The impacts of forestry on breeding curlew distribution – a report to the Forestry Commission

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SUMMARY

We were commissioned by Forestry Commission (FC) to utilise the large-scale Breeding Waders of English Upland Farmland (BWEUF) data and the wider literature to provide evidence for several questions posed by the Curlew Recovery Partnership relating to impacts of forestry on breeding Eurasian curlew *Numenius arquata* and inform the evidence base for guidelines relating to assessing the impact of new woodland creation on breeding waders in England.

We used a range of spatial datasets to derive variables relating to land use and management, terrain, forest type and forest configuration within 1 km and 500m of >32,000 inbye fields across >400 tetrads and investigated their impact on probability of presence of curlew.

Broadleaf woodland made up the majority of forests in our analysis and the amount of broadleaf woodland within both 1km and 500 m of inbye fields was negatively associated with probability of curlew presence. However, this pattern was less strong where there was more non-woodland semi-natural habitats (e.g. grassland, heath and bog) close to the field, at higher altitudes and, for 500 m only, where there was moorland management in place or the topography was flatter. Within 1km, the number of forest patches was also negatively associated with curlew presence, particularly at lower altitudes.

Our work informs several sections of the guidelines, in some cases providing supporting evidence, in others urging caution. Our work supports the presumption of preventing woodland creation within 1 km of important sites for waders (such as SSSIs) and that slopes of >20° are unlikely to hold breeding curlew populations. However, aspects of the guidance that suggest lower risk of new woodland or patches of woodland that do not extend the "predator shadow" may need caution since they are still likely to increase amount (and, depending on scheme, number of patches) of woodland. Our results did not support the view that woodland out of sightlines is of less risk. Our results may suggest that where 'high wildlife value' broadleaf woodland creation is considered (which under current guidelines is a positive factor in determining suitability of woodland creation), other non-woodland semi-natural habitat creation might be considered in addition to the woodland to buffer any possible future negative impacts on curlews.

Our analysis has made progress towards the research questions posed by the curlew recovery partnership, and we suggest future research directions to address these further.

INTRODUCTION

Impacts of forest on curlew and other wader species

Our current understanding from existing research is that in general, open-habitat, ground-nesting birds such as Eurasian curlew (hereafter 'curlew') respond negatively to woodland. Previous studies have shown that forestry is a large driver of wader declines or absence, with presence of forestry and increasing proximity to forestry resulting in a decreased probability of wader presence (Wilson et al. 2014a, Bertholdt et al. 2017, Roos et al. 2018, Pálsdóttir et al. 2022) and nest survival (Chalfoun et al. 2002, Batáry and Báldi 2004, Stephens et al. 2004). Further research has shown that as forest cover increases in an area, there is a decrease in curlew nest success and a negative change in populations (Douglas et al. 2014, Franks et al. 2017, Kaasiku et al. 2022).

Wilson et al. (2014) used breeding bird surveys across three years in northern Scotland's Flow Country to test the effects of distance to forestry on breeding distributions of dunlin *Calidris alpina*, European golden plover *Pluvialis apricaria* and common greenshank *Tringa nebularia*, after controlling for habitat and topography. Their findings suggested that flatter, more exposed ground and ground closer to bog pools was preferred by these species and that vegetation structure in peatlands influenced their distribution. Crucially, dunlin and golden plover were negatively impacted by proximity to forest edge. This effect was strongest within 700 m, suggesting that these wader species are more likely to avoid forestry within this distance. Further research in Estonia by Kaasiku et al. (2022) showed that overall nest survival of several wader species was lower closer to forest edge, with a predicted decrease from 26% at 1,020 m from forest edge to 7% at 20 m from the forest edge. Moreover, nest success was negatively impacted by proportion of forest within a 1 km buffer around nest sites, with 19% of nests hatching at least one egg with no forest cover but only 3% hatching at least one egg with 50% forest cover.

Whilst afforestation on open landscapes undoubtedly results in loss and degradation of suitable breeding habitat for curlew and other wader species, the predation-pressure hypothesis is strongly suggested in the literature as one of the main mechanisms behind the effects of forestry. Studies that have directly considered predation have found increased curlew densities and breeding success on managed moorland where predator control is present, compared to in areas where there is no predator control. In northern England and southern Scotland, Douglas et al. (2014) found that both curlew population change and nesting success positively related to gamekeeper density, whilst they correlated negatively with area of surrounding woodland. This suggests that predation of nests could be a likely mechanism causing population declines. Douglas et al. provided model predictions which suggested that an increase in woodland cover from 0% to 10% within 1km of sites with curlew would require an increase in predator control effort of 48% to ensure population stability. Another study by Franks et al. (2017) looked at the extent of burning and average gamebird abundance as a measure of grouse management intensity and, by proxy, for predator control. They found that curlew abundance was negatively correlated with proportion of surrounding forestry but positively correlated with extent of protected area and gamebird abundance. Further, they found a negative correlation between curlew abundance and corvid and red fox *Vulpes vulpes* abundance, as well as increasing rate of curlew decline in areas with a high crow and fox abundance.

Baines et al. (2023) used a paired design to compare grouse moors with no predator control and grouse moors with predator control and found that significantly greater numbers of curlew successfully raised chicks where predator control was present. Additionally, curlew on grouse moors

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with predator control fledged a greater average number of chicks. Their results suggested that curlew hatching and fledging success is lower where abundance of generalist predators (foxes and corvids) is higher. They also suggested that grouse moors with predator control could act as population sources and grouse moors without predator control as population sinks. A paper by Calladine et al. (2022) used statistical modelling to create high-resolution maps of predicted wader abundance which were subsequently used to produce sensitivity maps that are used to inform high-level policy decisions (O'Connell, 2021). The high-resolution maps were also used to analyse wader distribution in relation to land use and found that relatively high proportions of populations were supported in areas carrying lower predation risk (such as grouse moors).

The predation-pressure hypothesis is further backed up by evidence that shows a strong link between forestry and predator abundance. Douglas et al. (2014) found that area of woodland in 1 km surrounding a site had a highly positive correlation with fox abundance. A report by Chadwick et al. (1997) for FC found that afforestation in the uplands contributed to increasing fox abundances. Amar et al. (2010) used upland bird survey data to investigate correlations between wader population change and several habitat variables at two spatial scales. They found that, overall, predator abundance correlated negatively with several wader species, although golden plover population change showed a negative correlation with grouse moor intensity, and both curlew and golden plover populations showed little difference in managed versus non-managed moorlands. Franks et al. (2017) used gamebird density as an index for grouse moor management, on the assumption that this could indicate areas of predator control. They found that where red grouse and pheasant densities were higher so, too, were curlew densities. Additionally, fox and crow abundances were negatively correlated with curlew abundance between 2007-11 and curlew population declines were highest in areas of high crow abundance across both 1995-99 and 2007-11.

A review paper by Roos et al. (2018) compared generalist predator densities in the UK to the rest of Europe and found that red fox *Vulpes vulpes* densities were third highest and carrion crow *Corvus corone* and hooded crow *C. cornix* densities were highest compared with all other countries. Population trends for avian and mammalian predator species were used to assess whether predator numbers have increased in the UK and found that eight of sixteen avian predator species increased significantly between 1995 and 2015, including raptors and generalist species. Long-term trends from the National Gamebag Census (NGC) between 1961 and 2009 indicated significant increases in four of eight mammalian predators including red fox, stoat *Mustela erminea*, grey squirrel *Sciurus carolinensis* and American mink *Neovison vison*, and short-term (between 1995 and 2009), there were increases in red fox, stoat, grey squirrel and weasel *Mustela nivalis* (though note BTO BBS trend report a 42% decline in foxes between 1996 and 2017 – Harris et al. 2019). It is worth noting that the data used to collect predator densities and population trends included BTO/RSPB/JNCC Breeding Bird Survey which is primarily a bird-monitoring scheme and the NGC which does not include catching effort, which means predator kill trends are not corrected for any variation in effort between years. It is therefore likely that mammalian predators are under-recorded.

Roos et al. (2018) additionally conducted a literature review to investigate the potential impacts of predation on individual or groups of bird species. They used 81 studies that examined the effect of predation on 90 prey species across over 900 cases, with 24 of these cases focusing on curlew and a further 46 on lapwing *Vanellus vanellus*. In a quarter of cases, waders were found to be negatively impacted by their predators. A study by Ewing et al. (2022) considered determinants of curlew nest survival to better understand how management could be applied to increase this. Between 2019 and 2021, in eight sites within the Breckland area of East Anglia, up to 80 pairs of curlew were monitored in arable or grassland-dominated habitat. They found a failure rate of 86% in 136 nests and that this

was predominantly due to nocturnal predators. This further supports the trends in the literature that suggest predation is one of the main drivers of curlew declines in the UK.

More recently, research has started to question the effect that configuration of forestry has on curlew, rather than simply presence, proximity or area of forestry. A paper by Pálsdóttir et al. (2022) in Iceland, used the results of transect surveys to simulate scenarios with alternative forest configurations. They found that in scenarios where simulated trees were arranged in one large block of 1,000 ha, compared with in fifty blocks of 20 ha, the resultant decline in bird abundance would be halved. Moreover, they found that this effect increased as patch size decreased: simulated trees arranged in one 1,000 ha block was predicted to result in just 11% of the total decline experienced if they were arranged in one thousand 1 ha blocks. Linking this to predation, a review paper by Chalfoun et al. (2002) examined the effects of habitat fragmentation on predation and found that nest predation was more likely in fragmented landscapes, though they note that not all predator species will respond to fragmentation in the same way and that the relationship between fragmentation and predation is complex and context-specific. Nevertheless, predation pressure is understood to increase with increasing proximity to edge habitats, as a result of increased edge effects. In the current guidance relating to assessing the impact of new woodland creation on breeding waders in England, a 500 m "predator shadow" is used to account for increased predation pressure as a result of forestry in open landscapes and is part of the decision-making process for new woodland creation (DEFRA et al. 2023). It is not clear where the term predator shadow stems from but, in essence, it is a defined zone where there is increased predation pressure at the edge of a forestry block and where this pressure extends across neighbouring habitat.

Outside of forest habitats, one of the strongest predictors in Franks et al. (2017) was the proportion of semi-natural habitat which positively influenced curlew density. Across both timeframes, 1995-99 and 2007-11, semi-natural habitat supported the highest predicted densities of breeding curlew compared to all other measured habitats (arable, improved grassland and mountain, heath and bog). They found a strong negative correlation between arable habitat and the curlew abundance in both periods. Taken together, these results imply the importance of semi-natural habitat for curlew breeding and the conversely negative impacts that increase in arable land has within curlew territories. This could be as a result of direct habitat loss when arable land is created or through the subsequent increase in agricultural practice.

UK and Scandinavian contexts

An aspect that has been discussed within the Curlew Recovery Partnership is a comparison between the UK and other countries in Europe with high forest cover yet where curlews are facing less severe decline. There is evidence breeding wader populations can exist in more highly afforested landscapes in other northern European countries and understanding why this is may help us to create more woodland in the UK without threatening wader populations. Finland is one such example, where over 73% of the country's land area is forested (Food and Agriculture Organization of the United Nations (FAO) 2020) but where curlew populations seem to have stabilised (BirdLife International 2022)., Direct comparison between the UK and Finland is made difficult by several factors. Finland has a much smaller human population of 5.5 million, compared to the UK's 67.3 million, but is approximately 1.2 times the size in land mass area (The World Bank 2017). Most of the human population live in southern Finland and as such, this is where there is a high quantity of agricultural land. As a result, most of the curlew population breed here and are rarely found in the northern regions (Colwell and Wynn 2023, Luonto Portti (Nature Gate) 2023).

Additionally, Finland contains areas of expansive peatland, around 26% of the country's land area (Tanneberger et al. 2017), where curlew and other breeding birds congregate throughout the season (Colwell and Wynn 2023). In comparison, whilst 11% of England's land area is covered by peatlands, only 1% of these have been mapped as "undamaged" (Natural England 2010). In this case, where they remain significantly waterlogged, are actively forming peat and so act to sequester carbon. The large expanses of peatland habitats in Finland are often difficult to access by predators due to ground saturation following winter melts and due to the high numbers of birds present, vigilance for predators is high so predation of eggs/chicks/adult birds in these areas is low (Colwell and Wynn 2023).

