; Hannigan *et al*., 2011; Beadle *et al*., 2015; Elo *et al*., 2015; Brown *et al*., 2016).

1.4.2 Predation

Dealing with the threat of predation depends on the predator species present and the densities of those species. Effective predator control, both lethal and non-lethal, should therefore be decided on a site-by-site basis (Bolton *et al*., 2007; Teunissen *et al*., 2008). Partially due to the misinterpretation that predation is always a problem, predator control can be ineffective at increasing wader populations and breeding success (Parr, 1993; Bolton *et al*., 2007; Isaksson *et al*., 2007; Ausden *et al*., 2009; Fletcher *et al*., 2010; Bodey *et al*., 2011; Buchanan *et al*., 2017). Sites where predation is limiting wader productivity, however, can exhibit successful predator control (Jackson, 2001; Tharme *et al*., 2001; Bolton *et al*., 2007; Isaksson *et al*., 2007; Fletcher *et al*., 2010; Smith *et al*., 2011; Rickenbach *et al*., 2011; Malpas *et al*., 2013; Douglas *et al*., 2014; Buchanan *et al*., 2017; Ludwig *et al*., 2019).

Lethal control typically involves killing red fox, corvids (carrion crow and hooded crow *Corvus cornix*) and mustelids (stoat *Mustela erminea*, weasel *Mustela nivalis*, and feral ferret *Mustela furo*). Meso-predator release (for example, stoat), compensatory predation, and protection of certain predator species in Britain (for example, badger *Meles meles*; Great Britain, 1992; Parr, 1993; Ausden *et al*., 2009; Ritchie and Johnson, 2009; Bodey *et al*., 2011) are potential issues regarding lethal predator control, especially if these species are limiting wader populations. Non-lethal control involves the use of nest exclosures which exclude large mammalian and avian predators, or exclusion fences which typically exclude large mammalian predators, and can exclude European hedgehog *Erinaceus europaeus* depending on fence design (Jackson, 2001; Smith *et al*., 2011; Rickenbach *et al*., 2011; Malpas *et al*., 2013). Neither technique excludes small mustelid predators (stoat, weasel, and feral ferret) and exclusion fences cannot impede avian predators (Nol and Brooks, 1982; Johnson and Oring, 2002; Ivan and Murphy, 2005; Smith *et al*., 2011). A key issue surrounding nest exclosures concerns incubation behaviour. When accepted by incubating adults, which occurs most of the time, nest survival improves (Smith *et al*., 2011); yet for species which sit tight on nests, such as redshank, this non-lethal control method can prove fatal as adults are inhibited by the cage and cannot escape predators quickly (Nol and Brooks, 1982; Johnson and Oring, 2002; Murphy *et al*., 2003; Neuman *et al*., 2004; Isaksson *et al*., 2007; Smith *et al*., 2011). This is not an issue with exclusion fences which are typically used to protect aggregations of breeding lapwing (Rickenbach *et al*., 2011). Exclusion fences, however, may only be appropriate on enclosed in-bye rather than open moorland due to installation and maintenance issues on large areas of moorland (Ausden *et al*., 2009). Nevertheless, recent projects in the UK and Germany have successfully used temporary electric fencing to protect single curlew nests, demonstrating the potential for use on unenclosed moorland in some cases (M. Shurmer, personal communication, 08 January 2018; Meyer, 2017).

Lethal and non-lethal control can prove effective at increasing breeding wader productivity where predation poses a key threat (Tharme *et al*., 2001; Bolton *et al*., 2007; Isaksson *et al*., 2007; Fletcher *et al*., 2010; Rickenbach *et al*., 2011; Malpas *et al*., 2013; Douglas *et al*., 2014; Buchanan *et al*., 2017). Control techniques should be chosen based on the target wader and culpable predator species with monitoring to check if predation rates are reduced or if alternative predators increase their predation rates. In the first instance, non-lethal control should be used unless small mustelids are the primary predators. If this is not feasible or successful, lethal control can be considered for non-protected predators, such as red fox, corvids, and small mustelids. Non-lethal predator exclusion fences can be used for all wader species but are most appropriate for those nesting at relatively high densities on in-bye land. Nest exclosures, however, are only successfully tested on lapwing so should be used exclusively for this species (Isaksson *et al*., 2007). If red fox and/or legally protected badgers are the primary predator(s) exclusion fences, or nest exclosures, are likely to be effective. If the primary predator is avian, nest exclosures could be used to protect lapwing. In other situations, diversionary feeding may be useful (Redpath *et al*., 2001; Kubasiewicz *et al*., 2016; Smart and Amar, 2018); many aspects need to be considered depending on the target predator and wader species (type and quantity of food; feeding site location; timing of feeding; potential feeding of non-target predators) before feeding is established to increase the likelihood of success (Smart and Amar, 2018). A less contentious technique for all predation scenarios, however, is to improve wader habitat at the landscape scale (Roos *et al*., 2018); create heterogeneous vegetation structures to provide suitable nesting sites, cover for chicks, and foraging areas (Whittingham *et al*., 2001; Pearce-Higgins and Grant, 2006; Smart *et al*., 2006; Roos *et al*., 2018). Habitat improvement or diversionary feeding could complement predator control or replace it, if sufficiently effective. Predator control is costly in terms of finances and effort, and is unlikely to be viable in the long-term, but can be an effective interim measure whilst habitat management is conducted (Roos *et al*., 2018). Understanding and dealing with the underlying causes of increased predator populations is highly important for ecological balance to be reinstated.

1.4.3 Reduced invertebrate resources

The conservation interventions discussed here are linked with other connected threats as many land-uses and climatic change impact invertebrate resources. Interventions need to primarily focus on increasing invertebrate populations and their accessibility. As Hallmann *et al*. (2017) found that flying insect populations declined by 75% in 27 years on German nature reserves, it is likely that declines have been worse in the general countryside, including agricultural land, which is principally not managed for wildlife. This pattern of widespread decline almost certainly is the case for many soil-dwelling species given widespread problems including soil drainage and acidification (Galbraith *et al*., 1993; Hoodless *et al*., 2007; Pearce-Higgins *et al*., 2010; Douglas and Pearce-Higgins, 2014; Carroll *et al*., 2015; McCallum *et al*., 2016; Franks *et al*., 2017).

To increase invertebrate populations, both vegetative and edaphic conditions need to be improved. Regarding vegetative conditions, creation of heterogeneous vegetation structure and high plant diversity will provide a wide range of structural and compositional conditions attracting a greater abundance and diversity of invertebrates (Vickery *et al*., 2001; Newton, 2004; Buchanan *et al*., 2006). This could be achieved through conservation interventions regarding livestock farming. In terms of edaphic conditions, blocking drainage ditches and creation or extension of wet features such as pools or scrapes will increase soil moisture, benefitting key invertebrate prey such as Tipulidae larvae and creating soft substrates for feeding (Vickery *et al*., 2001; Newton, 2004; Buchanan *et al*., 2006; Smart *et al*., 2013). Soil aeration will also improve foraging conditions. Sub-soiling and spiking ameliorate soil structure, benefiting surface (slug) and sub-surface (earthworm) invertebrates by alleviating soil compaction (Vickery *et al*., 2001). Furthermore, as soil acidification is an important factor associated with lapwing declines, liming areas of acidic grassland could produce less acidic soils and increase earthworm densities for older foraging lapwing chicks (McCallum *et al*., 2015). Low to moderate rates of farmland manure could be applied on grassland to increase soil invertebrate populations, such as larval insects and earthworms. High levels of manure and any application of inorganic fertiliser, however, should be avoided due to overall adverse effects on invertebrates (Vickery *et al*., 2001; Buchanan *et al*., 2006).

Direct mortality of invertebrates through agricultural and grouse moor management-associated pesticides is a more controversial issue which should also be questioned. Avermectin, a pesticide used to treat parasites and insect pests of livestock, enters the environment through dung. Although invertebrates could potentially avoid avermectin-infected dung (Halley *et al*., 1993; McCracken, 1993; Vickery *et al*., 2001), spreading manure from avermectin-treated livestock could negate the intended benefits of farmland manure application; avermectin-free sources should be used. Anthelmintic drugs used to treat red grouse, administered via medicated grit or direct treatment, can also impact invertebrate growth, reproduction, and survival (Oh *et al*., 2006; Thompson *et al*., 2016). The true impact on moorland invertebrates has yet to be determined (Thompson *et al*., 2016), but a moratorium on its use on upland grouse moors should be considered.

1.4.4 Climate change

The effects of climate change are not yet particularly evident and are difficult to predict, but are likely to become more pronounced in the coming years as climate change progresses (Smart and Gill, 2003; Franks *et al*., 2017; Machín *et al*., 2019). As the threat cannot be halted by local conservation interventions, site-based adaptation management is required to reduce the impact on upland waders (Pearce-Higgins, 2011; Greenwood *et al*., 2016). This management can attempt to weaken the adverse changes that climate change generates, otherwise known as counteracting management, such as increasing soil moisture via blocking of drainage ditches (Pearce-Higgins, 2010; Pearce-Higgins, 2011; Carroll *et al*., 2011). It can also take the form of compensatory management, which involves manipulating an alternative mechanism to increase the population, such as predator control. The most successful conservation intervention would combine the two management options to improve the resistance of upland waders to these impending climatic events (Pearce-Higgins, 2011).

1.4.5 Human disturbance

The ideal measure to remove disturbance would be to limit access to key upland wader breeding sites during the breeding season. This is unlikely to be possible for disturbance caused by rural commercial activities such as those relating to agriculture; recreational disturbance is, however, more manageable. For example, the density of anglers permitted along upland waterways could be limited by installing refuge areas (Yalden, 1992). Nevertheless, it may not be possible to limit access in some areas. Therefore, as mentioned in the corresponding threat section (1.3.5), installing or re-surfacing footpaths away from key wader breeding sites will reduce the likelihood of disturbance and the public straying from footpaths (Pearce-Higgins and Yalden, 1997; Finney *et al*., 2005; Pearce-Higgins *et al*., 2007), narrowing the width of the area used by walkers. Moreover, as walkers are often accompanied by dogs, which trigger a greater response from breeding waders than humans (Gómez-Serrano, 2021), keeping dogs on a leash to also prevent them straying from footpaths would be beneficial.

1.5 Research synthesis and knowledge gaps

This review has confirmed that the threats facing upland waders are linked in a complex web (Fig. 1.1). Universal pressures impacting all species comprise intensification of livestock farming, predation, reduced invertebrate resources, climate change, and recreational disturbance. Moorland breeding waders (wader groups 1 and 2) appear to be under more pressure than other upland waders due to additional moorland-associated threats: game management, afforestation, peat extraction, wind farms, and disease and parasitism (sheep ticks). In light of these conclusions, however, knowledge gaps and inconsistent findings have been identified with sufficient research lacking for several threats and associated conservation interventions.

We use the evidence presented above to summarise current understanding of the threats facing upland waders and the effectiveness of potential conservation interactions. We assign an overall confidence score (low (L), medium (M), or high (H)) for each evidence-based statement (see sections 1.5.1 and 1.5.2), based on the methodology used for reporting climate change impacts (for example, see Morecroft and Speakman (2015)), using the level of agreement and quantity of the underlying evidence included in this review (Fig. 1.2). Level of agreement is defined by the degree of consensus between studies, and quantity of evidence is defined by the amount and quality of evidence. The amount of evidence is based on the number of research papers found during the literature review search. The quality of evidence is based on a consideration of the magnitude of replication (spatially, i.e. within and across study sites/regions and temporally, i.e. duration of the study), study design (for example, studies using a before-after control-impact (BACI) design are considered higher quality than only after studies), and data collection and analysis methods. In addition, peer reviewed material is considered higher quality than non-peer reviewed material. The overall confidence scores act as a guide for identifying priorities for future research, with the rationale for the assignment of each confidence score presented in Table A.1.

1.5.1 Overall confidence scores for threats

Overall confidence scores for the five land-use threats range from low to high (Fig. 1.2). Intensification of livestock farming in the uplands can cause changes in habitat quality, reductions in invertebrate accessibility and abundance, and increased egg and chick mortality rates (H). Heather burning on grouse moors and deer hunting estates appears to be detrimental for some wader species and beneficial for others (M). Predator control as part of grouse moor management improves wader breeding success and abundance (M). Afforestation displaces waders and lowers breeding success (M). Wind farms negatively impact some upland waders (snipe, curlew, and golden plover) by reducing breeding densities, but some species do not appear to be adversely affected (lapwing and dunlin; M). Peat extraction reduces habitat quality for peatland-breeding waders, with drainage extending the spatial extent of this land-use (L).

The discussed upland land-uses (intensification of livestock farming, game management, wind farms, afforestation, peat extraction) are known to impact invertebrate resources through various aspects of the associated land management practices; yet the extent of this effect and whether invertebrate declines are a primary driver of wader breeding success and population declines are unclear (L). High predation rates of wader nests and chicks reduces breeding success and limits wader populations (H). Climate change will alter population dynamics of upland waders (M). Human disturbance can alter upland wader distribution (M). Sheep ticks and its associated disease, Louping-ill virus could adversely impact wader chick survival, but it is unlikely to be a significant contributor to wader declines (L).

1.5.2 Overall confidence scores for conservation interventions

Whilst most conservation interventions have been designed based on empirical evidence concerning wader ecology and habitat requirements, there have been very few empirical assessments of the effectiveness of these interventions and thus in most cases confidence in the ability of these interventions to increase breeding wader populations is low (Fig. 1.2). The exceptions are conservation interventions relating to the intensification of livestock farming and predation. AES, the principal intervention used to manage intensive livestock farming, can improve wader breeding habitat at the local scale but not at the landscape scale. Targeted prescriptions need to be applied over a larger area to provide landscape scale benefits (M). Lethal and non-lethal predator control increases upland wader breeding success and abundance (M).

1.5.3 Future research

Using the overall confidence scores for each threat and associated conservation interventions, and the relative importance of each of the threats for upland waders, key areas for future research can be identified. With only two studies, disease and parasitism seems as if it would benefit from further research, but current evidence suggests this is unlikely to be a significant contributor to wader declines and thus research effort will likely be better spent on other threats. For example, as a large proportion of the UK uplands are farmed, with sheep grazing often on unenclosed moorland as well as enclosed pasture, intensification of livestock farming is the threat with the greatest spatial extent and thus the greatest potential to adversely impact habitat quality. Despite extensive analysis of agricultural practice impacts, evaluation of conservation interventions to improve farmed habitats for waders, particularly AES prescriptions, is deficient. Prescriptions such as those involving rush management thus require further empirical evidence to verify the beneficial effects.

Another land-use that covers a large proportion of the UK uplands is game management, particularly grouse moorland. Like intensification of livestock farming, despite the relatively high number of game management studies, more research effort is required to identify the best management for upland vegetation to benefit moorland-breeding waders. Evidence does, however, show that predator control conducted as part of grouse moor management has an overall positive impact on waders. Yet, as identified within this review, ascertaining and subsequently managing the underlying causes of increased predator populations (the main driver of low wader breeding success; Roodbergen *et al*., 2012; Roos *et al*., 2018) is of key importance because this will lessen the need for costly predator control.

Despite comprising a much lower proportion of the uplands than livestock farming and game management, future studies of wind farms and associated mitigation measures should employ a BACI approach, recognised as the best standard for monitoring the ecological impacts of infrastructure (European Commission, 2020). In the face of climate change (which itself poses a risk to waders) and the drive to reduce greenhouse gas emissions, wind farms could become an increasingly prominent threat. Moreover, because of climate change and land-use threats, invertebrate populations in the uplands are vulnerable but the magnitude of these impacts on invertebrates requires investigation. Due to the invertebrate-feeding nature of waders, reduced invertebrate prey availability has the potential to limit wader populations and thus should be a priority for future research.

1.6 Conclusion

The range of interlinked threats driving the upland wader population declines need to be combatted. As the evaluation of some potentially important threats, such as reduced invertebrate resources and game management, are incomplete due to deficient quantity of evidence, entirely effective conservation interventions cannot be implemented. Thus, to ensure the survival of upland waders in the UK, researchers, landowners, and policy makers need to unite to determine the importance of the unresolved threats and the effectiveness of conservation interventions.

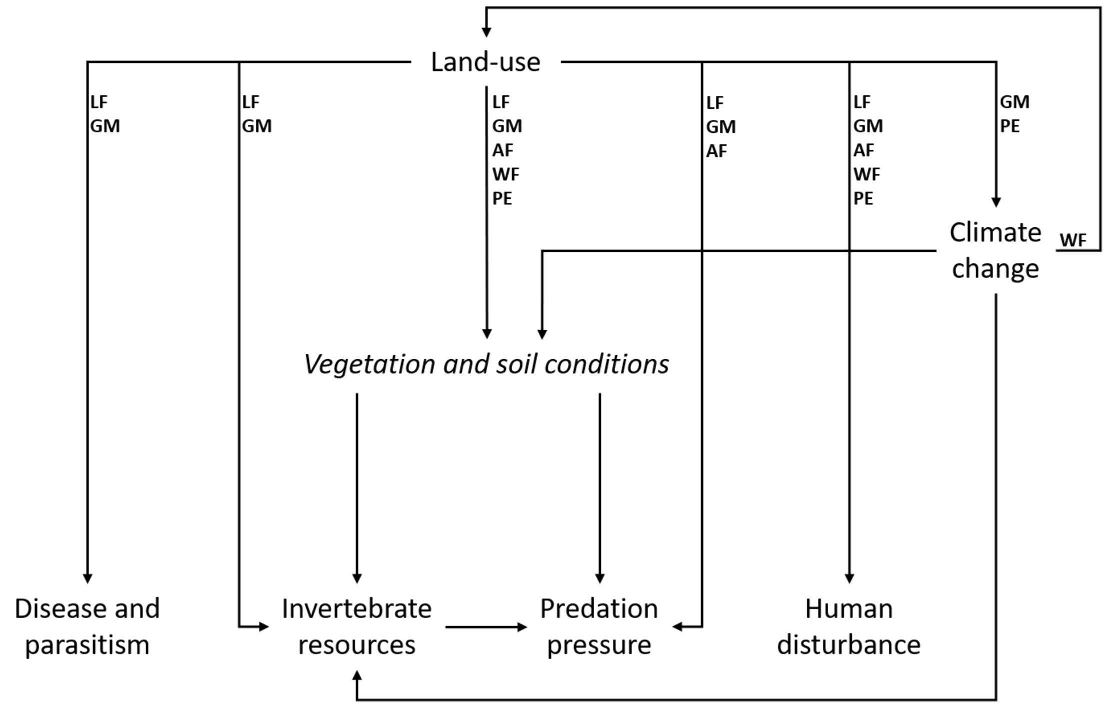

Figure 1.1. Major connections between the threats facing upland waders in the UK. 'Vegetation and soil conditions' is included (italicised) as it is a common factor that is influenced by several threats. Codes adjacent to lines indicate the types of land-use connected to other threats. LF = livestock farming; GM = game management; AF = afforestation; WF = wind farms; PE = peat extraction.

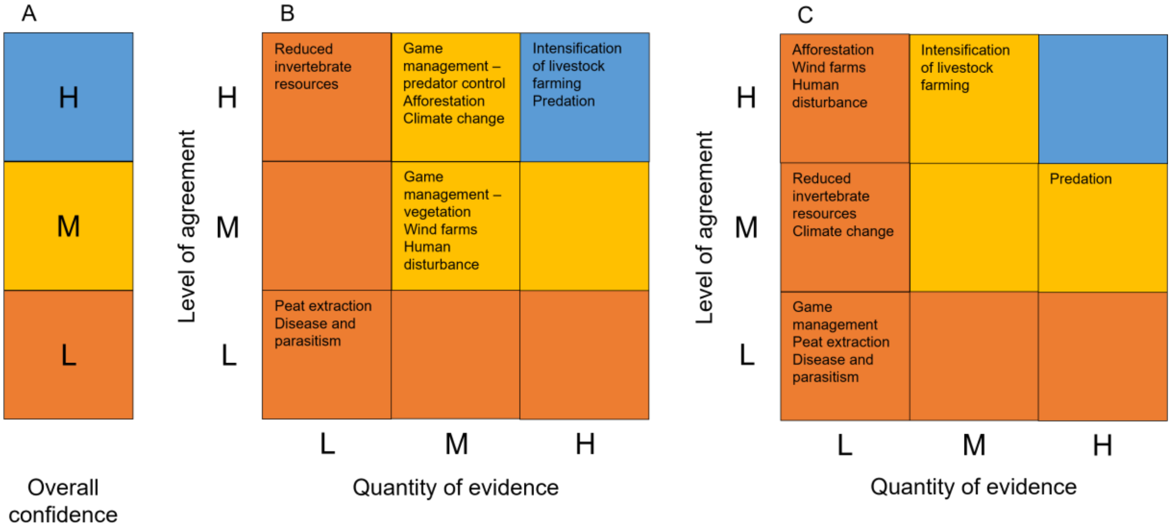

Figure 1.2. A guide for identifying priorities for future research of threats facing upland waders and the conservation interventions to mitigate each threat. A) Overall confidence scores of high (H), medium (M), and low (L) are assigned to a theme based on the amalgamation of level of agreement and quantity of evidence, indicating the areas which should be prioritised for further research. Level of agreement is defined by the degree of consensus between studies. Quantity of evidence is defined by the amount (number of research papers found during the literature review search), and quality (based on a consideration of the magnitude of replication, study design, and data collection and analysis methods) of evidence available. For an overall confidence score of high, the theme must have high level of agreement and high quantity of evidence. For an overall confidence score of low, the theme must be scored as low for either level of agreement or quantity of evidence. B) Assessment of evidence available for the impacts of each threat on breeding upland waders. C) Assessment of evidence available for the conservation interventions to combat each threat facing breeding upland waders.

1.7 Thesis aims and rationale

The overall aim of this thesis is to provide novel information that will help improve conservation for upland waders. It is presented as a publication format thesis because Chapter 3 is published, and the remaining chapters are intended for publication. This thesis forms a part of the South West Peak Landscape Partnership's Working for Waders project, funded by the National Lottery Heritage Fund. Research conducted for this thesis thus focused on the south-west of the Peak District National Park (South West Peak, hereafter "SWP") as the primary study area. The SWP is representative of other upland regions of the UK, comprising a mosaic of upland habitats including semi-improved pasture, hay meadows, and heather moorland. In this region, wader groups 1a (curlew, snipe, and redshank) and 1b (lapwing) are most abundant, with smaller populations of wader groups 2 (golden plover, and dunlin; excluding greenshank) and 4 (oystercatcher, and common sandpiper). Research aims were thus selected based on (i) the most important and understudied threats and conservation interventions facing the key wader groups 1a and 1b, and (ii) available habitat types with the potential for sufficient sample sizes. Encompassing key wader breeding habitats in the SWP, I first quantify invertebrate resources to explore which environmental conditions influence invertebrate abundance and in turn, the potential for invertebrate prey to influence upland wader abundance. With declining invertebrate populations evident, identifying influential environmental conditions is vital for advising effective land management for invertebrate and thus waders. Within the most prevalent habitat type of in-bye pastures, I then focus on a widespread but poorly studied conservation intervention available within agri-environment scheme (AES) prescriptions, rush management, with the potential to influence both foraging and breeding success. This research aims to assess rush management from both ecological and social science perspectives, with the ecological research partially conducted in another upland region outside of the SWP (Geltsdale reserve, Cumbria) to increase the amount of replication and transferability of the research. Although many studies have researched conservation interventions relating to farming, rush management lacks scientific evidence despite its common occurrence, within and outside of AES, across upland areas such as the SWP.

Chapter 2 – Bottom-up effects on upland waders: soil parameters, vegetation structure, and invertebrate resources

Declining invertebrate populations could be an important, understudied contributor to the low breeding productivity that is driving upland wader declines. Low invertebrate prey abundance could reduce adult fitness and produce weaker chicks which are more susceptible to predation. In this chapter, I examine bottom-up effects of environmental conditions on invertebrate resources, and the subsequent effects of these on breeding wader abundance. By sampling across a range of upland breeding habitats, the main objectives of this chapter were to: (i) assess environmental drivers of invertebrate abundance; and (ii) assess the influence of invertebrate abundance on upland breeding wader densities.

Invertebrate sampling was conducted by Leah Kelly, with help from MBiolSci student, Laura Turner. Analysis of invertebrate samples in the laboratory was conducted primarily by Leah Kelly, with help from two undergraduate students, Georgia Clifton-Dey and Hannah Ronan-Brown. Waders were surveyed by Leah Kelly and a host of volunteer breeding wader surveyors: Andy Banks, Bowy den Braber, Cheshire and Wirral Ornithology Society, Frances Horsford (PDNPA), Geraint Richards, Hazel Crowther (PDNPA), Mark Eddowes, Mike Shurmer (RSPB), Paul Beresford, Sarah Bird (PDNPA), Scott Petrek (WWT), Simon Mills, Staffordshire Wildlife Trust). Leah Kelly conducted the primary analyses and acted as lead author for this chapter, with analysis advice and writing edits provided by David Douglas, Karl Evans, and Mike Shurmer. The literature review of upland wader diet presented in Appendix A was conducted by MSc student, Nathaniel Dargue. Historic regional breeding wader data used for site selection was provided by Sara Barrett (Natural England).

Chapter 3 – Upland rush management advocated by agri-environment schemes increases predation of artificial wader nests

This chapter has been published as:

Kelly, L.A., Douglas, D.J.T., Shurmer, M.P., & Evans, K.L. (2021). Upland rush management advocated by agri-environment schemes increases predation of artificial wader nests. *Animal Conservation*. **24**(4), pp.646-658. doi: 10.1111/acv.12672.

Rush management is a conversation intervention available within AES prescriptions. This management practice aims to tackle encroachment of rush in agricultural grasslands by reducing the extent of tall, dense rush and thus improving vegetation structure for waders. Yet, effects on wader breeding success are unknown. This is critically important as high nest predation is one of the primary causes of wader declines. Using artificial wader nests as a proxy for real wader nests, the main objectives of this chapter were to: (i) assess whether artificial nests in fields with rush management experience higher predation rates than those in fields without rush management, and (ii) assess whether vegetation structure surrounding nests influences predation rates. Data collection for this chapter was partially conducted in another upland region outside of the SWP, at Geltsdale reserve in Cumbria.

The manuscript is replicated in its entirety in this thesis, with minor adjustments to the formatting. Leah Kelly conducted the fieldwork and primary analyses, and acted as lead author for this chapter alongside Karl Evans, with analysis advice and writing edits provided by David Douglas and Mike Shurmer.

Chapter 4 – Inter-specific variation in the potential for upland rush management advocated by agri-environment schemes to increase breeding wader densities

This chapter is currently in review at *Frontiers in Ecology and Evolution* as:

Kelly, L.A., Douglas, D.J.T., Shurmer, M.P., & Evans, K.L. Inter-specific variation in the potential for upland rush management advocated by agri-environment schemes to increase breeding wader densities

Another approach to evaluate the effectiveness of rush management is through wader abundance. There is, however, insufficient assessment and understanding of how rush management influences upland waders. Previous studies suggest that rush management can increase wader abundance, but do not compare areas with and without rush management. The objective of this chapter was to assess how the number of breeding wader pairs responds to rush management by surveying waders in fields with and without rush management. Data collection for this chapter was partially conducted in another upland region outside of the SWP, at Geltsdale reserve in Cumbria.

The manuscript is replicated in its entirety in this thesis, with minor adjustments to the formatting. Leah Kelly conducted the fieldwork and primary analyses, and acted as lead author for this chapter alongside Karl Evans, with analysis advice and writing edits provided by David Douglas and Mike Shurmer.

Chapter 5 – Farmers' views and understanding of rush management

Rush encroachment not only impacts waders but also causes problems for farmers by reducing farm productivity. As the majority of rush management is implemented by farmers, it is important to understand the reasons why farmers undertake rush management, and the factors that influence whether management is through AES. Uptake and efficacy of AES is crucial for success as conservation interventions need to be implemented at a sufficiently large spatial scale to generate the desired environmental benefits. Through interviews with SWP farmers, this chapter's objectives were to: (i) assess which factors influence farmers' decisions to participate in AES prescriptions for rush management, (ii) assess farmers' perceived effectiveness of rush management AES prescriptions, (iii) improve our understanding of the factors that motivate farmer to manage rush and whether to do this management within or outside of an AES prescription, and (iv) explore the possibility of improving the efficacy and uptake of rush management AES prescriptions by co-designing prescriptions with farmers.

Interviews were conducted by Leah Kelly, with David Cooper assisting one interview. Transcript organisation was undertaken by Leah Kelly, Joanna Shurmer, and Marina Aucejo. Quantitative and qualitative analyses were primarily conducted by Leah Kelly, with Karl Evans providing overall advice and contributing to the qualitative analyses. Leah Kelly acted as lead author for this chapter, with writing edits provided by David Douglas, Karl Evans, and Mike Shurmer.

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Chapter 2

Bottom-up effects on upland waders: soil parameters, vegetation structure, and invertebrate resources

Invertebrate sampling in an improved pasture field with views over the South West Peak

2.1 Abstract

Declining invertebrate populations are a cause of concern for threatened wading bird species reliant on invertebrates for food. Agricultural intensification is frequently cited as a key driver of invertebrate declines, particularly in Europe where much research has taken place. There are relatively few studies, however, in habitats such as upland grasslands and moorlands in the UK which hold important populations of waders such as threatened Eurasian curlew *Numenius arquata*. In this study, we investigated the environmental conditions that influence invertebrate biomass across upland grassland and moorland in the south-west of the Peak District National Park, England. Key dietary items identified from our literature review of upland wader diet supported the assessment of five invertebrate biomass metrics: total invertebrate dietary components, earthworms, Diptera larvae, Gastropoda, and other key invertebrate dietary components. We then tested whether invertebrate biomass and other potentially influential environmental conditions were associated with breeding densities of Eurasian curlew, northern lapwing *Vanellus vanellus*, and common snipe *Gallinago gallinago* pairs. In the early spring, and the remarkably dry late spring and summer of 2018, we found that other key invertebrate dietary components significantly increased over the wader breeding season whereas the abundance of our other four invertebrate metrics, particularly earthworms, declined. Whilst relationships between biomass and environmental conditions were variable across taxonomic groups, soil pH, soil moisture, and vegetation height were generally the most important for determining biomass out of the edaphic and vegetative conditions considered. Inter-specific variation was also observed in the responses of wader pair densities to biotic and abiotic environmental conditions, although densities did not appear to be driven strongly by invertebrate abundance. Curlew were detected in a high proportion of acidic, moorland sites which contained fewer earthworms, clarifying the significant negative relationship we found between curlew pair density and earthworm biomass. Lapwing pair density was associated with moderate earthworm biomass and low Gastropoda biomass, yet paradoxically both taxonomic groups were associated with the less acidic soils found in agricultural grasslands. For snipe, density effectively decreased, except for a small peak at 0.007 g, as other key invertebrate biomass increased; neither snipe nor this invertebrate metric were significantly associated with any habitat type. Despite these generally negative associations, the limiting potential of vegetation height and soil penetration resistance on food accessibility for waders was not evidenced by our mains effects plus interaction term models. Overall, our results show that invertebrate abundance does not currently appear to be limiting upland wader populations in our study region as other factors such as high predation pressure are likely limiting population size. Nevertheless, the associations we found between environmental conditions, invertebrates, and waders could be used in combination with individual wader species' selected breeding habitats and key invertebrate dietary components to advise upland land management aimed at maximising invertebrate availability and thus bolster wader numbers in areas where availability does regulate them. For example, a mosaic of shorter and taller swards would

respectively benefit Diptera larvae and Gastropoda, and reducing drainage of less acidic agricultural grasslands to promote high soil moisture (without waterlogging) could boost earthworm abundance.

2.2 Introduction

Monitoring of invertebrate populations, primarily in northern and western Europe, has reported worrying population declines (for example, Brooks *et al.*, 2012; Fox *et al.*, 2015, 2021; Valtonen *et al.*, 2017; Montgomery *et al.*, 2020). As this region has experienced increasingly intensive agriculture over several decades, habitat loss and associated agricultural practices such as pesticide application are often cited as primary drivers of invertebrate declines (Wilson *et al.*, 1999; Sánchez-Bayo and Wyckhuys, 2019; Montgomery *et al.*, 2020). Even on German nature reserves, over 75% declines in winged insect biomass between 1989 and 2016 are reported (Hallmann *et al.*, 2017). Such declines are likely worse in the surrounding countryside, including agricultural land, as it is typically not managed solely to benefit wildlife. Supplemental to land use changes, other drivers including light pollution, climate change, and invasive plant species are also a cause of concern (Sánchez-Bayo and Wyckhuys, 2019; Owens *et al.*, 2020; Montgomery *et al.*, 2020; Johansson *et al.*, 2020; Boyes *et al.*, 2021; Tallamy *et al.*, 2021).

Although most research has been European-focused, some ecologically important habitats within Europe lack sufficient assessment of invertebrate populations. Upland grassland and moorland habitats in the UK, for example, have experienced major environmental changes with potential adverse impacts on invertebrates. Such changes include intensification of agriculture, alongside land drainage associated with non-native commercial afforestation, peat extraction, and farming (Baines, 1988; Fuller and Gough, 1999; Holden *et al.*, 2004; Newton, 2020). Indeed, higher livestock densities as a result of these changes have been shown to reduce invertebrate biomass by simplifying sward structures (Dennis *et al.*, 2008; Evans *et al.*, 2015). Exacerbating these upland environmental changes, particularly drainage, climate change is also predicted to reduce the abundance of desiccation-prone taxa such as craneflies (Diptera: Tipulidae) due to higher summer temperatures and lowered water tables (Pearce-Higgins *et al.*, 2010; Carroll *et al.*, 2011; Carroll *et al.*, 2015). Notably, such upland habitats hold important invertebrate-feeding bird species, many of which are exhibiting population declines (Sim *et al.*, 2005; Scridel, 2014; BirdLife International, 2017; Buchanan *et al.*, 2017), and thus the cascading trophic impacts of invertebrate declines are of great concern.

Many terrestrial bird species rely on invertebrates as the primary food source for offspring (due to high protein content and easier digestion) and adults of many species also predominately consume invertebrates (Morse, 1971; Diaz, 1996; Snow and Perrins, 1998). Invertebrate availability is thus a key driver of bird populations that can regulate fecundity (clutch size) especially in income breeders (Poulsen *et al.*, 1998), nest success (Poulsen *et al.*, 1998; Brickle *et al.*, 2000; Møller, 2013; Schöll and Hille, 2020; Seress *et al.*, 2020), post-fledging survival rates (Pearce-Higgins and Yalden, 2002; Pearce-Higgins and Yalden, 2004; Newton, 2004; Douglas and Pearce-Higgins, 2014; Seward *et al*., 2014) and adult survival rates (Green *et al*., 1990; Newton, 2004; Siriwardena *et al*., 2008; Møller, 2013; Seward *et al*., 2014). A small number of studies have directly linked farmland bird declines to reduced invertebrate populations attributable to intensive agricultural practices (Benton *et al*., 2002; Hart *et al*., 2006; Hallmann *et al*., 2014; Bowler *et al*., 2019).

One groups of invertebrate-feeding farmland birds inhabiting upland grasslands and moorlands in the UK are wading birds, populations of which are declining across Europe including in the UK (Siriwardena *et al*., 2017; PECBMS, 2020). Three such species, the Eurasian curlew *Numenius arquata* (hereafter curlew), northern lapwing *Vanellus vanellus* (hereafter lapwing), and Eurasian oystercatcher *Haematopus ostralegus* (hereafter oystercatcher) are now listed as globally Near Threatened and Vulnerable in Europe (BirdLife International, 2015; IUCN, 2020). Low breeding productivity is driving wader population declines (Roodbergen *et al*., 2012) and thus, management of breeding habitats could make a crucial contribution to stabilising and ultimately reversing declines (O'Brien and Wilson, 2011; Fisher and Walker, 2015; Buchanan *et al*., 2017). Research and conservation efforts currently focus on either improving nesting and chick rearing habitats via measures such as manipulation of vegetation structure, or improving breeding success through controlling predation and increasing invertebrate availability (by creating wet features, for example; Eglington *et al*., 2010; Douglas and Pearce-Higgins, 2014; Douglas *et al*., 2014; Buchanan *et al*., 2017; Franks *et al*., 2017). Although invertebrate resources form part of the focus for such interventions, they do not always consider the specific habitat requirements for key invertebrate prey taxa.

Reduced invertebrate resources could diminish fitness of adult waders and produce weaker chicks which are more susceptible to predation (Galbraith, 1988a; Galbraith, 1988b; Galbraith, 1988c; Hegyi and Sasvári, 1998; Smart and Gill, 2003; Pearce-Higgins and Yalden, 2004; Schekkerman *et al*., 2009). Eurasian golden plover *Pluvialis apricaria* (hereafter golden plover) abundance, for example, is linked to reduced adult cranefly abundance (Pearce-Higgins *et al*., 2010), with larval craneflies susceptible to desiccation through low soil moisture (Coulson, 1962). Similar studies exploring the direct effect of invertebrate resources on wader populations are, however, lacking for priority Red-listed species such as curlew (Harris *et al*., 2020). Moreover, while both larval and adult craneflies form a key part of many wader diets, a wider range of invertebrate taxa also require monitoring as diet composition varies between wader species and ages (Buchanan *et al*., 2006; Pearce-Higgins, 2010). Important invertebrate taxa are gleaned by waders from the vegetation and soil surface, and probed from below ground; therefore, environmental changes within all breeding habitat strata have the potential to adversely impact invertebrates and hence require investigation.

Here, we explore bottom-up effects on invertebrate and breeding wader abundance by measuring environmental (vegetative and edaphic) conditions; sampling foliar, surface, and sub-surface invertebrates; and surveying breeding waders across broad upland habitat types within the south-west of the Peak District National Park, England. We first evaluate seasonal variation in invertebrate biomass, investigating differences between the early (start of April to mid-May) and late (mid-May to end of June) wader breeding season. Within these two breeding season stages, we then explore the environmental drivers of invertebrate biomass. Finally, we assess the influence of invertebrate biomass and other potentially influential environmental conditions on the density of upland breeding wader pairs.

2.3 Methods

2.3.1 Study area

Research was conducted in the south-west of the Peak District National Park (South West Peak, hereafter 'SWP'). The region is a mosaic of enclosed grassland (improved, semi-improved, and unimproved) and unenclosed moorland (grassland, dwarf shrub heath, and bog), and representative of UK upland farmed landscapes (primarily sheep and cattle grazing), land management, and land use. It supports populations of breeding waders including lapwing, curlew, and common snipe *Gallinago gallinago* (hereafter snipe).

We selected 42 fields using random stratification based on habitat type (unenclosed heather or *Molinia* dominated moorland; UM), improved pasture (IP) and semi-improved/unimproved pasture (SUP) and wader presence recorded during the most recent comprehensive wader survey in the region (2009; Carr, 2009). The goal was for 25% UM, 25% IP and 50% SUP across our selected fields – with the unequal representation being justified by the greater environmental variation in SUP fields. We sought to ensure that there was variation in the suitability of fields for waders to facilitate testing the core objective of quantifying how breeding wader densities were associated with food availability. Consequently, our random stratification aimed to select a suite of fields (within each habitat type) across which, in 2009, each focal wader species (lapwing, curlew, and snipe) was detected (one-third for each species). We then selected an additional 14 fields from the same landholdings (aiming for the same habitat type ratio using Google Earth aerial imagery in 2018) across which waders were not detected in 2009. Due to long-term wader population declines in our focal region (Carr, 2009) and species' fidelity to breeding sites (Thompson *et al*., 1994; Berg, 1994), this approach (75% wader fields; 25% control fields) increased the likelihood that our surveys would detect waders in a sufficient proportion of our fields relative to the alternative approach of equal weighting towards selecting fields with and without waders in 2009. Our final set of 56 fields deviated slightly from the ideal habitat ratio (Table B.1).

2.3.2 Breeding wader surveys

Breeding wader surveys were conducted following a method which combined elements of the standard protocols of O'Brien and Smith (1992) and Brown and Shepherd (1993). Each field was surveyed once during the early (9/4/2018 to 22/5/2018) and late (23/5/2018 – 30/6/2018) breeding seasons, with at least seven days between the early and late surveys in each field. Surveys were conducted by experienced wader surveyors from 8:30 or two hours post-sunrise (whichever was earlier) to 18:00 or 2 hours pre-sunset (whichever was later). Surveys were only conducted when there was no heavy rain or fog (defined as < 250 m visibility), and wind speeds were less than Beaufort force 5. Surveys consisted of walking to within at least 100 m of every part of the field; shorter distances were used when terrain obscured the view. All detected waders were recorded on a map, except overflying individuals that were not displaying, with symbols to note behaviour. Based on movement of individuals between fields, multiple registrations were noted as either the same individual in a new location or a new individual. Surveys recorded lapwing, curlew, snipe, and golden plover; the latter was only observed in two fields and is thus excluded from further analyses.

We calculated the number of breeding pairs using standard species-specific criteria, excluding non-breeding flocks (i.e. more than four individuals; Sim *et al*., 2005; Douglas *et al*., in press). For lapwing, the number of individuals observed during only the early breeding season visit (due to the earlier breeding cycle of this species) was divided by two (O'Brien and Smith, 1992; Bolton *et al*., 2011; O'Brien and Wilson, 2011; Smart *et al*., 2014). For snipe and curlew, both early and late breeding season visits were used with two conspecific birds together or a single individual regarded as a breeding pair (O'Brien and Smith, 1992; Henderson *et al*., 2002; Hoodless *et al*., 2006; Pearce-Higgins and Grant, 2006). The number of snipe and curlew pairs were estimated as the maximum per-visit number across the early and late breeding season stages (Green, 1985; Smart *et al*., 2008; O'Brien and Wilson, 2011; Buchanan *et al*., 2017; Douglas *et al*., 2017).

The above approach can yield inaccurate estimates of snipe populations, especially in locations with high snipe densities (Hoodless *et al*., 2006). We thus conducted additional dedicated evening snipe surveys using the methodology recommended by Hoodless *et al*. (2006) in 28 of our 56 fields selected at random. These recorded all snipe seen or heard (drumming and chipping) whilst walking to within 100 m of each part of the field between one hour pre- and post- sunset in dry conditions with wind speeds less than Beaufort force 4. Flight paths of displaying birds were followed to prevent double counting. The number of pairs is estimated as half the number of detected individuals. There were no significant differences in the number of snipe pairs estimated by the two methods (Wilcoxon signed-rank test with continuity correction; V = 17, P = 0.198) and we thus use estimates from the general breeding wader surveys in all analyses.

2.3.3 Invertebrate surveys

During the early breeding season, invertebrate surveys were conducted in all 56 fields. Of these fields, 30 (7 IP fields; 16 SUP fields; 7 UM fields; 2018 survey habitat type classifications) were haphazardly selected for additional invertebrate surveys during the late breeding season to assess potential seasonal variation in invertebrate abundance. Our focal fields varied in size from 1.2 ha to 19.9 ha (median = 4 ha). As UM sites are unenclosed, they were assigned a 200 m x 200 m (4 ha, which matches the typical size of other field types) block of land. Invertebrate sampling was conducted at 4 points in fields <2 ha, 5 points in fields 2-6 ha, and 6 points in fields > 6 ha. Sampling points were at least 25 m apart and 20 m away from the field edge.

The general location of the sampling points was determined prior to visiting each field depending on the number of within-field sampling points (see Fig. 2.1). When visiting each field, precise sampling point locations were selected by walking an estimated number of paces required to stop at the pre-selected locations. A total of 436 sampling points were sampled, with 286 in the early breeding season and 150 in the late breeding season.

At each sampling point, we obtained indices of invertebrate abundance in the vegetation (many wader species, especially their chicks, glean invertebrates from foliage; Devereux *et al*., 2004; Fisher and Walker, 2015) and soil. Invertebrates in the vegetation were sampled from three 15 cm x 15 cm squares – one centred on the sampling point and the others immediately adjacent to the central square. Sampling was conducted by placing a bag comprised of sweep net material over the sampling site, and cutting the vegetation as close to the ground as possible (approximately 5 cm). Vegetation samples were then searched by hand up to three minutes per sample - at sites with dense vegetation multiple cuts were needed, in which case each cut was searched for a maximum of three minutes. Searching stopped after one minute if no invertebrates were found. All invertebrates were removed using a pooter and then stored in 100% Industrial Methylated Spirit (IMS). This methodology was adopted as alternative approaches based on sweep netting or use of vacuum sampling are biased when vegetation is damp (which is frequently the case in upland habitats), and in the case of vacuum sampling changes in vegetation density alter sampling efficiency (Brook *et al*., 2008; Sanders and Entling, 2011).

The remaining turf and soil, to a depth of 10 cm, was then removed from the central 15 cm x 15 cm square. A sub-sample to a depth of 3 cm was separated, placed in a white sampling tray, and then broken up and hand searched for invertebrates for up to six minutes. Searching stopped after three minutes if no invertebrates were found. All invertebrates were stored in 100% IMS. This sub-sample represents soil invertebrates that are available to waders with short bills, such as lapwing (Ausden *et al*., 2003). Invertebrates were extracted from the remaining soil using the same methodology and these, when combined with invertebrates from the shallow soil layer, provide an index of invertebrate abundance for longer billed waders. A total depth of 10 cm was used as it is the midpoint between the average snipe bill length (approximately 7 cm) and average curlew bill length (approximately 13 cm; Cramp and Simmons, 1982).

Invertebrates from each sample were subsequently identified to order and life cycle stage (egg, larva, nymph, pupa, and adult), using a stereo microscope, when necessary, based on the criteria of Chinery (1993) and Tilling (2014). The wet biomass (g) of each group was obtained as the drying process to measure dry biomass could damage the specimens, preventing future research using the specimens.

Invertebrate biomass in the vegetation and shallow soil were combined to provide an index of invertebrate abundance that is theoretically available to all waders including short-billed species, such as lapwing (hereafter referred to as V-S layer). A few deep soil layer samples contained adult Diptera, Hymenoptera (wasps), and Hemiptera, which are all capable of flight and are likely to have landed in sampling trays during processing and extremely unlikely to be present in the deep soil layer – they were thus removed from the dataset for this soil layer. We then combined invertebrate biomass in the vegetation, shallow and deep soil to provide an index of invertebrate abundance that is theoretically available to longer billed waders, such as snipe and curlew (hereafter referred to as V-S-D layer).

Using data from a literature review, we identified which invertebrate groups (taxa and life cycle stages) were included in the diet of upland breeding waders, and which of these invertebrate groups are key dietary components - defined as comprising ≥ 20% of at least one wader species' diet (see Appendix B for full details and results). We calculated the cumulative biomass of invertebrate dietary components (hereafter referred to as total invertebrate dietary biomass), the cumulative biomass of key invertebrate dietary components (hereafter referred to as key invertebrate dietary biomass), and the biomass of each key invertebrate dietary component that comprised at least 4% of the invertebrate dietary biomass in either the early or late breeding season (i.e. Lumbricidae (hereafter earthworm), Diptera larvae, Gastropoda). We used the 4% threshold because groups that are rarer than this are highly unlikely to be important components of wader diet in our study region. Thus, for components whose cumulative biomass comprised less than 4% of the invertebrate dietary biomass (i.e. Diptera adult, Coleoptera adult, Coleoptera larva, Lepidoptera larva, Hymenoptera adult), we combined these into a single group (hereafter other key invertebrate dietary biomass) whose cumulative biomass exceeded the 4% threshold (see Table B.5 for cumulative biomass of total invertebrate dietary components and of key invertebrate dietary components individually and combined). As total invertebrate dietary biomass and key invertebrate dietary biomass were highly correlated ($r_s > 0.95$), key invertebrate dietary biomass was excluded from further analyses.

We thus generated five metrics of invertebrate biomass for the V-S layer and the V-S-D layer: i) total invertebrate dietary biomass, ii) earthworm biomass, iii) Diptera larva biomass, iv) Gastropoda biomass, and v) biomass of other key invertebrate groups.

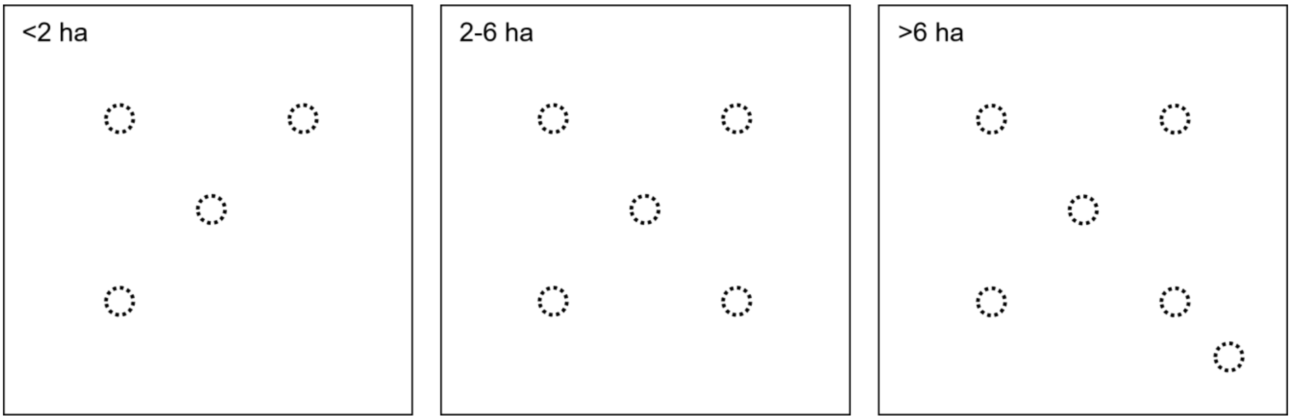

Figure 2.1. Schematic diagrams demonstrating the number and approximate locations of within-field sampling points (dashed circles) in <2 ha, 2-6 ha, and >6 ha survey fields.

2.3.4 Environmental conditions

During the breeding wader surveys, we recorded field level environmental conditions: i) rush *Juncus* percentage cover (to the nearest 5% as estimated during the early breeding season visit because shorter grass swards improve the accuracy and there is negligible spread of rush cover during the time separating early and late visits), ii) livestock dung presence/absence (separate early and late breeding season recordings), and iii) five broad habitat types (Table B.6). These habitat types were unenclosed white moor (characterised by dominance of *Molinia caerulea*), and heather moor (characterised by dominance of *Calluna vulgaris*), and enclosed improved pasture (species-poor grassland dominated by grasses and lacking forb diversity; intensive agricultural improvement), semi-improved pasture (relatively species-rich grassland; moderate agricultural improvement), and unimproved pasture (species-rich grassland; no or negligible agricultural improvement; Joint Nature Conservation Committee, 2010). Additional environmental variables were measured at each invertebrate sampling point during the early and late breeding season visits: i) vegetation height was recorded as the tallest piece of vegetation touching a measuring stick (to the nearest 5 cm), ii) vegetation density following the methodology of Sansom *et al*. (2016), i.e. the number of fully concealed white bands (2 cm width) out of five that are placed at 10 cm intervals from 0 to 40 cm on a measuring stick), iii) rush dominance (defined as rush being amongst the two most abundant plant groups at the survey location as judged by percentage cover), iv) mean soil penetration resistance (kgF) from three readings using a soil penetrometer (based on Green (1988); 20 kg Pesola macro-line spring scale and pressure set, NHBS, Devon, England), which involves pushing a 5 mm diameter metal pressure rod into the soil to 10 cm depth, v) mean soil moisture content (%) from three readings using a SM150T soil moisture sensor and HH150 readout meter (Delta-T Devices, Cambridge, England) with soil type (peat mix or mineral) deduced from the soil samples taken during invertebrate surveys, vi) soil pH at 6 cm depth using a HI-12922 HALO wireless soil pH electrode (Hanna Instruments, Woonsocket, Rhode Island, USA).

2.3.5 Statistical analysis

All statistical analyses were performed in R version 3.6.3 (R Core Team, 2020). Following Whittingham *et al*. (2006), we used a full model approach.

2.3.5.1 Seasonal variation in invertebrate biomass

To assess seasonal patterns in invertebrate biomass, each of the five metrics of invertebrate biomass from i) V-S layer, and ii) V-S-D layer, were modelled as a function of breeding season stage (early or late) and sampling point nested within field identity as a random effect. Each response variable was natural logarithm transformed following the addition of 0.001 to account for zeros in the data. We constructed log-normal generalised linear mixed effect models (GLMMs) with gaussian error structure (identity link) using the lme4 package (Bates *et al*., 2015).

2.3.5.2 Influence of environmental conditions on invertebrate biomass

We modelled each of our five invertebrate biomass responses variables (natural logarithm transformed; x + 0.001) as a function of soil penetration resistance (kgF), soil moisture (%), soil pH, vegetation height (cm; square root transformed), vegetation density, rush dominance, presence/absence of livestock dung, and habitat type (white moor, heather moor, improved pasture, semi-improved pasture, unimproved pasture), with field identity as a random effect. These models were constructed as log-normal GLMMs (gaussian error structure with identity link) using the lme4 package. Separate models were constructed using data from each of our two spatial layers (i.e. V-S layer, and V-S-D layer) and for the early and late sampling periods (Table B.7). Prior to fitting full models, we checked for simple non-linear effects of each continuous predictor variable by modelling the response as a function of the linear term of the focal predictor (linear models) and linear and quadratic terms (quadratic models). If these quadratic models had an Akaike information criterion value corrected for small sample sizes (AICc) that was at least two points lower than that of a linear model and a P value for the quadratic term < 0.1, we used linear and quadratic terms for that predictor variable in the full model; we adopt the $P < 0.1$ rather than $P < 0.05$ threshold to ensure a more cautious approach in rejecting the potential for non-linear relationships to be detected in the full models. We calculated generalised variance inflation factors (GVIFs) using the ‘vif’ function from the car package (Fox and Weisberg, 2019) for all our full models, with GVIFs greater than five advocating the removal of habitat type from some models. As it is easier to compare across response variables when they are modelled with the same suite of predictor variables, habitat type was dropped from all full models; the subsequent GVIFs of all predictor variables were less than five, providing strong evidence that inference from our models is not adversely impacted by collinearity (Harrison *et al*., 2018).

