Modelling mitigation of bird population declines in the UK through landscape-scale environmental management

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Copyright Statement

Abstract

Biodiversity is declining on a global scale despite efforts to the contrary. Birds are effective indicators of ecosystem health, occurring in almost every habitat on Earth. However, many UK birds have declined since the 1960s, and are now classified as endangered or rare. Knowledge of factors influencing the presence and abundance of such species is therefore vital for their conservation. Habitat diversity affects avian diversity attesting that birds are a vital resource to conservationists. Not only are breeding birds influenced directly by their immediate habitat, they are also indirectly affected by the surrounding landscape, indicating the need for local and landscape-level studies and management.

This study takes a multi-scale approach to examine the consequences of habitat and landscape changes on bird populations in two contrasting and mixed land-use sites: heathland and woodland in the New Forest (Hampshire) and arable farmland with scattered woodlands in Cambridgeshire. Recently acquired, high resolution airborne remote sensing datasets (Light Detection and Ranging, LiDAR) were used to develop metrics that quantified vegetation structure within the two study landscapes. These variables, together with vegetation composition (recorded from field surveys) were examined in relation to a series of bird indices (density, species richness, diversity, number of declining species, conservation priority, and rarity), as species richness and diversity alone can mask effects on more vulnerable species. Relationships with bird community composition and the habitat variables were also investigated using Multidimensional Scaling (MDS). Although bird communities are known to differ between broad habitat classifications, this has not explicitly been quantified. The results from these two studies were used to predict the effects of landscape change on the bird indices and to identify the bird species affected, with a view to providing management recommendations for the relevant authorities.

The most diverse habitat in the New Forest for bird species was the scrubland (despite low bird density), represented by a positive relationship with scrubby vegetation variables, such as the percent cover of vegetation at 2-5 m and height Vertical Distribution Ratio (VDR). Beech woodlands supported the greatest number of declining bird species. Pine was overall poor bird habitat, signified by a negative relationship of the percent cover of pine with the majority of the bird indices. Other conifer was positively related to the Index of Relative Rarity (IRR), and supported

the rare Firecrest. Heathland also had a high IRR value on account of the rare Dartford Warbler, supported by a positive relationship with the percent cover of heather, indicating that alternative habitats to those that increase diversity were extremely important to habitat specialists. The habitat associations of these bird species were confirmed by the MDS analysis. Furthermore, the MDS also showed that although poor in terms of the bird indices, pine provided habitat for other rare and declining birds, including Common Crossbills.

Woodland edges in Cambridgeshire were the most diverse for bird species (a ‘classic’ edge effect), but which conversely resulted in adjacent fields being poor bird habitat. The MDS analysis showed that corvids were strongly associated with these edge habitats creating an exclusion zone. An increase in the proportional length of woody hedge vegetation in field boundaries supported more declining bird species than the other habitats. Hedges also increased (and were positively related with) the majority of the bird indices in the field-only analysis. Rarity and IRR were positively related to variables depicting woodland vegetation (percent cover of oak and vegetation height), suggesting that rare birds, such as Marsh Tits or Ravens, were in taller oak woodlands. Furthermore, a negative relationship of rarity with wood area suggests that the woodlands were sufficiently interconnected over the Cambridgeshire landscape to allow populations to persist. Overall, the MDS results showed that in both landscapes, bird community composition was more similar between the woodlands and most dissimilar between the non-woodland habitats. However, once separated, the woodlands were found to vary by vegetation composition (and habitat class) in the New Forest and by particular vegetation species and structure (scrubbier vs taller woodlands) in Cambridgeshire.

Predictions of landscape change, such as scrub removal, in the New Forest, reduced bird density, and would also reduce bird diversity, and affect scrub preferring species such as Willow Warblers. Pine removal would increase many of the bird indices, but would affect conifer specialists, Common Crossbills and Wood Warblers. Beech decline locally was predicted to reduce the number of declining bird species supported, affecting the Hawfinch population. In Cambridgeshire, declines in hedge length would reduce the number of declining bird species supported (e.g. Yellowhammers), and most of the bird indices over this agricultural landscape. The spread of improved grass would reduce species richness and diversity, and increase corvid density. Declines in oak and tree height, through tree

disease or felling, would reduce the number of rare species in the woodlands, including Marsh Tits.

Contrasting habitat composition, structure and configuration of both the woodland and non-woodland habitats in these two landscapes, results in contrasting bird indices and community composition. Unsurprisingly, the New Forest was overall better for birds, however, Cambridgeshire supported bird species that were absent from the New Forest, such as the extremely rare and declining Turtle Dove. Bird species habitat preferences also differed between the landscapes, for example, the Goldfinch was associated with conifer in the New Forest, but with hedges in agricultural Cambridgeshire.

These two landscape studies had the same conclusions; biodiversity should not be taken alone to measure habitat health as this often masks trends in rare and declining species, as represented by metrics detailing the number of declining bird species, species priority, rarity, IRR and community composition, being related to different habitat variables. This leads on to the second conclusion; that landscape heterogeneity is vital to maintain gamma diversity by providing habitat for as many species as possible. Thus, conservation should be targeted at a landscape scale and incorporate all bird measures, including conservation priority, rarity and community composition as well as diversity.

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List of Acronyms and Abbreviations

AES	Agri-Environment Schemes
AEM	Agri-Environmental Management
AICc	Aikaike’s Information Criterion (corrected)
AOD	Acute Oak Decline
ArcGIS	Arc Geographical Information Systems
ARSF	Airborne Research and Survey Facility
BBS	Breeding Bird Survey
BTO	British Trust for Ornithology
BU	Bournemouth University
CBC	Common Bird Census
CEH	Centre for Ecology and Hydrology
CI	Confidence Interval
DEFRA	Department for Environment, Food and Rural Affairs
EU	European Union
IRR	Index of Relative Rarity
IUCN	The International Union for Conservation of Nature
JNCC	Joint Nature Conservation Committee
LiDAR	Light Detecting and Ranging
NDVI	Normalised Difference Vegetation Index
NERC	Natural Environmental Research Council
OS	Ordnance Survey
RSPB	Royal Society for the Protection of Birds
SAC	Special Areas of Conservation
SE	Standard Error
SPA	Special Protection Areas
SSSI	Site of Special Scientific Interest
VDR	Vertical Distribution Ratio

List of Bird Species Relevant to this Study

Species Code	Species	Scientific Name	Species Code	Species	Scientific Name
B.	Blackbird	*Turdus merula*	M.	Mistle Thrush	*Turdus viscivorus*
BC	Blackcap	*Sylvia atricapilla*	MA	Mallard	*Anas platyrhynchos*
BF	Bullfinch	*Pyrrhula pyrrhula*	MG	Magpie	*Pica pica*
BT	Blue Tit	*Cyanistes caeruleus*	MP	Meadow Pipit	*Anthus pratensis*
C.	Carrion Crow	*Corvus corone*	MT	Marsh Tit	*Poecile palustris*
CC	Chiffchaff	*Phylloscopus collybita*	NH	Nuthatch	*Sitta europaea*
CD	Collared Dove	*Streptopelia decaocto*	PH	Pheasant	*Phasianus colchicus*
CH	Chaffinch	*Fringilla coelebs*	PW	Pied Wagtail	*Motacilla alba*
CK	Cuckoo	*Cuculus canorus*	R.	Robin	*Erithacus rubecula*
CR	Common Crossbill	*Loxia curvirostra*	RB	Reed Bunting	*Emberiza schoeniclus*
CT	Coal Tit	*Periparus ater*	RN	Raven	*Corvus corax*
CU	Curlew	*Numenius arquata*	RL	Red-legged Partridge	*Alectoris rufa*
D.	Dunnock	*Prunella modularis*	RO	Rook	*Corvus frugilegus*
DW	Dartford Warbler	*Sylvia undata*	RT	Redstart	*Phoenicurus phoenicurus*
ET	Little Egret	*Egretta garzetta*	S.	Skylark	*Alauda arvensis*
FC	Firecrest	*Regulus ignicapilla*	SC	Stonechat	*Saxicola rubicola*
G.	Green Woodpecker	*Picus viridis*	SD	Stock Dove	*Columba oenas*
GC	Goldcrest	*Regulus regulus*	SF	Spotted Flycatcher	*Muscicapa striata*
GL	Grey Wagtail	*Motacilla cinerea*	SG	Starling	*Sturnus vulgaris*
GO	Goldfinch	*Carduelis carduelis*	SK	Siskin	*Carduelis spinus*
GR	Greenfinch	*Chloris chloris*	SN	Snipe	*Gallinago gallinago*
GS	Great Spotted Woodpecker	*Dendrocopos major*	ST	Song Thrush	*Turdus philomelos*
GT	Great Tit	*Parus major*	TC	Treecreeper	*Certhia familiaris*
GW	Garden Warbler	*Sylvia borin*	TD	Turtle Dove	*Streptopelia turtur*
H.	Grey Heron	*Ardea cinerea*	TP	Tree Pipit	*Anthus trivialis*
HF	Hawfinch	*Coccothraustes coccothraustes*	WH	Whitethroat	*Sylvia communis*
J.	Jay	*Garrulus glandarius*	WO	Wood Warbler	*Phylloscopus sibilatrix*
JD	Jackdaw	*Corvus monedula*	WP	Wood Pigeon	*Columba palumbus*
L.	Lapwing	*Vanellus vanellus*	WR	Wren	*Troglodytes troglodytes*
LI	Linnet	*Linaria cannabina*	WW	Willow Warbler	*Phylloscopus trochilus*
LT	Long-tailed Tit	*Aegithalos caudatus*	YW	Yellow Wagtail	*Motacilla flava*
LW	Lesser Whitethroat	*Sylvia curruca*	Y.	Yellowhammer	*Emberiza citrinella*

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Author’s Declaration

I declare that the work presented in this thesis is my own work.

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1 Introduction

Biodiversity is a valuable resource that is declining rapidly, even with international efforts and in the wealthiest and most technologically advanced regions (Hansen et al. 1993, Butchart et al. 2010, Batáry et al. 2015). Often, biodiversity management is implemented when species reach the endangered category and they are at risk of extinction, which typically proves to be insufficient and/or too late (Hansen et al. 1993). Areas of habitat have been protected across the world as nature reserves, national parks, sites of scientific interest and to protect rare flora and fauna (e.g. Site of Special Scientific Interest (SSSI), Special Areas of Conservation (SAC) and Special Protection Areas (SPA)) as well for their aesthetic value (Wilcove et al. 1986). However, these areas are often selected for protection because they are poor agricultural land or inaccessible which can place a bias on the species protected (Margules and Pressey 2000). Furthermore, some protected areas are too small, preventing natural processes from occurring (Hansen et al. 1993, Margules and Pressey 2000) and are liable to become smaller and fewer due to demand from human populations for food and housing (Margules and Pressey 2000).

To meet these concerns, conservation needs to broaden its approach to include the wider landscape, usually dominated by farmland, and to increase emphasis on the importance of artificially created areas, such as gardens and parks, for wildlife (Gregory and Baillie 1998). Such patches of semi-natural habitat, along with nature reserves and other protected areas can constitute a network of habitats to create metapopulations, i.e. individual patches, more or less sustainable in their own right, but which are connected by the movement of animals (and plants) between patches creating a larger, divided, but ecologically functional, population (Hanski and Gilpin 1991, Hanski et al. 1997). All species may not be present in all patches all of the time, but connectivity between patches allows for recolonization should local extinction occur. Depending on size and quality, some patches (sinks) may require frequent immigration from more stable ones (sources) to maintain their populations (Harrison 1991). In essence this is an extension of the long-established Theory of Island Biogeography (MacArthur and Wilson 1967), but where the ‘sea’ between ‘islands’ is less hostile, and may sustain individuals, especially dispersers, in the short term and/or provide a non-breeding refuge and source of new individuals when space becomes available in patches (Pulliam and Danielson 1991). Connectivity between patches can be the key for metapopulation success and may typically comprise hedgerows, tree lines or a series of small patches acting as stepping stones through the landscape. However, the utility of such corridors depends on many factors including individual species' home ranges, population sizes and dispersal abilities (Lindenmayer and Nix 1993). Corridors may also have negative effects such as providing access for predators and increasing the spread of disease (Simberloff et al. 1992, Eybert et al. 1995).

The concept of Land Sharing versus Land Sparing (Green et al. 2005, Phalan et al. 2011) takes a different view in which activities damaging to biodiversity are concentrated and enhanced in areas already degraded by such activities. In the UK for example, this might entail focusing, and further intensifying, arable agriculture in eastern England. The plus side for biodiversity then comes from reducing impacts and prioritising the natural environment in areas less suitable for agriculture. One obvious downside to this approach is the impact on the people and communities who are obliged to live in the intensive areas. The wildlife of such areas is likely to become dominated by relatively few generalist species (so called 'biotic homogenisation'; Olden and Rooney 2006) with the potential to constitute a serious pest problem (Inglis et al. 1990).

Another aspect of separating natural areas from the consequences of human activities is the concept of rewilding (Vera 2000, Merckx and Pereira 2015). This allows large areas of land to revert to a more natural state which is shaped by the activities of free-ranging grazing animals and other livestock as deemed appropriate. The current flagship project for rewilding in the UK is the Knepp Estate in West Sussex (Tree 2018). Since its inception in 2003 this project has seen large gains for biodiversity and makes it clear that where land can be 'spared' more or less entirely, nature can bounce back rapidly. In concept, this approach is similar to the practice of marine conservation areas which allow fish stocks and marine habitats to recover in the absence of fishing and other forms of exploitation, and have the added bonus of acting as reservoirs and sources of fish for non-protected areas (Edgar et al. 2014).

Birds inhabit almost every niche on Earth and are an important and popular wildlife resource worthy of conservation in their own right (Baillie 1991). As relatively well studied and easy to identify organisms at the top of the food chain, birds can be used as indicator species to monitor the state of the environment and its vulnerability to drivers of change (Baillie 1991, Gottschalk et al. 2005, DEFRA 2017, 2019). Overall, UK bird populations have been in decline since the 1960s due to pressures such as climate change, agricultural intensification, increasing urbanisation, and ultimately landscape change (Fuller et al. 1995, DEFRA 2017, 2019), but the details differ across habitat types. In the UK, farmland and woodland are two of the main land uses with farmland alone comprising ~72% of lowland Britain (DEFRA 2018). Woodland cover is less extensive at ~13%, but is of increasing importance as a means of storing carbon to mitigate climate change. The BTO's Woodland Bird Indicator showed that woodland birds declined by 30% between 1970 and 2018, with woodland specialists declining by 46% in the same period (DEFRA 2019). Whilst the farmland bird decline is greater (~50%) as a result of agricultural intensification, the drivers of woodland bird decline are much less clear and are most likely due to multiple factors (Fuller et al. 2005b). Possible drivers include: a) deer damage to woodland including coppice; b) woodland maturation and reduction in woodland management – maturation and canopy closure suits some species, e.g. Marsh Tit, but not species that prefer younger stages, and canopy closure may increase negative effects of shading, less management means less variety in woodland age structure; c) reductions in invertebrate food supplies (Bell et al. 2020); d) climate change effects, e.g. changes in phenology of plants and invertebrates; e) habitat/climate changes on wintering grounds for migrant species.

Agricultural intensification after the Second World War caused rapid and ongoing declines in farmland birds (Fuller et al. 1995, Siriwardena et al. 1998a, Robinson and Sutherland 2002). This resulted in part from the removal of hedgerows to enlarge fields for the manoeuvrability of larger machinery in order to produce more food more efficiently (Krebs et al. 1999). Farming practises also changed in regards to the type of crop grown, often becoming a monoculture of cereal, at the expense of mixed farming. Such changes, along with drainage, a switch to autumn sowing and the widespread use of unregulated pesticides, contributed to agricultural areas becoming increasingly inhospitable for wildlife (Krebs et al. 1999, Robinson and Sutherland 2002, Wilson et al. 2005). The annual joint report from the Royal Society for the Protection of Birds (RSPB), British Trust for Ornithology (BTO), and Joint Nature Conservation Committee (JNCC) on the state of the UK's birds from 2015, and more recently from 2017 (Hayhow et al. 2015, 2017), and the

DEFRA report on Wild Bird Populations in the UK (DEFRA 2017, 2019) all evidence the extent of the declines.

Species declines and overall reduction of biodiversity and health of environments led the European Union (EU) and national governments to step in. Their solution was to provide a revenue incentive for farmers to enhance biodiversity within the agricultural landscape via the implementation of Agri-Environment Schemes (AES) or through Agri-Environmental Management (AEM) (Kleijn and Sutherland 2003, Kleijn et al. 2011, Batáry et al. 2015). AES operate at the farm level and are designed to improve habitat management, for example of hedgerows, and to provide new habitat, such as beetle banks, flower/grass margins or set-aside/fallows. AEM targets farmland management, such as changing to organic farming.

Whilst there have been some studies on the effectiveness of AES/AEM (e.g. Kleijn and Sutherland 2003), implementation has not always been followed up. The EU's attempts to thwart biodiversity decline, while being well intentioned, have not necessarily worked in all cases, and new goals have been set using the same techniques as before without critical assessment (Batáry et al. 2015). Chamberlain (2018) stated that not enough farmland was under the right kind of management to make a difference to bird populations at larger scales, and in order to reverse biodiversity losses governments either need to change the AES prescriptions to encourage wider uptake or to develop new initiatives. Schemes need to focus on effective prescriptions rather than offering easy options, and to ensure that their implementation is successful, e.g. that bird food patches deliver significant quantities of seed.

Habitat structure and composition at local to landscape scales determine bird community composition, distribution and individual species abundances. Thus, assessing the consequences of habitat change requires an effective means of quantifying habitat structure and composition. Field-based assessment is time consuming, expensive and very limited in extent (Hinsley et al. 2002). The use of remote sensing techniques has aided research by providing a means of acquiring high resolution habitat data at a landscape scale (Newton et al. 2009, Coops et al. 2016). As well as structure and habitat type, remote sensing can also be used to investigate landscape change and to examine the important features in promoting biodiversity in the landscape (Bradbury et al. 2005, Wallis et al. 2016, 2017).

Remote sensing includes satellite and aerial optical imaging across tens or hundreds of wavelengths of light, collecting multi- or hyper-spectral imagery (Campbell and Wynne 2011). This uses the characteristics of reflected light to assess a wide range of parameters including plant species identity, vegetation biomass (e.g. Normalised Difference Vegetation Index, NDVI), chlorophyll content, water deficit and plant health (Lefsky et al. 2002, Duro et al. 2014).

Of other remote sensing techniques, Light Detection and Ranging (LiDAR) is the most direct means to acquire vegetation structural information including the two most common measures of tree/shrub canopy height and canopy cover (Lefsky et al. 2002, Vierling et al. 2008). LiDAR data can be acquired from a satellite, but is more typically acquired from an aircraft flying over the site of interest, providing a more detailed view of the area at a resolution suitable for organism-habitat studies (Bradbury et al. 2005, Hill and Hinsley 2015). LiDAR is an active remote sensing technology which sends a laser beam of near-infrared light from an aircraft to the ground and records the timing and strength of the return signal after backscattering from ground features (Vierling et al. 2008). The return signal from a complex surface, such as a woodland canopy, will contain information from surfaces at varying depths through the canopy with the first-return (i.e. first part of the laser echo) measuring the top of the trees and the last-return (i.e. end of the returned echo) coming from the last object encountered (Broughton et al. 2012a). This could potentially be from the forest floor or above it, if the woodland contains thick shrub under the canopy preventing any of the laser pulse from reaching the ground (Lefsky et al. 2002, Bradbury et al. 2005, Broughton et al. 2012a). Once processed, LiDAR data can be manipulated as a point cloud or can be rasterized into a Digital Surface Model (DSM) and subsequently separated into a Digital Terrain Model (DTM) and Canopy Height Model (CHM). A LiDAR point cloud can provide information on the vertical and horizontal structure of a landscape, identify differently structured habitats in the landscape, and quantify variation in vegetation structure within habitats (Lefsky et al. 2002, Bradbury et al. 2005, Vierling et al. 2008). LiDAR allows data to be acquired quickly with high resolution at a landscape scale, including for remote and inaccessible locations. Acquisition may be costly, but outweighs both the capabilities and costs of field collection, and many data sets are freely available (Vierling et al. 2008, Swift et al. 2017).

Habitat assessment and monitoring requires data at both local and landscape scales. For example, Hill et al. (2014) discussed the utilisation of LiDAR data for habitat assessment and detailed the advantages of using such information to gain a three-dimensional understanding of complex landscapes and land types, such as multi-layered forest habitats. Adoption of remote sensing for ecological evaluation of habitats has been used in a number of ways such as to predict the whereabouts of a species within a landscape based on its ecology; to utilise the abundance or numbers of species in a particular location to infer habitat suitability; and to use field ecological activity data to quantify habitat quality (Hill and Hinsley 2015). Furthermore, Newton et al. (2009) stated that remote sensing will provide landscape ecologists with relevant information for the suitable implementation of agri-environment schemes, and provide evidence for conservation areas and management directives. While remote sensing is a good way to quickly gain an overview of the landscape, the use of remote sensing data in combination with field data from habitat surveys on the ground increases the accuracy of predicting diversity or habitat suitability (Rhodes et al. 2015). For example, Broughton et al. (2012a) used LiDAR and territory mapping of Marsh Tits (*Poecile palustris*), to determine vegetation structure and composition of preferred breeding habitat of the species in Monks Wood National Nature Reserve in Cambridgeshire (Broughton et al. 2012a).

This current PhD project uses landscape ecology and remote sensing methods (LiDAR) to investigate measures of bird diversity, priority, rarity and bird community composition within habitats in two contrasting lowland landscapes, namely, woodland (the New Forest) and farmland (Cambridgeshire). As discussed above, these two landscapes types comprise two of the most important, and extensive, land uses in lowland Britain representative of wider woodland and farmland landscapes, but with unique characteristics. Diversity assumes all species are equal, whereas the bird indices included in the current study measure species which are also rare or have declining populations, which are perceivably of greater importance and therefore require additional protection. Diversity measures alone may obscure community composition and the needs of those often rare, specialist species which are more susceptible to change (Miller and Cale 2000, Radford et al. 2005). This study seeks to incorporate measures of diversity but with particular emphasis on rare and declining bird species. Bird-habitat relationships and bird community models are developed and compared to investigate how such relations vary between the two landscapes. The models are also used to investigate the consequences of a number of landscape change scenarios on the bird populations of the two landscapes.

1.1 Aims and Objectives

This study will take a multi-scale approach to examine the consequences of habitat and landscape changes on bird populations in two contrasting and mixed land-use sites, New Forest and Cambridgeshire. Recently acquired, high resolution airborne remote sensing datasets will be used to develop measures that quantify vegetation structure, and will be complimented with field acquired vegetation composition metrics. At the landscape scale, measures of bird indices (density, species richness, diversity, declining species, conservation priority and rarity) and bird community composition will be assessed in relation to vegetation structure, composition and condition of the two study landscapes. Scenarios of how changes in land-use affect bird population persistence will be tested. The overall aim of this study is to provide realistic measures to develop sustainable landscapes in order to conserve overall avian diversity, whilst providing habitat for rare and declining bird species (i.e. gamma diversity).