As reported by Colwell & Wynn (2023), a difference between Finland and the UK is the density of mesopredators (predator species that would tend to occupy a middle trophic level). As mentioned previously, the UK has some of the highest densities of red fox and crows in Europe (Roos et al. 2018) but in Finland mesopredators exist at much lower densities due to limited food availability, particularly through the winter, and exposure to predator control used to preserve shooting interests (Collwell & Wynn 2023). Finland, like much of Europe and unlike the UK, also has a greater suite of existing populations of apex predators such as grey wolf *Canis lupus*, Eurasian lynx *Lynx lynx* and brown bear *Ursus arctos*. These species are unevenly distributed and subject to predator control for hunting management but also help to limit mesopredator numbers. A trip in May 2023 by Curlew Action suggested that breeding curlews can persist close to trees provided mesopredator densities are kept low enough to allow curlew to be productive (Colwell and Wynn 2023).

Eurasian curlew status in the UK

Alongside several other wader species, curlews in Britain are experiencing significant population decline, reflecting wider trends seen across Europe (Pearce-Higgins et al. 2017). Britain is globally important for curlew, hosting around a quarter of the world's breeding population (Brown, 2015) but, in 2008, the curlew was classified as globally Near Threatened by the IUCN and, in 2015, the species was moved from Green to Red on the UK Birds of Conservation Concern (BoCC) List after a 49% breeding population decline between 1990 and 2015 (Eaton et al. 2015). Table 1 shows the conservation status of the curlew within UK, European and international designations.

Table 1. Conservation status of Eurasian curlew in the UK and internationally.

Directory	Classification
NERC Act (2006) Section 41 species of "principal importance" for the purposes of conserving biodiversity in England	YES
UK Birds of Conservation Concern 5	Red
IUCN Red List GB	Endangered
IUCN Red List (EU)	Near threatened
IUCN Red List (global)	Near threatened

Woodland creation in England

At present, in England, as in other parts of the UK, there is growing pressure to increase woodland cover as a method of improving climate change mitigation and adaptation, increasing domestic timber production, restoring nature and reducing carbon emissions. In England, tree planting has been occurring at an average rate of approximately 2,000 hectares per year since 2011 (Reid et al., 2021). However, an increase in this rate will be necessary in order to meet the peak national afforestation target of 50,000 hectares per year in 2035, to be maintained until at least 2050, as outlined in the UK

Net Zero Strategy (UK Government 2021). This has resulted in a land use conflict in the uplands whereby land that is suitable and targeted for forestry often also carries high conservation value in relation to waders such as curlew. The Forestry Commission provides guidance in line with relevant legislation on when woodland creation is likely to be appropriate as well as when further survey or action is needed prior to woodland creation (DEFRA et al. 2023).

AIMS & RESEARCH QUESTIONS

Given the threatened status of the curlew in the UK, the drive to plant more forests in England and the known negative associations between curlew and afforestation, we were commissioned by the Forestry Commission (FC) to conduct research that would contribute further towards a better understanding of the impacts of forestry and woodland creation on breeding curlew distribution. We were guided by questions posed to the FC by the Curlew Recovery Partnership (Table 2a).

Table 2. Questions posed by the Curlew Recovery Partnership and our matched objectives for this study

	(a) Question	(b) Objective
1	How does predation risk differ between woodland types?	To analyse the impact of different woodland types on curlew abundance and distribution
2	How does predation pressure impact breeding waders beyond a 1km buffer of woodland sites?	To analyse the impact of forestry within various buffer distances on curlew abundance and distribution
3	How does topography influence the extent of edge effects and increased predation pressure from woodland sites?	To investigate the interaction between topography and forestry on curlew abundance and distribution
4	Why do breeding waders in the UK suffer more severe declines in wooded landscapes than in other European countries?	To suggest possible hypotheses that may explain this, through a combination of literature review and implications from our wider analysis
5	What are the effects of extent and configuration of woodland on changes in wader abundance and productivity?	To analyse the effects of several landscape ecology metrics on curlew abundance and distribution

These questions drew on knowledge gaps in the existing literature, such as the potential differences in impact of woodland type and configuration, scales of impact and topography/terrain, as well as the comparative differences in state of curlew populations in other European countries. Table 2b outlines our objectives in response to these questions, where we analysed fine-scale spatial data from the Breeding Waders of English Upland Farmland (BWEUF) survey to address aspects of Questions 1, 2, 3 and 5. Additionally, through a combination of our results and literature review, we aimed to contribute towards Question 4. Ultimately, this report and its findings aim to provide further evidence that will be useful in an applied forestry context, and specifically to inform updates to the guidance on where woodland creation is likely to be appropriate (DEFRA et al. 2023)

METHODS

Data sources

We used curlew data from the BWEUF 2016 survey and five additional open datasets to provide data on forests, land use and management, and terrain listed in Table 3. These datasets were chosen based on their ability to facilitate analysis on a range of potentially important predictor variables, as well as their temporal match to the BWEUF data. For the National Forest Inventory, Corine Land Cover and Moorland Change maps, we selected for years that most closely matched the 2016 BWEUF survey. The 2023 Ancient Woodland Inventory and UK Digital Elevation Model datasets were used, as only the most recent versions are available, but these are highly unlikely to vary between years, since Ancient Woodland is protected and long-established.

Table 3. Sources of data used in our analysis.

Data	Provider	Year	Source
Breeding Waders of English Upland Farmland	British Trust for Ornithology (BTO)	2016	Provided by the BTO. Details at: https://www.bto.org/our-science/projects/breeding-waders-english-upland-farmland/
National Forest Inventory	Forestry Commission	2016	https://www.forestresearch.gov.uk/tools-and-resources/national-forest-inventory/
Ancient Woodland Inventory	Natural England	2023	https://www.data.gov.uk/dataset/9461f463-c363-4309-ae77-fdcd7e9df7d3/ancient-woodland-england
Corine Land Cover	Copernicus Land Monitoring Service (CLMS)	2018	https://land.copernicus.eu/pan-european/corine-land-cover/clc2018
Moorland Change Map	Natural England	2015-16, 2016-17	https://www.data.gov.uk/dataset/e204e870-37f8-48b1-adb3-e0cdcd044ed0/moorland-change-map-england
UK Digital Elevation Model	Ordnance Survey	2022	https://osdatahub.os.uk/downloads/open/Terrain50

The Breeding Waders of English Upland Farmland (BWEUF) survey

The BWEUF survey was conducted in 2016 by the British Trust for Ornithology (BTO) and RSPB (funded by EN and Defra) across upland England. The stated aims of the survey were to "estimate the population size of waders and other important bird species supported by 'In-bye' farmland" and "provide [a] baseline against which to measure population changes and the effect of agri-environment scheme management" (https://www.bto.org/our-science/projects/breeding-waders-english-upland-farmland). In total, 522 (18.4%) out of 2837 English tetrads containing at least 20% (80 ha) of inbye land were surveyed, 54% by volunteers, and the rest by professional fieldworkers employed by RSPB (Phil Grice, Natural England, *personal communication*). This consisted of a combination of volunteer and professional fieldworker surveys across two visits in the breeding season to 2km x 2km tetrads. All fields within each tetrad were surveyed and total numbers of adults, chicks and estimated breeding pairs were recorded per field. More information regarding the survey methodology and results can be obtained in the URL in Table 3. Figure 1 provides an overview of the extent of the survey, showing tetrads from Northumberland down to Devon. This map also shows results of curlew presence and absence within tetrads.
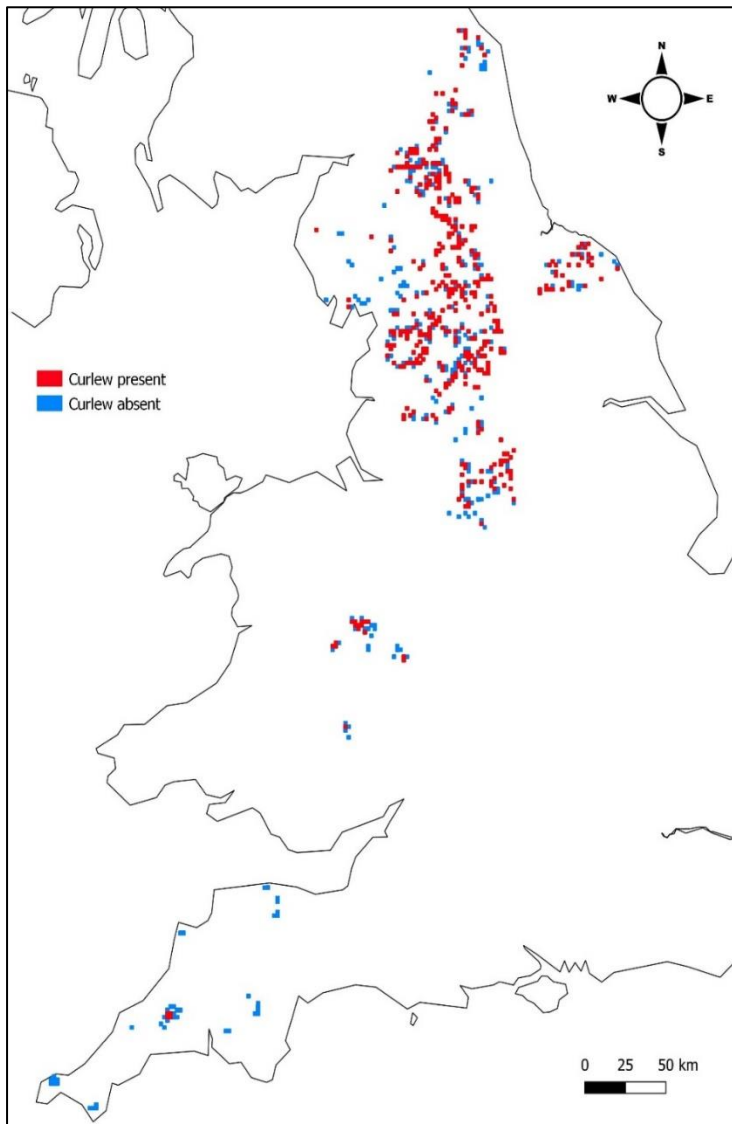
Figure 1. Map showing distribution of surveyed tetrads. Red indicates a tetrad where there was a minimum of one curlew record in any field. Blue indicates tetrads in which no curlews were recorded. © Base map obtained from IGISMAP (2023), curlew data obtained from BTO (2016) and tetrad data obtained using TomBio QGIS plugin Field Studies Council (2023).

Extent of analysis and selection of response variable

The total BWEUF dataset we received from BTO contained 716 tetrads. As the methodology (https://www.bto.org/sites/default/files/bweuf-full-survey-instructions.pdf) specified 'moorland' and 'other fields' as optional for surveyors, we only analysed data from inbye fields, so that surveyor coverage was consistent and not a confounding variable in our analysis. The total number of inbye fields surveyed was 43,326. Of the 716 tetrads surveyed, 406 (56%) contained at least one curlew record, although these 56% of tetrads represented 75% of in-bye fields in the dataset (32,446). Many of the tetrads where curlews were absent were relatively geographically isolated areas, for example in South West England, or generally appeared to be at the edges of the surveyed regions (Figure 1),

and so may have represented landscapes where curlew were absent. As such, we focused our analysis only on the tetrads where at least one curlew was located, since this suggests there is a local population of breeding curlews. This meant our analysis focussed on the suitability of individuals fields (based on the habitat, land use and terrain in their vicinity) within landscapes with some curlew. All subsequent text refers to this modified extent.

Of the 32,446 fields in the extent of our analysis, 2,127 (7.0%) contained any record of curlew, of which 1,228 (4.1%) contained identified breeding pairs and only 22 (<0.001%) contained identified chicks (Table 4a). It should be noted that measuring breeding success was not part of the survey methodology, so these chick records can only be considered incidental records and may be biased by factors such as field worker identity. Where adults or breeding pairs where present, the majority of fields contained only a single adult or breeding pair (Table 4b). Because variation in abundance was very low, and to maximise sample size of 'presence' records, we chose to analyse the data using presence/absence of adult curlew as our response variable.

Table 4. Summary of curlew presence and abundance in fields used in analysis by (a) presence of adults, breeding pairs and young, and (b) median and range of observed abundance. All combinations that represented fields in the sample are shown. For (a) all combinations represented in the data are shown.