There is value in understanding how food abundance varies across habitat types. We thus constructed a simpler set of models to describe variation in food abundance across habitat types by modelling each response variable as a function of habitat type and field identity as a random effect (see Table B.8 for results).

2.3.5.3 Influence of invertebrate biomass on wader pair densities

We constructed generalised linear models (GLMs; poisson error structure with log link) of the number of pairs per field for each wader species (curlew, lapwing, and snipe) as a function of invertebrate biomass, soil penetration resistance (kgF), vegetation height (cm), and rush cover (%; natural logarithm (x + 2.5) transformed). We constructed five such models, with each using one of our five invertebrate biomass metrics (total invertebrate dietary biomass; earthworm biomass; Diptera larva biomass; Gastropoda biomass; other key invertebrate biomass). Due to variation in wader bill lengths the lapwing models used invertebrate biomass data from the V-S layer, whereas curlew and snipe models used invertebrate data from the V-S-D layer. Each model included field size (ha; natural logarithm transformed) as an offset to convert wader pairs to wader densities, ensuring that field size was accounted for within the models (Table B.9). Habitat type was not included in these models as it inflated the GVIFs of most of our models above the threshold of five, over which model inference would be hindered by collinearity (Harrison *et al.*, 2018). Rush cover rather than rush dominance was included in these models because rush cover was measured at the field level which is more appropriate for predicting wader densities within a field. Other environmental variables were not included in these models due to strong correlation with other more informative variables (i.e. vegetation height and vegetation density: $r_s > 0.7$, density was excluded; Dormann *et al.*, 2013) and because they will only influence wader densities through their impact on invertebrate biomass (i.e. soil pH, soil moisture, and livestock dung) which is included in the models.

Environmental data collected as the sampling point level (i.e. invertebrate biomass metrics, soil penetration resistance, and vegetation height) were averaged for each field. The lapwing model only used environmental data, including invertebrate biomass metrics, from the early breeding season to reflect the survey phase used to estimate lapwing pairs. Snipe and curlew models used environmental data from both the early and late breeding seasons, averaging data over the entire breeding season, as the number of pairs for these species were estimated from both survey phases.

Using the same approach as the invertebrate biomass models (see above), we checked for simple non-linear effects of each continuous predictor variable for each response variable. For all our full models, we checked for overdispersion and zero inflation using the DHARMa package (Hartig, 2020). Using AICc and McFadden's pseudo-R^2 values, we compared between the models for each of our focal wader species in cases where more than one invertebrate metric was statistically significant. Wader densities may be influenced by the actual availability of food items rather than their abundance, with harder soils and taller vegetation potentially restricting access to food (Green, 1988; Devereux *et al*., 2004). We thus constructed two additional sets of full models that included the interaction term between the invertebrate biomass metric and i) soil penetration resistance, or ii) vegetation height when soil penetration resistance or vegetation height was statistically significant ($P < 0.05$) in the main effects only models. We concluded that interaction terms influenced wader densities when $P < 0.05$ (see Tables B.10 and B.11 for model structure and results).

As habitat type was dropped from the wader models, we constructed a simpler set of models to describe variation in wader densities across habitat types by modelling each response variable as a function of habitat type (see Table B.12 for results).

2.4 Results

2.4.1 Seasonal variation in invertebrate biomass

The biomass of total invertebrate dietary components and earthworms were significantly reduced in the late breeding season in both spatial layers (V-S and V-S-D). Inspection of parameter estimates and their standard errors indicate that these declines are greater when the deep soil layer is excluded (Table 2.1). Diptera larvae and Gastropoda biomass did not exhibit significant declines in biomass in either layer (Table 2.1). Conversely, the biomass of other key dietary components was significantly higher in the late than the early breeding season in both spatial layers (Table 2.1).

2.4.2 Influence of environmental conditions on invertebrate biomass

Our models explained substantial amounts of spatial variation in the biomass of total invertebrate dietary components (model McFadden's pseudo-R^2 values range from 0.359 to 0.695; Table 2.2). In all four models, invertebrate biomass was consistently significantly associated with soil pH along a unimodal quadratic curve, with biomass (depending on the season and spatial layer) peaking at pH 5.1-5.8 (Figs. 2.2d, 2.2e, 2.3a, 2.3b). In the early breeding season models, invertebrate biomass was significantly associated with soil penetration resistance along a unimodal quadratic curve, with biomass (depending on the spatial layer) peaking at 7.5-7.9 kgF (Figs. 2.2a, 2.2b). Soil penetration resistance was not significant in either late breeding season model. In the early breeding season, invertebrate biomass (V-S spatial layer only) was significantly associated with soil moisture along a unimodal quadratic curve, with biomass peaking at 68.8% (Fig. 2.3c). In the late breeding season, biomass (V-S-D spatial layer only) was significantly linearly negatively associated with soil moisture (Fig. 2.3c).

Earthworms accounted for most of the total invertebrate dietary biomass, although this proportion was lower in the late breeding season, particularly in the V-S spatial layer (see Table B.5). As a result, the biomass of earthworms exhibited similar associations with soil conditions as total invertebrate dietary biomass (Table

2.2). In the early breeding season, earthworm biomass was consistently significantly associated with soil penetration resistance, moisture, and pH along unimodal quadratic curves, with biomass (depending on spatial layer) peaking at 6.2-6.5 kgF, 61.2-67.0%, and pH 5.8-5.9 (Figs. 2.4a-f). In the late breeding season, earthworm biomass was consistently significantly associated with soil pH along a unimodal quadratic curve, with biomass (depending on the spatial layer) peaking at pH 5.2-5.5 (Figs. 2.5a, 2.5b), and in the V-S spatial layer only, earthworm biomass was significantly higher at sampling points where rush was a dominant vegetation type.

Soil pH was the only edaphic condition to significantly influence biomass of Diptera larvae and Gastropoda. For Diptera larvae (Table 2.2), biomass was consistently significantly associated with soil pH along a unimodal quadratic curve in the early breeding season only, with biomass (depending on the spatial layer) peaking at pH 5.4-5.5 (Figs. 2.6a, 2.6b). In the late breeding season, Diptera larvae biomass within the V-S spatial layer was significantly higher at sampling points with shorter vegetation (linear relationship; Fig. 2.7). No predictor variables significantly influenced Diptera larvae biomass in the V-S-D spatial layer during the late breeding season.

For Gastropoda (Table 2.2), biomass was significantly higher at sampling points with less acidic soil during both breeding seasons and within both spatial layers (linear relationship; Figs. 2.8a, 2.8b, 2.9a, 2.9b). In the late breeding season only, Gastropoda biomass (in the V-S and V-S-D spatial layers) was significantly higher at sampling points with taller vegetation (linear relationship; Figs. 2.9c, 2.9d) and in fields with livestock dung present.

For other components whose cumulative biomass comprised <4% of the invertebrate dietary biomass, i.e. other key invertebrate dietary components (Table 2.2), biomass was significantly higher biomass at sampling points with lower soil moisture (linear relationship; albeit marginally non-significant in the V-S spatial layer model during the early breeding season; Figs. 2.10, 2.11a, 2.11b). Moreover, other key invertebrate dietary biomass was significantly higher at sampling points where rush was a dominant vegetation type, except during the late breeding season where rush dominance was marginally non-significant in the V-S-D spatial layer model and non-significant in the V-S spatial layer model. In the V-S spatial layer model, instead of rush dominance, biomass was significantly higher at sampling points with more acidic soil (linear relationship; Fig. 2.11c).

2.4.3 Influence of invertebrate biomass on wader pair densities

When modelling the density of wader pairs as a function of soil penetration resistance, vegetation height, rush cover and one of five alternative metrics of invertebrate biomass, the lapwing models that included earthworm biomass and Gastropoda biomass had similar AICc values that were >2 AICc points lower than models with alternative invertebrate biomass metrics (Table 2.3). The density of lapwing pairs was marginally non-significantly associated with earthworm biomass (square root transformed) along a unimodal curve, with lapwing density peaking at 0.568 g (Fig. 2.12a); whilst this model had the highest explanatory capacity this was still limited. Lapwing density was marginally non-significantly, and negatively linearly correlated with Gastropoda biomass (natural logarithm (x + 0.001) transformed; Fig. 2.12b). No other predictor variables were significantly associated with lapwing density.

For snipe, the model that included other key invertebrate biomass was >2 AICc points lower than models with alternative invertebrate biomass metrics (Table 2.3). The density of snipe pairs was significantly associated with other key invertebrate biomass (natural logarithm (x + 0.001) transformed) along a unimodal curve, with snipe density peaking at 0.007 g (Fig. 2.12c). Snipe density was also significantly higher in fields with higher soil penetration resistance, and fields with greater rush cover. Rush cover was the only predictor variable that was significant or marginally non-significant in all five snipe models. No other predictor variables were significantly associated with snipe density.

For curlew, the models that included total invertebrate dietary biomass and earthworm biomass had similar AICc values that were >2 AICc points lower than models with alternative invertebrate biomass metrics (Table 2.3). The density of curlew pairs was significantly higher in fields with lower total invertebrate dietary biomass (square root transformed; linear relationship; Fig. 2.12d) and fields with lower earthworm biomass (square root transformed; linear relationship; Fig. 2.12e). In both models, curlew density was consistently significantly associated with vegetation height, with a relatively constant density (peaking at 7.121 cm) until a vegetation height of approximately 14 cm followed by a steady decline. Curlew density was also significantly associated with vegetation height in the other three alternative invertebrate biomass metric models and followed a unimodal quadratic curve with density (depending on invertebrate biomass metric) peaking at 15.667-16.379 cm. No other predictor variables were significantly associated with curlew density.

Table 2.1. Results of log-normal generalised linear mixed effects models investigating whether invertebrate biomass (represented by five different metrics) differs between the early and late breeding season stages at two different spatial layers (all sub-samples combined; vegetation and shallow soil sub-samples combined). Parameter estimates (β) with standard errors (SE), and Type II Wald F tests with Kenward-Roger degrees of freedom are presented, with significant effects highlighted with an asterisk. Early breeding season is the reference level for breeding season stage, so significant negative relationships indicate reduced food abundance in the late breeding season.

Response variable	**Spatial layer**	**Breeding season predictor variable**		
		β ± SE	*F*	*P* value
Total invertebrate dietary biomass	V-S-D	-0.485 ± 0.185	6.876	0.010 *
	V-S	-1.344 ± 0.241	30.993	1.174^{-7} *
Earthworm biomass	V-S-D	-1.221 ± 0.224	29.675	2.064^{-7} *
	V-S	-2.791 ± 0.274	104.130	2.200^{-16} *
Diptera larvae biomass	V-S-D	-0.291 ± 0.237	1.498	0.223
	V-S	-0.367 ± 0.234	2.464	0.119
Gastropoda biomass	V-S-D	-0.258 ± 0.207	1.553	0.215
	V-S	-0.319 ± 0.205	2.407	0.123
Other key invertebrate biomass	V-S-D	0.484 ± 0.190	6.523	0.012 *
	V-S	0.451 ± 0.170	7.031	0.009 *

Table 2.2. Results of log-normal generalised linear mixed effects models (GLMM) investigating the influence of environmental conditions on invertebrate biomass (represented by five different metrics). Parameter estimates with standard errors (L = linear term; Q = quadratic term when both are modelled), and Type II Wald F tests with Kenward-Roger degrees of freedom are presented for each predictor variable, with significant effects highlighted with an asterisk. Rush not being dominant and absence of livestock dung are reference levels for these categorical predictors. Theoretical conditional R^2 (Nakagawa and Schielzeth, 2013; Nakagawa *et al*., 2017) are presented for each GLMM.

Response variable	**Spatial layer**	**Breeding season**	**Model R^2**	**Predictor variables**						
				Soil penetration resistance	*Soil moisture*	*Soil pH*	*Vegetation height (square root transformed)*	*Vegetation density*	*Rush dominance*	*Livestock dung*
Total invertebrate dietary biomass	V-S-D	Early	0.695	L 5.004 ±2.095 Q -6.603 ±1.707 F = 8.655 *P* = 0.0002 *	L -1.609 ±2.269 Q -3.774 ±1.742 F = 2.347 *P* = 0.098	L 30.400 ±2.116 Q -13.664 ±1.084 F = 112.109 *P* = 2.200^{-16} *	0.013 ±0.069 F = 0.035 *P* = 0.852	-0.274 ±0.200 F = 1.865 *P* = 0.173	-0.154 ±0.293 F = 0.272 *P* = 0.603	-0.063 ±0.346 F = 0.033 *P* = 0.857
	V-S	Early	0.635	L 5.526 ±2.115 Q -5.687 ±1.747 F = 7.130 *P* = 0.001 *	L 0.911 ±2.285 Q -5.288 ±1.775 F = 4.846 *P* = 0.009 *	L 27.517 ±2.115 Q -12.197 ±1.833 F = 91.448 *P* = 2.200^{-16} *	-0.031 ±0.070 F = 0.191 *P* = 0.663	-0.231 ±0.205 F = 1.250 *P* = 0.265	0.131 ±0.298 F = 0.190 *P* = 0.663	0.075 ±0.341 F = 0.049 *P* = 0.826
	V-S-D	Late	0.617	-0.041 ±0.059 F = 0.455 *P* = 0.501	-0.023 ±0.001 F = 5.626 *P* = 0.019 *	L 15.483 ±2.650 Q -6.629 ±2.205 F = 18.836 *P* = 1.272^{-7} *	0.053 ±0.099 F = 0.279 *P* = 0.598	0.475 ±0.295 F = 2.523 *P* = 0.114	0.369 ±0.423 F = 0.740 *P* = 0.391	1.332 ±1.088 F = 1.489 *P* = 0.231
	V-S	Late	0.359	-0.053 ±0.079	L -3.974 ±4.199	L 7.581 ±3.536	0.020 ±0.139	0.594 ±0.402	0.749 ±0.574	0.260 ±1.434

				F = 0.426 *P* = 0.515	Q -3.170 ±2.401 F = 1.381 *P* = 0.255	Q -8.230 ±2.721 F = 5.630 *P* = 0.005 *	F = 0.020 *P* = 0.888	F = 2.115 *P* = 0.148	F = 1.652 *P* = 0.201	F = 0.033 *P* = 0.858
Earthworm biomass	V-S-D	Early	0.693	L 0.899 ±2.427 Q -7.466 ±2.008 F = 6.893 *P* = 0.001 *	L -3.385 ±2.622 Q -4.840 ±2.039 F = 3.110 *P* = 0.047 *	L 35.667 ±2.426 Q -15.906 ±2.106 F = 117.040 *P* = 2.200^{-16} *	-0.082 ±0.081 F = 1.020 *P* = 0.313	-0.113 ±0.235 F = 0.228 *P* = 0.633	-0.677 ±0.343 F = 3.833 *P* = 0.051	-0.014 ±0.390 F = 0.001 *P* = 0.972
	V-S	Early	0.642	L 1.926 ±2.378 Q -6.533 ±2.002 F = 5.249 *P* = 0.006 *	L -0.162 ±2.563 Q -6.157 ±2.021 F = 4.695 *P* = 0.010 *	L 32.352 ±2.352 Q -14.043 ±2.081 F = 101.476 *P* = 2.200^{-16} *	-0.152 ±0.080 F = 3.566 *P* = 0.060	-0.042 ±0.235 F = 0.031 *P* = 0.861	-0.326 ±0.339 F = 0.903 *P* = 0.343	0.136 ±0.372 F = 0.133 *P* = 0.717
	V-S-D	Late	0.565	-0.124 ±0.078 F = 2.415 *P* = 0.123	-0.015 ±0.013 F = 1.298 *P* = 0.257	L 20.960 ±3.472 Q -11.532 ±2.666 F = 22.621 *P* = 1.027^{-8} *	L 2.422 ±3.458 Q -0.022 ±2.348 F = 0.242 *P* = 0.785	-0.074 ±0.391 F = 0.034 *P* = 0.853	-0.065 ±0.587 F = 0.012 *P* = 0.913	1.078 ±1.417 F = 0.575 *P* = 0.454
	V-S	Late	0.284	-0.020 ±0.078 F = 0.060 *P* = 0.807	L 5.737 ±4.210 Q -2.944 ±2.393 F = 1.536 *P* = 0.220	L 11.326 ±3.379 Q -9.795 ±2.660 F = 10.007 *P* = 0.0001 *	-0.010 ±0.140 F = 0.004 *P* = 0.947	0.136 ±0.409 F = 0.107 *P* = 0.744	1.288 ±0.587 F = 4.632 *P* = 0.033 *	-1.165 ±1.273 F = 0.829 *P* = 0.369
Diptera larvae biomass	V-S-D	Early	0.336	0.110 ±0.077 F = 1.984 *P* = 0.160	-0.003 ±0.010 F = 0.116 *P* = 0.733	L 10.625 ±3.095 Q -7.189 ±2.550 F = 7.991	-0.101 ±0.096 F = 1.082 *P* = 0.229	-0.098 ±0.274 F = 0.127 *P* = 0.722	-0.519 ±0.413 F = 1.559 *P* = 0.213	-0.101 ±0.547 F = 0.034 *P* = 0.854

						P = 0.0005 *				
	V-S	Early	0.306	0.095 ±0.077 F = 1.492 *P* = 0.223	L -1.795 ±3.200 Q -2.549 ±2.420 F = 0.591 *P* = 0.555	L 9.408 ±3.094 Q -5.516 ±2.567 F = 5.667 *P* = 0.004 *	-0.069 ±0.097 F = 0.497 *P* = 0.481	-0.202 ±0.277 F = 0.528 *P* = 0.468	-0.619 ±0.415 F = 2.192 *P* = 0.140	-0.171 ±0.537 F = 0.102 *P* = 0.751
	V-S-D	Late	0.364	L 1.880 ±3.929 Q -2.517 ±3.929 F = 0.364 *P* = 0.696	-0.024 ±0.015 F = 2.610 *P* = 0.109	0.246 ±0.330 F = 0.534 *P* = 0.467	-0.182 ±0.143 F = 1.572 *P* = 0.212	0.577 ±0.425 F = 1.800 *P* = 0.182	0.636 ±0.612 F = 1.055 *P* = 0.306	0.019 ±1.673 F = 0.0001 *P* = 0.991
	V-S	Late	0.417	L 1.276 ±3.699 Q -2.673 ±2.744 F = 0.463 *P* = 0.630	-0.017 ±0.014 F = 1.383 *P* = 0.242	L 0.890 ±3.792 Q -3.885 ±2.802 F = 0.927 *P* = 0.399	-0.312 ±0.134 F = 5.266 *P* = 0.023 *	0.488 ±0.393 F = 1.505 *P* = 0.222	0.823 ±0.567 F = 2.078 *P* = 0.152	-0.176 ± 1.699 F = 0.011 *P* = 0.918
Gastropoda biomass	V-S-D	Early	0.090	-0.020 ±0.060 F = 0.103 *P* = 0.749	0.006 ±0.007 F = 0.734 *P* = 0.393	0.498 ±0.126 F = 15.283 *P* = 0.0002 *	0.112 ±0.080 F = 1.909 *P* = 0.168	-0.132 ±0.241 F = 0.298 *P* = 0.586	0.389 ±0.336 F = 1.308 *P* = 0.254	0.149 ±0.352 F = 0.178 *P* = 0.675
	V-S	Early	0.090	-0.017 ±0.061 F = 0.077 *P* = 0.782	0.006 ±0.007 F = 0.745 *P* = 0.390	0.489 ±0.127 F = 14.604 *P* = 0.0002 *	0.101 ±0.080 F = 1.559 *P* = 0.213	-0.122 ±0.240 F = 0.256 *P* = 0.614	0.380 ±0.336 F = 1.244 *P* = 0.266	0.151 ±0.355 F = 0.181 *P* = 0.672
	V-S-D	Late	0.203	-0.051 ±0.064 F = 0.596 *P* = 0.442	-0.008 ±0.010 F = 0.568 *P* = 0.452	0.556 ±0.233 F = 5.475 *P* = 0.022 *	0.242 ±0.110 F = 4.633 *P* = 0.033 *	-0.130 ±0.333 F = 0.148 *P* = 0.701	0.205 ±0.478 F = 0.179 *P* = 0.673	2.192 ±1.047 F = 4.352 *P* = 0.045 *
	V-S	Late	0.178	-0.067 ±0.063 F = 1.104 *P* = 0.296	-0.006 ±0.010 F = 0.318 *P* = 0.574	0.557 ±0.226 F = 6.281 *P* = 0.015 *	0.256 ±0.108 F = 5.402 *P* = 0.022 *	-0.238 ±0.326 F = 0.514 *P* = 0.475	-0.174 ±0.468 F = 0.134 *P* = 0.715	2.160 ±1.006 F = 4.572 *P* = 0.041 *

Other key invertebrate biomass	V-S-D	Early	0.163	0.093 ±0.054	-0.015 ±0.007	-0.108 ±0.116	0.056 ±0.068	-0.308 ±0.200	0.593 ±0.290	-0.086 ±0.342
				F = 2.923	F = 5.368	F = 0.860	F = 0.664	F = 2.344	F = 4.100	F = 0.063
				P = 0.089	*P* = 0.022 *	*P* = 0.356	*P* = 0.416	*P* = 0.127	*P* = 0.044 *	*P* = 0.803
	V-S	Early	0.170	0.070 ±0.049	-0.011 ±0.006	-0.148 ±0.107	0.028 ±0.062	-0.201 ±0.182	0.764 ±0.265	-0.114 ±0.318
				F = 1.976	F = 3.237	F = 1.882	F = 0.199	F = 1.200	F = 8.168	F = 0.128
				P = 0.161	*P* = 0.074	*P* = 0.172	*P* = 0.656	*P* = 0.274	*P* = 0.005 *	*P* = 0.722
	V-S-D	Late	0.228	-0.023 ±0.064	-0.025 ±0.010	-0.436 ±0.236	-0.015 ±0.107	0.152 ±0.320	0.879 ±0.457	0.546 ±1.119
				F = 0.128	F = 6.099	F = 3.308	F = 0.019	F = 0.219	F = 3.589	F = 0.236
				P = 0.721	*P* = 0.015 *	*P* = 0.073	*P* = 0.891	*P* = 0.640	*P* = 0.060	*P* = 0.630
	V-S	Late	0.150	-0.077 ±0.060	-0.025 ±0.010	-0.508 ±0.220	0.053 ±0.103	0.155 ±0.311	0.255 ±0.446	0.480 ±0.998
				F = 1.559	F = 6.901	F = 5.139	F = 0.253	F = 0.240	F = 0.316	F = 0.230
				P = 0.215	*P* = 0.010 *	*P* = 0.026 *	*P* = 0.616	*P* = 0.625	*P* = 0.575	*P* = 0.635

Table 2.3. Results of the main effects only generalised linear models (GLMs) investigating the influence of invertebrate biomass on wader density, as well as environmental variables that could influence waders through invertebrate accessibility and breeding habitat selection. Parameter estimates with standard errors (L = linear term; Q = quadratic term when both are modelled), and χ^2 tests are presented for each predictor variable, with significant effects highlighted with an asterisk. Akaike information criterion value corrected for small sample sizes (AICc) and McFadden's pseudo-R^2 are presented for each GLM. The lapwing pairs models use environmental data from the early breeding season only and invertebrate biomass data from the vegetation and shallow soil spatial layer only. The snipe and curlew pairs models use environmental data from the overall breeding season and invertebrate biomass data from all sub-samples combined.

Response variable	**Invertebrate biomass metric**	**Model R^2**	**AICc**	**Predictor variables**			
				Soil penetration resistance	Vegetation height	Rush cover (natural logarithm (x + 2.5 transformed	Invertebrate biomass
Lapwing pairs	Total invertebrate dietary biomass	0.085	75.314	0.190 ±0.232 $\chi^2 = 0.664$ *P* = 0.415	0.006 ±0.054 $\chi^2 = 0.014$ *P* = 0.906	0.031 ±0.386 $\chi^2 = 0.006$ *P* = 0.936	L -1.890 ±4.878 Q -8.451 ±5.623 $\chi^2 = 3.401$ *P* = 0.183
	Earthworm biomass (square root transformed)	0.118	73.084	0.052 ±0.260 $\chi^2 = 0.040$ *P* = 0.841	0.019 ±0.054 $\chi^2 = 0.123$ *P* = 0.726	-0.189 ±0.409 $\chi^2 = 0.211$ *P* = 0.646	L 3.913 ±5.077 Q -9.515 ±5.134 $\chi^2 = 5.631$ *P* = 0.060
	Diptera larvae biomass (natural logarithm (x + 0.001 transformed)	0.035	76.169	0.268 ±0.212 $\chi^2 = 1.527$ *P* = 0.217	-0.017 ±0.048 $\chi^2 = 0.133$ *P* = 0.716	0.095 ±0.418 $\chi^2 = 0.052$ *P* = 0.820	0.038 ±0.215 $\chi^2 = 0.032$ *P* = 0.859
	Gastropoda biomass (natural logarithm (x + 0.001 transformed)	0.081	73.077	0.283 ±0.203 $\chi^2 = 1.843$ *P* = 0.175	-0.037 ±0.039 $\chi^2 = 1.033$ *P* = 0.309	0.425 ±0.439 $\chi^2 = 0.971$ *P* = 0.325	-0.351 ±0.211 $\chi^2 = 3.123$ *P* = 0.077

	Other key invertebrate biomass (natural logarithm (x + 0.001 transformed)	0.044	75.591	0.250 ±0.202 $\chi^2 = 1.472$ *P* = 0.225	-0.029 ±0.040 $\chi^2 = 0.600$ *P* = 0.439	0.086 ±0.394 $\chi^2 = 0.048$ *P* = 0.827	0.176 ±0.223 $\chi^2 = 0.609$ *P* = 0.435
Snipe pairs	Total invertebrate dietary biomass (square root transformed)	0.084	99.602	0.268 ±0.186 $\chi^2 = 2.058$ *P* = 0.151	L 6.179 ±3.230 Q -1.395 ±2.316 $\chi^2 = 4.836$ *P* = 0.124	0.758 ±0.468 $\chi^2 = 3.058$ *P* = 0.080	L 1.867 ±2.820 Q -1.446 ±3.587 $\chi^2 = 1.038$ *P* = 0.595
	Earthworm biomass (square root transformed)	0.091	98.986	0.279 ±0.195 $\chi^2 = 2.006$ *P* = 0.157	L 6.636 ±3.189 Q -1.257 ±2.288 $\chi^2 = 4.175$ *P* = 0.089	0.754 ±0.481 $\chi^2 = 2.721$ *P* = 0.099	L 2.880 ±2.902 Q -1.287 ±3.905 $\chi^2 = 1.654$ *P* = 0.437
	Diptera larvae biomass (natural logarithm (x + 0.001 transformed)	0.086	99.445	0.333 ±0.172 $\chi^2 = 3.823$ *P* = 0.051	L 1.796 ±3.930 Q -0.253 ±2.376 $\chi^2 = 0.255$ *P* = 0.880	1.287 ±0.586 $\chi^2 = 6.722$ *P* = 0.010 *	L -3.448 ±3.512 Q -0.428 ±2.357 $\chi^2 = 1.195$ *P* = 0.550
	Gastropoda biomass (natural logarithm (x + 0.001 transformed)	0.073	97.943	0.290 ±0.164 $\chi^2 = 3.150$ *P* = 0.076	L 4.596 ±2.891 Q -0.840 ±2.281 $\chi^2 = 3.345$ *P* = 0.188	0.914 ±0.449 $\chi^2 = 5.415$ *P* = 0.020 *	0.047 ±0.167 $\chi^2 = 0.078$ *P* = 0.781
	Other key invertebrate biomass (natural logarithm (x + 0.001 transformed)	0.180	90.897	0.464 ±0.185 $\chi^2 = 6.346$ *P* = 0.012 *	L 3.737 ±3.190 Q 0.918 ±2.636 $\chi^2 = 3.973$ *P* = 0.137	1.265 ±0.482 $\chi^2 = 10.026$ *P* = 0.002 *	L -3.905 ±2.934 Q -7.517 ±3.550 $\chi^2 = 9.743$ *P* = 0.008 *

Curlew pairs	Total invertebrate dietary biomass (square root transformed)	0.216	84.060	0.012 ±0.136 χ^2 = 0.007 *P* = 0.933	L -7.101 ±3.727 Q -3.264 ±2.843 χ^2 = 8.708 *P* = 0.013 *	-0.158 ±0.319 χ^2 = 0.240 *P* = 0.624	-1.712 ± 0.806 χ^2 = 6.036 *P* = 0.014 *
	Earthworm biomass (square root transformed)	0.214	84.291	0.009 ±0.137 χ^2 = 0.004 *P* = 0.950	L -7.210 ±3.838 Q -3.288 ±2.903 χ^2 = 8.393 *P* = 0.015 *	-0.172 ±0.323 χ^2 = 0.275 *P* = 0.600	-1.619 ±0.782 χ^2 = 5.805 *P* = 0.016 *
	Diptera larvae biomass (natural logarithm (x + 0.001 transformed)	0.150	90.009	-0.130 ±0.154 χ^2 = 0.740 *P* = 0.390	L -2.436 ±3.543 Q -6.251 ±2.820 χ^2 = 8.377 *P* = 0.015 *	-0.472 ±0.351 χ^2 = 1.795 *P* = 0.180	0.054 ±0.183 χ^2 = 0.087 *P* = 0.768
	Gastropoda biomass (natural logarithm (x + 0.001 transformed)	0.161	89.011	-0.105 ±0.137 χ^2 = 0.613 *P* = 0.434	L -3.037 ±3.213 Q -5.777 ±2.696 χ^2 = 8.070 *P* = 0.018 *	-0.313 ±0.292 χ^2 = 1.118 *P* = 0.290	-0.148 ±0.145 χ^2 = 1.085 *P* = 0.298
	Other key invertebrate biomass (natural logarithm (x + 0.001 transformed)	0.149	90.080	-0.105 ±0.141 χ^2 = 0.574 *P* = 0.449	L -2.747 ±3.338 Q -6.096 ±2.759 χ^2 = 8.268 *P* = 0.016 *	-0.404 ±0.269 χ^2 = 2.210 *P* = 0.137	-0.024 ±0.188 χ^2 = 0.016 *P* = 0.900

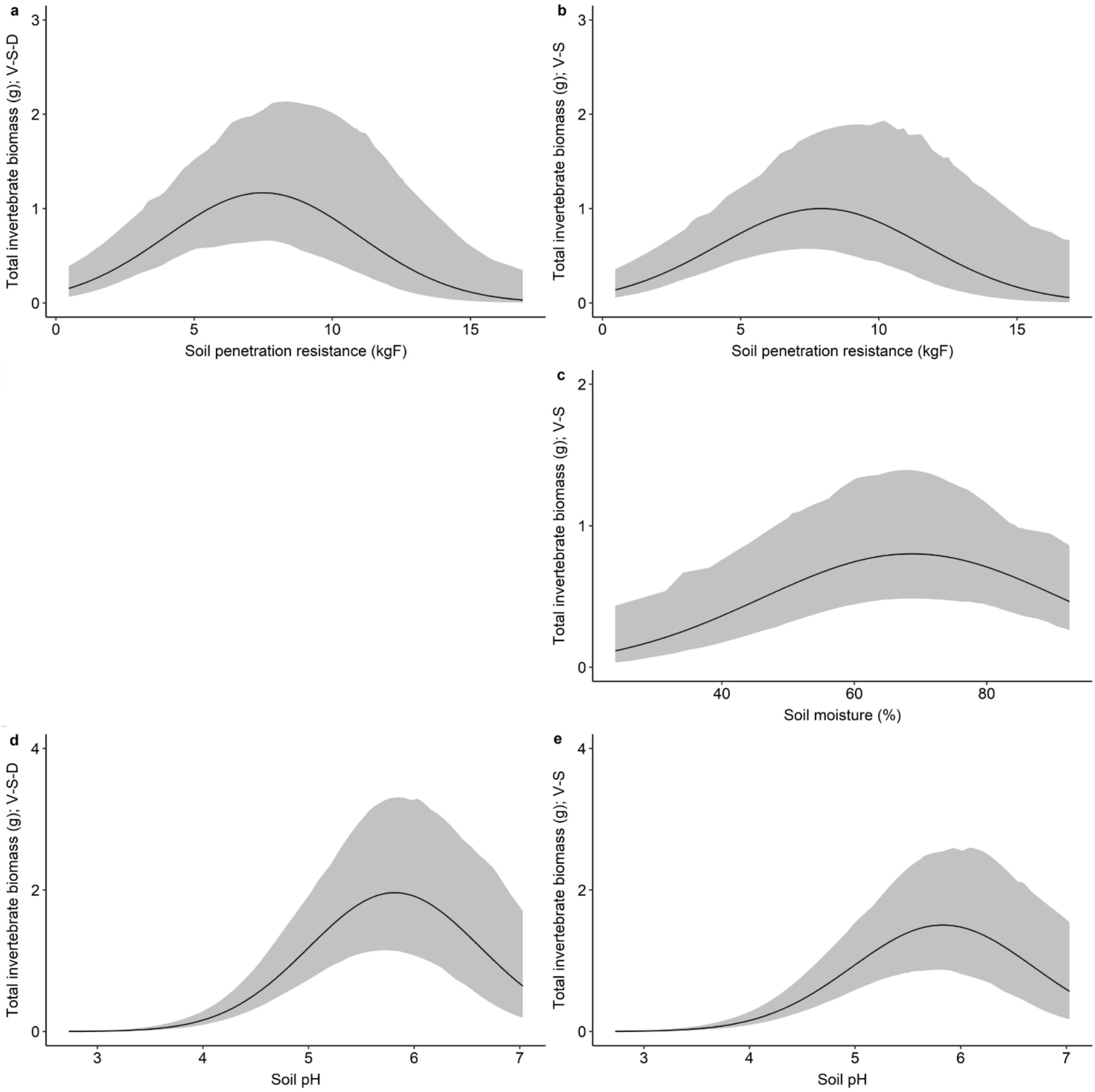

Figure 2.2. The significant relationships between total invertebrate biomass and soil conditions in the early breeding season, showing the predicted total invertebrate dietary biomass (back-transformed) with differing soil penetration resistance (a) V-S-D spatial layer; b) V-S spatial layer), soil moisture (c) V-S spatial layer), and soil pH (d) V-S-D spatial layer; e) V-S spatial layer), generated from the associated generalised linear mixed effects models. Shaded ribbons show the 95% bootstrap confidence intervals.

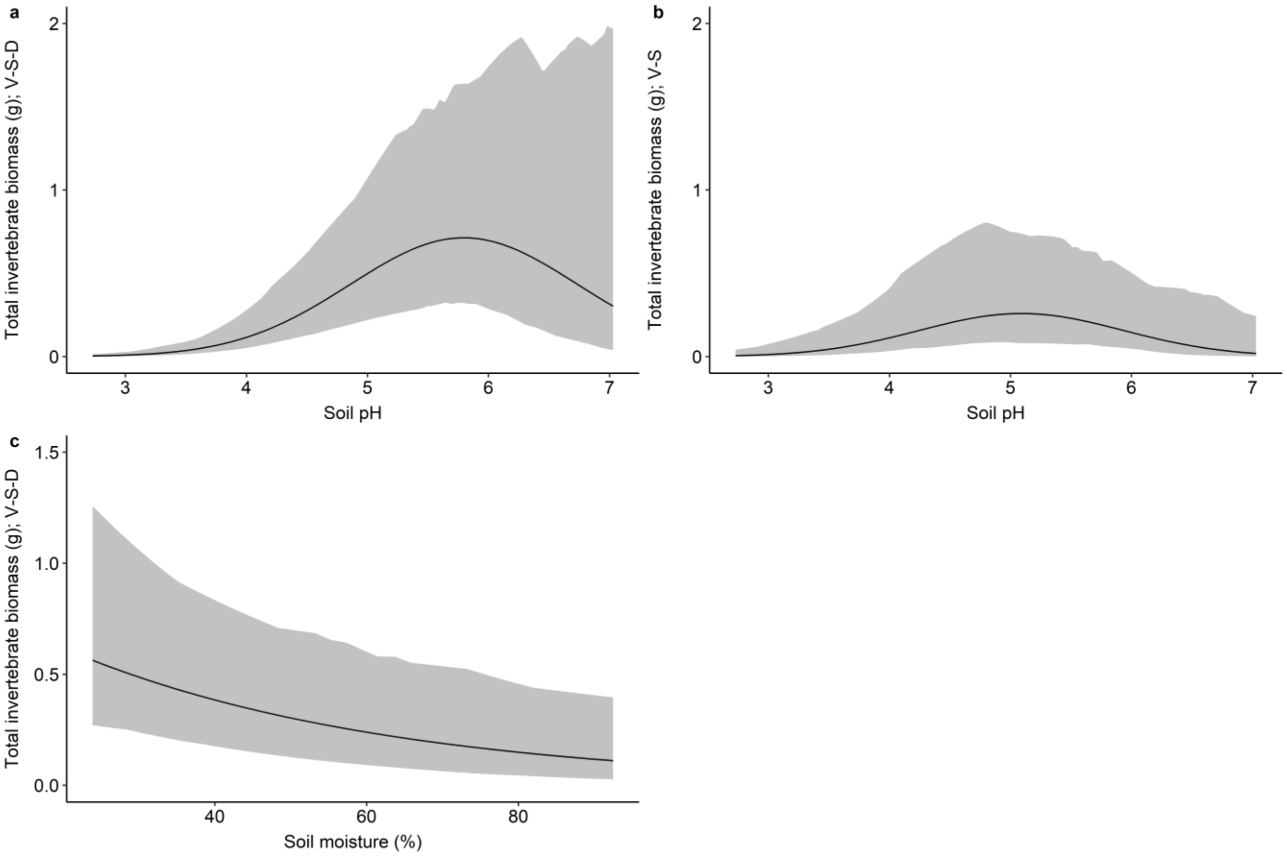

Figure 2.3. The significant relationships between total invertebrate biomass and soil conditions in the late breeding season, showing the predicted total invertebrate dietary biomass (back-transformed) with differing soil pH (a) V-S-D spatial layer; b) V-S spatial layer), and soil moisture (c) V-S-D spatial layer), generated from the associated generalised linear mixed effects models. Shaded ribbons show the 95% bootstrap confidence intervals.

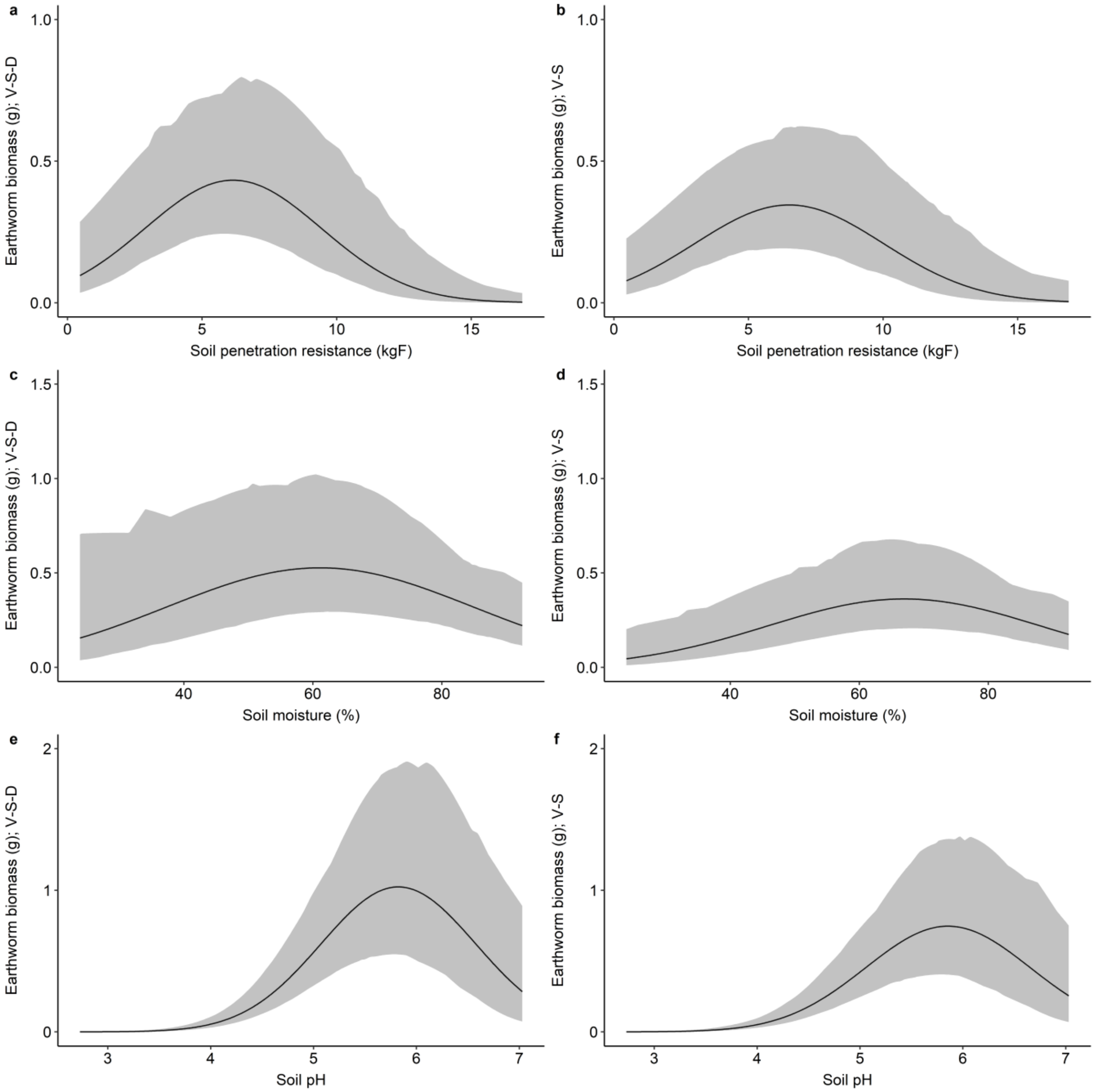

Figure 2.4. The significant relationships between earthworm biomass and soil conditions in the early breeding season, showing the predicted earthworm biomass (back-transformed) with differing soil penetration resistance (a) V-S-D spatial layer; b) V-S spatial layer), soil moisture (c) V-S-D spatial layer; d) V-S spatial layer), and soil pH (e) V-S-D spatial layer; f) V-S spatial layer), generated from the associated generalised linear mixed effects models. Shaded ribbons show the 95% bootstrap confidence intervals.

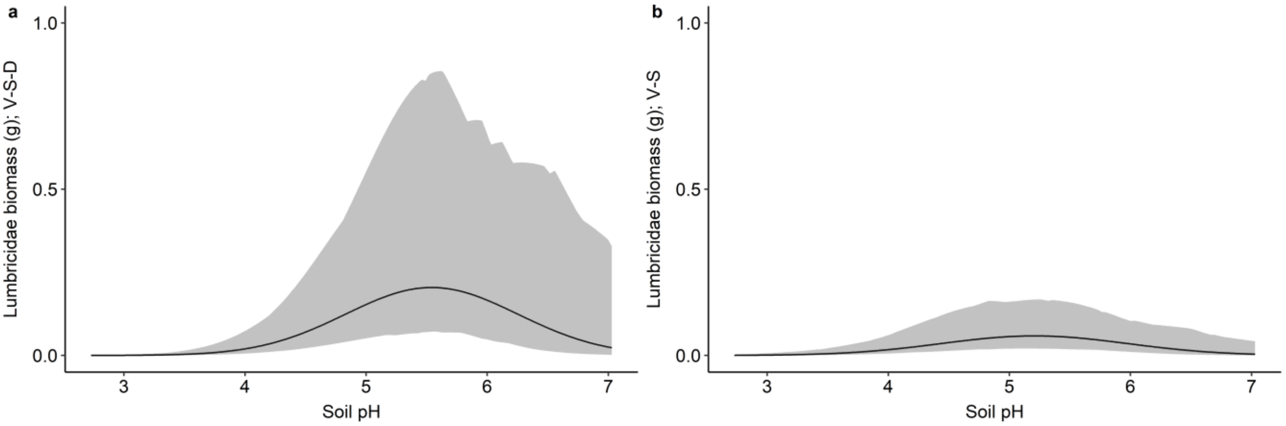

Figure 2.5. The significant relationships between earthworm biomass and soil conditions in the late breeding season, showing the predicted earthworm biomass (back-transformed) with differing soil pH for the (a) V-S-D spatial layer and (b) V-S spatial layer, generated from the associated generalised linear mixed effects models. Shaded ribbons show the 95% bootstrap confidence intervals.

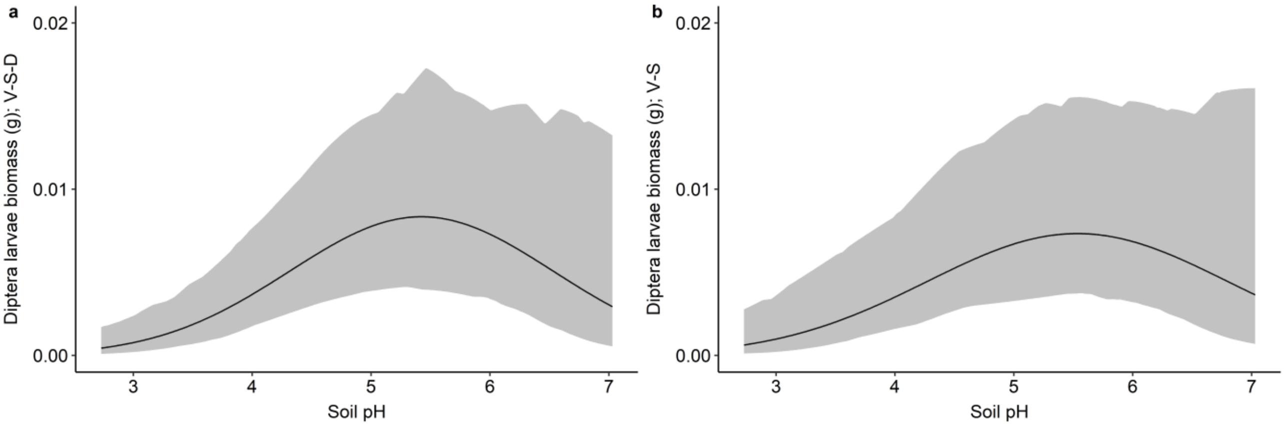

Figure 2.6. The significant relationships between Diptera larvae biomass and soil conditions in the early breeding season, showing the predicted Diptera larvae biomass (back-transformed) with differing soil pH for the (a) V-S-D spatial layer and (b) V-S spatial layer, generated from the associated generalised linear mixed effects models. Shaded ribbons show the 95% bootstrap confidence intervals.

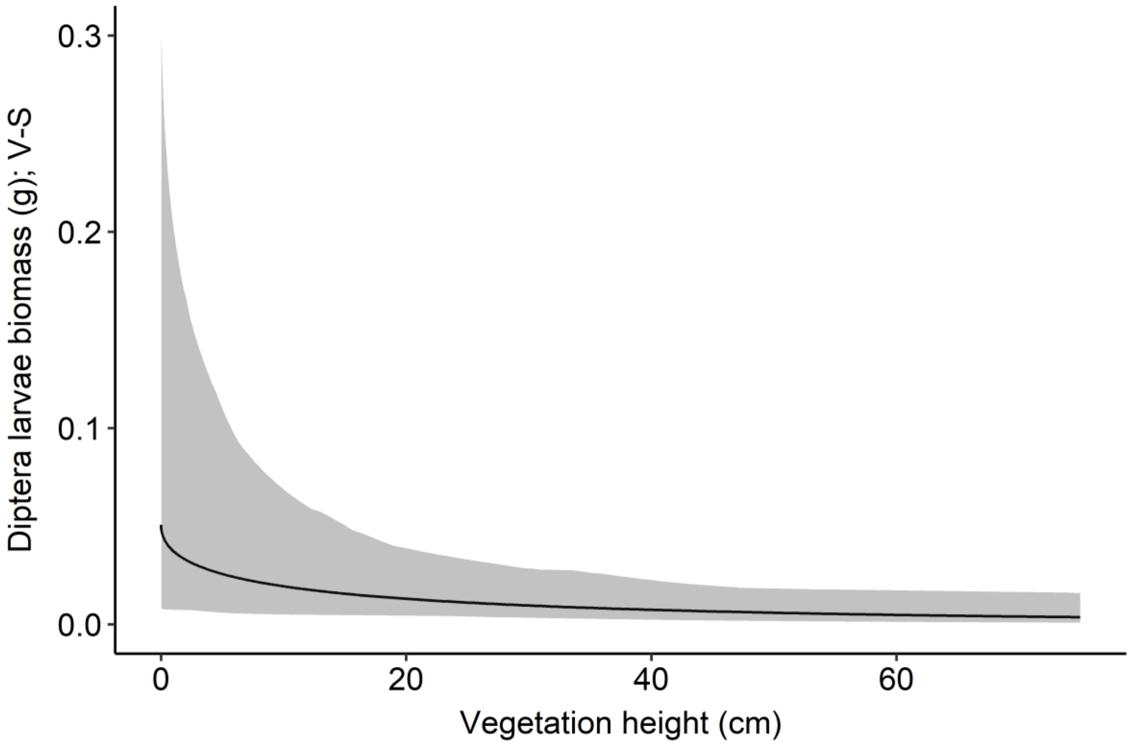

Figure 2.7. The significant relationship between Diptera larvae biomass and vegetation height in the early breeding season, showing the predicted Diptera larvae biomass (back-transformed) with differing vegetation height (back-transformed) for the V-S spatial layer, generated from the associated generalised linear mixed effects model. Shaded ribbon shows the 95% bootstrap confidence intervals.

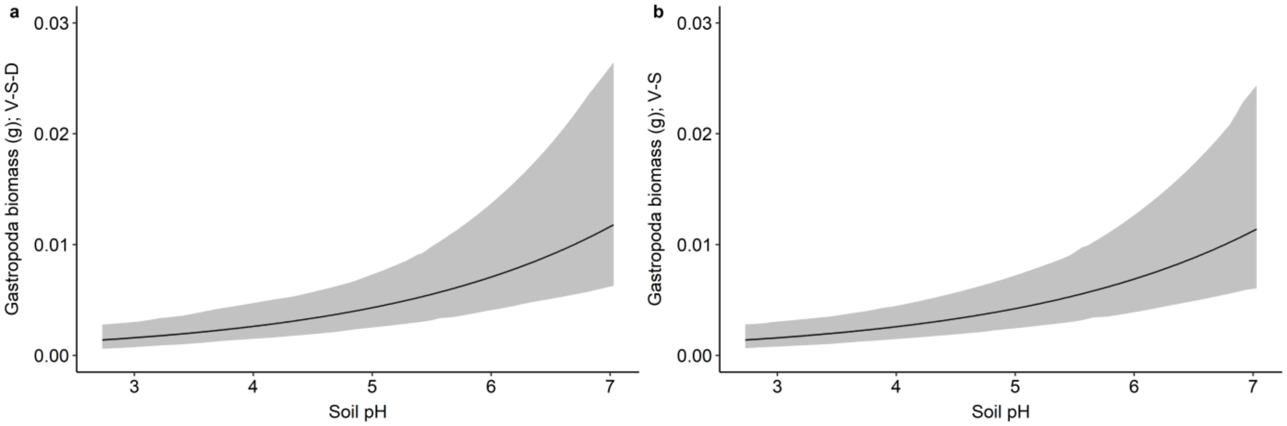

Figure 2.8. The significant relationships between Gastropoda biomass and soil conditions in the early breeding season, showing the predicted Gastropoda biomass (back-transformed) with differing soil pH for the (a) V-S-D spatial layer and (b) V-S spatial layer, generated from the associated generalised linear mixed effects models. Shaded ribbons show the 95% bootstrap confidence intervals.