Objective 1. Assess Bird-Habitat Relationships per Landscape Type:

1.1. Quantify landscape structure and vegetation composition using remotely sensed data (i.e. LiDAR data) and field survey in two contrasting landscapes; the New Forest and Cambridgeshire.

1.2. Record and calculate bird indices and bird community composition in the two landscapes and evaluate any differences between the habitat types within each landscape.

1.3. Determine relationships between the bird indices and bird community composition in relation to this detailed habitat analysis at a landscape scale.

Objective 2. Compare Bird-Habitat Relationships between Landscape Types:

2.1. Compare and contrast the two mixed land-use study sites (New Forest and Cambridgeshire) in terms of the bird indices and bird species composition.

Objective 3. Assess Bird Responses to Landscape Change and Possible Management Recommendations:

3.1. Use the bird-habitat relationships derived in Objective 1 to test the consequences of various scenarios of habitat and landscape change (e.g. scrub encroachment; woodland planting/felling, including pine removal, beech/oak decline and ash dieback; alteration of the hedgerow network; and the spread of improved grassland) on the bird indices and community composition.

3.2. Provide suitable, sound and realistic management recommendations for the New Forest and agricultural areas, such as Cambridgeshire, based on the results from previous Objectives.

Research Question 1. Bird-Habitat Relationships:

1.1. What habitat characteristics, in terms of vegetation structure and composition, maximise the bird indices within the two landscapes?

1.2. Do the habitat classes that support a higher number of declining and rare bird species differ from highly diverse habitats within the two landscapes?

Research Question 2. Bird Community and Habitat Relationships:

2.1. Does bird community composition differ between the habitat classes within the two landscapes and if so what bird species are driving these differences?

2.2. What habitat characteristics are influencing any differences in bird community composition: (i) considering all plots and woodland-only plots in the New Forest; and (ii) considering all transects, woodland-only and farmland-only transects in Cambridgeshire?

Research Question 3. Scenario Testing and Management Recommendations:

3.1. Using the results from Research Questions 1 and 2, how and to what extent would potential land-use changes affect the bird indices and community over the two landscapes?

3.2. What recommendations of land management and/or combination of 'best habitats' would improve avian diversity and composition of rare and conservation priority bird species?

Research Question 4. Comparison of Bird-Habitat Relationships between Landscape Types:

4.1. How do the two landscapes compare and contrast to one another in terms of the bird indices and bird community composition, particularly in similar habitat types?

4.2. What are the differences in specific bird species driving high diversity and conservation priority in the New Forest and Cambridgeshire?

2 Literature Review

The purpose of this literature review was to collate existing landscape or habitat research on birds, using the following search terms: (i) “Remote Sensing AND Birds OR Avian AND Diversity AND Habitat OR Landscape”, (ii) “Landscape Ecology AND Conservation” and (iii) “Land Use Change AND Remote Sensing AND Birds OR Avian”. Therefore, studies relating bird species diversity to habitat types and characteristics were collated and reviewed. It includes studies involving single and multiple bird species, often in homogeneous habitats, which investigate habitat selection and the specificity of particular bird species. Many of these studies are defined as landscape-scale, investigating bird community composition, and increasingly utilising advancements in technology (e.g. remote sensing) to assess habitat structural components. The review also discusses studies that focus on the challenges to biodiversity of landscape change due to human intervention and the subsequent strategies currently implemented to mediate the effects of agriculture, as the most prominent land-use in the UK.

2.1 Habitat and Bird Species Diversity Relationships

Bird species have evolved to occupy almost every habitat on Earth; they have adapted to specific habitats based on food substrate, nesting location and breeding strategies. Habitat is defined as “*the place in which an organism lives, which is characterised by its physical features or by the dominant plant types.*” (Oxford Dictionary of Biology), and is ultimately scale dependent. Habitat selection by bird species depends on the ability of the composition and structure of the habitat to meet the species requirements. It is believed that the diversity of fauna in an area, including birds, is related to the habitat diversity (Wiens and Rotenberry 1981, Poulsen 2002); increasing the variety of vegetation allows more species of bird to occupy a given area (Freemark and Merriam 1986).

MacArthur and MacArthur (1961) were the first to show that bird species diversity was related to vegetation structure; in particular the height profile of foliage density, however, bird species diversity was deemed to be unrelated to plant species diversity. Forest structure was investigated in more detail by MacArthur et al. (1962) who stated that the bird species present can be predicted from the amount of foliage in three horizontal layers in the vegetation. MacArthur et al. (1962) also found that abundance is determined by the number of suitable habitat patches, and that a greater variety of habitat patches with differing heights of vegetation increases the number of bird species present. However, once again they concluded that the variety of plant species had no direct effect on bird species diversity (MacArthur et al. 1962). MacArthur (1964) showed that vegetation layers were sufficient to account for breeding bird diversity in homogeneous habitats, but this was no longer supported if the area contained more complex habitat types, such as deciduous and coniferous patches or sparse and dense vegetation. In line with this latter finding, Adams and Edington (1973) demonstrated that bird diversity was higher in broadleaved oak (*Quercus robur*) and beech (*Fagus sylvatica*) woodlands than in coniferous woodlands, such as Douglas fir (*Pseudotsuga menziesii*) and Norway spruce (*Picea abies*), suggesting that tree species identity is a factor in bird-habitat relationships. By contrast, Wesołowski et al. (2002) found that in their survey area, coniferous woodland had the same avifauna as broadleaved woodland, but at a lower density. This may be due to the configuration of the habitat patches, but still shows that tree species had an effect on the avifauna. More recently, Ekroos et al. (2013) agreed that the correlation between bird species and plant species is weak and not significant in agricultural areas and homogeneous landscapes. However, at a landscape-scale the area will be more complex and consist of more than one habitat type, increasing the number of factors contributing to bird species diversity (Huston 1979).

Bird-habitat relationships are studied at different spatial scales, and differ in their interpretation of landscape. Therefore, studies address either, alpha diversity at the habitat or community level, beta diversity which is the measure of heterogeneity between these habitats or communities, or gamma diversity which is the regional diversity of all the habitats combined at the landscape scale. A review by Tews et al. (2004) showed that 85% of the relevant published literature reports a positive correlation with animal diversity and the habitat structural variable that was studied. The review highlighted that there was a positive influence of vegetation physiognomy (general form or appearance) and floristics (plant species) on bird species diversity and abundance and that the structural complexity in forests supports higher bird guild diversity. Poulsen (2002) also stated that increasing the number of old trees, and the variety in tree size and tree species will increase avian richness and abundance. More recently, Swift et al. (2017) found a weak relationship between avian richness and vegetation structure, and canopy composition was the most important factor affecting total species richness. This clarifies that both vegetation structure and tree species composition are significant in determining bird-habitat diversity relationships.

According to Böhning-Gaese (1997), total species richness is correlated with different habitat types at different spatial scales, and thus the spatial scale selected in relation to the organism in question is important for conservation purposes. Species diversity is also related to ecosystem and landscape diversity (Böhning-Gaese 1997), so increasing the habitat types in the landscape would increase diversity over a purely forested, albeit heterogeneous, landscape. Tews et al. (2004) thus stated that when the habitats in the landscape are sufficiently distinct, the number of habitat types will most likely influence bird species diversity. Therefore, bird species diversity must be related to habitat diversity, and landscape-scale studies within heterogeneous habitats should help elucidate this. Bird species inhabit a variety of niches, with preferred habitat expected to be species specific, therefore conservation of heterogeneous landscapes offers a means to maximise bird species diversity.

2.1.1 Bird Habitat Selection

Vegetation structure determines habitat quality, influences prey abundance and availability, predator detection and avoidance, and organism thermoregulation (Bradbury et al. 2005). The relationship between vegetation structure and foraging, particularly in farmland, often results in a trade-off between foraging success and predator detection as explored, for example, within Optimal Foraging Theory (Charnov 1976, Whittingham and Evans 2004, Whittingham et al. 2004, Butler et al. 2005). For example, small seed eaters such as Linnets (*Linaria cannabina*) were more likely to forage in stubble fields with shorter vegetation and more bare ground facilitating vigilance for predators (Moorcroft et al. 2002). Differences in bird species habitat preferences are well known, but defining the rationale behind specific species-habitat relationships is complex. Tews et al. (2004) showed that a third of the studies they reviewed on habitat-species relationships considered various habitat types, but forest ecosystems predominated. Moreover, they also found that studies in anthropogenic habitats, rather than natural heterogeneous habitats, were often dominated by anthropogenic disturbance. Landscape heterogeneity was investigated by Neumann et al. (2016) and demonstrated that vegetation composition and configuration were interrelated when considering bird species community. When studied simultaneously, composition had more relevance to the response of the bird species community than landscape configuration (Neumann et al. 2016). In addition, multiple variables influenced bird species assemblages in the landscape which could not be explained by landscape composition and configuration alone, for example, structural variables and broader variables, such as climate (Neumann et al. 2016).

2.1.1.1 Individual Bird Species Studies

Much research has explored the principles underlying individual species' habitat selection and preferences in an area of specific habitat. In woodland, for example, Hill et al. (2004) demonstrated that canopy structure affects Great Tit (*Parus major*) nestling mass and reproductive performance, via inferred effects on the abundance and availability of tree-dwelling lepidopteran prey. Therefore, vegetation structure influences habitat quality by affecting food supplies and ultimately demographic rates, such as survival. Furthermore, Hinsley et al. (2002) and Hill et al. (2004) found a significant negative relationship between Great Tit mean nestling mass and mean vegetation height in relation to poor weather conditions, indicating that a varied vegetation structure, rather than a tall, closed canopy, could offer more shelter (for both birds and prey), and thus more favourable foraging conditions in poor weather. However, Hinsley et al. (2002) found a positive relationship of mean vegetation height with Blue Tit (*Cyanistes caeruleus*) nestling mass due to the Blue Tits foraging in taller canopy vegetation (Bradbury et al. 2005). At the time of the study in 2001, there was a cold and late spring which also affected Great Tit nestling mass, suggesting that the Great Tit is possibly more susceptible to harsh and changing weather than the Blue Tit, thus demonstrating an effect of environmental variables on bird species reproductive success and habitat relationships.

Extensive research on habitat preference has been carried out on species, such as the Marsh Tit, in terms of woodland composition and structure (Broughton et al. 2012a, 2012b, Broughton and Hinsley 2015). Broughton et al. (2012b) noted, through territory mapping in the breeding season, that occupation by the Marsh Tit was positively and significantly related to the vegetation structural measures of overstorey height, tree canopy closure and understorey coverage. The Marsh Tit preferred the interior of the wood, rather than the woodland edge, but had no particular preference for any tree species (Broughton et al. 2012b). Conversely, Broughton et al. (2014) showed that in winter the Marsh Tit preferred old English oak (*Quercus robur*) which was present in the core wintering areas, but showed no relationship with any of the woodland structural variables. Variation in habitat use between seasons should therefore also be reviewed when making conservation management decisions. Huber et al. (2016) investigated the territory characteristics of the Wood Warbler (*Phylloscopus sibilatrix*), and showed an association with the structural metrics of vegetation height and vertical diversity. Furthermore, the Wood Warbler preferred broadleaved forests with a homogeneous structure in fairly steep areas on nutrient poor soils. These results indicated that management providing open and structurally diverse woodlands would be detrimental to the Wood Warbler. On the other hand, the Willow Warbler (*Phylloscopus trochilus*), preferred low mean vegetation height of 3.7-5.3 m as either early successional or open canopy woodlands and in patches greater than 0.5 ha (Bellamy et al. 2009). Kosicki et al. (2015) observed that the Goldcrest (*Regulus regulus*) and the Firecrest (*Regulus ignicapilla*) responded to tree species composition and the number of tree species in a woodland, indicating that vegetation composition could also be a relevant factor. In contrast, the Skylark (*Alauda arvensis*) is an open countryside species, with higher densities in short crops and sparse vegetation cover, and is deterred by the proximity of tall vegetation (Bradbury et al. 2005). Predictive models of Skylark abundance are greatly improved by including crop and boundary height, along with crop species and field area (Donald et al. 2001b), further indicating an influence of vegetation composition as well as structure.

Most knowledge of habitat has been derived simply by observing the species in that habitat, but often little is known regarding the reasons why a species may inhabit one and not another habitat of apparently the same type, or indeed certain parts of the area, but not its entirety. Factors in addition to habitat variables per se are also often important. For example, the species population size, the presence of other competitor species or a predator, and the wider landscape context of location of the habitat, effecting the connectivity and ability to be populated by a particular species. Even from these few bird species mentioned above, it demonstrates that they all have particular and varied habitat preferences and requirements. Some species are more specific than others; specialists require a certain environment or environmental factor to survive, whereas generalists are more adaptable to a range of environments.

Therefore, in order to increase habitat suitability for a number of species, the avian community of an area must be investigated, and to increase and protect avian diversity a variety of habitat types is required, most likely on a large scale.

2.1.1.2 Multiple Bird Species Studies

Birds have evolved to occupy certain habitats and yet two areas which appear to be similar can contain a different bird assemblage. Since the composition and structure of vegetation has been shown to be important in determining bird species diversity (Welsh 1987, Böhning-Gaese 1997, Estades 1997, Poulsen 2002, Tews et al. 2004, Neumann et al. 2016) and habitat selection in birds (Caprio et al. 2009), a number of studies have investigated the variables that affect avian diversity in single habitat types. Some studies have detailed an avian community by studying a number of species in a particular habitat and the relationships with habitat features, such as vegetation structure. For example, Anderson and Shugart (1974) investigated the avian community in an east Tennessee deciduous forest by examining each of the 28 bird species present against 28 habitat variables independently of one another, to determine which variables influenced each bird species, and observed obvious habitat preferences for families of birds. They then analysed 13 of the more common species and discovered that some were distributed based on specific habitat variables. Bird community was also investigated briefly in three dimensional space using a Principal Components Analysis (PCA; Anderson and Shugart, 1974). This process would have been very time consuming, conducting hundreds of analyses for each species individually, constituting a conservative avian community analysis.

More recently, studies have investigated avian community structure as a whole, in relation to the habitat, with sophisticated statistical analysis. Sallabanks et al. (2006), for example, studied bird communities in Grand fir (*Abies grandis*) forests in Oregon, USA, and found that certain habitat features explained variation in composition and abundance of the bird species. They determined that the most important features accounting for an increased variance in relative abundances of birds, were at the habitat level (canopy cover, density of large-diameter trees and understory structure), reiterating that vegetation structure is a major factor in species habitat selection. In broadleaved woodlands, vegetation structure has also been shown to affect bird species composition and distribution. The difference between young and older oak woodland can be apparent, whilst a detailed metric, such as canopy height, can be used to determine the more subtle structural variation in mature woodlands, and possibly account for inconsistent species composition (Hinsley et al. 2009b). Caprio et al. (2009) focused on habitat structure, configuration and connectivity of deciduous woodland habitat patches and their relationship with guild parameters of the birds in winter and spring, and showed that retaining native oak in these woodlands is important to conserve specialist bird species. Specialist birds were correlated with oak biomass, but generalist birds were not, and birds were primarily influenced by tree species composition (in terms of biomass and distribution) and secondarily by the shape of the patches, regardless of the tree species.

The study by Cadieux and Drapeau (2017) looked at a functional group of cavity and bark nesting bird species that relied on deadwood trees in old mixed and coniferous boreal forests in Canada. They investigated variables that affected this functional group along a forest age gradient in coniferous and mixed woodlands and established that it responded highly to structural diversity, degradation stages and diameter of trees. They also analysed individual species which revealed that the quality and quantity of dead trees accounted for the presence of certain species. Cadieux and Drapeau (2017) concluded that the old mixed forests should be a conservation priority as they are highly important to bird species that rely on deadwood in this region. This contradicts Adams and Edington (1973), who stated that deciduous, broadleaf woodlands were more productive and species rich than coniferous woodlands. However, this confirms that single habitat types are not preferential to all bird species, and that tree species composition and the structural diversity of the habitat are both important in bird-habitat selection. Moreover, Hanzelka and Reif (2016) investigated the effect of non-native tree species on bird species richness, and whether non-native trees weakened the relationship between bird species richness and vegetation heterogeneity. Although non-native trees provided heterogeneity in the habitat, there were other limitations to avian diversity with these tree species, such as reduced food availability. However, the effect on bird community composition of tree origin was small compared with the importance of the leaf morphology of the tree (i.e. being coniferous or broadleaved). They therefore recommended that increasing vegetation heterogeneity of native trees will maximise avian species richness and maintain specialist birds (Hanzelka and Reif 2016).

Some researchers have studied the ecology of single habitats in order to improve avian diversity, understand the characteristics that are required for avian-specific habitat selection, and for conservation purposes (Riffell et al. 2001). For example, Sauerbrei et al. (2017) investigated the effect of hedge cutting for biofuels on hedge dwelling bird species. The analysis was carried out on 25 bird species grouped together using a cluster analysis, into 3 groups of similar hedge preference from an extensive literature review. The first group, containing the Yellowhammer (*Emberiza citrinella*), preferred long and broad hedges, whereas the second group, containing the Blackbird (*Turdus merula*), preferred high hedges, and both groups preferred hedges with trees (Sauerbrei et al. 2017). The third group, containing the Common Whitethroat (*Sylvia communis*), preferred small hedges with gaps. This indicates that a single prescription does not exist, as bird species have different preferred hedge characteristics, supporting the rationale for heterogeneous habitats with varying vegetation structure. Sauerbrei et al. (2017) also found that all the studied bird species, regardless of their preferences, were affected by hedge cutting which caused a reduction in hedge suitability.

Bird-habitat relationships have also been investigated in habitats other than forest. For example, Riffell et al. (2001) investigated bird-habitat relationships in a wetland ecosystem and noted that a suite of characteristics in this habitat type, such as vegetation density, structural diversity and the presence of a particular plant species, were related to higher values of bird abundance and bird species presence. Wiens and Rotenberry (1981) studied shrub-steppe environments in North America, which are low rainfall natural grasslands with enough moisture to support grasses and shrubs. They ascertained that variations in bird abundances were more related to the coverage of different shrub species and that bird species richness increased with increasing structural diversity. Between-habitats the birds responded to habitat configuration, but within the habitat, vegetation floristics influenced bird species associations, in line with the theory of alpha and beta diversity. Wiens and Rotenberry (1981) concluded that they could not guarantee that their findings were completely reliable due to other biotic factors, such as predation, effecting avian population structure, but still demonstrated that bird-habitat relationships are influenced by habitat composition, structure and configuration.

Ecologists have utilised evidence from research in bird-habitat relationships to predict abundance and or composition of the avifauna in a particular habitat type.

Further investigations have also used modelling to unveil the most relevant habitat feature(s) for the species, in order to advise conservation and management strategies (e.g. Sallabanks et al. 2006, Sauerbrei et al. 2017). Furthermore, the effectiveness of management strategies varies between regions as well as between species, suggesting that over-arching national scale strategies may not be the answer (Whittingham et al. 2007). However, this multiple-species analysis does allow for a more thorough insight into avian community structure within habitats. There are complex explanations for habitat selection and preference at smaller ecosystem scales, therefore regional strategies may be more practical (Whittingham et al. 2007). However, as with all ecological studies, additional factors in the wider landscape also affect avian diversity and population abundance, such as the connectivity between similar habitats, enabling the distribution and genetic movement of birds between populations. The variety of suitable habitats across the landscape is also important, along with the level of disturbances and agricultural practises, including deforestation, silviculture, grazing and monocultures, and proximity to urban areas, which need to be factored into habitat analyses.

2.1.2 Landscape-Scale Studies

Avian communities are influenced by multi-dimensional vegetation structure, composition and landscape configuration and bird species select their habitat using these traits (Bergen et al. 2007). However, studies often neglect the effect on avian community composition of the habitat in a wider context, for instance: the configuration of habitat in relation to other suitable and also non-suitable habitat patches; the ability of species to disperse to suitable habitat utilizing connecting patches (colonization and distribution); and the distance between suitable habitat patches (isolation), specifically, over landscape-scale features.

In order to broaden the field of view, studies have encompassed landscape information as well as smaller scale habitat information. The habitat context or home range of species in the landscape can then be examined to provide a more detailed explanation for the presence of said species or the community composition in a certain area. However, landscape-scale can be interpreted at various sizes depending on the spatial extent of an organism's habitat. An invertebrate's landscape would be perceived as much smaller than that of a large mammal.

In the context of avian ecology, landscape is also perceived in a number of ways in various studies. Neumann et al. (2016) used 2 km by 2 km tetrad survey sites spread out over the landscape to investigate the avian community over a woodland-agriculture gradient in relation to habitat configuration as well as the composition of vegetation. They concluded that multiple landscape scales need to be examined as some birds do not respond to a 2 km by 2 km scale, and found no relationship between spatial correlation and landscape heterogeneity (Neumann et al. 2016). Sallabanks et al. (2006) analysed data from a range of forest stands between 16 and 213 ha, investigating landscape-scale and habitat-scale features. They maintained that whilst the habitat level features accounted for more variance in relative abundance of avifauna, the landscape level features (stand isolation and percentage of surrounding forest cover in a 4 km radius) also accounted for some of the variance and can enhance abundance predictions (Sallabanks et al. 2006). Huber et al. (2016) agreed, stating that habitat requirements must be addressed at different spatial scales for conservation purposes, as using solely coarse, large scale analysis will provide incomplete data.

The scale at which landscape metrics and bird responses were analysed has been shown to affect some correlations, often weakening correlations and creating the opposing trend (Mayer and Cameron 2003). Mayer and Cameron (2003) demonstrated that bird species richness was correlated with mean patch size and number of forest patches at several scales. Bird diversity was mainly related to mean forest patch size, suggesting that the larger patches contributed higher bird abundances, as opposed to a greater number of forest patches. The landscape metrics that Mayer and Cameron (2003) analysed mainly portrayed the configuration and the number of forest patches in the landscape (which they related to the abundance, diversity and richness of birds), not necessarily encompassing the entirety of the landscape. Bellamy et al. (1996) also analysed a range of scales in order to investigate the effects of small woods on bird species in the landscape. Woodland area accounted for 70% of the variation in breeding species numbers, and the length of the perimeter influenced edge species numbers, with the smaller woods containing more breeding edge species and fewer breeding woodland species. Interior woodland species, such as the Marsh Tit, have been found to favour larger woodlands which contain their preferred habitat and were often found in fewer numbers in small woodlands (Broughton et al. 2012b). Buchanan et al. (2016)

carried out a long-term study to investigate whether the diversity and abundance of woody plant species still explained changes in bird populations when landscape metrics were also considered. Vegetation species composition and diversity of trees and shrubs had the strongest relationship with most bird species groups, but to protect these bird communities, they suggested that the whole landscape must be taken into account due to the negative relationships of some bird guilds to the index of ‘forest edge to forest interior’.