(a) Presence

Adults	Breeding pairs	Young	Number of fields
None	None	None	30,319
Present	None	None	877
Present	Present	None	1,228
Present	Present	Present	22
Total			32,446

(b) Median, inter-quartile range (first parentheses) and range (second parentheses) of observed abundance where curlew were present

Adults	Breeding pairs	Young
1.0 (1-2) (1-30)	1.0 (1-1) (1-6)	1.5 (1-2) (1-6)

Spatial data extraction

QGIS 3.22.14-Białowieża (QGIS.org 2023) was used to visualise, analyse and extract spatial data from the datasets in Table 3. First, BWEUF bird data recorded only in inbye fields were extracted, removing any data from moorland or 'other' field types. Following this, buffers were created around each field. In line with current guidance, a 1 km buffer was used for the first round of analysis and, following discussion with the FC, a second analysis was conducted using a 0.5 km buffer to assess impacts on a smaller, more localised scale. This multi-scale approach addresses Question 2 in Table 2a.

Location and extent of woodlands were identified from the National Forest Inventory (NFI) and Ancient Woodland Inventory (AWI) datasets where they fell within the extent of a field buffer. Any areas of overlap of AWI and NFI data were classified as AWI (e.g. a patch of woodland that was classified 'Ancient' in the AWI and 'Broadleaf' in the NFI was classed as 'Ancient') for the purposes of our analysis. The total area of each forest type classification per buffer was extracted for statistical analysis. Due to the complexity of our final analysis, following consultation with FC, we chose to use

five levels of forest type from this data, derived directly from the AWI and NFI datasets: ancient, young, low density, mature broadleaf and mature conifer. Mature broadleaf and mature conifer included data from their respective categories combined with the associated 'mixed' categories in the NFI ('Mixed predominantly broadleaved' and 'Mixed predominantly coniferous'). This variable allowed us to consider impacts across different woodland types, in line with Question 1 in Table 2a.

Data from the Corine Land Cover (CLC) 2018 dataset was similarly extracted where it fell within the field buffers. Our literature review had revealed that semi-natural habitats can be positively associated with curlew abundance. The area of each land use classification was totalled per buffer and categorised as either 'semi-natural' or 'improved', according to Appendix 2 of the consultation on proposal to introduce zonal maps to guide wader conservation and forest expansion (DEFRA et al. 2023), as seen in Table 5. Forest-related land uses were removed because these were analysed separately.

Table 5. Classification of Corine Land Use (CLC) land use categories as semi-natural or improved. Broadleaved forest (311), coniferous forest (312), mixed forest (313) and transitional woodland/scrub (324) were removed because forestry variables were being analysed separately.

CLC Code	Land use	Category
111	Continuous urban fabric	Improved
112	Discontinuous urban fabric	Improved
121	Industrial or commercial units	Improved
122	Road and rail networks and associated land	Improved
124	Airports	Improved
131	Mineral extraction sites	Improved
132	Dump sites	Improved
141	Green urban areas	Improved
142	Sports and leisure facilities	Improved
211	Non-irrigated arable land	Improved
231	Pastures	Improved
243	Land principally occupied by agriculture, with significant areas of natural vegetation	Improved
321	Natural grassland	Semi-natural
322	Moors and heathland	Semi-natural
332	Bare rock	Semi-natural
333	Sparsely vegetated areas	Semi-natural
412	Peat bogs	Semi-natural
421	Salt marshes	Semi-natural
423	Intertidal flats	Semi-natural
512	Water bodies	Semi-natural
522	Estuaries	Semi-natural
523	Sea and ocean	Semi-natural

Because previous research had shown predator control to be a strong positive predictor of curlew population and productivity (Douglas et al. 2014, Baines et al. 2023, see Introduction for detail), Natural England's Moorland Change (MC) maps covering 2015-16 and 2016-17 were used to derive areas of recent burning that fell within field buffers, as a way of quantifying areas of moorland management and, by proxy, assumed presence of predator control of any level. This approach was

used following previous research such as Douglas et al. (2014) and Franks et al. (2017) who used gamekeeper density and grouse management intensity as proxies for predator control. Total area in square metres per field buffer of any zones classified as 'Recent burn or cut in vegetation' in the MC maps was extracted for statistical analysis.

Question 3 in Table 2 specifically asks about the potential interaction between topography and effects of forests on breeding waders. Using the OS Terrain 50 Digital Elevation Model (DEM), a range of topography/terrain metrics could be calculated including the mean and variance values for altitude, slope and ruggedness. These six metrics were calculated and extracted for each field buffer for statistical analysis. The slope is the inclination angle of the terrain, expressed in degrees, and thus, the mean slope is the mean incline calculated across each field buffer. Ruggedness is a quantitative measurement of terrain heterogeneity, developed by Riley et al. (1999). The change in elevation within a 3x3 grid of cells within the raster DEM data is calculated for every cell and summarised. Each cell contains the difference in elevation from a centre cell and the 8 cells that surround it. The mean ruggedness was the mean value calculated across each field buffer. Similarly, the mean altitude, measured in meters, is the mean altitude value calculated across each field buffer.

While we considered extent of woodland by type, above, Question 5 in Table 2 asks about the configuration of woodland. Woodland configuration is a complex concept since we can consider it across several potentially interrelated dimensions such as amount, shape, dispersion, fragmentation, prevalence of edges and others. The R package 'landscapemetrics' (Hesselbarth et al. 2019) was used to analyse a range forest patch configuration metrics, using a raster layer of all the woodland present across the survey area, derived from the NFI and AWI. This was overlaid with the field buffers and a range of variables was calculated within each buffer area to use in statistical analysis. The metrics selected were mean number of patches, mean patch area, Largest Patch Index (LPI), total edge, edge density, and Euclidean nearest-neighbour index. Descriptions and justification for each of these can be seen later in Table 7 where we summarise all variables analysed.

To investigate potential interactions between forestry and topography, further addressing Question 3 in Table 2, the concept of a Visibility Encroachment Index (VEI) was developed. This built on the concept of viewshed analysis, a terrain analysis technique that delineates the area of landscape visible from a given location, based on terrain surface. In our analysis, we used a custom Python script using the GDAL library combined with the DEM raster data to calculate a viewshed output for each inbye field which reflected visibility from a viewpoint in the field centroid. A height of 0.5m was used to simulate the roughly estimated viewpoint of an individual curlew (British Trust for Ornithology (BTO) 2023) although curlew may sit on hummocks or boundary features. The viewsheds were calculated up to 5km away from the field centroid; this value was arbitrary but seemed a reasonable maximum distance and setting a limit reduced processing time. The viewshed was first calculated just using the DEM layer, which essentially assumes a viewshed with no vegetation or man-made boundary features (such as drystone walls) present. This analysis was the re-run using a modified DEM raster where we added assumed tree heights to all forest patches, Assumed heights were provided by FC and were specific to the five levels of forest type considered in our analysis (

Table 6). Assumed tree heights were used because tree height is not a variable included in the NFI or AWI and time/cost constraints precluded the use of additional datasets such as LIDAR. The forest patches added into the modified DEM were taken directly from the AWI and NFI layers created for forest type analysis previously.

Table 6. Forest type classes and the heights added to each for Visibility Encroachment Index (VEI)

Forest type	Assumed representative height (m)
Young	4
Mature broadleaved	15
Mature coniferous	20
Ancient	20
Low density	10

This generated a second series of viewsheds for each field centroid based on topography and forestry effects. The proportional difference between these was calculated to produce the VEI that showed the proportion of viewshed that was inhibited by forest for each field centroid. This was calculated as follows, where Vis_D is the proportional difference, Vis_T is the visibility based solely on topography and Vis_F is the visibility when forestry is added:

$$Vis_D = \frac{Vis_T - Vis_F}{Vis_T}$$

Fundamentally, Vis_T represents an estimate of the area visible from the field centroid if the landscape had no vegetation (or other non-terrain features) on it and Vis_F is the area visible once tree height is factored in. In cases where forest patches 'blocked' the view from field centroids, this would reduce the area visible; therefore, we assumed that where there were more trees and/or trees closer to fields, and provided they were not hidden by the topography of the landscape, they would reduce the viewshed. Thus the VEI was designed to capture the difference between landscape where the trees were hidden from view from inbye fields (for example if the topography hid them), and landscapes where they were not, an aspect not considered by the other forest variables in our analysis.

Initial variables considered for analysis

Following the above spatial data extraction, we had a list of nineteen potential predictor variables to consider compiled based on the existing literature, the available data and the questions posed in Table 2. These are listed in Table 7 with justifications for their consideration.

Table 7. Variables chosen for statistical data analysis. Variables have been grouped broadly into field characteristics, land use and terrain variables, forest type and extent variables, forest configuration variables, and interaction between forestry and topography. Variables shaded grey were later excluded because of high correlation with other variables in their group (see main text).

Group	Variable	Justification
Field characteristics	Field size (ha)	Several mechanisms could mean field size influences curlew presence or breeding. Larger fields might contain more curlew territories, and larger fields will have a lower density of boundaries which may modify predation pressure.
Land use and terrain variables	Altitude (m)	The mean altitude in metres of the buffer area. Curlew are typically found in upland areas so assessing their presence against altitude could be useful in establishing whether there are high altitude areas that curlew are not likely to be present in and would therefore be suitable for forestry.
	Slope (°)	The mean slope in degrees of the buffer area. Curlew are unlikely to breed on steep slopes but there is a gap in the knowledge around what this threshold might be. This could facilitate woodland creation schemes in upland areas where the slope makes it unlikely breeding habitat for curlew but still suitable for planting.

	Ruggedness	The mean topographical variance across a buffer area. A high ruggedness could indicate a landscape that would be less desirable to curlew because of reduced visibility and increased perceived risk of predation.
	Proportion of semi-natural habitat	The proportion of the buffer area covered by semi-natural or improved habitat. Curlew use surrounding land to forage and are often found in arable farmland, moorland and heath habitats. We could expect to find fewer curlew or reduced presence probability in buffers with a lower proportion of semi-natural habitat. As land uses were categorised as semi-natural or improved (Table 5), these classifications are complementary.
	Proportion of improved habitat	
	Presence/absence of recent moorland burning	The presence/absence habitat with recent burn or cut in vegetation within a buffer area. Buffers containing recent burning indicate they contain an area of moorland that is managed and, as a proxy, an area where predator control is used. Existing literature shows that curlew are more abundant and more likely to breed in areas where predator control is present, so we would expect to find a higher presence probability in areas with recent burning.
Forest type variables	Proportion of broadleaved woodland	The proportion of each forest type class within a buffer area. Differences in forest type and age could influence curlew in different ways and influence how curlew may perceive the woodland in relation to predation risk, which is believed to be a major driver in wader declines in forested landscapes (Franks et al., 2017; Roos et al., 2018; Hancock et al., 2020). Previous research by Wilson et al. (2014) and Palsdottír et al. (2022) suggests that there occurs a decrease in probability of presence, relative abundance and the number of breeding pairs of curlew with increasing proximity to older forests which are more likely to be taller, denser and darker.
	Proportion of coniferous woodland	
	Proportion of ancient woodland	
	Proportion of young woodland	
	Proportion of low-density woodland	
Forest configuration variables	Number of patches	Number of patches within a buffer area. An increase in number of patches will increase the total length of woodland edge within a landscape. Previous literature suggests that increasing the amount of edge habitat leads to a decrease in wader populations as a result of edge effects and increased predation (Palsdottír et al. 2022).
	Mean patch area	Mean area of all patches within a buffer area. As mean patch area increases, conversely, the area of open habitat suitable for curlew will decrease and proximity of curlew to forest edges will increase.
	Largest Patch Index (LPI)	The proportion (%) of coverage of the largest patch within a buffer area. An increase in the LPI suggests that a larger proportion of the buffer area is being dominated by one woodland patch. As forest cover increases, probability of curlew presence decreases (Franks et al. 2017).
	Total edge	The total length of all patch perimeters in a buffer area. An increase in total edge length in an area with curlew will result in their increased exposure to edge effects and decreases in presence probability and breeding success (Amar et al. 2011, Calladine et al. 2022).
	Edge density	The total length of all patch perimeters divided by the area of the buffer. Edge density is linked to total edge and is defined as the total edge length divided by the buffer area. As edge density increases, so will curlew exposure to edge effects.
	Mean Euclidean nearest-neighbour distance	The mean distance between all patches within a buffer area. This measures the average distance between patches as a measure of connectivity of forestry throughout the buffer area.