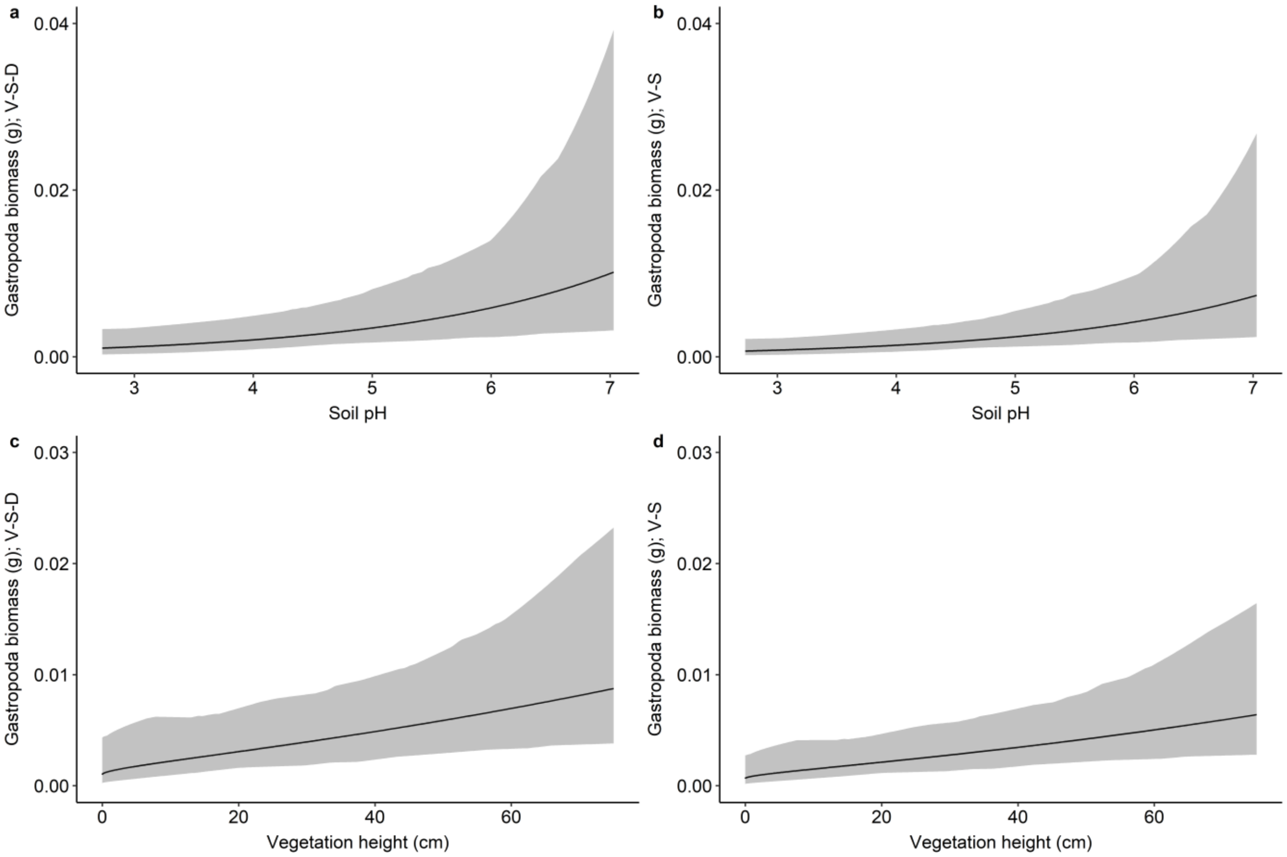

Figure 2.9. The significant relationships between Gastropoda biomass and environmental conditions in the late breeding season, showing the predicted Gastropoda biomass (back-transformed) with differing soil pH (a) V-S-D spatial layer; b) V-S spatial layer), and vegetation height (back-transformed; c) V-S-D spatial layer; d) V-S spatial layer), generated from the associated generalised linear mixed effects models. Shaded ribbons show the 95% bootstrap confidence intervals.

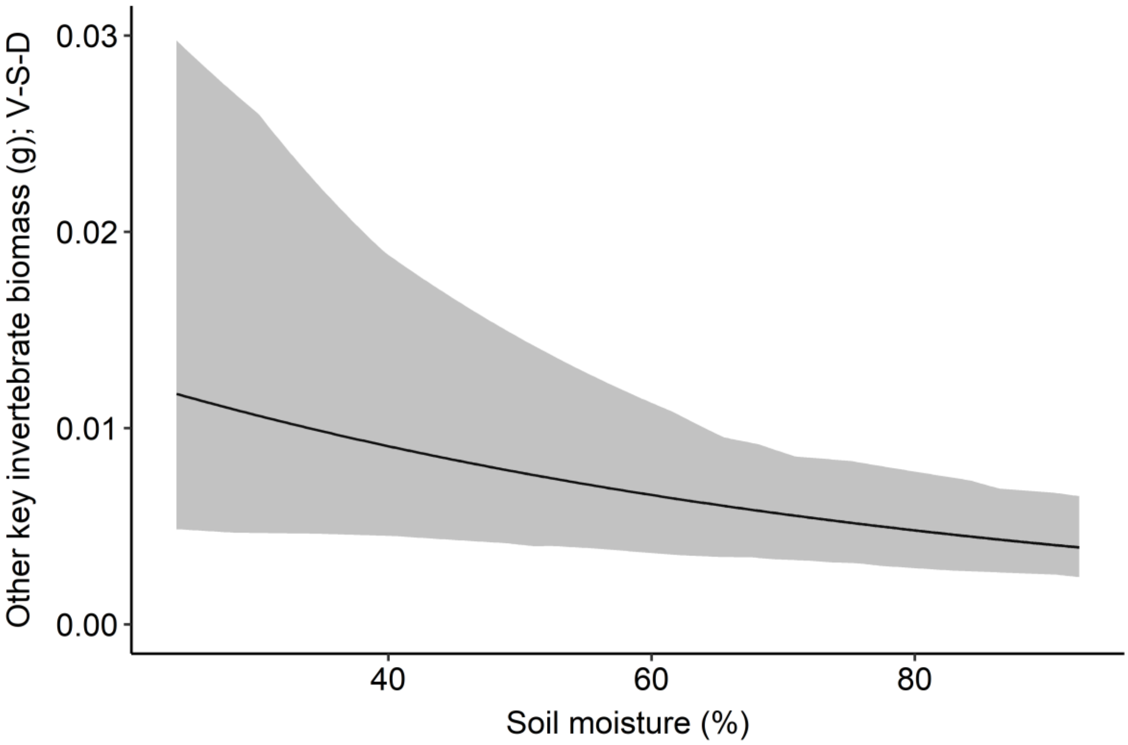

Figure 2.10. The significant relationship between other key invertebrate biomass and soil moisture in the early breeding season, showing the predicted other key invertebrate biomass (back-transformed) with differing soil moisture for the V-S-D spatial layer, generated from the associated generalised linear mixed effects model. Shaded ribbon shows the 95% bootstrap confidence intervals.

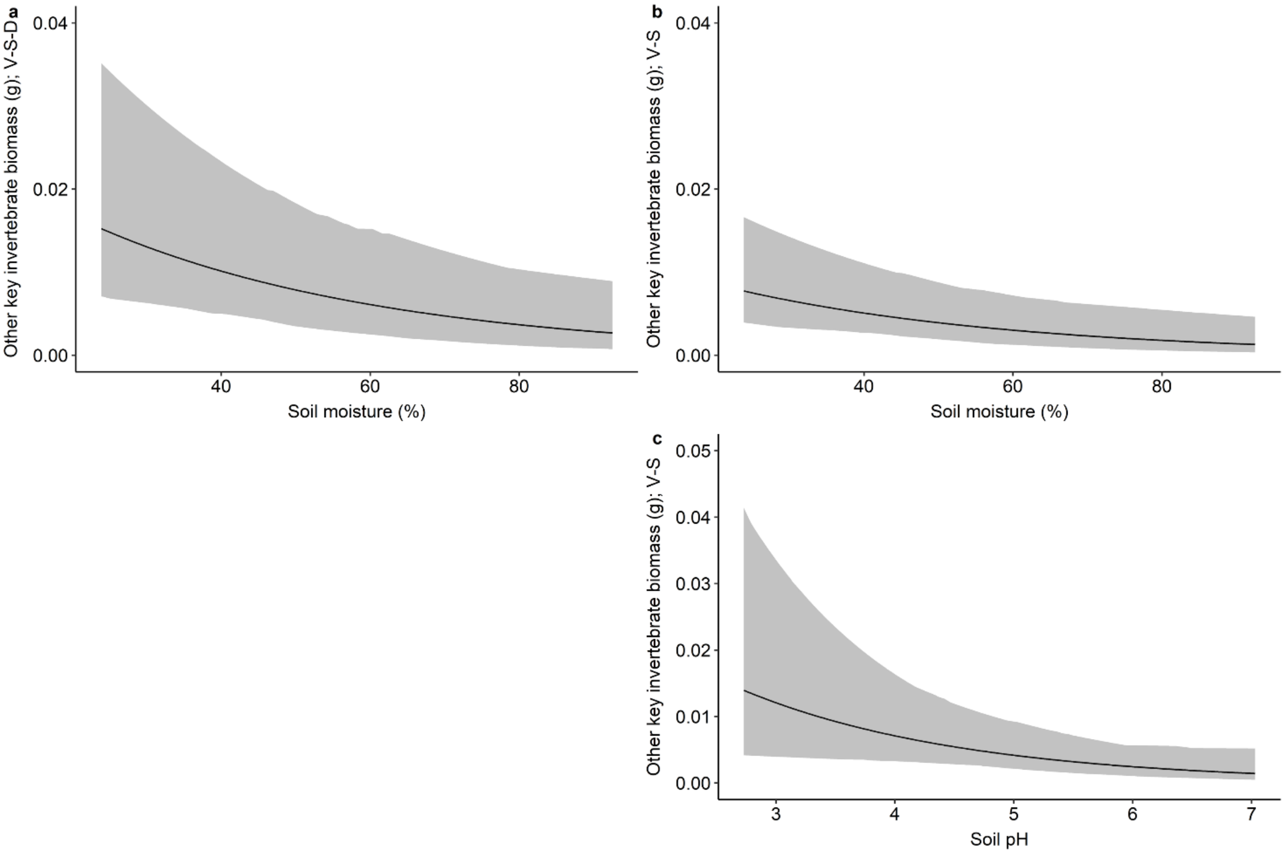

Figure 2.11. The significant relationships between other key invertebrate biomass and soil conditions in the late breeding season, showing the predicted other key invertebrate biomass (back-transformed) with differing soil moisture (a) V-S-D spatial layer; b) V-S spatial layer), and soil pH (c) V-S spatial layer), generated from the associated generalised linear mixed effects models. Shaded ribbons show the 95% bootstrap confidence intervals.

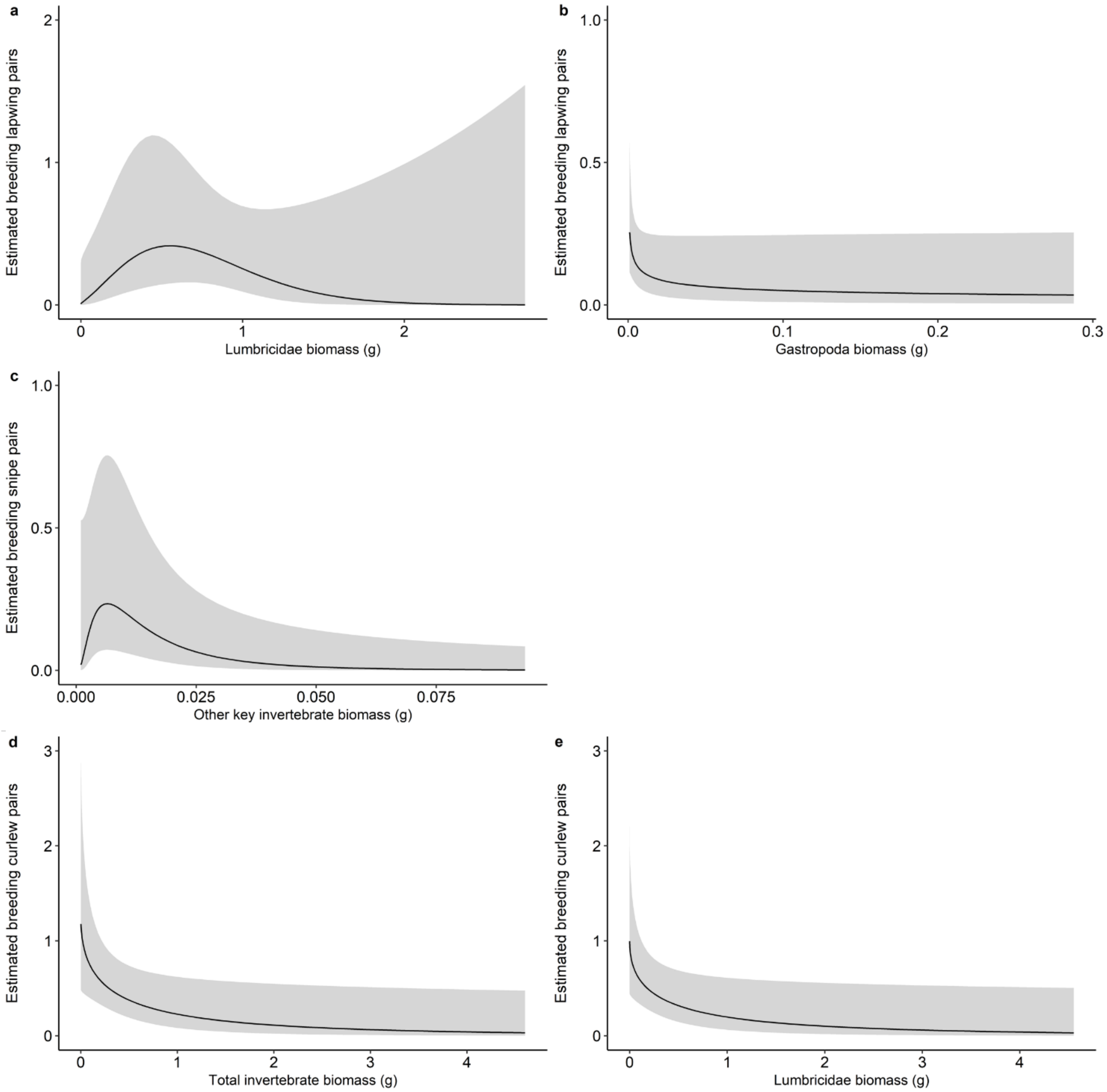

Figure 2.12. The significant (or marginally non-significant; $P < 0.1$) relationships between breeding wader pairs and invertebrate biomass metrics (back-transformed) showing predicted pairs with differing invertebrate biomass, generated from the associated generalised linear models: a) lapwing pairs and earthworm biomass; b) lapwing pairs and Gastropoda biomass; c) snipe pairs and other key invertebrate biomass; d) curlew pairs and total invertebrate dietary biomass; e) curlew pairs and earthworm biomass. Shaded ribbons show the 95% bootstrap confidence intervals.

2.5 Discussion

2.5.1 Influence of environmental conditions on invertebrate biomass over the wader breeding season

As expected for different invertebrate taxa, our study found a diverse range of responses from the five invertebrate biomass metrics to different environmental conditions over the wader breeding season. Four of the five invertebrate biomass metrics (total invertebrate dietary components, earthworms, Diptera larvae, Gastropoda) were lower in the late breeding season compared to the early, and significantly so for earthworm biomass and total invertebrate dietary biomass (of which earthworms comprised most of the biomass; see Table B.5). This decline was likely caused by lower soil moisture and higher soil penetration resistance in the late breeding season (Tables B.13 and B.14). Yet, these soil conditions only significantly influenced biomass in the early breeding season for earthworms and total invertebrate dietary components (except soil moisture in the total invertebrate dietary biomass V-S-D spatial layer model) with a higher biomass at sampling points with intermediate soil wetness and compactness. The lack of impact in the late breeding season could be affected by the biomass earlier in the breeding season, other factors that affect invertebrate distribution, and the mobility of the invertebrate groups. For example, earthworms are restricted by soil pH to habitats with less acidic soils, but habitats such as improved and semi-improved pasture (which contained the highest biomass; see Table B.15) experienced a greater loss in soil moisture over the breeding season; earthworms would have been unable to migrate to acidic habitat types which experienced a less considerable loss in soil moisture and thus biomass would have been influenced more by soil pH than soil moisture. Nevertheless, important implications for future climate change projections are highlighted by these results; less precipitation in summer months (Met Office, 2019) could affect the abundance or distribution of invertebrates, for example by forcing earthworms deeper into the soil out of reach of foraging waders.

The non-significant decline exhibited by Diptera larvae and Gastropoda over the wader breeding season could signify that these groups cannot adjust their distribution in response to drier conditions, or possibly that they are less affected by warmer, drier weather. Diptera larvae biomass and Gastropoda biomass were not significantly influenced by soil moisture or soil penetration resistance in either the early or late breeding season models. The lower Diptera larvae biomass in the late breeding season could, however, be due to the emergence of Tipulidae larvae as adults. Most of the Diptera larvae biomass was comprised of the family Tipulidae (early breeding season = 89.5%; late breeding season = 95.0%) and Tipulidae typically emerge as adults from late May (Pearce-Higgins and Yalden, 2004) which coincides with our late breeding season. No Tipulidae adults were however sampled during the late breeding season, but the lifespan of adults is only a few days (Coulson, 1962), so it is possible that our sampling dates missed the peak emergence. Moreover, our vegetation sub-sample sampling method was not tailored to capture quick flying insects.

In contrast to these four invertebrate biomass metrics, other key invertebrate biomass was significantly higher in the late breeding season compared to the early, substituting a proportion of the earthworm biomass from earlier in the breeding season. This group includes Diptera, Coleoptera and Hymenoptera adults which are more likely to be active in warmer weather (Williams, 1961; Høye and Forchhammer, 2008). Thus, the negative relationship with soil moisture could be a product of the invertebrate groups' behaviour in the warmer but also drier late breeding season. Interestingly, the biomass of these other key invertebrate groups was higher where rush, a plant species typically associated with wet conditions (Preston *et al*., 2002), was a dominant vegetation type. Of sampling points within white moor, the habitat that contained the highest average other key invertebrate biomass in the late breeding season (Table B.15), 50.8% had rush as a dominant vegetation type.

For all invertebrate biomass metrics, soil pH is the only environmental condition that consistently predicted invertebrate biomass in at least one of each metrics' models. This constant impact of soil pH likely relates to habitat type; unimproved pasture and moorland habitats were more acidic than improved and semi-improved pasture (Table B.14). The relationship between soil pH and earthworms (and total invertebrate dietary components) was most significant, with lower biomass at points with more acidic soil (McCallum *et al*., 2016) and hence at points within white moor and heather moor habitats (Table B.15). A similar, albeit linear, pattern emerged for Gastropoda with a significantly higher biomass in less acidic soils i.e. improved, semi-improved and unimproved pastures. In contrast, Diptera larvae biomass was only significantly influenced by soil pH in the early breeding season, with biomass peaking at pH 5.5-5.6. For these four metrics, of which two are sub-surface taxa, there is a general trend towards greater biomass in less acidic soils. For other key invertebrate groups, soil pH only significantly influenced biomass in the V-S spatial layer model during the late breeding season. Yet, biomass was significantly higher at sampling points with more acidic soil and is likely attributable to the white moor habitat type which was the second most acidic habitat type whilst also containing other important environmental conditions for this invertebrate biomass metric (see rush dominance above). The lack of relationship in the V-S-D spatial layer model may be because these invertebrate groups (Diptera adults, Coleoptera larvae and adults, Lepidoptera larvae, and Hymenoptera adults) are unlikely to inhabit the deep soil.

Other non-edaphic conditions also influenced invertebrate biomass. Similar to other key invertebrate groups, earthworms were associated with rush dominance but only in the V-S spatial layer during the late breeding season. As mentioned previously, rush is typically associated with wet conditions (Preston *et al*., 2002) and hence earthworms may have migrated to areas with rush as the soil may have retained more water. However, soil moisture did not significantly influence earthworm biomass in the late breeding season when it was drier. Again, this lack of relationship could relate to habitat type. Unimproved pasture was the only habitat type to experience greater earthworm biomass in the late breeding season compared to the early. Moreover, this habitat type experienced a less considerable decrease in soil moisture between the early and late breeding season compared to improved and semi-improved pastures, and 17.5% of sampling points had rush as a dominant vegetation type. Yet, improved and semi-improved pasture still contained higher mean biomass, despite experiencing a decrease over the breeding season.

Vegetation also impacted Diptera larvae and Gastropoda, but through vegetation height rather than type for these invertebrate groups. In the late breeding season, higher Diptera larvae biomass was associated with shorter vegetation (V-S spatial layer model only). Tipulidae larvae, which comprise most of the biomass, usually reside near the surface of the soil and feed on roots. This association with shorter vegetation could thus relate to root damage reducing shoot growth (Dawson *et al*., 2004). The opposite relationship was found for Gastropoda in the late breeding season, with significantly higher biomass at sampling points with taller vegetation. Due to the drier conditions later in the season (see Table B.13; mean soil moisture ± SE: early breeding season = 78.422 ± 1.115; late breeding season = 41.755 ± 2.160), taller vegetation may have provided damp refuges for Gastropoda to prevent desiccation. Also, in the late breeding season only, the finding that Gastropoda biomass was associated with livestock dung presence needs to be cautiously interpreted as livestock dung was only absent in one field.

2.5.2 Influence of invertebrate biomass and environmental conditions on wader pair density

When exploring the next level of the “bottom-up” effect, our study found that wader pair densities responded differently to invertebrate abundance and other environmental conditions but in all cases, densities did not appear to be driven strongly by invertebrate abundance. Higher curlew pair densities were found where earthworm (and total invertebrate dietary component) biomass was lower, with soil pH and habitat type clarifying this association. A higher proportion of white moor ‘fields’ (62.5%) contained curlew compared to the other habitat types (improved pasture = 27%; heather moor = 25%; semi-improved pasture = 19%; unimproved pasture = 12.5%). Within white moor ‘fields’, soil pH was lower (see Table B.14; mean soil pH ± SE: early breeding season = 3.723 ± 0.102; late breeding season = 3.920 ± 0.105) and we found that earthworm biomass was lower in more acidic soils (see above) typically found in moorland habitats. This result suggests that the wader surveys were primarily detecting adult curlew on their nesting or chick rearing sites rather than their foraging sites, as agricultural grasslands provide important foraging areas for off-duty adults (Robson, 1998; Ewing *et al.*, 2017) likely feeding on the higher abundance of earthworms and Diptera larvae (Table B.15). It is therefore possible that, based on standard breeding wader surveys, conservation efforts for curlew in the uplands may overlook these important habitats essential for adult fitness. For example, lime application and less intensive inorganic nitrogenous fertiliser application could be advised to prevent acidification of soils in upland improved grasslands and thus retain high earthworm abundance (McCallum *et al.*, 2016). Indeed, our literature review highlighted that quantitative curlew diet data was only available from one moorland study (Robson, 1998) with no empirical evidence that off-duty adults rely on earthworms in agricultural grasslands, despite anecdotal observations suggesting the contrary.

Earthworm biomass was also associated with lapwing pair density, albeit marginally non-significantly, with lapwing density peaking at 0.568 g. The mean biomass values recorded in semi-improved pasture fields and unimproved pasture fields during the early breeding season straddle this peak suggesting that lapwing may prefer semi-improved or unimproved pasture to improved pasture where the highest mean earthworm biomass was recorded. No environmental conditions significantly influenced lapwing density to indicate which conditions in semi-improved or unimproved pasture were preferable. Surprisingly, higher lapwing pair densities were found where Gastropoda biomass was lower, yet Gastropoda were associated with less acidic soils (see above) such as those in semi-improved pasture, contradicting the earthworm result. According to our literature review, however, earthworms and Gastropoda are both key components of lapwing diet (Table B.4). These conflicting results may stem from the need for a larger sample of fields with lapwing present as lapwing were only detected in 14.3% of survey fields (cf. 23.2% for snipe and 26.8% for curlew).

For snipe, other key invertebrate biomass was the only metric with which density was associated. Snipe pair densities essentially decreased where biomass was higher, with a small peak in density at very low quantities (0.007 g) of these prey items. Based on our literature review, the only invertebrate group within the other key invertebrate metric that comprised at least 20% of snipe diet is Coleoptera adults (Table B.4), which only formed 18.0% and 21.0% of other key invertebrate biomass respectively in the early and late breeding seasons. This relatively small proportion could possibly explain the peak in snipe density at very low biomass. As discussed previously, however, other key invertebrate groups were associated with points where rush was a dominant vegetation type, and our models found that snipe density was higher in fields with higher rush cover (supporting previous research by Baines (1988) and Hoodless *et al*. (2007)). It is possible that other key invertebrate groups' relationships with drier and more acidic soils (see above) negated the link with rush, as the habitat types with the greatest proportion of fields with snipe present were the less acidic unimproved and semi-improved pastures (50.0% and 28.6% respectively). These habitat types, as well as improved pasture and white moor, also had relatively hard soil in the late breeding season which exceeded the soil penetration resistance threshold revealed by Green *et al*. (1990). The significant association between snipe density and harder soil could be a product of the 2018 wader breeding season which experienced unseasonably low precipitation (Fig. B.1).

Rather than rush cover, we found that intermediate vegetation height was key for curlew, supporting previous research (Durant *et al*., 2008). In our study, field level vegetation height was generated by calculating the mean from the vegetation measured at the within-field sampling points. Therefore, fields with intermediate vegetation height could either have homogeneous intermediate vegetation height, or a heterogeneous mixture of shorter and taller vegetation. The latter is more likely as Pearce-Higgins and Grant (2006) found that a heterogeneous sward structure was more beneficial for curlew.

2.5.3 Conclusion

Despite finding that invertebrate abundance does not currently appear to be limiting upland wader populations in our study region, and thus other factors such as high predation pressure are likely limiting population size (Smart *et al*., 2013; Roos *et al*., 2018), implementing upland land management that promotes healthy invertebrate populations will bolster wader numbers in areas where availability does regulate them. Our five invertebrate biomass metrics were all associated with soil pH and thus habitat type, with each group associated with different environmental conditions. This knowledge could be used in combination with upland wader species' selected breeding habitats and key invertebrate dietary components highlighted by our literature review (Table B.4) to enable habitat suitability checks for those vital invertebrate groups. It is however important that all habitat types utilised by waders throughout the breeding season for nesting, chick-rearing, and foraging are incorporated, particularly for wide-ranging species such as curlew (Robson, 1998; Ewing *et al*., 2017).

Different invertebrate taxa may preferably be consumed by wader adults and different aged chicks, with these preferences often depending on the seasonal availability of those taxa (Pearce-Higgins and Yalden, 2004; Buchanan *et al*., 2006). For example, for our three focal wader species, other key invertebrate groups was not positively associated with adult density, but for golden plover, another moorland breeding wader, taxa within this metric such as Coleoptera and Diptera adults are key components of chick diet (Whittingham *et al*., 2001; Pearce-Higgins and Yalden, 2004). Diet of wader chicks should therefore be considered too (our study focused on adult waders) due to the principal threat of low breeding productivity for waders (Roodbergen *et al*., 2012; Cook *et al*., 2021). Higher chick fitness through plentiful foraging conditions could potentially reduce the risk of starvation, and predation by reducing chicks' required foraging time (Pearce-Higgins and Yalden, 2004; Schekkerman *et al*., 2009). Such chick dietary information could supplement our results to advise upland land management for invertebrate taxa and thus breeding waders. Complementing previous advice for waders, provision of a mosaic of habitats has been recommended to maximise the suitable environmental conditions available for invertebrates (Buchanan *et al*., 2006; Buchanan *et al*., 2017; Arnott *et al*., 2021). Creation of heterogenous sward structures would provide the shorter and taller vegetation that is respectively beneficial for Diptera larvae and Gastropoda. In the face of climate change and hence more frequent warmer, drier breeding seasons as experienced during our 2018 study, the implementation of counteracting adaptation management, such as blocking drainage ditches for desiccation-prone taxa (Pearce-Higgins, 2011; Carroll *et al*., 2011; Carroll *et al*., 2015), to maximise the abundance and accessibility of invertebrate prey for waders will become increasingly important. According to our findings, reducing drainage of less acidic agricultural grasslands to promote high soil moisture (without waterlogging) could boost earthworm abundance.

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Chapter 3

Upland rush management advocated by agri-environment schemes increases predation of artificial wader nests

An artificial wader nest in a patch of rush

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3.1 Abstract

Farmland birds, including breeding waders, have declined across Europe. One frequently advocated strategy to facilitate population recovery is using agri-environment schemes (AES) to improve vegetation structure. A key example is cutting dense rush *Juncus* to open the sward which aims to increase the abundance of wading birds, for example by improving foraging conditions. Effects on breeding success are, however, unknown. This is a critical knowledge gap as high nest and chick predation rates are a key driver of wader declines. For wader species that nest across a range of sward structures, e.g. Eurasian curlew *Numenius arquata* and common snipe *Gallinago gallinago*, converting denser swards to more open ones may reduce opportunities for nest concealment and thus increase predation risk. Due to the difficulties of locating large numbers of wader nests, we assess rush management impacts on nest predation risk using artificial wader nests (n = 184) in two upland areas of England, using fields in which rush is managed according to AES prescriptions (treatment; n = 21) or un-managed (control; n = 22) fields. Daily nest predation rates (DPRs) were twice as high in treatment (0.064 day^{-1}) than control fields (0.027 day^{-1}). Within treatment fields, DPRs were twice as high for nests in cut rush patches (0.108 day^{-1}) than in uncut rush (0.055 day^{-1}). Modelling links higher DPRs associated with rush cutting to the resultant shorter and less dense vegetation. Our results highlight the need to assess how AES prescriptions that alter vegetation structure impact all aspects of the target species' fitness and thus determine population recovery. Studies using real wader nests should test whether AES rush management inadvertently creates an ecological trap by altering vegetation structure, and identify the sward structure and configuration that optimises trade-offs between foraging conditions and nest predation risk.

3.2 Introduction

Widespread declines in European farmland birds have arisen from changes in agricultural practices, in particular intensification (Donald, Green, & Heath, 2001; BirdLife International, 2015). Despite three decades of conservation interventions, principally agri-environment schemes (AES), that attempt to reverse these declines many previously common species are still declining (Kleijn & Sutherland, 2003; BirdLife International, 2015). Targeted AES interventions can benefit locally distributed, threatened species (e.g. corn bunting *Emberiza calandra*, Perkins *et al.*, 2011) but non-targeted AES typically primarily benefit common and widespread species of least conservation concern (Kleijn *et al.*, 2006; Batáry *et al.*, 2015). Ongoing declines in farmland bird populations are probably due to insufficient landscape-scale implementation of AES (Franks *et al.*, 2018) combined with limited availability and uptake of detailed prescriptions targeted to the specific habitat requirements of each species (Kleijn *et al.*, 2001; Franks *et al.*, 2018).

A common goal of AES prescriptions is to reverse the trend towards homogenous swards that have become much more common due to agricultural intensification. Sward homogeneity frequently limits avian diversity and abundance (Perkins *et al.*, 2000; Benton, Vickery, & Wilson, 2003; Wilson, Whittingham, & Bradbury, 2005). Homogenous swards drive avian population declines as vegetation structure determines food availability and nest predation risk (of ground-nesting birds), but the optimum structure varies for each species, and many require different structures for feeding and nest sites (Whittingham & Evans, 2004; Wilson *et al.*, 2005). Shorter and less dense swards enable soil- and surface-foraging birds to access food more easily, but extremely short swards rarely provide suitable nesting sites as predators can locate nests more easily. Taller or denser swards provide more concealment for nests, but very tall or dense swards may be avoided as nesting sites because incubating parents cannot readily detect approaching predators, thus increasing their predation risk (Vickery *et al.*, 2001; Whittingham & Evans, 2004). Changing vegetation structure through AES to generate more diverse sward structures, whilst avoiding a dominance of extremely short, tall or dense swards, is one widely advocated approach to tackling farmland bird declines (Wilson *et al.*, 2005).

Breeding waders are experiencing widespread and marked population declines across Europe (BirdLife International, 2015; BirdLife International, 2017) despite being targeted by AES (Natural England, 2012a; Franks *et al.*, 2018). The primary driver of wader declines is low breeding productivity, mainly due to high nest and chick predation rates that arise through numerous factors including land use change (Roodbergen, van der Werf, & Hötker, 2012; Douglas *et al.*, 2014; Roos *et al.*, 2018). Habitat loss and degradation have also contributed to population declines (Franks *et al.*, 2018). Waders have a diverse range of requirements regarding sward structure. Some species mainly nest in tall, denser vegetation (e.g. common redshank *Tringa tetanus* – threatened within some European countries including the UK; Smart *et al.*, 2006; Eaton *et al.*, 2015), or short, more open vegetation (e.g. northern lapwing *Vanellus vanellus* – globally Near Threatened; Milsom *et al.*, 2000; IUCN, 2020). Other species, including Eurasian curlew *Numenius arquata* (globally Near Threatened; IUCN, 2020) and common snipe *Gallinago gallinago* (threatened within some European countries including the UK; Eaton *et al.*, 2015), use nest sites across much of the gradient in vegetation structure from short, open swards to tall, denser patches of vegetation (Valkama, Roberston, & Currie, 1998; Fisher & Walker, 2015; Wentworth, 2015; Zielonka *et al.*, 2019).

The UK uplands support important breeding populations of declining wader species (Balmer *et al.*, 2013). These regions are targeted by AES that attempt to create more favourable vegetation structures by promoting management of dense *Juncus* spp. (hereafter termed “rush”) swards and other rank vegetation. This management aims to generate less homogenous swards that contain patches of uncut rush as well as cut rush patches that provide shorter, more open vegetation. In theory, this provides open swards that are suitable for foraging and a variety of sward structures that provide suitable nesting sites for upland waders (Natural England, 2012a). This system thus provides a useful framework for assessing the consequences of AES-induced changes in vegetation structure on wader nest predation rates.

In the UK uplands, rush encroachment on grasslands has increased in recent decades (Silcock, Brunyee, & Pring, 2012; Ashby *et al.*, 2020). Rush encroachment is facilitated by high livestock densities due to grazing of other more palatable vegetation (Tweel & Bohlen, 2008) and trampling creating patches of bare ground that enables rush seeds to germinate and establish (Agnew, 1961; Bilotta, Brazier, & Haygarth, 2007). Other potential catalysts include increased soil wetness due to inadequate drainage and soil compaction; insufficient grazing by traditional cattle and pony breeds which are more likely to eat rush (e.g. arising from a switch from mixed grazing to sheep grazing) and land abandonment; reduced fertiliser and lime application; and increased precipitation and warmer winters (Silcock *et al.*, 2012; Ashby *et al.* 2020). Rush encroachment generates tall, dense swards that will limit wader foraging opportunities and reduce the availability of nesting sites, especially for those species that prefer to nest in more open areas (see above). This has been tackled by incorporating rush management within AES prescriptions to improve foraging and nesting conditions for waders (Natural England, 2018).

As of 2009, 83% of the eligible area of purple moor grass *Molinia caerulea* and rush pasture priority habitat in England was managed under AES prescriptions (Natural England, 2009). Current broad AES prescriptions incorporating rush management in the UK (precise prescriptions deviate slightly between component countries) require at least one-third of a field to be covered in rush for a field to qualify for the prescriptions. Within a qualifying field, one-third of the total area of rush needs to be cut annually in rotation (e.g. Natural England, 2012a), although farmers may often cut more than this. The overall objective is to reduce rush cover to less than 30% of the field, with continued management over a minimum of two years required due to the high regrowth capacity of rush (Nielsen *et al.*, 2014; Natural England, 2018; Shellswell & Humpidge, 2018; Kaczmarek-Derda *et al.*, 2019). Targeted prescriptions can vary the extent of rush cutting and desired rush cover depending on the target wader species (Natural England, 2012b; Welsh Government, 2017), with lower rush cover typically desired for lapwing than curlew (Glastir Advanced Management Options 164 and 168; Welsh Government, 2017). Supplementary rush management techniques involve aftermath grazing following cutting to reduce the rate of regrowth (livestock type and number is highly variable and there are no clear guidelines; Natural England, 2018; Shellswell & Humpidge, 2018) and occasionally herbicide application (Natural England, 2018).

Although rush management is a major component of UK upland AES prescriptions, research assessing its effectiveness for reversing wader population declines is limited. Wader abundance may increase following targeted rush management (Holton & Allcorn, 2006; Robson & Allcorn, 2006) or cutting of rank moorland vegetation that includes rush (Fisher & Walker, 2015; Douglas *et al.*, 2017). However, the mechanisms through which rush management influences wader populations are unclear. A key unanswered question, which is especially important given the role of nest predation as a driver of wader population declines, is how rush management influences breeding success. Evidence from other agricultural systems strongly suggests that simpler and more open swards arising from rush management may increase nest predation (Whittingham & Evans, 2004).

Detecting and monitoring a large sample of wader nests is logistically extremely challenging but the relative predation rates of artificial nests that closely mimic real nests, and attract similar predator guilds, can provide useful information for evaluating conservation interventions (Major & Kendal, 1996; Villard & Pärt, 2004). We thus use predation rates of artificial wader nests as an index of predation pressure in treatment fields that follow or emulate AES prescriptions for rush cutting and in nearby untreated control fields. Artificial nests are located in vegetation patches with a wide range of vegetation structures and our results are thus most applicable to waders that nest across this gradient in vegetation structure, such as snipe and curlew (see above). We first test how rush management influences wader nest predation rates by assessing if a) artificial nests in treatment fields, i.e. those with rush management, have higher daily nest predation rates (DPRs) than those in control fields without rush management, and b) artificial nests in cut rush patches within treatment fields have higher DPRs than those in uncut rush patches within the same fields. We then test if the structure of vegetation surrounding nests varies between nests located in treatment and control fields, and between cut and uncut rush patches within treatment fields. These results enable us to confirm that rush management influences vegetation structure. Finally, we model DPRs as a function of vegetation structure and other potentially confounding environmental variables.

3.3 Material and methods

3.3.1 Study areas

Research was conducted in two English upland regions during the wader breeding season (April-June 2019) in the south-west of the Peak District National Park (South West Peak, hereafter “SWP”) and Geltsdale reserve (hereafter “Geltsdale”) in Cumbria (Fig. 3.1), which is jointly owned by the Royal Society for the Protection of Birds and the Weir Trust. Both regions are representative of UK upland farmed landscapes (a mosaic of grassland and moorland) in terms of land management and use, and support populations of breeding waders including curlew and snipe. Study fields were mostly semi-improved pasture with additional unimproved pasture, hay meadow and ‘white moor’ fields (rough grassland with a mixture of rush and *Molinia*). Rush cutting within treatment fields had been managed between autumn 2018 and spring 2019 following or emulating the EK4 and EL4 Entry Level Stewardship prescriptions (Table C.1; Natural England, 2012a). These AES prescriptions are available throughout the UK regardless of whether they are located within a National Park or a reserve. All treatment fields had at least one-third rush cover prior to management, in accordance with AES prescription requirements, and received rush management in the preceding autumn/winter. All control fields had not been managed for rush in the two years prior to the study but had a similar range of rush cover as treatment fields (c.30%, although three fields had 10-30%) to limit the potential for other environmental variables to differ between control and treatment fields and generate confounding factors. Control fields were located close to treatment fields (mean distance = 90 m (95% confidence interval (CI) 23.36 m to 156.64 m)) and were similar in size (Fig. 3.1; Table C.2). In the SWP, we used 12 treatment fields and 13 control fields across 10 farms. At Geltsdale, we used 9 treatment and 9 control fields.

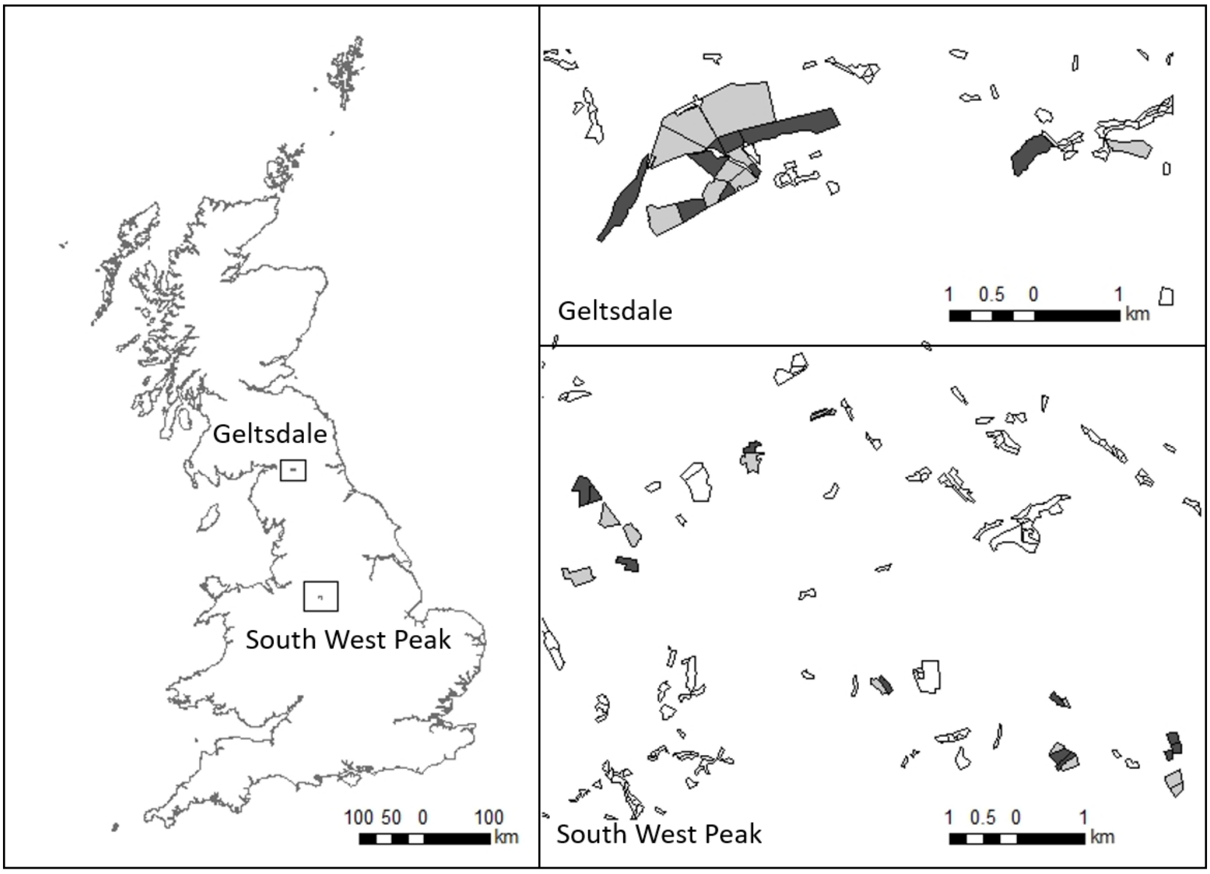

Figure 1. Locations of treatment (light grey) and control (dark grey) fields, with blocks of woodland (white; defined as areas with >20% tree cover, from Land Cover Map 2015; Rowland *et al.*, 2017) in the surrounding landscape.

3.3.2 Artificial nest deployment and predation rates

Artificial wader nests were deployed within the typical breeding season of upland waders (Joys & Crick, 2004) from 1st April – 28th April (early breeding season) and 28th May – 18th June (late breeding season) in the SWP and 3rd May – 20th May (early breeding season) at Geltsdale (late season nests were not deployed in Geltsdale due to logistical constraints). Nests contained three fresh Japanese quail *Coturnix japonica* eggs and a plasticine egg (to aid separation of avian and mammalian predators). The latter was made by adapting the methods of Martin, Dueser, and Moncrief (2010; Fig. C.1) using Newplast modelling clay (Newclay Product Ltd, Devon, England) and PlastiDip® coating (PlastiDip UK Ltd, Hampshire, England) to minimise plasticine scent which could influence predators (Purger *et al.*, 2012). The markings of plasticine and quail eggs were similar to each other and those of curlew and snipe, and their dimensions are close to those of snipe eggs (Cramp & Simmons, 1982; Robinson, 2005; Smith, Gilchrist, & Smith, 2007; Fig. C.2). Green garden twine (30 cm) was embedded in each plasticine egg and ground-tethered to hinder removal by predators.

Latex gloves were worn to minimise human scent and disturbance of vegetation around nests was minimised. The number of nests varied with field size using estimates of snipe nest densities (Green, 1985) which are typically intermediate between those of more colonial lapwing and less aggregated curlew (Cramp & Simmons, 1982). Fields <4 ha received two nests (22 fields; median = 2.03 ha; range = 0.40-3.55 ha; four <1.05 ha fields within the same landholding (two treatment and two control) in the SWP received only one nest), and fields >4 ha (21 fields; median = 7.06 ha; range = 4.16-40.64 ha) received four nests (Table C.3).

Nest placement was conducted using an approach that ensured that there was no systematic bias in nest placement which could confound our analyses. In control fields, nests were placed in patches of uncut rush (the only type of rush available). In treatment fields, nests were placed in patches of cut rush except for fields >4 ha where one nest was placed in uncut rush (Table C.4). Placing nests in cut and uncut rush patches within the same field provides an additional check that differing nest predation rates between treatment and control fields is due to rush cutting rather than unrelated attributes of treatment fields. Upon entering a field, patches of cut and/or uncut rush were identified via a scan of the field. To select precise nest locations, the number of paces required to stop in one of the rush patches, without bias towards the centre or edge of the patch, was estimated. After walking this number of paces, a natural depression in the immediately adjacent ground was selected for the nest scrape. If a natural depression was unavailable, a scrape (15 cm diameter x 5 cm depth) was created using a small trowel. Eggs were placed in the scrape which was lined with a handful of dried vegetation; artificial nests were thus similar in appearance and location to real wader nests including those of snipe and curlew (Cramp & Simmons, 1982; Fig. C.2). To aid relocation, a blue wooden golf tee was discreetly placed flush to the ground and a 60 cm bamboo cane topped with red tape was placed 10 m away in a random direction (Smith *et al.*, 2007; Pedersen *et al.*, 2018); the use of such canes does not alter wader nest predation rates (Zámečník, Kubelka, & Šálek, 2018). The mean distance (95% CI) between nests within a field was 80.97 m (75.80 m to 86.14 m), and from each nest to the nearest field boundary was 50.66 m (47.49 m to 53.84 m).

Nests were deployed until predated, or for 15 days. The length of this maximum exposure period was determined by a trade-off between use of a longer exposure period that would provide a closer match to wader incubation periods (e.g. 18-20 days for snipe; Robinson, 2005) and maximising the number of nests that could be deployed and monitored; the duration of our maximum exposure period is sufficient to generate a reliable estimate of DPRs. Nests were checked every 5 days (±1 day in both cases depending on weather conditions to avoid disturbing real wader nests in these fields during inclement weather). Nests were classified as predated if at least one egg was missing, damaged or outside the nest scrape in the immediate surroundings (Smith *et al.*, 2007; Pedersen *et al.*, 2018). Trampled nests (7.1% of 184 nests; crushed eggs with contents remaining in the shell or on surrounding ground) were excluded from further analyses and trampling rates were similar in treatment (7.1% of 98 nests) and control fields (7.0% of 86 nests).

For predated nests, the plasticine eggs (if found) were assigned to predator type using bill or tooth marks following Trnka, Prokop, and Batáry (2008) and Bocz *et al.* (2017). Two assessors working independently classified each plasticine egg predator as unknown, avian, mammalian, or signs of both avian and mammalian predators.

3.3.3 Environmental variables

Habitat around nests was recorded during the first monitoring visit. Vegetation height (to the nearest 1 cm) and density were measured at four equally spaced points on the nest scrape edge, and four equally spaced points 1 m away from the nest. Mean values were calculated from all eight measures. Vegetation density was the number of concealed white bands – five 2 cm wide bands at 10 cm intervals from 0-40 cm on a pole – and was measured visually by viewing the upright pole at a height of approximately 85 cm from a point approximately 45 cm horizontally from the pole (Sansom, Pearce-Higgins, & Douglas, 2016). More concealed bands indicate denser vegetation. Rush cover within a 5 m radius of each nest was estimated visually to the nearest 5%.

Field size (ha) was measured from 1:25 000 maps (Ordnance Survey, 2019) using ArcMap™ (v10.4.1; Esri, Redlands, CA, USA); some fields had identical sizes. Straight-line distance (m) from each nest to the nearest field boundary was measured using the 'Near (Analysis)' tool to account for potential edge effects in nest predation risk. Similarly, we measured the straight-line distance from each nest to the nearest block of woodland (defined as areas with >20% tree cover, from Land Cover Map 2015; Rowland *et al.*, 2017) as this represents a metric of real or perceived nest predation risk for waders (e.g. Wilson *et al.*, 2014). We used a straight-line distance, rather than a metric which attempts to measure routes potentially taken by predators (such as through gates) as many important mammalian predators of wader nests (mustelids and rodents) can pass through gaps in fields boundaries (fence lines, small gaps in dry stone walls etc.) and avian predators are not constrained by boundary features.

3.3.3.1 Predator control

Predator control was classified across each of the 11 landholdings in which our study fields were located using a semi-structured interview with land managers and owners (Table C.5) following approval from the University of Sheffield Research Ethics Committee (application number 030271). Informed consent was gained from all interviewees. 'Regular' predator control comprised 3-4 nightly patrols for red foxes *Vulpes vulpes* per week (January-June inclusive) and daily Larsen trapping of corvids (March-April inclusive) with a full-time contractor (with one landholding also conducting daily mustelid trap checks; January-June inclusive). 'Negligible/no' control comprised no corvid control and no fox control, except one landholding with occasional fox control.

3.3.3.2 Avian predator abundance

Avian predator surveys were conducted in each study field – two during the early breeding season in all fields and two during the late breeding season in SWP fields only. Surveys were not undertaken in the first hour after sunrise or last hour before sunset, during heavy rain, in fog (<250 m visibility) or if wind speed was greater than Beaufort Force 5. The entirety of each study field, to within a distance of 50 m, was walked during each survey and all avian predators (regardless of their activity) were recorded except those flying more than 30 m above the ground (which were considered unlikely to be using or searching for resources in the field). We calculated two indices of potential avian predator activity: corvid abundance (carrion crow *Corvus corone*; rook *Corvus frugilegus*; Eurasian jackdaw *Corvus monedula*; Eurasian magpie *Pica pica*; unidentified corvid; Leigh, Smart, & Gill, 2017) and total avian predator abundance (corvids, gulls, raptors, and herons). The indices were generated for the early and late breeding seasons separately by calculating the mean number of individuals observed over the two surveys per study field. Because gulls, raptors, and herons were rarely observed, corvid abundance and total avian predator abundance were highly correlated (early breeding season: $r_s = 0.971$, $P = 2.2^{-16}$; late breeding season: $r_s = 0.980$, $P = 2.2^{-16}$). Therefore, corvid abundance was the only measure included in further analyses (carrion crow are the primary avian predator of wader nests; MacDonald & Bolton, 2008; Teunissen *et al.*, 2008). Mammalian predator surveys were logistically unfeasible due to time constraints.

3.3.4 Statistical analyses

All analyses were conducted in R 3.6.3 (R Core Team., 2020). Our general approach is to use full models to test our core hypotheses that i) rush management and ii) metrics of vegetation structure influenced by rush management (vegetation density, vegetation height, and rush cover) influence nest predation rates whilst taking into account potentially confounding variables (Table C.6). This full model approach is a suitable method, especially in experimental settings (Burnham & Anderson, 2002), and superior to selection of a single best model through step-wise model selection techniques which can generate biased parameter estimates (Whittingham *et al.*, 2006; Mundry & Nunn, 2009).

We modelled daily nest predation rates (DPRs) using Mayfield logistic regressions following the Hazler (2004) method and constructed generalised linear mixed effects models (GLMMs; fit by maximum likelihood with Laplace approximation) using the lme4 package (Bates *et al.*, 2015). The response variable was DPR i.e. nest outcome (1 = predated; 0 = not predated)/exposure days. Exposure days was the number of days between nest establishment and failure date, or date of final monitoring visit. Failure date was calculated as the mid-point between the monitoring visit when the nest was last observed intact and the subsequent visit when the nest had failed. Thus, exposure days could have non-integer values. These were converted to integer values by rounding up for odd numbered nests and down for even numbered nests to avoid problems generated by consistently over-estimating exposure days if 0.5 values were consistently rounded upwards (Johnson, 2007). All models of DPRs were constructed with a binomial error structure (logit link) and field identity as a random effect as each field contained more than one artificial nest. Continuous predictor variables were centred and scaled using the scale function. Theoretical conditional R^2 values were calculated for each model using the MuMIn package (Barton, 2019) and represent model fit (Nakagawa & Schielzeth, 2013; Nakagawa, Johnson, & Shielzeth, 2017). We report profile and bootstrap 95% CIs of parameter estimates.

3.3.4.1 Effect of rush management on daily nest predation rates

We first modelled DPRs, using data from all artificial nests, as a function of location in treatment or control fields whilst accounting for region, field identity (random factor), deployment date, and other environmental variables (woodland distance, boundary distance, field size, predator control, and corvid abundance) except those relating to vegetation structure (Table 3.1, model i). We then used data from treatment fields >4 ha and modelled DPRs as a function of artificial nest location in cut or uncut rush whilst accounting for region, field identity (random factor), deployment date, and other nest level environmental variables except those relating to vegetation structure (Table 3.1, model ii).

3.3.4.2 Associations between rush management and environmental variables

We conducted subsidiary analyses to test whether vegetation structure differed between nests in treatment and control fields and whether other environmental variables (woodland distance, boundary distance, field size, and corvid density) differed between treatment and control fields. Similarly, we tested whether there were differences in the structure of vegetation surrounding nests in cut and uncut rush patches within treatment fields that contained nests in both habitat types (fields > 4 ha). We used GLMMs (lme4 package; Gaussian error structure and identity link) when there was more than one data point per field (nest level variables; with field identity as a random factor) and generalised linear models (GLMs; Gaussian error structure and identity link) in other cases (field level variables; see Table C.7, C.8).