Böhning-Gaese (1997) investigated the fact that at small, habitat scales (e.g. 0.0025 to 0.4 km^2), habitat diversity affects avian biodiversity, whereas at larger, geographical scales (e.g. 400 to 50,000 km^2), variables relating to available energy become more important. They also found that avian species richness was influenced by habitat diversity rather than by the available energy at intermediate spatial scales between 4 and 36 km^2. Böhning-Gaese (1997) discovered that at the smaller size of the intermediate range, bird species richness was affected by forest area and at the larger size it was affected by the presence of uncommon habitats such as water, cliffs or bogs, providing evidence that additional habitat types in a woodland environment can increase avian diversity in the landscape. Species diversity was also found to be associated with diversity both at the individual ecosystem and landscape-scale, therefore dictating that both scales should be protected in terms of conservation strategies (Böhning-Gaese 1997). Hinsley et al. (2009b) indicated that in order for heterogeneity in the landscape to be able to support higher bird diversity, the birds must have the ability to travel the distances required to locate suitable habitat patches and/or be able to survive in smaller patches. Many studies have characterised the woodland across landscapes and the connectivity between woodland patches. However, the avian community as a whole over multiple land-use types and at a large landscape-scale (e.g. hundreds of square kilometres), and also over multiple landscape types, has not been fully studied.

2.1.2.1 Bird Community Composition – Landscape-Scale

Landscapes comprise numerous types of habitat each containing a variety of species adapted and suited to each particular habitat. The habitats must be suitable and also accessible in order for colonization to occur. As previously mentioned, the co-habitation by multiple species in each habitat creates a community of bird species which are dependent on the habitat, and are affected by the presence of other species. Each community will be unique due to a number of factors, but fundamentally as a result of habitat composition and structure.

A number of studies have investigated bird species composition at landscape-scales using diversity indices and species richness. Seoane et al. (2017), for example, found that species richness depended more on the number of individuals than on habitat or climate; however, this varied with habitat type. They concluded that the two mechanisms they were testing were not sufficient to account for the variability in the populations. Instead, the way in which the species fill the available space (niche packing) determines the energy input, translating to species richness through abundance. Seoane et al. (2017) continued that structurally complex habitats provide foraging niche division which allows for a greater abundance of species, therefore increasing species richness in that habitat. However, their habitat data were based on estimations of percentage land-cover, (i.e. trees, scrub, grassland or rock), assuming that this variable effected bird niche occupation. Typically, birds are thought to be affected by the level of structural complexity in the habitat, although Seoane et al. (2017) did not quantify the habitat vertical structure, therefore neglecting the discrete dissimilarities between them. They assumed that the simpler habitats, such as grassland, would have lower species richness due to the lack of structure (Seoane et al. 2017). However, they found that the energy inputs explained more in the low-energy environments, and vegetation complexity was the main driver in high energy environments, endorsing the need for further structural analysis.

At the 25 hectare scale, Heikkinen et al. (2004) showed that the distribution of important habitat was related more to the abundance of bird pairs rather than the landscape heterogeneity. Heikkinen et al. (2004) used models to investigate landscape effects on avian diversity, and concluded that in order to avoid biased results, the relationships between avian diversity and landscape structure must also take the ecological importance of different habitats into account. This elucidates the need for landscape-based studies to encompass every aspect of the landscape, including habitat features, in order to gain complete understanding of the mechanisms involved with the bird species composition in each landscape. However, the use of diversity indices, including species richness, does not provide a complete picture of the avifaunal community as it can mask changes in species composition and can obscure losses in specialist species which are more sensitive to landscape change (Miller and Cale 2000, Radford et al. 2005).

Some studies have looked at the individual bird species occurring in a particular landscape in order to evaluate biodiversity for management purposes (Hansen et al. 1993, Hannah et al. 2017). Hannah et al. (2017) investigated habitat level and landscape level variables on bird species in a forest landscape and used priority bird species to indicate the health of that ecosystem for other birds. The landscape-scale of the study was 155,046 ha, however, only one habitat type was assessed, longleaf pine (*Pinus palustris*; Hannah et al. 2017). Thus, focussing on priority species was useful for the habitat they were found in, whereas, in a landscape of mosaic habitats in various proportions, single species may not wholly represent the bird community in every habitat due to various other factors. Gregory and Baillie (1998) studied eight bird species using data from the whole of Britain to investigate habitat preference across the country. Whilst this was a great use of the Breeding Bird Survey (BBS) data collated by the British Trust for Ornithology (BTO), the species studied preferred different habitat types which varied over the country, therefore management would need to be addressed at the landscape-scale and within the context of each landscape. They also disregarded the wider bird species community due to the overwhelming scale and the broad scope of the study.

Ultimately, community analysis allows for a more detailed understanding of the habitat and the extent of the species diversity it can maintain, therefore depicting habitat quality. Investigating bird composition in the landscape is important to evaluate the health of ecosystems. However, this has not been carried out in great detail. In order to gain a complete understanding of the ecosystem, future studies would benefit from encompassing the landscape variables along with habitat metrics, including structural components of the habitat and the composition of avian species present.

2.1.2.2 Remote Sensing for Landscape-Scale Studies

Since the research by MacArthur and MacArthur (1961) found that bird species diversity was related to habitat vegetation structure, studies have been developing methods to analyse the response of birds to vegetation structure (e.g. Caprio et al. 2009, Hanzelka and Reif 2016, Swift et al. 2017). Early studies calculated vegetation structure by hand using field-acquired vegetation surveys (e.g.

MacArthur and MacArthur 1961, Riffell et al. 2001, Sallabanks et al. 2006). However, this was time consuming, costly, and difficult to collect in inaccessible areas, and often lacked accuracy (Bradbury et al. 2005, Clawges et al. 2008).

Advances in technology and the adoption of remote sensing techniques has allowed vegetation structural data to be acquired more easily and can efficiently cover larger areas, providing the opportunity for research into large scale habitat-species studies (Newton et al. 2009, Coops et al. 2016). One advancement has been the use of Light Detection and Ranging (LiDAR) data, described in Vierling et al. (2008) and Lefsky et al. (2002), which enables the full three dimensional structure of habitats to be quantified (Clawges et al. 2008, Hill et al. 2014, Hill and Hinsley 2015). LiDAR data can supply information on the vertical and horizontal structure of habitats with the main focus often being on woodland habitats (Tews et al. 2004, Newton et al. 2009). Metrics can be derived to describe vegetation structure, such as canopy cover openness or closure, maximum and average height of trees, information on the understorey detailing the height and depth of below-canopy features, the permeability or density of the canopy from laser penetration depth, and ground level topography. Use of this remote sensing technique has been widely adopted in forest ecosystem studies and has facilitated the process of habitat assessments (Clawges et al. 2008, Müller et al. 2009, Hill et al. 2014). Hill et al. (2013) showed that LiDAR-derived structural variables can predict forest habitat types with a similar accuracy as ground data on soil, vegetation composition and climate. Müller et al. (2009) also demonstrated that the predictive power of LiDAR was similar to that of aerial photography and superior to field acquired metrics. Goetz et al. (2007) found that LiDAR metrics performed better than optical remote sensing at describing habitat heterogeneity, while combining the two did not improve the result. Hill and Hinsley (2015) investigated the relationship between Great Tit mean nestling body mass and vegetation structure using LiDAR data in an area around each nest. They found a significant relationship between mean nestling body mass and structural variables of mean canopy height, mean overstorey height and the standard deviation of canopy height, demonstrating a correlation between habitat structure and reproductive success (Hill and Hinsley 2015). Hill and Hinsley (2015) concluded that in relatively stable habitats, such as mature woodland, LiDAR data can be used reliably to investigate organism-habitat relationships. Goetz et al. (2007) used satellite acquired LiDAR metrics to investigate the influence of canopy structural diversity on bird species richness in a temperate forest in the US, and found that canopy vertical distribution information had the strongest influence on bird species richness. LiDAR data have been used to assess the structural components of the habitat surrounding nest placement in species, such as the Marsh Tit (Broughton et al. 2012b). The presence of breeding Marsh Tits and the relationship with habitat structure has also been investigated using LiDAR data to gain a greater understanding of habitat preference in Monks Wood National Nature Reserve in Cambridgeshire (Broughton et al. 2012a). There was a positive relationship with some of the LiDAR metrics as well as a relationship with vegetation species, and the Marsh Tit was also found to be affected by proximity to the woodland edge (Broughton et al. 2012a). Bellamy et al. (2009) utilised the vegetation height profile from LiDAR data at the location of bird sightings to identify suitable Willow Warbler habitat. Model prediction was moderate or good. For example, in the three study woods, birds were recorded in 0%, 7% and 30% of locations predicted to be unsuitable habitat.

In order to investigate landscape-scale bird-habitat relationships a number of studies used satellite imagery to estimate the percentage land-cover (Osborne et al. 2001, Radford et al. 2005). Radford et al. (2005), for instance, used Geographic Information Systems (GIS) software to map and calculate habitat cover variables over the landscape (tree cover extent, altitude, habitat condition, number of patches etc.) and patch configuration was calculated using FRAGSTATS (McGarigal et al. 2002). This was to test the relationship between habitat cover and configuration on bird species richness to calculate a threshold value of cover needed to maintain bird species richness over the landscape (Radford et al. 2005). However, there are other factors in the landscape and also at the habitat level which will be affecting the bird species. Radford et al. (2005) deduced that in order to conserve woodland birds in the landscape the species' demography must also be taken into account, such as the bird population age structure, breeding success and trends in population size. Fuller et al. (2005a) also used remote sensing at a landscape-scale; they used satellite imagery to classify habitats in south eastern England, and associated the bird species to the habitats and vice versa. Lindbladh et al. (2019) also used remotely sensed satellite imagery to investigate differences in bird diversity and community composition in pine and Norway spruce reserves of different ages. They found distinct but overlapping communities, and found that the older spruce (80 years old) had higher diversity due to the higher proportion of broadleaved trees and volume of dead wood (Lindbladh et al. 2019).

Other studies, such as Bergen et al. (2007), used radar to derive forest biomass along with satellite (Landsat) imagery to derive vegetation types. They investigated the relationship between bird presence and vegetation type, with descriptions of the surrounding vegetation and vegetation biomass, but stated that further work should be carried out using remote sensing methods to quantify the forest and landscape structure. O'Connell et al. (2015), on the other hand, examined agricultural landscapes using remotely sensed high spatial resolution colour infrared aerial photography (CIR) to classify the non-cropped area for habitat suitability. Duro et al. (2014) calculated the area of vegetation in an agricultural landscape using the Normalized Difference Vegetation Index (NDVI) from multi-spectral satellite imagery (from the Landsat series of satellites), and predicted the diversity of birds, butterflies and plants.

A review by Newton et al. (2009) showed that of the literature investigating landscape ecology at the time, only 36% mentioned remote sensing, and of those the most common technique was aerial photography and Landsat satellite sensor images, which have not changed substantially since the 1970s. This suggests that ecologists have been slow to recognise the potential of remote sensing and to gain a more comprehensive understanding of habitat selection by birds at a landscape-scale, finer details of habitat features, as well as the features themselves must be included in the analysis. A review by Nagendra et al. (2012) agreed that remote sensing techniques have not been used to their full potential and the use of spatial imagery in the form of Landsat data has been the main provider of data for assessing habitat. This is attributed to its capability of gaining broad information on habitat change and disturbance (Nagendra et al. 2012). However, it is less able to provide the finer details in the habitat relating to habitat quality, species distributions and fine-scale disturbances. The review also stated that LiDAR has the potential to collect data on the three dimensional structure of the vegetation and these data should be incorporated into habitat suitability modelling as, at the time of the review, most studies focussed on two dimensional type of vegetation, and disregarded the structure (Nagendra et al. 2012).

Some studies, such as Rhodes et al. (2015), stated that both remotely sensed data and field survey data should be used collaboratively to achieve the most accurate predictions of bird distributions. However, the remotely sensed data they used in the analysis was of low spatial resolution and satellite based, providing only broad habitat classifications and no landscape feature data (Rhodes et al. 2015). Kosicki et al. (2015) also noted that remote sensing alone was not enough to explain the complex features of habitats, and that models based on both remote sensing and detailed environmental factors concerning the species were better predictors than either alone. Wallis et al. (2016) investigated the use of both LiDAR and satellite imagery to model the distribution of bird diversity in a tropical rainforest ecosystem. They found that both LiDAR and satellite imagery predicted bird community and concluded that multispectral data could replace costly LiDAR for some biodiversity modelling.

Numerous studies demonstrate individual species relationships to the structure of the habitat using remote sensing in terms of LiDAR (e.g. Barnes et al. 2016, Huber et al. 2016, Huber et al. 2017). Remote sensing data, such as LiDAR acquired vegetation features, have also been used to investigate the avian diversity in a particular habitat and for habitat assessment (e.g. grassland by Besnard et al. 2015, woodland understory by Martinuzzi et al. 2009). LiDAR can also be used to look at the wider landscape structure due to its capabilities of capturing a large area in a relatively short period of time, when compared to field collection. There are also large costs with this data collection, but Clawges et al. (2008) and Swift et al. (2017) stated that the cost of acquiring LiDAR data far outweighed the cost and time of collecting the data manually through field based methods. Bradbury et al. (2005), for example, demonstrated that LiDAR can be used to predict certain species distribution, such as the Skylark in a farmed landscape, and Blue Tits and Great Tits in woodland. They referred to the landscape as greater than 10 ha, and stated that future LiDAR data needs to be at a higher resolution over a larger spatial extent to continue with landscape-scale studies (Bradbury et al. 2005). Müller et al. (2009) agreed that due to the accuracy of LiDAR data, it should be incorporated into ecological studies of bird-habitat relationships. The majority of landscape remote sensing studies observed the landscape features using imagery, often from satellites with low spatial resolution, and produced a broad classification of habitat neglecting the fine details that LiDAR can provide through the full vertical and horizontal profile of the habitats. Research in landscape ecology using remotely sensed data for multiple bird species in multiple habitats is lacking and is a research area to be address in more detail in the future.

2.2 Landscape Change

Over the past century the landscape has changed dramatically as a result of anthropogenic behaviour, with advances in technology causing agricultural intensification, and with increasing urbanisation (Fuller et al. 1995). The European landscape has been managed by humans for almost the last 10,000 years through the cultivation of crops, grazing of livestock for the provision of food and the felling of trees for timber or coppicing for poles (Batáry et al. 2015). Consequently, there is very little natural land left and much of the countryside in Europe is artificial with the open landscape being maintained by farm animals and agriculture, rather than indigenous grazers and natural disturbances (Batáry et al. 2015). Up until the Second World War, wildlife often benefited from landscape change such as agriculture (e.g. the Skylark; Bradbury et al. 2005), but since then rapid changes, particularly in agricultural practises and urbanisation, have significantly altered the landscape, ultimately affecting indigenous avifauna (Robinson and Sutherland 2002, Hayhow et al. 2015, 2017).

Birds have been used as indicator species to evaluate the health of ecosystems, as they are easy to detect and identify and can be indicative of overall biodiversity change (Harrison et al. 2014). Avian species have either adapted or suffered as a result of landscape change, causing shifts in avian species' ranges and declines in many farmland and woodland bird species (Fuller et al. 1995, Vanhinsbergh et al. 2003, Fuller et al. 2005b). Harrison et al. (2014) showed declining trends in biodiversity in southern and eastern England, but found more positive trends in northern Britain in line with the north-south gradient. They also found positive changes in common species and a negative change in rarer species, possibly owing to the adaptability of more common generalists in comparison to rarer specialists (Hinsley et al. 2009b, Harrison et al. 2014). Forestry practises have also changed, causing habitat loss, isolation and fragmentation (Caprio et al. 2009). Farmland birds have seen the greatest declines in the UK over the last 60 years due to changes in agricultural practises (Hayhow et al. 2015, 2017), but many species have suffered as a result of landscape change, mainly attributable to the human population.

Farmland birds were affected by the rapid changes in agriculture, whereas woodland bird decline could not be explained by a single factor or overall changes in land management (Vanhinsbergh et al. 2003, Fuller et al. 2005b). Surprisingly, the declines occurred when the habitat was favourable, due to trends in forestry increasing the wooded area, and when management became more environmentally aware (Fuller et al. 2005b). Many of the declines occurred in woodlands where management and tree species had not changed in decades. The review by Fuller et al. (2005b) suggested seven possible factors explaining woodland bird decline in Britain: increased pressures during migration or in winter for migrants; climate change effecting the breeding areas; reduction in invertebrate food; land use impacting woodland edges and habitats next to woodland and hedges; reduced woodland management; increasing grazing pressure by deer; and increased predation pressure from Grey Squirrels (*Sciurus carolinensis),* Great Spotted Woodpeckers (*Dendrocopos major)* and corvids. Vanhinsbergh et al. (2003) further suggested that each bird species was unlikely to be influenced by a single factor, and that multiple factors were most probably affecting each species simultaneously to reduce reproductive success and survival.

Climate change has been debated for decades. The warming of areas has caused species to shift their ranges to find cooler, more appropriate habitats (Mason et al. 2015), but some species cannot move and adapt to change as easily as others resulting in their decline and, at worst, their extinction (Bradshaw et al. 2014, Barbet-Massin and Jetz 2015). Climate change affects the landscape in an alternative, indirect way to other physical, more direct landscape changes which will be addressed here. Along with the changes in climate, the human population is affecting biodiversity in a number of ways, and conservation priorities should be addressing these effects in order to reverse the declines. The following section will review research on the effects of the various aspects of landscape change on birds.

2.2.1 Anthropogenic Change

As well as the effects of climate change, the main anthropogenic effects on avian species are agricultural practises and urbanisation. Urbanisation is causing problems due to the ever-declining amount of ‘green’, natural space for wildlife. With the human population set to rise to 9 billion by 2050 the required space will be increasingly sought after and is predicted to double in order to house and provide food for the human population (Godfray et al. 2010, Tilman et al. 2011).

Birds, as indicator species, are often used to measure the state of ecosystems and, due to some species’ adaptability, they have utilised urban areas to their advantage. Miller et al. (2003) found that bird communities and local habitat conditions were affected by the increasing urbanisation in their study area. Bird species richness was also found to decrease with increasing urbanisation and was also affected by human activity, such as trail use (Miller et al. 2003). Melles et al. (2003) also showed that species richness declined with increasing urbanisation, and stated that the configuration of local habitat within the landscape may be as important as the composition of the habitat. They concluded that urban development must incorporate parks, reserves and the surrounding urban area into the planning process. Neumann et al. (2016) demonstrated that generalist bird species that nested in holes or foraged on the ground were positively correlated with urban land-cover, whereas specialist broadleaved bird species avoided landscapes with urban areas. Feeding of birds in urban areas has increased in popularity for aesthetic, wellbeing and educational purposes, and can help to subsidise the food for birds that the landscape is lacking (Cox et al. 2016). However, recent studies have also suggested that this may be detrimental to bird fitness and be increasing the spread of diseases, such as trichomonosis (Lawson et al. 2018). Other bird species nest on buildings, feed on anthropogenic food waste and utilise the small green spaces in gardens, parks and golf-courses for example, as a substitute for their natural habitat (e.g. Merola-Zwartjes and DeLong 2005). Gregory and Baillie (1998) stated that these areas should not be neglected by conservationists as they still provide refuges and habitats for some bird species, and also pollinators (Baldock et al. 2019).

Anthropogenic actions have caused the landscape to change dramatically, altering the land to cater for the human population. This has caused disruption and drastic changes to many bird habitats, and in some cases has resulted in habitat loss, and consequently birds, from the landscape. Foley et al. (2005) observed that land-use transition varies in intensity and timing across the world. They clearly showed that the largest proportion of landscape usage (at the intensive land use stage) was intensive agriculture, which is the main cause of farmland biodiversity decline (Benton et al. 2003). They also stated that humans have altered the Earth in order to supply fresh running water to homes, for irrigation of crops and for industry (Foley et al. 2005). The area of irrigated cropland has doubled over the world in the last 50 years, which mostly likely rerouted and degraded water courses (Foley et al. 2011). Global fertiliser use has also increased by 500% in the last 50 years, and an increase in energy use, and pollution through leaching of fertilisers and by-products from factories has also degraded water courses and habitats (Foley et al. 2011). Therefore, the main focus of the following sections is the effects attributable to land management regimes, land-use change and agricultural practises that influence the immediate avian populations.

2.2.1.1 Agriculture

Agricultural land covers approximately 38% of the Earth's terrestrial surface (FAO 2014 in O'Connell et al. 2015): 12% (1.53 billion ha) is cropland and 26% (3.38 billion ha) is pasture land (Foley et al. 2011). After the Second World War, mechanisation resulted in the intensification of farming due to technological advancements, allowing for larger machines to cultivate larger areas of land more effectively (Fuller et al. 1995, Siriwardena et al. 1998a, Robinson and Sutherland 2002). In Britain, this resulted in the removal of 50% of the hedgerows and the advent of extremely large monocrop fields (Robinson and Sutherland 2002, Foley et al. 2011, Batáry et al. 2015), which resulted in the intensification at the field scale and landscape scale (Batáry et al. 2011). Agriculture has also expanded into new areas, which has removed natural ecosystems globally, including 70% of grassland, 50% of savannah, 45% of temperate deciduous forest and 27% of tropical forest (Foley et al. 2011). In the 1960s the introduction of broad spectrum pesticides and herbicides to increase the crop yield caused a reduction in wild flowers which provided seed and insect food for farmland birds (Smil 1999 in Batáry et al. 2015). Dichlorodiphenyltrichloroethane (DDT) was a successful and efficient pesticide. However, after prolonged and widespread use it was discovered that it bioaccumulated in the food chain causing egg shell thinning and the rapid decline of many bird populations, with top predators (raptors) being worst effected (Ratcliffe 1967). More recently it has been discovered that neonicotinoids, also used as widespread pesticides, have lethal effects on bees, which in turn affects other organisms, including a decrease in pollination of flowers and crops (Kremen et al. 2002, Whitehorn et al. 2012, Scheper 2015). The altered timing of agricultural practises has also affected birds, such as earlier harvest, earlier ploughing and autumn sewing of crops, which results in a lack of available winter food usually provided by spilt seed and crop stubble left in fields (Robinson and Sutherland 2002). Furthermore, advances in crop breeding and improvements to grassland have increased crop density, affecting invertebrate prey density and seed availability for birds, as well as predator detection and avoidance (Vickery et al. 2001, Wilson et al. 2005)

Agricultural intensification has had proven consequences on European bird populations (Benton et al. 2003, Bradbury et al. 2005, Wilson et al. 2005, Vickery and Arlettaz 2012). Due to the numbers of farmland birds almost halving over the last forty years, it is suggested that they may now prefer alternative habitats (Fuller et al. 2005a). Tucker and Evans (1997 in Wilson et al. 2005) found that agricultural habitats contained more European bird species of conservation concern than other habitats. Additionally, Heikkinen et al. (2004) showed that the total number of bird pairs was negatively related to agricultural land and positively related to forest cover and landscape heterogeneity. It is not only birds that have been affected by the changes, but also plants, invertebrates and mammals (e.g. Smart et al. 2000, Smith et al. 2005), influencing the entire food web. Donald et al. (2001a) demonstrated that the anthropogenic effect on biodiversity is detectable at a continental scale and is comparable to deforestation and climate change. Rodewald and Yahner (2001) compared the effects of agriculture and silviculture on bird community composition, and found that the extent of the disturbance was less of an influence than the type of disturbance. Agriculture caused fewer species to be present in the forest areas, and therefore caused a greater effect on bird community composition (Rodewald and Yahner 2001). Haslem and Bennett (2011) reported that 65% of the regional species were found in agricultural mosaics in Australia and that the presence of wooded areas increased the numbers of bird species.