Interaction between forestry and topography	Visibility Encroachment Index (VEI)	The VEI measures the level of encroachment by trees on visibility of the surrounding area from a field. Literature shows breeding curlew abundance can be negatively associated with increased area of forestry (Franks et al. 2017) but there is a gap in the knowledge on the impacts of any interaction between topography and forestry on curlew presence probability or population change. Additionally, there is interest to know whether there is an interaction from an applied forestry perspective e.g. could woodland creation in valleys, thereby reducing visibility encroachment on the horizon, reduce the negative impacts of forestry on curlew?

Data transformations and multicollinearity between variables

Each of our area-based variables (field area, proportion of semi-natural habitat, proportion of each forest type, visibility encroachment index) showed very heavy positive skew (i.e. many low values and a long tail of higher values). To reduce skew we applied a square-root transformation to each of these variables before analysis.

Predictor variables in observational data can have a high degree of correlation, called multicollinearity, which can reduce precision of parameter estimates for the individual predictor variables. To assess and mitigate multicollinearity, we took a two-step approach. Firstly, we took the three variable categories that contained multiple variables in Table 7 (land use and terrain variables, forest type and extent variables, forest configuration variables) and in each generated a Pearson correlation matrix to look for any linear correlation between each pair of variables, taking a value of 0.7 as a threshold for 'high' correlation. We only conducted this analysis at the 1km buffer scale as this represented a larger portion of the landscape and we assumed that any correlations would be similar at the smaller 0.5 km buffer scale but with less precision available in analysis.

For land use and terrain variables, proportion of semi-natural and improved habitats within buffers were very highly negative correlated (r = -0.87) so only proportion of semi-natural habitat was used in analysis since Franks et al. (2017) found that proportion of semi-natural habitat positively influenced curlew density. There was a large amount of correlation between terrain variables, except for mean altitude which did not correlate with any other terrain variable by greater than r = 0.29. Mean slope and mean ruggedness correlated positively at r = 0.99. Since it appeared that these variables were mostly measuring a similar effect (how topographically complex the buffer area was) we decided to only keep the most intuitively simple index, that of mean slope.

There were no apparent substantial correlations between proportions of the five forest type variables, with the largest being a weak positive correlation between proportion of mature broadleaf and ancient woodland (r = 0.35). For forest configuration variables, edge density and total edge were very highly positive correlated (r = 0.99). Both were also highly positively correlated with number of patches (r = 0.70 and 0.71), so we took edge density forward as it was marginally less correlated to number of patches. Mean patch area and largest patch index were highly positively correlated (p = 0.75). As largest patch index we felt was a more intuitive index, we excluded mean patch area from analyses.

Following initial tests for multicollinearity within variable categories we had a final list of fifteen of the original nineteen potential explanatory variables, with those excluded shown with grey shading in Table 7. The second part of our analysis of multicollinearity was to generate a 'global model' containing all fifteen explanatory variables (but no interactions) and presence/absence of adult curlew as the binary response variable. The model was a generalised linear model with binomial error distribution and logit link function. A Variance Inflation Factor (VIF) was calculated for each

explanatory variable in the global model, using the 'car' package in R (Fox and Weisberg 2019). We used the rule of thumb that a VIF value >10 gives some cause for concern (Forthofer et al. 2006), and found that none of our explanatory variables showed a VIF > 7.9 (Appendix 1), so we were satisfied multicollinearity was not a problem in our variables and by extension our model sets detailed below.

Candidate models and model selection

Initially the global model was also used to have an initial examination of the potential strength of main effects of each of the explanatory variables. We used the R package 'sjPlot' (Lüdecke 2019) to plot marginal effects of each variable (the predicted effect of each variable on probability of curlew presence with each other variable held at its mean). It was clear from these plots (Appendices 2a and 2b) that square-root transformed field area was a very strong positive predictor of probability of curlew presence. Because the BWEUF method is a field-based survey (as opposed to a fixed area such as with the BTO Breeding Bird Survey) we were concerned that this relationship could have caused a methodological artefact in our analysis in that, by definition, larger fields had more chance to contain curlew territories and so they were more likely to be detected in those fields. As such, we decided to include this variable in all multivariate models (described below), so that this strong effect was accounted for while we investigated whether land use, terrain or forest variables (or their interactions) impacted probability of curlew presence on top of that effect.

Our main inferential modelling approach was to develop a set of candidate models that represented as series of competing hypotheses about predictors of probability of presence of adult curlew at the field scale. We included a null model, fifteen univariate models, and then a series of multivariate interaction models. The number of possible interactions between our predictor variables is substantial, so we restricted our analysis to two-way interactions, and only one interaction per model, to aid interpretation. For interactions we divided the variables into those we saw as providing important 'landscape context' (field characteristics, land use and terrain variables in Table 7) and those describing the 'forest context' (all other variables in Table 7 which describe the type, extent and configuration of forests), and looked for each possible pairwise interaction between variables from landscape context and forest context variable sets. A summarised list of the final candidate set of 66 models is shown in Table 8.

Table 8. Candidate set of models considered. Models are all generalised linear models with binomial errors and logit-link functions, with presence/absence of adult curlew as a binary response variable. For brevity model subsets are summarised. Square brackets indicate that in each model one of that variable set was included, with all variables in the set included once. Variables and variable sets are detailed in Table 7. Variables linked by an asterisk indicate that both variables plus their interaction term are included in the model.

Model specification	Number of models
Null model	1
All univariate models	15
√Field area*[FOREST TYPE VARIABLE]	5
√Field area*[FOREST CONFIGURATION VARIABLE]	5
√Field area + [LANDUSE OR TERRAIN VARIABLE]*[FOREST TYPE VARIABLE]	20
√Field area + [LANDUSE OR TERRAIN VARIABLE]*[FOREST CONFIGURATION VARIABLE]	16
√Field area + [LANDUSE OR TERRAIN VARIABLE]* √Visibility encroachment index	4
Total	**66**

All candidate models were ranked by their small-sample Akaike information criterion (AICc) using the 'MuMIn' package in R (Barton 2016) and inferences were taken from the smallest set of models with a summed Akaike's weight of ≥ 0.99.

Assessment of spatial autocorrelation

Top performing models were assessed for effects of spatial autocorrelation on the parameter estimates by calculating variograms based on the standardised residuals, using the R packages 'gstat' and 'sp' (Pebesma 2004, Pebesma and Bivand 2005).

RESULTS

Summary of land use, terrain and forest variables by presence/absence of curlew

Table 9 shows summaries of our predictor variables separated by fields with adult curlew in the BWEUF survey and those without, for both the 1 km and 0.5 km buffers. We have also provided summaries for the 22 fields where chicks were detected for comparison purposes. It is clear from the summaries that fields containing curlew were on average larger (Table 9a) and tended to have more semi-natural habitat (as defined in Table 5) and be slightly more likely to have moorland management present, within both the 1 km or 0.5 km buffer. Fields where chicks were detected had a very low sample size but at both scales had a higher chance of being close to moorland management. In terms of other explanatory variables, there was a large degree of overlap in terms of inter-quartile ranges between fields with and without curlew. Broadleaf woodland dominated in terms of what woodland types were within either 1 km or 0.5 km of inbye fields, particularly at 0.5 km where the medians for each other forest type were all zero.

Table 9. Explanatory variables summarised by inbye fields with and without curlew. Unless indicated values are medians, with interquartile ranges in parentheses. Some additional summary variables (shaded grey) have been added for interest or later discussion. Variables are separated by those that are specific to (a) field only, (b) the 1km buffer around each field, (b) the 0.5 km buffer around each field and (d) the viewshed of the field (up to 5 km maximum). The small sample of fields where chicks were detected has also been added for comparison.

(a) Field only

Variable subset	Variable	Without curlew (n = 30,530)	With curlew (n = 2,129)	With chicks (n=22)
Field characteristics	Field area (ha)	1.36 (0.75-2.51)	2.31 (1.42-4.23)	1.55 (1.37-1.84
	Mean slope of field (°)	5.6 (3.7-8.1)	5.3 (3.5-7.6)	5.3 (3.5-8.,2)

(b) Within 1 km buffer

Variable subset	Variable	Without curlew (n = 30,530)	With curlew (n = 2,129)	With chicks (n=22)
Land use and terrain variables	Percentage of semi-natural habitats	29.8 (13.4-52.0)	33.9 (16.8-53.6)	35.0 (21.0-53.9)
	Moorland management present?	23.5% of fields	26.9% of fields	36.4% of fields
	Mean slope (°)	6.2 (4.9-7.6)	5.9 (4.5-7.5)	5.7 (4.2-7.2)

16

Variable subset	Variable	Without curlew (n = 30,530)	With curlew (n = 2,129)	With chicks (n = 22)
	Mean altitude(m)	266 (219-318)	264 (221-318)	262 (207-314)
Forest type variables	Percentage mature conifer	0.5 (0.0-1.8)	0.6 (0.0-2.2)	0.6 (0.0-1.2)
	Percentage mature broadleaf	1.5 (0.5-3.1)	1.2 (0.4-2.5)	1.2 (0.5-2.2)
	Proportion ancient woodland	0.0 (0.0-1.4)	0.0 (0.0-1.1)	0.0 (0.0-1.7)
	Proportion young trees	0.0 (0.0-0.3)	0.0 (0.0-0.3)	0.0 (0.0-0.2)
	Proportion low density	0.0 (0.0-0.0)	0.0 (0.0-0.0)	0.0 (0.0-0.0)
	TOTAL FOREST	3.6 (1.5-8.0)	3.4 (1.4-7.3)	3.2 (1.4-7.8)
	Any forest present within 1km	96% of fields	97% of fields	100% of fields
Forest configuration variables	Number of patches	7 (4-10)	6 (4-10)	6 (4-12)
	Largest patch index (%)	2.0 (0.8-4.7)	1.9 (0.8-4.7)	1.9 (0.9-3.0)
	Euclidean nearest neighbour distance (m)	212 (154-317)	231 (168-344)	227 (148-387)
	Edge density (m/ha)	14.7 (7.3-23.9)	14.3 (7.1-21.7)	14.6 (6.0-25.1)

(c) Within 0.5 km buffer

Variable subset	Variable	Without curlew (n = 30,530)	With curlew (n = 2,129)	With chicks (n = 22)
Land use and terrain variables	Percentage of semi-natural habitats	44.5 (13.2-69.3)	49.4 (21.4-71.5)	45.9 (39.8-58.2)
	Moorland management present?	8.3% of fields	10.3% of fields	22.7% of fields
	Mean slope (°)	6.0 (4.7-7.8)	5.7 (4.4-7.7)	6.1 (3.9-7.5)
	Mean altitude(m)	263 (215-314)	263 (218-315)	261 (205-313)
Forest type variables	Percentage mature conifer	0.0 (0.0-11.1)	0.0 (0.0-12.0)	0.0 (0.0-7.6)
	Percentage mature broadleaf	9.7 (0.0-17.7)	8.1 (0.0-15.1)	11.6 (0.9-16.0)
	Proportion ancient woodland	0.0 (0.0-0.0)	0.0 (0.0-0.0)	0.0 (0.0-0.0)
	Proportion young trees	0.0 (0.0-0.0)	0.0 (0.0-0.0)	0.0 (0.0-0.0)
	Proportion low density	0.0 (0.0-0.0)	0.0 (0.0-0.0)	0.0 (0.0-0.0)
	TOTAL FOREST	16.3 (7.4-27.0)	15.0 (7.0-25.2)	17.3 (9.3-28.8)
	Any forest present within 500m	82% of fields	82% of fields	95% of fields
Forest configuration variables	Number of patches	2 (1-4)	1 (1-4)	2 (1-4)
	Largest patch index (%)	1.9 (0.6-5.4)	1.8 (0.6-5.1)	1.6 (0.8-5.7)
	Euclidean nearest neighbour distance (m)	135 (0-244)	141 (0-265)	160 (0-281)
	Edge density (m/ha)	12.8 (3.7-25.6)	11.3 (3.3-22.9)	11.8 (5.7-22.8)

(d) Whole viewshed

Variable subset	Variable	Without curlew (n = 30,530)	With curlew (n = 2,129)	With chicks (n = 22)
Interaction between forestry and topography	Proportion viewshed encroachment (%)	23.4 (10.3-41.8)	22.9 (11.2-39.8)	27.9 (20.0-33.0)

Model selection

For the 1 km buffer, one model had an Akaike's weight of 0.95 (meaning an estimated 95% probability it is the best model given the model set and data), which contained field area and an interaction between semi-natural habitat and broadleaf woodland (Table 10a). The second and third ranked models contained an interaction between altitude and either number of forest patches, or proportion broadleaf forest respectively, although their Akaike's weights were substantially lower (0.03 and 0.01 respectively).