3.3.4.3 Effect of vegetation structure on daily nest predation rates

Finally, we modelled DPRs as a function of vegetation structure whilst accounting for other environmental variables. Vegetation density and vegetation height were highly correlated so could not be included in the same model (r = 0.74; Dormann *et al.*, 2013). We first used GLMMs (lme4 package) to model DPRs, using data from all artificial nests, as a function of vegetation density whilst accounting for region, field identity (random factor), deployment date, and other environmental variables (Table 3.1, model iii). We repeated these models replacing vegetation density with vegetation height (Table 3.1, model iv) as this is an easier metric to obtain in the field and may be more practical for conservation managers.

Table 3.1. Structure of the four generalised linear mixed effects models (binomial (logit)) of daily nest predation rate (DPR), the dataset used, the optimiser used, the model distribution and link function, and the predictor variables (fixed and random) included. The bobyqa optimiser was used for model iii following the recommendation of lme4 package author, Ben Bolker, as the model failed to converge with the default optimiser (combination of Nelder-Mead and bobyqa). For a detailed breakdown of sample sizes, see Table C.3, C.4.

Model	Dataset	Optimiser	Distribution (link)	Predictor variables
Objective – test effect of rush management on daily nest predation rates				
i	All fields (n = 43) 91 treatment nests; 80 control nests	Combination of Nelder-Mead and bobyqa	Binomial (logit)	Treatment + Woodland distance + Boundary distance (square root transformed) + Deployment date + Field size (natural logarithm transformed) + Predator control + Corvid abundance (natural logarithm (x+1) transformed) + Region + (1|Field identity)
ii	Treatment fields >4 ha (n = 13) 50 nests in cut rush; 17 nests in uncut rush	Combination of Nelder-Mead and bobyqa	Binomial (logit)	Cut or uncut rush + Woodland distance + Boundary distance (square root transformed) + Deployment date + Region + (1|Field identity) * * Note that the three variables measured at the field level (field size, predator control, and corvid abundance) were excluded from this model because it is comparing nests within the same field. Field identity was retained as a random factor.
Objective – test effect of vegetation structure on daily nest predation rates				
iii	All fields (n = 43) 171 nests	bobyqa	Binomial (logit)	Vegetation density + Rush cover + Woodland distance + Boundary distance (square root transformed) + Deployment date + Field size (natural logarithm transformed) + Predator control + Corvid abundance (natural logarithm (x+1) transformed) + Region + (1|Field identity)
iv	All fields (n = 43) 171 nests	Combination of Nelder-Mead and bobyqa	Binomial (logit)	Vegetation height + Rush cover + Woodland distance + Boundary distance (square root transformed) + Deployment date + Field size (natural logarithm transformed) + Predator control + Corvid abundance (natural logarithm (x+1) transformed) + Region + (1|Field identity)

3.4 Results

3.4.1 Predator type

Marked plasticine eggs (n = 45) suggest that mammals were the main nest predators, being identified as the sole predators in 64% (n = 39 eggs assigned to a known predator; assessor #1) and 85% (n = 41 eggs assigned to a known predator; assessor #2) of predation events. Equivalent estimates for avian predators were 23% (assessor #1) and 10% (assessor #2), with joint avian and mammalian predation events estimated at 13% (assessor #1) and 5% (assessor #2).

3.4.2 Effect of rush management on daily nest predation rates

DPRs were significantly higher in treatment than control fields (z = 3.038, P = 0.002; Table 3.2, C.9, model i). This equates to 2.35 times higher DPR in treatment fields (0.064 day^{-1}; bootstrap 95% CI 0.029 to 0.118 day^{-1}) than control fields (0.027 day^{-1}; bootstrap 95% CI 0.009 to 0.059 day^{-1}; Fig. 3.2a). Within treatment fields in which nests were located in cut and uncut rush patches (fields >4 ha), DPRs were significantly higher in cut rush patches (z = 1.989, P = 0.047; Table 3.2, C.9, model ii). This equates to 1.96 times higher DPR in cut rush patches (0.108 day^{-1}; bootstrap 95% CI 0.052 to 0.180 day^{-1}) than uncut rush patches (0.055 day^{-1}; bootstrap 95% CI 0.021 to 0.106 day^{-1}; Fig. 3.2b). No other predictor variable had a consistent significant influence on DPRs (Table 3.2).

3.4.3 Associations between rush management and environmental variables

Environmental variables (woodland distance and boundary distance) around artificial nests in treatment and control fields were similar except that nests in treatment fields were surrounded by shorter, less dense vegetation and lower rush cover – although the difference in rush cover was marginally non-significant in the late breeding season (Table C.2, C.7; Fig. C.3). Within treatment fields >4 ha, artificial nests located in cut rush patches were surrounded by significantly shorter, less dense vegetation and lower rush cover than nests in uncut rush patches (Table C.8, C.10; Fig. C.3). Field level variables (field size and corvid density) did not differ significantly between treatment and control fields (Table C.2, C.7).

3.4.4 Effect of vegetation structure on daily nest predation rates

DPRs were significantly higher for nests surrounded by less dense vegetation and shorter vegetation (vegetation density: z = -5.061, P = 4.165^{-7}; vegetation height: z = -4.483, P = 7.367^{-6}; Table 3.2, C.9, models iii and iv). Predicted DPRs and bootstrap 95% CIs across the observed range of vegetation density and vegetation height values are shown in Figs. 2c and 2d. No other predictor variables had a significant influence on DPRs (Table 3.2).

Table 3.2. Results of generalised linear mixed effects models (GLMMs) investigating the effect of rush management and vegetation structure on daily nest predation rates (DPRs) of artificial nests in all study fields (models i, iii, and iv) and treatment fields > 4 ha (model ii). For each GLMM, parameter estimates and profile 95% confidence intervals (CIs) are presented for the predictor variables (see Table C.9 for alternative bootstrap CIs), with significant predictor variables in bold. Control fields are the reference level for treatment; uncut rush patches are the reference level for cut or uncut rush; negligible/no predator control is the reference for predator control; Geltsdale is the reference for region. Theoretical conditional R^2 (Nakagawa & Shielzeth, 2013; Nakagawa, Johnson, & Shielzeth, 2017), Akaike information criterion with correction for small sample size (AICc), and the dispersion parameter are presented for each GLMM.

Model	Predictor variables in models	Estimate (profile 95% CIs)	*z* statistic	*P*	R^2	AICc	Dispersion parameter
Effect of rush management on daily nest predation rates							
i	Treatment	**0.894 (0.310 to 1.501)**	**3.038**	**0.002**	0.156	409.65	1.218
	Woodland distance	-0.050 (-0.402 to 0.297)	-0.291	0.771			
	Boundary distance (square root transformed)	0.156 (-0.143 to 0.459)	1.022	0.307			
	Deployment date	**-0.287 (-0.506 to -0.074)**	**-2.612**	**0.009**			
	Field size (natural logarithm transformed)	-0.221 (-0.632 to 0.183)	-1.088	0.276			
	Predator control	-0.474 (-1.202 to 0.206)	-1.357	0.175			
	Corvid abundance (natural logarithm (x+1) transformed)	-0.181 (-0.470 to 0.113)	-1.230	0.219			
	Region	0.093 (-0.846 to 1.072)	0.195	0.845			
ii	Cut or uncut rush	**0.728 (0.038 to 1.496)**	**1.989**	**0.047**	0.127	174.42	0.998
	Woodland distance	0.260 (-0.360 to 0.874)	0.864	0.388			
	Boundary distance (square root transformed)	-0.245 (-0.596 to 0.097)	-1.409	0.159			
	Deployment date	0.040 (-0.264 to 0.345)	0.259	0.795			
	Region	0.065 (-1.239 to 1.432)	0.102	0.919			
Effect of vegetation structure on daily nest predation rates							
iii	Vegetation density	**-0.735 (-1.027 to -0.453)**	**-5.061**	**4.165^{-7}**	0.220	387.54	1.264
	Rush cover	-0.051 (-0.324 to 0.217)	-0.372	0.710			
	Woodland distance	-0.106 (-0.421 to 0.201)	-0.693	0.489			
	Boundary distance (square root transformed)	0.209 (-0.093 to 0.515)	1.357	0.175			
	Deployment date	-0.176 (-0.412 to 0.058)	-1.477	0.140			
	Field size (natural logarithm transformed)	-0.113 (-0.489 to 0.267)	-0.597	0.551			
	Predator control	-0.246 (-0.896 to 0.377)	-0.783	0.434			

	Corvid abundance (natural logarithm (x+1) transformed)	-0.206 (-0.478 to 0.067)	-1.506	0.132			
	Region	0.639 (-0.167 to 1.491)	1.571	0.116			
iv	Vegetation height	**-0.766 (-1.105 to -0.432)**	**-4.483**	**7.367^{-6}**	0.204	392.79	1.579
	Rush cover	0.046 (-0.246 to 0.319)	0.319	0.750			
	Woodland distance	-0.101 (-0.411 to 0.197)	-0.676	0.499			
	Boundary distance (square root transformed)	0.203 (-0.101 to 0.512)	1.305	0.192			
	Deployment date	-0.083 (-0.334 to 0.167)	-0.651	0.515			
	Field size (natural logarithm transformed)	-0.119 (-0.493 to 0.254)	-0.640	0.522			
	Predator control	-0.406 (-1.051 to 0.205)	-1.312	0.190			
	Corvid abundance (natural logarithm (x+1) transformed)	-0.168 (-0.433 to 0.104)	-1.252	0.211			
	Region	0.395 (-0.398 to 1.238)	0.991	0.322			

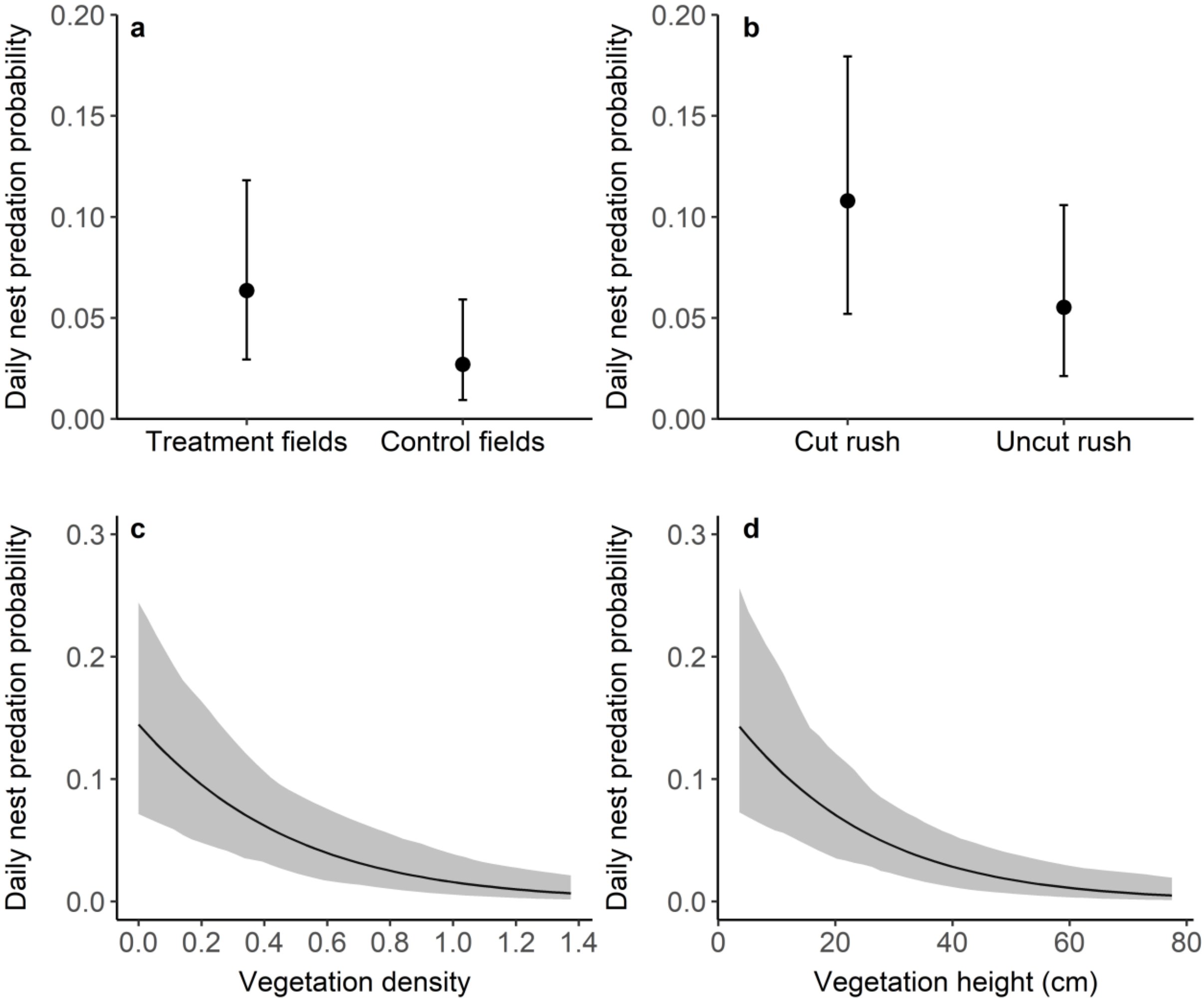

Figure 3.2. Predicted daily nest predation probability of artificial nests in relation to (a) treatment and control fields, (b) cut and uncut rush patches within treatment fields > 4 ha, (c) vegetation density and, (d) vegetation height. In (a) and (b), points represent mean values and vertical lines represent bootstrap 95% confidence intervals from models i and ii presented in Table 3.2. In (c) and (d), shaded ribbons represent bootstrap 95% confidence intervals from models iii and iv presented in Table 3.2.

3.5 Discussion

When taking other environmental variables into account, artificial wader nests located in areas of rush that had been cut in accordance with AES prescriptions had daily nest predation rates that were approximately double those of nests in unmanaged areas of rush. This pattern was consistent when comparing nests in treatment and control fields, and when comparing patches of cut and uncut rush within treatment fields. Nests in cut rush were surrounded by shorter and less dense vegetation than nests in unmanaged rush, and the risk of nest predation increased as vegetation height and density decreased.

3.5.1 Artificial nests as indicators of predation risk

Artificial nest experiments require careful interpretation. The absence of parental nest defence could increase artificial nest predation rates (Berg, 1996), especially for species such as lapwing which exhibit strong nest defence, but less so for other waders such as curlew and snipe which exhibit less active nest defence and rely more on nest concealment (Cramp & Simmons, 1982; Vickery *et al.*, 2001). The absence of potential additional camouflage provided by the plumage of incubating adults could also increase artificial nest predation rates relative to actual nests (Troscianko *et al.*, 2016). Conversely, the lack of parental cues (nest visits for incubation, odour of an incubating adult) could reduce predation rates of artificial nests compared to real nests (Berg, 1996). The deposition of human scent on artificial nests could also influence predation rates and counteract the lack of odour from incubating adults (Zanette, 2002), although our experimental design followed protocols to minimise human scent trails.

Nevertheless, artificial nest predation rates can provide useful information for addressing key questions including relative predation risk between experimental treatments and quantifying variation in predation risk along environmental gradients (Ibáñez-Álamo *et al.*, 2015). This is reinforced when artificial nests closely mimic real nests and attract similar predator guilds (Major & Kendal, 1996; Villard & Pärt, 2004). We believe that our experimental study meets these requirements for three reasons. First, artificial nests were in similar locations to those of snipe and curlew nests including in terms of their vegetation structure (Cramp & Simmons, 1982; Durant *et al.*, 2008; Fig. C.2), with snipe and curlew frequently nesting in sward structures across a gradient from short and open swards to relatively tall and/or dense swards (Valkama *et al.*, 1998; Fisher & Walker, 2015; Wentworth, 2015; Zielonka *et al.*, 2019). Notably, we found real snipe and curlew nests during the 2019 fieldwork in rush patches that had been cut the previous winter, and these were found in the early breeding season before any substantial regrowth had occurred (L. Kelly, pers. obs). This demonstrates that some individuals nest in areas managed under AES rush cutting prescriptions despite higher predation rates of artificial nests in such locations. Second, our DPRs (treatment fields = 0.064 day^{-1}; control fields = 0.027 day^{-1}) are within the range of those reported in studies of real snipe and curlew nests (MacDonald & Bolton, 2008). Finally, the high rate of mammalian predation concurs with research on real wader nests (MacDonald & Bolton, 2008).

3.5.2 Rush management and daily nest predation rates

Rush management following or emulating AES prescriptions can generate suitable habitat conditions for breeding waders in locations which would otherwise be unsuitable due to rush encroachment (Holton & Allcorn, 2006; Robson & Allcorn, 2006; Fisher & Walker, 2015; Douglas *et al.*, 2017). Yet, such rush management doubled DPRs compared to a control that lacked rush management, and these differences are attributable to rush management impacts on vegetation structure. These patterns probably arise due to shorter and sparser vegetation increasing the visibility of nests to predators (Whittingham & Evans, 2004) and mammalian predators are more likely to travel through less dense vegetation, increasing the detection of ground nests (Donald *et al.*, 2002). This is particularly pertinent for snipe and curlew as concealment is one of the primary forms of nest defence (Cramp & Simmons, 1982; Vickery *et al.*, 2001).

If breeding waders select areas of recently cut rush for nesting, then rush management that follows or emulates AES prescriptions appears likely to create an ecological trap due to higher nest predation rates in such locations. Such ecological traps are plausible as it cannot be assumed that waders always select nesting locations that minimise the risk of nest predation (e.g. Hegyi & Sasvári, 1997) as many factors determine nest site choice (Blomqvist & Johansson, 1995; Smart *et al.*, 2013). Indeed, some waders will select nest sites that increase the probability of parents detecting, and thus escaping, approaching predators even though the risk of nest predation is greater at such sites (Whittingham & Evans, 2004; Gómez-Serrano & López-López, 2014). Further evidence for the possibility of ecological traps is provided by their occurrence in other ground-nesting farmland birds (e.g. nest site selection by western yellow wagtail *Motacilla flava*, Gilroy *et al.*, 2011). Ecological traps only arise when suboptimal habitats that reduce fitness are not selected against (Battin, 2004) and demonstration of an ecological trap in our study system thus requires robust data on nest site selection patterns. However, even if rush cutting does not create an ecological trap, it is clear that curlew and snipe do sometimes nest in cut rush and these individuals are likely to experience reduced breeding success that could reduce population growth rates. Indeed, wader population declines are driven largely by poor reproductive output (Roodbergen *et al.*, 2012) and the intended benefits of rush cutting may not occur. This situation is most likely to arise in wader species that nest across a wide range of sward structures, such as curlew and snipe (Valkama *et al.*, 1998; Fisher & Walker, 2015; Wentworth, 2015; Zielonka *et al.*, 2019).

3.5.3 Implications for managing upland sward structure and further research requirements

Our artificial nest experiment meets the conditions required to provide a reasonable indicator of environmental variation in DPRs (see 'Artificial nests as indicators of predation risk' section). It provides evidence that rush management through AES prescriptions could reduce breeding success for individuals nesting in those locations (especially snipe and to a lesser extent curlew due to these species' reliance on nest crypsis for reducing predation risk; although, curlew do exhibit some active nest defence; Cramp & Simmons, 1982; Vickery *et al.*, 2001). This could generate ecological traps but nest site selection studies combined with assessments of predation rates of real wader nests are required to demonstrate this. Achieving this will require a major investment in fieldwork over multiple seasons to achieve sufficient sample sizes, which is why we initially assessed patterns using the more rapid assessment that could be conducted using artificial nests. Our results demonstrate that cut rush is likely to increase nest predation rates, and thus a mosaic of cut and uncut rush will be required to generate heterogeneous swards that breeding waders require.

Future work should, however, explore solutions to the trade-off between the need to manage rush to generate open swards that improve foraging conditions, and the adverse impacts of such swards for breeding success, particularly for wader species which rely on nest concealment. It may also be important to confirm whether rush management provides a universal benefit to species such as lapwing, which primarily nest in shorter swards (Milsom *et al.*, 2000). This research would require assessing (at a range of spatial scales) how different wader species respond to variation in the relative amounts of cut and uncut rush patches, and their spatial configuration.

Concern over the encroachment of rush and other rank vegetation in upland areas that support important breeding populations of waders and other-ground nesting birds (Silcock *et al.*, 2012; Ashby *et al.*, 2020) suggests that ongoing management of vegetation structure is required. Our study highlights the need, however, for investment in robust evaluation of AES prescriptions targeting vegetation structure in order to ensure that prescriptions balance trade-offs against all demographic factors influenced by vegetation structure, including nest predation risk.

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3.7 Data accessibility

Data are expected to be archived in the Dryad Digital Repository.

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Chapter 4

Inter-specific variation in the potential for upland rush management advocated by agri-environment schemes to increase breeding wader densities

A rush pasture field at Geltsdale reserve, Cumbria

This chapter is currently in review at *Frontiers in Ecology and Evolution* as:

Kelly, L.A., Douglas, D.J.T., Shurmer, M.P. and Evans, K.L. Inter-specific variation in the potential for upland rush management advocated by agri-environment schemes to increase breeding wader densities.

4.1 Abstract

Encroachment of rush *Juncus* spp. in the UK uplands poses a threat to declining wader populations due to taller, denser swards that can limit foraging and breeding habitat quality for some species. Rush management implemented through agri-environment schemes (AES) could thus increase wader abundance, but there is insufficient assessment and understanding of how rush management influences upland waders. Across two upland regions of England (South West Peak (SWP) and Geltsdale nature reserve, Cumbria), we surveyed waders over four visits in fields where rush was managed according to AES prescriptions (treatment; n = 21) and fields without rush management that were otherwise ecologically similar (control; n = 22) to assess how the densities of breeding wader pairs respond to rush management in the short-term. We find some evidence for regional variation in the response of waders to rush management, with densities of Common Snipe *Gallinago gallinago* significantly higher in treatment than control fields in the SWP, but not Geltsdale. There were no statistically significant responses to treatment on densities of Eurasian Curlew *Numenius arquata* or Northern Lapwing *Vanellus vanellus*. The 95% confidence intervals for the treatment parameter estimates suggest that this may be due to limited statistical power in the case of Lapwing, but not Curlew, for which any potential increases in densities are negligible. There was no evidence that variation in rush cover, which reached a maximum of 70%, influenced densities of any of our three focal species. Our results suggest that rush management through AES prescriptions delivered in isolation of other interventions may not lead to general increases in breeding wader densities in the short-term, but benefits may arise in some situations due to regional and inter-specific variation in effectiveness. Rush management supported with interventions that improve soil conditions and thus food availability, or reduce predation pressure, may enable AES rush management to generate benefits. Additional research is required to maximise the potential benefits of rush management for each species through the development of prescriptions that tailor to individual species' optimum sward structure.

4.2 Introduction

Waders are one of several taxonomic groups of farmland breeding birds that have undergone severe Europe-wide declines in recent decades (PECBMS, 2020), with species threatened at both the national level (for example, Common Snipe *Gallinago gallinago* (hereafter Snipe); Amber-listed, UK; Eaton et al., 2015) and international level (for example, Eurasian Curlew *Numenius arquata* (hereafter Curlew) and Northern Lapwing *Vanellus vanellus* (hereafter Lapwing); globally Near Threatened; IUCN, 2020). In the UK, wader population declines are occurring throughout the lowlands and uplands, with the latter containing the majority of remaining grassland-breeding wader populations (Wilson et al., 2005a; Balmer et al., 2013; Siriwardena et al., 2017). Poor nest and chick survival, primarily attributable to predation and habitat degradation, are thought to be the major drivers of these declines (Roodbergen et al., 2012; Franks et al., 2018; Roos et al., 2018). Land use change including agricultural intensification, and to a lesser extent land abandonment, is driving degradation of wader habitat in UK upland regions (Baines, 1988; Fuller and Gough, 1999; Amar et al., 2011; Silcock et al., 2012; Douglas et al., 2017; Johnstone et al., 2017).

To prevent further degradation of grasslands, agri-environment schemes (AES) encourage farming practices that benefit breeding waders through improved habitat quality (O'Brien and Wilson, 2011; Smart et al., 2013, 2014; Franks et al., 2018). Such AES prescriptions have exhibited mixed success; population trends of some wader species have been reversed at the local level, yet nationally wader population declines continue (O'Brien and Wilson, 2011; Smart et al., 2013, 2014; Siriwardena et al., 2017; Franks et al., 2018). One mechanism used in these AES prescriptions is the manipulation of vegetation structure (for example, Natural England, 2018) with the aim of creating a mosaic of short and tall vegetation that is beneficial for foraging and breeding waders (Wilson et al., 2005b). Due to the substantial inter-specific variation in wader breeding habitat requirements, creation of a mosaic aims to simultaneously provide suitable habitat for several wader species. Lapwing, for example, favour short swards with a few tussocks (Baines, 1988; Milsom et al., 2000; Durant et al., 2008), whereas Curlew and Snipe may be more tolerant of a range of sward structures with a greater preference for taller vegetation (Baines, 1988; Pearce-Higgins and Grant, 2006; Hoodless et al., 2007; Durant et al., 2008).

A recent threat to the maintenance of structurally heterogeneous grasslands in the UK uplands is encroachment by *Juncus* spp. (hereafter termed "rush"), with rush frequency approximately doubling between 2005 and 2018 (Ashby et al., 2020). Rush encroachment could significantly contribute to wader population declines by creating expanses of tall, dense, rush-dominated swards that restrict physical access to the soil for foraging (Devereux et al., 2004; Robson and Allcorn, 2006), and reduce waders' ability to detect predators and thus their willingness to breed and forage in such locations (Whittingham and Evans, 2004; Robson and Allcorn, 2006). Although, the taller, denser vegetation generated by increased rush cover could provide nests and chicks with greater concealment from predators (Valkama et al., 1998; Kelly et al. 2021).

Rush management prescriptions within AES have been developed to address the adverse impacts of rush encroachment (Natural England, 2018). These prescriptions, which typically comprise a long-term aim to reduce the extent of dense rush swards within a field to <30%, involve mowing, aftermath grazing, and occasionally herbicide application (precise prescriptions deviate slightly between UK countries; Natural England, 2012; Welsh Government, 2017; Shellswell and Humpidge, 2018). In the short-term, rush management opens up the sward and reduces vegetation height and density (Kelly et al., 2021). There is, however, little published data supporting the assumed beneficial impacts of these changes in sward structure on breeding waders. Whilst previous studies suggest that targeted rush management, or cutting of rank vegetation including rush, can increase wader abundance, these studies do not experimentally compare areas with and without rush management and in some cases are combined with additional interventions (Holton and Allcorn, 2006; Robson and Allcorn, 2006; Douglas et al., 2017). Consequently, there is insufficient assessment and understanding of how rush management influences upland waders, despite the importance of evaluating the effectiveness of AES prescriptions (Kleijn and Sutherland, 2003).

Here, we assess how the number of breeding wader pairs responds to within-field rush management in the short-term by surveying waders in treatment fields (where rush is managed according to AES prescriptions) and control fields (without rush management) across two upland regions of England. We first test whether field size and environmental conditions that could influence wader abundance (rush cover, Holton and Allcorn (2006), Robson and Allcorn (2006); soil conditions (pH, moisture, and penetration resistance), Smart et al., (2006), Hoodless et al., (2007), Smart et al., (2008), McCallum et al., (2016); woodland distance, Wilson et al., (2014), Douglas et al., (2014)) are similar between control and treatment fields. We then test whether the density of breeding wader pairs differs between treatment and control fields whilst accounting for environmental conditions and region, and test if the effects of rush management vary between regions and with the amount of rush cover.

4.3 Material and Methods

4.3.1 Study areas

This study was conducted during the wader breeding season (April-June 2019) in the south-west of the Peak District National Park (South West Peak, hereafter “SWP”; in fields managed by various landowners including private farmers, tenant farmers on Peak District National Park Authority owned land, and (in three cases) the Staffordshire Wildlife Trust), and Geltsdale nature reserve in Cumbria, jointly owned by the Royal Society for the Protection of Birds (RSPB) and the Weir Trust (hereafter “Geltsdale”; in fields managed by tenant farmers and RSPB staff; Fig. 4.1). Both regions support important breeding wader populations including Curlew, Lapwing, and Snipe (Carr, 2009; Balmer et al., 2013; Douglas et al., 2017). Survey fields within the two regions were characteristic of UK upland farmed landscapes and were mostly semi-improved pasture with a smaller number of unimproved pasture, hay meadow and ‘white moor’ fields (rough grassland with rush and *Molinia*). The dominant rush species was *Juncus effusus* with smaller amounts of other species present at some sites, particularly *Juncus acutiflorus* and *Juncus conglomeratus* at Geltsdale.

The study design is described in full by Kelly et al. (2021). Field selection was performed without prior knowledge on wader use of the selected fields, and was based on fields meeting our criteria on rush management, spatial configuration and obtaining permission from landowners to conduct the research. Treatment fields were selected if fields had received rush management between autumn 2018 and spring 2019 (fields may also have received management in previous years) following the EK4 and EL4 rush cutting prescriptions in Entry Level Stewardship (Natural England, 2012). These are standard AES prescriptions that are applicable to any field with at least one-third rush cover, including those on nature reserves. Rush management in treatment fields involved cutting one-third of the rush present once or twice annually on rotation (Table D.1 provides more information on the AES prescriptions). Control fields were selected if fields had not undergone rush management in the previous two years, had a similar extent of rush cover to treatment fields (mean rush cover ± standard error, treatment = 46.70 ± 3.67%, control = 40.00 ± 4.21%; Mann-Whitney test: W = 186, P = 0.255) and were in close proximity to treatment fields (mean distance = 90 ± 34 m standard error; Fig. 4.1; Table D.2). Information on field rush cover was provided by the landowners prior to selection of our survey fields. For both treatment and control fields, we only selected those that had greater than one-third rush cover so that all survey fields, regardless of treatment, qualified for the AES rush management prescriptions. Control fields were not deliberately selected to contain different levels of rush cover than treatment fields and thus both control and treatment fields constituted a representative sample of the rush cover in fields with and without AES rush management. Rush cover in our survey fields was subsequently assessed during fieldwork and varied across fields from 10% to 70% (three control fields contained less than 30% rush cover). In the SWP, there were 12 treatment and 13 control fields (one treatment field that had initially been selected was excluded as insufficient rush cutting had been conducted) and at Geltsdale, there were 9 treatment and 9 control fields, giving a total sample size of 21 treatment and 22 control fields.

4.3.2 Wader surveys

We estimated the number of breeding wader pairs using a modified version of the standard field-by-field survey method of O’Brien & Smith (1992). Four visits were made to each survey field - two visits in the early breeding season (SWP: 16th April – 28th April; Geltsdale: 5th May – 18th May) and two visits in the late breeding season (SWP: 28th May – 18th June; Geltsdale: 21st June – 25th June). Successive visits within the early or late breeding season were on average seven days apart in the early breeding season and six days apart in the late breeding season. All visits were conducted by one researcher to ensure consistency of survey estimates and thus both regions could not be surveyed concurrently. As Geltsdale is at a higher latitude than the SWP (Fig. 4.1), the wader breeding season commences slightly later in the former region. Survey fields in the SWP were thus visited first in both the early and late breeding seasons. Moreover, surveys were not undertaken during the first hour after sunrise or last hour before sunset, or in heavy rain, fog (< 250 m visibility) or wind greater than Beaufort Force 5.

Within each field, observations were made along a survey route that started 50 m from the field edge and took the observer to within 50 m of every part of the field. All individual waders were marked on a field map with symbols to note behaviour. Surveys recorded Lapwing, Curlew, Snipe, and Common Redshank *Tringa totanus*, but the latter was only detected in three fields at Geltsdale (two control and one treatment) and is not considered further.

An index of the number of breeding pairs of each species per field was calculated using standard species-specific criteria. For all species, groups of more than four individuals were excluded as these may represent non-breeding flocks (following Sim et al., 2005; Douglas et al., in press). For Lapwing, we divided the maximum number of individuals across the two early breeding season visits by two (detectability of Lapwing is high and this approach follows O'Brien and Smith, 1992; Bolton et al., 2011; O'Brien and Wilson, 2011; Smart et al., 2014). For Curlew and Snipe, two birds together, or a single (detectability of these species is expected to be lower than that of Lapwing), either in a field or associating with the field (displaying or mobbing birds above the field) were treated as a pair (following O'Brien and Smith, 1992; Henderson et al., 2002; Hoodless et al., 2006; Pearce-Higgins and Grant, 2006). The number of pairs were then estimated as the maximum per-visit number across all four visits (following Green, 1985; Smart et al., 2008; O'Brien and Wilson, 2011; Buchanan et al., 2017; Douglas et al., 2017). The restricted date range of visits used for calculating Lapwing pairs, compared to Snipe and Curlew, follows standard protocols (O'Brien and Smith, 1992; Bolton et al., 2011; O'Brien and Wilson, 2011; Smart et al., 2014). Estimates of breeding Snipe pairs from diurnal observations are likely to be robust in regions, such as our survey locations, where they do not occur at very high densities (Hoodless et al., 2006) and our surveys detected displaying (drumming and chipping) snipe as well as flushed individuals.

4.3.3 Environmental variables

Rush cover was estimated once per field to the nearest 10% from multiple vantage points during the early breeding season when more accurate estimates can be obtained due to lower vegetation height (note that whilst rush grows tall, it typically spreads relatively slowly in horizontal extent (Ashby et al., 2020) and thus any spread in extent of rush cover within a field is negligible during the survey period).

Field size (ha) was measured in ArcMap TM (v10.4.1; Esri, Redlands, CA, USA) using 1:25 000 Ordnance Survey maps (Ordnance Survey, 2019). Straight-line distance (m) from the centroid of each survey field to the nearest block of woodland (defined as areas with >20% tree cover, from Land Cover Map 2015; Rowland et al., 2017) was measured using the 'Near (Analysis)' tool as woodland proximity can influence breeding wader distributions or abundance (Douglas et al., 2014; Wilson et al., 2014).

Soil conditions were measured during one early, and one late, breeding season visit to account for potential seasonal variation. Soil penetration resistance (kgF) and soil moisture content (%) were recorded at three locations within each field (field centre and two randomly selected locations towards opposite ends of the field) and at two separate points (approximately 15 cm apart) at each of these three locations - giving six measurements per field on each of the two visits. Soil penetration resistance was measured, following Green (1988), using a soil penetrometer with a 5 mm diameter metal pressure rod (20 kg Pesola macro-line spring scale and pressure set, NHBS, Devon, England). Soil moisture content (%) was measured using a soil moisture sensor and readout meter (SM150T soil moisture sensor and HH150 readout meter, Delta-T Devices, Cambridge, England). This sensor had a maximum measurement threshold of 85% and when this threshold was exceeded, we used a value of 92.5% (the mid-point between this threshold and 100%). Soil pH was recorded, using a direct soil pH meter (HI-12922 HALO wireless soil pH electrode, Hanna Instruments, Woonsocket, Rhode Island, USA), at one of the points at each of the three locations - giving three measurements per field on each visit and six measurements per field overall.

Mean soil penetration resistance, soil moisture, and soil pH values were calculated per field for the early breeding season visit (for use in models of the number of Lapwing pairs as these are estimated only using data from the early breeding season), and across the overall breeding season (for use in Curlew and Snipe models as these use data from all site visits). We note, however, that early breeding season and overall breeding season soil conditions were highly correlated (soil penetration resistance: $r = 0.958$; soil moisture: $r = 0.931$; soil pH: $r = 0.919$; $P < 0.001$ and $n = 43$ in all cases).

4.3.4 Statistical analyses

All analyses were conducted in R version 3.6.3 (R Core Team, 2020).

4.3.4.1 Environmental conditions in treatment and control fields

We tested whether treatment and control fields had similar environmental conditions. We fitted generalised linear models (GLMs) with a Gaussian error structure and identity link that modelled each environmental variable (rush cover, soil penetration resistance, soil pH, soil moisture, woodland distance, and field size (natural logarithm transformed prior to inclusion in the models to remove the influence of outliers due to its skewed distribution)) as a function of treatment (treatment or control field) whilst accounting for region (SWP or Geltsdale).

4.3.4.2 Wader responses to rush management

We modelled the density of breeding waders for each species by constructing GLMs with a response variable of the number of pairs per field with a Poisson error structure (log link) and field size (ha; natural logarithm transformed) as an offset in all models. This offset variable converts wader pairs into densities and ensures that field size is accounted for within the models. McFadden's pseudo-R^2 was calculated to represent model fit.

We first ran preliminary checks for simple non-linear effects of our environmental variables (rush cover (%), soil penetration resistance (kgF), soil moisture (%), soil pH, and woodland distance (km)) by modelling each species' density as a function of the selected environmental variable linear term (linear models), and linear and quadratic terms (quadratic models), whilst including region as a fixed factor and field size (ha; natural logarithm transformed) as an offset. There was no strong evidence for non-linear associations, defined as Akaike information criterion value corrected for small sample sizes (AICc) being two points or more lower than that of a linear model (Table D.3) and all subsequent modelling thus used only linear terms.

Following these preliminary checks, we followed Whittingham et al. (2006) and constructed a full model of the main effects to test the prediction that rush management increased wader density, i.e. that there were significantly higher densities in treatment than control fields, whilst accounting for other environmental variables (Table D.4). For each species, we modelled estimated breeding pairs as a function of treatment (treatment or control field as a fixed factor), region (SWP or Geltsdale as a fixed factor), rush cover, soil moisture, soil pH, soil penetration resistance, and woodland distance, with field size (ha; natural logarithm transformed) as an offset. Early breeding season soil conditions were included in the Lapwing models, and overall breeding season soil conditions were included in the Curlew and Snipe models to match the survey dates that were used to estimate the number of pairs of these species (see section 4.3.2).

In addition to the main effects full model, we constructed two extra models that also included i) the interaction between rush cover and treatment/control field (to test if the effects of rush management varied across different amounts of rush cover), or ii) the interaction between region and treatment/control field (to test if rush management effects differed between regions; which could be the case if the factors regulating population size or the capacity of populations to respond to management vary regionally).

For each species, we compared the three full model types (main effects only, main effects plus treatment and rush cover interaction, and main effects plus treatment and region interaction) using each model's AICc, and when interaction terms were present their statistical significance (using a $P < 0.05$ threshold; Table D.5). For Curlew and Lapwing, the main effects only models had the lowest AICc values and interaction terms were not significant; inference is thus based only on the main effects model as there is no evidence that the effects of treatment varied with region or rush cover. For Snipe, the model with the lowest AICc value was that with the treatment and region interaction (ΔAICc relative to model with the next lowest AICc (main effects only model) = 3.605, interaction term $P = 0.009$). The interaction term's parameter estimate did, however, have a very large standard error (SE = 3621.325) demonstrating uncertainty in its effect size and we thus also report the results from a full model that only contains the main effects (Table 4.1).

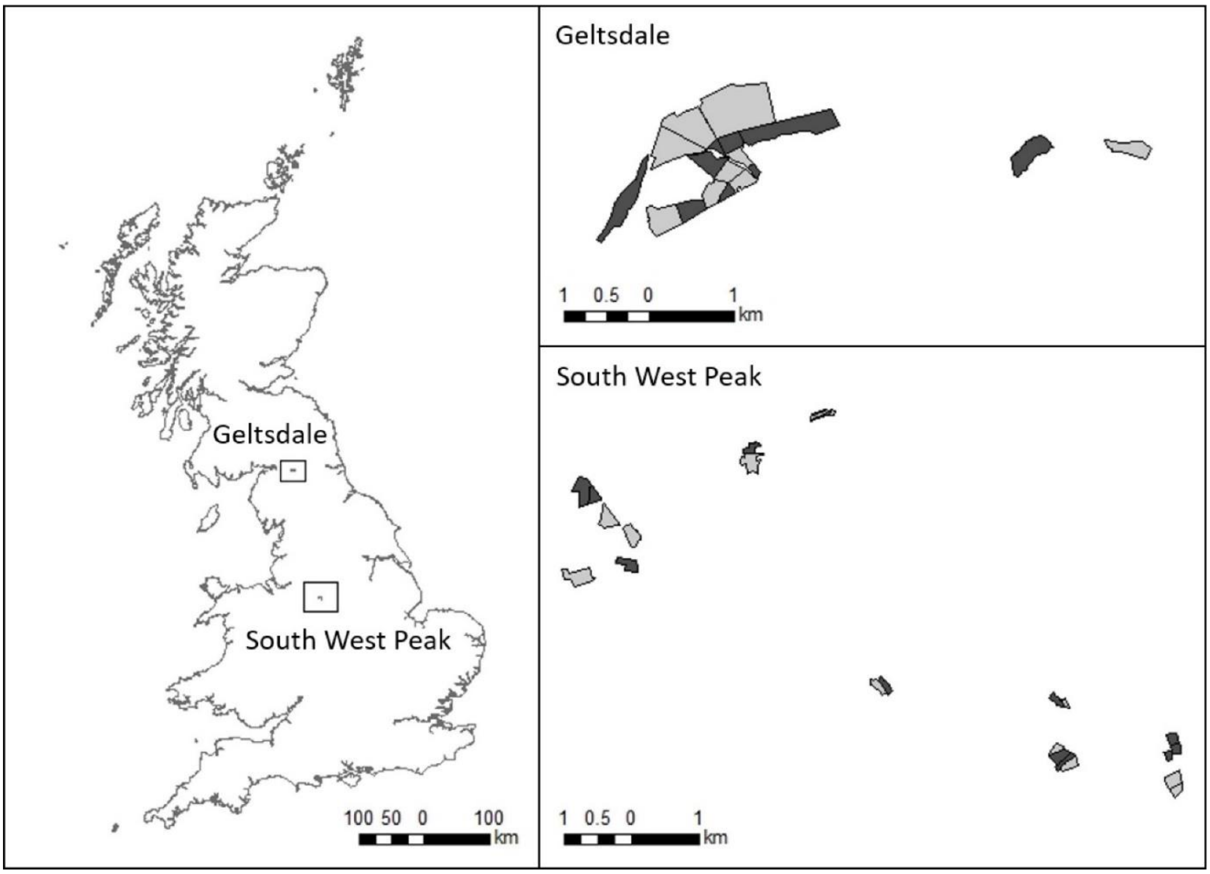

Figure 4.1. The two study regions showing locations of treatment fields (rush management, light grey) and control fields (no rush management, dark grey), with blocks of woodland in the surrounding landscape (white; defined as areas with >20% tree cover, from Land Cover Map 2015; Rowland et al., 2017).

4.4 Results

4.4.1 Environmental conditions in treatment and control fields

Environmental conditions (rush cover, soil penetration resistance, soil moisture, woodland distance, and field size) did not differ significantly between treatment and control fields, except for soil pH (Tables D.2, D.6). In both the early and overall breeding season metrics, treatment fields had slightly more alkaline soil (approximately half a pH unit difference).

4.4.2 Effects of rush management on breeding wader pair densities

Models that took region, rush cover, woodland distance, and soil conditions into account found no evidence that the density of Curlew pairs varied between treatment and control fields (Fig. 4.2a, b; Table 4.1; profile 95% confidence interval (CI) for treatment parameter estimate = -0.73 to 1.12). Rush cover, which varied from 10% to 70% (Table D.4), was not associated with breeding Curlew densities (Table 4.1). Similarly, there was no evidence that Lapwing pair densities differed between control and treatment fields or were influenced by rush cover (Fig. 4.2c, d; Table 4.1; profile 95% CI for treatment parameter estimate = -0.76 to 1.91) – although it is important to note that Lapwings were extremely rare in the SWP survey fields, being observed in just a single control field (Table D.7). For Snipe, when treatment was modelled as an interaction with region, there were higher Snipe densities in treatment fields than control fields in the SWP but similar densities in the two field types at Geltsdale, and no evidence that rush cover was associated with Snipe densities (Fig. 4.2g; Table 4.1). When the interaction between treatment and region was excluded from the model there was no evidence that Snipe pair densities differed between treatment and control fields or were influenced by rush cover (Fig. 4.2e, f; Table 4.1; profile 95% CI for treatment parameter estimate = -0.64 to 2.09).

4.4.3 Effects of woodland distance, soil conditions, and region on breeding wader pair densities

There were trends, albeit only marginally significant ones, for higher densities of Curlew and Snipe in fields with more alkaline soil conditions (Table 4.1). Snipe densities were also higher in fields with wetter soils and in the SWP than Geltsdale (Table 4.1; Fig. 4.2; Fig D.1; Fig. D.2). No other environmental variables influenced breeding wader pair densities (Table 4.1).

Table 4.1. Generalised linear models of breeding wader pair density for Curlew, Lapwing, and Snipe – treatment, region, rush cover, soil conditions (moisture, pH, and penetration resistance), and woodland distance were included as predictor variables, with field size (natural logarithm transformed) included as an offset. Following preliminary tests, Snipe densities were modelled with and without the interaction term between treatment and region; the densities of other species were modelled with just the main effects. Parameter estimates (β) and profile 95% confidence intervals (CIs; in brackets) are presented, with significant effects highlighted with an asterisk. CIs cannot be generated for the Snipe model with the interaction term and thus standard errors are presented for this model. Geltsdale is the reference level for region. Control fields are the reference level for treatment. McFadden's pseudo-R^2 are presented for each model.

Wader species	**Environmental variable**								**McFadden's pseudo-R^2**
	Treatment	Region	Rush cover	Soil moisture	Soil pH	Soil penetration resistance	Woodland distance	Treatment x region	
Curlew	β = 0.182 (-0.725 to 1.124) χ^2 = 0.152 *P* = 0.670	β = 0.181 (-1.259 to 1.458) χ^2 = 0.069 *P* = 0.793	β = -0.014 (-0.044 to 0.018) χ^2 = 0.753 *P* = 0.386	β = 0.025 (-0.011 to 0.061) χ^2 = 1.913 *P* = 0.167	β = 0.592 (-0.075 to 1.243) χ^2 = 3.049 *P* = 0.081	β = 0.198 (-0.274 to 0.687) χ^2 = 0.672 *P* = 0.412	β = 1.385 (-1.056 to 3.910) χ^2 = 1.226 *P* = 0.268	-	0.229
Lapwing	β = 0.509 (-0.757 to 1.911) χ^2 = 0.595 *P* = 0.441	β = -1.504 (-4.571 to 0.464) χ^2 = 2.100 *P* = 0.147	β = -0.035 (-0.085 to 0.009) χ^2 = 2.444 *P* = 0.118	β = -0.007 (-0.049 to 0.038) χ^2 = 0.101 *P* = 0.750	β = 0.384 (-0.324 to 1.073) χ^2 = 1.165 *P* = 0.281	β = 0.069 (-0.499 to 0.654) χ^2 = 0.057 *P* = 0.811	β = -1.567 (-4.566 to 1.346) χ^2 = 1.122 *P* = 0.290	-	0.437
Snipe	β = 0.632 (-0.636 to 2.093) χ^2 = 0.902 *P* = 0.342	β = 1.837 (0.023 to 3.420) χ^2 = 3.933 *P* = 0.047 *	β = -0.026 (-0.066 to 0.015) χ^2 = 1.585 *P* = 0.208	β = 0.048 (0.001 to 0.100) χ^2 = 4.042 *P* = 0.044 *	β = 0.699 (-0.113 to 1.509) χ^2 = 2.870 *P* = 0.090	β = 0.104 (-0.544 to 0.749) χ^2 = 0.104 *P* = 0.747	β = -2.537 (-5.675 to 0.885) χ^2 = 2.147 *P* = 0.143	-	0.168
	β = -0.377 (± 0.734) χ^2 = 0.902 *P* = 0.342	β = -16.871 (± 3621.325) χ^2 = 3.933 *P* = 0.047 *	β = -0.032 (± 0.022) χ^2 = 2.009 *P* = 0.156	β = 0.070 (± 0.030) χ^2 = 5.948 *P* = 0.015 *	β = 0.804 (± 0.433) χ^2 = 3.218 *P* = 0.073	β = 0.132 (± 0.373) χ^2 = 0.126 *P* = 0.723	β = -2.702 (± 1.651) χ^2 = 2.400 *P* = 0.121	β = 19.636 (± 3621.325) χ^2 = 6.824 *P* = 0.009 *	0.273

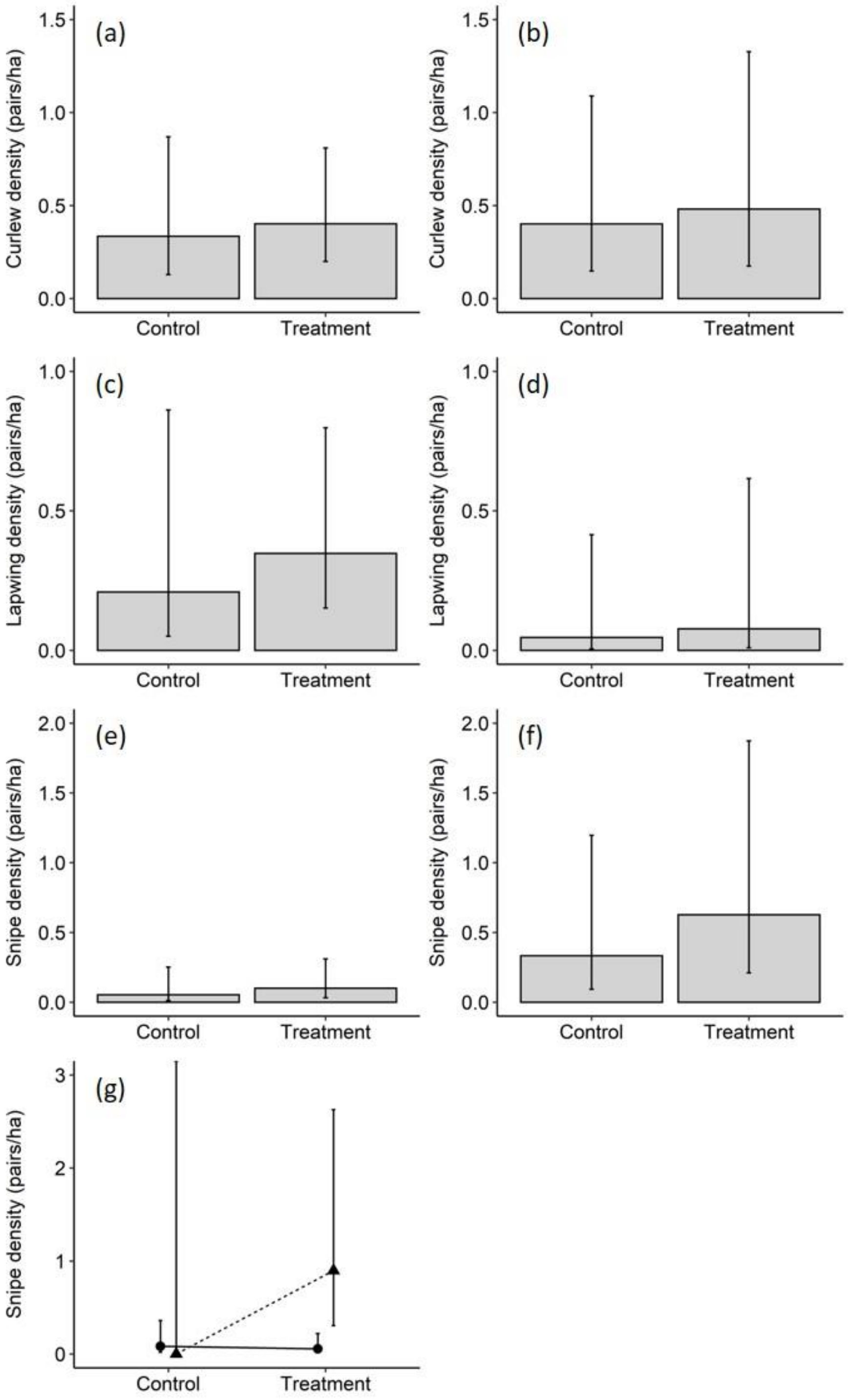

Figure 4.2. Poisson model (main effects only models) predicted breeding wader pair densities in control fields and treatment fields within Geltsdale (left hand column) and the SWP (right hand column) when taking into account rush cover, woodland distance, and soil conditions (moisture, pH and penetration resistance) for Curlew (a and b), Lapwing (c and d), and Snipe (e and f). Bars represent model predicted densities, and errors represent model predicted 95% confidence intervals. The best fitting model (judged by AICc values) for Snipe densities included an interaction between region and treatment with model predicted densities (g) being represented by triangles (SWP) and circles (Geltsdale); error bars again represent 95% confidence intervals but note that for Snipe densities in SWP control fields these are infinite due to singularity issues with the model as no Snipe were observed in such fields.

4.5 Discussion

Our results reveal potential regional variation in the short-term response of Snipe breeding densities to AES rush management prescriptions, with benefits arising from rush management in the SWP but not Geltsdale. Certainty around the strength of this effect is, however, limited by the large standard error around the parameter estimate's interaction term. When regional variation is omitted, there is no firm evidence for benefits of rush management. Breeding Lapwing densities also did not appear to be significantly influenced in the short-term by rush management. Yet, positive impacts on breeding densities cannot be excluded for either Snipe or Lapwing as the 95% CIs for the treatment parameter estimate suggest that the largest plausible values are approximately two (Snipe = 2.09; Lapwing = 1.91). For Curlew, we found negligible evidence for positive effects of rush management on breeding densities in the short-term (no significant effect; 95% CIs indicate that the largest plausible treatment parameter estimate is 1.12).