Nelson et al. (2011) stated that a balance must be achieved between the conservation of avifauna and agricultural practises, such as cattle grazing. In order to meet the human demand for food and provide a sustainable environment, food production must increase substantially, whilst decreasing the environmental footprint caused by agriculture (Foley et al. 2011). Gregory and Baillie (1998) noted that of the bird species they surveyed, 50% had populations in agricultural land and they suggested that sympathetic changes in farming would help to conserve other species, whilst demonstrating that agricultural land is important for a number of species. For instance, the Skylark, probably benefitted from farming practises originally, but the recent changes in farming to quick growing crops of uniform height, has now caused the species to be in decline (Bradbury et al. 2005).

2.2.1.2 Semi-Natural vs. Natural Habitat

Agricultural land is often a mosaic of artificial, natural and semi-natural land, with natural areas now sparse and difficult to reinstate once removed. Studies have looked at whether semi-natural land is a suitable replacement, in order to restore and conserve avian biodiversity (e.g. Hinsley et al. 2009b, Neumann et al. 2016). In agricultural areas it is predominantly the non-cropped land (such as trees, hedgerows and grassy margins) that provides avian habitat for breeding and foraging for most species (O'Connell et al. 2015). O'Connell et al. (2015) showed that the abundance, quality and configuration of non-cropped land affected their usefulness for ecosystem services. Similarly, generalist bird species, as well as specialist bird species, were found to struggle in highly modified secondary habitats, such as small woods, parks and gardens, and suffered from decreased breeding success and increased energy costs for parents (Hinsley et al. 2009b). Haslem and Bennett (2011) demonstrated that native woodland vegetation was critical for the conservation of bird species in agricultural environments, and careful management of scattered trees and plantations could provide beneficial habitat for woodland-dependant bird species. Estrada et al. (1993) investigated the effect of agriculture on birds in Mexico, and found that pastures were deficient in forest bird species, but the agricultural areas that grew coffee, cacao, citrus, pepper and mixed crops, acted as temporary habitats and links between the small forest patches. Moreover, Heikkinen et al. (2004) found a positive relationship between semi-natural grassland and the number of agricultural bird pairs. The total number of bird pairs was found to be positively related to conifer cover and negatively related to cultivated field cover, stressing that semi-natural habitat and even coniferous woodlands were important for increasing avian diversity and abundance. This suggests that, depending on the type of agriculture, semi-natural habitat can enhance and support bird populations, providing there are enough suitable habitats, such as woodlands, in close proximity to allow dispersion between the populations (Gregory and Baillie 1998).

Around 90% of forest cover in Britain is plantation (Donald et al. 1997, Hartley 2002), and most European woodlands having been altered at some stage. In Britain, 43% of all woodland cover was coniferous in 2010 (Forestry Commission 2013).

Calladine et al. (2018) discussed the effects of ‘un-natural’ forests, which contained mainly non-native conifer trees, on birds. A number of factors affected bird species assemblages in the conifer plantations, such as the surrounding habitat and the responses of species to vegetation structure and composition (Hewson et al. 2011, Calladine et al. 2018). Adams and Edington (1973) suggested that conifer forests were more impoverished than broadleaved woodlands for birds, but found that this was only partly the case as bird species diversity, but not the abundance of birds, differed between the two. Furthermore, Hanzelka and Reif (2016) stated that birds were affected more by leaf morphology and their specialisation to conifers or broadleaved trees, than whether the tree species was native or non-native. Since the heterogeneity of tree species increased bird diversity, they suggested increasing tree species heterogeneity in native forests, but reducing non-native woodlands to prevent the loss of specialist bird species (Hanzelka and Reif 2016). Caprio et al. (2009) also noted that introduced non-native vegetation altered bird community composition and nesting success. If non-native vegetation occupies a substantial area, this may reduce invertebrate prey and predator populations (Hanzelka and Reif 2016). In contrast, McNab et al. (2019) found that in Scotland, the Common Crossbill (*Loxia curvirostra*) was preferentially utilising non-native Sitka spruce (*Picea sitchensis*) in place of native conifers, as a result of the physical properties of the cones and seeding phenology, which allowed easier access and higher energy consumption. Furthermore, Calladine et al. (2018) noted that a number of bird species with low or declining populations, such as the Goshawk (*Accipiter gentilis*), Capercaillie (*Tetrao urogallus*), Firecrest and Crested Tit (*Lophophanes cristatus*), benefited from mature plantations.

2.2.1.3 Disturbance and Deforestation

Changes in land-use practises have contributed to changes in the global carbon cycle and consequently climate change (Foley et al. 2005). The destruction and deforestation of tropical rainforests has resulted in increased CO_2 emissions and greenhouse gases that would normally be locked away, or sequestered into the growing trees (Foley et al. 2007). It is estimated that clearing tropical forests releases approximately 1.1 x 10^{15} grams of carbon as greenhouse gases each year, which is 12% of the total anthropogenic CO_2 emissions (Foley et al. 2005). Since 80% of agricultural land gained in the 1980s-1990s came from the deforestation of intact and undisturbed forests, instead of already cleared land, the resultant loss of tropical rainforest, as well as contributing to climate change will also result in the loss of biodiversity (Foley et al. 2007, Gibbs et al. 2010). Furthermore, Foley et al. (2007) stated that the deforestation of the Amazon rainforest not only results in the loss of trees, but also damages the surrounding forest through drying out of the forest floor, increasing forest fires, and lower productivity. In addition, this results in the degradation of ecosystem services, such as carbon storage, the regulation of water balance and river flow, regional climate moderation and the ability to prevent the spread of infectious diseases (Foley et al. 2007). Gamfeldt et al. (2013) added that increasing the number of tree species in a production forest was advantageous to promoting ecosystem services.

For a long time, forestry management focussed on wood production with little thought to the value of forest ecosystems (Caprio et al. 2009). Rodewald and Yahner (2001) showed that the disturbance caused by agriculture resulted in the adjacent forests having fewer forest bird species present and negatively affected bird communities, more so than disturbance by silviculture. Nesting success was also greater and nest predators were lower in forests with silviculture than forested landscapes with agriculture (Rodewald and Yahner 2001). They therefore suggested that the type of disturbance, rather than the extent of the disturbance, should be considered in conservation and management strategies. Moreover, Hansen et al. (1993) demonstrated that managing the forest for maximum timber production would negatively affect the bird community and suggested multiple-use management to support both birds and timber production.

A study by Ram et al. (2017), showed forest birds on average had more increasing trends than non-forest birds in their study site in Sweden, and suggested that recent legislative changes in forestry practises, improving forest quantity and quality, have contributed to the increases. Changes included increasing middle-aged and mature forest and improvements in forest features, including dead wood, retention trees on clear cuts, multi-layer forests, old forest and broadleaved forest (Ram et al. 2017). Poulsen (2002) also found that an increase in old trees, tree species, and tree sizes, as well as site quality and chance of colonization, were correlated with an increase in bird species richness and abundance. Due to the reduction in forest cover, the source habitats maintaining species in isolated patches may suffer if the distance between patches is too great or the species is sedentary (Poulsen 2002).

The effects on birds of different forest management and silviculture practises were discussed in Fuller and Robles (2018). Management such as clear felling, where a large patch (>1 ha) is felled on rotation, results in patches of similar growth stages; whereas continuous felling removes a few trees in a forest whilst retaining canopy cover and little change to forest structure. Paquet et al. (2006) noted that clear cut, open areas in artificial coniferous plantation forest and edge habitats held higher conservation value and species richness, compared with the forest interior and agricultural land. They also found that the open, felled and replanted areas of the forest did not contain a mix of woodland and agricultural species, as expected, but rather had its own suite of bird species. Therefore, the clear-cutting of these plantations provides temporary habitat for early succession-preferring bird species, replacing the open land that has been lost to the plantations (Costello et al. 2000, Gram et al. 2003, Paquet et al. 2006). Welsh (1987) showed that following tree cutting, new growth was rapid, but bird density and abundance did not follow the pattern of vegetation succession. Some species persisted throughout the stages of succession, whereas, the majority of bird species followed the pattern of growth and were often only found in one stage resulting in turnover within communities. Caprio et al. (2009) agreed that clear-cutting can increase bird diversity by providing successional habitats, but their results suggested that overall diversity in a forest context may be misleading. Forest integrity should be maintained, as logging non-native tree species in their study area had a negative effect on forest change and biodiversity. Calladine et al. (2015) also demonstrated that clear felling provides habitat for ‘young-growth’ species, and that continuous cover felling, where only patches were felled maintaining the canopy cover and overall structure of the woodland, increased bird diversity, whereas Costello et al. (2000) found the opposite trend. However, Du Bus De Warnaffe and Deconchat (2008) found no difference in bird assemblage diversity between the two techniques in conifer plantations, and also found no differences in bird assemblages when applied to beech forests. This demonstrates that not all disturbances have a negative effect on avian populations and deforestation for timber production can be maintained through suitable management practises. The management practises, however, must be effectively carried out and prior investigation and knowledge of the habitat is required to sustain the local, endemic bird population.

Disturbances and deforestation have major impacts, not only on the global climate, but on the ecosystem services the forests provide and also directly on the avifauna that inhabits them. For centuries humans have benefitted from forestry, often at the expense of the ecosystem and biodiversity. As a result of thorough research, management schemes can now be implemented to allow the continuation of sustainable forestry practises for timber whilst maintaining the forest for the avifauna.

2.2.1.4 Isolation and Fragmentation

Land-use change, particularly as a result of agriculture, caused a reduction in size of many habitats which were split up or disappeared from the landscape altogether. Consequently, as well as habitat loss, this causes isolation or separation of habitats of the same type, fragmenting into plots dispersed throughout the landscape. Fragmented habitat often deteriorates as, for example, what was once a large heterogeneous expanse of forest, loses the original heterogeneity when split into fragments (Wilcove et al. 1986). Many species, including birds, suffer as the habitat becomes too spread out, resulting in uninhabited inaccessible areas of suitable habitat or local extinction of a particular species if it is sedentary or the habitat patch is out of its dispersal range (Wilcove et al. 1986, van Dorp and Opdam 1987). A reduction of territory size through fragmentation could also affect some species, with the resultant patches of habitat being too small for species with larger ranges or territories (Wilcove et al. 1986). These patches subsequently become islands of suitable habitat in a sea of agricultural land or urban dwellings, following the theory of Island Biogeography (MacArthur and Wilson 1967, Wilcove et al. 1986, Knaapen et al. 1992). Whilst the geographical distance of islands in the sea influences the isolation of species, the terrestrial habitats are influenced by the characteristics of the intervening landscape (Knaapen et al. 1992). These uninhabitable areas can also affect the patch populations by increasing the abundance of harmful species and predators (Wilcove et al. 1986).

The loss of forests has led to pockets of often ancient woodland being spread out over the landscape. Larger and therefore more heterogeneous forest areas contained more bird species and can support a higher abundance of bird species, namely interior and resident birds (MacArthur and Wilson 1967, Freemark and Merriam 1986). Conversely, Howell et al. (2000) found higher species richness and diversity in smaller forest fragments, but higher percentage of migrants in the main unfragmented woodland, along with significant differences in the mean number of birds detected in each. However, they also found that not all increased diversity was good, as the edge and disturbed habitats in the fragmented forests were more likely to contain a higher diversity of often generalist and edge specific bird species. Large, continuous forests may support a lower diversity compared with the disturbed, fragmented forests, but contain interior species and, in this case, neo-tropical migrants, indicating that the species of birds present must also be evaluated (Howell et al. 2000). Cushman and McGarigal (2003) also found lower species richness in mature forest dominated landscapes, but found that bird species richness and density responded more to mature forest area than to fragmentation. Bellamy et al. (1996) studied small woods in south east England and separated the woodland and edge species for the analysis. They found that woodland species were related to woodland area while the edge species had a poor relationship with area and a positive relationship with perimeter edge. Therefore, the small woods held more edge specific birds than woodland birds in the equivalent area of large woods (Bellamy et al. 1996). Melin et al. (2018) found that even in small woods, bird diversity increased towards the woodland edge and demonstrated that this was related to vegetation structure. Wilcove et al. (1986) also stated that forest edge habitat increases species richness. However, the introduction of edge influenced the habitat by increasing shade intolerant plants and shrubs, and therefore changed the floristics of the habitat and created a noticeable increase in predator numbers, which caused many bird species to nest further into the interior of the wood (Wilcove et al. 1986). Moreover, the disturbed areas were more likely to have reduced reproductive output (Robinson et al. 1995).

In pasture dominated landscapes, Graham and Blake (2001) recommended that large blocks of forest be maintained to conserve certain species, with smaller forests able to support generalist species that are more resistant to disturbance. Moreover, Estrada et al. (1993) showed that smaller patches of forest had higher species richness than larger forests, and agricultural areas contained 58% of the species detected in the study, and 34% of all the individual birds censused. The man-made, agricultural ‘islands’ of vegetation provided biotic connectivity between isolated forest fragments and foraging areas. Estrada et al. (1993) concluded that rather than clearing forests for pastoral land, if agricultural land, such as the type in their study, was used it would enhance the persistence of biodiversity and regenerate the ecosystem with direct benefits to the human population. Howell et al. (2000) also found that the landscape variables influenced species more than the local variables, demonstrating the importance of the landscape to bird species, and therefore indicating that landscape variables should be considered when making management decisions.

Focusing on species richness alone may mask individual responses to habitat fragmentation as Miller and Cale (2000) found that the remnant area in which the fragments were embedded explained community structure, but other variables explained foraging guilds and individual species. The foraging guilds were influenced by whether the fragment was fenced off from grazing, embedded in the remnant forest or degraded because of grazing (Miller and Cale 2000). Lopes et al. (2016) found that the connectivity of the landscape was more important for bird communities than individual species. The intrinsic characteristics of the species affected its sensitivity to fragmentation more than landscape connectivity (Lopes et al. 2016). This indicates the importance of understanding species ecology in order to protect and restore habitats correctly. Hinsley et al. (2009b) stated that even generalist bird species suffered in terms of reproductive output and parental energy costs in a landscape with small, fragmented patches of suitable habitat. Specialist bird species experienced the greatest threat from modified, scattered land patches, which provided insufficient resources and increased competition from generalist bird species.

A number of studies have investigated the effect on birds of island biogeography through fragmentation and isolation of habitats in terrestrial landscapes, proving that agricultural land and urban expanse have an effect (e.g. Wilcove et al. 1986, Knaapen et al. 1992, Hinsley et al. 1996). These studies indicate the complexity of ecosystems and the factors that affect bird species. As a result, managing landscapes to enhance biodiversity is difficult and each landscape is different, therefore appropriate investigations and research must be carried out. However, species richness or diversity may not be the best variable to explain the state of birds, therefore bird community and individual species analysis must also be considered for successful habitat restoration or preservation management strategies.

2.2.1.5 Grazing

Degradation and habitat loss are major factors in the declines of bird populations globally, and human use of the landscape conflicts with the means of restoring and conserving avian habitat (Nelson et al. 2011). Nelson et al. (2011) showed that grazing caused degradation of riparian areas, and that compromise was needed to allow the grazing of cattle and the conservation of bird species. Both total exclusion of cattle and seasonal grazing of cattle on the land influenced bird species abundance, whilst species richness remained the same in both prescriptions. Although total exclusion of grazing resulted in more abundant bird populations, seasonal exclusion was a viable and acceptable method of compromise to allow the recovery of the bird species (Nelson et al. 2011). However, avian management should be addressed by complete exclusion of grazing, and seasonal grazing should be attempted with the associated conservation measures, where total exclusion is not an option (Nelson et al. 2011).

Links between habitat degradation by grazing and decreased insect prey habitat were noted by Miller and Cale (2000). As a result of habitat fragmentation and then further degradation by grazing, Miller and Cale (2000) suggested that fences around isolated fragments of natural forest habitat were needed to maintain habitat quality. Donald et al. (1998) found that grazing significantly altered vegetation structure and bird communities, with ungrazed stands containing more small, broadleaved trees and a higher proportion of migrants than the grazed stands. However, they found no significant differences in species richness, the proportion of hole-nesters or overall abundance between grazed and ungrazed stands. Other studies discussed by Donald et al. (1998) showed that conservation efforts have often been based on three migrants that prefer mature, grazed stands, namely the Redstart (*Phoenicurus phoenicurus),* Pied Flycatcher (*Ficedula hypoleuca*) and Wood Warbler. However, at Donald et al.’s site, these species were present in low numbers compared to other migrant species that preferred ungrazed, younger growth and scrub habitats, such as the Willow Warbler, Blackcap (*Sylvia atricapilla)* and the Garden Warbler (*Sylvia borin*). Martin and Possingham (2005) demonstrated that most bird species in their study decreased with increased grazing pressure, and that the birds’ foraging height preferences in the vegetation predicted the species’ susceptibility to grazing. However, grazing can create pasture woodland, which is a rare habitat in Europe, and benefits certain bird species, such as the Green Woodpecker (*Picus viridis*) (Dorresteijn et al. 2013, Hartel et al. 2014, Fuller and Robles 2018).

Grazing contributes to both habitat deterioration for certain avian species and to habitat suitability for others. Excessive grazing negatively affects some avian species, therefore lowering bird species richness in that area. For conservation purposes, a detailed analysis of the bird species and bird community present in the habitat should be carried out before any management is put in place. A heterogeneous landscape containing multiple habitat types with varying degrees of grazing could ultimately provide the greatest number of habitats and the most diverse communities, but not without prior analysis of the ecology of the bird species present in the landscape.

2.2.2 Mediating the Effects of Agriculture

To mediate the effects of agricultural intensification and keep up with the increasing demand for human consumption, conservation measures must be implemented at a regional scale. Practises to enhance the landscape for biodiversity must be monitored in order to evaluate the effectiveness of the implemented scheme. The following sections will detail studies that have investigated methods to mediate biodiversity decline attributed to agriculture.

2.2.2.1 Management Practises

Landscapes vary, so management strategies must be suitable for individual landscapes and the species to be conserved. Robinson and Sutherland (2002) reviewed post-War changes in agriculture and stated that biodiversity may benefit from integrated farming techniques, but these should be incorporated directly as an environmental objective and not as an addendum to current practises. Melles et al. (2003) found that local-scale habitat features, such as large conifer trees, berry-producing shrubs and freshwater streams, were important features affecting bird presence. Forest cover, within 500 m, and park area also increased the likelihood of bird species occurrence (Melles et al. 2003).

Increasing habitat heterogeneity in the landscape has been shown to enhance biodiversity (Benton et al. 2003). However, most of the management practises to enhance biodiversity have been carried out at the field level (e.g. AES), but to maximise heterogeneity, a simple, low cost option (for example, field margins or in-field strips), should be employed in multiple fields over the wider landscape (Benton et al. 2003). Furthermore, Freemark and Merriam (1986) suggested that regional conservation strategies should maximise both size and habitat heterogeneity of forests. As this is not always possible, management must include investigations into the habitat extent that can sustain the highest diversity whilst allowing for land diversification for agriculture (Freemark and Merriam 1986). Multiple habitats in the landscape will sustain the highest diversity; therefore a blanket forest landscape would ultimately reduce the diversity of other species, for example, wetland, heathland and scrubland species (Freemark and Merriam 1986).

Due to the difficulties in acquiring all relevant ecological data on every bird species, Hansen et al. (1993) suggested rather than ignoring incomplete ecological data, conservationists and managers are encouraged to use the best available data at the time to make decisions. This therefore, highlights the areas with less information available and indicates where more data are needed (Hansen et al. 1993). Hannah et al. (2017) utilised priority bird species as indicators of suitable habitat management and restoration, due to the relationship between priority bird species and bird species richness. If suitable avian indicator species are present and identified in the habitat, it would allow management to be prioritised to enhance bird species diversity without the need for advanced ecological knowledge for every species occurring in the habitat. Opdam et al. (2003) proposed indices in order to explain and portray the habitat elements of species. Since each species has a different requirement of the landscape and the landscape needs to be maintained for multiple species, these can be combined and used as a tool in GIS for landscape assessments (Opdam et al. 2003).

Foley et al. (2011) broadly stated that solutions to food production and agricultural degradation of the environment could be solved by following a number of management strategies. For example, halting agricultural expansion, closing “yield gaps” on land not producing enough, increasing cropping efficiency, shifting diets and reducing waste would mediate the effects of agriculture. Foley et al. (2011) also indicated that preventing loss of tropical rainforests will reduce global warming, biodiversity losses and continue to provide ecosystem services. In order to completely mediate and reverse biodiversity decline, management practises need to be implemented globally. However, this is difficult and has many social and ethical implications especially in poorer, malnourished countries.

To mediate the effects of agriculture and keep up with the increasing demand for human consumption, farming must integrate conservation management into normal practises. This is problematic as landowners, farmers and managers must come to an agreement and implement strategies that will benefit wildlife. Conservation strategies need be implemented landscape-wide in order to maximise biodiversity, but also considered on a regional case-by-case basis (Whittingham et al. 2007). Increasing habitat heterogeneity by providing natural habitats over the landscape will increase biodiversity, although the management schemes implemented must be monitored, in order to evaluate and understand their effectiveness on biodiversity trends (Kleijn et al. 2011).   However, biodiversity may not necessarily be the most reliable method of determining the effectiveness of conservation strategies (Miller and Cale 2000, Radford et al. 2005).

2.2.2.2 Agri-Environment Schemes (AES)

To reverse the effects of agricultural intensification, the UK government and the EU introduced quotas and subsidies for farmers to adopt and implement Agri-Environment Schemes (AES) and Agri-Management Schemes (AMS) to enhance the environment and improve biodiversity (Kleijn and Sutherland 2003, Kleijn et al. 2011). This was suggested to be the main solution in the 21st century to mitigate declines in biodiversity, as AES directly supports necessary agricultural management (Batáry et al. 2015), and each EU state was obliged to implement an agri-environment initiative (Batáry et al. 2011). The schemes varied between countries, but the main objectives were to reduce nutrient and pesticide usage, protect biodiversity, restore landscapes and prevent depopulation (Kleijn and Sutherland 2003). This involved reinstating ponds, re-planting and maintaining hedgerows, ‘set-aside’ which was land left alone to grow naturally, and beetle banks, grassy margins or strips of unsprayed wild flowers, providing a refuge for pollinators and wildlife (Kleijn et al. 2011).