For the 0.5 km, model selection was less dominated by a single model, and the top five models cumulatively had an Akaike's weight of ≥0.99, but with none >0.34 (Table 10b). The top two models matched models selected for the 1 km buffer (an interaction between semi-natural habitat and broadleaf woodland, and an interaction between altitude and proportion broadleaf woodland). The third ranked model contained an interaction between field area and proportion broadleaf woodland. Each of the top three models had Akaike's weights between 0.27 and 0.34, while two other models made up the cumulative set with a combined Akaike's weight ≥0.99, with Akaike's weights of 0.04 and 0.02 respectively. These contained either moorland management or slope interacting with proportion broadleaf forest.

Table 10. Model selection table for presence/absence of curlew and variables within (a) a 1 km buffer and (b) a 0.5 km buffer. For brevity, only best models with a summed Akaike's weight ≥ 0.99 and the null model are directly shown. The number of models not 'selected' that were above or below the null model are indicated. √ = square root; df = model degrees of freedom, logLik = negative log-likelihood of model, AICc = small sample Akaike's information criterion, ΔAICc = absolute difference in AIC between top ranked model and given model, w_i = Akaike's weight of model.

### (a) 1 km buffer

Model	df	logLik	AICc	Δ$_{AICc}$	w_i
Field area + √proportion semi-natural * √proportion broadleaf	5	-7554.07	15118.15	0.00	0.95
Field area + altitude * number of forest patches	5	-7557.52	15125.05	6.90	0.03
Field area + altitude * √proportion broadleaf	5	-7558.57	15127.13	8.99	0.01
… 60 models…					
Null model	1	-7851.49	15704.99	586.84	0.00
…2 models…					

(b) 0.5km buffer

| Model	df	logLik	AICc	Δ$_{AICc}$	w_i
Field area + altitude * √proportion broadleaf	5	-7553.36	15116.73	0.00	0.34
Field area + √proportion semi-natural * √proportion broadleaf	5	-7553.37	15116.75	0.02	0.33
Field area * √proportion broadleaf	4	-7554.58	15117.17	0.44	0.27
Field area + moorland management * √proportion broadleaf	5	-7555.52	15121.03	4.30	0.04
Field area + slope * √proportion broadleaf	5	-7556.43	15122.86	6.13	0.02
… 57 models…					
Null model	1	-7851.49	15704.99	588.26	0.00
…3 models…					

The marginal effects of the top ranked models (not including the effect of field area for ease of display) are shown for the 1 km buffer in Figure 2. For this scale, it is clear that the predicted probability of curlew in an inbye field is generally relatively high where proportion broadleaf woodland or number of forest patches is low, and the proportion of semi-natural habitat or the altitude has a relatively low impact on this prediction. However, at higher values of broadleaf woodland or number of forest patches, the prediction depends on the context. For broadleaf woodland, the relationship with probability of curlew presence is negative where there is less semi-natural habitat (Figure 2a) or at lower altitudes (Figure 2c), but this negative relationship is not evident at higher proportions of semi-natural habitat or higher altitudes. Similarly for number of forest patches (Figure 2b), at lower altitudes there appears a strong negative correlation to probability of curlew presence, but this does not hold at higher altitudes.
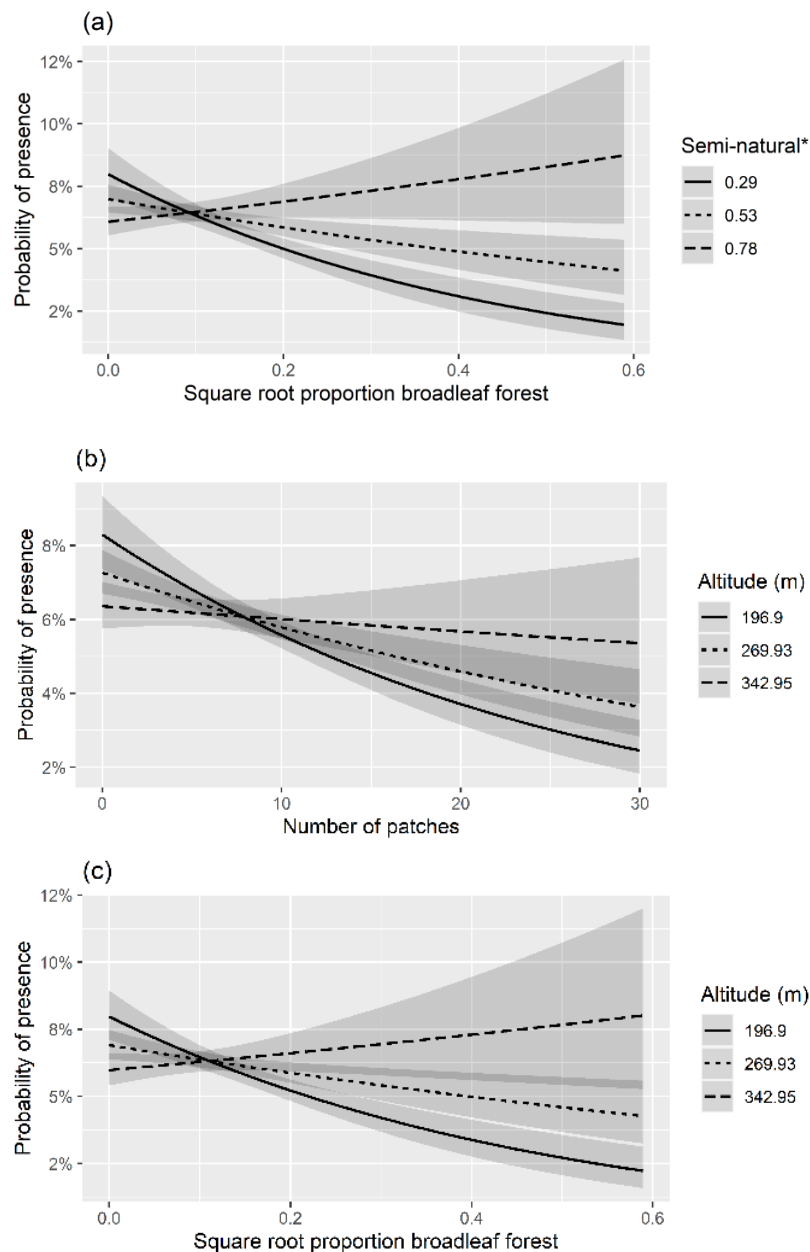
Figure 2. Predicted relationships between probability of curlew presence and, within 1 km of inbye fields: (a) interaction between square-root proportion of broadleaf forest and square-root proportion of semi-natural habitat, (b) interaction between number of forest patches and altitude,

*and (c) interaction between square-root proportion broadleaf woodland and altitude. Plots show marginal effects with square-root field size held at its mean value. Values for square-root proportion semi-natural habitat and altitude are depicted with three representative values (mean ± SD). These plots correspond to models shown in Table 10a. *this is a square-root proportion*

The marginal effects of the top ranked models (not showing the effect of field area unless it was included as an interaction term), are shown for the 0.5 km buffer in Figure 3. Figures 3a and 3b can be interpreted very similarly to the analogous models shown in Figure 2. Figure 3c shows further that the effect of proportion broadleaf woodland appears to be negative across all sizes of inbye field, but generally there is a greater probability of presence of curlew in larger fields. Figures 3d and 3e show again the apparent negative relationship between broadleaf woodland and probability of curlew presence, but indicate that this relationship is context dependent, and becomes less certain where there is moorland management in place (note, however, very wide confidence intervals) or on where mean slope within the buffer is lower (i.e. flatter local landscapes).
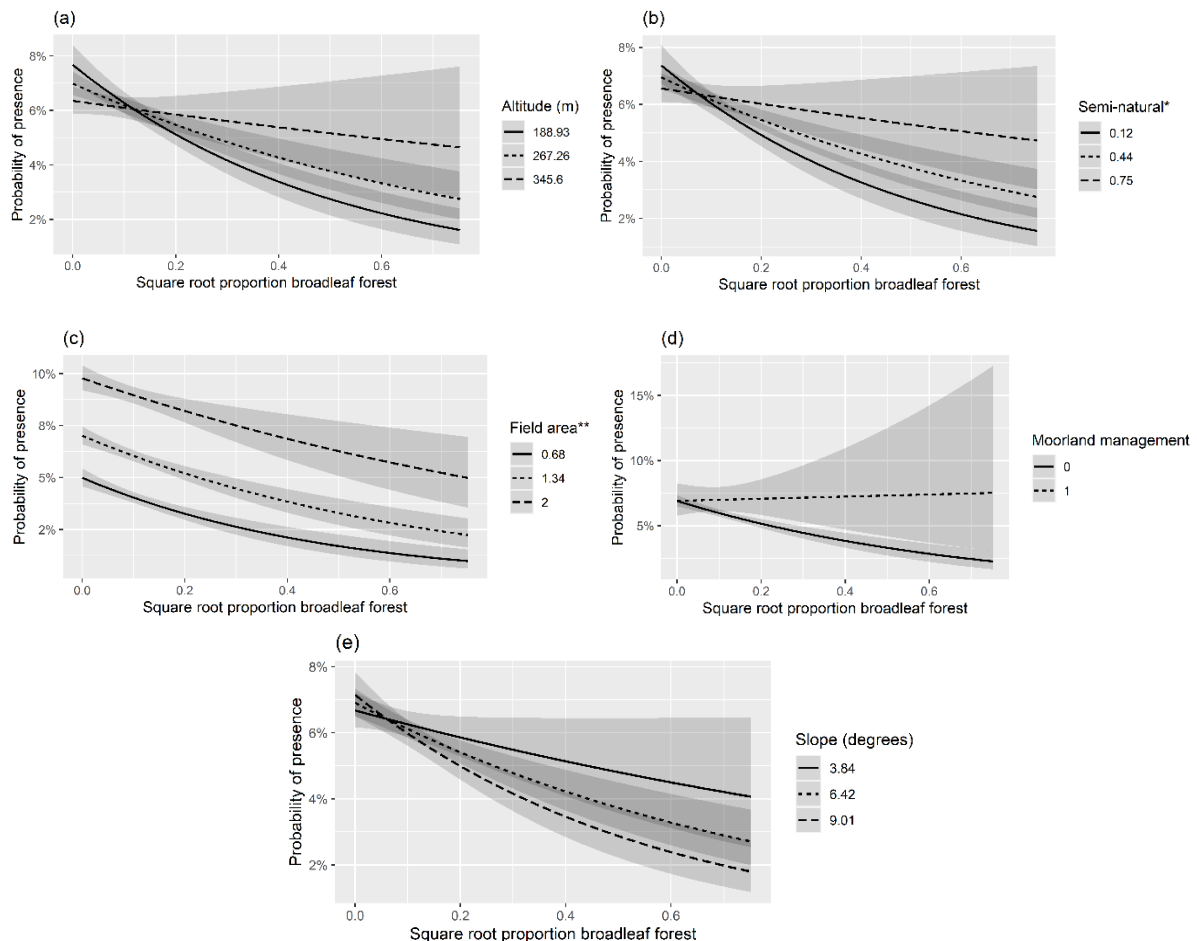

Figure 3. Predicted relationships between probability of curlew presence and, within 0.5 km of inbye fields: (a) interaction between square-root proportion of broadleaf forest and square-root proportion of semi-natural habitat, (b) interaction between square-root proportion of broadleaf forest and altitude, (c) interaction between square-root proportion of broadleaf forest and square-root field size, (d) interaction between square-root proportion of broadleaf forest and presence(1)/absence(0) of moorland management, and (e) interaction between square-root proportion of broadleaf forest and mean slope. Values for square-root proportion semi-natural

*habitat, square-root field area, altitude and slope are depicted with three representative values (mean ± SD). These plots correspond to models shown in Table 10b. *this is a square-root proportion*

Spatial autocorrelation assessment

Variogram plots of the top supported models showed no evidence of any significant spatial autocorrelation in the residuals, which would be indicated by low values of semivariance at short distances between sites. Figure 4 shows the variograms for the top supported model for the 1 km and 0.5 km buffers. For brevity, variograms for other selected models are not shown, but they showed similar patterns of lacking low semivariance at shorter distances. As such we did not include any spatial information in our models.
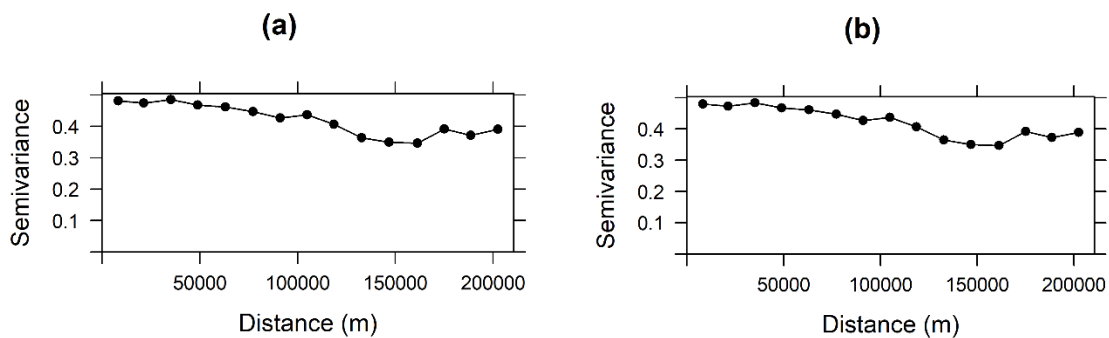
Figure 4. Variograms for top models for presence/absence of adult curlews for (a) a 1km buffer, and (b) a 0.5 km buffer. Top models are shown in Table 10. 200,000 m = 200 km.