Whilst our findings are not indicative of strong, and regionally uniform increases in Lapwing and Snipe breeding densities arising from rush management, they do suggest that these species are more likely to respond positively than Curlew, especially in the case of Snipe. This is perhaps logical given i) the preference of nesting Lapwing for short, open vegetation (Baines, 1988; Milsom et al., 2000; Durant et al., 2008) that is generated by rush cutting (Robson and Allcorn, 2006; Kelly et al., 2021), and ii) smaller- and medium-bodied species (Snipe and Lapwing respectively) may be particularly negatively impacted by taller and denser swards that will obscure their view and thus ability to detect predators (limiting their willingness to forage and nest in such habitats) to a greater extent than larger species such as Curlew (Devereux et al., 2004; Whittingham and Evans, 2004; Wilson et al., 2005b).

There is some evidence for regional variation in Snipe responses to AES rush management, with Snipe densities being higher in treatment than control fields in the SWP but not Geltsdale. Such situations are expected to arise if there is regional variation in the extent to which habitat availability regulates Snipe populations. Snipe densities were significantly higher in the SWP than Geltsdale. This situation could arise if most of the habitat with structurally suitable vegetation is occupied in the SWP, whilst other regulating factors limit the Geltsdale population and prevent it from occupying all suitable habitat, including that created through AES rush management. Indeed, the general lack of strong evidence for beneficial impacts of rush management could highlight that habitat improvements will not enable breeding densities to increase because other factors are regulating population sizes (Smart et al., 2013). This links to the buffer effect, through which there is higher likelihood that high quality habitat remains unoccupied (Kluyver and Tinbergen, 1954; Brown, 1969; Gunnarsson et al., 2005). Thus, our results will be most applicable to wader populations at similar or lower densities to those at our study sites and we cannot exclude the possibility that rush management impacts would be greater in populations whose size is regulated by availability of fields with suitable vegetation structure. Given that increased nest and chick predation rates are a key driver of wader declines (Roodbergen et al., 2012; Roos et al., 2018), management may be required that simultaneously tackles rush encroachment and predation pressure to enable wader populations to recover and respond positively to AES rush management – especially for Curlew which exhibited negligible evidence for increased densities in response to rush management.

Alternatively, rush management may not be creating sufficiently optimal conditions for some of our focal wader species to generate consistent and detectable increases in breeding densities. Some AES prescriptions aim to reduce rush cover within a field to <30% (for example, Natural England (2018)), yet all treatment fields had >30% rush cover due to the study design. Nevertheless, we found no evidence that rush cover (which ranged between 30% and 70% in all survey fields barring three control fields with 10-30% rush cover) influenced wader densities. As our study spanned a single breeding season, results are most applicable to the influence of rush management on breeding waders through its short-term impact on vegetation structure (i.e. vegetation height and density). Given that Lapwing favour short grass swards with sparse tussocks comprising rush and grass (Baines, 1988; Milsom et al., 2000), rush management may need to ensure a large proportion of short vegetation is retained throughout the breeding season to generate substantial increases in breeding Lapwing densities, which in the long-term could be achieved by reducing rush cover to lower than 30%. Curlew require a heterogenous sward structure for breeding (Pearce-Higgins and Grant, 2006; Durant et al., 2008), with denser areas to provide chicks with concealment from predators (Valkama et al., 1998), more open areas for foraging (Robson and Allcorn, 2006; Fisher and Walker, 2015), and a range of vegetation heights for nesting (Valkama et al., 1998; Fisher and Walker, 2015; Zielonka et al., 2019). It is plausible that current AES rush management prescriptions are not delivering sufficient within-field heterogeneity in sward structure to provide the complex habitat matrix required by Curlew. Such a situation could arise either because the current prescriptions to cut one-third of the rush within a field on an annual basis are insufficient, or because such prescriptions are too difficult for farmers to follow as they feel that they should cut a larger proportion of the field when they are able to access fields for rush cutting (this is often difficult in winter due to waterlogged conditions). It is also important to note that rush cutting through AES prescriptions has been found to increase the risk of artificial wader nest predation (Kelly et al., 2021) and thus birds may be avoiding nesting in such fields due to perceived, and realised, increases in nest predation risk.

4.5.1 Implications for management of wader breeding habitat

Accounting for variation in field size via inclusion as an offset in the models, our results suggest that rush management through AES prescriptions can increase Snipe breeding densities in some but not all regions, and such benefits could also arise for Lapwing (although low population sizes, especially in the SWP, limited our ability to detect such effects). In contrast, we found evidence that Curlew are, out of our focal species, the least likely to respond to the implementation of AES rush management prescriptions. Whilst ideally surveys would be repeated in subsequent years, this was not possible due to logistical constraints. Our study of two distinct upland regions, however, enables testing of rush management across different environmental conditions and population densities of our focal species (see Fig. 4.2). Moreover, our results provide a snapshot of wader densities in fields with and without rush management, with results revealing the potential for rush management to increase densities of Snipe and Lapwing in the short-term. Our study thus advocates further research exploring both the short- and long-term impacts of AES rush management prescriptions on upland breeding waders. We also found evidence that more alkaline soils were associated with higher Curlew and Snipe breeding densities, which for Curlew is consistent with previous research showing lower densities where soil organic carbon (assumed to be more acidic, peaty soils) is higher (Franks et al., 2017). These patterns are presumably due to higher pH increasing abundance of soil invertebrates such as earthworms (McCallum et al., 2016), and wetter soils also increased Snipe densities. Combining rush management with additional interventions to improve habitat quality may thus be beneficial, such as installation of wetter depressions or flushes and blocking of drainage ditches (Smart et al., 2006; Douglas and Pearce-Higgins, 2014; Douglas et al., in press), or liming (but with targeted use; McCallum et al., 2016). Rush management prescriptions may, however, benefit from potential revision to increase their efficacy. Potentially beneficial changes that merit further investigation include researching the optimal total area and spatial configuration of cut and uncut rush within fields, thus ensuring heterogeneity in sward structure, perhaps particularly for Curlew (as shown by beneficial mosaic grassland management for Black-tailed Godwits *Limosa limosa* in The Netherlands; Schekkerman et al., 2008), and in the case of Lapwing contrastingly ensuring rush cover is below 30% (yet retaining some taller vegetation patches; Laidlaw et al., 2017) which will limit heterogeneity in the sward. In addition, where additional drivers to habitat degradation, such as increased nest predation risk, are suppressing wader populations, interventions that focus solely on habitat improvements are unlikely to fully meet their potential to reverse population declines (Smart et al., 2013).

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Chapter 5

Upland rush management within agri-environment schemes: farmers' motivations and constraints to participation, and suggestions to increase efficacy and uptake

A ewe and two lambs in an in-bye field with cut rush

5.1 Abstract

Agri-environment schemes (AES) are a key strategy for combatting biodiversity loss in Europe. Potential factors contributing to the current limited effectiveness of AES include insufficient uptake by farmers and consequent lack of landscape scale implementation. AES management of rush *Juncus spp.* encroachment is widely considered essential for improving the conservation status of breeding waders in upland regions of the UK, some species of which are on the global IUCN Red List, but populations are continuing to decline. Little attention has been given to a socio-economic assessment of factors influencing uptake of AES prescriptions for rush management. We use semi-structured interviews with farmers in the south-west of the Peak District National Park, England, and a mixed methods approach to explore these issues. All farmers confirmed that rush was present on their farm, with 70.3% reporting an increase in the quantity of rush over the last 10 years. We then used quantitative analyses to test the predictions that AES participation will be associated with farmers' pro-environmental attitudes, environmental knowledge, perceived social norms, and perceived effectiveness of AES prescriptions for rush management. We, however, only found evidence for one of these hypotheses – perceived effectiveness of AES prescriptions for rush management. Farmers' perceptions of rush management prescriptions at improving land for farming and wading birds were strongly co-associated; although there was marginal evidence that non-AES participating farmers were more likely to perceive rush management prescriptions to be effective for waders than those who did participate. Qualitative analyses highlight that the core motivations for participation were financial benefits, pro-environmental attitudes, and limited inconvenience as prescriptions fitted in with farmers' typical activities and lifestyle. Conversely, the strongest constraints to participation were insufficient financial incentive, scheme inflexibility, and unawareness of rush management prescriptions. Farmers' suggestions for changes that would increase the efficacy of AES rush management prescriptions for farming and wading birds were similar, and primarily focused on expanding treatment methods to include herbicide application via weed wiping and more flexible rush cutting requirements. Such changes, and increasing awareness of rush management prescriptions, are likely to be required to increase the uptake of rush management prescriptions. Achieving increased flexibility whilst also ensuring that prescriptions are sufficiently targeted, and detailed, to deliver the required habitat improvements is likely to be a challenge without increasing access to specialised advisors on the ground.

5.2 Introduction

Loss of biodiversity in Europe is largely attributable to agricultural intensification and, in some regions abandonment of farmland (European Commission, 2020). Agri-environment schemes (AES) which subsidise farmers to implement environmentally beneficial land management practices are a key strategy for combatting this loss and achieving biodiversity targets (Batáry *et al*., 2015). Evidence suggests, however, that success of AES has been variable across taxa and participating countries (Kleijn *et al*., 2001; Kleijn and Sutherland, 2003; Kleijn *et al*., 2006), and AES have thus far failed to deliver effective biodiversity conservation across large spatial extents (O'Brien and Wilson, 2011; Princé *et al*., 2012; Franks *et al*., 2018). Potential factors contributing to this limited effectiveness include insufficient uptake and consequent lack of landscape scale implementation (Whittingham, 2007; Dallimer *et al*., 2010; Perkins *et al*., 2011; Baker *et al*., 2012; Burgess *et al*., 2015; Franks *et al*., 2018). Skylark plots (current AES in England: Countryside Stewardship, option AB4) provide an example of an AES prescription that, despite scientific robustness, has received too limited uptake to deliver the intended ecological benefits, with weed invasion within the plots and mechanical difficulties regarding the creation of plots reported to limit uptake by farmers (Evans and Green, 2007; Dillon *et al*., 2009).

Increasing the uptake of AES is critically important for improving the scale of impact and effectiveness. Thus, identifying and understanding the factors which influence participation is key. The link between AES and farmer engagement is influenced by a range of potential factors including (but not restricted to) AES design, social norms, and parameters relating to the farm and the farmer. Careful design of AES is essential for farmer engagement as the magnitude of financial incentives and the degree of scheme complexity can affect the likelihood of participation (Morris *et al*., 2000; Defrancesco *et al*., 2008; Ruto and Garrod, 2009; Karali *et al*., 2014; Schroeder *et al*., 2015). Indeed, these two design attributes can be intertwined – more complex paperwork, set up and management require greater financial compensation (Ruto and Garrod, 2009; Karali *et al*., 2014). The perceived complexity and thus applicability of individual schemes will, however, vary between farmers. Parameters such as farm size, the intensity of current farming practices and labour requirements, and the impact that prescriptions may have on farm productivity can all influence participation (Vanslembrouck *et al*., 2002; Defrancesco *et al*., 2008; Hynes and Garvey, 2009; Karali *et al*., 2014). Moreover, individuals' intrinsic characteristics can also influence farmer engagement with AES such as their farming and environmental attitudes (Vanslembrouck *et al*., 2002; Defrancesco *et al*., 2008; Karali *et al*., 2014; Schroeder *et al*., 2015; Cullen *et al*., 2020). Previous experience with AES can have a positive impact on likelihood of participation (Vanslembrouck *et al*., 2002; Defrancesco *et al*., 2008), although this can depend on the specific prescription (Vanslembrouck *et al*., 2002). Other individuals can also intentionally and unintentionally impose their influence on farmer engagement, particularly neighbouring farmers, and family members (Vanslembrouck *et al*., 2002; Defrancesco *et al*., 2008; Schroeder *et al*., 2015). In the case of neighbouring farmers, the extent that these social norms influence farmer engagement can be related to the relationship with and opinions of neighbouring farmers as well as the participation of neighbouring farmers in AES (Vanslembrouck *et al*., 2002; Defrancesco *et al*., 2008; Cullen *et al*., 2020). Despite this range of factors that may influence farmers' engagement with AES, their relative importance, especially for specific AES prescriptions, is poorly understood. This is an important knowledge gap, especially as the factors that influence participation will vary across specific AES programmes and their prescriptions (Lastra-Bravo *et al*., 2015; Pavlis *et al*., 2016).

An example of a management practice available within AES which has received little attention is the management of *Juncus spp.* (hereafter rush). The extent of rush cover, which is native to the UK, has increased in recent decades on upland grasslands (Ashby *et al*., 2020). This reduces farm productivity by lowering the amount and quality of forage for the dominant livestock types, i.e. sheep and cattle (Grant *et al*., 1984; Nielsen and Søegaard, 2000; Tweel and Bohlen, 2008; Cairns, 2013; Hamilton *et al*., 2018). Rush encroachment also negatively impacts breeding waders, which are increasingly concentrated in upland regions and have declined throughout the UK and much of Europe (Hayhow *et al*., 2017; PECBMS, 2020). Some species are now globally Near Threatened, e.g. Eurasian curlew *Numenius arquata*, northern lapwing *Vanellus vanellus*, and Eurasian oystercatcher *Haematopus ostralegus* (IUCN, 2021). Expanses of tall, rank vegetation such as rush can adversely affect foraging and breeding success of waders which typically require a mosaic of shorter and taller vegetation (Vickery *et al*., 2001; Whittingham and Evans, 2004; Pearce-Higgins and Grant, 2006; Durant *et al*., 2008; Kelly *et al*., 2021).

Rush management through AES can thus benefit farmers by improving farm productivity and deliver conservation gains by improving wader habitat (Robson and Allcorn, 2006; Holton and Allcorn, 2006; Fisher and Walker, 2015). AES rush management prescriptions aimed at breeding waders are available throughout the UK and are split into broad and narrower, more targeted prescriptions (Natural England, 2012a; Natural England, 2012b; Welsh Government, 2013; Welsh Government, 2017). Broad prescriptions typically require one-third of the rush in a field to be cut annually in rotation and advise grazing with cattle (Natural England, 2012a; Welsh Government, 2013), usually aiming to reduce rush cover within a field to less than 30% (Natural England, 2018). Targeted prescriptions are more variable, with the ability to tailor the prescription to specific wader species (Natural England, 2012b; Welsh Government, 2017).

Intensive rush management could considerably reduce rush cover in fields and thus improve farm productivity to a greater extent than that permitted through AES, and many upland farmers do not manage rush through AES schemes. There is currently, however, very limited understanding of the motivations and constraints that determine farmers' decisions regarding participation in AES rush management prescriptions. In addition to the general factors influencing AES engagement, farmers' perceptions regarding the effectiveness of AES rush management could influence their participation. Rush management can increase numbers of breeding waders (Holton and Allcorn, 2006; Robson and Allcorn, 2006) but could also inadvertently increase nest predation risk for waders which nest across a gradient of vegetation structures, such as Eurasian curlew and common snipe *Gallinago gallinago* (Kelly *et al.*, 2021).

Here, we used semi-structured questionnaire-style interviews with farmers from an upland region of England and a mixed-methods approach to assess which factors influence farmers' decisions to participate in AES prescriptions for rush management. In doing so, we explore four broad hypotheses which link participation to farmers' pro-environmental attitudes (bird conservation), environmental (wader habitat) knowledge, perceived social norms, and perceived effectiveness of AES prescriptions for rush management. With a view to future changes that increase participation, we also assess specific motivations and barriers to participation, and use the principle of co-design to elicit farmers' views on potential changes to a current management prescription within AES.

5.3 Methods

5.3.1 Study area

Research involved farmers from the south-west of the Peak District National Park (hereafter the South West Peak, "SWP"), England. The SWP is typical of UK upland farmed landscapes, comprising a mosaic of grassland and moorland, and supporting breeding waders including Eurasian curlew, common snipe, northern lapwing, and Eurasian golden plover *Pluvialis apricaria*. The AES options available here are the same as those available in all other upland regions of England and similar to schemes elsewhere in the UK.

5.3.2 Respondent recruitment

Ethical approval was obtained from the University of Sheffield's Research Ethics Committee (application number 032479), and informed consent was obtained from all respondents (see Appendix E.1).

Potential respondents were identified from a list of 58 farmers, provided by the South West Peak Landscape Partnership, whose land was considered likely to be suitable for rushes. The list was broadly representative of the types of farmers within the study region and included farmers' details regardless of their interest in wildlife conservation or engagement with AES. It did, however, record farmers' participation in rush management through AES for many farmers (28 AES participants; 15 non-AES participants; 15 unknown status). Initial contact with potential respondents was conducted via telephone or in person. We contacted potential respondents at 55 of these 58 farms and asked if they wished to take part in a study about rush management and AES to see how rush management within AES could be improved using the knowledge and understanding of SWP farmers (the 55 farms consisted of 26 AES participants, 15 non-AES participants, and 14 respondents with originally unknown AES participation status).

We completed interviews with 37 farmers (response rates were 73% of AES participants; 73% of non-AES participants; and 50% of respondents with originally unknown AES participation). Final sample sizes were fairly evenly balanced between farmers currently (n = 20) and not currently (n = 14) participating in AES prescriptions to manage rush. Three farmers did not manage rush on their farm at all. These 37 interviews comprised 16 face-to-face interviews conducted in early March 2020, with the remainder conducted remotely via telephone from mid-March to early September 2020 in accordance with COVID-19 regulations. Each interview phase comprised a mix of AES participants and non-AES participants. Most interviews (32 out of 37) were held with a single respondent, the principal farmer for the landholding. However, some interviews (5 out of 37) were held with multiple respondents from the same farm – the principal farmer and one or two relatives who also played a role in making farming decisions.

5.3.3 Data collection

Interviews were conducted using a semi-structured approach supplemented with questionnaire-style questions, with the interviewer filling in the forms based on verbal responses from the respondents. Interviews collated data on rush management practices, their perceived efficacy and potential for improvement, AES participation, environmental attitudes and knowledge, and social norms. The questions were asked in the order of the questionnaire (see Appendix E.2) unless a respondent unintentionally answered a forthcoming question in response to an earlier question in which case the order varied to enable a free-flowing conversation. Interviews were audio-recorded using a Dictaphone (Sony ICD-UX560 Digital Dictation Machine). All interview audio-recordings were transcribed using Trint transcription software (Trint Ltd., London, England) and checked for errors.

5.3.3.1 Description of the study system

Information regarding rush cover, its impact on the farm and environment, and its management provided an insight into the study system. Respondents were asked *“Do you have rush on your farm?”* (potential responses: ‘yes’, and ‘no’) and *“Has the amount of rush on your farm changed over the last 10 years?”* (potential responses: 1 – ‘considerably decreased, 2 – ‘moderately decreased’, 3 – ‘slightly decreased', 4 – 'no change, 5 – 'slightly increased', 6 – 'moderately increased', and 7 – 'considerably increased'). Those who reported a change in rush cover were asked for the overall impact on the quality of their land from a) a farming perspective, and b) an environmental perspective (potential responses: 1 – 'considerably detrimental, 2 – 'moderately detrimental', 3 – 'slightly detrimental', 4 – 'no impact, 5 – 'slightly beneficial', 6 – 'moderately beneficial', and 7 – 'considerably beneficial'). All respondents were then asked, *"Have you managed rush on your farm in the last 5 years?"* (potential responses: 'yes', and 'no'). Respondents who had managed rush in the last 5 years were asked to describe how they typically managed rush within this time period regarding cutting rush (including what they did with the rush that had been cut), aftermath grazing, herbicide application, and the proportion of rush managed within a field.

5.3.3.2 Farmers' ecological understanding of the system

Respondents' wader identification skills and knowledge of wader population trends were assessed to qualitatively describe farmers' ecological understanding of the system. The first was a measure of respondents' ability to identify four wading bird species that breed in the SWP (northern lapwing, Eurasian curlew, common snipe, and Eurasian golden plover) from a photograph of each species (see Appendix E.2 for photographs). Colloquial names were accepted; for example, 'peewit' was accepted for northern lapwing (Table E.1). Respondents were scored one point per correctly identified species and this scale thus varied from zero to four. Respondents' knowledge of wader population trends was assessed by asking *"How have breeding wading bird populations changed in the South West Peak over the last 20 years?"* scored on a seven-point Likert scale (1 – 'considerably decreased', 2 – 'moderately decreased', 3 – 'slightly decreased', 4 – 'no change', 5 – 'slightly increased', 6 – 'moderately increased', and 7 – 'considerably increased').

5.3.3.3 Agri-environment scheme participation

Participation in rush management AES prescriptions could be influenced by a plethora of reasons. Respondents who had managed rush in the last five years (n = 34) were asked "*In the last five years, were any of your fields that received rush management managed as part of an agri-environment scheme*" (potential responses: 1 - 'always managed as part of an agri-environment scheme', 2 - 'started managing outside an agri-environment scheme but now managed as part of one', 3 - 'started managing as part of an agri-environment scheme but now managed outside one', and 4 - 'always managed outside an agri-environment scheme'). Based on these responses, farmers were assigned to one of two groups for the quantitative analyses: 'currently managing rush as part of an agri-environment scheme' (responses 1 and 2) and 'currently managing rush outside of an agri-environment scheme' (responses 3 and 4).

For farmers currently managing rush as part of an AES (responses 1 and 2) and those who started managing rush as part of an AES but now were managing outside one (response 3), we requested the AES and the specific prescription most frequently used within this AES that the rush was managed under.

5.3.3.4 Perceived effectiveness of farmers' own rush management approach

As there are multiple options and combinations for managing rush, the 'success' of the respondents' rush management approach may vary. Two statements captured respondents' perceived 'success' by asking to what extent they agreed or disagreed, using a seven-point Likert scale (potential responses: 1 – 'strongly disagree', 2 – 'moderately disagree', 3 – 'slightly disagree', 4 – 'neither agree nor disagree', 5 – 'slightly agree', 6 – 'moderately agree', and 7 – 'strongly agree'), with i) "*My typical approach to rush management has improved the quality of my land from a farming perspective*", and ii) "*My typical approach to rush management has improved the quality of my land from an environmental perspective*".

5.3.3.5 Perceived effectiveness of rush management AES prescriptions

To gauge farmers' perceptions of the efficacy of rush management AES prescriptions, based on either their own direct experience or through discussions with other people, we asked respondents to what extent they agreed or disagreed with two statements: i) "*Rush management agri-environment scheme prescriptions are effective at reducing rush cover*", and ii) "*Rush management agri-environment scheme prescriptions are effective at improving habitat for breeding wading birds*", using a seven-point Likert scale from strongly disagree to strongly agree. Separate indices of perceived effectiveness were generated for these two statements: index of perceived effectiveness at reducing rush cover, and index of perceived effectiveness at improving wading bird habitat. Farmers who reported that they did not know (n = 13 for reducing rush cover; n = 11 for improving wader habitat) were removed from the dataset for models including either variable.

5.3.3.6 Attitudes to bird conservation

To quantify the general attitude of respondents towards bird conservation, respondents were asked to what extent they agreed or disagreed with two statements, using a seven-point Likert scale from strongly disagree to strongly agree. These statements assessed the importance of birdlife on their farm and conserving wading bird species in the wider landscape: i) "*I see value in having large numbers* *and variety of birdlife on my farm*", and "*Conservation of breeding wading bird populations in the South West Peak is important*". Scores for these two statements were positively correlated ($r_s = 0.660$; $P = 9.012^{-6}$; N = 37) and thus summed to generate a single index of bird conservation attitudes.

5.3.3.7 Wading bird habitat knowledge

Two statements concerning farmers' knowledge of wading bird habitat were based on four wading bird species: northern lapwing, Eurasian curlew, common snipe, and Eurasian golden plover, which are the most widespread and abundant waders breeding on farmland in the SWP. To assess respondents' knowledge of the vegetation structure requirements of the wading birds, they were asked to what extent they agreed or disagreed with the statements that i) "*Optimum rush cover differs between breeding wading bird species*", and ii) "*A mosaic of shorter and taller vegetation is beneficial for breeding wading birds*", using a seven-point Likert scale from strongly disagree to strongly agree. Scores for these two statements were positively correlated ($r_s = 0.615$; $P = 5.141^{-5}$; N = 37) and thus summed to generate a single index of wading bird habitat knowledge.

5.3.3.8 Perceived social norms

To assess whether the respondents prescribed to social norms, respondents were given two statements concerning neighbouring farmers' perceptions using a seven-point Likert scale from strongly disagree to strongly agree. We asked to what extent the respondents agreed or disagreed with the statements: i) "*It is important that neighbouring farmers approve of the way I manage my farm"* (0 – 'strongly disagree', 6 – 'strongly agree'), and ii) "*If I managed rush according to agri-environment scheme prescriptions, neighbouring farmers would consider me to be a good farmer*" (1 – 'strongly disagree', 7 – 'strongly agree'). Note that we used scores from 0 to 6 instead of 1 to 7 for the first statement because we calculated the product of the two statements' scores to generate a single index of perceived social norms. A 0 score for the first statement (i.e. strongly disagree that it matters if neighbouring farmers approve) will thus generate an overall score of 0 for the importance of social norms, and high product scores indicate that social norms encourage the farmer to manage according to AES prescriptions.

5.3.3.9 Importance of rush encroachment for wading birds

Respondents were asked to state whether they thought rush encroachment had a positive or a negative effect on breeding wading bird population sizes and to rank the importance of this effect from zero to ten (0 = not important; 10 = extremely important). Based on the direction of effect and importance rank, each respondent was then assigned a score ranging from -10 (strongly disadvantageous) to +10 (strongly beneficial). One respondent who did not provide a score was assigned a score of 0.

5.3.3.10 Index of grazing livestock intensity

Respondents were asked for the number of animals present on their farm within different grazing livestock categories (Table E.2). The number of animals in each category was multiplied by the corresponding livestock units (Natural England, 2017). The sum of these values for each livestock category was divided by the total farm area (hectares) to generate an index of grazing livestock intensity (livestock units per hectare).

5.3.3.11 AES participation motivations

Farmers who were currently managing rush as part of an AES (see section 5.3.3.3) and farmers who started managing rush as part of an AES but now were managing outside one were asked "*What influenced your decision to participate in this specific agri-environment scheme prescription?*". Farmers who were currently managing rush outside of an AES were asked "*Why do you not/no longer manage rush as part of an agri-environment scheme?*" (open-ended questions).

5.3.3.12 Co-designing rush management AES prescriptions

Farmers' wealth of knowledge and experience concerning interventions such as rush management provides an important source of information for improving the efficacy and uptake of rush management AES prescriptions. We asked all respondents to read a summary of the GS16 rush infestation control supplement, the main prescription targeted at rush-dominated fields that was available at the time of the interviews within the UK government's Countryside Stewardship (the current English AES in 2020; see Appendix E.2 for GS16 summary; Natural England, 2018). We then asked respondents two open-ended questions, "*What, if anything, would you change within the current Countryside Stewardship GS16 rush infestation control supplement in order to improve the effectiveness at reducing rush cover and why?*" and "*If you were focusing solely on managing rush to benefit breeding wading birds, what (if anything) would you change within the current Countryside Stewardship GS16 rush infestation control supplement and why?*". Respondents were prompted, when required, for any potential changes regarding cutting rush, aftermath grazing, adding in herbicide application, or the proportion of rush to be managed within a field. For each of the two questions, if respondents suggested changes (answers could be the same or different for the two questions), we asked "*If these proposed changes were implemented in an agri-environment scheme, do you think they would increase or decrease the likelihood of farmers' participation in the scheme?*", using a seven-point Likert scale from considerably decrease to considerably increase (potential responses: 1 – 'considerably decrease', 2 – 'moderately decrease', 3 – 'slightly decrease', 4 – 'no change', 5 – 'slightly increase', 6 – 'moderately increase', and 7 – 'considerably increase').

Rush cutting constitutes a major part of rush management. The effort required by farmers to cut rush will be dependent on the rush cutting pattern, with rush cut in a single, large block likely to be less time-consuming. There is anecdotal evidence, however, that managing rush to create a mosaic of smaller cut and uncut patches may benefit breeding wading birds by reducing perceived and actual predation risk, for example (Fisher and Walker, 2015). Respondents' likelihood to implement this mosaic rush cutting pattern was assessed by asking "*Would you consider changing your rush cutting pattern within a field to multiple, smaller cuts if there was a benefit to breeding wading birds?*" (n = 34), using a seven-point Likert scale from considerably unlikely to considerably likely (potential responses: 1 – 'considerably unlikely', 2 – 'moderately unlikely', 3 – 'slightly unlikely', 4 – 'neither likely nor unlikely', 5 – 'slightly likely', 6 – 'moderately likely', and 7 – 'considerably likely').

5.3.3.13 Socio-demographics

We collected data on respondents' gender, age, highest education qualification, years spent actively farming, and land ownership (Table E.3). Age was recorded in categories (25-34; 35-44; 45-54; 55-64; 65-74; 75-84; 85-94) with the midpoint of each category used in analyses. For highest education qualification, 'O level/GCSE or equivalent' and 'A level or equivalent' were merged into a single category ('school level education'), and 'undergraduate degree' and 'higher degree' were merged into an additional single category ('higher level education').

5.3.4 Data analyses

5.3.4.1 Quantitative statistical analyses

All statistical analyses were conducted in R version 3.6.3 (R Core Team, 2020) and modelled farmers' i) current participation in AES (generalised linear model with binomial error structure and logit link), ii) perceived effectiveness of AES rush management prescriptions at improving wader habitat (cumulative link model (CLM) with logit link, using the ordinal package (Christensen, 2019)), iii) perceptions of whether their typical approach to rush management improved the agricultural quality of their land (CLM with logit link), and iv) the environmental quality of their land (CLM with logit link). For CLMs, we checked for proportional odds assumption violations using the nominal_test and scale_test functions in the ordinal package (Christensen, 2019) and when predictors violated this CLM assumption, we included these variables in the model using the respective nominal or scale argument (Christensen, 2019). In all cases we use McFadden's pseudo R^2 as a measure of explanatory capacity.

Our approach was to conduct modelling in a two-stage process to avoid over-fitting models. In stage one, we assessed which (if any) socio-demographic predictors were associated with each of our four response variables. We did this by constructing a null model and a series of socio-demographic models that consisted of all possible combinations of our five socio-demographic variables (gender (two categories: male, female); age (continuous); highest education qualification (three categories: none, school level education, higher level education); years spent actively farming (continuous); land ownership (three categories: landowner, tenant, landowner and tenant); Table E.2). We retained, for stage two, socio-demographic variables contained in the model with the lowest Akaike information criterion with correction for small sample size (AICc) if this model's AICc was lower than that of the null model; if the null model had the lowest AICc, we did not retain any socio-demographic variables for use in stage 2 models (see Table E.4 for results of stage 1 modelling). In stage two, we constructed a series of models for each response variable that focused on testing specific hypotheses whilst taking into account the socio-demographic variables selected in stage one and grazing livestock intensity (natural logarithm transformed). We first constructed base models that only contained socio-demographic variables and grazing livestock intensity. Each of the initial base models contained either a single socio-demographic variable or grazing livestock intensity. The final base model contained all socio-demographic variables and grazing livestock intensity. To the final base model, we then constructed a series of models, each of which included a single additional predictor which was selected to test a specific hypothesis. We then distinguished between these competing models based on their AICc values, and checked that their AICc values were lower than those of the null model (i.e. model without any predictors) and the base models.

5.3.4.1.1 Farmers' decisions to participate in rush management AES prescriptions

When modelling farmers' current participation in AES (0 = currently managing rush outside of an AES prescription; 1 = currently managing rush as part of an AES prescription) in stage two, we included the following predictors in addition to the base model (grazing livestock intensity and age): i) bird conservation attitudes index (testing the hypothesis that farmers were more likely to participate in AES if they had stronger pro-conservation attitudes), ii) wading bird habitat knowledge index (testing the hypothesis that participation is higher when farmers had greater ecological knowledge), iii) perceived social norms index (testing the hypothesis that farmers were more likely to participate if they were influenced by their neighbours' approval and their neighbours perceived AES positively), iv) perceived AES effectiveness at reducing rush cover, and v) perceived AES effectiveness at improving wading bird habitat (respectively testing hypotheses that farmers' who perceived AES as effective were more likely to participate). All predictors were assumed to be continuous.

5.3.4.1.2 Farmers' perceived effectiveness of rush management AES prescriptions at improving wading bird habitat

When modelling farmers' perceived effectiveness of AES rush management prescriptions at improving wader habitat in stage two, we included the following predictors in addition to the base model (grazing livestock intensity): i) bird conservation attitudes index (testing the hypothesis that farmers were more likely to perceive AES rush management prescriptions to be effective if they had stronger pro-conservation attitudes), ii) wading bird habitat knowledge index (testing the hypothesis that farmers were more likely to perceive AES rush management prescriptions to be effective when farmers' had greater ecological knowledge), iii) perceived AES effectiveness at reducing rush cover (testing the hypothesis that farmers' who perceived AES as effective at reducing rush cover were more likely to perceive AES as effective at improving wading bird habitat), and iv) rush encroachment importance (natural logarithm + 11 transformed; testing the hypothesis that farmers' who perceived rush encroachment to be an important negative driver for wading bird populations were more likely to perceive AES as effective at improving wading bird habitat). All predictors were assumed to be continuous. As the model with rush encroachment importance failed the scale effects test and thus violated the proportional odds assumption of CLMs, this predictor was included in the model using the scale argument (Christensen, 2019).

5.3.4.1.3 Farmers' perceived effectiveness of own rush management approach from a farming perspective

When modelling farmers' perceptions of whether their typical approach to rush management improved the agricultural quality of their land in stage two, we included AES participation (testing the hypothesis that farmers who currently participated in AES were less likely to perceive their rush management approach as improving their agricultural land quality; two categories: currently participating, not currently participating) in addition to the base model (grazing livestock intensity and highest education qualification).

5.3.4.1.4 Farmers' perceived effectiveness of own rush management approach from an environmental perspective

When modelling farmers' perceptions of whether their typical approach to rush management improved the environmental quality of their land in stage two, we included the following predictors in addition to the base model (grazing livestock intensity): i) AES participation (testing the hypothesis that farmers who currently participated in AES were more likely to perceive their rush management approach as improving their environmental land quality) ii) bird conservation attitudes index (testing the hypothesis that farmers were more likely to perceive their rush management approach as improving their environmental land quality if they had stronger pro-conservation attitudes), iii) wading bird habitat knowledge index (testing the hypothesis farmers were more likely to perceive their rush management approach as improving their environmental land quality if they had greater ecological knowledge). AES participation included two categories: currently participating (1), and not currently participating (0). Bird conservation attitudes index and wading bird habitat knowledge index were assumed to be continuous.

5.3.4.2 Qualitative content analyses

5.3.4.2.1 Thematic analysis

Thematic analysis of open-ended questions was performed by manual coding whereby mutually exclusive themes were deciphered from the responses to each question. We initially analysed responses to questions regarding the factors that motivate farmers to manage rush within or outside of an AES prescription: "*What influenced your decision to participate in this specific agri-environment scheme prescription?*" and "*Why do you not/no longer manage rush as part of an agri-environment scheme?*". Then, to see if and how farmers would improve current rush management AES prescriptions, we analysed responses to two further questions: "*What (if anything) would you change within the current Countryside Stewardship GS16 rush infestation control supplement in order to improve the effectiveness at reducing rush cover and why?*" and "*If you were focusing solely on managing rush to benefit breeding wading birds, what (if anything) would you change within the current Countryside Stewardship GS16 rush infestation control supplement and why?".*

Themes were independently identified by two researchers which were then compared and discussed to generate a final set of key themes for each open-ended question. For each question, responses were then independently assigned to the selected themes by the two researchers, and any disagreements discussed until full agreement was reached. When assigning themes, a single response could be aligned with multiple themes and thus the cumulative number of themes identified for each question is greater than the number of respondents. The relative importance of each theme is indicated by the number of responses. Exerts from responses to open-ended questions are included to illustrate themes and provide additional detail.

5.4 Results

5.4.1 Description of sample population and study system

Respondents were typically older (median age 59.5) males (75.7%) with lots of farming experience (median 40 years) who owned their own land (83.8%; although 25.8% also rented land), and a small majority (64.9%) had school or higher level education qualifications (Table S2). Farm size (including owned and rented land) varied from smallholdings to vast moorland and in-bye landholdings (median = 60.70 ha; range = 7.69 to 404.69 ha). All respondents grazed livestock, especially cattle (78.4%) and sheep (75.7%), and 89.2% grew fodder crops (56.8% hay; 40.5% haylage; 43.2% silage). Rush was present on all farms and had increased on 70.3% over the last 10 years (Table 5.1), and perceived to cause detrimental impacts in most cases for farming (88.5% overall: 30.8% considerably detrimental; 30.8% moderately detrimental; 26.9% slightly detrimental) and the environment (53.9% overall: 3.9% considerably detrimental; 19.2% moderately detrimental; 30.8% slightly detrimental). The majority of respondents (91.9%; n = 34) had managed rush on their farm in the last 5 years, with 58.8% of these (n = 20) currently undertaking management as part of an AES and 64.7% (n = 22) having done so at some point in the last five years. Of this latter group, 95.5% of farmers recalled the specific AES scheme, but only 59.1% recalled the specific AES prescription code (Table E.5). Most respondents that managed rush did so with a topper or mower (n = 32), and two did so with a strimmer/brush cutter. The vast majority of respondents left cut rush in the field (88.2%), with the percentage doing so being relatively similar for those managing within (60%) and outside (40%) AES schemes. Similarly, the vast majority of respondents used aftermath grazing after cutting (94.1% in total; 86% of non-AES participants and 100% of AES participants). More information on rush management practices is provided in Table E.6.

5.4.9 Farmers’ ecological understanding of the system

Most respondents correctly identified northern lapwing (94.6%), Eurasian curlew (91.9%) and common snipe (70.3%). Very few respondents (16.2%) correctly identified Eurasian golden plover, but two respondents (5.4%) did identify this species as a plover (worth 0.5 points). This resulted in a median score for wader identification of 3 (mean = 2.76 ± 0.14). The vast majority of respondents correctly stated that waders had declined in the SWP over the last 20 years (83.8%; Table E.7).

Table 5.1. Number of respondents with varying degrees of rush cover change on their farm over the last 10 years that selected different Likert scale responses to questions concerning the impact of this change on land quality. Respondents who reported no change in the amount of rush on their farm over the last 10 years did not provide responses to the impact on land quality. Likert scale responses: 1 = considerably detrimental; 2 = moderately detrimental; 3 = slightly detrimental; 4 = no impact; 5 = slightly beneficial; 6 = moderately beneficial; 7 = considerably beneficial.

Has the amount of rush on your farm changed over the last 10 years? (n = 37)	**Overall impact on land quality from a farming perspective**							**Overall impact on land quality from an environmental perspective**						
	1	*2*	*3*	*4*	*5*	*6*	*7*	*1*	*2*	*3*	*4*	*5*	*6*	*7*
Considerably decreased (n = 2)					1		1				1		1	
Moderately decreased (n = 1)					1									1
Slightly decreased (n = 5)					2	3					1	2	2	
No change (n = 3)														
Slightly increased (n = 8)		1	5	2						1	5	1	1	
Moderately increased (n = 9)	2	5	1	1				5	1		1	1		1
Considerably increased (n = 9)	6	2	1					3	4			2		

5.4.2 Quantitative statistical analyses

5.4.2.1 Participation in rush management AES prescriptions

The model testing the hypothesis that participation was associated with farmers' perceived effectiveness of rush management AES prescriptions at improving wading bird habitat had the lowest AICc value (substantially lower than the null and base models) and the greatest explanatory power (McFadden's pseudo R^2 = 0.539; Table 5.2). Farmers who perceived rush management AES prescriptions to be more effective at improving wading bird habitat were less likely to currently participate in AES, but the relationship was marginally non-significant (β = -0.709, SE = 0.576, *P* = 0.085). The model testing the alternative hypothesis that participation was associated with farmers' perceived effectiveness of rush management AES prescriptions at reducing rush cover had a slightly higher AICc value, slightly lower explanatory power, and negligible evidence for an association between participation and perceived effectiveness (β = -0.259, SE = 0.295, *P* = 0.352; Table 5.2). There was no evidence from these models that participation was associated with the intensity of livestock grazing or farmer age (Table 4). The other three models testing the hypotheses that participation was associated with farmers' bird conservation attitudes, wading bird habitat knowledge, and perceived social norms had AICc values higher than the null model and thus, these hypotheses received negligible support and are thus not reported (see Table 5.2).

5.4.2.2 Perceived effectiveness of rush management AES prescriptions

Farmers' perceptions of the ability of rush management AES prescriptions to improve habitat for waders was significantly positively associated with perceived effectiveness at reducing rush cover (McFadden's pseudo R^2 = 0.290; β = 1.127, SE = 0.324, *P* = 0.000002). This was the only model with a lower AICc than the null model, and there was no evidence that farmer's grazing livestock intensity influenced their perceptions of rush management effectiveness (Table 5.2). The other three models testing the hypotheses that perceived effectiveness was associated with farmers' bird conservation attitudes, wading bird habitat knowledge, and rush encroachment importance score had AICc values higher than the null model and thus, these hypotheses received negligible support and are thus not reported (see Table 5.2).

Whilst 29.7% (n = 11) of respondents reported that they 'did not know' whether rush management AES prescriptions were effective at improving wading bird habitat, all 11 of these respondents had not recently (i.e. in the last 5 years) managed rush as part of an AES. Similarly, 35.1% (n = 13) of respondents 'did not know' if prescriptions were effective at reducing rush cover, and most (92.3%) had not recently managed rush with AES prescriptions.

5.4.2.3 Perceived effectiveness of farmer's rush management approach from a farming perspective

The base model including highest education qualification was the only model with an AICc lower than the null model and this explained limited amounts of variation (McFadden's pseudo $R^2 = 0.085$; Table 5.2). There was no evidence for our singular hypothesis testing model that AES participation was associated with whether farmers perceived their farm-specific rush management approach to improve their land from a farming perspective; the model had an AICc value higher than the null model, only explained 8.6% of the variation and is thus not reported (Table 5.2).

5.4.2.4 Perceived effectiveness of farmer's rush management approach from an environmental perspective

The null model had the lowest AICc and the three hypotheses testing models explained negligible amount of variation; there was thus no evidence for our hypotheses that perceived effectiveness of farmer's rush management approach from an environmental perspective was associated with AES participation, farmers' bird conservation attitudes, and farmers' wading bird habitat knowledge (Table 5.2).

Table 5.2. Quantitative models for each of the four response variables from stage two of the modelling process. Base and hypothesis testing models with Akaike information criterion with correction for small sample size (AICc) values higher than the null model are not reported and they provide no evidence supporting the associations with the response variables. Delta AICc values of the unreported models relative to the null model for response variable (i) were 1.695 (grazing livestock intensity and age base model), 2.250 (grazing livestock intensity base model), 3.970 (bird conservation index hypothesis model), 4.098 (wader knowledge index hypothesis model), and 4.112 (social norms index hypothesis model). Delta AICc values of the unreported models relative to the null model for response variable (ii) were 0.017 (bird conservation index hypothesis model), 3.662 (grazing livestock intensity base model), 5.553 (rush encroachment important hypothesis model), and 7.845 (wader knowledge index hypothesis model). Delta AICc values of the unreported models relative to the null model for response variable (iii) were 0.959 (grazing livestock intensity and highest education qualification base model), 2.493 (grazing livestock intensity base model), and 4.623 (agri-environment scheme (AES) participation hypothesis model). Delta AICc values of the unreported models relative to the null model for response variable (iv) were 2.732 (grazing livestock intensity base model), 4.229 (bird conservation index hypothesis model), 5.769 (AES participation hypothesis model), and 6.099 (wader knowledge index hypothesis model). 'Higher level education qualification' is the reference level for highest education qualification. Grazing livestock intensity is natural logarithm transformed.

Response variable	**Model structure**	**Predictor variable**					**McFadden's pseudo R^2**	**AICc**
		Grazing livestock intensity	Age	Highest education qualification	Perceived AES effectiveness – wading birds	Perceived AES effectiveness – rush cover		
(i) AES participation	Generalised linear model; binomial error structure; logit link			*Null model i.e. no predictors*				48.195
			β = -0.048 SE = 0.031 P = 0.096				0.060	47.687
		β = -0.300 SE = 0.783 P = 0.699	β = -0.033 SE = 0.046 P = 0.443		B = -0.709 SE = 0.576 P = 0.085		0.539	31.219
		β = -0.325 SE = 0.804 P = 0.682	β = -0.059 SE = 0.052 P = 0.210			B = -0.259 SE = 0.295 P = 0.352	0.512	32.588
(ii) Perceived AES effectiveness – wading birds	Cumulative link model; logit link			*Null model i.e. no predictors*				96.481
		β = -0.230 SE = 0.555				β = 1.127 SE = 0.324	0.290	82.453

		P = 0.678		P = 0.000002		
(iii) Perceived farm-specific rush management effectiveness – farm quality	Cumulative link model; logit link		*Null model i.e. no predictors*			114.196
			None β = 2.850 SE = 1.020 *School level education qualification* β = 1.908 SE = 0.921 P = 0.013		0.085	111.703
(iv) Perceived farm-specific rush management effectiveness – environment quality	Cumulative link model; logit link		*Null model i.e. no predictors*			118.323

5.4.3 Qualitative content analyses

5.4.3.1 Motivations for AES participation

The 22 respondents who had managed rush as part of an AES prescription within the last five years (including two respondents who no longer currently managed rush as part of an AES) provided six themes (five main themes and one sub-theme) as reasons for their participation. Themes are described in order of their frequency for those that were reported by more than 15% of the respondents (see Table 5.3). The most frequent stated reason (54.6% of respondents in this group) was increased income, e.g. "*I mean, the point is you can't, there's precious little money in farming now, particularly in these upland hills and therefore you've got to generate cash in other ways, and the HLS* [Higher Level Stewardship AES] *scheme is one particular way of doing that*" [respondent 36]. The second most frequent stated reason was to help the environment, wildlife, and plants (n = 9; 40.9%; e.g. "*Well, because we've always farmed this farm for wildlife actually, and managing it for environmental reasons. So, yeah, it* [AES prescription for rush management] *was an opportunity to continue doing it, really*" – respondent 35). The third most frequent reason was convenience as rush management AES prescriptions fit in with their farming activities and lifestyle, particularly because they were already managing rush on their farm (n = 8; 36.4%), e.g. "*I didn't have to make any radical changes to my farming practices when I joined the scheme because I was already farming that way. I was already farming extensively. Not intensively*" [respondent 30]. In two cases, this was because their farm contained land designated as a Site of Special Scientific Interest (SSSI) and rush management was mandatory as part of the SSSI management agreement; thus the farmers decided to join a rush management AES prescription as they would have to manage the rush regardless due to the farm's SSSI status, and it therefore provide them with a financial benefit.

5.4.3.2 Motivations for not participating in AES rush management

The 14 respondents not currently managing rush as part of an AES provided eight themes regarding decisions to not or no longer participate in a rush management AES prescription. Themes are described in order of their frequency for those that were reported by more than 15% of the respondents (see Table 5.3). The most frequent stated reason (35.7% of respondents in this group) was that the schemes were insufficiently flexible, e.g. "*We found for the different schemes that we could get, just like for what ground we have, we'd have to jump through serious hoops to go into the things. And for what we could actually get out of it, weren't a great deal compared to how much we'd have to change our farming policy*" [respondent 3]. The following two most frequent stated reasons were provided by four farmers each (28.6%). The first reason was that the schemes provided insufficient financial incentive (e.g. “*There was that much to it* [Higher Level Stewardship prescriptions] *for very little return, we decided it wasn't worth it*” – respondent 6) and the second reason was that the farmers were not aware of rush management prescriptions. The fourth most frequent stated reason (21.4% of respondents in this group) was that the schemes lacked understanding of local conditions, e.g. “*I don't think it's at all helpful by having your rules made by people that have no idea what your conditions are like and aren't really interested*” [respondent 13].

5.4.3.3 Co-designing rush management AES prescriptions

Many farmers thought improvements were required to the current major AES rush management prescription (GS16 rush infestation control supplement) to benefit farming (n = 20, 54.1%) and wader populations (n = 15, 40.5%). Nine themes for improvements were provided; these are described in order of their frequency for themes that were reported by more than 15% of the respondents who suggested improvements (see Table 5.3). The most frequent suggested change was regarding rush cutting requirements (farming: n = 12; wader populations: n = 7), with most farmers requesting more flexible cutting dates (e.g. “*It's really good not to cut them* [rush] *when wading birds are nesting, but if the season starts early or late, I wonder if those dates are a bit too inflexible*” – respondent 19); two farmers specifically mentioned that cutting date flexibility should be on a farm-by-farm basis (e.g. “*I think, you know, you should be able to have an on farm thing where if you say, ‘well, could we move it two weeks one way, two weeks the other?’. As long as you had their* [Natural England’s] *permission or, you know, the scheme's permission and they* [Natural England] *said ‘yeah, they've all gone’, fine. A bit more, yeah, just not complete flexibility where they* [farmers] *just go willy-nilly sort of thing but yeah*” – respondent 3). Other rush cutting requirement suggestions included two cuts within a year, increasing the rush cutting area, and not cutting on rotation. The second most frequent suggested change was to permit the use of herbicides (farming: n = 10; wader populations: n = 5), with most of these farmers seeking permission to use weed wipers due to the ability to target specific areas of the field (e.g. “*What you could try is where they* [rush cover] *are very bad, you could weed wipe patches, you know, strips and stripes. You know, you've got to feel you can sort of leave a few metres* [of rush] *in a strip. And then you could weed wipe a piece and then leave another strip and do it like that*” – interview 37), although a couple of farmers thought boom spraying would also be useful (e.g. “*The big secret to getting farmers and get me onside would be allowing a certain amount of boom spraying because I think that's 100% more effective than weed wiping. I mean in a controlled way*” – respondent 27).

The improvements suggested by farmers were similar for farming and waders. Ten farmers suggested the same changes for both. Those farmers that indicated different improvements were required (n =

4) suggested fewer or less intensive changes to benefit wader populations compared to farming, with differences relating to herbicide application (n = 2; "*Herbicides wouldn't go anywhere with the wildlife*" – respondent 5) and rush cutting and rush cover targets (n = 2; "*I don't think you'd cut as much, would you? If you were managing it for the wading birds…*" – respondent 26). Furthermore, six farmers suggested changes to benefit farming but no changes for wader populations, whereas one farmer suggested no changes to benefit farming but changes for wader populations. Most farmers who suggested improvements felt that they would moderately or considerably increase participation (farming related improvements: 80.0% of the 20 respondents, mean Likert score 5.9 ± 0.2; wader related improvements: 80.0% of the 15 respondents, mean Likert score 5.5 ± 0.4).

In addition to questioning farmers about improvements to AES prescriptions for rush management, when asked "*Would you consider changing your rush cutting pattern within a field to multiple, smaller cuts if there was a benefit to breeding wading birds?"* (n = 34), most farmers (n = 29; 85.3%) stated that they would be likely to implement a mosaic rush cutting pattern on their farm, with the majority moderately or considerably likely (n = 27; 79.4%).

Table 5.3. The two aspects within which open-ended questions were thematically coded, the number of respondents who provided answers to those questions, the themes identified from those responses, and the number of respondents who were assigned to each theme. The number of assigned respondents are noted for farmers who are currently managing rush as part of an AES (AES), farmers who are currently managing rush outside of an AES (Non-AES), and all respondents combined (Total). SSSI denotes Site of Special Scientific Interest.

Aspect	**Open-ended question**	**Number of respondents who responded**	**Themes**	**Number of respondents reporting each theme**		
				Total	**AES**	**Non-AES**
AES participation motivations	What influenced your decision to participate in this specific agri-environment scheme prescription?	22	1. Implement appropriate rush management	2	2	0
			2. Increased income	12	11	1
			3. Help environment, wildlife, and plants	9	8	1
			4. Compulsory participation with land	3	3	0
			5. Fits in with farming activities or lifestyle	6	6	0
			5a. Fits in with farming activities or lifestyle – SSSI	2	2	0
	Why do you not/no longer manage rush as part of an agri-environment scheme?	14	1. Insufficient financial incentive	4	0	4
			2. Contract period too long	1	0	1
			3. Scheme insufficiently flexible	5	0	5
			4. Lack of local conditions understanding	3	0	3
			5. Rush cover minor problem	2	0	2
			6. Did not qualify	1	0	1
			7. Not aware of rush management prescriptions	4	0	4
			8. Current restructuring of schemes and payments	1	0	1
Co-designing rush management AES prescriptions	What, if anything, would you change within the current Countryside Stewardship GS16 rush infestation control supplement in order to improve the effectiveness at reducing rush cover and why?	37	1. Herbicide application	10	6	4
			2. Lime application	1	1	0
			3. Cutting requirements	12	7	5
			4. More practical advice	2	2	0
			5. Cutting aftermath removal	2	2	0
			6. Rush cover target	2	2	0
			7. Increase stocking rates	3	3	0

		8. Rush prevention work	2	1	1
		9. Organic fertiliser application	2	1	1
		10. No changes	17	7	10
If you were focusing solely on managing rush to benefit breeding wading birds, what (if anything) would you change within the current Countryside Stewardship GS16 rush infestation control supplement and why?	37	1. Herbicide application	5	4	1
		2. Lime application	1	1	0
		3. Cutting requirements	7	4	3
		4. More practical advice	2	2	0
		5. Cutting aftermath removal	1	1	0
		6. Rush cover target	2	2	0
		7. Increase stocking rates	1	1	0
		8. Rush prevention work	1	0	1
		9. Organic fertiliser application	0	0	0
		10. No changes	22	9	13

5.5 Discussion

5.5.1 What influenced farmers' decisions to participate in AES prescriptions for rush management?

In our SWP study population of upland farmers, motivations and constraints that influence participation in rush management via AES prescriptions were highlighted by both quantitative and qualitative analyses. One of the core motivations for participation was farmers' pro-environmental attitudes, extending this previously discovered reason for other AES prescriptions to rush management prescriptions in the UK uplands (Wilson and Hart, 2000; Vanslembrouck *et al.*, 2002; Defrancesco *et al.*, 2008; Barreiro-Hurlé *et al.*, 2010; Karali *et al.*, 2014). Benefiting the environment, including birds, was often quoted indicating that these farmers believed such prescriptions could benefit waders, possibly through reducing rush cover and thus improving wader breeding habitat. Paradoxically, our quantitative models found marginal evidence that non-participating farmers were more likely to perceive such prescriptions to be more effective at improving wader habitat, and no evidence regarding the effectiveness at reducing rush cover. It is important to note, however, that our relatively small sample size could limit the ability to detect statistical significance. Moreover, positive attitudes to bird conservation did not translate into increased likelihood of participation, similar to findings by Guillem and Barnes (2013) for farmers in Scotland. In contrast to their results, however, SWP farmers had a good grasp of habitat requirements for wading birds and thus, this was not limiting the influence of their positive attitudes on participation; a knowledge-action gap is therefore established (Kollmuss and Agyeman, 2002; Knapp *et al.*, 2021).