In England, there are two-levels to the AES: Entry Level Schemes (ELS) which are broader management strategies, and Higher Level Schemes (HLS) which are more targeted schemes (Chamberlain 2018). AEM is more concerned with altering farmland management practises, such as conversion to organic farming. Studies, such as Walker et al. (2018) and Davey et al. (2010), presented evidence that HLS could increase bird abundance, but that overall, farmland birds were still declining (Hayhow et al. 2015, 2017). Moreover, the majority of the targeted bird species were not responding, suggesting poor implementation (Chamberlain 2018, Walker et al. 2018). Batáry et al. (2015) combined meta-analyses to review the effectiveness of AES in Europe and they found that the areas around or beside the cropped area were more effective at enhancing species richness than trying to enhance the cropped area itself. They also stated that the first studies looking into AES, such as Kleijn and Sutherland (2003), showed that half of the schemes lacked a positive effect on biodiversity. Successful AES concentrated on rare species with input and supervision from scientists, whilst more general AES only increased common species or had no effect at all (Kleijn and Sutherland 2003, Batáry et al. 2015). Batáry et al. (2015) continued that since the earlier studies, there had been more positive outcomes from AES with modest increases in species richness locally. They concluded that AES is effective at conserving farmland biodiversity, but expensive and must be carefully designed and managed to enhance biodiversity over the selected landscape and species, without detriment to species that depend on the farmland and farming techniques (Batáry et al. 2015). Nonetheless, both AES and AEM were deemed less effective in more complex landscapes (Tscharntke et al. 2005, Batáry et al. 2011), possibly due to less clear cut contrasts between intervention and non-intervention (Kleijn et al. 2011). Complex, heterogeneous landscapes already support a higher diversity of organisms, therefore these landscapes should not be neglected but preserved and maintained (Batáry et al. 2011). Conversely, Whittingham (2011) contested that in heterogeneous landscapes with greater biodiversity AES will improve ecosystem services (e.g. pollination) more effectively than in homogeneous landscapes, and Kleijn and Sutherland (2003) showed that AEM had more pronounced effects in extensive rather than intensive farmland. Furthermore, Duelli and Obrist (2003) concluded that populations are more likely to survive if source populations are supported in nearby natural and semi-natural habitat. The meta-analysis by Batáry et al. (2011) revealed that the efficiency of AEM (or AES) is dependent on the landscape context as well as the habitat type, and should therefore be prescribed for the particular landscape and target species.

Poor habitat quantity and quality is to blame for the continued population decline of specialist farmland birds in the UK (Vickery et al. 2004). The required ‘beneficial’ or ‘sympathetically managed’ land needed to stop the decline is predicted to be considerably large and specific knowledge of relevant bird ecology and careful management is required (Vickery et al. 2004). Redhead et al. (2013) established that Blue Tits and Great Tits can act as indicators of the efficiency of AES to manage local habitat quality and spatial arrangement at the field scale. However, they cannot indicate overall success or failure of the scheme, but rather assess specific management on local habitat quality and resource provision. Redhead et al. (2013) also found that AES may have little effect in the summer, but can provide a buffer for certain species against extreme weather in the winter. More recently, Walker et al. (2018) showed that High Level Schemes of AES in England increased bird abundance, however, they showed that this was often temporary and the schemes were not resilient to poor weather conditions. Furthermore, they also indicated that the extent of the schemes currently implemented were not large enough to reverse the declines in farmland birds, and the schemes were dependent on the farming practises as well as the weather conditions (Chamberlain 2018, Walker et al. 2018). Conversely, Whittingham (2011) stated that AES has only delivered moderate biodiversity gains, and that heterogeneous landscapes which support higher levels of biodiversity were more likely to increase the benefits of AES than homogenous landscapes. Whittingham (2011) agreed that providing clear guidance to land managers is essential in implementing AES, which should be modified for the specific area through adaptive management and applied to small fragments of land to enhance farmland bird populations.

The focus of current AES from the EU is not explicitly on increasing landscape heterogeneity of farmland (Benton et al., 2003). AES along with land management practises (AEM), such as organic and integrated farming, is thought to be the answer to meet the governmental targets to enhance biodiversity (Vickery et al. 2004). AES have been shown to increase diversity of farmland birds, but often only at a local, habitat scale. In order to increase overall (gamma) avian diversity the landscape context needs to be taken into account, and management needs to be at a landscape scale in order to conserve the bird populations. More work needs to be carried out to assess the bird communities over multiple habitat types, at a landscape scale and over multiple landscape types, in order to provide and implement the most effective management practises to conserve avian diversity.

2.2.2.3 Reserves and Protected Areas

Preserving natural habitats for wildlife is not a new concept. Historically, estate owners and royalty designated areas of land, such as the New Forest, for hunting game, and for recreation or aesthetic purposes (Margules and Pressey 2000). A number of areas have been protected by law to conserve specific habitats, often because they are rare, ancient or sustain rare species not found anywhere else. There are various designations that can be given to an area, depending on the type of area, the reason for its preservation and, for instance, whether people are allowed to also inhabit and alter the area. Batáry et al. (2015) noted that it is unusual that protected areas in Europe are inhabited by people, cultivated, or have grazing livestock, or that woodlands in nature reserves are managed, such as through coppicing, to provide open areas for conservation purposes. This is resultant of the long history of thousands of years of intensive human management across Europe (Batáry et al. 2015).

Areas designated for birds can be either quite specific or very general, and will also provide protection for other species. Sparse habitats, such as wetland areas and marshes, are often protected as a specialised habitat for certain bird species. For instance, the Tay reed beds are the largest reed beds in Britain and are protected and managed for the Bearded Tit (*Panurus biarmicus*) population. Poole Harbour has been designated as a Site of Specific Scientific Interest (SSSI) due to the natural harbour providing wetland habitats for over-wintering waders and wildfowl. Forests and woodlands are often protected as they are the last remaining remnants of ancient forest and support a diverse range of woodland bird species. Patches of trees regularly need to be protected otherwise the ecosystem services they provide often go unrecognised and the woods could be removed for agriculture.

Wilcove et al. (1986) detailed guidelines for allocating protection to areas and stated that all remaining fragments of a habitat should be protected, with priority given to the largest fragments which provide habitat required by certain species, such as raptors, and to provide a source population for smaller fragmented woodlands. Fragmented habitats within close proximity to one another should also be protected in order to allow movement between them (Wilcove et al. 1986). Regional protection should encompass multiple habitat types in order to attain variation and preserve diversity over the landscape. Wilcove et al. (1986) also suggested that a reserve be circular in shape in order to protect interior species from any changes at the boundaries i.e. ‘edge effects’. Effective and active management is also required to adequately protect species in the reserve from further fragmentation and human activity, whilst maintaining successional and open habitats. This includes the eradication of non-native vegetation species and the culling and controlling of animals, either deemed a nuisance or lacking a natural predator to control the population (Wilcove et al. 1986). Miller and Cale (2000) agreed that since larger reserves were more likely to hold more species, they should be prioritised as they will enhance biodiversity of native species in fragmented landscapes and will be beneficial if little is known about the ecology of the present species. Diversity and quality, including the variation of features influencing the species present, must be examined if protection is given to smaller reserves, as well as their proximity to other reserves of the same remnant (Miller and Cale 2000).

The surrounding landscape and landscape features must be taken into account when allocating, designing and planning reserves, as well as the biotic composition of the proposed area (Howell et al. 2000). Howell et al. (2000) also stated that in order to manage for multiple species, trade-offs are inevitable due to the varying sensitivity of species to vegetation and landscape characteristics. Margules and Pressey (2000) discussed a strategy for designing and locating reserves, and also agreed that whole landscapes must be managed for conservation purposes, as reserves alone are only the starting point for nature conservation. Reserves should represent the biodiversity in each landscape, and not be skewed towards specific species or habitats, but rather encapsulate them all and these areas should be separated from processes that threaten the diversity. Most reserves are placed in remote, inaccessible areas or areas unsuitable for agriculture, which places a bias on the species being conserved. In order to successfully conserve biodiversity a strategic conservation plan must be put in place to capture the highest diversity in the landscape, whilst maintaining areas for agriculture and commercial use (Margules and Pressey 2000). Nagendra et al. (2012) demonstrated that remote sensing data, in the form of LiDAR, can be used to evaluate protected areas and assist in their management by informing managers of changes in the landscape that would impact on the biodiversity, allowing long-term restoration and protection from adverse effects of climate change. Remote sensing can also provide managers with spatial and temporal data on the extent and condition of protected habitats and the effect of changes over different time scales (Nagendra et al. 2012).

In order for the preservation and conservation of avian species to be a success, it falls to the landowners of many of these remnant and rare habitats to work with conservationists and managers to collate the information required and implement relevant strategies, often through some incentive (Miller and Cale 2000). If the destruction of these habitats continues, then many of the species that rely on them will become extinct either locally or worse, globally, and reserves are a viable approach of maintaining habitats for biodiversity. However, management efforts must be maintained in the reserves once they are allocated. The interaction and influence of the surrounding landscape and habitats is imperative to the success of the reserves and for any biodiversity enhancing scheme, therefore, thorough investigation and planning is essential.

2.3 Summary and Conclusion

This review of the literature demonstrates that landscape level studies over multiple habitats and of multiple bird species are lacking, as studies often focus on one or a select number of species in a single or few habitats. Remote sensing is an effective and useful tool for conservationists, but uptake has been slow. It is clear that heterogeneous landscapes create greater bird diversity, but this diversity can mask more subtle relationships and those of rarer bird species. Anthropogenic activity has altered landscapes, often to the detriment of bird species, particularly of farmland birds. The extent of the change and the factors contributing are analysed more often on a species-by-species or biodiversity basis, rather than the bird community. The bird community can supply an overview of the avian ecology of a given habitat or area and provide insight into its likely history, health and resilience to change. Measures and strategies have been implemented to prevent further declines, but their effectiveness is often not monitored, therefore preventing informative implementation of schemes elsewhere. Reserves should also be used in conjunction with management strategies as population sources for smaller patches of habitat, providing they contain suitable habitat to sustain viable bird populations.

3 Sites, Materials and Methods

3.1 Field Sites

3.1.1 The New Forest

The New Forest National Park was designated in 2005 and covers an area of 571 km^2 in the counties of Hampshire and Wiltshire (Chatters 2006), situated on the south coast of England between the conurbations of Southampton to the east and Bournemouth to the west (Figure 3.2a). The survey area for this project is approximately 53 km^2 and is located in the triangle formed between the towns of Lyndhurst, Brockenhurst and Beaulieu, between grid references SU 3406 to SU 3201 (Figure 3.2). The survey area is predominantly enclosed broadleaved forest interspersed with patches of conifer plantations, unenclosed forest and open heathland to the north-east and south of the woodland extent (Figure 3.1 and Figure 3.2).

The New Forest is maintained and managed mostly by the New Forest National Park Authority and the Forestry Commission, working with nearby authorities, regional decision-makers and people who manage and live in the National Park (Newton 2010). It is a large visitor attraction due to the fact that it is the largest area of wild, “unsown”, unenclosed forest in Britain, termed the Open Forest, and contains three now rare habitats of ancient pasture woodland, heathland and valley mire, which are not found in as large a scale or in as particular a mosaic of habitats anywhere else (Tubbs 2001). The New Forest is of great conservation value with over 50% (29,000 ha) of the National Park designated for its international importance for nature, which includes 20 SSSIs, six Natura 2000 sites and two Ramsar Convention sites (Cantarello et al. 2010, Newton 2010). The New Forest also has Special Protection Areas (SPA) for five bird species, including the UK’s largest breeding population of Dartford Warbler (*Sylvia undata*), Nightjar (*Caprimulgus europaeus*) and Honey Buzzard (*Pernis apivorus*) (JNCC 2001). Special Areas of Conservation (SAC) have also been designated in the New Forest to protect important habitats, such as the wet and dry heath, *Molinia* meadows, beech and oak woodland, bog woodland and oligotrophic waters (JNCC 2019). The SAC is also to protect the southern damselfly (*Coenagrion mercurial),* stag beetle (*Lucanus cervus)* and the great crested newt (*Triturus cristatus*) (JNCC 2019). The New Forest also supports

73% of the British breeding dragonflies, all British species of reptile, numerous rare plant species and 13 of the 18 British bat species (New Forest National Park Authority 2013).

Historically the forest perambulation, or boundary where the forest laws existed, covered 37,907 ha, and today this is where the New Forest Verderers graze their livestock (Tubbs 2001, Newton 2010). The unenclosed forest area covers 20,000 ha and in 2001, it was noted that nearly 3,700 ha of the unenclosed forest was mainly oak, beech and holly (*Ilex aquifolium)* woodland, much of it having been woodland for the last 5,000 years or more (Tubbs 2001, Newton 2010). The unenclosed forest also comprised 12,500 ha of heathland and acid grassland, 2,900 ha of valley, seepage step mire and wet heath, and 837 ha of plantations, 40% of which are broadleaved (Tubbs 2001, Newton 2010). The enclosed areas are privately owned farmland or Silvicultural Inclosures in the crown land, enclosed by fences and cattle grids to prevent grazing (Tubbs 2001, Newton 2010). There is around 8646 ha of Silvicultural Inclosures of mainly conifer plantations (4744 ha), hardwood (2744 ha) and mixed plantations (Tubbs 2001). As such, the New Forest National Park Management Plan 2010-2015 (updated in the Partnership Plan (2015-2020), Natural England et al. 2015) was set out to preserve the New Forest's outstanding natural beauty, nationally and internationally important habitats and the resulting biodiversity (New Forest National Park Authority 2010, 2013). This also includes plans to maintain the unique historic, cultural and archaeological heritage of the land and the communities, along with the historic commoning system including grazing of New Forest ponies (*Equus caballus*), and a healthy recreational environment for locals and tourists to enjoy (New Forest National Park Authority 2010).

The estates, interspersed between a number of small villages and towns throughout the New Forest, have common rights to allow the grazing of certain species on the land. In recent years up to 6000-7400 ponies, donkeys (*Equus asinus*), cattle, pigs and sheep have been depastured in the "Open Forest" (Newton 2010). There are also at least 2,500 wild deer with access to the entire forest (Putman 2010). The most widespread and common is the fallow deer (*Dama dama*); roe deer (*Capreolus capreolus*) are not as common, but are patchily distributed over the New Forest; Sika (*Cervus nippon*) and red deer (*Cervus elaphus*) are relatively recent to the New Forest and have limited distribution, and lastly, Reeves muntjac (*Muntiacus reevesi*) are regularly reported, but at low numbers (Putman 2010). European pine marten

(*Martes martes*) have also been sighted and are thought to have been released into the National Park (*pers. comm.* Wild New Forest, February 2016).

The action of grazing has caused changes in the botanical composition and morphology of vegetation species (Tubbs 2001, Putman 2010). Intensive grazing coupled with the low productivity of the poor, acid forest soils also influences the productivity and behaviour of the animals (Tubbs 2001). The heavy grazing has resulted in large *Molinia* meadows occurring in the New Forest, which contain a heathy form of *Molinia caerulea* and *Cirsium dissectum* fen-meadow thistle (JNCC 2019). The meadows in the New Forest are unusual as they are species rich with an abundance of small sedges, including carnation sedge (*Carex panicea*), common sedge (*C. nigra*) and yellow-sedge (*C. viridula* ssp. *oedocarpa),* and the more frequent occurrence of mat-grass (*Nardus stricta*) and petty whin (*Genista anglica*), compared with elsewhere in the UK, therefore resulting in SAC protection of the grasslands, as well as the heathlands (JNCC 2019).

Tubbs (2001) provides great detail of the geology of the New Forest, and stated that the New Forest is situated in a chalk syncline in the Hampshire basin which consists of soft, sedimentary clays and sands of the Tertiary age. The land has been subjected to earth movements and erosion from the sea, laying down and removing superficial gravel or brickearth, and partially infilling valleys and hollows with the transported material creating a veneer over the solid strata (Tubbs 2001). Therefore the chemical composition and texture of the soil varies, characterising the variation in vegetation in the New Forest (Tubbs 2001). The highest point in the New Forest is 128 m at SU 248162, Black Bush Plain, and the land then descends south and east to the Solent and south and west to the River Avon (Tubbs 2001). The survey area in the New Forest only reaches 43 m above sea level at the highest point and the lowest point is 14 m, descending towards the Solent.

The climate in the New Forest is relatively dry and temperate with a recorded average annual precipitation of 866 mm between 1958 and 1981, with almost all falling as rain (Tubbs 2001). The New Forest has hot, humid summers and relatively cold, frosty winters not exceeding -9°C. When snowfall has occurred it rarely lasts very long (Tubbs 2001).

Figure 3.1. Photographs, clockwise, of a typical broadleaved woodland near Denny Lodge looking south, open heathland in the southern edge of the survey area looking west, a typical conifer plantation at the northern edge of the survey area near Matley Passage looking south, and in Denny Inclosure looking south.

a)

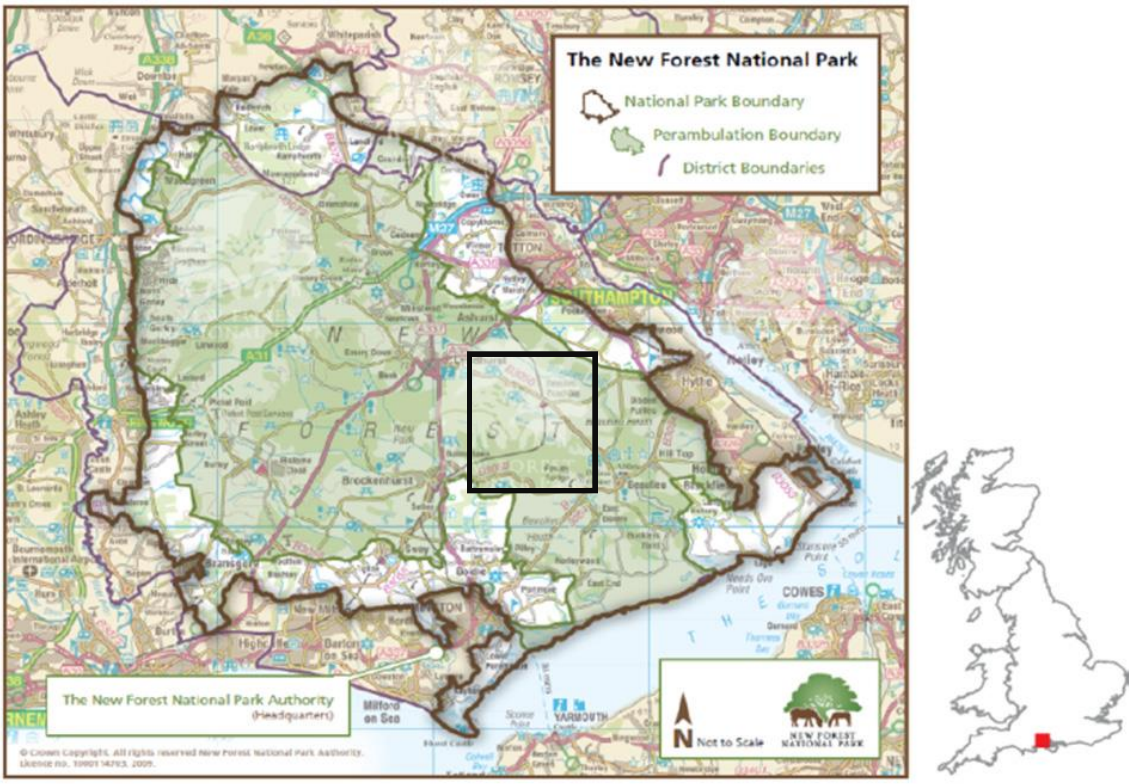


Figure 3.2. a) The extent of the New Forest National Park in the south of England and the location of the survey site within it indicated by the black box (New Forest National Park Authority n.d., Pogue et al. 2015).

b)

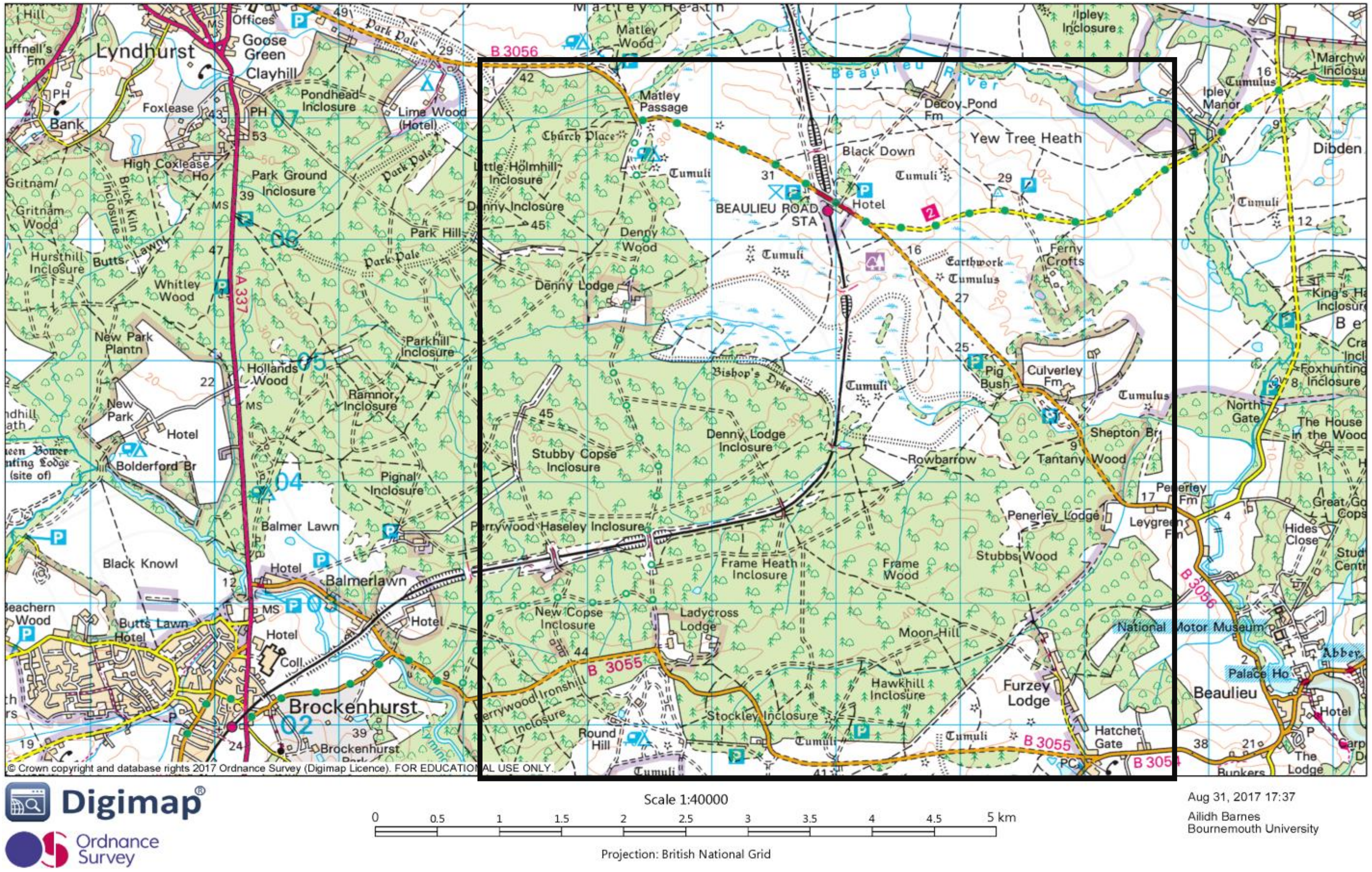


Figure 3.2. b) Ordnance Survey Map (OS) of the survey area in the New Forest within the black box. © Crown copyright and database rights 2017 Ordnance Survey (100025252) (Edina Digimap, 2017/2018).