DISCUSSION

Summary and context of findings

Although survey data were collected in a single year (2016) the BWEUF was a comprehensive survey that offers an opportunity for a fine-scale analysis across 30-40,000 consistently surveyed inbye fields. It should be noted that due to the survey methods itself, counts on moorland and other field types are optional, so were not analysed, so the impacts of curlew breeding on semi-natural habitats such as moorland and bog does not form part of our analysis, although many of the sites studied are adjacent to such habitats. Another important consideration for the context of our analysis is that broadleaf woodland made up most of the woodland, being present in more buffers, at larger amounts. Although not strong, there were positive correlations across all woodland types (i.e. in general the more broadleaf woodland present, the more likely other woodland was present) so the fact that we found relationships with broadleaf woodland might be reflective of woodland extent in general. Indeed, the proportion of broadleaf woodland and total woodland within a buffer were highly correlated (r = 0.76 at 1 km).

The key patterns from model selection within both 1 km and 500 m of inbye fields, when the positive effect of field size was accounted for, was a negative relationship between probability of curlew being present and proportion of broadleaf woodland. This was seen across almost all selected models and broadly matches with previous findings on studies on curlew and other breeding waders that have found that presence, extent and proximity of forest result in decreased probability of wader presence

(Bertholdt et al., 2017; Pálsdóttir et al., 2022; Roos et al., 2018; Wilson et al., 2014) which, in part, might be explained by reduced breeding success where forest cover and proximity increase (Batáry & Báldi, 2004; Chalfoun et al., 2002; Stephens et al., 2004; Douglas et al., 2014; Franks et al., 2017; Kaasiku et al., 2022).

The only selected model not to include amount of broadleaf woodland instead included a negative impact of the number of forest patches within 1 km of fields. This pattern was not detected within 500 m, but variation in number of forest patches was low at that distance. It should be noted that at 1 km there was a positive, but not high, correlation between number of forest patches and both proportion of broadleaf woodland (r = 0.57) and total forest (r = 0.50), so number of patches is explained by more than just amount of forest. Interestingly, this aligns to simulations based on data from Iceland in Pálsdóttir et al. (2022) which suggested that for a given amount of forest, a more fragmented structure (i.e. more patches) would more negatively impact wader populations. The mechanisms behind such an effect could relate to the impact of more fragmented habitat on behaviour of predator species and so called 'edge effects' although the evidence-base for this is not clear (Chalfoun et al. 2002).

Despite these apparent negative effects of amount and number of patches of woodland, one of the striking findings from the analysis was that these effects appeared less strong in certain land use, management or terrain contexts, such that it appeared other factors potentially mitigated negative impacts of woodland cover within the BWEUF data. Notably, for the top ranked model for both 1 km and 500 m buffers, where semi-natural habitat was more prevalent the relationship between broadleaf woodland and probability of curlew presence was less pronounced. This appears to match with findings by Franks et al. (2017) who found semi-natural habitat to be a strong predictor of curlew density (split in their study as 'semi-natural grassland' and 'mountain, heath or bog' but both showing a positive relationship).

In addition, we found that at higher altitudes, the negative effect of broadleaf woodland (both at 1 km and 500 m) and number of patches (at 1 km) were less certain (i.e. less negative and with broader confidence bands), although for the 1 km buffer the altitude models had much less support than that with semi-natural habitat amount. In these upland areas, higher altitude will likely mean closer proximity to higher altitude moorland, and indeed there was a positive, though not strong, correlation between altitude and proportion of semi-natural habitats (r = 0.48) and a negative correlation between altitude and improved habitats (r = -0.44), which could partially explain the pattern. Franks et al. (2017) also found a negative effect of arable land cover and soil organic carbon on curlew density and these factors, not directly measured here, might be linked to altitude.

Two patterns were seen within the 500 m buffer that were relatively highly ranked (fourth and fifth) but with relatively low Akaike's weights (0.04 and 0.02) were that the apparent negative effect of proportion broadleaf woodland was less certain (i.e. less negative and with broader confidence bands) when there was evidence of nearby moorland management (as proxied by evidence of moorland burning in NE's Moorland Change Map) or where the mean slope of ground in the buffer was shallower. Neither mean slope nor presence/absence of moorland management were even moderately correlated to amount of semi-natural habitat (r = 0.29 and 0.00 respectively) suggesting these are independent of the top model. The indication that presence of moorland management, which we assume to be correlated to presence of intensive predator control (although we did not directly measure this), potentially mediated negative effects of woodland is supported by a range of studies. In northern England and southern Scotland, Douglas et al. (2014) found that both curlew population change and nesting success positively related to gamekeeper density, and predicted that an increase in woodland cover from 0% to 10% within 1km of sites with curlew would require an

increase in predator control effort of 48% to ensure population stability. Franks et al. (2017) also found that that curlew abundance was positively correlated with gamebird abundance, which they used as an alternative proxy of intensity of game management. Mechanistically, Baines et al. (2023) in a paired design study found that curlew breeding success was higher in the presence of predator control and suggested that grouse moors with predator control could act as source population. Thus, it may be that through a combination of higher breeding success locally or from nearby source populations, curlew populations on inbye fields close to managed moorland were more resilient to negative impacts of woodland.

The result that curlew populations in inbye fields in areas with lower mean slopes also appeared less impacted by amount of broadleaf woodland partially matches results of Wilson et al. (2014) who found that flatter, more exposed ground was positively associated with presence of a suite of wader species in the Flow Country (although not including curlew). Mean slope was highly correlated to a suite of other terrain variables including variance of slope, variance of altitude and ruggedness, indicating that a higher mean slope means, on average, a more topographically complex landscape. Given this effect was found for the 500 m buffer only, this is referring to topographical complexity within a relatively close proximity to the field. One possible mechanism is that topographical complexity influences actual or perceived predation risk, as it might mean poorer lines of sight for foraging, displaying or nesting curlew. It is worth noting that the Visible Encroachment Index analysis was looking at a different scale (up to 5km) so is a distinct variable to this possible effect. Various studies have shown that perceived predation risk might be a factor in terms of foraging selection patterns or even nest success of various bird species (Beale and Monaghan 2004, Thornton et al. 2021) and so it could be that areas with local topographical complexity or greater mean slopes are less likely to attract breeding curlew or have lower breeding success and/or survival. The guidance document for which this report was commissioned to inform (DEFRA et al. 2023) does state slope as an important factor, stating "areas of land that have consistent slope more than 20 degrees and which do not have intervening areas of lower degree slope can be excluded from the need for survey as waders are unlikely to nest on ground with this degree of slope". Although few of our buffers reached mean slopes approaching 20°, with a mean of about 6° across all fields and a maximum of 19°, a higher mean slope would imply that there are parts of the buffer above that value, and so it may be that fields containing higher mean slopes within 500 m generally have less suitable breeding areas within the vicinity and can thus hold few breeding territories, reduces local breeding population density and thus probability of occurrence.

Implications for Woodland Creation Guidance

The 2023 'Guidance to help inform when an upland breeding wader survey is needed and when woodland creation is likely to be appropriate' (DEFRA et al. 2023) in part aims to understand potential risk to wader conservation and appropriateness/suitability of sites for woodland creation. While our previous research informed the appropriateness of new survey guidelines for assessing if a site had breeding waders, and at what density (Borthwick et al. 2023), the current research might provide an evidence-base to inform the suitability of sites where breeding wader survey information is available (i.e. Appendix 3 of DEFRA et al., 2023), and this is discussed below with reference to specific boxes within the flow charts of that report.

Implications for Appendix 2, Box 7

In the guidance, it is advised that if the slope of the ground within a proposal's footprint or 1km buffer is consistently >20° then surveys are not required, and this is justified based on the assertion that "waders are unlikely to nest on ground with this degree of slope". Our main analysis results do suggest a potential role of slope in probability of curlew presence, with areas with greater means slopes with 500 m appearing to be more negatively impacted by the amount of broadleaf woodland, although mean slope is an average taken across the area so does not necessarily show how much is above or below the 20° threshold. However, we did extract the mean slope of each individual field and found that for all fields the maximum mean slope was 30° (n = 32,446), while fields containing adult curlew (n = 2,129) or breeding pairs (n = 1,228) it was 21° and 20° respectively, and for fields where chicks were detected (n = 22) it was 14°. The low sample size for the latter (and the fact that the survey did not require recording of chicks, so these are incidental records) implies caution should be taken with respect to this 14° maximum value, although these 22 chick records were spread across 21 different tetrads in several different regions (see Appendix 3 of this report), so this is geographically quite a good representation. Frequency distributions of mean field slope for fields without curlew, with adult curlew and with chicks are shown in Appendix 4. A Mann-Witney U test showed that the mean field slope was significantly different between fields with and without curlew (W = 337,459,150, P < 0.001). It should be noted that we looked at inbye fields only, but our results would suggest that there is a very low chance of curlew presence during the breeding season where the slope is consistently >20°, supporting the guidance.

Implications for Appendix 3, Boxes 1 and 9

In the guidance, a threshold value of 1km is used to contraindicate the creation of woodland within 1km of a SSSI that has breeding waders as a notified feature or a wader recovery area (Boxes 1 and 9). If the proposal is within 1km, the applicant is directed to contact Natural England. Our results generally support this part of the guidance, since we found a negative impact of broadleaf woodland (which in our data represented most woodland) within a 1km buffer of inbye fields. Our results did find this negative impact appeared to be mitigated by the presence of more semi-natural habitat near to the fields and was not as strong at higher altitudes. The mean altitude of buffers and the proportion of semi-natural habitats were positively, if only moderately, correlated (r = 0.48) so these may represent similar drivers. Since SSSIs in the uplands will comprise semi-natural habitats, then it may be that curlew populations in these areas are less sensitive to woodland creation but if a precautionary approach is taken, we would advise this regulation is supported, particularly as there could be cumulative impacts of other landscape factors that are not considered in our analysis.

Implications for Appendix 3, Box 6

In contrast to Boxes 1 and 9, if waders are present within 1 km of a woodland creation proposal, as indicated by targeted surveys (Appendix 2 of the same document) but not in a SSSI or wader recovery area (nor is there an SPA within 2km) then woodland creation can go ahead if the proposal does not "significantly extend the predator shadow onto open land" (Box 6). Consideration of the extension of the predator shadow becomes complex because it relates to distance and configuration rather than amount of woodland. The threshold used for predator shadow in the guidelines is set at 500 m. Our data suggest a potential impact of both total amount of broadleaf woodland and number of patches of woodland within 1 km, while within 500 m we did not find an effect of number of patches, just amount of broadleaf woodland (although it needs to be considered that at 500 m the number of forest patches did not vary much between fields: Table 9c). Nevertheless, this could indicate that at 500 m, and 1 km the amount of woodland (not just its configuration or distance) could, in some contexts, be strongly negatively associated with curlew presence. Some of the example proposals given in the notes for Box 6 that are not deemed to extend the predator shadow would, however, appear to

increase woodland within 500 m or 1 km of breeding waders. Our results also suggest that a predator shadow for curlew might exist beyond 500 m. Therefore this part of the guidance might be reviewed in case it inadvertently increases risk to breeding curlew populations outside of protected areas. Our data did suggest that negative impact of woodland might be mitigated partially by presence of semi-natural habitat, and so it could be that decisions on the extent of predator shadow is considered as context dependent.