Alongside benefiting the environment, financial incentives was a similarly, if not more, important reason behind participation, with both participating and non-participating farmers providing reasons relating to money (Morris *et al.*, 2000; Wilson and Hart, 2000; Defrancesco *et al.*, 2008; Ruto and Garrod, 2009; Karali *et al.*, 2014; Lastra-Bravo *et al.*, 2015). Whether the financial incentive offered is deemed sufficient to promote participation could relate to the convenience of the rush management AES prescriptions. Indeed, some non-participating farmers found the prescriptions to be insufficiently flexible and did not feel that the payments were enough to warrant the restrictions that the prescriptions imposed on their farm (Wilson and Hart, 2000; Ruto and Garrod, 2009; Espinosa-Goded *et al.*, 2010). Moreover, some felt that the schemes did not understand that conditions differ between farms, and this therefore acted as a further constraint to participation; for example, wet ground conditions often prevent the use of rush cutting machinery within the prescriptions' dates when such management is permitted. Conversely, for participating farmers the rush management prescriptions often fit in with their farming activities and lifestyle, particularly because they were already managing rush on their farm. The prescriptions therefore provided them with additional income without the need to change most aspects of their current farming practices (Wilson and Hart, 2000; Karali *et al*., 2014), perhaps raising concerns regarding how much additional gain is delivered through AES schemes. Interestingly, most participating SWP farmers were in Higher Level Stewardship agreements for rush management (Table E.5) which provide greater flexibility than Entry Level Stewardship (Schroeder *et al*., 2015). Farm-specific factors such as farm size, farm management intensity, and other sources of income also likely influence whether AES payment is deemed sufficient (Defrancesco *et al*., 2008; Karali *et al*., 2014; Lastra-Bravo *et al*., 2015).

The influence of social norms through neighbouring farmers can also influence AES participation as shown by Vanslembrouck *et al*., (2002), Hynes and Garvey, (2009), Defrancesco *et al*., (2008) and Cullen *et al*., (2020), yet we found no evidence that neighbouring farmers' opinions on AES engagement motivated or constrained participation in rush management prescriptions, concurring with Schroeder *et al*., (2015). Perhaps, communication with neighbouring farmers may be limited (Emery and Franks, 2012) or neighbours may themselves not have knowledge of specific AES prescriptions because a core constraint to participation for SWP farmers was unawareness of rush management AES prescriptions. This highlights the need for clearer communication of AES options and for Natural England (the non-departmental public body operating AES, and sponsored by the Department for Environment, Food & Rural Affairs) advisors to visit farmers to discuss appropriate options available on individual farms (Morris *et al*., 2000). Nevertheless, combining neighbouring farmers' opinions on rush management AES prescriptions (as we have used here) with neighbouring farmers' participation (as used by Vanslembrouck *et al*. (2002), and Cullen *et al*. (2020)) may help develop a more holistic metric of social norms and further understanding of their impacts on AES participation.

5.5.2 How effective do farmers perceive rush management, within and outside of AES, to be?

When examining farmers' perceptions of rush management, specifically within AES, efficacy at improving land for farming and waders were strongly co-associated indicating that farmers believe waders require lower rush cover. Thus, if AES achieve their aim at reducing rush cover, then wader habitat quality should increase. Indeed, most ecological research does suggest that rush management is beneficial for waders (Holton and Allcorn, 2006; Robson and Allcorn, 2006; Fisher and Walker, 2015) but there is concern about potential increases in nest predation risk for species that nest across a range of vegetation structures (e.g. Eurasian curlew and common snipe) and use concealment as a form of nest defence (Kelly *et al*., 2021). As we found that perceived effectiveness for waders is greater for farmers not participating in AES (see section 5.5.1), this suggests that participating farmers may not have observed increased wader abundance on their farm. This could nonetheless be due to other drivers limiting wader populations in the SWP such as high predation pressure (Smart *et al*., 2013; Roos *et al*., 2018), as well as the wider declines in UK wader populations (Harris *et al*., 2020). Interestingly, on farms where rush cover had increased over the last 10 years, farmers expressed opposing views regarding the overall impact of rush encroachment on the environment (Table 5.1); the actual quantity of rush, however, may influence an individual farmer's opinion.

When asking about the effectiveness of farmer's own rush management approach for farming and waders, current participation in AES prescriptions did not influence this. This finding provides further indication that not all farmers implementing AES rush management perceive it to be effective; although, implementation of management and interpretation of prescriptions is likely to be variable between farmers (Hejnowicz *et al*., 2016). Even farmers' wading bird habitat knowledge and bird conservation attitudes, which were encouragingly positive overall (satisfactory ecological knowledge and attitudes have been recorded in other regions and agricultural systems; for example, Guillem and Barnes, (2013) and Hevia *et al*., (2021)), were not evidenced to influence farmers' perceptions; although, this lack of variation in responses potentially explains these non-significant associations.

5.5.3 Can the efficacy and uptake of rush management AES prescriptions be improved by co-designing prescriptions with farmers?

Farmers' suggestions for changes that would increase the efficacy of rush management AES prescriptions for farming and wading birds were similar, indicating that few farmers thought there was a trade-off between rush management to benefit farming and wading birds. Yet, more farmers suggested improvements to reduce rush cover for farming than to improve habitat for waders. Such differences often related to implementing fewer or less intensive changes for waders; these farmers thus emulated the usual aims of AES which promote more extensive farming practices to benefit the environment (Kleijn and Sutherland, 2003). It is possible, however, that suggestions for farming or waders were not provided in some cases due to lack of ideas during the interview.

Suggested improvements primarily focused on expanding treatment methods to include herbicide application via weed wiping and more flexible rush cutting requirements (including cutting dates, cutting areas, and rush cover targets; Wilson and Hart, 2000; Ruto and Garrod, 2009; Espinosa-Goded *et al*., 2010). Individual farmers had variable opinions about the optimum weed wiping and rush cutting methods, possibly due to environmental differences between farms such as ground conditions, adding further support to the proposal of increased flexibility. Efficacy of such methods should be investigated and if beneficial, could advise future prescriptions, promoting co-design of AES with farmers (Department for Environment Food & Rural Affairs, 2020). Importantly, weed wiping can be requested in Higher Tier agreements of the GS16 rush infestation control supplement within England's current Countryside Stewardship (Natural England, 2018), and has been permitted as part of Eurasian curlew management on the Royal Society for the Protection of Birds' Lake Vyrnwy reserve, Powys, Wales (Fisher and Walker, 2015). It also encouraging to note that SWP farmers were willing to implement mosaic rush cutting patterns on their farm and thus create smaller cut and uncut rush patches, which anecdotal evidence suggests may benefit breeding wading birds by reducing waders' perceived and actual nest predation risk (Fisher and Walker, 2015). This suggests that farmers are open to applying more time-consuming management for conservation purposes (Herzon and Mikk, 2007; Guillem and Barnes, 2013; van Dijk *et al.*, 2016).

These farmer-endorsed changes are likely to increase the uptake of rush management prescriptions according to the SWP farmers, providing further promising evidence for AES co-design. Implementing increased flexibility and other suggested improvements whilst also ensuring that prescriptions are sufficiently targeted, and detailed, to deliver the required habitat improvements for waders is likely to be a challenge without increasing access to specialised advisors on the ground (Perkins *et al.*, 2011). Importantly, advisors would also help increase awareness of rush management prescriptions, a lack of which constrained participation in the SWP. Furthermore, some SWP farmers felt that prescriptions required better explanations regarding the best rush management methods and reasons why these benefit waders (Schroeder *et al.*, 2015; Hejnowicz *et al.*, 2016; Mills *et al.*, 2018), with a specific request for more visits from advisors who understand the variation in environmental conditions between farms. To enable increased flexibility on a farm-by-farm basis, advisors would need to consider annual variation in weather and the onset of spring, as well as circumstances on individual farms such as the presence of different wader species due to distinct breeding season timings (Joys and Crick, 2004).

5.5.4 Conclusions and implications

Our interviews with upland farmers in the SWP have revealed the motivations behind participation and non-participation in AES prescriptions for rush management, the perceived effectiveness of such prescriptions, and farmer-endorsed improvements that could increase the efficacy and likelihood of uptake. Based on the identified motivations and constraints, one easy approach to increase uptake of rush management AES is to ensure that all upland farmers who are eligible for such prescriptions are made aware through improved communication of AES options. Financial payment is also evidently an important driver and thus, the feasibility of increasing payment for such prescriptions should be assessed – with enhanced payments perhaps being linked to changes in management rather than maintaining the status quo. To ensure that this funding is contributing to worthwhile wader conservation, however, rush management prescriptions must be evaluated to ensure they benefit breeding waders, especially as not all SWP farmers implementing AES rush management perceived it to be effective.

Farmers' suggestions for changes that could increase the efficacy and uptake of rush management AES prescriptions primarily focused on expanding treatment methods to include herbicide application via weed wiping and more flexible rush cutting requirements. Recommendations for specific rush cutting procedures require empirical evidence support from upland areas; although, optimal cutting methods in terms of timing, height, and frequency have already received attention (Kaczmarek-Derda *et al*., 2014; Shellswell and Humpidge, 2018; Kaczmarek-Derda *et al*., 2019). Flexible management dates, an important aspect for SWP farmers, would necessitate increased government funding for greater access to specialised advisors on the ground. For herbicide application, if carried out sympathetically with breeding waders in mind and prescribed alongside cutting at the right time of year to reduce rush vigour, permission to weed wipe could help to control rushes (Fisher and Walker, 2015) and greatly increase farmer uptake in areas experiencing substantial rush encroachment.

Overall, we show that co-designing AES prescriptions for rush management with farmers (Emery and Franks, 2012; Lastra-Bravo *et al*., 2015; Department for Environment Food & Rural Affairs, 2020), who work with the land on a daily basis and have accumulated years of land management experience, could be a promising option for both increasing the effectiveness and uptake of such prescriptions. This is timely considering AES in England are currently transitioning to the new Environmental Land Management scheme (Department for Environment Food & Rural Affairs, 2020) and thus our findings could advise rush management prescriptions. Increasing uptake by farmers is key to ensuring landscape scale implementation of beneficial habitat management for waders (Whittingham, 2007; Dallimer *et al*., 2010; Perkins *et al*., 2011; Baker *et al*., 2012; Burgess *et al*., 2015; Franks *et al*., 2018), and ultimately the success of AES prescriptions.

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Chapter 6

General discussion

A view from a heather moor over rush pasture fields

6.1 Key findings

Breeding populations of European waders are declining, particularly across north-west Europe including the UK (BirdLife International, 2017). Importantly, the UK holds important breeding wader populations; for example, 19-27% of the global Eurasian Curlew *Numenius arquata* population resides in the UK (Brown et al., 2015). Several species including the Curlew breed in the UK uplands, often in agricultural grasslands and moorlands (Snow and Perrins, 1998; Balmer et al., 2013). Many environmental change drivers threaten breeding wader populations at these upland sites. These are reviewed in Chapter 1, which highlights the overall importance of habitat change and predation, as well as the limited understanding of how to optimise vegetation management for different wader species in order to improve habitat quality and manage predation risk. Indeed, the low breeding productivity that is driving population declines is often attributed to poor habitat quality and high nest and chick predation (Roodbergen et al., 2012; Franks et al., 2018; Roos et al., 2018). The literature review also highlights that many potentially important threats are understudied, including reduced invertebrate resources. Finally, the literature review highlights the scarcity of high quality evidence for many conservation interventions that have been proposed to benefit upland wader populations. Whilst large proportions of upland landscapes are managed via agri-environment schemes (AES) often with a specific objective of supporting breeding waders, their populations are still declining (Franks et al., 2018).

The overall aim of this thesis was to provide novel information that will help improve conservation for upland waders. Due to concerns about declining prey populations for these invertebrate-feeding birds, the first aim was to investigate the environmental conditions influencing invertebrate abundance and in turn, the potential for invertebrate prey to influence upland wader abundance. From a literature review of upland wader diet, Chapter 2 revealed eight key invertebrate dietary groups that comprised at least 20% of the diet for at least one wader species: earthworms, Gastropoda, Diptera larvae, Diptera adults, Coleoptera larvae, Coleoptera adults, Lepidoptera larvae, and Hymenoptera adults. Chapter 2 also revealed environmental conditions that are important for these key invertebrate prey for waders. Vegetation height is important for Diptera larvae (shorter vegetation is associated with higher abundance), and Gastropoda (taller vegetation is associated with higher abundance). High soil moisture, without waterlogging, is important for earthworms. Interestingly, presence of rush can also be important for earthworms, potentially during drier conditions, as well as for other key invertebrate groups (for example, Diptera adults and Coleoptera). Managing breeding sites to retain sufficient invertebrate abundance will bolster wader populations where food availability is limiting. Indeed, cranefly (Diptera: Tipulidae) abundance is a key determinant of European golden plover *Pluvialis apricaria* breeding success in two separate UK populations (Pearce-Higgins and Yalden, 2004; Douglas and Pearce-Higgins, 2014).

The second aim was to assess rush management from both ecological and social science perspectives. Rush management is a widespread conservation intervention for upland waders and is available within AES prescriptions (Natural England, 2018). The efficacy of rush management for benefiting waders is, however, unclear. Apart from two studies showing increased wader abundance following implementation of rush management alongside other conservation interventions (Holton and Allcorn, 2006; Robson and Allcorn, 2006), no studies have experimentally examined the impacts of rush management on wader population dynamics such as breeding productivity. Rush management was thus explored in this thesis from three perspectives: nest success (Chapter 3), wader abundance (Chapter 4), and upland farmers' perspectives (Chapter 5).

In terms of nest success, the artificial nest experiment revealed that daily nest predation rates (DPRs) were two times higher for nests located in fields with rush management and in cut rush patches due to the resultant shorter, less dense vegetation making nests more visible to predators (Kelly et al., 2021). Importantly, at the same study sites, the potential for rush management advocated by agri-environment schemes to increase breeding wader densities varied between species, with some evidence for increases in Snipe and Lapwing densities, but not Curlew. Due to our nest experiment design, the higher DPRs are most applicable to species that nest across a range of sward structures (for example, Curlew and Snipe; Valkama et al., 1998; Fisher and Walker, 2015; Wentworth, 2015; Zielonka et al., 2019) rather than species which nest exclusively in short vegetation (for example, Lapwing; Milsom et al., 2000). Yet, as rush management has the potential to attract Snipe to breeding sites, impact on nest success is an important factor to consider when designing rush management AES prescriptions.

When interviewing upland farmers in the South West Peak (SWP) about rush management, both within and outside of AES, the effectiveness of such management at benefiting upland waders was called into question by some farmers. To increase the efficacy and uptake of rush management within AES prescriptions, farmers commonly suggested, amongst other options, herbicide application such as weed wiping and improving the flexibility of management dates. An additional recommendation to improve the likelihood of prescription uptake, which is important to ensure beneficial landscape scale implementation, was increasing financial payments; for this funding to contribute to worthwhile wader conservation, ensuring the effectiveness of rush management prescriptions is vital.

If rush management is effective at improving breeding wader habitat, managing the environment to ensure removal of other potentially limiting factors is essential. Food availability, for example, cannot be ignored. Yet, Chapter 2 suggests that invertebrate biomass is not a crucial driving factor for upland wader populations in this thesis' primary study region, the SWP. This lends greater weight to rush management and the balance between foraging and predation; predation aspects of rush management arguably may be more important than foraging (in part due to the results from Chapter 2 assessing food availability).

6.2 Wider applicability of findings

Waders are charismatic birds that breed in farmed habitats in the UK uplands (Snow and Perrins, 1998; Balmer et al., 2013). Most species are experiencing population declines, with the plight of the Red-listed Curlew in particular drawing attention in recent years (Burns et al., 2020). The results of this thesis regarding the threats and conservation interventions for upland waders are relevant across the UK uplands because the main SWP study region comprises a mosaic of enclosed grassland and unenclosed moorland which are representative of UK upland habitats, land management, and land-use (for example, livestock farming and driven grouse moors). Some aspects such as land-use and land management will also be relevant to other upland farmland and moorland birds such as Meadow Pipit *Anthus pratensis* and Skylark *Alauda arvensis* because such species are also ground-nesting and insectivorous, and vegetation characteristics are important drivers of species abundance (Fuller et al., 2002; Pearce-Higgins and Grant, 2006; Douglas et al., 2017; Buchanan et al., 2017). Furthermore, many of the same threats identified in Chapter 1 (intensification of agriculture, predation, reduced invertebrate resources, climate change, human disturbance) are also facing waders in the lowlands, including those that also breed in the uplands (for example, Lapwing) and those that are lowland-specialists (for example, Black-tailed Godwit *Limosa limosa*; Schekkerman et al., 2009; Smart et al., 2014). Although game management such as driven grouse shooting is exclusive to the UK uplands (Thompson et al., 2016), other threats such as intensification of agriculture and reduced invertebrate resources are prevalent across Europe (Donald et al., 2001; Habel et al., 2019) and thus the results of this thesis regarding management of vegetation structure and invertebrate abundance will be applicable to breeding wader populations in other European countries.

6.3 Recommendations for conservation

Based on results of this thesis, recommendations can be made to help conserve upland waders. To maximise the suitable environmental conditions available for invertebrate prey, provision of a mosaic of habitats is recommended. Within a particular habitat, creation of heterogeneous vegetation would provide the shorter and taller swards that are beneficial for different invertebrate prey taxa (Buchanan et al., 2006; Arnott et al., 2021). The same advice relates to waders and rush management. A mosaic of different sward structures across the landscape, with short vegetation for foraging and a range of vegetation structures for different wader species to nest and rear chicks, would ensure favourable conditions for multiple wader species (Buchanan et al., 2017). For this to be successful, however, other components of the conservation toolkit for waders need to be implemented alongside rush management. Potential drivers of low breeding productivity, such as food availability and predation pressure, could be reduced by improving environmental conditions for key invertebrate prey taxa and by implementing predator control, ideally non-lethal but lethal where necessary. To ensure land management practices are implemented at a sufficiently large scale, co-designing AES prescriptions such as rush management with farmers should increase both efficacy and uptake.

6.4 Recommendations for future research

Although the Chapter 2 results suggest that invertebrate availability was not limiting wader populations in the SWP study region, the number of fields including those with waders present was relatively low. Therefore, recommendations for future research regarding invertebrate prey would involve surveying invertebrates and waders over more years with a larger sample of sites to increase the likelihood of fields containing waders. Extending the research to other upland regions in the UK would also reveal whether invertebrate availability limits wader populations in other regions.

Similarly, for Chapters 3 and 4 regarding the ecological perspective of rush management, conducting research over more years and a larger number of fields with real wader nests will test whether wader species select cut or uncut rush for nesting and whether AES rush management inadvertently creates an ecological trap by altering vegetation structure. Other recommended future research involves identification of the optimal spatial configuration of cut and uncut rush patches, and overall sward structure, that trade-offs between foraging conditions and nest predation risk. Ultimately, future research should aim to develop AES prescriptions that tailor to individual wader species' optimum rush cover and sward structure. From the social science perspective of rush management in Chapter 5, interviews should be conducted with more farmers and in other upland areas within the UK. In addition, the effectiveness of the changes suggested by farmers to improve the GS16 rush infestation control supplement, an example rush management AES prescription, should be tested. It would also be prudent to repeat this social science research for other conservation interventions within AES prescriptions that lack empirical evaluation. Overall, Chapters 3, 4 and 5 highlight the need to assess conservation intervention and AES prescription effectiveness to ensure they are delivering their intended benefits.

In addition to invertebrate prey resources and rush management which were studied in Chapters 2 to 5 of this thesis, other evidence gaps were highlighted in Chapter 1 and identified as priorities for future research. Similar to rush management, more research effort is required on upland moorlands, including grouse moors, to identify the best vegetation management to benefit moorland-breeding waders. Also linked to grouse moor management, the underlying causes of increased predator populations need to be identified and managed to reduce the need for costly predator control. Finally, the other main priority highlighted in Chapter 1 relates to climate change and wind farms, and the need to employ a before-after control-impact approach in future studies of wind farms and associated mitigation measures.

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Appendix A

Supporting information for Chapter 1

Table A.1. Low, medium, and high scores assigned each of the threats and associated conservation interventions for quantity of evidence and level of agreement, with the rationale for the scores provided.

Threats and conservation interventions	**Score**		**Rationale**
	Quantity of evidence	*Level of agreement*	
Threats			
Intensification of livestock farming	High	High	Extensive research surrounding the impacts of agricultural intensification on upland waders has been conducted, with the consensus that intensive livestock farming negatively impacts waders.
Game management	Medium (vegetation management) Medium (predator control)	Medium (vegetation management) High (predator control)	Despite the relatively high number of studies, more research effort is required to identify the best moorland vegetation management methods to provide beneficial vegetation structure for different wader species, and investigate the long-term impacts of vegetation management such as burning (greater than a few years post-burn) on waders. Moreover, the degree of consensus between current studies regarding the impacts of heather burning on different wader species is moderate, particularly for some species such as curlew and dunlin. A similar number of studies also investigated the effects of predator control, with a high degree of consensus between studies. Yet, few studies have successfully teased apart the effects of predator control and heather burning.
Afforestation	Medium	High	Relatively high number of studies, although essentially all studies examine commercial conifer plantations rather than planting of native broadleaved woodland, with virtually all showing negative impacts of plantations on waders.
Wind farms	Medium	Medium	Wind farm studies generally agreed on the impacts for most wader species, except golden plover. However, due to the relatively small number of studies and the lack of ‘before-after control-impact’ studies, recognised as the best standard for monitoring the ecological impacts of infrastructure, the quantity of evidence was assigned as medium.
Peat extraction	Low	Low	Little research has directly examined the impacts of peat extraction on waders, yet adverse effects can be construed via the influence on peatland habitat. Due to the low number of direct studies, this threat has been assigned low quantity of evidence.

Predation	High	High	Extensive research of predation pressure on upland waders has been conducted, with the consensus that predation is a key driver of population declines.
Reduced invertebrate resources	Low	High	The low number of studies directly investigating the impacts of invertebrate abundance on waders has resulted in this threat being assigned low quantity of evidence.
Climate change	Medium	High	Adverse impacts of climate change are expected based on the few empirical studies on waders, particularly due to effects on invertebrate prey populations. Most predictive climate change studies forecast detrimental effects on waders including wader population declines, range contractions, northward shifts, and loss of southern populations. More empirical studies would be beneficial.
Human disturbance	Medium	Medium	Relatively high number of studies with some providing evidence and others providing no evidence to support negative impacts.
Disease and parasitism	Low	Low	Only two studies have investigated the effect of disease and parasitism on waders, with a low level of agreement between these studies.
Conservation interventions			
Intensification of livestock farming	Medium	High	Evaluation of conservation interventions to improve farmed habitats for waders (particularly prescriptions within AES such as rush management) is deficient, but most studies agree that they have the potential to improve conditions for upland waders.
Game management	Low	Low	The best moorland vegetation management methods to provide beneficial vegetation structure for different wader species is uncertain, and the use of cutting has only been explored in a small number of studies compared to heather burning. There is no empirical evidence that deer culls benefit waders.
Afforestation	Low	High	Despite a high level of agreement, there are only a low number of empirical studies.
Wind farms	Low	High	Recommended interventions to mitigate wind farm development are inferred from studies exploring the impacts of wind farms on upland breeding waders. Most suggest avoiding installation of wind farms in key upland breeding areas.
Peat extraction	Low	Low	A few studies have demonstrated interventions that can restore peatlands following extraction but direct evidence for impacts of restoration on waders is lacking.

Predation	High	Medium	A high number of studies have investigated the effects of lethal and non-lethal predator control on waders, but not all studies found that it sufficiently increases wader breeding success to stabilise or increase wader populations. Research needs to investigate the underlying causes of the high predator densities.
Reduced invertebrate resources	Low	Medium	A few studies have investigated the impact of interventions to improve conditions, particularly soil conditions, for invertebrate prey and foraging waders, but most interventions are inferred from the ecology of invertebrate taxa. Moreover, the true impact of anthelmintics on invertebrate prey has yet to be determined.
Climate change	Low	Medium	A few studies exploring the impacts of lowered soil moisture on invertebrate prey (craneflies) agree that blocking drainage ditches to increase soil moisture could help mitigate the effects of climate change. However, because the effects of climate change are not yet particularly evident and are difficult to predict, only a small number of studies have examined potential mitigations for waders.
Human disturbance	Low	High	Only a small number of studies have demonstrated mitigation measures that reduced disturbance of upland breeding waders, but agree that reducing numbers of visitors to key upland wader breeding sites during the breeding season is important.
Disease and parasitism	Low	Low	No studies have empirically tested conservation interventions to reduce the impact of sheep ticks, but the first priority is to identify whether disease and parasitism exerts population level impacts on waders before testing interventions.

Appendix B

Supporting information for Chapter 2

Table B.1. The number of survey fields recorded within each habitat type category in 2009 and 2018. Control fields were those in which no waders were detected during the 2009 wader surveys. Wader fields were those in which at least one species of wader (lapwing, curlew and/or snipe) was detected during the 2009 wader surveys. Habitat type categories recorded in 2009: IP = improved pasture, SUP = semi-improved and unimproved pasture, UM = unenclosed moorland i.e., white moor and heather moor. Habitat type categories recorded in 2018: IP = improved pasture, SI = semi-improved pasture, UP = unimproved pasture, WM = white moor, HM = heather moor. Habitat type was unknown for control fields in 2009 as it was not recorded during the surveys for fields with no waders. The habitat type category recorded for a field in 2009 did not always match the habitat type category recorded during our surveys in 2018.

Field type	**Total number of fields**	**2009 habitat type**			**2018 habitat type**				
		IP	*SUP*	*UM*	*IP*	*SI*	*UP*	*WM*	*HM*
Control	14				5	4	3	0	2
Wader	42	9	23	10	10	17	5	8	2

B.1 Upland wader diet literature review

B.1.1 Methods

Based on the habitat types surveyed in the South West Peak (SWP) during our study, seven wader species were selected for the literature search regarding upland wader diet: northern Lapwing *Vanellus vanellus*, Eurasian curlew *Numenius arquata*, common snipe *Gallinago gallinago*, Eurasian golden plover *Pluvialis apricaria*, common redshank *Tringa totanus*, Eurasian oystercatcher *Haematopus ostralegus*, and dunlin *Calidris alpina*. Quantitative data on the diets of these species were collected using Google Scholar (https://scholar.google.co.uk), Web of Science (https://app.webofknowledge.com), and StarPlus (University of Sheffield's library catalogue and online discovery tool). The search comprised three rounds of terms: i) 'diet' AND one wader species' common name e.g. 'Eurasian golden plover'), ii) 'diet' AND one wader species' Latin name e.g. '*Pluvialis apricaria*', and iii) key invertebrate taxa (identified from literature obtained from rounds one and two) e.g. 'Tipulid' AND one wader species' Latin name e.g. '*Pluvialis apricaria*'. These three rounds of the literature search yielded 163, 57, and 6 results, respectively. Subsequently, raw quantitative wader diet data was extracted from articles where available (Table B.2) and grouped by i) wader species, ii) invertebrate taxa and (where applicable) life cycle stage, and iii) type of data available (% dry biomass, % occurrence, and % proportion of diet items). The mean of each group was then calculated.

B.1.2 Limitations

This literature review highlighted the dearth of research concerning wader diet composition in upland habitats and thus, quantitative data deficiency was a limitation for several of our focal wader species. For example, the single upland study obtained for curlew focused on moorland habitats; this overlooks the key foraging areas on agricultural grasslands (Robson, 1998; Ewing et al., 2017). For lapwing, redshank, and oystercatcher, however, quantitative data within upland habitats was absent entirely and thus, diet composition data had to be obtained from lowland inland studies. An additional limitation of the extracted quantitative data was the different sources of the diet composition data. The selected studies from which data was extracted collected samples from at least one of three different sources: stomach, pellets, and faeces. Consumed invertebrates may have experienced varying degrees of digestion when collected from each of the three sources which could bias the identified dietary component compositions, but this potential difference in unknown.

B.1.3 2018 data

Invertebrate groups (taxa and life cycle stages) which were not listed, by at least one study obtained from the literature review, as comprising part of at least one of our seven focal wader species' diet were excluded from the analyses of our 2018 study. Prior to the removal of the excluded taxa (Diplopoda, Chilopoda, Collembola, Acari, Isopoda, Dermaptera, Enchytraeidae, and specimens classified as unknown), such excluded taxa accounted respectively for 0.280% and 0.614% of the total invertebrate biomass in the early breeding season and late breeding season (Table B.3).

Table B.2. The source(s) of quantitative data on wader diet for the seven focal wader species.

Wader species	Diet data source(s)
Northern lapwing	Klomp, 1954; Galbraith, 1989; Linsley, 1999; Ausden et al., 2003
Eurasian curlew	Robson, 1998
Common snipe	Green et al., 1990; Hoodless et al., 2007
Eurasian golden plover	Pearce-Higgins and Yalden, 2003, 2004; Machín et al., 2017
Common redshank	Ausden et al., 2003; Sánchez et al., 2005
Eurasian oystercatcher	Heppleston, 1972
Dunlin	Holmes, 1966; Baker, 1977

Table B.3. Cumulative biomass (g) across all within-field sampling points for excluded invertebrate taxa and total invertebrate taxa (prior to the removal of the excluded invertebrate taxa) with mean ± standard error (SE) provided in parentheses. Biomass values are given for all sub-samples combined during the early and late breeding seasons separately.

Breeding season	**Cumulative biomass (mean ± SE; g)**	
	Excluded taxa	*Total invertebrate taxa*
Early	1.036 (0.004 ± 0.0009)	370.012 (1.294 ± 0.083)
Late	0.520 (0.004 ± 0.001)	84.760 (0.565 ± 0.055)

Table B.4. The key dietary components (comprising ≥ 20% of the diet) for each of our seven focal wader species. A tick mark indicates that the invertebrate group (taxa and, where applicable, life cycle stage) was identified as a key dietary component for that wader species. Even if only one family from an invertebrate order was listed as a major part of a wader species' diet, the order is highlighted as a key dietary component. Diptera larvae and Diptera adults include the family Tipulidae.

Wader species	**Key dietary component (taxa and life cycle stage)**							
	Lumbricidae	*Gastropoda*	*Diptera larvae*	*Diptera adults*	*Coleoptera larvae*	*Coleoptera adults*	*Lepidoptera larvae*	*Hymenoptera adults*
Northern lapwing	✓	✓	✓		✓	✓	✓	
Eurasian curlew				✓		✓		
Common snipe	✓	✓	✓			✓		
Eurasian golden plover	✓		✓	✓		✓		
Common redshank			✓	✓		✓		✓
Eurasian oystercatcher	✓		✓					
Dunlin			✓	✓				

Table B.5. Cumulative biomass (g) across all within-field sampling points for ‘total’ invertebrate dietary components and for key invertebrate dietary components individually and combined (mean ± standard error (SE) provided in parentheses). Biomass values are given for each of the five spatial layers during the early and late breeding seasons separately.

Breeding season stage	**Invertebrate group**	**Cumulative biomass (mean ± SE; g)**				
		Vegetation	*Shallow soil*	*Deep soil*	*All sub-samples combined*	*Vegetation and shallow soil combined*
Early	‘Total’ dietary components	1.247 (0.004 ± 0.001)	220.768 (0.772 ± 0.054)	146.961 (0.514 ± 0.043)	368.976 (1.290 ± 0.083)	222.015 (0.776 ± 0.053)
	All key dietary components	0.312 (0.001 ± 0.0004)	208.129 (0.728 ± 0.052)	145.238 (0.508 ± 0.043)	367.432 (1.285 ± 0.083)	220.531 (0.771 ± 0.053)
	Lumbricidae	0.068 (0.0002 ± 0.0002)	184.974 (0.647 ± 0.049)	144.163 (0.504 ± 0.043)	329.205 (1.151 ± 0.079)	185.042 (0.647 ± 0.049)
	Diptera larva	0.041 (0.0001 ± 0.0001)	22.504 (0.079 ± 0.010)	0.963 (0.003 ± 0.001)	23.508 (0.082 ± 0.010)	22.545 (0.079 ± 0.010)
	Diptera adult	0.102 (0.0004 ± 0.0002)	0.078 (0.0003 ± 0.0001)	0.000 (0.000 ± 0.000)	0.180 (0.0006 ± 0.0003)	0.180 (0.0006 ± 0.0003)
	Coleoptera adult	0.101 (0.0004 ± 0.0002)	0.573 (0.002 ± 0.0005)	0.112 (0.0004 ± 0.0003)	0.786 (0.003 ± 0.0006)	0.674 (0.002 ± 0.0005)
	Coleoptera larva	0.022 (0.00008 ± 0.00006)	1.094 (0.004 ± 0.001)	0.630 (0.002 ± 0.0009)	1.746 (0.006 ± 0.001)	1.116 (0.004 ± 0.001)
	Lepidoptera larva	0.279 (0.001 ± 0.0006)	0.709 (0.003 ± 0.001)	0.659 (0.002 ± 0.002)	1.647 (0.006 ± 0.002)	0.988 (0.004 ± 0.001)
	Gastropoda	0.473 (0.002 ± 0.001)	9.473 (0.033 ± 0.006)	0.370 (0.001 ± 0.001)	10.316 (0.036 ± 0.006)	9.946 (0.035 ± 0.006)
	Hymenoptera adult	0.009 (0.00003 ± 0.00002)	0.031 (0.0001 ± 0.00006)	0.004 (0.00001 ± 0.00001)	0.044 (0.0002 ± 0.00006)	0.040 (0.0001 ± 0.00006)
Late	‘Total’ dietary components	3.109 (0.021 ± 0.007)	33.676 (0.225 ± 0.027)	47.455 (0.316 ± 0.040)	84.240 (0.562 ± 0.056)	36.785 (0.245 ± 0.030)
	All key dietary components	0.467 (0.003 ± 0.001)	25.119 (0.168 ± 0.022)	46.340 (0.309 ± 0.040)	81.469 (0.543 ± 0.055)	34.056 (0.227 ± 0.027)
	Lumbricidae	0.000 (0.000 ± 0.000)	13.030 (0.087 ± 0.014)	42.484 (0.283 ± 0.036)	55.514 (0.370 ± 0.043)	13.030 (0.087 ± 0.014)
	Diptera larva	0.207 (0.001 ± 0.001)	11.521 (0.077 ± 0.017)	3.760 (0.025 ± 0.009)	15.488 (0.103 ± 0.019)	11.728 (0.078 ± 0.017)
	Diptera adult	0.028 (0.0002 ± 0.0002)	0.092 (0.0006 ± 0.0005)	0.000 (0.000 ± 0.000)	0.120 (0.001 ± 0.001)	0.120 (0.001 ± 0.001)
	Coleoptera adult	0.232 (0.002 ± 0.001)	0.476 (0.003 ± 0.001)	0.096 (0.001 ± 0.0004)	0.804 (0.005 ± 0.001)	0.708 (0.005 ± 0.001)
	Coleoptera larva	0.025 (0.0002 ± 0.0001)	0.690 (0.005 ± 0.001)	0.234 (0.002 ± 0.0006)	0.949 (0.006 ± 0.001)	0.715 (0.005 ± 0.001)
	Lepidoptera larva	0.094 (0.0006 ± 0.0005)	1.322 (0.009 ± 0.004)	0.101 (0.0007 ± 0.0007)	1.517 (0.010 ± 0.004)	1.416 (0.009 ± 0.004)
	Gastropoda	1.850 (0.012 ± 0.006)	4.255 (0.028 ± 0.008)	0.543 (0.004 ± 0.002)	6.648 (0.044 ± 0.012)	6.105 (0.041 ± 0.011)
	Hymenoptera adult	0.016 (0.0001 ± 0.00008)	0.218 (0.002 ± 0.001)	0.195 (0.001 ± 0.001)	0.429 (0.003 ± 0.002)	0.234 (0.002 ± 0.001)

Table B.6. Environmental conditions recorded during breeding wader surveys and invertebrate surveys, and the spatial scale at which metrics were obtained (S = variable measured at the within-field sampling points during invertebrate surveys; F = variable recorded at the field level during breeding wader surveys).

Environmental variable	Unit	Spatial scale	Description
Broad habitat type	Category	F	Visual assessment Five categories: heather moor; white moor; improved pasture; semi-improved pasture; unimproved pasture
Rush cover	To the nearest 5%	F	All rush species
Livestock dung presence/absence	Category	F	Two categories: present; absent
Soil penetration resistance	kgF	S	Soil penetrometer measurements of the force to drive a metal rod (5 mm diameter) to a depth of 10 cm Mean value calculated from the three measurements taken at each within-field sampling point
Soil moisture	%	S	SM150 soil moisture kit measurements Soil type setting used (mineral; peat mix) dependent on broad soil type Mean value calculated from the three measurements taken at each within-field sampling point
Soil pH	pH	S	HI-12922 HALO wireless soil pH electrode, 0-14 pH
Vegetation height	cm	S	Recorded to the nearest 5 cm, using a measuring stick
Vegetation density	Number	S	Number of concealed white bands on a pole
Rush dominance	Category	S	Visual assessment Two categories: dominant; not dominant/absent

Table B.7. Structure of the twenty log-normal generalised linear mixed effects models (gaussian (identity)) of the five metrics of invertebrate biomass with the dataset used, the spatial layer of the response variable, and the predictor variables (fixed and random) included. 'Quadratic' in parentheses after a predictor variable indicates that the linear and quadratic terms of this variable were included in the model. All response variables were natural logarithm (x + 0.001) transformed. Note that the Gastropod biomass model for the early breeding season had a singular fit, indicating that the random effect term had zero variance – following advocacy from Ben Bolker (lme4 package author; https://bbolker.github.io/mixedmodels-misc/glmmFAQ.html), we retained the random effect term. V-S-D spatial layer indicates that invertebrate biomass data was used from all sub-samples: vegetation, shallow soil, and deep soil. V-S spatial layer indicates that invertebrate biomass data was used from the vegetation and shallow soil sub-samples only.

Response variable	**Dataset**	**Spatial layer**	**Predictor variables**
Total invertebrate biomass	Early breeding season (*n* = 286)	V-S-D	Soil penetration resistance (quadratic) + Soil moisture (quadratic) + Soil pH (quadratic) + Vegetation height (square root transformed) + Vegetation density + Rush presence + Livestock dung + (1|Field identity)
Total invertebrate biomass	Early breeding season (*n* = 286)	V-S	Soil penetration resistance (quadratic) + Soil moisture (quadratic) + Soil pH (quadratic) + Vegetation height (square root transformed) + Vegetation density + Rush presence + Livestock dung + (1|Field identity)
Total invertebrate biomass	Late breeding season (*n* = 150)	V-S-D	Soil penetration resistance + Soil moisture + Soil pH (quadratic) + Vegetation height (square root transformed) + Vegetation density + Rush presence + Livestock dung + (1|Field identity)
Total invertebrate biomass	Late breeding season (*n* = 150)	V-S	Soil penetration resistance + Soil moisture (quadratic) + Soil pH (quadratic) + Vegetation height (square root transformed) + Vegetation density + Rush presence + Livestock dung + (1|Field identity)
Lumbricidae biomass	Early breeding season (*n* = 286)	V-S-D	Soil penetration resistance (quadratic) + Soil moisture (quadratic) + Soil pH (quadratic) + Vegetation height (square root transformed) + Vegetation density + Rush presence + Livestock dung + (1|Field identity)
Lumbricidae biomass	Early breeding season (*n* = 286)	V-S	Soil penetration resistance (quadratic) + Soil moisture (quadratic) + Soil pH (quadratic) + Vegetation height (square root transformed) + Vegetation density + Rush presence + Livestock dung + (1|Field identity)
Lumbricidae biomass	Late breeding season (*n* = 150)	V-S-D	Soil penetration resistance + Soil moisture + Soil pH (quadratic) + Vegetation height (square root transformed; quadratic) + Vegetation density + Rush presence + Livestock dung + (1|Field identity)
Lumbricidae biomass	Late breeding season (*n* = 150)	V-S	Soil penetration resistance + Soil moisture (quadratic) + Soil pH (quadratic) + Vegetation height (square root transformed) + Vegetation density + Rush presence + Livestock dung + (1|Field identity)
Diptera larvae biomass	Early breeding season (*n* = 286)	V-S-D	Soil penetration resistance + Soil moisture + Soil pH (quadratic) + Vegetation height (square root transformed) + Vegetation density + Rush presence + Livestock dung + (1|Field identity)
Diptera larvae biomass	Early breeding season (*n* = 286)	V-S	Soil penetration resistance + Soil moisture (quadratic) + Soil pH (quadratic) + Vegetation height (square root transformed) + Vegetation density + Rush presence + Livestock dung + (1|Field identity)
Diptera larvae biomass	Late breeding season (*n* = 150)	V-S-D	Soil penetration resistance (quadratic) + Soil moisture + Soil pH + Vegetation height (square root transformed) + Vegetation density + Rush presence + Livestock dung + (1|Field identity)

Diptera larvae biomass	Late breeding season (*n* = 150)	V-S	Soil penetration resistance (quadratic) + Soil moisture + Soil pH (quadratic) + Vegetation height (square root transformed) + Vegetation density + Rush presence + Livestock dung + (1|Field identity)
Gastropoda biomass	Early breeding season (*n* = 286)	V-S-D	Soil penetration resistance + Soil moisture + Soil pH + Vegetation height (square root transformed) + Vegetation density + Rush presence + Livestock dung + (1|Field identity)
Gastropoda biomass	Early breeding season (*n* = 286)	V-S	Soil penetration resistance + Soil moisture + Soil pH + Vegetation height (square root transformed) + Vegetation density + Rush presence + Livestock dung + (1|Field identity)
Gastropoda biomass	Late breeding season (*n* = 150)	V-S-D	Soil penetration resistance + Soil moisture + Soil pH + Vegetation height (square root transformed) + Vegetation density + Rush presence + Livestock dung + (1|Field identity)
Gastropoda biomass	Late breeding season (*n* = 150)	V-S	Soil penetration resistance + Soil moisture + Soil pH + Vegetation height (square root transformed) + Vegetation density + Rush presence + Livestock dung + (1|Field identity)
Other key invertebrate biomass	Early breeding season (*n* = 286)	V-S-D	Soil penetration resistance + Soil moisture + Soil pH + Vegetation height (square root transformed) + Vegetation density + Rush presence + Livestock dung + (1|Field identity)
Other key invertebrate biomass	Early breeding season (*n* = 286)	V-S	Soil penetration resistance + Soil moisture + Soil pH + Vegetation height (square root transformed) + Vegetation density + Rush presence + Livestock dung + (1|Field identity)
Other key invertebrate biomass	Late breeding season (*n* = 150)	V-S-D	Soil penetration resistance + Soil moisture + Soil pH + Vegetation height (square root transformed) + Vegetation density + Rush presence + Livestock dung + (1|Field identity)
Other key invertebrate biomass	Late breeding season (*n* = 150)	V-S	Soil penetration resistance + Soil moisture + Soil pH + Vegetation height (square root transformed) + Vegetation density + Rush presence + Livestock dung + (1|Field identity)

Table B.8. Results of log-normal generalised linear mixed effects models (GLMM) investigating the influence of habitat type on invertebrate biomass (represented by five different metrics). Parameter estimates with standard errors, and Type II Wald F tests with Kenward-Roger degrees of freedom are presented, with significant effects highlighted with an asterisk. Improved pasture is the reference level. HM = heather moor, SI = semi-improved pasture, UP = unimproved pasture, WM = white moor. Theoretical conditional R^2 (Nakagawa and Schielzeth, 2013; Nakagawa et al., 2017) are presented for each GLMM. For the Diptera larvae biomass GLMM in the V-S-D spatial layer and early breeding season, bobyqa optimiser was used due to lack of convergence with the default optimiser (nloptwrap; used in all other models).

Response variable	**Spatial layer**	**Breeding season**	**Model R^2**	**Habitat type predictor variable**					
				β ± SE				*F*	*P* value
				HM	*SI*	*UP*	*WM*		
Total invertebrate dietary biomass	V-S-D	Early	0.601	-5.728 ± 0.591	-0.728 ± 0.354	-3.408 ± 0.458	-4.648 ± 0.456	49.950	2.200^{-16} *
	V-S	Early	0.551	-5.165 ± 0.594	-0.798 ± 0.356	-2.951 ± 0.461	-4.242 ± 0.458	39.085	6.464^{-15} *
	V-S-D	Late	0.438	-4.064 ± 0.703	-0.536 ± 0.542	-2.134 ± 0.662	-2.183 ± 0.774	10.851	3.037^{-5} *
	V-S	Late	0.245	-1.837 ± 0.908	0.132 ± 0.700	-0.176 ± 0.854	-0.127 ± 0.999	1.454	0.246
Lumbricidae biomass	V-S-D	Early	0.614	-6.797 ± 0.626	-1.065 ± 0.375	-4.556 ± 0.485	-5.815 ± 0.482	67.801	2.200^{-16} *
	V-S	Early	0.578	-6.154 ± 0.620	-1.171 ± 0.372	-4.036 ± 0.481	-5.403 ± 0.478	55.410	2.200^{-16} *
	V-S-D	Late	0.469	-5.591 ± 0.771	-0.863 ± 0.594	-2.317 ± 0.727	-4.213 ± 0.848	17.954	4.512^{-7} *
	V-S	Late	0.154	-2.287 ± 0.751	-0.412 ± 0.578	0.473 ± 0.709	-1.012 ± 0.827	3.471	0.022 *
Diptera larvae biomass	V-S-D	Early	0.281	-2.404 ± 0.790	0.050 ± 0.474	-0.795 ± 0.613	-1.549 ± 0.611	4.351	0.004 *
	V-S	Early	0.257	-2.305 ± 0.768	0.004 ± 0.461	-0.933 ± 0.596	-1.466 ± 0.593	4.196	0.005 *
	V-S-D	Late	0.289	-2.103 ± 0.915	0.092 ± 0.705	-1.831 ± 0.861	-0.359 ± 1.007	2.867	0.044 *
	V-S	Late	0.306	-1.520 ± 0.915	0.380 ± 0.705	-1.247 ± 0.860	0.224 ± 1.007	1.974	0.129
Gastropoda biomass	V-S-D	Early	0.054	-0.435 ± 0.489	0.758 ± 0.292	0.147 ± 0.377	-0.287 ± 0.374	3.572	0.012 *
	V-S	Early	0.058	-0.436 ± 0.489	0.746 ± 0.292	0.147 ± 0.377	-0.333 ± 0.374	3.622	0.011 *
	V-S-D	Late	0.231	-0.603 ± 0.715	0.733 ± 0.551	0.522 ± 0.673	-0.225 ± 0.787	1.339	0.283
	V-S	Late	0.208	-0.481 ± 0.672	0.750 ± 0.518	0.634 ± 0.633	-0.103 ± 0.740	1.380	0.269
Other key invertebrate biomass	V-S-D	Early	0.143	0.013 ± 0.519	0.185 ± 0.311	0.038 ± 0.403	0.147 ± 0.400	0.112	0.978
	V-S	Early	0.147	0.299 ± 0.463	0.378 ± 0.278	0.437 ± 0.360	0.494 ± 0.358	0.715	0.586
	V-S-D	Late	0.140	0.064 ± 0.591	0.402 ± 0.455	-0.241 ± 0.557	1.220 ± 0.651	1.355	0.278

	V-S	Late	0.094	0.243 ± 0.526	0.207 ± 0.405	-0.064 ± 0.497	1.301 ± 0.579	1.503	0.231

Table B.9. Structure of the fifteen main effects only generalised linear models (poisson (log)) of the three focal wader species pairs with the temporal level of the environmental condition variables, the spatial layer of the invertebrate biomass metric predictor variable, and the predictor variables included (field size is included as an offset in all models). 'Quadratic' in parentheses after a predictor variable indicates that the linear and quadratic terms of this variable were included in the model. Type of transformation, where applicable, is shown in parentheses after a predictor variable. V-S-D spatial layer indicates that invertebrate biomass data was used from all sub-samples: vegetation, shallow soil, and deep soil. V-S spatial layer indicates that invertebrate biomass data was used from the vegetation and shallow soil sub-samples only.

Response variable	Breeding season	Invertebrate biomass spatial layer	Predictor variables
Lapwing pairs	Early	V-S	Soil penetration resistance + Vegetation height + Rush cover (natural logarithm (x + 2.5) transformed) + **Total invertebrate dietary biomass (quadratic)** + Field size (offset; natural logarithm (x + 0.001) transformed)
Lapwing pairs	Early	V-S	Soil penetration resistance + Vegetation height + Rush cover (natural logarithm (x + 2.5) transformed) + **Lumbricidae biomass (square root transformed; quadratic)** + Field size (offset; natural logarithm (x + 0.001) transformed)
Lapwing pairs	Early	V-S	Soil penetration resistance + Vegetation height + Rush cover (natural logarithm (x + 2.5) transformed) + **Diptera larvae biomass (natural logarithm (x + 0.001) transformed)** + Field size (offset; natural logarithm (x + 0.001) transformed)
Lapwing pairs	Early	V-S	Soil penetration resistance + Vegetation height + Rush cover (natural logarithm (x + 2.5) transformed) + **Gastropoda biomass (natural logarithm (x + 0.001) transformed)** + Field size (offset; natural logarithm (x + 0.001) transformed)
Lapwing pairs	Early	V-S	Soil penetration resistance + Vegetation height + Rush cover (natural logarithm (x + 2.5) transformed) + **Other key invertebrate biomass (natural logarithm (x + 0.001) transformed)** + Field size (offset; natural logarithm (x + 0.001) transformed)
Snipe pairs	Overall	V-S-D	Soil penetration resistance + Vegetation height (quadratic) + Rush cover (natural logarithm (x + 2.5) transformed) + **Total invertebrate dietary biomass (square root transformed; quadratic)** + Field size (offset; natural logarithm (x + 0.001) transformed)
Snipe pairs	Overall	V-S-D	Soil penetration resistance + Vegetation height (quadratic) + Rush cover (natural logarithm (x + 2.5) transformed) + **Lumbricidae biomass (square root transformed; quadratic)** + Field size (offset; natural logarithm (x + 0.001) transformed)
Snipe pairs	Overall	V-S-D	Soil penetration resistance + Vegetation height (quadratic) + Rush cover (natural logarithm (x + 2.5) transformed) + **Diptera larvae biomass (natural logarithm (x + 0.001) transformed; quadratic)** + Field size (offset; natural logarithm (x + 0.001) transformed)
Snipe pairs	Overall	V-S-D	Soil penetration resistance + Vegetation height (quadratic) + Rush cover (natural logarithm (x + 2.5) transformed) + **Gastropoda biomass (natural logarithm (x + 0.001) transformed)** + Field size (offset; natural logarithm (x + 0.001) transformed)
Snipe pairs	Overall	V-S-D	Soil penetration resistance + Vegetation height (quadratic) + Rush cover (natural logarithm (x + 2.5) transformed) + **Other key invertebrate biomass (natural logarithm (x + 0.001) transformed; quadratic)** + Field size (offset; natural logarithm (x + 0.001) transformed)
Curlew pairs	Overall	V-S-D	Soil penetration resistance + Vegetation height (quadratic) + Rush cover (natural logarithm (x + 2.5) transformed) + **Total invertebrate dietary biomass (square root transformed)** + Field size (offset; natural logarithm (x + 0.001) transformed)

Curlew pairs	Overall	V-S-D	Soil penetration resistance + Vegetation height (quadratic) + Rush cover (natural logarithm (x + 2.5) transformed) + **Lumbricidae biomass (square root transformed)** + Field size (offset; natural logarithm (x + 0.001) transformed)
Curlew pairs	Overall	V-S-D	Soil penetration resistance + Vegetation height (quadratic) + Rush cover (natural logarithm (x + 2.5) transformed) + **Diptera larvae biomass (natural logarithm (x + 0.001) transformed)** + Field size (offset; natural logarithm (x + 0.001) transformed)
Curlew pairs	Overall	V-S-D	Soil penetration resistance + Vegetation height (quadratic) + Rush cover (natural logarithm (x + 2.5) transformed) + **Gastropoda biomass (natural logarithm (x + 0.001) transformed)** + Field size (offset; natural logarithm (x + 0.001) transformed)
Curlew pairs	Overall	V-S-D	Soil penetration resistance + Vegetation height (quadratic) + Rush cover (natural logarithm (x + 2.5) transformed) + **Other key invertebrate biomass (natural logarithm (x + 0.001) transformed)** + Field size (offset; natural logarithm (x + 0.001) transformed)

Table B.10. Structure of the eleven main effects plus interaction term generalised linear models (GLMs; poisson (log)) of two focal wader species pairs with the predictor variables included (field size is included as an offset in all models). 'Quadratic' in parentheses after a predictor variable indicates that the linear and quadratic terms of this variable were included in the model. Type of transformation, where applicable, is shown in parentheses after a predictor variable. The environmental condition variables included data from both the early and late breeding season, where available. The invertebrate biomass metric predictor variable included data from the V-S-D spatial layer (i.e all sub-samples: vegetation, shallow soil, and deep soil). These GLMs are in addition to the main effects only models in Tables 2.3 and B.9, and include interaction terms between soil penetration resistance or vegetation height and invertebrate biomass metrics. The decision to construct a GLM with an interaction term was based on whether soil penetration resistance or vegetation height in the main effects only GLM were significant (*P* < 0.05). If either variable in the main effects only GLM was significant, an additional version of the main effects only GLM was constructed including an interaction between the significant environmental variable and the invertebrate biomass metric.