3.1.2 Cambridgeshire

The survey site in Cambridgeshire is situated in east England, between Huntingdon to the south and Peterborough to the north (Figure 3.4). There are a number of towns and many villages spread throughout the survey area with Sawtry, Alconbury, Abbots Ripton and Upwood bordering the site. The survey extent is approximately 58 km^2 and is located between grid reference TL 5328 and TL 5127 (Figure 3.4). The highest point in the survey area is 51 m above sea level south of Aversley Wood and the lowest point is ~0 to 5 m in the north of the survey extent. The landscape is relatively flat with gentle slopes characteristic of the Fens; however, there is a rapid incline to the OS triangulation point at 51 m. The A1 and the main railway line from London to Edinburgh interject through the field site, with Monks Wood National Nature Reserve in the centre (Figure 3.4).

In contrast to the New Forest, the majority of this site is agricultural land interspersed with various sizes of isolated woodlands, many of which are Nature Reserves. The woodlands also differ from the New Forest as they are predominantly broadleaved woodlands of oak, ash (*Fraxinus excelsior*) and field maple (*Acer campestre*). Some of the woodlands are maintained by various trusts (e.g. the Wildlife Trust, Natural England and the Woodland Trust) and are open to the public; however, other woods in the area, such as Wennington Wood, are still privately owned and managed.

The fragmented woodlands in the survey area are remnants of a 27 km woodland belt that once covered most of the landscape. Monks Wood is the main and largest ancient woodland remnant in the area measuring 157 ha, managed by Natural England and has been a National Nature Reserve since 1953 (Hill and Hinsley 2015). It was designated a Site of Special Scientific Interest (SSSI) in 1986 having been identified as one of Britain’s most important lowland woods (Natural England 2019a). The geology in Monks Wood is Jurassic Oxford clay, with chalky boulder clay on high ground (Steele and Welch 1973). Aversley Wood, measuring 61.6 ha, is located outside the village of Sawtry (Figure 3.4), and has been managed by the Woodland Trust since 1979 and was also designated a SSSI in 1983 as it is another ancient woodland remnant, parts of which date back to the last Ice Age (Woodland Trust 2019a). Archers Wood, near to Aversley, is also managed by the Woodland Trust and is only 18.6 ha in size. The wood used to extend to the A1, but was cut back to within a bow-shot as highwaymen used to ambush travellers on the old Roman road (Woodland Trust 2019b). Lady's Wood, Raveley Wood, and Gamsey Wood, near Ramsey, are managed by the Bedfordshire, Cambridgeshire and Northamptonshire Wildlife Trust and are also remnants of the ancient forest that covered this area in the Saxon times (The Wildlife Trusts 2019). Lady's Wood is only 7.13 ha and was traditionally managed for coppice until many of the trees were cut down in 1951 and is now a county wildlife site (Figure 3.4 and Figure 3.3). Not far from Lady's Wood, Raveley Wood measures only 5.6 ha, and is notable for containing elm (*Ulmus minor* 'Atinia') trees, as well as ash, oak and field maple, after many were killed by Dutch Elm Disease in the 1970s. Gamsey Wood is the smallest surveyed woodland of only 4.7 ha and is located where the clay uplands adjoin the Fenland Basin (The Wildlife Trusts 2019). The remaining woodlands in the survey area are privately owned and managed: Wennington Wood, Little Less Wood, Upton Wood, Bevil's Wood, Coppingford Wood, Hill Wood, Hermitage Wood and Holland Wood. Therefore, public access is restricted to these woods, although some of them have public footpaths running alongside them, as shown in Figure 3.4. Upton Wood was accessible and was surveyed, and measured 29.1 ha, an edge of Little Less Wood and Wennington Wood, near Abbots Ripton, were also surveyed along the public right of way.

Agricultural land covers the majority of the survey area (Figure 3.3 and Figure 3.4) and is mainly arable crops of oil seed rape (*Brassica napus*) and wheat (*Triticum aestivum*, winter and spring), some field beans (*Vicia faba*), barley (*Hordeum vulgare*), improved grass/silage and peas (*Pisum sativum*). There was very little pasture and only some grazing livestock of cattle and horses at the time of the bird survey (April-June 2017). The farmland is privately owned, interspersed with a network of public roads, tracks and public footpaths. The rights of way were sometimes bordered by hedgerows of various qualities linking the isolated woodlands across the landscape. Muntjac deer were sighted in the woodlands and along field boundaries. Other mammals, such as brown hare (*Lepus europaeus*), red foxes (*Vulpes vulpes*), brown rats (*Rattus norvegicus*) and mustelids were also observed during the study. Game birds, such as Pheasants (*Phasianus colchicus*) and Red-Legged Partridge (*Alectoris rufa*), are released into the Cambridgeshire landscape for sport every year, with holding pens and feeding stations, often near farms, being notable signs in the landscape.

Steele and Welch (1973) describe the climate in Monks Wood as continental due to the remoteness from the sea creating variation in the weather with a greater proportion of annual rainfall in the summer. Between 1964 and 2003 an average daily maximum temperature of 21.8°C, an average daily minimum of 0.7°C and an average annual rainfall of 570.2 mm was recorded in Monks Wood (Heywood 2003).   The maximum highest temperature between 1964 and 1972 was 31.1°C, the lowest minimum temperature was -15°C, and the greatest rainfall in a day was 92.7 mm (Steele and Welch 1973). Compared to the average annual rainfall for Great Britain, Monks Wood is situated in an area of one of the lowest rainfalls in the country (Heywood 2003).

Figure 3.3. A selection of surveyed woods, on the left and field sites on the right. Top left and Middle left: Lady's Wood, Bottom left: Upton Wood. The fields are all south of Raveley Wood and north of Wennington Wood looking south.

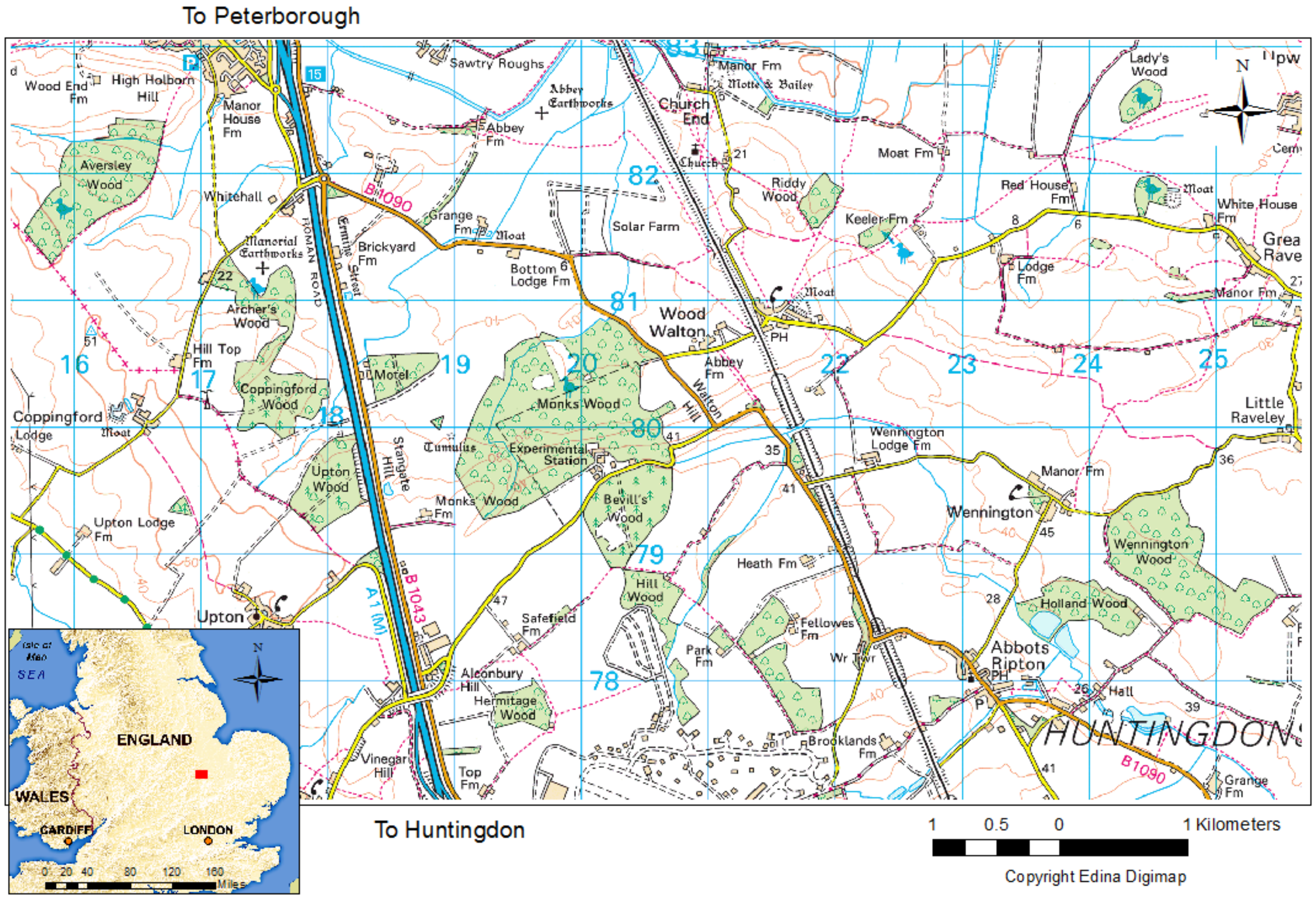


Figure 3.4. The location of the survey area in Cambridgeshire with an insert of the location in the UK on an Ordnance Survey map. The red dashed lines represent public rights of way. © Crown copyright and database rights 2018 Ordnance Survey (100025252) (Edina Digimap, 2017).

3.2 Materials

3.2.1 Tree Species Map

A tree species map of the New Forest survey area was provided from a previous PhD thesis at Bournemouth University (Sumnall 2013). The tree species map was developed using remotely sensed data. An in-depth classification of tree species types within coniferous and deciduous categories was carried out, using species expected to occur in the study area according to field survey and Forestry Commission inventory data (Sumnall 2013). The deciduous sub-classes included species, such as oak, beech, silver birch (*Betula pendula*), sweet chestnut (*Castanea sativa*) and holly, and the coniferous sub-classes contained species, such as Scots pine (*Pinus sylvestris*), Corsican pine (*Pinus nigra*), Douglas fir or Norway spruce (Sumnall, 2013). This tree species map was used to stratify and select field plots in the New Forest. However, after commencing fieldwork, inaccuracies in the tree species map were apparent, and therefore, the tree species map was not used in further analyses, and the habitat classifications were re-evaluated. This is explained further in Section 3.3.1.1.

3.2.2 LiDAR Data

The vegetation structure over the survey area was analysed using Light Detecting and Ranging (LiDAR) data acquisitioned before the start of the project. Small-footprint LiDAR data were flown by the Natural Environment Research Council's (NERC) Airborne Research and Survey Facility (ARSF) using a Leica ALS50-II airborne laser scanner with an upgrade to allow simultaneous recording of discrete return and full waveform data.

The New Forest was flown on 6th July 2010 collecting leaf-on data. This was acquired at an altitude of approximately 1600 m, a pulse repetition frequency of 147 kHz, a beam divergence of 0.22 mr and a scan angle of 10 degrees (Sumnall et al. 2016). The geometric accuracy for the scanner is 0.05 m to 0.10 m vertical accuracy and 0.13 m to 0.61 m horizontal accuracy as stated by Leica Geosystems (Sumnall et al. 2016). The data were processed as per Sumnall et al. (2016). The ground hits were filtered from the vegetation hits to determine ground elevation and this was then removed to terrain normalise the data and determine the vegetation height (Sumnall et al. 2016). The ground elevation was rasterized to form the digital terrain model (DTM) and the vegetation height raster gave the canopy height model (CHM). The LiDAR metrics for the current study were calculated from the normalised point cloud.

The Cambridgeshire site was flown on the 15th September 2012, representing leaf-on data as leaf drop does not take place until October-November at this site (Hill and Hinsley 2015). The data were also acquired at an altitude of around 1600 m, a pulse repetition frequency of 144 kHz, a scan angle of 10 degrees, with four returns per laser pulse and a footprint of around 35 cm (Hill and Hinsley 2015). The LiDAR data were processed according to Hill and Hinsley (2015). To investigate the vertical foliage distributions, a terrain corrected vegetation point cloud was used, where the heights of laser echoes were subtracted from an interpolated DTM (Zellweger et al. 2013).

3.3 Methods

3.3.1 Survey Area Selection

3.3.1.1 New Forest

Potential field plots in the New Forest were identified using the Canopy Height Model (CHM) from the LiDAR data underlying an OS map in ArcGIS software (version 10.2.2); obvious physical boundaries, such as footpaths, breaks between vegetation and drainage ditches, were used to demarcate areas to survey. Polygons were drawn (in a shapefile) in ArcGIS software (version 10.2.2) and amended using the LiDAR data to make sure they lined up precisely with the edge of the plot. This was to avoid paths and non-vegetative areas from contributing to the calculation of average canopy cover or openness over the plot from the LiDAR.

The area of each polygon was calculated and assigned a habitat-classification based on the tree species map (Sumnall 2013). The plots were first stratified according to whether they were woodland, heathland or scrubland. The woodland plots were then further stratified based on the dominant tree species in the plot (from the tree species map) into broadleaved, pine or ‘other conifer’ woodland categories. The ‘Heathland’ was categorised as wet or dry heath, and the ‘Scrubland’ was split into broadleaved, coniferous or mixed scrub species, in order to capture the variety of vegetation in the landscape. The ‘Scrubland’ plots were discriminated from the ‘Heathland’ plots as having multiple successional tree species, such as silver birch, present in the plot.

Plots were then pre-selected by appropriate and reasonable survey size. This was chosen to be between 3 and 6 ha for the woodland plots, and 5 ha to ~10 ha for the ‘Heathland’ and ‘Scrubland’ plots, due to the greater range of visibility and detectability of the birds. From these categories a selection of plots covering the variety of habitats were randomly stratified to survey (see Figure 3.5). The plots were distributed across the following land-cover types: ‘Heathland’ (4 plots), ‘Scrubland’ (4 plots) ‘Beech’ (6 plots), ‘Oak’ (6 plots), ‘Pine’ (6 plots) and ‘Other Conifer’ (6 plots).

However, upon starting fieldwork it was apparent that the classifications were not correct due to errors in the tree species map (Sumnall 2013), and so the plots were re-classified after a field-based vegetation survey (see Figure 3.5). The 32 plots were re-classified into 4 ‘Heathland’ plots, 4 ‘Scrubland’ plots, as before, 3 ‘Beech’ dominated plots, 3 ‘Oak’ dominated plots, 5 ‘Beech/Oak’ mixed plots, 3 ‘Pine/Broadleaved’ mixed plots, 5 ‘Pine’ plots and 5 ‘Other Conifer’ plots (Figure 3.5).

The ‘Pine’ class contained both Scots pine and Corsican pine, due to the difficulties in distinguishing between the species and the presence of hybrids in the New Forest. The ‘Other Conifer’ class consisted of the remaining non-pine conifer species in the New Forest, i.e. Douglas fir, Norway spruce, western hemlock (*Tsuga heterophylla*) and coastal redwood (*Sequoia sempervirens*). The ‘Beech’, ‘Oak’ and ‘Beech/Oak’ classes were predominantly beech or oak trees or a mix of the two species, but also contained other tree species, such as silver birch. ‘Pine/Broadleaved’ was a mix of pine spp. and predominantly either oak or beech. Fewer ‘Heathland’ and ‘Scrubland’ plots were surveyed as these plots were generally bigger in size. The ‘Heathland’ plots contained the occasional tree, but were predominately heather (*Calluna vulgaris*), gorse (*Ulex europaeus*) and various grass species.

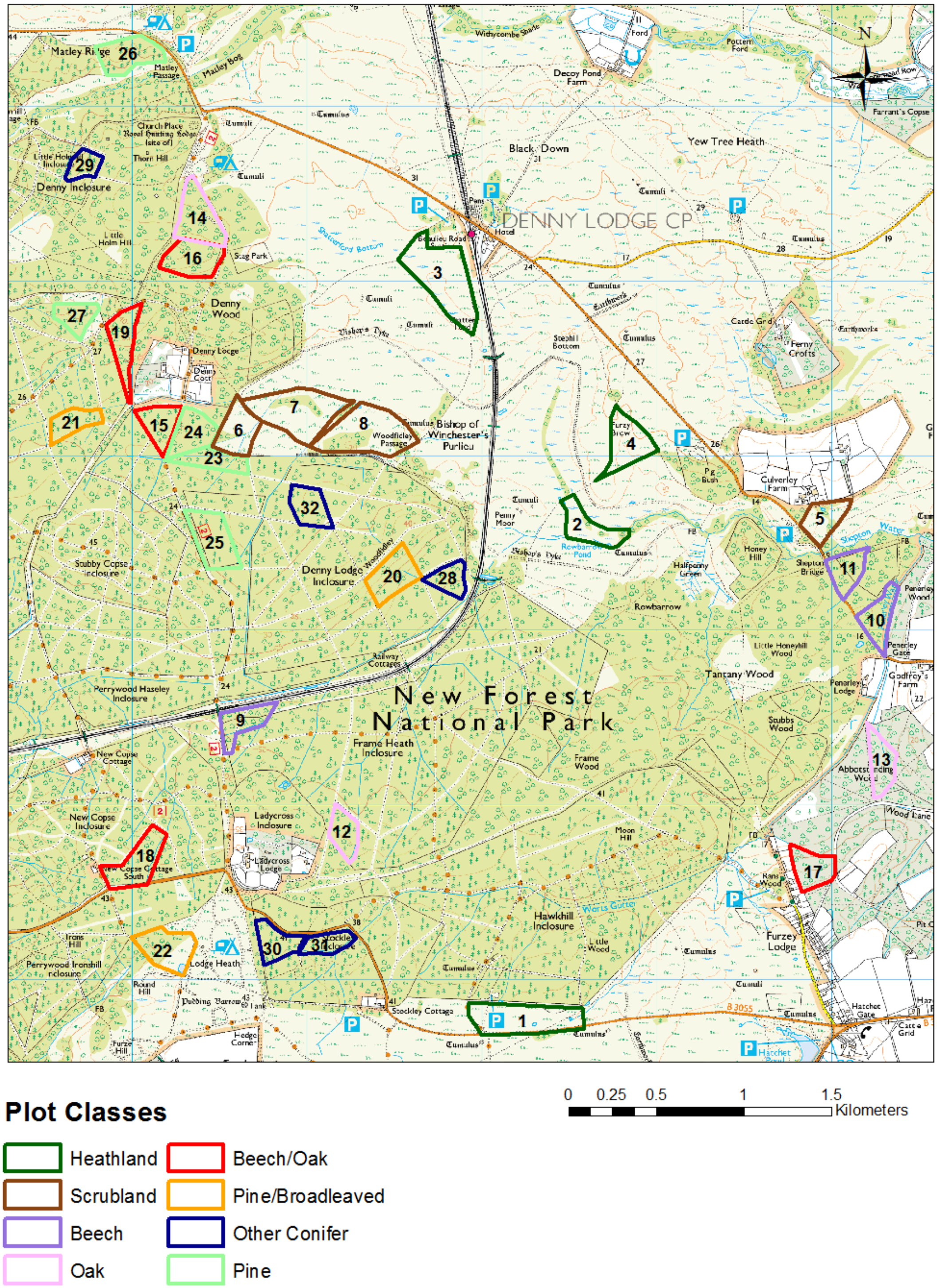


Figure 3.5. The 32 survey plots in the New Forest, numbered and colour coded based on the habitat classification from the vegetation field survey. (OS Map provided by Edina digimap © Crown copyright and database rights 2017 Ordnance Survey (100025252), ArcGIS software version 10.2.2).

3.3.1.2 Cambridgeshire

Line transects were randomly selected on an OS map as survey areas in Cambridgeshire (Figure 3.6). These followed public rights of way to minimise access issues, as a result, only interiors of woodlands open to the public were surveyed, and the edge of private woodlands were surveyed where paths allowed. The transects covered the interior or edge of small, medium and large woods and field margins with and without hedges, which were mostly along field boundaries and grassy margins (often maintained for walking), or gravel paths between fields. Transects were selected ensuring the distance between tangential transects was ~150-200 m in the woodland and ~250-500 m in open areas (Bibby et al. 1992).

Polylines were drawn on ArcGIS software (version 10.2.2) along the survey routes, the length of each transect was calculated and shortened or split into two separate transects (if long enough) in order for the length not to exceed ~1.3 km maximum (Figure 3.6). As a result, some of the transects were adjoining, for ease and to reduce travelling and survey time (Bibby et al. 1992). The longest transect was 1340 m on a field boundary and the shortest transect was 459 m along a woodland edge. The field transects were longer than the woodland transects as the survey time was lower due to the greater field of view and species-poor habitat (Bibby et al. 1992). The line transects were buffered, at 50 m in the woodlands and 100 m in the fields (as detectability was greater in the open fields), on an OS excerpt to demarcate the area to be surveyed from the transect line.

The survey consisted of 7 ‘Wood Interior’ transects (4 large wood interiors, 3 small wood interiors) and 9 ‘Wood Edge’ transects (Figure 3.6). The 5 ‘Field By_Wood’ transects are the buffered area of the fields adjacent to the surveyed woodland edges where the field was visible, numbered 18-22 in Figure 3.6. The 17 field boundary transects were classified using the proportion of hedge length to transect length (*P_HedgeLen*, see Table 3.4), as: 5 ‘Field_low%_Hedge’ (where *P_HedgeLen* was <30%), 6 ‘Field_med%_Hedge’ (where *P_HedgeLen* was 30-80%) and 6 ‘Field_high%_Hedge’ (where *P_HedgeLen* was >80%), colour coded in Figure 3.6.