Implications for Appendix 3, Boxes 7 and 8

If there are breeding curlew within 1km of the proposed woodland, but no significant woodland cover within 500 m, but the proposal delivers habitats of high wildlife value and the biodiversity benefits are perceived to outweigh the impact of breeding waders, the proposal can go ahead. Our results give a mixed picture with respect to this part of the guidelines, because high wildlife value habitats listed include both the types of non-wooded semi-natural habitats in our study that we found appeared to mitigate the negative relationship between broadleaf woodland or number of forest patches and curlew presence, but also some high wildlife value broadleaf woodland habitats themselves.

Woodland priority habitats named in the guidance include "Wood Pasture & Parkland, Upland Oakwood, Upland Mixed Ashwoods, Upland Birchwoods, Wet Woodland and/or Native Scrubs". In our analysis we did not distinguish woodland types, so we cannot comment on the extent to which these categories made up our 'broadleaf woodland' habitat category or their relative impacts. Non-woodland high wildlife value habitats listed are: Upland Calcareous Grassland, Upland Hay Meadows, Coastal and Floodplain Grazing Marsh, Upland Heathland, Upland Flushes, Fens and Swamps, Purple Moor Grass and Rush Pastures, Blanket Bog, Mountain Heaths and Willow Scrub, Inland Rock Outcrop and Scree Habitats, Calaminarian Grasslands, Limestone Pavement." Many of these habitats overlap the Corine Land Classes we categorised as semi-natural habitats, which included 'natural grassland', 'moors and heathland', 'peat bog' and 'bare rock' (Table 5).

Overall, our results suggest some caution should be applied if the high wildlife value habitats delivered in a proposal solely consist of broadleaf woodland types, because this potentially could result in future reductions in nearby curlew breeding populations if not mediated by other factors. Our results do, however, suggest may be that if the high wildlife value habitats also include what we classed as non-woodland semi-natural habitats (and in the guidance list this would include several grassland, heath, and bog habitats) then might provide a potential buffer against possible future negative impacts of the new woodland.

In the notes for Box 8 the guidance makes a couple of assertions that our results neither contraindicate nor support. One such statement is that "tree establishment through natural colonisation or low-density planting below 600/Ha will have greater wildlife benefits". We did look at low density woodland as a variable, but since the amount within our 1 km and 500 m buffers was very low (medians of zero for both), it is likely the BWEUF 2016 data do not allow a sufficient analysis of this as a variable. A future study could be carried out when this habitat is more prevalent, or that specifically targets areas with sufficient low-density woodland that contrasts sufficiently to control areas without this habitat.

Another statement in the notes of Box 8 is that "tree establishment in areas that will not be in sightlines to areas used by waders, e.g. in cloughs and valleys, will have less impact". We did investigate this using a novel approach of assessing viewshed from inbye fields both with and without realistic assumptions about tree heights. This variable uniquely was able to measure the extent to which trees were visible or invisible in the landscape, since we would expect where trees are more

visible they will tend, on average, to reduce potential sightlines. We did not find strong support for an influence of this variable on curlew presence in the BWEUF data, although we think this type of analysis, that combines information on tree cover and topography, could be a useful future tool for studying this hypothesis. Thus, our analysis does not provide support for this assertion that woodland creation that is out of sightlines will have less impact and urge caution with this assumption since it depends on the mechanisms operating, and whether they impact curlew behaviour, predator behaviour, or an interaction between the two. For example, in theory, if woodland out of sightlines affected curlews' perception of predation risk differently to actual predation risk created from woodland, there is a risk of an ecological trap forming (Hale and Swearer 2016).

Curlew Recovery Partnership Questions

By analysing the fine-scale BWEUF 2016 data alongside a suite of other spatial datasets, combined with a review of the current literature on impacts of forest and other variables on breeding waders we have addressed aspects of the five questions posed by the Curlew Recovery Partnership (Table 2). Although geographically widespread (from Cornwall to Northumbria), by definition the core BWEUF is spatially restricted to inbye fields and a single year.

Questions 1 and 5

We found that broadleaf woodland had a generally negative relationship with probability of curlew presence. We did not find such a relationship with conifer forest or the three other forest types considered, but within the scale we considered (up to 1 km), the prevalence of these were not high, so insight into and differential impacts of woodland type (Question 1: "How does predation risk differ between woodland types?") is somewhat limited. However, our analysis did provide insight in to Question 5 ("What are the effects of extent and configuration of woodland on changes in wader abundance and productivity?"), since we showed that woodland extent (proportion of buffer area covered in broadleaf woodland) and, to some extent, configuration (number of patches of woodland) both negatively influenced probability of breeding curlew in some contexts. Interestingly, however, this effect appeared to be reduced where there was more non-wooded semi-natural habitat in the buffer as well. This latter finding is potentially important in terms of planning where impact of woodland creation might have the largest (or smallest) impacts, as discussed in terms of the guidance, above. There was not sufficient breeding data (i.e. fields with chicks) for a robust analysis of the effect of woodland extent or configuration on breeding productivity, and such an analysis may require a more targeted (but thus more spatially restricted) approach where curlew nest success and overall productivity are directly measured.

Question 2

In terms of Question 2 ("How does predation pressure impact breeding waders beyond a 1km buffer of woodland sites?") we found relatively similar effects of woodland when we looked at either a 1 km or a 500 m buffer suggesting that the negative effect of woodland extends at least this far, matching other research that have found a negative effect of woodland at 1 km distance. We did not test the effect beyond 1 km, because in discussion with FC it was decided to focus on 500 m because this value is important within the guidelines (DEFRA et al. 2023), but the framework we have developed could be extended to consider larger buffers (e.g. 1.5 or 2 km), and all of the processed datasets are available to do so.

Question 3

For Question 3 ("How does topography influence the extent of edge effects and increased predation pressure from woodland sites?"), using a unique variable intended to differentiate landscapes with woodland that was out of sightlines of inbye fields and those where it was visible, we did not find evidence that where forest is hidden from view its effect on curlew presence is modified. In fact, we found in one model, albeit not with very strong support, that the negative effect of broadleaf woodland was less pronounced where mean slope of the ground was lower, i.e. flatter landscapes. However, this does not directly measure where in the landscape woodland is placed , so this aspect requires further investigation. One approach might be to identify "paired landscapes" that are similar in terms of a range of variables (land use, management, extent and fragmentation of woodland) but differ in terms of the topographical placement of that woodland) and examine effects between pairs.

Question 4

Question 4 ("Why do breeding waders in the UK suffer more severe declines in wooded landscapes than in other European countries?") is a complex question with potentially multifaceted answers. We looked at the existing literature and our analysis to provide some potential insights. In our review of the literature we highlighted that there are some potentially important and substantial differences in land use and ecosystem context in England/UK and other Northern European countries, particularly Scandinavia, with different human population density and distribution, different predator communities, and different extents of non-wooded semi-natural habitats. In addition, woodland cover and configuration appear to differ between the UK/England and Scandinavian countries. A key finding of our study, both within 1 km and 500 m was that negative effects of woodland extent, and for 1 km, number of patches, were somewhat mediated through the composition of non-wooded habitat, namely the presence semi-natural habitats. This does suggest that the negative effect of woodland, which the literature suggests is likely, at least partially, due to increased predation risk and partly through loss of foraging habitat, can be reduced by the presence of semi-natural foraging or breeding habitats. As such differences in both predator communities (and the balance of apex and meso-predators) and non-wooded habits might partially explain differences in population trajectories of waders between UK and Scandinavian countries. A comparative detailed study of curlew populations in the UK and other northern European countries (examining in detail demography, inter-specific interactions, habitat/resource selection) might help to provide further evidence on this question.

Conclusions

With the pressure for woodland creation in England and the UK, it is imperative that a balance is found between the benefits woodland creation may bring and any potential negative environmental impacts, including those to breeding waders such as curlew. Our analysis of the fine-scale BWEUF data provides further evidence of the potential relationships between forest extent and configuration and curlew populations, and some new insights into the role non-wooded semi-natural habitats might play in those relationships. This analysis provides some new evidence to contribute to the review of the DEFRA, NE and FC guidelines on woodland creation near to breeding wader populations and contributes towards addressing several questions posed by the Curlew Recovery Partnership. More intensive studies of curlew breeding success, foraging, predator behaviour in different landscapes would give insights into the mechanism behind the observed patterns. Although we were commissioned to study curlew, the BWEUF data also contain counts of lapwing, oystercatcher, redshank and snipe, and our approach provides a framework and processed data to address similar questions across a wider suite of wading species.

ACKNOWLEDGEMENTS

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APPENDICES

Appendix 1. Variance inflation factors (VIF) from a global model for presence/absence of adult curlew and each explanatory variable within a 1 km buffer.