Response variable	**Predictor variables**
Snipe pairs	Soil penetration resistance + Vegetation height (quadratic) + Rush cover (natural logarithm (x + 2.5) transformed) + **Other key invertebrate biomass (natural logarithm (x + 0.001) transformed; quadratic)** + Soil penetration resistance***Other key dietary components biomass (natural logarithm (x + 0.001) transformed) +** Field size (offset; natural logarithm (x + 0.001) transformed)
Snipe pairs	Soil penetration resistance + Vegetation height (quadratic) + Rush cover (natural logarithm (x + 2.5) transformed) + **Other key invertebrate biomass (natural logarithm (x + 0.001) transformed; quadratic)** + Soil penetration resistance***Other key dietary components biomass (natural logarithm (x + 0.001) transformed; quadratic) +** Field size (offset; natural logarithm (x + 0.001) transformed)
Curlew pairs	Soil penetration resistance + Vegetation height (quadratic) + Rush cover (natural logarithm (x + 2.5) transformed) + **Total invertebrate dietary biomass (square root transformed)** + **Vegetation height***'Total' invertebrate dietary components biomass (square root transformed) + Field size (offset; natural logarithm (x + 0.001) transformed)
Curlew pairs	Soil penetration resistance + Vegetation height (quadratic) + Rush cover (natural logarithm (x + 2.5) transformed) + **Total invertebrate dietary biomass (square root transformed)** + **Vegetation height (quadratic)***'Total' invertebrate dietary components biomass (square root transformed) + Field size (offset; natural logarithm (x + 0.001) transformed)
Curlew pairs	Soil penetration resistance + Vegetation height (quadratic) + Rush cover (natural logarithm (x + 2.5) transformed) + **Lumbricidae biomass (square root transformed)** + **Vegetation height***Lumbricidae biomass (square root transformed) + Field size (offset; natural logarithm (x + 0.001) transformed)
Curlew pairs	Soil penetration resistance + Vegetation height (quadratic) + Rush cover (natural logarithm (x + 2.5) transformed) + **Lumbricidae biomass (square root transformed)** + **Vegetation height (quadratic)***Lumbricidae biomass (square root transformed) + Field size (offset; natural logarithm (x + 0.001) transformed)
Curlew pairs	Soil penetration resistance + Vegetation height (quadratic) + Rush cover (natural logarithm (x + 2.5) transformed) + **Diptera larvae biomass (natural logarithm (x + 0.001) transformed)** + **Vegetation height***Diptera larvae biomass (natural logarithm (x + 0.001) transformed) + Field size (offset; natural logarithm (x + 0.001) transformed)

Curlew pairs	Soil penetration resistance + Vegetation height (quadratic) + Rush cover (natural logarithm (x + 2.5) transformed) + **Diptera larvae biomass (natural logarithm (x + 0.001) transformed)** + **Vegetation height (quadratic)***Diptera larvae biomass (natural logarithm (x + 0.001) transformed) + Field size (offset; natural logarithm (x + 0.001) transformed)
Curlew pairs	Soil penetration resistance + Vegetation height (quadratic) + Rush cover (natural logarithm (x + 2.5) transformed) + **Gastropoda biomass (natural logarithm (x + 0.001) transformed)** + **Vegetation height***Gastropoda biomass (natural logarithm (x + 0.001) transformed) + Field size (offset; natural logarithm (x + 0.001) transformed)
Curlew pairs	Soil penetration resistance + Vegetation height (quadratic) + Rush cover (natural logarithm (x + 2.5) transformed) + **Gastropoda biomass (natural logarithm (x + 0.001) transformed)** + **Vegetation height (quadratic)***Gastropoda biomass (natural logarithm (x + 0.001) transformed) + Field size (offset; natural logarithm (x + 0.001) transformed)
Curlew pairs	Soil penetration resistance + Vegetation height (quadratic) + Rush cover (natural logarithm (x + 2.5) transformed) + **Other key invertebrate biomass (natural logarithm (x + 0.001) transformed)** + **Vegetation height***Other key dietary components biomass (natural logarithm (x + 0.001) transformed) + Field size (offset; natural logarithm (x + 0.001) transformed)
Curlew pairs	Soil penetration resistance + Vegetation height (quadratic) + Rush cover (natural logarithm (x + 2.5) transformed) + **Other key invertebrate biomass (natural logarithm (x + 0.001) transformed)** + **Vegetation height (quadratic)***Other key dietary components biomass (natural logarithm (x + 0.001) transformed) + Field size (offset; natural logarithm (x + 0.001) transformed)

Table B.11. Results of main effects plus interaction term generalised linear models (GLMs) investigating the influence of invertebrate biomass on wader density, as well as environmental variables that could influence waders through invertebrate accessibility and breeding habitat selection. Parameter estimates with standard errors (L = linear term; Q = quadratic term when both are modelled), and χ^2 tests are presented, with significant effects highlighted with an asterisk. Rush cover is natural logarithm (x + 2.5) transformed. Invertebrate biomass metrics are transformed (see Table B.10 for details). Akaike information criterion value corrected for small sample sizes (AICc) and McFadden's pseudo-R^2 are presented for each GLM. The lapwing pairs models use environmental data from the early breeding season only and invertebrate biomass data from the vegetation and shallow soil spatial layer only. The snipe and curlew pairs models use environmental data from the overall breeding season and invertebrate biomass data from all sub-samples combined.

Response variable	**Invertebrate biomass metric**	**Model R^2**	**AICc**	**Predictor variables**					
				Soil penetration resistance	*Vegetation height*	*Rush cover*	*Invertebrate biomass*	*Soil penetration resistance x invertebrate biomass*	*Vegetation height x invertebrate biomass*
Snipe pairs	Other key invertebrate biomass	0.180	93.628	0.452 ±0.893 $\chi^2 = 6.346$ $P = 0.012$ *	L 3.749 ±3.301 Q 0.916 ±2.637 $\chi^2 = 3.657$ $P = 0.161$	1.265 ±0.483 $\chi^2 = 9.743$ $P = 0.002$ *	L -3.778 ±9.599 Q -7.514 ±3.557 $\chi^2 = 6.788$ $P = 0.034$ *	-0.003 ±0.188 $\chi^2 = 0.002$ $P = 0.989$	
Snipe pairs	Other key invertebrate biomass	0.181	96.388	0.535 ±0.302 $\chi^2 = 6.346$ $P = 0.012$ *	L 3.857 ±3.364 Q 0.897 ±2.672 $\chi^2 = 3.721$ $P = 0.156$	1.294 ±0.498 $\chi^2 = 9.830$ $P = 0.002$ *	L -3.675 ±9.722 Q -11.723 ±14.653 $\chi^2 = 9.743$ $P = 0.008$ *	L -0.062 ±1.644 Q 0.741 ±2.462 $\chi^2 = 0.089$ $P = 0.956$	
Curlew pairs	Total invertebrate dietary biomass	0.219	86.442	0.015 ±0.138 $\chi^2 = 0.012$ $P = 0.914$	L -4.338 ±6.834 Q -5.029 ±4.836 $\chi^2 = 7.491$ $P = 0.024$ *	-0.113 ±0.335 $\chi^2 = 0.112$ $P = 0.738$	-0.800 ±2.078 $\chi^2 = 6.036$ $P = 0.014$ *		-0.070 ±0.149 $\chi^2 = 0.236$ $P = 0.627$
Curlew pairs	Total invertebrate dietary biomass	0.250	86.336	0.005 ±0.138 $\chi^2 = 0.001$ $P = 0.972$	L 1.124 ±8.638 Q -0.109 ±5.265 $\chi^2 = 8.708$ $P = 0.013$ *	0.041 ±0.370 $\chi^2 = 0.013$ $P = 0.911$	-4.845 ±2.784 $\chi^2 = 6.036$ $P = 0.014$ *		L -31.158 ±26.617 Q -14.987 ±10.606 $\chi^2 = 3.073$ $P = 0.215$
Curlew pairs	Lumbricidae biomass	0.214	86.909	0.009 ±0.137 $\chi^2 = 0.004$	L -7.103 ±5.617 Q -3.367 ±4.232	-0.170 ±0.333 $\chi^2 = 0.252$	-1.576 ±1.814 $\chi^2 = 5.805$		-0.003 ±0.120 $\chi^2 = 0.001$

				P = 0.949	χ^2 = 7.648 *P* = 0.022 *	*P* = 0.616	*P* = 0.016 *		*P* = 0.979
Curlew pairs	Lumbricidae biomass	0.242	87.097	-0.020 ±0.137 χ^2 = 0.021 *P* = 0.886	L -6.357 ±5.398 Q 1.503 ±4.952 χ^2 = 8.393 *P* = 0.015 *	-0.076 ±0.358 χ^2 = 0.045 *P* = 0.833	-3.437 ±1.896 χ^2 = 5.805 *P* = 0.016 *		L -15.086 ±17.361 Q -12.267 ±8.807 χ^2 = 2.543 *P* = 0.280
Curlew pairs	Diptera larvae biomass	0.182	89.728	-0.141 ±0.153 χ^2 = 0.879 *P* = 0.349	L -17.016 ±10.558 Q -10.030 ±3.924 χ^2 = 10.719 *P* = 0.005 *	-0.369 ±0.392 χ^2 = 0.879 *P* = 0.348	0.667 ±0.437 χ^2 = 0.087 *P* = 0.768		-0.043 ±0.027 χ^2 = 2.900 *P* = 0.089
Curlew pairs	Diptera larvae biomass	0.182	92.457	-0.141 ±0.154 χ^2 = 0.876 *P* = 0.349	L -17.170 ±11.458 Q -10.261 ±7.685 χ^2 = 8.377 *P* = 0.015 *	-0.369 ±0.392 χ^2 = 0.880 *P* = 0.348	-0.060 ±0.267 χ^2 = 0.087 *P* = 0.768		L -3.217 ±2.057 Q -0.058 ±1.655 χ^2 = 2.901 *P* = 0.234
Curlew pairs	Gastropoda biomass	0.199	88.188	-0.087 ±0.142 χ^2 = 0.391 *P* = 0.532	L -21.921 ±11.689 Q -6.597 ±2.876 χ^2 = 11.434 *P* = 0.003 *	-0.225 ±0.311 χ^2 = 0.510 *P* = 0.298	0.418 ±0.348 χ^2 = 1.085 *P* = 0.475		-0.046 ±0.026 χ^2 = 3.442 *P* = 0.064
Curlew pairs	Gastropoda biomass	0.202	90.676	-0.092 ±0.142 χ^2 = 0.440 *P* = 0.507	L -28.635 ±18.992 Q -13.031 ±13.736 χ^2 = 8.070 *P* = 0.018 *	-0.203 ±0.313 χ^2 = 0.409 *P* = 0.523	-0.466 ±0.324 χ^2 = 1.085 *P* = 0.298		L -4.438 ±2.989 Q -1.105 ±2.277 χ^2 = 3.685 *P* = 0.159
Curlew pairs	Other key invertebrate biomass	0.155	92.171	-0.098 ±0.141 χ^2 = 0.504 *P* = 0.478	L -9.943 ±10.570 Q -5.853 ±2.696 χ^2 = 8.771 *P* = 0.013 *	-0.419 ±0.266 χ^2 = 2.412 *P* = 0.120	0.218 ±0.380 χ^2 = 0.016 *P* = 0.900		-0.022 ±0.030 χ^2 = 0.528 *P* = 0.467
Curlew pairs	Other key invertebrate biomass	0.156	94.792	-0.096 ±0.142 χ^2 = 0.473 *P* = 0.492	L -5.876 ±15.887 Q -1.583 ±13.007 χ^2 = 8.268 *P* = 0.016 *	-0.393 ±0.277 χ^2 = 1.987 *P* = 0.159	-0.065 ±0.367 χ^2 = 0.016 *P* = 0.900		L -0.728 ±3.538 Q 0.920 ±2.790 χ^2 = 0.638 *P* = 0.727

Table B.12. Results of the main effects only generalised linear models (GLMs) investigating the influence of habitat type on wader density. Parameter estimates with standard errors, and χ^2 tests are presented. McFadden's pseudo-R^2 are presented for each GLM. Improved pasture is the reference level. HM = heather moor, SI = semi-improved pasture, UP = unimproved pasture, WM = white moor. The snipe pairs poisson (log link) GLM was overdispersed, so the model was run as a quasipoisson (log link) GLM; thus McFadden's pseudo-R^2 was not quantifiable.

Response variable	**Model R^2**	**Habitat type predictor variable**					
		β ± SE				X^2	*P* value
		HM	*SI*	*UP*	*WM*		
Lapwing pairs	0.033	-15.855 ± 1732.780	-0.320 ± 0.671	-0.287 ± 0.866	-1.014 ± 1.118	2.331	0.675
Snipe pairs		-15.162 ± 2819.262	1.066 ± 1.260	1.505 ± 1.329	0.373 ± 1.627	2.729	0.604
Curlew pairs	0.087	0.060 ± 1.118	-0.320 ± 0.671	0.119 ± 0.764	0.932 ± 0.627	4.761	0.313

Table B.13. Mean ± standard error (SE) and range for environmental variables in the early breeding season and late breeding season. The 'measure' column shows the spatial scale at which each variable was measured (S = variable measured at the within-field sampling points during invertebrate surveys; F = variable recorded at the field level during breeding wader surveys).

Environmental variable	**Measure**	**Mean ± SE; range**	
		Early breeding season	*Late breeding season*
Soil moisture (%)	S	78.422 ± 1.115; 23.900 to 92.500	41.755 ± 2.160; 8.967 to 92.500
Soil penetration resistance (kgF)	S	4.950 ± 0.135; 0.467 to 16.900	7.287 ± 0.277; 1.900 to 20.000
Soil pH	S	4.798 ± 0.062; 2.730 to 7.030	4.717 ± 0.079; 2.930 to 6.610
Vegetation height (cm)	S	14.091 ± 0.912; 0.000 to 75.000	23.400 ± 1.511; 0.000 to 80.000
Vegetation density	S	0.409 ± 0.035; 0.000 to 3.000	0.587 ± 0.045; 0.000 to 2.000

Table B.14. Mean ± standard error (SE) and range for environmental variables in each of the five upland habitat types that were recorded. Data are given for the early breeding season and late breeding season separately.

Habitat type	**Breeding season**	**Environmental variable (mean ± SE; range)**				
		Soil moisture (%)	*Soil penetration resistance (kgF)*	*Soil pH*	*Vegetation height (cm)*	*Vegetation density*
Improved pasture	Early	75.129 ± 1,672; 42.033 to 92.500	5.368 ± 0.202; 0.900 to 8.933	5.585 ± 0.054; 4.670 to 6.630	6.053 ± 0.637; 0.000 to 30.000	0.158 ± 0.046; 0 to 2
	Late	26.912 ± 1.744; 9.067 to 49.433	8.433 ± 0.384; 4.333 to 13.600	5.757 ± 0.073; 5.030 to 6.610	14.286 ± 2.774; 0.000 to 80.000	0.486 ± 0.086; 0 to 1
Semi-improved pasture	Early	70.492 ± 2.178; 23.900 to 92.500	4.980 ± 0.206; 0.567 to 11.700	5.135 ± 0.075; 3.240 to 7.030	12.570 ± 1.415; 0.000 to 65.000	0.355 to 0.055; 0 to 3
	Late	29.726 ± 2.684; 8.967 to 88.300	7.877 ± 0.453; 2.700 to 14.300	4.952 ± 0.087; 3.290 to 6.600	23.482 ± 2.636; 0.000 to 70.000	0.536 ± 0.076; 0 to 2
Unimproved pasture	Early	87.842 ± 2.043; 31.200 to 92.500	5.056 ± 0.505; 1.267 to 16.900	4.367 ± 0.154; 3.120 to 7.000	20.366 ± 2.713; 0.000 to 70.000	0.537 ± 0.086; 0 to 2
	Late	56.116 ± 5.444; 14.733 to 92.500	6.211 ± 0.945; 2.133 to 20.000	4.365 ± 0.138; 3.270 to 5.270	31.250 ± 3.738; 5.000 to 65.000	0.750 ± 0.090; 0 to 1
White moor	Early	90.085 ± 1.569; 27.967 to 92.500	4.402 ± 0.390; 0.467 to 12.067	3.723 ± 0.102; 2.730 to 5.480	18.691 ± 2.872; 0.000 to 75.000	0.595 ± 0.084; 0 to 2
	Late	50.000 ± 6.402; 21.867 to 92.500	6.944 ± 0.781; 3.100 to 13.033	3.920 ± 0.105; 3.320 to 4.950	24.000 ± 5.350; 5.000 to 70.000	0.600 ± 0.131; 0 to 1
Heather moor	Early	89.567 ± 1.719; 59.000 to 92.500	4.170 ± 0.386; 2.100 to 7.600	3.154 ± 0.075; 2.790 to 4.470	30.250 ± 3.067; 0.000 to 55.000	1.000 ± 0.205; 0 to 3
	Late	77.993 ± 3.777; 45.600 to 92.500	5.180 ± 0.516; 1.900 to 8.533	3.257 ± 0.048; 2.930 to 3.640	29.250 ± 2.092; 10.000 to 45.000	0.700 ± 0.147; 0 to 2

Table B.15. Mean ± standard error (SE), median, and range for the five invertebrate biomass metrics in each of the five upland habitat types that were recorded. Data are given for the early breeding season and late breeding season separately.

Habitat type	**Breeding season**	**Invertebrate biomass metric (mean ± SE; median; range)**				
		Total invertebrate dietary components	*Earthworms*	*Diptera larvae*	*Gastropoda*	*Other key invertebrate groups*
Improved pasture	Early	2.282 ± 0.167; 2.048; 0.115 to 7.365	2.128 ± 0.167; 1.854; 0.115 to 7.365	0.096 ± 0.017; 0.003; 0.000 to 0.606	0.036 ± 0.012; 0.000; 0.000 to 0.559	0.020 ± 0.007; 0.000; 0.000 to 0.394
	Late	0.830 ± 0.116; 0.700; 0.011 to 3.395	0.666 ± 0.107; 0.499; 0.000 to 2.838	0.121 ± 0.033; 0.000; 0.000 to 0.667	0.027 ± 0.016; 0.000; 0.000 to 0.500	0.012 ± 0.004; 0.000; 0.000 to 0.104
Semi-improved pasture	Early	1.664 ± 0.126; 1.544; 0.000 to 7.983	1.447 ± 0.114; 1.205; 0.000 to 6.909	0.130 ± 0.023; 0.003; 0.000 to 1.339	0.063 ± 0.014; 0.000; 0.000 to 0.931	0.014 ± 0.003; 0.000; 0.000 to 0.203
	Late	0.688 ± 0.091; 0.561; 0.000 to 3.270	0.418 ± 0.061; 0.240; 0.000 to 2.063	0.173 ± 0.045; 0.000; 0.000 to 1.587	0.051 ± 0.015; 0.000; 0.000 to 0.446	0.028 ± 0.010; 0.003; 0.000 to 0.487
Unimproved pasture	Early	0.291 ± 0.063; 0.143; 0.000 to 1.866	0.223 ± 0.058; 0.000; 0.000 to 1.757	0.035 ± 0.011; 0.000; 0.000 to 0.375	0.015 ± 0.005; 0.000; 0.000 to 0.119	0.015 ± 0.008; 0.000; 0.000 to 0.303
	Late	0.531 ± 0.173; 0.105; 0.000 to 3.630	0.334 ± 0.124; 0.047; 0.000 to 2.590	0.027 ± 0.020; 0.000; 0.000 to 0.455	0.118 ± 0.061; 0.000; 0.000 1.247	0.021 ± 0.013; 0.000; 0.000 to 0.297
White moor	Early	0.117 ± 0.031; 0.020; 0.000 to 0.792	0.079 ± 0.027; 0.000; 0.000 to 0.741	0.021 ± 0.009; 0.000; 0.000 to 0.266	0.005 ± 0.002; 0.000; 0.000 to 0.046	0.009 ± 0.003; 0.000; 0.000 to 0.081
	Late	0.181 ± 0.047; 0.157; 0.000 to 0.568	0.053 ± 0.027; 0.000; 0.000 to 0.338	0.057 ± 0.021; 0.000; 0.000 to 0.258	0.003 ± 0.002; 0.000; 0.000 to 0.028	0.067 ± 0.031; 0.010; 0.000 to 0.377
Heather moor	Early	0.031 ± 0.011; 0.000; 0.000 to 0.175	0.008 ± 0.004; 0.000; 0.000 to 0.060	0.000 ± 0.000; 0.000; 0.000 to 0.000	0.003 ± 0.002; 0.000; 0.000 to 0.030	0.018 ± 0.010; 0.000; 0.000 to 0.164
	Late	0.061 ± 0.026; 0.008; 0.000 to 0.455	0.000 ± 0.000; 0.000; 0.000 to 0.000	0.003 ± 0.003; 0.000; 0.000 to 0.059	0.000 ± 0.000; 0.000; 0.000 to 0.000	0.017 ± 0.008; 0.000; 0.000 to 0.150

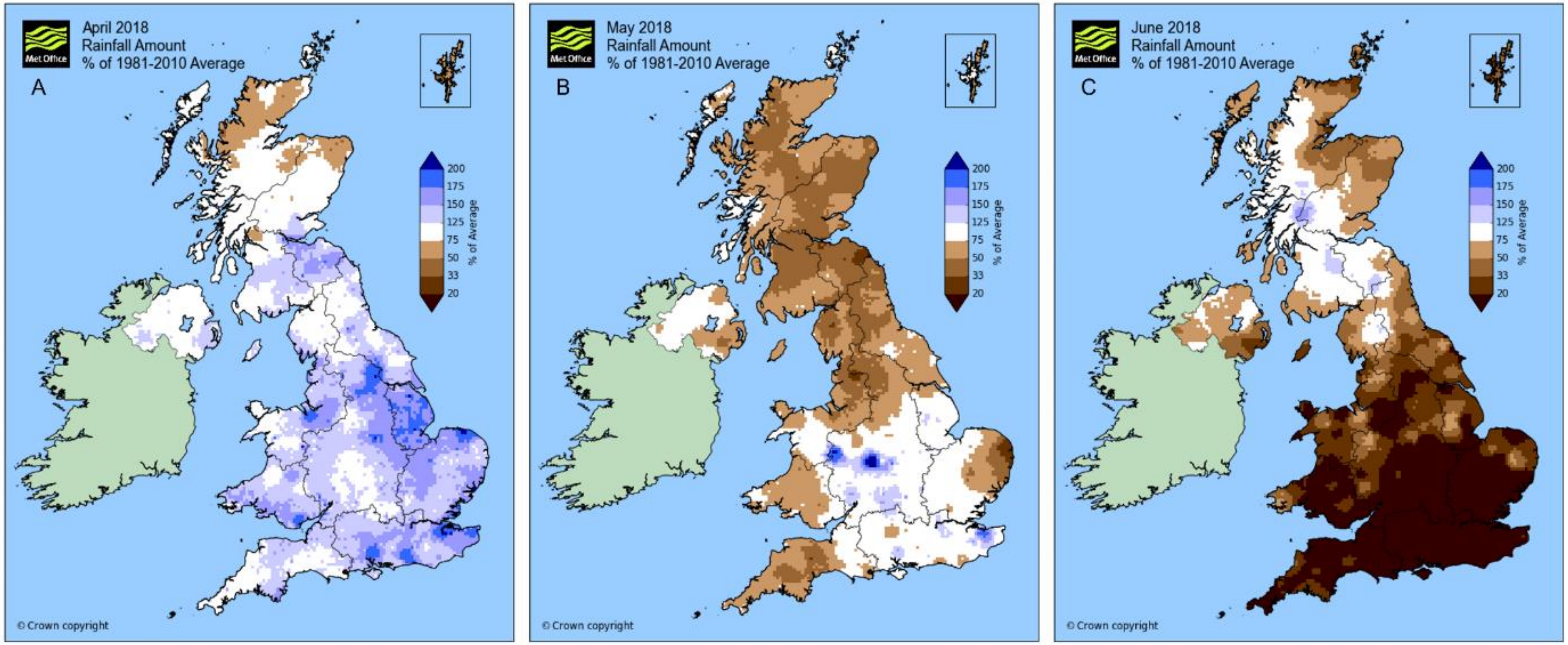

Figure B.1. UK maps showing, as a percentage of the 1981-2010 average, typical amounts of rainfall in April 2018 (A) but considerably lower rainfall in May 2018 (B) and June 2018 (C). Maps are modified from https://www.metoffice.gov.uk/research/climate/maps-and-data/uk-actual-and-anomaly-maps.

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Appendix C

Supporting information for Chapter 3

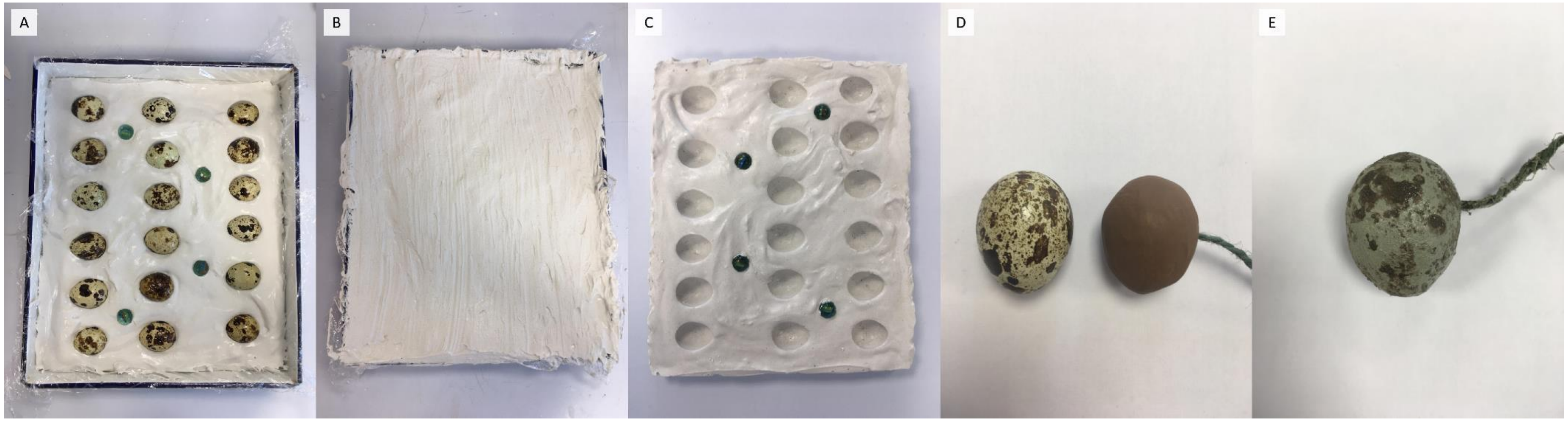

Figure C.1. A standardised method was employed to produce large quantities of plasticine eggs for the artificial nest experiment. To mimic the real Japanese quail *Coturnix japonica* eggs in size and shape, moulds of quail eggs were formed based on the Martin, Dueser, and Moncrief (2010) method. A clingfilm-lined baking tray was filled to half its depth with plaster of Paris and the mixture levelled. Petroleum jelly-coated, hard-boiled quail eggs were pressed, on their long axis, into the plaster of Paris at approximately 1 cm intervals until half of the egg was submerged (A). Following a 1-hour drying period, the top of the plaster of Paris and quail eggs were coated in a thin layer of petroleum jelly. The tray was then filled with more plaster of Paris so that the eggs were submerged (B). Following a second 1-hour drying period, the plaster of Paris was removed from the tray and the two halves prised apart. The quail eggs were extracted, and excess petroleum wiped off creating two half-quail egg moulds (C). Multiple quail eggs were used to produce the moulds which generated slight variation in mould size. To produce the plasticine eggs, brown or terracotta Newplast modelling clay (Newclay Product Ltd, Devon, England) was heated on a heating block (50-70 °C) until malleable. Pressing 11.5 g (±0.3 g) of plasticine into a clingfilm-lined half-quail egg mould produced half of a plasticine quail egg. A double-knotted end of 30 cm dark green garden twine was placed into the centre of the flat side of one of the half plasticine eggs. Two half plasticine eggs were then placed together and joined by smudging the plasticine along the seam (D – left egg real is a quail egg; right egg is a plasticine egg). Once plasticine eggs had cooled and hardened, each egg was spray-coated in a layer of tan-coloured PlastiDip® (PlastiDip UK Ltd, Hampshire, England), based on Purger *et al.* (2012), to minimise the scent of the plasticine which could influence predator activity at the nests, and mimic the base colour of real quail and wader eggs. Following a 4-hour drying period, speckles and spots were applied to the coated plasticine eggs with a small brush and brown-coloured PlastiDip®, to improve the mimicry of real eggs (E).

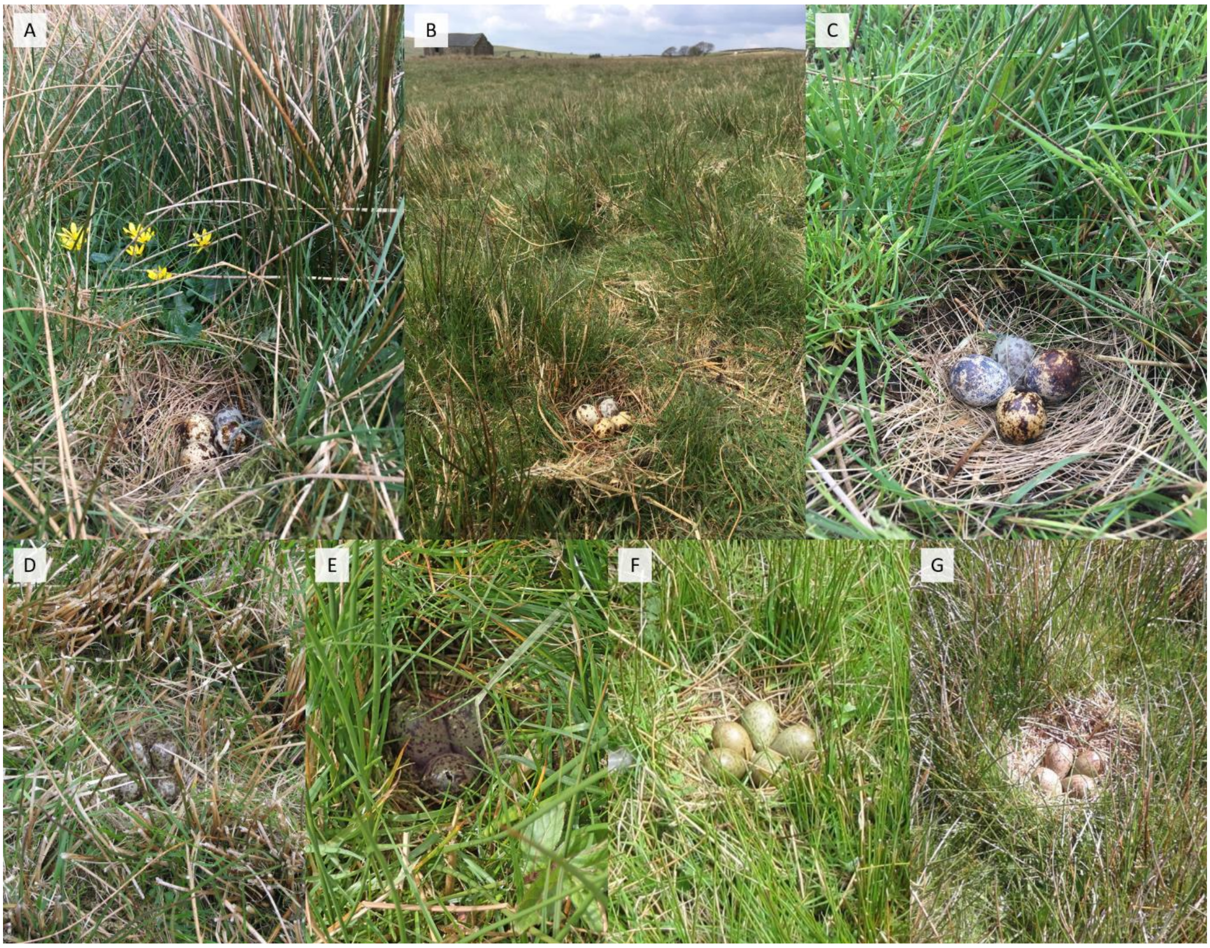

Figure C.2. A) An example artificial wader nest located in a patch of uncut rush demonstrating nest placement (the egg furthest from the camera is the plasticine egg). B) An example artificial wader nest located in a patch of cut rush demonstrating nest placement (the egg furthest from the camera is the plasticine egg). C) A close-up of the contents of an artificial wader nest (the egg furthest from the camera is the plasticine egg). D) A real snipe nest found in an area of cut rush in the SWP in 2015 (photo credit: Mark Eddowes). E) A close-up photograph of a real snipe nest in an area of cut rush at Geltsdale in 2019 (photo credit: Mike Shurmer, RSPB). F) A real curlew nest found in an area of cut rush in the SWP in 2015 (photo credit: Mark Eddowes). G) A real curlew nest found in an area of uncut rush in the SWP in 2018 (photo credit: Jonathan Groom, Staffordshire Wildlife Trust).

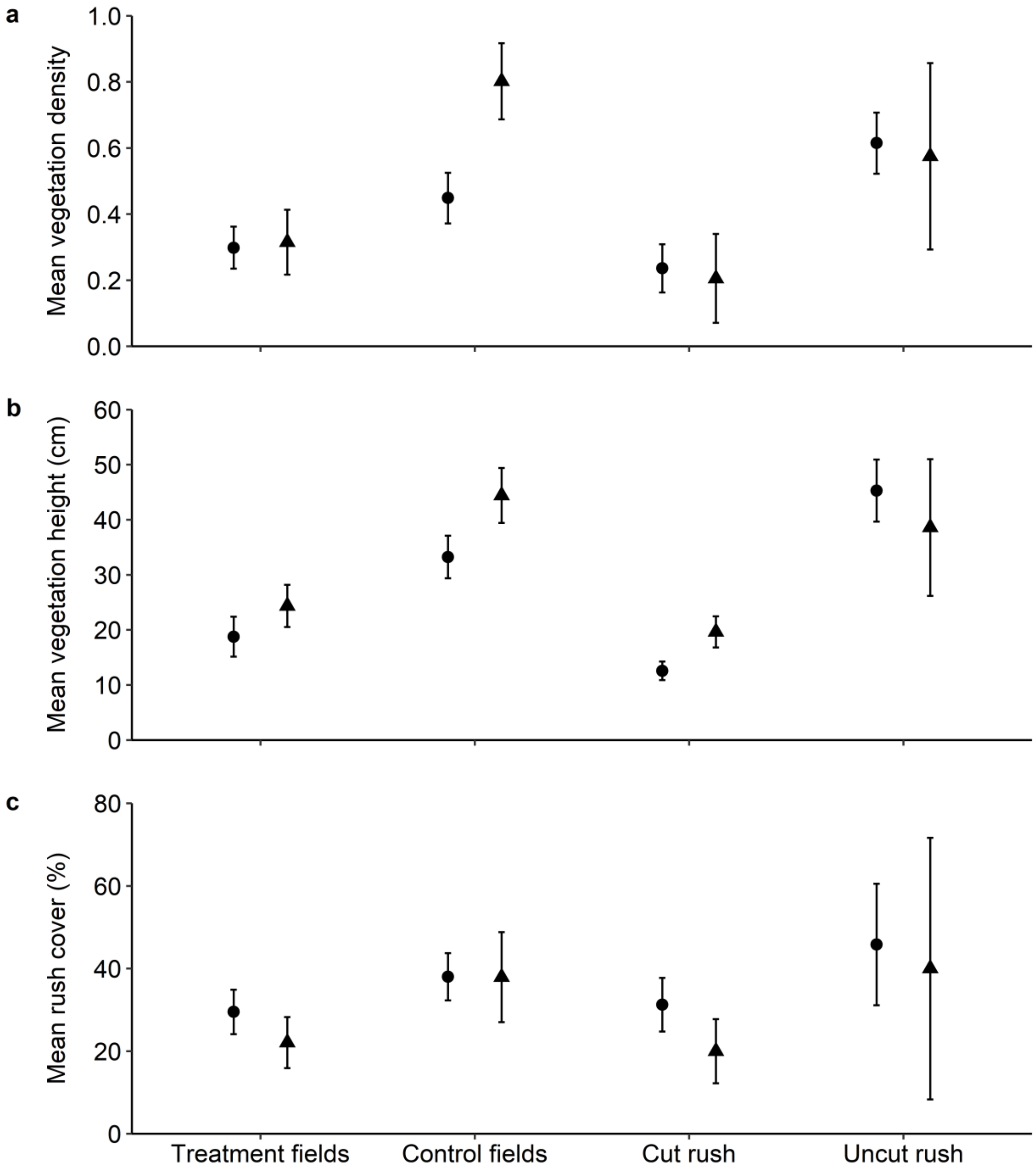

Figure C.3. Mean observed (a) vegetation density, (b) vegetation height and (c) rush cover surrounding artificial nests located in i) treatment and control fields, and ii) cut and uncut rush patches within treatment fields >4 ha during the early (circles) and late (triangles) breeding seasons. Points represent mean values and vertical lines represent 95% confidence intervals.

Table C.1. Summaries of the Entry Level Stewardship agri-environment scheme prescriptions (adapted from Natural England (2012)) that were followed or emulated in the treatment fields. Descriptions relate to rush management aspects of the prescriptions only (cutting regime and grazing).

Prescription code	Prescription name	Pre-requisites	Description of rush management
EK4	Management of rush pastures	• Outside SDAs • Below the Moorland Line • At least one-third of the field area covered by rush	• Cut no more than one-third of the total area of rush in rotation • Cutting must not be conducted between 15 March and 31 July • Graze the aftermath, ideally with cattle • If there is rapid regrowth of rush, cut again within 8 weeks of the first cut (not between 1 April and 31 July) • Repeat management each year
EL4	Management of rush pastures in SDAs	• Inside SDAs • Below the Moorland Line • Above the Moorland Line on parcels <15 ha • At least one-third of the field area covered by rush	• Cut no more than one-third of the total area of rush in rotation • Cutting must not be conducted between 1 April and 31 July • Graze the aftermath, ideally with cattle • If there is rapid regrowth of rush, cut again within 8 weeks of the first cut (not between 1 April and 31 July) • Repeat management each year

SDA = Severely Disadvantaged Area; LU = Livestock Unit; ha = hectare

Table C.2. Mean and 95% confidence intervals (CIs) for observed values of environmental variables in treatment and control fields in the SWP and at Geltsdale. Data are given for four different groups (treatment fields, control fields, SWP and Geltsdale) in the early and late periods of the breeding season. Bootstrap CIs were calculated for distance to nearest field boundary (m) and field size (ha), corvid abundance, and corvid density (corvid abundance divided by field size to take field size into account) because the data were positively skewed.

Response variable	**Group**	**Mean (95% CIs)**	
		Early breeding season	Late breeding season
Vegetation density	Treatment fields	0.30 (0.24 to 0.36)	0.31 (0.22 to 0.41)
	Control fields	0.45 (0.37 to 0.52)	0.80 (0.69 to 0.92)
	SWP	0.37 (0.30 to 0.44)	-
	Geltsdale	0.37 (0.30 to 0.45)	-
Vegetation height (cm)	Treatment fields	18.76 (15.14 to 22.39)	24.35 (20.51 to 28.19)
	Control fields	33.24 (29.39 to 37.10)	44.41 (39.41 to 49.40)
	SWP	22.69 (18.45 to 26.92)	-
	Geltsdale	28.48 (24.47 to 32.50)	-
Rush cover (%)	Treatment fields	29.52 (24.13 to 34.90)	22.07 (15.90 to 28.24)
	Control fields	38.04 (32.32 to 43.75)	37.92 (27.01 to 48.82)
	SWP	34.74 (28.14 to 41.35)	-
	Geltsdale	32.42 (27.84 to 36.99)	-
Woodland distance (km)	Treatment fields	0.50 (0.44 to 0.56)	0.62 (0.57 to 0.68)
	Control fields	0.54 (0.47 to 0.61)	0.72 (0.63 to 0.81)
	SWP	0.68 (0.63 to 0.73)	-
	Geltsdale	0.36 (0.31 to 0.41)	-
Boundary distance (m)	Treatment fields	55.31 (49.64 to 62.31)	47.55 (42.42 to 53.47)
	Control fields	48.56 (43.73 to 53.97)	43.38 (37.64 to 49.58)
	SWP	45.79 (41.73 to 50.28)	-
	Geltsdale	58.21 (52.04 to 65.66)	-
Field size (ha)	Treatment fields	7.18 (4.70 to 13.50)	-
	Control fields	4.68 (3.16 to 7.56)	-
	SWP	3.00 (2.30 to 3.95)	-
	Geltsdale	9.92 (6.74 to 16.00)	-
Corvid abundance	Treatment fields	2.98 (2.19 to 3.76)	2.25 (0.96 to 5.54)
	Control fields	3.50 (2.05 to 5.84)	1.15 (0.62 to 1.62)
	SWP	2.44 (1.64 to 3.36)	-
	Geltsdale	4.36 (2.69 to 7.00)	-
Corvid density	Treatment fields	0.80 (0.52 to 1.29)	0.91 (0.30 to 2.53)
	Control fields	0.97 (0.58 to 1.53)	0.73 (0.35 to 1.41)
	SWP	1.10 (0.71 to 1.60)	-
	Geltsdale	0.59 (0.37 to 1.16)	-

Table C.3. The number of treatment and control fields of <4 ha and >4 ha in area surveyed in each region during the early and late breeding seasons. The number of artificial nests deployed in each group of fields (and used in analyses) are given, with the total number prior to the exclusion of trampled nests shown in parentheses.

Breading season stage	Region	Field type	Field size category	Number of fields	Number of artificial nests
Early	SWP	Treatment	<4 ha	7	12 (12)
			>4 ha	5	19 (20)
		Control	<4 ha	11	19 (20)
			>4 ha	2	8 (8)
	Geltsdale	Treatment	<4 ha	1	2 (2)
			>4 ha	8	29 (32)
		Control	<4 ha	3	6 (6)
			>4 ha	6	23 (24)
	Total			43	118 (124)
Late	SWP	Treatment	<4 ha	7	10 (12)
			>4 ha	5	19 (20)
		Control	<4 ha	11	17 (20)
			>4 ha	2	7 (8)
	Total			25	53 (60)

Table C.4. The number of artificial nests deployed in cut and uncut rush patches (and used in analyses) within treatment fields >4 ha during both the early and late breeding seasons within each region. The total number of artificial nests deployed prior to the exclusion of trampled nests are shown in parentheses.

Breading season stage	Region	Rush type	Number of artificial nests
Early	SWP	Cut	14 (15)
		Uncut	5 (5)
	Geltsdale	Cut	22 (24)
		Uncut	7 (8)
	Total		48 (52)
Late	SWP	Cut	14 (15)
		Uncut	5 (5)
	Total		19 (20)

Table C.5. The questionnaire used for the semi-structured interviews with land managers and owners to obtain information on the intensity of any predator control conducted across each land holding in which our focal fields were located.

Date:
Time:
Respondent name:
Farm name:
Field ID codes:
Consent
Do you understand the project?
Do you give consent for the data to be provided to the Work for Waders project within the South West Peak Landscape Partnership Scheme?
Are you happy to take part and answer some questions about predator control on your farm?
Predator control questions
1) Were predator control measures conducted on your farm in 2019? If the answer is yes, go to question 2. If the answer is no, the interview is complete.
2) When was predator control started in 2019?
3) For which of the following species was predator control attempted on your farm: a) Corvids b) Red foxes c) Mustelids (e.g. stoats, weasels)
4) How intensive was the predator control effort for each group of species? *Corvids* a) How regular was predator control conducted? b) Over what area was the predator control conducted (e.g. the whole farm, around the farmhouse)? c) Was more effort involved in controlling corvids compared to red foxes and mustelids? *Red foxes* a) How regular was predator control conducted? b) Over what area was the predator control conducted (e.g. the whole farm, around the farmhouse)? c) Was more effort involved in controlling red foxes compared to corvids and mustelids? *Mustelids* a) How regular was predator control conducted? b) Over what area was the predator control conducted (e.g. the whole farm, around the farmhouse)? c) Was more effort involved in control corvids compared to corvids and red foxes?

Table C.6. Descriptions, ranges, and means (95% confidence intervals (CIs)) of the predictor variables used in each model of daily nest predation rate (models are described in Table 3.1). Data relate to untransformed variables, although predictor variables with skewed distributions (boundary distance, field size, and corvid abundance) were transformed prior to inclusion in the models to reduce the influence of outliers.

Predictor variable	**Type**	**Units**	**Description**	**Range; mean (95% CI)**	
				Models i, iii, and iv	Model ii
Treatment	Categorical		2 categories: treatment field; control field	Treatment field n = 21 fields; control field n = 22 fields [included in model i only]	Not included
Cut or uncut rush	Categorical		2 categories: cut rush patch; uncut rush patch; treatment fields >4 ha only	Not included	Cut rush patch n = 50 fields; uncut rush patch n = 17 fields
Vegetation density	Continuous		Mean density of vegetation surrounding each artificial nest; four scrape edge and four 1 m distance measures averaged	0.000-1.375; 0.421 (95% CI 0.374 to 0.468) [included in model iii only]	Not included
Vegetation height	Continuous	cm	Mean height of vegetation surrounding each artificial nest; four scrape edge and four 1 m distance measures averaged	3.625-77.500; 28.051 (95% CI 25.635 to 30.468) [included in model iv only]	Not included
Rush cover	Continuous	%	Visual estimate of rush cover (to the nearest 5%) within a 5 m radius of each artificial nest	5-90; 32.222 (95% CI 28.920 to 35.525) [included in models iii and iv only]	Not included
Woodland distance	Continuous	km	Straight-line distance to the nearest block of woodland from each artificial nest	0.069-1.052; 0.564 (95% CI 0.527 to 0.600)	0.215-0.818; 0.536 (95% CI 0.489 to 0.583)
Boundary distance	Continuous	m	Straight-line distance to the nearest field boundary from each artificial nest	6.626-134.748; 50.109 (95% CI 46.911 to 53.306)	25.213-134.748; 59.690 (95% CI 54.214 to 65.167)

Deployment date	Continuous		Day after 1 January when each artificial nest was deployed	91-151; 121.912 (95% CI 118.551 to 125.274)	91-151; 122.970 (95% CI 118.008 to 127.932)
Field size	Continuous	ha	Area of the nest field	0.403-40.638; 5.898 (95% CI 3.704 to 8.092)	Not included
Predator control	Categorical		2 categories: regular or negligible/no	Regular n = 12 fields; negligible/no n = 31 fields	Not included
Corvid abundance	Continuous		Mean number of corvids observed over 2 visits during each breeding season stage	0-16; 2.669 (95% CI 1.883 to 3.455)	Not included
Region	Categorical		2 categories: SWP; Geltsdale	SWP n = 25 fields; Geltsdale n = 18 fields	SWP n = 5 fields; Geltsdale n = 8 fields
Field identity	Categorical		Identity code for the nest field	n = 43 unique fields	n = 13 unique fields

Table C.7. Modelling environmental variables of treatment and control fields whilst taking region (SWP or Geltsdale) into account. Separate models of each variable were constructed for the early and late breeding seasons except for field size which did not differ throughout the breeding season. Generalised linear mixed effects models (GLMMs) were fitted in the lme4 package (Bates *et al.*, 2015) using restricted maximum likelihood, Gaussian error structure and identity link, with field identity as a random factor for environmental variables with multiple measures per field. The exception was vegetation height in the early breeding season, modelled using a generalised linear model (GLM; restricted maximum likelihood) with Gaussian error structure and identity link as singularity prevented a mixed model. Field size and corvid density (per ha) were also modelled using GLMs (restricted maximum likelihood) with Gaussian error structure and identity link. For all models, we included treatment (treatment or control field) and region (SWP or Geltsdale) as predictor variables. For late breeding season models, region was not included because data were only available from the SWP. Profile (P) 95% confidence intervals (CIs) are reported in all cases, alongside bootstrap (BS) CIs when using GLMMs. Type II F tests are presented, corrected using Type II Wald F tests with Kenward-Roger degrees of freedom for GLMMs. Significant effects ($P < 0.05$) are in bold. Control fields are the reference level for treatment; Geltsdale is the reference level for region. Skewed response variables were transformed to meet models' assumptions.

Response variable	**Predictor variable**	**Early breeding season**				**Late breeding season**			
		Estimate (95% CIs)	*Sample size*	*F statistic*	*P*	*Estimate (95% CIs)*	*Sample size*	*F statistic*	*P*
Vegetation density	Treatment	**-0.154 (P -0.273 to -0.036; BS -0.284 to -0.035)**	**118**	**6.296**	**0.017**	**-0.486 (P -0.632 to -0.336; BS -0.647 to -0.330)**	**53**	**39.590**	**5.638^{-6}**
	Region	0.001 (P -0.117 to 0.120; BS -0.110 to 0.126)	118	0.0003	0.986	-	-	-	-
Vegetation height	Treatment	**-14.380 (P -19.481 to -9.279)**	**118**	**30.528**	**2.077^{-7}**	**-19.786 (P -26.096 to -13.288; BS -25.736 to -13.360)**	**53**	**35.584**	**9.678^{-6}**
	Region	**-5.542 (P -10.637 to -0.447)**	**118**	**4.544**	**0.035**	-	-	-	-
Rush cover	Treatment	**-11.722 (P -22.209 to -1.287; BS -22.185 to -2.312)**	**118**	**4.707**	**0.036**	-14.067 (P -27.676 to -0.266; BS -27.601 to -0.656)	53	3.986	0.060
	Region	0.111 (P -10.396 to 10.576; BS -10.587 to 10.568)	118	0.0004	0.984	-	-	-	-
Woodland distance	Treatment	-0.045 (P -0.164 to 0.074; BS -0.171 to 0.083)	118	0.540	0.467	-0.080 (P -0.255 to 0.095; BS -0.245 to 0.110)	53	0.791	0.384
	Region	**0.292 (P 0.172 to 0.412; BS 0.191 to 0.411)**	**118**	**22.114**	**3.084^{-5}**	-	-	-	-

Boundary distance (square root transformed)	Treatment	0.425 (P -0.242 to 1.091; BS -0.260 to 1.095)	118	1.508	0.227	0.312 (P -0.524 to 1.148; BS -0.556 to 1.091)	53	0.529	0.475
	Region	**-0.870 (P -1.541 to -0.203; BS -1.569 to -0.125)**	**118**	**6.299**	**0.017**	-	-	-	-
Field size (natural logarithm transformed)	Treatment	0.302 (P -0.193 to 0.797)	43	1.428	0.239	-	-	-	-
	Region	**-1.077 (P -1.579 to -0.576)**	**43**	**17.716**	**1.409^{-4}**	-	-	-	-
Corvid density (natural logarithm (x+1) transformed)	Treatment	-0.029 (P -0.304 to 0.246)	43	0.043	0.837	-0.006 (P -0.417 to 0.406)	25	0.001	0.979
	Region	0.218 (P -0.060 to 0.497)	43	2.359	0.132				

Table C.8. Modelling environmental variables of artificial nests located in uncut and cut rush patches within treatment fields >4 ha whilst taking region (SWP or Geltsdale) into account. Separate models of each variable were constructed for the early and late breeding seasons. Generalised linear mixed effects models (GLMMs) were fitted in the lme4 package (Bates *et al.*, 2015) using restricted maximum likelihood, Gaussian error structure and identity link, with field identity as a random factor. The exception was boundary distance in the late breeding season, modelled using a generalised linear model (GLM; restricted maximum likelihood) with Gaussian error structure and identity link as singularity prevented a mixed model. For all models, we included uncut or cut rush and region (SWP or Geltsdale) as predictor variables. For late breeding season models, region was not included because data were only available from the SWP. Profile (P) 95% confidence intervals (CIs) are reported in all cases, alongside bootstrap (BS) CIs when using GLMMs. Type II F tests are presented, corrected using Type II Wald F tests with Kenward-Roger degrees of freedom for GLMMs. Significant effects (*P* < 0.05) are in bold. Cut rush patches are the reference level for uncut or cut rush; Geltsdale is the reference level for region. Skewed response variables were transformed to meet models' assumptions.