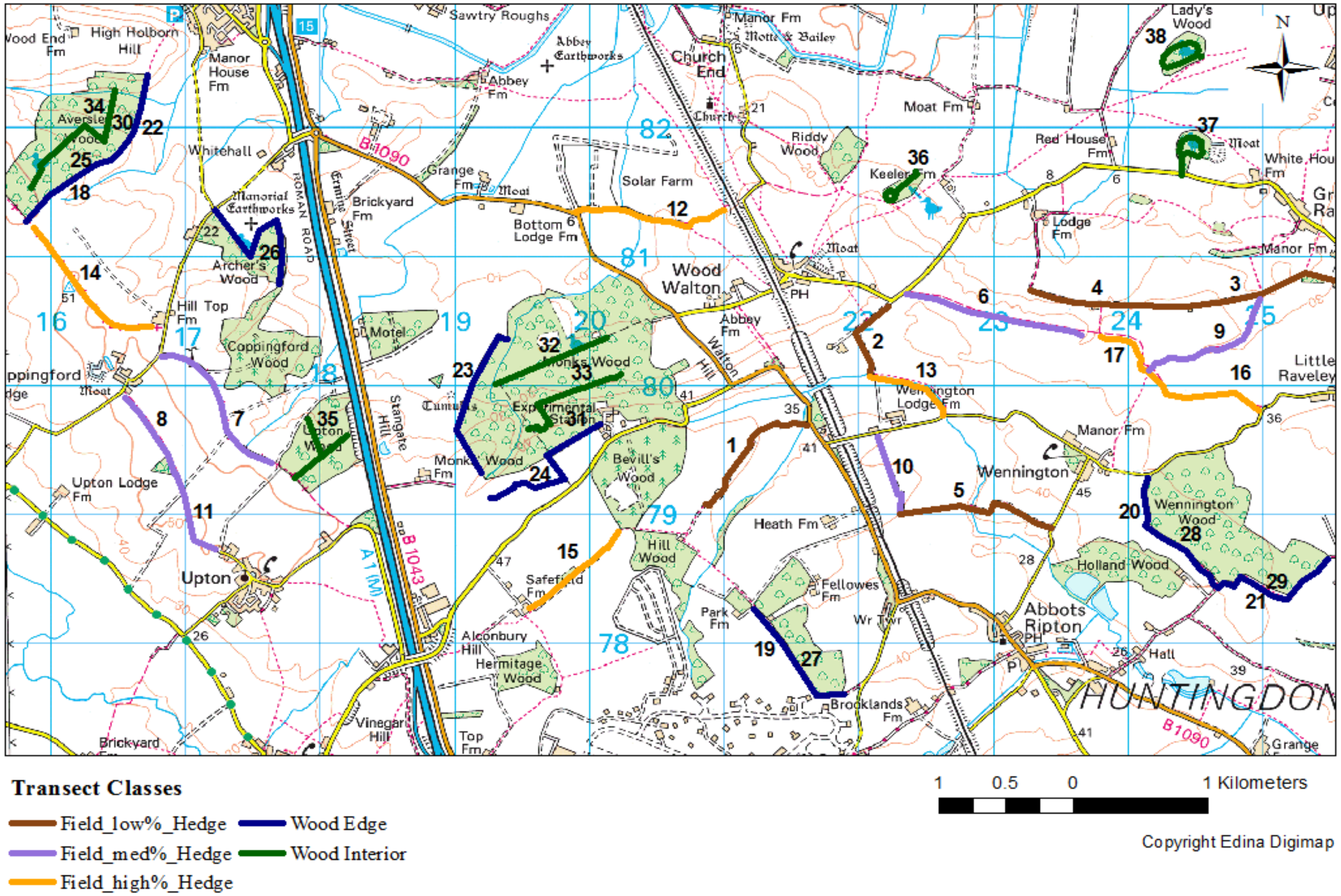


Figure 3.6. The 38 line transects surveyed in Cambridgeshire coloured by transect classification. 'Field By_Wood' transects are at the same location as the 'Wood Edge' transects, numbered 18-22 on the outer edge of the woods (OS Map provided by Edina digimap © Crown copyright and database rights 2018 Ordnance Survey (100025252), ArcGIS software version 10.2.2).

3.3.2 Vegetation Composition

3.3.2.1 New Forest

Due to the unreliability of the available tree species map, a survey of the vegetation composition in each of the plots in the New Forest was carried out in September/October 2016. This represented the dominant vegetation species, and therefore composition, of each habitat which will be used as a dependant variable in the analysis to investigate relationships with the bird measures. The percentage composition of tree species present in each plot was estimated through field observations and recorded in a table. The observer walked through the survey plot and noted all the woody species present in the plot (i.e. canopy and understorey species) and estimated a percentage composition of each of the canopy species. The actual percentage cover of the main canopy vegetation was then calculated (to account for the openness of the canopy) using the observed composition and the overall canopy cover of the plot as estimated from LiDAR data (i.e. *PCov_>5m*, see Table 3.3 for variable explanation), using Equation 1:

Equation 1

$$\% \boldsymbol{Species\ Cover} = (\% \boldsymbol{Species\ composition}) * \left(\frac{\boldsymbol{PCov\_ > 5m}}{\mathbf{100}}\right)$$

For example, if there was 75% composition of oak in a plot with 50% cover of overstorey (*PCov_>5m*) there would be 37.5% cover of oak in the plot (*PCov_Oak*: Table 3.1).

Shrub and ground vegetation species were also recorded, particularly in the ‘Heathland’ and ‘Scrubland’ plots. The percent cover of the lower vegetation in the ‘Heathland’ and ‘Scrubland’ was calculated from the field vegetation survey data and using aerial imagery to calculate approximate percentage cover. Due to difficulties in establishing the percent composition of understorey and ground vegetation, the presence or absence of holly, hawthorn (*Crataegus monogyna*), bracken (*Pteridium aquilinum*), birch and willow (*Salix spp.*) were represented as binary data for the analysis for all habitats, as these species were deemed to be potentially influential to the birds (Table 3.1).

Table 3.1. Vegetation composition variables based on the surveyed vegetation and explanations for the New Forest.

Variable Name	**Explanation**
PCov_Pine	Percent cover of pine species (Corsican, Scots and hybrids) using Equation 1.
PCov_ConsOther	Percent cover of 'Other Conifer' species using Equation 1, including: Douglas fir, western hemlock and coastal redwood.
PCov_Oak	Percent cover of oak using Equation 1.
PCov_Beech	Percent cover of beech using Equation 1.
PCov_BroadOther	Percent cover of other broadleaved species using Equation 1, including: sweet chestnut, rowan and sycamore.
PCov_Heather	Percent cover of heather.
PCov_Gorse	Percent cover of gorse.
Pres_Bracken	Presence (or absence) of bracken.
Pres_Holly	Presence (or absence) of holly.
Pres_Hawthorn	Presence (or absence) of hawthorn.
Pres_Birch	Presence (or absence) of silver birch.
Pres_Willow	Presence (or absence) of willow species.

3.3.2.2 Cambridgeshire

A vegetation survey was also carried out for the Cambridgeshire transects between 26th September 2017 and 6th October 2017 using similar methods as in the New Forest. Transects were re-walked and an estimate of percent composition of vegetation species present were recorded. Understorey and ground species were also recorded, however, again due to difficulty in estimating percent composition this was only recorded as presence of species and represented as binary data for the analysis. Hedge species along the field transects and any tree species within that transect or hedge were included in the vegetation composition survey. The species of vegetation making up patches of trees in the arable field transects were also recorded as these contribute to the overall vegetation composition of each plot. The percent cover of vegetation was calculated using Equation 1 with *PCov_2-5m* and *PCov_>5m* (Table 3.3) for Cambridgeshire, as the woody vegetation in the field transects was often lower than 5 m so this would encompass the lower strata. The most abundant vegetation in Cambridgeshire differed from the New Forest, therefore the vegetation composition variables differ, with *PCov_Ash*, *PCov_Oak* and *PCov_Maple* being the most abundant tree species (Table 3.2).

The main vegetation in the field transects was the crop, or the relevant land-use, and the vegetation species in the fields along each transect was noted at the time of the bird survey (April-June 2017). The proportion of crop per transect was calculated as a percent of the total transect area by isolating the individual fields as polygons on ArcGIS software (version 10.2.2). The crop types were grouped into *PCov_ImpGrass,* for any grass or possible pasture, *PCov_Cereal*, including winter and spring wheat, barley, and oats (*Avena sativa*), and *PCov_CropOther* which included non-cereal crop, such as oilseed rape, potatoes (*Solanum tuberosum)* and field beans (Table 3.2).

Table 3.2. Vegetation composition variables and explanation for Cambridgeshire.

Variable Name	**Explanation**
PCov_Ash	Percent cover of ash using Equation 1.
PCov_Oak	Percent cover of oak using Equation 1.
PCov_Maple	Percent cover of field maple using Equation 1.
PCov_Elm	Percent cover of elm using Equation 1.
PCov_BroadOther	Percent cover of other broadleaved woody species using Equation 1, including: silver birch, wild service, horse chestnut and sycamore.
PCov_Cons	Percent cover of conifer species using Equation 1, including: western hemlock and pine.
Pres_Thorns	Presence (or absence) of hawthorn and blackthorn (*Prunus spinosa*).
Pres_Bramble	Presence (or absence) of bramble (*Rubus fruticosus*).
Pres_Shrub	Presence (or absence) of Shrub vegetation including: willow, apple, cherry, hazel, elder, and dogwood.
PCov_ImpGrass	Percent cover of improved grass in fields.
PCov_Cereal	Percent cover of cereal crop: wheat, barley and oats.
PCov_CropOther	Percent cover of other crop: oil seed rape, field beans, potatoes.

3.3.3 Vegetation Structure

3.3.3.1 New Forest

87 metrics detailing the vertical and horizontal structure of the vegetation in each of the 32 plots were derived from the LiDAR point cloud data in the New Forest. From the 87 metrics produced, a range of nine ecologically meaningful LiDAR metrics were selected for the analysis, as shown in Table 3.3. As appropriate for each metric, some were derived using all LiDAR returns and some using only the first returns (i.e. top of vegetation canopy).

The height limit of 5 m for denoting the canopy trees and 2 m for separating shrubby vegetation (e.g. bramble and gorse) from understorey trees was established from field data in the New Forest as part of a separate project (Evans 2018). 0.5 m was determined as representing the ground layer whilst allowing for errors in the DTM.

Table 3.3. Explanation of the LiDAR derived metrics detailing the structural attributes of the vegetation chosen as ecologically meaningful for the analysis.

Variable Name	**Full Name**	**Derived From**	**Equation**	**Explanation**
Ht_Av	Height_ Average	First Returns	Average height of all first returns per plot (i.e. excluding any subsequent returns per pulse).	Details the height at the top of the vegetation canopy.
Ht_StDev	Height_ Standard Deviation	First Returns	The standard deviation of the height of the first returns per plot.	This indicates canopy surface roughness (at the plot level).
Ht_VDR	Height_ Vertical Distribution Ratio	All Returns	(95th percentile-50th percentile) / 95th percentile (i.e. similar to max-median/max).	Height Vertical Distribution Ratio (VDR) details the spread of vegetation through the vertical profile.
Ostorey_Ht	Overstorey Height	First Returns	Mean height of all first returns >5 m (taken as signifying an overstorey canopy return).	Mean overstorey height details the height of the trees above 5 m in the plot.
Ostorey_ PenDepth	Overstorey Penetration Depth	All Returns	The average distance between the first and final return of laser pulses with multiple returns (m).	Average laser penetration depth shows the openness of the canopy and may indicate canopy depth.
PCov_<0.5m	% Cover <0.5m	All Returns	Percentage of returns that come from below 0.5 m (signifying ground vegetation).	Indicates the percent of ground in the plot and all ground vegetation, such as grasses. This allows for small errors in the DTM.
PCov_0.5-2m	%Cover 0.5-2 m	All Returns	Percent of returns that come from between 0.5-2m (signifying shrubby vegetation).	Indicates the percent of shrub vegetation that contributes to the woodland structure.
PCov_2-5m	% Cover 2-5m	All Returns	Percent of returns that come from between 2-5 m (signifying understorey).	This details the percent of smaller trees, possibly under the canopy.
PCov_>5m	% Cover >5m	All Returns	Percent of returns that come from above 5 m (signifying overstorey or mature trees).	This details the percent cover of bigger trees that possibly make up the woodland canopy overstorey.

3.3.3.2 Cambridgeshire

The same metrics in Table 3.3 were calculated from the LiDAR data acquired for Cambridgeshire. It is worth noting, that the height threshold for separating understorey and overstorey, sub-dominant and dominant trees, in Monks Wood had previously been determined as 8 m (Hill and Broughton 2009). However, for the purposes of this study it was kept consistent with the New Forest (5 m) in order to allow cross comparison.

P_HedgeLen was also included in the vegetation structural metrics for Cambridgeshire, measuring the total hedge length as a proportion of transect length (Table 3.4). Hedge was defined as a linear, woody feature with foliage ranging the full vertical height of the vegetation, bordering the fields. Tree lines were not treated as hedges, but the hedge could contain trees (Tansey et al. 2009, Aksoy et al. 2010). There were no linear features representing hedges in the woodland plots, any shrubby vegetation species of similar height range were considered as understorey. The length of parallel and also perpendicular linear features (hedges) in the field transects were measured from the LiDAR on ArcGIS software (version 10.2.2) and calculated as a proportion of the transect length (*P_HedgeLen;* Table 3.4). If the field transect had a hedge running along its full length, then *P_HedgeLen* would be 100%, and if there were also hedges running perpendicular to the main hedge running the length of the transect then *P_HedgeLen* would be more than 100%.

Table 3.4. Extra structural variable for Cambridgeshire in addition to the structural variables in Table 3.3.

Variable Name	**Equation**	**Explanation**
P_HedgeLen	(Σ(Length of hedges)/Length of transect) x 100	Percent of hedge length as a proportion of the transect length.

Extra woodland variables were also calculated for the analysis in Cambridgeshire to investigate island biogeography and metapopulation theories as shown in Table 3.5. The area (*Wood.Area*) and perimeter (*Wood.Perimeter*) of each woodland was calculated by creating a polygon in ArcGIS software (version 10.2.2) and calculating both using the geometry function, then calculating the perimeter to area ratio (*Wood_P:A*). The percent area of other woodlands in a 500 m (*%_Woods_500m*) and 1 km (*%_Woods_1km*) buffer around each woodland polygon was also calculated. Distance to nearest wood (*Dist_Wood*) was calculated by simply measuring the shortest distance from the woodland survey transect to another wood, not necessarily a surveyed woodland, on the LiDAR in ArcGIS software (version 10.2.2).

Table 3.5. Extra woodland variable explanations for the Cambridgeshire woodland analysis.

Variable Name	**Explanation**
Wood.Area	Area of each surveyed woodland in hectares.
Wood.Perimeter	The length of the woodland perimeter of each surveyed woodland in metres.
Wood_P:A	Ratio of woodland perimeter to area.
%_Woods_500m	The percent cover of other woodlands in a 500 m buffer as a proportion of the area of the surveyed woodland.
%_Woods_1km	The percent cover of other woodlands in a 1 km buffer as a proportion of the area of the surveyed woodland.
Dist_Wood	The distance to nearest wood from transect.

3.3.4 Bird Survey Method

The bird survey approximately followed the British Trust for Ornithology's (BTO) Breeding Bird Survey (BBS) and Common Bird Census (CBC) transect method. The surveys were carried out in the breeding season, spring 2016 in the New Forest (from 11th April to the 15th June 2016), and spring 2017 in Cambridgeshire (from the 5th April to 26th June 2017). The surveys were carried out twice, once in April to early May, to capture mainly resident birds, then again in mid-May to June, to capture the migrant birds, ensuring all species were equally represented due to differences in breeding timing (Bibby et al. 1992, Heikkinen et al. 2004).

The birds were surveyed in the morning, starting around 6 am and stopping at midday (or earlier if they went quiet early due to high temperatures; Bibby et al. 1992). The surveys were carried out in favourable weather conditions of no precipitation and a maximum wind speed of 15 mph, as the birds would not be visible or audible in wet or windy conditions (Bibby et al. 1992). A walk and stop method was employed in both sites, where the surveyor walked for approximately 10-20 m so that the birds recorded would not be the same ones recorded already (Bibby et al. 1992). The birds were observed at that point for approximately 10 minutes or until all bird species could be identified, and then the observer walked further on, repeating the process along the route (Bibby et al. 1992). The species were identified by visual and/or audio cues (songs and calls; Goetz et al. 2007), and the individual's location in the survey area was recorded as accurately as possible on an OS map extract of the survey area using the BTO's two letter bird species code (see Appendix A1). Google Maps with a GPS on a mobile telephone was used to determine the location of the observer (and therefore the birds) in the plot or transect, as this was not always obvious especially in dense woodland or a featureless landscape (Google 2017).

The individual's activity was also recorded using the BTO's BBS/CBC codes (see Appendix A1), for example, singing, calling, alarm calling, and flying. The abundance, if there was more than one bird in the same location (often as a pair or flock), was recorded as a subscript number on the survey sheet and the sex of the bird was noted by the female and male sign (♀♂) if it was distinguishable. The survey was repeated with the same surveyor to reduce between-observer variation in effort, detectability and distance discrepancies (Bibby et al. 1992, Heikkinen et al. 2004). The time taken to complete each survey plot or transect varied with size and shape, therefore approximately 2-2 ½ hours maximum for each survey area was needed to allow thorough recording of the bird composition present.

In addition to bird data, the total recording time, weather information, and often the number of people walking and any incidents of any mammals present, were noted as extra data on the survey sheet.

3.3.4.1 New Forest

Plot sampling was carried out in the New Forest as the landscape was conveniently parcelled into habitat plots demarcated by paths, fences and/or ditches. The interior edge of the plot was surveyed first, then the middle of the plot was surveyed (often in a zigzag pattern) following methods in Bibby et al. (1992), to cover as much of the plot as possible. The route taken varied depending on the size, shape and the visibility in the plot, examples are shown in Figure 3.7. If possible the second survey was undertaken in the opposite direction from the first in order to maintain stochastisity in the data collection. If there was a significant edge, where the landscape parcel was adjacent to a different habitat type (namely open ground, heathland or scrubland), the outside edge of the parcel was also surveyed to record any birds specifically found at the edge (but not in the neighbouring parcel).

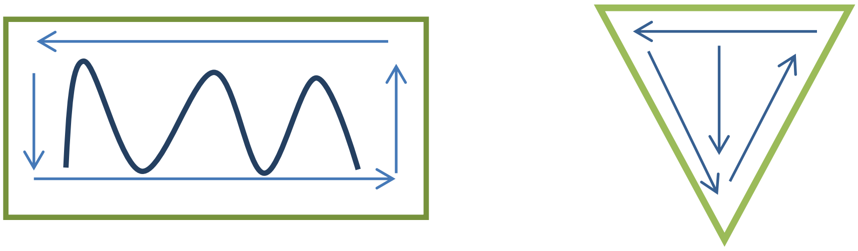


Figure 3.7. These diagrams represent a typical route taken to survey the plots in order to cover as much of the plot area as possible.

3.3.4.2 Cambridgeshire

Due to the layout of the landscape in Cambridgeshire and a result of access issues, line transect sampling was implemented, recording birds present either side of the transect line as it was walked. The birds were mapped with an approximate distance from the transect line, using buffers as guides and to demarcate the area of the plot to be surveyed on the OS map extract (50 m in the woodland and up to 100 m in the field transects). Transects were surveyed in only one direction in each survey to avoid double recording the same birds on the return (Bibby et al. 1992). If there were large hedges along the route the surveyor would look along the other side of the hedge or through a gap into the adjacent field, if possible. If feasible the transect was walked in the opposite direction on the second survey, however, due to access and parking issues this was often not possible. The buffered area on either side of the transect created a survey area that was comparable to the plots in the New Forest analysis.

3.3.5 Bird Data Manipulation and Analysis

3.3.5.1 New Forest

The field survey sheets were scanned and then digitised by hand as a point shapefile in ArcGIS (ArcMap version 10.2.2). Each point on ArcGIS software (version 10.2.2) was a bird encounter (which can be any number of the same species) denoted by the two letter bird code. Each point contained attributes detailing: the number of individuals, the sex if distinguishable, the activity of the bird, comments, the number of the survey plot (Plot ID) and the date and time of the survey. The two surveys from each site were entered into two separate shapefiles in ArcGIS software (version 10.2.2) with a total of 4,426 bird points of 67 bird species.

The data were then cleaned up by removing birds that were recorded as “possibly same bird” in the same survey, birds which were not directly associated with the plot (i.e. flying over) and raptors, due to their large habitat range (Newson et al. 2008). Hirundines and House Sparrows (*Passer domesticus*) were also removed due to their association with buildings and urban areas (Müller et al. 2009, Seoane et al. 2017). The juveniles, predominantly encountered in the second survey, were omitted as they did not contribute to the breeding population in the survey year, and birds located outside the survey area were also omitted from the analysis (Newson et al. 2008).

Bird abundance data of each species from each plot was selected as the highest abundance from the two surveys, as recommended by Bibby et al. (1992). This was then converted into a density measure (birds per hectare) in order to account for the variation in plot size. The data were stored as an Excel file and separate comma delimited (.CSV) files so they could to be read into R statistical software (version 3.5.2) for analysis (R Core Team 2018).

3.3.5.2 Cambridgeshire

In total 6,046 points of 60 bird species were recorded over the two breeding bird surveys in Cambridgeshire and the data were treated following the methods for the New Forest, above, with the exception of the bird density calculation. Hirundines, House Sparrow, raptors and birds flying over were again omitted from the data, along with Peacock (*Pavo cristatus*) as this was domestic.

Since the bird survey in Cambridgeshire was carried out using line transect sampling, distance sampling was carried out as the detectability of individual birds decreases with distance from the transect (Buckland et al. 1993a). Bird species differ in detectability, which is related to their behaviour and song (Buckland et al. 1993b). Distance sampling accounts for the birds not recorded at greater distances from the observer and was carried out for both bird surveys to estimate bird density following Buckland et al. (1993c), Thomas et al. (2005) and Miller et al. (2016). The probability of detection from the survey line was determined for either each species singularly or grouped. The bird species were grouped according to similarities in detectability in order to acquire the recommended 60-80 registrations for the analysis (Buckland et al. 1993d, 2008).

The woodland preferring birds were grouped as follows: Wrens (*Troglodytes troglodytes*) were numerous enough to be analysed alone, Chiffchaff (*Phylloscopus collybita*) and Willow Warbler were combined into a warbler group. The Thrush species (Blackbird, Song Thrush (*Turdus philomelos*) and Mistle Thrush (*Turdus viscivorus*)) were grouped together. Robin (*Erithacus rubecula*), Blackcap, Dunnock (*Prunella modularis*), Garden Warbler and Common Whitethroat were grouped together based on their similar level of detectability. The doves: Woodpigeon (*Columba palumbus*), Stock Dove (*Columba oenas*), Collared Dove (*Streptopelia decaocto*) and Turtle Dove (*Streptopelia turtur*), were grouped together along with the few Pheasants that were present in the woodland. The tit group contained Blue Tit and Great Tit based on their higher level of detectability; whereas Coal Tit (*Periparus ater*) was grouped with the quieter birds, along with Bullfinch (*Pyrrhula pyrrhula*), Goldcrest, Treecreeper (*Certhia familiaris*), Long-tailed Tit (*Aegithalos caudatus*) and Marsh Tit. The finch group contained the Goldfinch (*Carduelis carduelis*) and Chaffinch (*Fringilla coelebs*). Since Nuthatch (*Sitta europaea*) was considered to have a similar call as the Green Woodpecker, it was placed in the woodpecker group along with Great Spotted Woodpecker and the Corvids (Carrion Crow (*Corvus corone*), Jay (*Garrulus glandarius*), Jackdaw (*Corvus monedula*), Raven (*Corvus corax*) and Magpie (*Pica pica*)), due to their being more likely to be heard in the woodland rather than seen and were present in low numbers.