| Variable	VIF 1 km
Field area (ha)	1.1
Presence/absence of moorland management	1.1
√Proportion semi-natural habitat	1.6
√Proportion broadleaf forest	3.5
√Proportion conifer forest	3.2
√Proportion ancient woodland	1.9
√Proportion young forest	1.4
√Proportion low density woodland	1.1
Mean altitude (m)	1.8
Mean slope (°)	1.5
Number of patches	3.0
Largest patch index (%)	3.3
Edge density (m/ha)	7.9
Euclidean nearest neighbour distance (m)	1.1
√Proportion viewshed encroachment	1.5
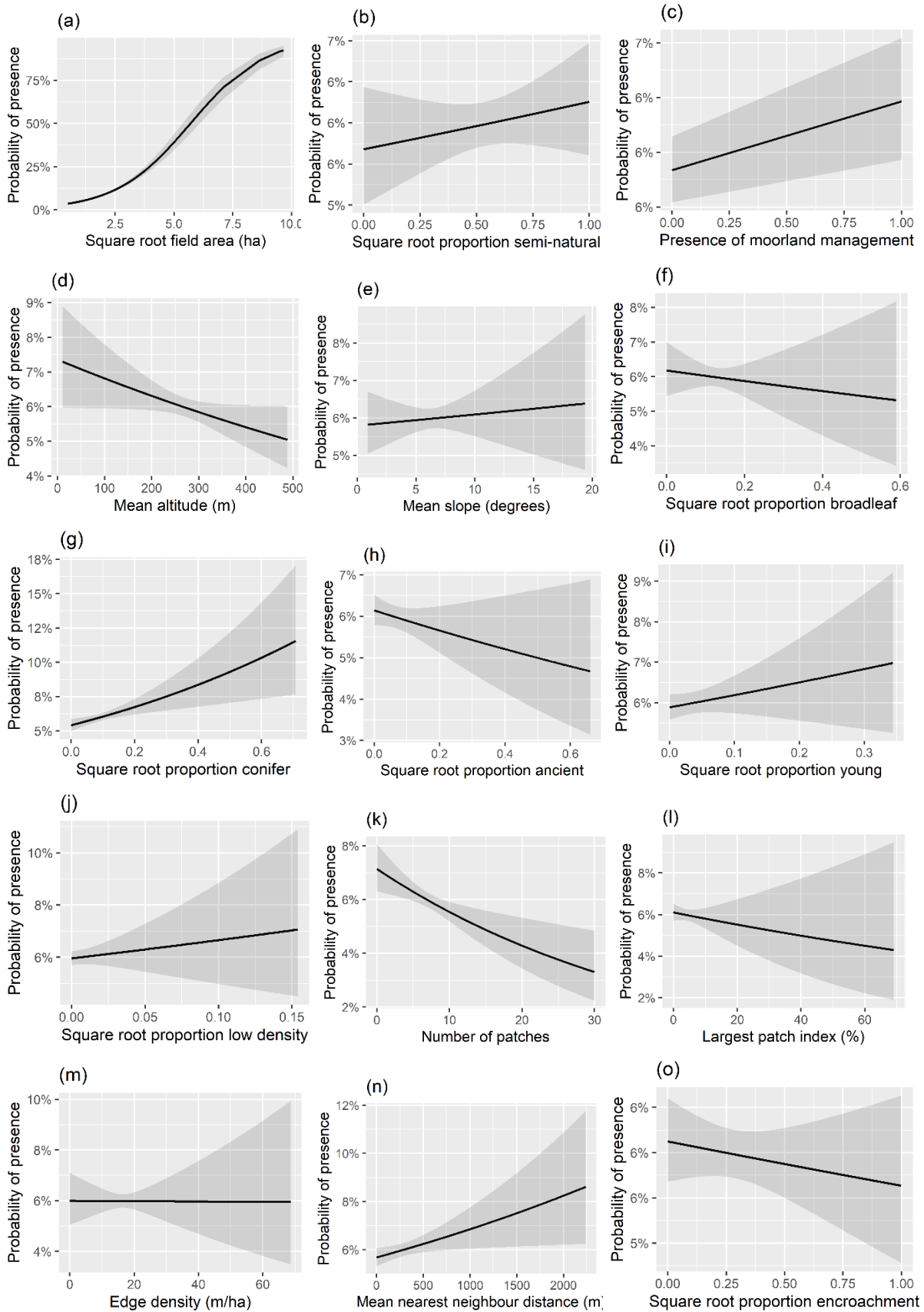
Appendix 2a. Predicted marginal effects for a 1 km buffer of all 15 terms in the global model, i.e. the predicted effect of each where each other variable is held at its mean value. Grey shading represents 95% confidence regions.
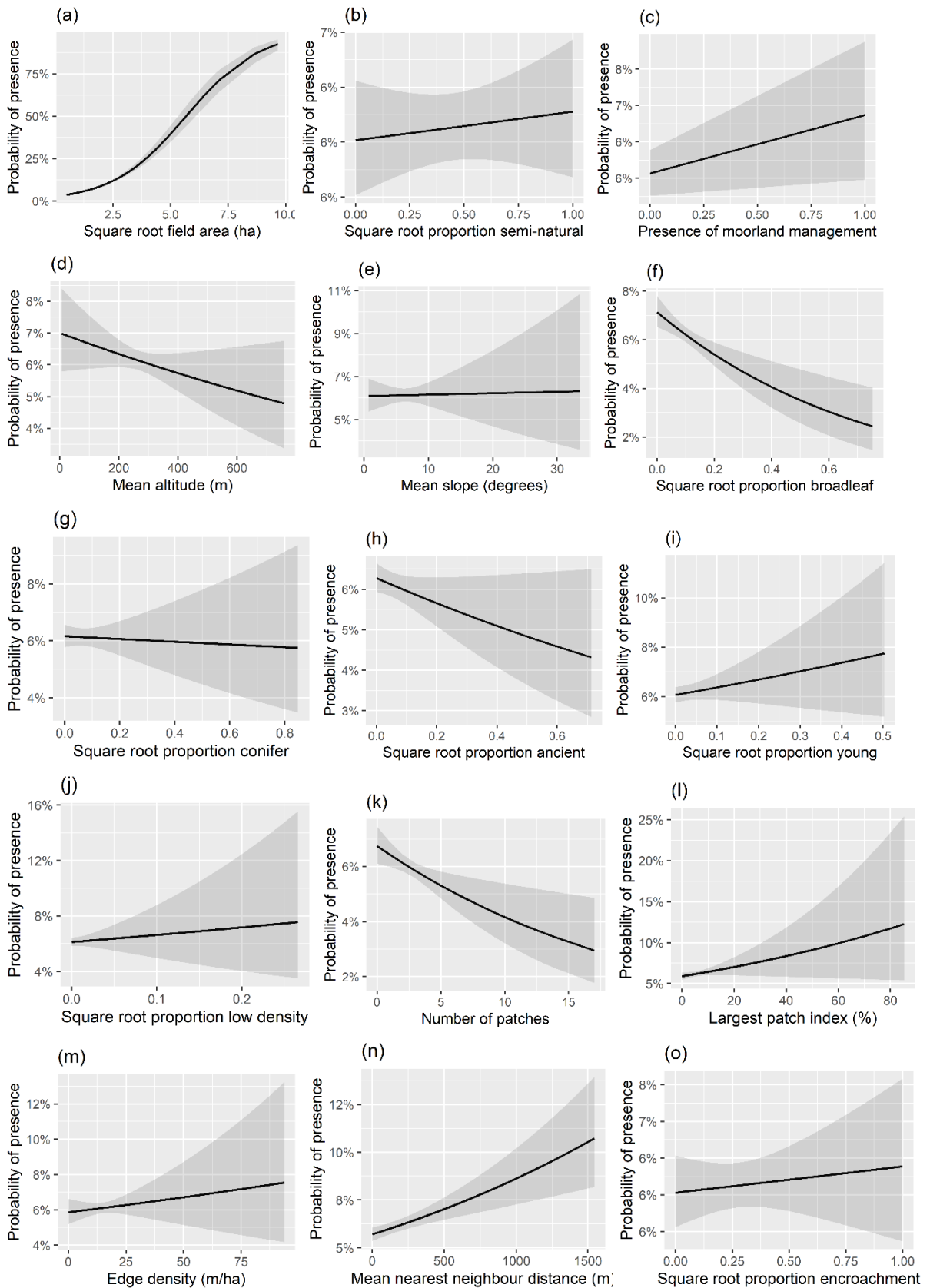
Appendix 2b. Predicted marginal effects for a 0.5 km buffer of all 15 terms in the global model, i.e. the predicted effect of each where each other variable is held at its mean value. Grey shading represents 95% confidence regions.
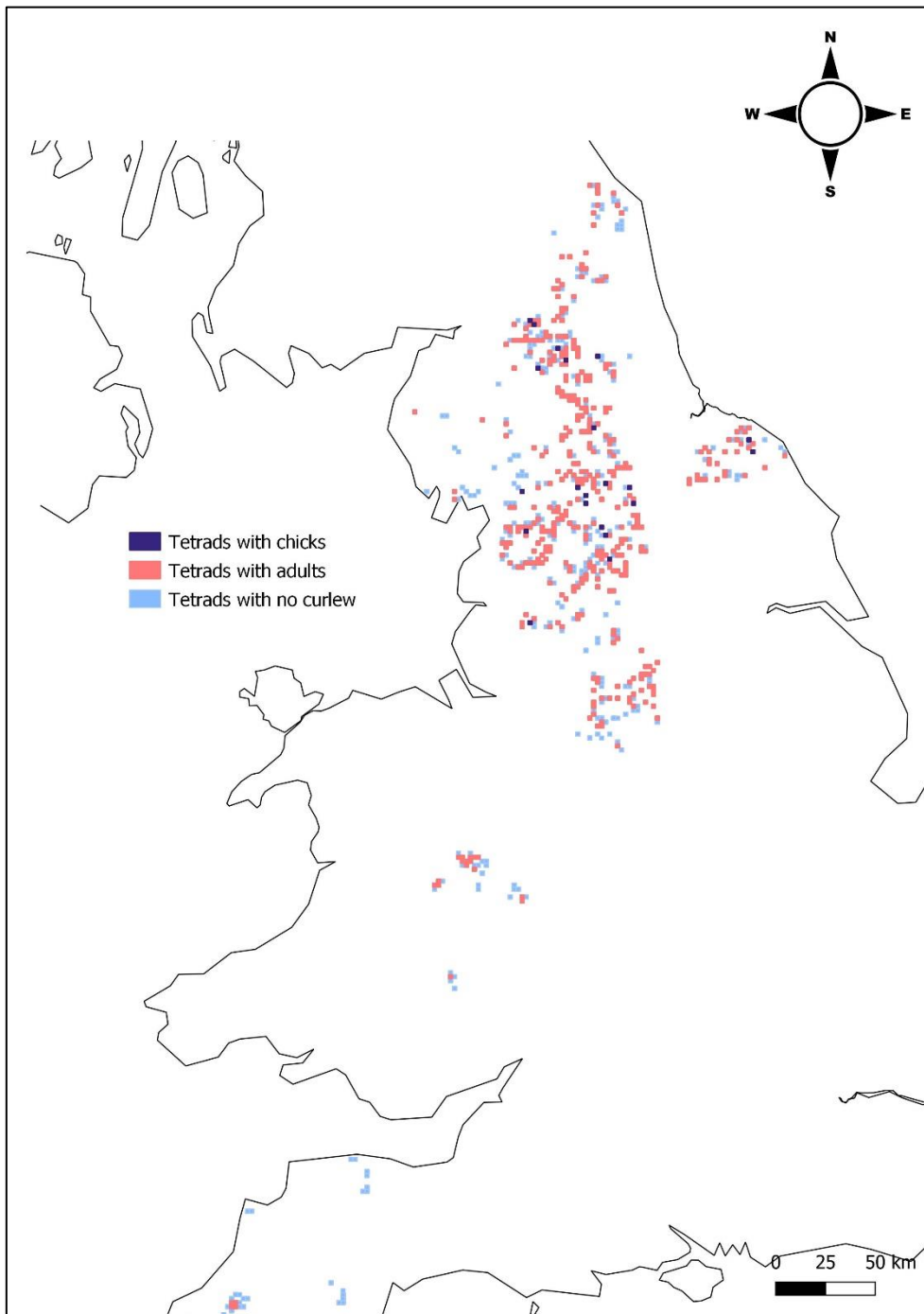
Appendix 3. Map showing distribution of surveyed tetrads. Red indicates a tetrad where there was a minimum of one curlew record in any field. Blue indicates tetrads in which no curlews were recorded. Purple represents the 21 tetrads where at least one chick record was recorded. © Base map obtained from IGISMAP (2023), curlew data obtained from BTO (2016) and tetrad data obtained using TomBio QGIS plugin Field Studies Council (2023).

(a) fields without curlew

(b) fields with curlew

(c) fields with chicks
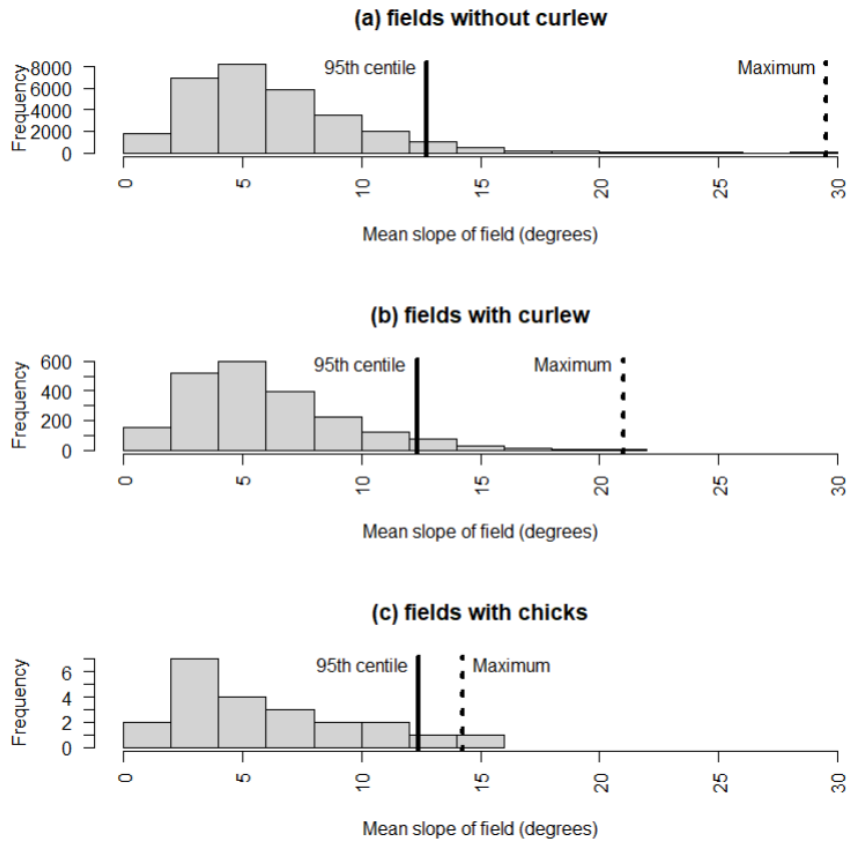
Appendix 4. Frequency distributions of fields (a) without curlew (30,317), (b) with adult curlew present (n = 2,129) and (c) with curlew chicks present (n = 22). Indicated are the 95th centile and the maximum values per distribution. Note differences in y-axis scale.