Response variable	**Predictor variable**	**Early breeding season**				**Late breeding season**			
		Estimate (95% CIs)	*Sample size*	*F statistic*	*P*	*Estimate (95% CIs)*	*Sample size*	*F statistic*	*P*
Vegetation density	Uncut or cut rush	**0.376 (P 0.263 to 0.489; BS 0.254 to 0.484)**	**48**	**43.772**	**1.280^{-7}**	**0.370 (P 0.134 to 0.605; BS 0.121 to 0.599)**	**19**	**9.407**	**0.009**
	Region	0.053 (P -0.108 to 0.216; BS -0.138 to 0.217)	48	0.409	0.536	-	-	-	-
Vegetation height	Uncut or cut rush	**32.766 (P 29.900 to 35.653; BS 30.036 to 35.555)**	**48**	**508.826**	**$<2.000^{-16}$**	**19.033 (P 12.931 to 25.072; BS 13.210 to 25.144)**	**19**	**40.055**	**2.558^{-5}**
	Region	-5.267 (P -10.060 to -0.481; BS -10.584 to -0.491)	48	4.565	0.056	-	-	-	-
Rush cover	Uncut or cut rush	**14.571 (P 5.049 to 24.050; BS 5.547 to 23.649)**	**48**	**9.263**	**0.005**	**20.132 (P 2.650 to 37.350; BS 4.139 to 37.773)**	**19**	**5.349**	**0.038**
	Region	12.495 (P -5.038 to 29.959; BS -6.637 to 29.713)	48	1.926	0.193	-	-	-	-
Woodland distance	Uncut or cut rush	-0.013 (P -0.062 to 0.035; BS -0.065 to 0.032)	48	0.295	0.591	0.005 (P -0.045 to 0.054; BS -0.044 to 0.058)	19	0.047	0.833
	Region	**0.311 (P 0.168 to 0.453; BS 0.159 to 0.450)**	**48**	**18.022**	**0.001**	-	-	-	-
	Uncut or cut rush	-0.040 (P -0.948 to 0.866;	48	0.008	0.931	0.583 (P -0.312 to 1.479)	19	1.629	0.219

		BS -0.966 to 0.922)							
Boundary distance (square root transformed)	Region	-0.700 (P -1.799 to 0.382; BS -1.716 to 0.423)	48	1.554	0.239	-	-	-	-

Table C.9. Bootstrap 95% confidence intervals for parameter estimates in generalised linear mixed effects models (GLMMs) investigating the effect of rush management and vegetation structure on daily nest predation rates (DPRs) of artificial nests in all study fields (models i, iii, and iv) and treatment fields > 4 ha (model ii) for comparison with profile CIs reported in Table 3.2. Significant predictor variables are in bold. Control fields are the reference level for treatment; uncut rush patches are the reference level for cut or uncut rush; negligible/no predator control is the reference for predator control; Geltsdale is the reference for region.

Model	Predictor variables in models	Estimate (bootstrap 95% CIs)
Effect of rush management on daily nest predation rates		
i	Treatment	**0.894 (0.370 to 1.529)**
	Woodland distance	-0.050 (-0.387 to 0.251)
	Boundary distance (square root transformed)	0.156 (-0.130 to 0.480)
	Deployment date	**-0.287 (-0.493 to -0.070)**
	Field size (natural logarithm transformed)	-0.221 (-0.571 to 0.147)
	Predator control	-0.474 (-1.252 to 0.170)
	Corvid abundance (natural logarithm (x+1) transformed)	-0.181 (-0.515 to 0.130)
	Region	0.093 (-0.771 to 1.056)
ii	Cut or uncut rush	**0.728 (0.054 to 1.579)**
	Woodland distance	0.260 (-0.422 to 0.853)
	Boundary distance (square root transformed)	-0.245 (-0.636 to 0.105)
	Deployment date	0.040 (-0.256 to 0.405)
	Region	0.065 (-1.122 to 1.542)
Effect of vegetation structure on daily nest predation rates		
iii	Vegetation density	**-0.735 (-1.066 to -0.456)**
	Rush cover	-0.051 (-0.325 to 0.188)
	Woodland distance	-0.106 (-0.392 to 0.193)
	Boundary distance (square root transformed)	0.209 (-0.097 to 0.517)
	Deployment date	-0.176 (-0.421 to 0.059)
	Field size (natural logarithm transformed)	-0.113 (-0.490 to 0.272)
	Predator control	-0.246 (-0.969 to 0.309)
	Corvid abundance (natural logarithm (x+1) transformed)	-0.206 (-0.476 to 0.063)
	Region	0.639 (-0.119 to 1.476)
iv	Vegetation height	**-0.766 (-1.162 to -0.449)**
	Rush cover	0.046 (-0.259 to 0.342)
	Woodland distance	-0.101 (-0.358 to 0.172)
	Boundary distance (square root transformed)	0.203 (-0.075 to 0.537)
	Deployment date	-0.083 (-0.332 to 0.203)
	Field size (natural logarithm transformed)	-0.119 (-0.520 to 0.264)
	Predator control	-0.406 (-1.044 to 0.169)
	Corvid abundance (natural logarithm (x+1) transformed)	-0.168 (-0.473 to 0.135)
	Region	0.395 (-0.378 to 1.236)

Table C.10. Mean and 95% confidence intervals (CIs) for observed values of environmental variables of artificial wader nests located in cut and uncut rush patches within treatment fields >4 ha in the SWP and at Geltsdale. Data are given for four different groups (cut rush patches, uncut rush patches, SWP and Geltsdale) separated into the early and late breeding seasons. Bootstrap CIs were calculated for distance to nearest field boundary (m) because the data were positively skewed.

Response variable	**Group**	**Mean (95% CIs)**	
		Early breeding season	Late breeding season
Vegetation density	Cut rush patches	0.24 (0.16 to 0.31)	1.21 (0.07 to 0.34)
	Uncut rush patches	0.62 (0.52 to 0.71)	1.58 (0.29 to 0.86)
	SWP	0.36 (0.24 to 0.48)	-
	Geltsdale	0.31 (0.21 to 0.41)	-
Vegetation height (cm)	Cut rush patches	12.57 (10.90 to 14.25)	19.63 (16.81 to 22.46)
	Uncut rush patches	45.29 (39.66 to 50.92)	38.60 (26.19 to 51.01)
	SWP	18.03 (10.58 to 25.49)	-
	Geltsdale	22.53 (16.60 to 28.47)	-
Rush cover (%)	Cut rush patches	31.25 (24.77 to 37.73)	20.00 (12.24 to 27.76)
	Uncut rush patches	45.83 (31.13 to 60.54)	40.00 (8.34 to 71.66)
	SWP	43.16 (31.37 to 54.94)	-
	Geltsdale	29.48 (23.17 to 35.80)	-
Woodland distance (km)	Cut rush patches	0.49 (0.42 to 0.55)	0.67 (0.63 to 0.70)
	Uncut rush patches	0.48 (0.35 to 0.62)	0.67 (0.56 to 0.77)
	SWP	0.67 (0.64 to 0.70)	-
	Geltsdale	0.36 (0.30 to 0.43)	-
Boundary distance (m)	Cut rush patches	61.96 (55.03 to 70.16)	51.63 (46.37 to 57.81)
	Uncut rush patches	61.85 (49.28 to 84.11)	60.75 (48.83 to 78.18)
	SWP	54.72 (48.88 to 62.16)	-
	Geltsdale	66.65 (57.31 to 78.17)	-

C.1 References

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Natural England (2012). *Entry level stewardship: environmental stewardship handbook*. Available at http://publications.naturalengland.org.uk/publication/2798159.

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Appendix D

Supporting information for Chapter 4

Table D.1. Summaries of the Entry Level Stewardship agri-environment scheme prescriptions (adapted from Natural England (2012)) that were followed or emulated in the treatment fields. Descriptions relate to rush management aspects of the prescriptions only (cutting regime and grazing).

Prescription code	Prescription name	Pre-requisites	Description of rush management
EK4	Management of rush pastures	• Outside SDAs • Below the Moorland Line • At least one-third of the field area covered by rush	• Cut no more than one-third of the total area of rush in rotation • Cutting must not be conducted between 15 March and 31 July • Graze the aftermath, ideally with cattle • If there is rapid regrowth of rush, cut again within 8 weeks of the first cut (not between 1 April and 31 July) • Repeat management each year
EL4	Management of rush pastures in SDAs	• Inside SDAs • Below the Moorland Line • Above the Moorland Line on parcels <15 ha • At least one-third of the field area covered by rush	• Cut no more than one-third of the total area of rush in rotation • Cutting must not be conducted between 1 April and 31 July • Graze the aftermath, ideally with cattle • If there is rapid regrowth of rush, cut again within 8 weeks of the first cut (not between 1 April and 31 July) • Repeat management each year

SDA = Severely Disadvantaged Area: land on which agricultural production is severely restricted by environmental conditions (Natural England, 2013); ha = hectare

Table D.2. Mean and 95% confidence intervals for observed values of environmental variables for treatment and control fields in the SWP and Geltsdale. Data are given for four different groups (treatment fields, control fields, SWP and Geltsdale) in the early breeding season and overall breeding season. Bootstrapped CIs were calculated for field size (ha) because the data were positively skewed.

Response variable	Group	Mean (95% CIs)	
		Early breeding season	Overall breeding season
Rush cover (%)	Treatment fields	-	46.70 (39.00 to 54.30)
	Control fields	-	40.00 (31.20 to 48.80)
	SWP	-	48.80 (41.10 to 56.50)
	Geltsdale	-	35.60 (27.70 to 43.40)
Soil penetration resistance (kgF)	Treatment fields	5.17 (4.70 to 5.65)	5.34 (4.98 to 5.71)
	Control fields	5.38 (4.45 to 6.31)	5.44 (4.72 to 6.17)
	SWP	5.65 (4.93 to 6.36)	5.68 (5.17 to 6.18)
	Geltsdale	4.76 (4.05 to 5.48)	5.01 (4.36 to 5.65)
Soil moisture (%)	Treatment fields	65.00 (57.30 to 72.60)	65.20 (58.80 to 71.70)
	Control fields	60.90 (52.90 to 68.90)	62.60 (55.40 to 69.90)
	SWP	60.00 (52.20 to 67.70)	63.30 (56.60 to 70.00)
	Geltsdale	66.90 (59.60 to 74.20)	64.70 (57.80 to 71.70)
Soil pH	Treatment fields	5.36 (5.05 to 5.66)	5.34 (5.03 to 5.64)
	Control fields	4.87 (4.59 to 5.15)	4.88 (4.60 to 5.17)
	SWP	5.01 (4.78 to 5.25)	4.90 (4.68 to 5.12)
	Geltsdale	5.24 (4.83 to 5.66)	5.39 (5.01 to 5.78)
Field size (ha)	Treatment fields	-	7.18 (4.67 to 13.50)
	Control fields	-	4.68 (3.21 to 7.68)
	SWP	-	3.00 (2.33 to 3.90)
	Geltsdale	-	9.92 (6.87 to 16.20)
Woodland distance (km)	Treatment fields	-	0.486 (0.372 to 0.601)
	Control fields	-	0.528 (0.414 to 0.641)
	SWP	-	0.629 (0.536 to 0.722)
	Geltsdale	-	0.339 (0.249 to 0.429)

Table D.3. The number of fields containing different estimated breeding Curlew, Snipe, and Lapwing pairs. The number of fields is given for four groups: treatment fields (21 fields in total), control fields (22 fields in total), the SWP (25 fields in total), and Geltsdale (18 fields in total).

Wader species	Estimated breeding pairs	Number of fields			
		Treatment		Region	
		Treatment	Control	SWP	Geltsdale
Curlew	0	9	13	15	7
	1	7	7	9	5
	2	4	2	1	5
	3	1	0	0	1
Snipe	0	10	18	20	8
	1	10	4	4	10
	2	1	0	1	0
Lapwing	0	12	16	24	4
	1	5	4	1	8
	2	4	0	0	4
	3	0	2	0	2

Table D.4. Modelling estimated breeding wader pairs as a function of a selected environmental variable whilst taking region (SWP or Geltsdale) and field size (ha; natural logarithm transformed; as an offset) into account. Each environmental variable was modelled independently of the others. Linear models for each environmental variable include the linear term. Quadratic models for each environmental variable include the linear and quadratic terms. The Akaike information criterion with correction for small sample size (AICc) value for each model is shown, with the models with the lowest AICc values (the lowest AICc model plus any models with a value <2 AICc higher) highlighted by an asterisk. Evidence for an important non-linear association requires the AICc value of a quadratic model to be >2 AICc lower than that of a linear model.

Wader species	**Model**	**AICc**				
		Environmental variables				
		Rush cover	Soil penetrability	Soil moisture	Soil pH	Woodland distance
Curlew	Linear	83.263 *	83.685	83.647	82.619	67.894 *
	Quadratic	85.700	84.493	81.918	84.028	70.267
Snipe	Linear	70.033 *	69.335 *	68.871	68.099 *	67.894 *
	Quadratic	72.450	71.636	68.177	70.349	70.267
Lapwing	Linear	60.945 *	62.074 *	61.430	62.369 *	62.690
	Quadratic	63.352	64.114	60.127	64.473	64.672

Table D.5. Descriptions, ranges, and means (95% confidence intervals (CIs)) of the predictor variables included in the models (see Table 4.1). Data pertain to untransformed variables, although field size was natural logarithm transformed prior to inclusion in the models to remove the influence of outliers due to its skewed distribution.

Predictor variable	**Type**	**Units**	**Description**	**Range; mean (95% CI)**	
				Early breeding season	Overall breeding season
Treatment	Categorical		2 categories: treatment field; control field		Treatment field n = 21; control field n = 22
Rush cover	Continuous	%	Visual estimate of rush cover (to the nearest 10%) within each survey field		10-70; 43.256 (95% CI 37.568 to 48.943)
Region	Categorical		2 categories: SWP; Geltsdale		SWP n = 25 fields; Geltsdale n = 18 fields
Field size	Continuous	ha	Area of the survey field		0.403-40.638; 5.898 (95% CI 3.704 to 8.092)
Woodland distance	Continuous	km	Straight-line distance to the nearest block of woodland from the centroid of each survey field		0.084-1.000; 0.508 (95% CI 0.430 to 0.585)
Soil moisture	Continuous	%	Mean soil moisture within each survey field; 6 and 12 measurements averaged respectively for the early and overall breeding seasons	31.333-92.500; 62.895 (95% CI 57.565 to 68.225)	35.392-92.500; 63.888 (95% CI 59.218 to 68.559)
Soil pH	Continuous	pH	Mean soil pH within each survey field; 3 and 6 measurement averaged respectively for the early and overall breeding seasons	3.800-6.595; 5.109 (95% CI 4.898 to 5.321)	3.768-6.578; 5.105 (95% CI 4.894 to 5.316)
Soil penetration resistance	Continuous	kgF	Mean soil penetration resistance within each survey field; 6 and 12 measurements averaged respectively for the early and overall breeding seasons	2.317-9.267; 5.278 (95% CI 4.769 to 5.786)	3.000-8.500; 5.395 (95% CI 5.000 to 5.790)

Table D.6. Comparisons between the three full model types for each wader species using each model's Akaike information criterion value corrected for small sample sizes (AICc) and the statistical significance (using a $P < 0.05$ threshold) of interaction terms when present. Models with the lowest AICc values are highlighted by an asterisk in the 'AICc' column.

Wader species	**Poisson model type**	**AICc**	**Statistically significant interaction term**
Lapwing	Main effects only	69.536 *	
	Main effects plus treatment and rush cover interaction	71.962	-
	Main effects plus treatment and region interaction	70.597	-
Curlew	Main effects only	91.734 *	
	Main effects plus treatment and rush cover interaction	94.949	-
	Main effects plus treatment and region interaction	94.563	-
Snipe	Main effects only	74.352	
	Main effects plus treatment and rush cover interaction	75.126	-
	Main effects plus treatment and region interaction	70.747 *	✓

Table D.7. Modelling environmental variables in treatment and control fields whilst taking region (SWP or Geltsdale) into account. Separate models of each variable were constructed for the early breeding season and overall breeding season except for rush cover and field size as these metrics were stable over the study period. Parameter estimates (± standard errors; SE) are presented, significant effects are in highlighted by an asterisk in the '*P*' column. Type II F tests are presented. Control fields are the reference level for treatment; Geltsdale is the reference level for region. Field size and rush cover did not change with breeding season stage so were only analysed for the overall breeding season.

Response variable	**Predictor variable**	**Early breeding season**				**Overall breeding season**			
		Parameter estimate (± SE)	*Sample size*	*F statistic*	*P*	*Parameter estimate (± SE)*	*Sample size*	*F statistic*	*P*
Rush cover	Treatment	-	-	-	-	6.927 (± 5.284)	43	1.719	0.197
	Region	-	-	-	-	13.383 (± 5.354)	43	6.249	0.017 *
Soil penetration resistance	Treatment	-0.189 (± 0.497)	43	0.144	0.706	-0.085 (± 0.387)	43	0.048	0.828
	Region	0.878 (± 0.503)	43	3.042	0.089	0.668 (± 0.392)	43	2.900	0.096
Soil moisture	Treatment	3.943 (± 5.268)	43	0.560	0.459	2.556 (± 4.722)	43	0.293	0.591
	Region	-6.881 (± 5.338)	43	1.662	0.205	-1.420 (± 4.785)	43	0.088	0.768
Soil pH	Treatment	0.482 (± 0.198)	43	5.918	0.020 *	0.441 (± 0.187)	43	5.528	0.024 *
	Region	-0.218 (± 0.201)	43	1.183	0.283	-0.485 (± 0.190)	43	6.510	0.015 *
Field size	Treatment	-	-	-	-	0.302 (± 0.253)	43	1.428	0.239
	Region	-	-	-	-	-1.077 (± 0.256)	43	17.716	1.409^{-4} *
Woodland distance	Treatment	-	-	-	-	-0.036 (± 0.064)	43	0.311	0.580
	Region	-	-	-	-	0.289 (± 0.065)	43	19.819	6.663^{-5} *

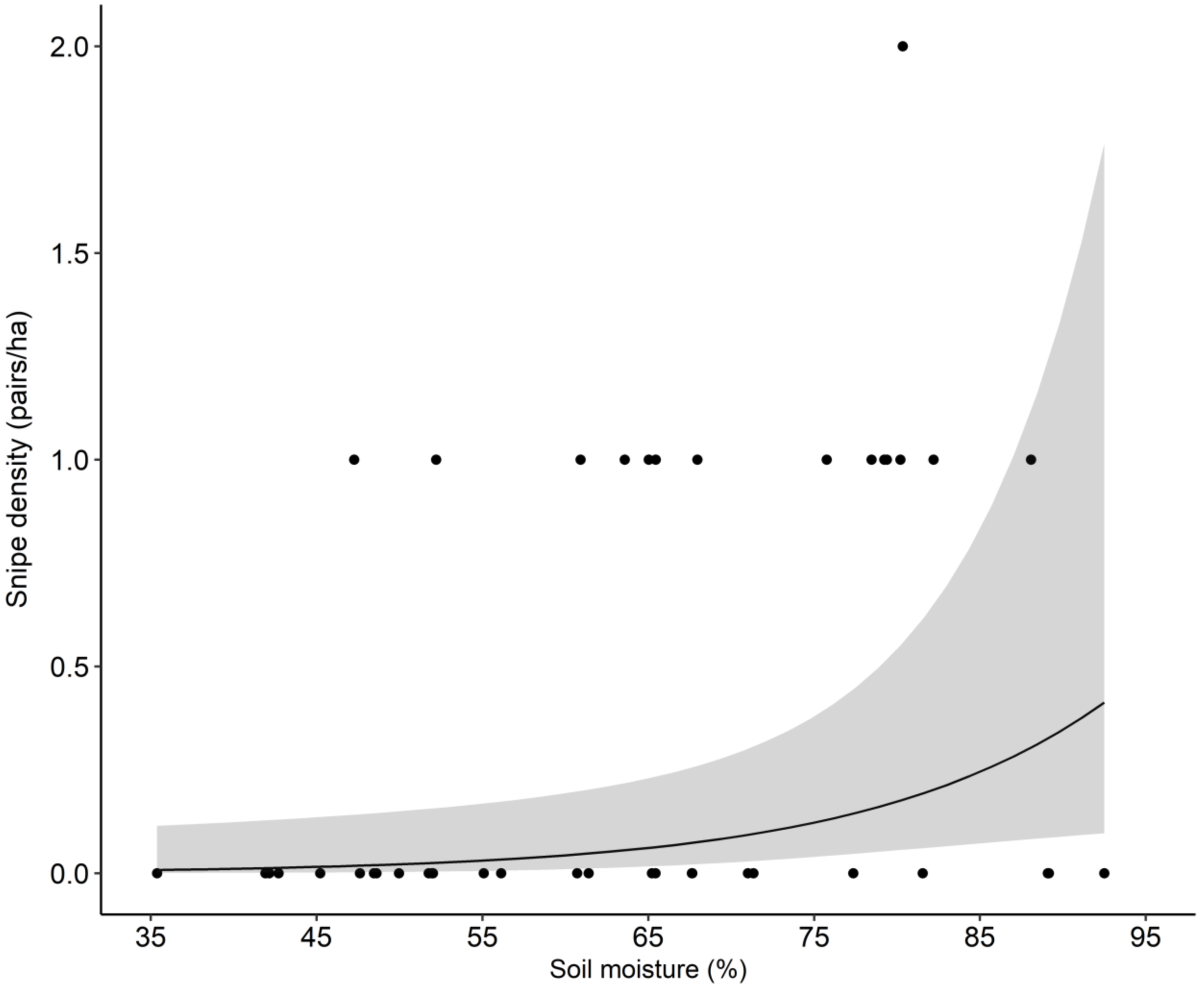

Figure D.1. Poisson model (with treatment and region interaction term) predicted breeding Snipe pair density across the range of soil moisture values recorded in the survey fields. Line represents the model predicted densities, ribbon represents the model predicted 95% confidence interval, and points represent the raw estimated breeding Snipe pairs.

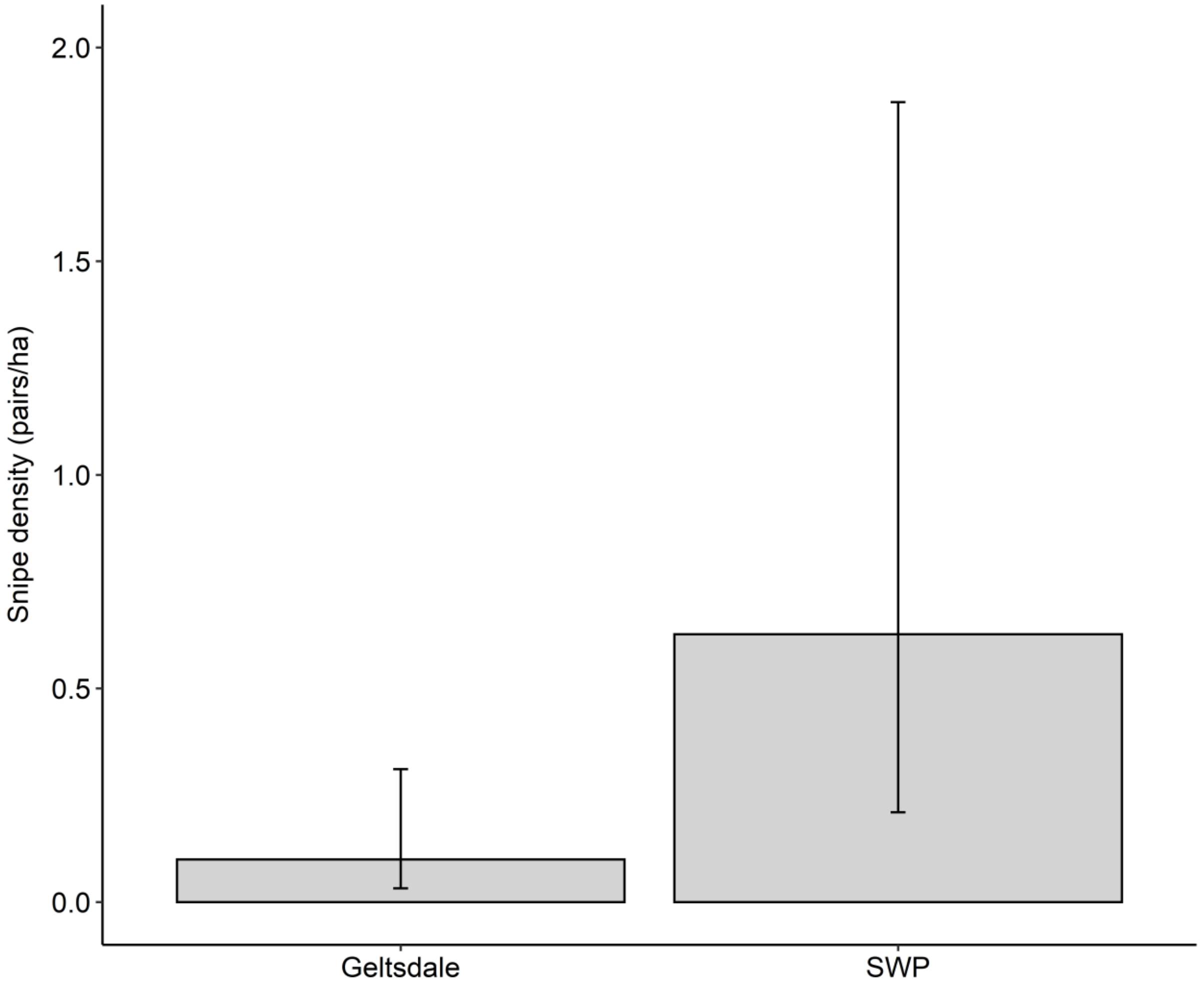

Figure D.2. Poisson model (main effects only model) predicted breeding Snipe pair density in the two study regions, Geltsdale and SWP. Bars represent model predicted densities, and errors represent the model predicted 95% confidence intervals.

D.1 References

Natural England. 2012. *Entry Level Stewardship: Environmental Stewardship Handbook* [online]. Available from: https://www.gov.uk/guidance/environmental-stewardship.

Natural England. 2013. *Hill farming: grants and requirements for upland farmers* [online]. Available from: https://www.gov.uk/guidance/hill-farming.

Appendix E

Supporting information for Chapter 5

E.1 Methods

Information about the project aims, the interview format, and the interviewer (Leah Kelly, PhD researcher, University of Sheffield, South West Peak Landscape Partnership Working for Waders project) was provided. In addition, potential participants were informed that participation in the research was completely voluntary, and that all data provided would be kept anonymous and safely stored. Upon verbal agreement to participate, an interview date and time was arranged.

Before interviews commenced, participants were provided with an information sheet and consent form to ensure that they were able to make an informed decision about whether they wanted to participate. Only after participants had read and agreed to this information (via a signature on the consent form for face-to-face interviews, or verbal consent for telephone interviews), and hence provided informed consent, was the interview allowed to commence.

E.2 Questionnaire

Farmers’ views and experience of rush management

Name:

Farm name:

1. Do you have rush on your farm?

☐ Yes (proceed to question 2)

☐ No (proceed to question 11)

2. Has the amount of rush on your farm changed over the last 10 years?

Considerably decreased	Moderately decreased	Slightly decreased	No change	Slightly increased	Moderately increased	Considerably increased
☐	☐	☐	☐	☐	☐	☐

- If ‘no change’, proceed to question 7.

3. What has been the overall impact of this change in rush on the quality of your land from a farming perspective?

Considerably detrimental	Moderately detrimental	Slightly detrimental	No impact	Slightly beneficial	Moderately beneficial	Considerably beneficial
☐	☐	☐	☐	☐	☐	☐

4. What are the two most important factors of this impact on the quality of your land from a farming perspective?

Prompts e.g. impact on fodder crop, grazing quality, land value

5. What has been the overall impact of this change in rush on the quality of your land from an environmental perspective?

Considerably detrimental	Moderately detrimental	Slightly detrimental	No impact	Slightly beneficial	Moderately beneficial	Considerably beneficial
☐	☐	☐	☐	☐	☐	☐

6. What are the two most important factors of this impact on the quality of your land from an environmental perspective?

Prompts e.g. impact on breeding wading birds, other wildlife, wildflower diversity

- Proceed to question 11

7. What would be the impact of an increase in the amount of rush on the quality of your land from a farming perspective?

Considerably detrimental	Moderately detrimental	Slightly detrimental	No impact	Slightly beneficial	Moderately beneficial	Considerably beneficial
☐	☐	☐	☐	☐	☐	☐

8. What would be the two most important factors of this impact on the quality of your land from a farming perspective?

Prompts e.g. impact on fodder crop, grazing quality, land value

9. What would be the impact of an increase in the amount of rush on the quality of your land from an environmental perspective?

Considerably detrimental	Moderately detrimental	Slightly detrimental	No impact	Slightly beneficial	Moderately beneficial	Considerably beneficial
☐	☐	☐	☐	☐	☐	☐

10. What would be the two most important factors of this impact on the quality of your land from an environmental perspective?

Prompts e.g. impact on breeding wading birds, other wildlife, wildflower diversity

11. Have you managed rush on your farm in the last 5 years?

☐ Yes (proceed to question 12)

☐ No (proceed to question 26)

12. In the last 5 years, were any of your fields that received rush management managed as part of an agri-environment scheme?

☐ Always as part of an agri-environment scheme (proceed to question 13)

☐ Started managing outside an agri-environment scheme but now managed as part of one (proceed to question 13)

☐ Started managing as part of an agri-environment scheme but now managed outside one (proceed to question 13)

☐ Always managed outside an agri-environment scheme (proceed to question 20)

13. In the last 5 years, which of the following agri-environment schemes was the rush managed under? Tick all that apply.

☐ Environmental Stewardship – Entry Level Stewardship

☐ Environmental Stewardship – Upland Entry Level Stewardship

☐ Environmental Stewardship – Higher Level Stewardship

☐ Countryside Stewardship

☐ Other (please specify):

14. In the last 5 years, which specific prescription was the most frequently used prescription for rush management?

Prompts – if you cannot recall it, can I call at a later date for the information?

15. What influenced your decision to participate in this specific agri-environment scheme prescription?

Prompts e.g. increased income, to improve habitat for breeding wading birds and other wildlife, aligned with current farming practices (easy to implement), advice or encouragement from other people (if so, who?)

16. For this specific prescription, which aspects did you find easiest to follow and why?

Prompts e.g. finding livestock for aftermath grazing, aftermath grazing dates, mowing dates, cutting area, rush cover target

17. For this specific prescription, which aspects did you find hardest to follow and why?

Prompts e.g. finding livestock for aftermath grazing, aftermath grazing dates, mowing dates, cutting area, rush cover target

- If ‘started managing as part of an agri-environment scheme prescription but now managed outside one’, proceed to question 18.
- If ‘always managed as part of an agri-environment scheme prescription’ or ‘started managing outside an agri-environment scheme prescription but now managed as part of one’, proceed to question 23.

18. So, you have previously managed rush as part of an agri-environment scheme prescription but are not currently within a scheme, is that correct?

☐ Yes

☐ No

19. Why do you no longer manage rush as part of an agri-environment scheme?

- Proceed to question 23.

20. So, you manage rush but have not done so as part of an agri-environment scheme in the last 5 years, is that correct?

☐ Yes

☐ No

21. Have you managed rush as part of an agri-environment scheme prior to the last 5 years?

☐ Yes

☐ No

22. Why do you not/no longer manage rush as part of an agri-environment scheme?

23. I would now like to ask for more specific details about how you have typically managed rush over the last 5 years.

Please describe if, and how, you cut rush - height of cut, cutting dates (within a year), number of cuts (within a year), number of years when you cut
Please describe if, and how, you grazed land following rush cutting, i.e. aftermath grazing - livestock type, numbers, stocking dates
Please describe if, and how, you applied herbicides to rush - application method, application date, application frequency, timing of application relative to cutting and weather conditions, name of herbicide(s)
Please describe the proportion of rush that was managed within a field and the spatial pattern of cut areas - was it all in one big block or separate smaller patches?

24. What did you do with the rush that had been cut and why?

Prompts e.g. left it, collected it in, end uses of cut rush if collected.

25. If you manage rush, what are the two most important reasons why you do?

Prompts e.g. improve fodder crop, improve grazing quality, improve land value, benefits breeding wading birds, perceptions of neighbouring farmers, agri-environment scheme payments

- Proceed to question 27

26. If you do not manage rush, what are the two most important reasons why you do not?

Prompts e.g. cost to the business, health reasons, lack of necessary equipment, not enough of a problem to warrant management, does not fit in with normal farming activities, lack of knowledge/advice, lack of time

- Proceed to question 32

To what extent do you agree/disagree with the following statement?

27. My typical approach to rush management has improved the quality of my land from a farming perspective.

Strongly disagree	Moderately disagree	Slightly disagree	Neither agree nor disagree	Slightly agree	Moderately agree	Strongly agree
☐	☐	☐	☐	☐	☐	☐

28. What (if any) are the primary reasons for these changes in the farming quality of your land following rush management?

Prompts e.g. fodder crop, grazing quality for a specific type of livestock, land value

To what extent do you agree/disagree with the following statement?

29. My typical approach to rush management has improved the quality of my land from an environmental perspective.

Strongly disagree	Moderately disagree	Slightly disagree	Neither agree nor disagree	Slightly agree	Moderately agree	Strongly agree
☐	☐	☐	☐	☐	☐	☐

30. What (if any) are the primary reasons for these changes in the environmental quality of your land following rush management?

Prompts e.g. breeding wading birds, other wildlife, wildflower diversity

31. Would you consider changing your rush cutting pattern within a field to multiple, smaller cuts if there was a benefit to breeding wading birds?

Considerably unlikely	Moderately unlikely	Slightly unlikely	Neither likely nor unlikely	Slightly likely	Moderately likely	Considerably likely
☐	☐	☐	☐	☐	☐	☐

To what extent do you agree/disagree with the following statements?

32. Rush management agri-environment scheme prescriptions are effective at reducing rush cover.

Strongly disagree	Moderately disagree	Slightly disagree	Neither agree nor disagree	Slightly agree	Moderately agree	Strongly agree
☐	☐	☐	☐	☐	☐	☐

If they do not manage rush at all or in an agri-environment scheme and do not know, say 'I understand that you may not have direct experience but based on your discussions with other people, could you provide an answer'.

33. Rush management agri-environment scheme prescriptions are effective at improving habitat for breeding wading birds.

Strongly disagree	Moderately disagree	Slightly disagree	Neither agree nor disagree	Slightly agree	Moderately agree	Strongly agree
☐	☐	☐	☐	☐	☐	☐

If they do not manage rush at all or in an agri-environment scheme and do not know, say 'I understand that you may not have direct experience but based on your discussions with other people, could you provide an answer'.

Please read the following card with information about the current Countryside Stewardship GS16 rush infestation control supplement.

34. What (if anything) would you change within the current Countryside Stewardship GS16 rush infestation control supplement in order to improve the effectiveness at reducing rush cover and why?

Prompts e.g.

- *Cutting rush (height of cut, cutting dates (within a year), number of cuts (within a year))*
- *Aftermath grazing (livestock type; numbers; stocking dates)*
- *Add in herbicide application (application method, application date, application frequency, timing of application relative to cutting and weather conditions, name of herbicide(s))*
- *Proportion of rush managed within a field, spatial pattern of cut areas*

35. If these proposed changes were implemented in an agri-environment scheme, do you think they would increase or decrease the likelihood of farmers' participation in the scheme?

Considerably decrease	Moderately decrease	Slightly decrease	No change	Slightly increase	Moderately increase	Considerably increase
☐	☐	☐	☐	☐	☐	☐

36. If you were focusing solely on managing rush to benefit breeding wading birds, what (if anything) would you change within the current Countryside Stewardship GS16 rush infestation control supplement and why?

Prompts e.g.

- *Cutting rush (height of cut, cutting dates (within a year), number of cuts (within a year))*
- *Aftermath grazing (livestock type; numbers; stocking dates)*
- *Add in herbicide application (application method, application date, application frequency, timing of application relative to cutting and weather conditions, name of herbicide(s))*
- *Proportion of rush managed within a field, spatial pattern of cut areas*

37. If these proposed changes were implemented in an agri-environment scheme, do you think they would increase or decrease the likelihood of farmers' participation in the scheme?

Considerably decrease	Moderately decrease	Slightly decrease	No change	Slightly increase	Moderately increase	Considerably increase
☐	☐	☐	☐	☐	☐	☐

To what extent do you agree/disagree with the following statements?

38. It is important to ensure that my farm is in a very favourable environmental condition whilst I am farming the land.

Strongly disagree	Moderately disagree	Slightly disagree	Neither agree nor disagree	Slightly agree	Moderately agree	Strongly agree
☐	☐	☐	☐	☐	☐	☐

39. I see value in having large numbers and variety of birdlife on my farm.

Strongly disagree	Moderately disagree	Slightly disagree	Neither agree nor disagree	Slightly agree	Moderately agree	Strongly agree
☐	☐	☐	☐	☐	☐	☐

40. What species are these?

A ………………………………………………………………………………………………………

B ………………………………………………………………………………………………………

C ………………………………………………………………………………………………………

D ………

Thinking about the four species I have just shown you, to what extent do you agree/disagree with the following statements?

41. Optimum rush cover differs between breeding wading bird species.

Strongly disagree	Moderately disagree	Slightly disagree	Neither agree nor disagree	Slightly agree	Moderately agree	Strongly agree
☐	☐	☐	☐	☐	☐	☐

42. A mosaic of shorter and taller vegetation is beneficial for breeding wading birds.

Strongly disagree	Moderately disagree	Slightly disagree	Neither agree nor disagree	Slightly agree	Moderately agree	Strongly agree
☐	☐	☐	☐	☐	☐	☐

43. Conservation of breeding wading bird populations in the South West Peak is important.

Strongly disagree	Moderately disagree	Slightly disagree	Neither agree nor disagree	Slightly agree	Moderately agree	Strongly agree
☐	☐	☐	☐	☐	☐	☐

If they are unsure what the South West Peak is, say 'the South West Peak is approximately the area of the Peak District National Park located between Whaley Bridge in the north and Onecote in the south and between Macclesfield in the west and Longnor in the east.'

44. How have breeding wading bird populations changed in the South West Peak over the last 20 years?

Considerably decreased	Moderately decreased	Slightly decreased	No change	Slightly increased	Moderately increased	Considerably increased
☐	☐	☐	☐	☐	☐	☐

45. For each of the following factors, please state how important it is in influencing breeding wading bird population sizes from 0-10 (0 = not important; 10 = extremely important) and the direction of effect (positive or negative).

Driver	**Direction of effect (+ or -)**	**Importance**										
		Not important										**Extremely important**
Rush encroachment		0	1	2	3	4	5	6	7	8	9	10
Agri-environment schemes		0	1	2	3	4	5	6	7	8	9	10
Draining farmland		0	1	2	3	4	5	6	7	8	9	10
Conversion of other grassland types to improved pasture		0	1	2	3	4	5	6	7	8	9	10
Disturbance from walkers/bird watchers		0	1	2	3	4	5	6	7	8	9	10
Climate change		0	1	2	3	4	5	6	7	8	9	10
Predation		0	1	2	3	4	5	6	7	8	9	10
Other (please specify):		0	1	2	3	4	5	6	7	8	9	10

To what extent do you agree/disagree with the following statements?

46. It is important that neighbouring farmers approve of the way I manage my farm.

Strongly disagree	Moderately disagree	Slightly disagree	Neither agree nor disagree	Slightly agree	Moderately agree	Strongly agree
☐	☐	☐	☐	☐	☐	☐

47. My opinions influence how my neighbours manage their farms.

Strongly disagree	Moderately disagree	Slightly disagree	Neither agree nor disagree	Slightly agree	Moderately agree	Strongly agree
☐	☐	☐	☐	☐	☐	☐

48. If I managed rush according to agri-environment scheme prescriptions, neighbouring farmers would consider me to be a good farmer.

Strongly disagree	Moderately disagree	Slightly disagree	Neither agree nor disagree	Slightly agree	Moderately agree	Strongly agree
☐	☐	☐	☐	☐	☐	☐

49. If I increased the number of breeding wading birds on my farm, neighbouring farmers would consider me to be a good farmer.

Strongly disagree	Moderately disagree	Slightly disagree	Neither agree nor disagree	Slightly agree	Moderately agree	Strongly agree
☐	☐	☐	☐	☐	☐	☐

50. Gender

- Don’t ask, just tick

☐ Male

☐ Female

51. Age

☐ 25-34

☐ 35-44

☐ 45-54

☐ 55-64

☐ 65-74

☐ 75-84

☐ 85-94

☐ Prefer not to say (put an estimate myself if this category is chosen)

52. Highest education qualification

☐ None

☐ O level/GCSE or equivalent

☐ A level or equivalent

☐ Undergraduate degree

☐ Higher degree

53. Tenant farmer or landowner

☐ Tenant

☐ Owner

54. Farm size

55. How many years have you been farming in total?

56. How many years have you been farming in the South West Peak?

57. How many livestock of each type do you have on your farm?

Cattle over 2 years old ………………………………………………………………...

Cattle 6 months to 2 years ………………………………………………………………...

Ram; lowland ewe and lamb ………………………………………………………………...

Store lamb; hogg; teg; hill ewe and lamb ………………………………………………………………...

Horse ………………………………………………………………...

Pony ………………………………………………………………...

Donkey ………………………………………………………………...

Goat ………………………………………………………………...

If cattle are reared, please specify if they are dairy or beef cattle.

☐ Dairy cattle

☐ Beef cattle

58. Typically, what proportion of the area of your farm was dedicated to silage production over the last 5 years (to the nearest 10%)?

59. Typically, what proportion of the area of your farm was dedicated to haylage production over the last 5 years (to the nearest 10%)?

60. Typically, what proportion of the area of your farm was dedicated to hay production over the last 5 years (to the nearest 10%)?

61. Is the farm certified organic?

☐ Yes

☐ No

62. Facilitation fund participation

☐ I am a member of a facilitation fund

☐ I am not a member of a facilitation fund

63. Wader plan participation

☐ I have a wader plan

☐ I am currently in consultation for a wader plan

☐ I do not have a wader plan

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GS16: Rush infestation control supplement

This supplement must be used with another option such as UP2: Management of rough grazing for birds.

This is a short-term incentive to reduce rush cover in parcels with heavy infestations. It will help prevent the loss of botanically rich grasslands or provide nesting areas for breeding wading birds.

Where to use this supplement

Only available on:

- permanent grassland
- a single parcel of land with continuous rush cover of more than 50% of the parcel
- a single parcel for a maximum of 3 years

Requirements

- graze or cut areas of dense rush growth every year so that rushes are less than 20cm high by 30 September
- keep bare ground cover at less than 10% by year 2
- reduce cover of dense rush growth to less than 30% of the parcel area by the end of year 3 of the agreement
- **DO NOT** cut rushes between 15 March and 15 July

Advice and suggestions

The aim is to reduce heavy infestations of rushes to a cover below 30%. Cover should then be maintained at between 10% and 30%.

Also:

- ensure damage to nests or disturbance of chicks is avoided when cutting
- avoid controlling rush where there is standing or flowing water on or close to the surface for most of the year
- avoid cutting areas where rush is growing in association with sphagnum mosses
- graze with cattle, as sheep normally avoid grazing rush
- wherever possible, remove cuttings to prevent low growing plants being smothered

The following machines are most appropriate for rush cutting:

- a grass mower trailed by a tractor
- a pasture topper
- a forage harvester

Table E.1. The scientific and colloquial names accepted for each of the wader species photograph identifications.

Wader species	**Accepted names**
Northern lapwing *Vanellus vanellus*	Northern lapwing, lapwing, peewit, pewit, green plover
Eurasian curlew *Numenius arquata*	Eurasian curlew, curlew
Common snipe *Gallinago gallinago*	Common snipe, snipe
Eurasian golden plover *Pluvialis apricaria*	Eurasian golden plover, golden plover (0.5 point for ‘plover’)

Table E.2. The livestock categories, and respective livestock units, used to collate information concerning farm type, size, and intensity. Categories taken from the UK government's Countryside Stewardship (Natural England, 2017; Annex C).

Livestock category	**Livestock units**
Cattle over 2 years	1.0
Cattle over 6 months to 2 years	0.6
Lowland ewe and lamb; ram	0.12
Store lamb, hill ewe and lamb; hogg; teg	0.08
Horse	1.0
Pony/Donkey	0.8
Goat	0.12

Table E.3. Descriptions, ranges, medians, and means (± standard error (SE)) of the variables included as subsets in the stage 1 and stage 2 models. Data relate to untransformed variables, although variables with skewed distributions (index of grazing livestock intensity, and rush encroachment score) were transformed prior to inclusion in the models to reduce the influence of outliers.

Variable	Type	Description	Range; median; mean ± SE
AES participation	Categorical	2 categories: currently participating; not currently participating	Currently participating n = 20; not currently participating n = 14
AES effectiveness for waders	Continuous (predictor) Ordinal (response)	Likert scale response to the effectiveness of AES rush management prescriptions at improving wader habitat. 'Do not know' responses removed.	1-7; 6; 5.308 ± 0.363
AES effectiveness for rush cover	Continuous (predictor) Ordinal (response)	Likert scale response to the effectiveness of AES rush management prescriptions at reducing rush cover. 'Do not know' responses removed.	1-7; 5; 4.458 ± 0.413
Index of attitudes to bird conservation	Continuous	Summed score of question 39 and question 43	10-14; 14; 13.730 ± 0.138
Index of wader habitat knowledge	Continuous	Summed score of question 41 and question 42	8-14; 13; 12.190 ± 0.312
Index of perceived social norms	Continuous	Product of the scores of questions 46 and 48	0-42; 15; 17.590 ± 2.339
Index of grazing livestock intensity	Continuous	The number of livestock units divided by the total farm area (ha)	0.107-2.377; 0.604; 0.837 ± 0.102
Rush encroachment score	Continuous	-10 to +10; negative score indicates a perceived negative impact on wading bird populations; positive score indicates a perceived positive impact on wading bird populations; scores closer to 0 indicate a perceived lower importance of rush encroachment on wading bird populations	-10 to 8; -8; -6.432 ± 0.778
Gender	Categorical	2 categories: male; female	Male n = 28; female n = 9
Age	Continuous	The midpoint of specified age categories (years) Categories: 25-34; 35-44; 45-54; 55-64; 65-74; 75-84; 85-94	29.5-79.5; 59.5; 62.5 ± 2.2

Highest education qualification	Categorical	3 categories: none; school level education; higher level education	None n = 13; school level education n = 15; higher level education n = 9
Farming years	Continuous	Number of years principal farmer has been farming	1.5-65; 40; 40.2 ± 2.1
Land ownership	Categorical	3 categories: landowner; tenant; landowner and tenant	Landowner n = 23; tenant n = 6; landowner and tenant n = 8
Perceived farm-specific rush management effectiveness - farming perspective	Ordinal	Likert scale response to the perceived effectiveness of farmers' typical approach to rush management at improving the quality of the land from a farming perspective	1-7; 6; 5.441 ± 0.278
Perceived farm-specific rush management effectiveness - environmental perspective	Ordinal	Likert scale response to the perceived effectiveness of farmers' typical approach to rush management at improving the quality of the land from an environmental perspective	1-7; 6; 5.559 ± 0.268

Table E.4. Results from stage 1 modelling showing, for each response variable, the socio-demographic predictor variable(s) included in the model with the lowest Akaike information criterion with correction for small sample size (AICc) plus any addition models with ΔAICc less than two. Model structure and degrees of freedom (df) are shown for each model.

Response variable	Model structure	Predictor variables	df	AICc	ΔAICc
AES participation	Generalised linear model; binomial error structure; logit link	Age	2	47.7	0.00
		Null	1	48.2	0.51
		Land ownership	3	48.7	1.04
AES effectiveness at improving wading bird habitat	Cumulative link model; logit link	*Null*	6	96.5	0.00
Farm-specific rush management effectiveness – farming perspective	Cumulative link model; logit link	Highest education qualification	7	111.7	0.00
		Highest education qualification + farming years	8	113.3	1.59
Farm-specific rush management effectiveness - environmental perspective	Cumulative link model; logit link	*Null*	6	118.3	0.00

Table E.5. a) The number of AES participating respondents who managed rush under Entry Level Stewardship (ELS) and Higher Level Stewardship (HLS) schemes in the last 5 years. b) The number of AES participating respondents who managed rush under different ELS and HLS scheme prescriptions. The prescriptions are those that were most frequently used for rush management in the last 5 years. Some respondents reported multiple prescription codes. HK7, HK8, HK15, HK16, and HL8 are HLS AES prescriptions. EL4 and UL23 are ELS AES prescriptions.

	Number of respondents
a) AES	
ELS only	6
HLS only	11
ELS and HLS	4
AES not provided	1
b) AES prescription(s) most frequently used for rush management	
HK15 Maintenance of grassland for target features (HLS)	2
HK16 Restoration of grassland for target features (HLS)	2
EL4 Management of rush pasture in Severely Disadvantaged Areas (ELS)	3
HL8 Restoration of rough grazing for birds (HLS)	6
UL23 Management of upland grassland for waders (ELS)	1
HK7 Restoration of species-rich, semi-natural grassland (HLS)	1
HK8 Creation of species-rich, semi-natural grassland (HLS)	1
Prescription not provided	9

Table E.6. For each aspect of rush management (rush cutting; aftermath grazing; herbicide application), the total number of respondents who were questioned and thus the number of respondents assigned to each of the associated categories are provided, with the proportion of total number of respondents shown in parentheses. Respondents are divided into those who are currently managing rush outside of an agri-environment scheme prescription (non-AES) and those who are currently managing rush within an agri-environment scheme prescription (AES). For rush cutting: spatial configuration of cuts, one non-AES respondent did not provide an answer. For rush cutting: proportion of rush cut, the sum of the respondents for each of the associated categories is higher than the total number of respondents as some respondents managed different proportions of rush in different fields. The 91-100% category includes respondents who stated either 100% or 'as much as possible'. For herbicide application: number of years applied (out of the last 5 years), one non-AES respondent did not provide an answer.

Rush management	Non-AES	AES
Rush cutting	n = 14	n = 20
Yes	14 (100%)	20 (100%)
No	0	0
Number of years cut (out of last 5 years)	n = 14	n = 20
1 or 2	3 (21%)	4 (20%)
3 or 4	4 (29%)	4 (20%)
5	7 (50%)	12 (60%)
Number of cuts within a year	n = 14	n = 20
1	12 (86%)	20 (100%)
2	2 (14%)	0
Cutting within the wader breeding season (April, May, June)	n = 14	n = 20
Yes	4 (29%)	1 (5%)
No	10 (71%)	19 (95%)
Spatial configuration of cuts	n = 13	n = 20
One block	12 (92%)	11 (55%)
Patches	1 (8%)	6 (30%)
Both	0	3 (15%)
Proportion of rush cut	n = 14	n = 20
1-10%	1	1
11-20%	0	0
21-30%	0	0
31-40%	1	5
41-50%	1	3
51-60%	2	0
61-70%	0	4
71-80%	3	3
81-90%	1	2
91-100%	9	4
Aftermath grazing	n = 14	n = 20
Yes	12 (86%)	20 (100%)
No	2 (14%)	0
Livestock type(s)	n = 12	n = 20
Cattle	5 (42%)	7 (35%)
Sheep	3 (25%)	4 (20%)
Cattle and Sheep	3 (25%)	8 (40%)
Cattle and Donkey	1 (8%)	0
Sheep and Horse	0	1 (5%)
Herbicide application	n = 14	n = 20

Yes	5 (35%)	2 (10%)
No	9 (65%)	18 (90%)
Number of years applied (out of last 5 years)	n = 4	n = 2
1 or 2	3 (75%)	2 (100%)
3 or 4	1 (25%)	0
5	0	0
Herbicide application within the wader breeding season (April, May, June)	n = 5	n = 2
Yes	2 (40%)	1 (50%)
No	3 (60%)	1 (50%)
Herbicide application method	n = 5	n = 2
Weed wipe	4 (80%)	1 (50%)
Boom spray	0	1 (50%)
Knapsack spray	1 (20%)	0
Herbicide type	n = 5	n = 2
Glyphosate	4 (80%)	1 (50%)
MCPA	1 (20%)	1 (50%)

Table E.7. The number of respondents who reported different changes in breeding wading bird populations in the South West Peak over the last 20 years.

Likert scale response	Number of respondents
Considerably decreased	12
Moderately decreased	15
Slightly decreased	4
Neither increased nor decreased	2
Slightly increased	3
Moderately increased	1
Considerably increased	0

E.3 References

Natural England (2017). Countryside Stewardship: Livestock record-keeping guidance on arable and grassland. Available at https://www.gov.uk/government/collections/countryside-stewardship.