Due to the detectability being higher in the open fields, farmland species were grouped based on their association with the habitat, as follows: the Skylark, Yellow Wagtail (*Motacilla flava*) and Starling (*Sturnus vulgaris*) were more open field species. The Reed Bunting (*Emberiza schoeniclus*), Common Whitethroat and Yellowhammer were low shrub/hedge dwelling species and the remaining warblers, finches etc. which are associated with higher vegetation (Willow Warbler, Goldfinch, Greenfinch (*Chloris chloris*), Spotted Flycatcher (*Muscicapa striata*) and Linnet), made up the hedge species group.

Probability detection functions were fitted on to histograms using the *Distance* package in R (Miller et al. 2016, R Core Team 2018). The truncation distance was calculated as the average width of the transects when the detection probability reached 0.15 (Buckland et al. 1993b) over all the species/groups for each habitat (woodland and fields). The (uncorrected) combined density for each of the species/groups was calculated by multiplying the bird abundance survey data by the area of each of the transects in hectares (to the truncated distance of 40 m in the woodland and 65 m in the fields).

Distance sampling analysis was then carried out for each bird group with the relevant truncation using the hazard rate, half-normal and uniform fits and suitable adjustment terms. The best model for each species or group of species was chosen using AICc (AIC corrected for small sample sizes), which was mainly the hazard rate model and where there was very little difference in the models (2ΔAIC) the hazard rate was selected as the best model. This provided the average probability of detection and the individual density estimate per square metre for the woodland interior and the open field transects. The groups were all cluster analysed as the registrations were not always singular and the individual density output was used (Miller et al. 2016).

In order to account for the differences in species in each transect, a Density Multiplier for each species group was calculated from dividing the corrected combined density estimate by the uncorrected surveyed density, as in Equation 2. This was then applied to the density of the relevant species in that group (Equation 3) to calculate the corrected density of each species in each transect.

The 'Wood Edge' and the 'Field By_Wood' transects were excluded from the distance sampling estimations as the *Distance* package could not feasibly calculate the distance estimate on only one side of the transect line, as is the case in this instance (Miller et al. 2016). The density multiplier from the 'Wood Interior' transects were therefore used to correct the bird density in the 'Wood Edge' transects, and the density multiplier from the other field transects was used to correct the 'Field By_Wood' transects. The maximum bird density of the new density estimate was selected from the two surveys for each species in each transect following Bibby et al. (1992) and subsequently used in the analysis.

Equation 2

$$\text{Density Multiplier} = \frac{\text{Corrected Combined Density}}{\text{Uncorrected Combined Density}}$$

Equation 3

$$\text{Corrected Density} = \text{Density Multiplier x Density}$$

3.3.6 Bird Variables

Bird_Density was calculated for each plot/transect as the number of birds per hectare, as described in the previous section (3.3.5), this provided a comparable variable between survey areas. Species richness (*Spp_Richness*) is simply a count of the number of species present in each survey plot or transect. The Shannon-Wiener Diversity Index (*Spp_Diversity*) was calculated using the equation in Table 3.6 from the bird abundance/density data (corrected density for Cambridgeshire) in R using the *vegan* package (Oksanen 2015, Oksanen et al. 2018).

Spp_Decline was the number of declining bird species in each survey area. The bird population trends from the BTO's CBC/BBS bird trend data in England (the UK for the Common Crossbill and the Wood Warbler), was used to establish the bird species with declining populations (Massimino et al. 2017). For example, if a declining species, such as the Marsh Tit, was present in the survey area then this was one declining species, irrespective of the abundance of the species. This was then summed over the survey area (plot/transect) to give a total number of declining species present.

To account for the 'endangeredness' or level of conservation priority of a species the index, *Spp_Priority*, was calculated by weighting each bird species based on the level/percentage of population decline in England (Massimino et al. 2017). Each species was assigned a weight of 0-5 based on the percentage population change over the 10 years from 2005 to 2015 (Table 3.6 and Appendix A2). The weights for each species in each survey area were summed to give the overall *Spp_Priority* index for the plot/transect, so that the higher the value of the index, the higher the conservation priority or the number of birds with higher percent population declines. This therefore indicates greater importance of the habitat or surveyed area. For example, the Turtle Dove declined by 87% and was therefore assigned a 5 (Appendix A2). Missing trends for Firecrest, Hawfinch (*Coccothraustes coccothraustes*) and Dartford Warbler were obtained from averaging the trends of the other species with the same listing: RED, AMBER or GREEN. The index was then further weighted by the density of each priority species to form another version of the index, $Spp\_Priority^{w}$.

Spp_Rarity was calculated similarly to *Spp_Priority* by assigning each bird species a weight or value from 0-8 (Table 3.6 and Appendix A2), based on the number of breeding pairs, territories or singing males in the UK collated by Robinson (2005) in the BTO's "BirdFacts" web pages (Newson et al. 2008, Eaton et al. 2009, Wotton et al. 2009, Musgrove et al. 2013). The weighted value assigned to each bird species was then summed across each surveyed area (plot/transect) which provided a level of *Spp_Rarity*. Rarer bird species, therefore, had higher rarity values, which created higher index values for the habitats with more rare bird species present.

The Index of Relative Rarity (IRR) created by Leroy et al. (2012), was used to calculate the *Spp_IRR* index, using the Irr function in the *Rarity* package in R (see equation in Table 3.6; Leroy 2016, R Core Team, 2018). Firstly the bird species were weighted based on the UK breeding populations for each species using the rweights function (Robinson 2005, Leroy 2016, R Core Team 2018). The function deemed a cut-off at 68,000 breeding pairs in the New Forest and 177,500 breeding pairs in Cambridgeshire was appropriate. Species with populations higher than the cut-off were weighted with a zero (i.e. not rare), and birds with population numbers lower than the cut-off were weighted with an exponential increase as population size decreased, with 1 being the most rare (see Appendix A2). This apportioned greater importance to rarer species with lower population numbers. The Irr function used the surveyed bird abundance along with the weights of the species to calculate an Index of Relative Rarity (IRR) for each plot, bounded by 0 and 1. If the plot/transect contained no rare species, in other words if all species present in the plot/transect were common, then it was assigned an IRR value of 0 (Appendix A2). A survey area with an IRR of 1 indicates that the plot contains the complete pool of rare species in that landscape.

In order to investigate isolation and fragmentation of the Cambridgeshire woodlands extra woodland variables were calculated. Woodland *Spp_Richness* and *Spp_Diversity* were calculated by combining *Bird_Density* from the survey transects located in each woodland (where there was more than one). *Spp_Richness* and *Spp_Diversity* were then calculated as a proportion of the total transect length in the woodland (*T_Length* per metre), and as a proportion of woodland area (*Wood.Area* per hectare). *Spp_Richness* was then plotted against *Wood.Area* to evaluate the relationship.

Table 3.6. Equations and explanations of the bird indices; Spp_Diversity, Spp_Priority, Spp_Rarity and Spp_IRR.

Variable Name	**Index**	**Explanation**
Spp_Diversity	$Shannon\ Index\ (H) = -\sum_{i=l}^{s} p_i lnp_i$ (Eq 4)	*p_i* is the proportion of individuals belonging to *i*th species, *S* is total number of species in the community (richness).
Spp_Priority	Priority Species	0 = positive/stable trend 1= 1-20% population decline 2 = 21-40% population decline 3 = 41-60% population decline 4 = 61-80% population decline 5=81-100% population decline
Spp_Rarity	Rarity	0= ≥1 million pairs/territories; 1= 500,000-1 million pairs/terr 2= 250,000-500,000 pairs/terr 3= 100,000-250,000 pairs/terr 4= 50,000-100,000 pairs/terr 5= 10,000-50,000 pairs/terr 6= 1000-10,000 pairs/terr 7= 500-1000 pairs/terr 8 = <500 pairs/territories.
Spp_IRR	$Index\ of\ Relative\ Rarity = \frac{\sum\frac{a_i\, w_i}{N} - w_{min}}{w_{max} - w_{min}}\ (Eq\ 5)$	a_i the abundance of the ith species, *N* the total number of individuals, w_i weight of the ith species, w_{min} the minimum species weight w_{max} the maximum species weight.

3.3.7 Statistical Analysis on Plot/Transect Data

Preliminary analysis was carried out using box plots in R (version 3.5.2) and an ANOVA, for all the bird indices and for the vegetation composition and structural variables in order to visualise the spread of the data. The ANOVA (aov) in R (version 3.5.2) was used to determine if there were any statistically significant differences between the habitat classifications to allow further analysis of the classes (Chambers et al. 1992, R Core Team 2018). The summary of the linear model (lm) provided the R^2 value, which indicates how much of the variation in the habitat classes is explained by said variable. Post-hoc Tukey tests were then carried out on the significant ANOVA results using Tukey Honest Significant Differences (TukeyHSD in R) for each variable, to determine which habitat pairs were responsible for the differences (Bates n.d., Thomas et al. 2013).

For the woodland analysis in Cambridgeshire, a one sample *t*-test was carried out in R (R Core Team 2018), in order to determine if there were any statistically significant differences of the *Spp_Richness* and *Spp_Diversity* variables (per metre transect and per hectare area) between the woodlands.

3.3.7.1 Correlation

A correlation matrix of Pearson's r was then calculated in R (version 3.5.2), using the rcorr function in the *Hmisc* package, to determine if there were any significant univariate relationships between the bird indices and habitat vegetation structure and composition variables (Harrell 2019). The P values for the correlations were also calculated using the rcorr function in the *Hmisc* package in R (R Core Team 2018, Harrell 2019) and the significant relationships ($P < 0.05$) were highlighted in bold on the correlation matrix. The habitat (vegetation structure and composition) variables were also assessed for collinearity, which would affect any further analysis, and was presented as a matrix in Appendix B4 for the New Forest and Appendix C6 for Cambridgeshire. The significant relationships with the highest Pearson's r value for each of the bird indices were then plotted as regression graphs to visualise and highlight the linear relationships between the bird indices and the habitat variables.

3.3.7.2 Multi-Model Inferencing

A multi-model inferencing and averaging approach was carried out in order to investigate multi-variate relationships between the bird indices and the vegetation composition and structural variables following Grueber et al. (2011) and Collop (2016). First a global model was created for each bird index with the independent (habitat) variables selected as the statistically significantly correlated habitat variables from the correlation matrix and from ecological knowledge. No more than 7 variables were included in the global model for the New Forest, and 13 variables per global model for Cambridgeshire to adhere as closely to the 3 survey plots to 1 variable rule.

The relevant generalised linear model family, such as Poisson for count data, Gamma/Inverse Gaussian for continuous data bounded by zero etc., with the best fitting link function (identity, log etc.) was selected for the global models for each of the bird indices (Burnham and Anderson 2002). *Spp_IRR* was ‘nudged’ away from 0 by adding a small transformation of 0.0001 to the data following Thomas et al. (2017). To prevent multi-collinearity, the variance inflation factor (vif in the *car* package) was calculated for each global model and any variables with a value over 10 were removed from the model (Dormann et al. 2013, Fox et al. 2018). The global models were then assessed visually for model validation of the assumptions.

The independent variables were then standardised using the standardize function in the *arm* package in R (Gelman et al. 2018, R Core Team 2018). Continuous data were rescaled by subtracting the mean and dividing by 2 standard deviations, and binary data were mean centered, with the data having the mean equal to 0 and the difference equal to 1 (Gelman 2008, Grueber et al. 2011). Standardising the variables allows for a direct comparison of variables that were on different scales, and allows for some multi-collinearity (Gelman 2008, Grueber et al. 2011, Cade 2015). The Negative Binomial was used for *Spp_Rarity* due to overdispersion of the data, and Beta Regression was used for *Spp_IRR* as the data were bounded by 0 and 1, following Thomas et al. (2017), therefore the rescale function, also in the *arm* package, was used to standardise each of the independent variables in these global models (Gelman et al. 2018).

The standardised global model was then passed through the dredge function from the *MuMIn* package, following Grueber et al. (2011). This generated a sub-set of models, limited to up to three variable combinations for the New Forest and up to four variable combinations for Cambridgeshire (two variable combinations for the separate wood/field habitat analysis), to adhere to the rule of 10 survey plots to 1 independent variable (Burnham and Anderson 2002, Grueber et al. 2011, Barton 2015). The top candidate model sets, up to two ΔAICc (corrected for small data sets: Akaike and Csaki, 1973), were then selected using get.models, following Grueber et al. (2011). Each model in the set was assessed further for collinearity and models containing correlated variables with $r > 0.5$ on the variable correlation matrix Appendix B4 and C6, were omitted from the model set (Dormann et al. 2013, Bani et al. 2018). The remaining candidate set of top models to two AICc and an accumulated weight (*acc w*) of ~1 was used, in order to acquire enough models to avoid having weak variables, and also to avoid having too many models leading to poor model weights and spurious results (Anderson and Burnham 2002, Burnham and Anderson 2002, Grueber et al. 2011). The candidate set of models were deemed the “best approximating models” as not one model could be coined the “best model” with so little difference in AICc (Symonds and Moussalli 2011). If the top model was more than two AICc different from the next model, inference was made from that single model.

Model averaging was then carried out on the candidate set of models using the model.avg function also in the *MuMIn* package, (Burnham and Anderson 2002, Grueber et al. 2011, Barton 2015). The R output from the model average summary provided the degrees of freedom, Log-likelihood, AICc (AIC corrected for small sample sizes, Akaike and Csaki, 1973), difference in AICc between models (ΔAIC) and the weight (w_i) of the model. From this, the accumulated weight (*acc* w_i) was calculated and the adjusted R^2 or Pseudo-R^2 (for Poisson, Negative Binomial and Beta Regressions) were calculated in R from the summary output of each individual generalised linear model run (R Core Team 2018). The relative importance and model averaged coefficients of the standardised predictors for the model set for each of the bird indices were also presented, which includes the full average estimate, adjusted standard error (SE), and the lower and upper 95% confidence intervals (CI) (Grueber et al. 2011, Collop et al. 2016).

Due consideration was taken to address the problems that multiple tests can cause in terms of Type I errors. Measures such as Bonferroni corrections (Napierala 2012) have been widely applied to address this, however, there are also significant criticisms of this approach including reductions in the probability of finding any significant results when sequential Bonferroni is applied. Following the recommendations of Moran (2013) and Cabin and Mitchell (2011) a reasoned approach was adopted which considered the conflicting responses of when, whether and how to use Bonferroni corrections and would allow analysis of this complex system, with the conclusion that it was not implemented. Thus, effect sizes and interpretation of the results were assessed alongside the P values for each analysis.

3.3.8 Community Composition Analysis

The community composition of birds in the plots was investigated by generating Multidimensional Scaling (MDS) plots from the density of each bird species in each survey plot or transect. The MDS graphs are a visual representation of the pattern of proximities, similarities and differences, among a set of objects (Borgatti 1997). All MDS graphs were produced in R version 3.5.1/3.5.2 (R Core Team 2018) and run using the function metaMDS in the *vegan* package (Oksanen 2010, 2015, Oksanen et al. 2018). The Bray-Curtis dissimilarity was selected in the metaMDS which transforms the data (if required to fit the data on to the plot) using the square root or Wisconsin double standardisation. Wisconsin divides the species by their maxima and the survey plots or transects are standardised to have equal totals (Oksanen 2015). The function metaMDS runs the analysis with multiple random starts and stops after a certain number of tries (often 20) or until it finds two similar configurations with minimum stress which produces the best fitting model (Oksanen 2010). It then scales and rotates the solution so that the largest variance of site scores will be on the first axis, and adds species scores to the configuration as weighted averages of site scores, but expands them so that the species and site scores equal the variance (Oksanen 2010, 2015). The plots are produced by plotting the distance matrix calculated from the data in N-dimensional space (2D is the default).

MDS plots were created to display similarities in bird species composition between the habitats in each landscape at the landscape level; all 32 plots for the New Forest and 38 transects in Cambridgeshire, for the woodland only; 24 plots in the New Forest and 16 transects in Cambridgeshire, and the 22 field transects on their own in Cambridgeshire. The birds responsible for the ordination of the survey areas were presented on a separate MDS plot. The envfit function, also in the *vegan* package, was used to associate the habitat variables as environmental factors to the ordination of the survey plots/transects by superimposing arrows of significantly ($P < 0.05$) related factors on the MDS of the survey areas (Oksanen 2015, Oksanen et al. 2018). MDS plots were produced (i) with the survey plots or transects on their own, (ii) with the survey plots or transects annotated, (iii) with the associated bird species to the plots or transects, (iv) with vegetation composition data, and (v) with vegetation structural variables. This was repeated for each landscape, for the woodland survey areas only, and the field transects only in Cambridgeshire. The non-woodland plots in the New Forest survey (‘Heathland’ and ‘Scrubland’) were not included on their own MDS as the separate ordination was comparable to the landscape ordination.

4 Bird-Habitat Relationships for the New Forest

4.1 Abstract

Biodiversity is often used to indicate habitat health, however, this often masks rare and declining bird populations with more common species. Therefore, the current study investigates measures of the number of declining bird species, conservation priority, rarity and the index of relative rarity, along with density, diversity and species richness, to understand bird-habitat relationships. Whilst biodiversity is important to maintain, the scale at which it is monitored should include larger landscapes, or gamma diversity, to maximise the conservation of the greatest number of species. The current study showed that the landscape of the New Forest contained multiple bird habitats, such as varied vegetation structure and composition in scrubland increasing bird diversity, beech woodlands providing habitat for declining bird species, and heath and ornamental conifer plots providing habitat for some of the rarest birds in southern England. This multitude of habitats is therefore vital to maintain gamma diversity, with different habitats often important for different reasons. The exception to this was pine plantations which were deemed poor bird habitat by all of the bird indices, as a result of the lack of understorey and the immature stage of growth.

4.2 Introduction

Woodland only covers 13% of the UK and 90% of forest cover in Britain is plantations (Donald et al. 1997, Hartley 2002). The New Forest, as stated in Section 3.1.1, is a National Park with a unique and protected landscape of enclosed and unenclosed woodland and heathland with high levels of grazing. The New Forest is highly managed, for coppice, plantation woodland and also heathland burning is carried out to prevent scrub encroachment and protect this rare and important habitat. However, successional scrubland habitat, as opposed to closed forest or homogeneous landscapes (in temperate regions) can support a greater diversity of birds, due to the gradient in tree growth (structure) and variety of tree species (composition) present (Ralph 1985, Tews et al. 2004). The more open structure before tree canopy closure favours vegetation diversity (Bazzaz 1975, Welsh 1987) and offers birds a range of habitat options (Wiens and Rotenberry 1981, Estades 1997).

Diversity has often been used as a measure of habitat health. However, as habitat preferences differ between species, how habitats differ in alternate bird measures, such as rarity, or numbers of declining species, is less well known. In order to investigate bird-habitat relationships in the New Forest, the usual bird indices of density, species richness and diversity, were coupled with indices depicting the numbers of declining and conservation priority bird species, and species rarity were related to vegetation composition and structural metrics to ascertain the important habitat characteristics.

4.3 Methods

A total of 32 plots (size 3 – 10 ha) were selected encompassing eight habitat classes: Heathland, Scrubland, Beech, Oak, Beech/Oak, Pine/Broadleaved, Pine and Other Conifer (3.3.1.1, pp. 57 – 59). The vegetation composition in each plot was recorded, either as percentage cover or as presence/absence, for the tree, shrub and field layers using both field and aerial data (3.3.2.1, pp. 62 – 63). Nine ecologically meaningful LiDAR metrics were selected for analysis from an original 87 describing the vertical and horizontal vegetation structure in each plot (3.3.3.1, pp. 64 – 65). Breeding bird surveys were carried out (twice) using a plot based method (3.3.4.1, pp. 68 – 69), the data for analyses were selected to maximise detected numbers (3.3.5.1, pp. 69 – 70), and from this the eight bird variables were calculated for each plot (3.3.6, pp. 73 – 75).

For data analysis, firstly, for vegetation composition and structure and the bird indices, the data were tabulated and graphed for a visual inspection of differences across the transects. Followed by an ANOVA for all three data sets (3.3.7, pp. 75 – 76), with the transects grouped into the habitat classifications (the results of this are presented in Supplementary Material 4.7). The bird variables were then correlated with the vegetation variables in order to establish if there were any significant bird-habitat relationships (3.3.7.1, pp. 76). Significantly related variables were then used to populate global models to carry out multi-model inferencing and averaging (3.3.7.2, pp. 76 – 78). Multiple variable models were fitted to explain bird-habitat relationships, as such, the number of models in each candidate set differed depending on the bird index. Model inference and averaging were made on standardised variables in a candidate set of models to six ΔAICc and an accumulated weight (*acc* w_i) of approximately one (~100% confidence set) for each bird index following Symonds and Moussalli (2011) and Burnham and Anderson (2002). However, if the top model was more than six AICc different from the next model then inference was made from that top model alone.

4.4 Results

4.4.1 Vegetation Composition Across the 32 plots

A total of 32 plots were enumerated across eight different habitat classes in the New Forest. ‘Heathland’ plots were composed of 70-95% heather (*Calluna spp.*) and/or semi-natural grassland, often with gorse (*Ulex spp.*) and woody vegetation, often as standalone trees or a small stand (Figure 4.1 and Table S 4.1). The fourth plot (plot 2) is notably different; a wet grassy bog area with 25% woody vegetation as a patch dominated by willow trees (*Salix spp.*). The ‘Scrubland’ plots were defined as an ecotonal habitat between the heathland and woodland, containing at least 10% cover of tree species representing the successional development of scrub, however, ‘Scrubland’ plot 5 was composed of more woody vegetation (~50%) with bracken, semi-natural grassland and gorse (Table S 4.1).

Broadleaved woodland plots were categorised into ‘Beech’, ‘Beech/Oak’ and ‘Oak’ classes, but all were a mixture of beech and oak trees in differing proportions (Figure 4.1 and Table S 4.2). Canopy cover for the broadleaved woodland classes varied between ~50-90%, with limited understorey (typically of holly and hawthorn, in addition to saplings of beech and other overstorey trees) and open areas dominated by grass, bracken and bramble. The conifer dominated plots were categorised into ‘Pine’, ‘Pine/Broadleaved’ and ‘Other Conifer’. The ‘Pine’ class was composed of ≥80% pine (Scots pine, Corsican pine, and hybrids), and the ‘Pine/Broadleaved’ class had a lower composition of pine (45-65%) and a higher composition of broadleaved species (45-60%), such as beech, oak or silver birch (Figure 4.1 and Table S 4.2). The ‘Other Conifer’ class was more heterogeneous: two plots were dominated by Douglas fir; another two by western hemlock; and one ‘Other Conifer’ plot was mostly coastal redwood (50%) mixed with Douglas fir

(Figure 4.1 and Table S 4.2). For the conifer plots, canopy cover was between ~65-95%, again with an understorey cover of < 10%, and ground cover dominated by grass, bracken and needles. Potentially ecologically important understorey plant species, such as bracken, holly, hawthorn, silver birch and willow spp., were also recorded for each survey plot as presence/absence data (Table S 4.3).

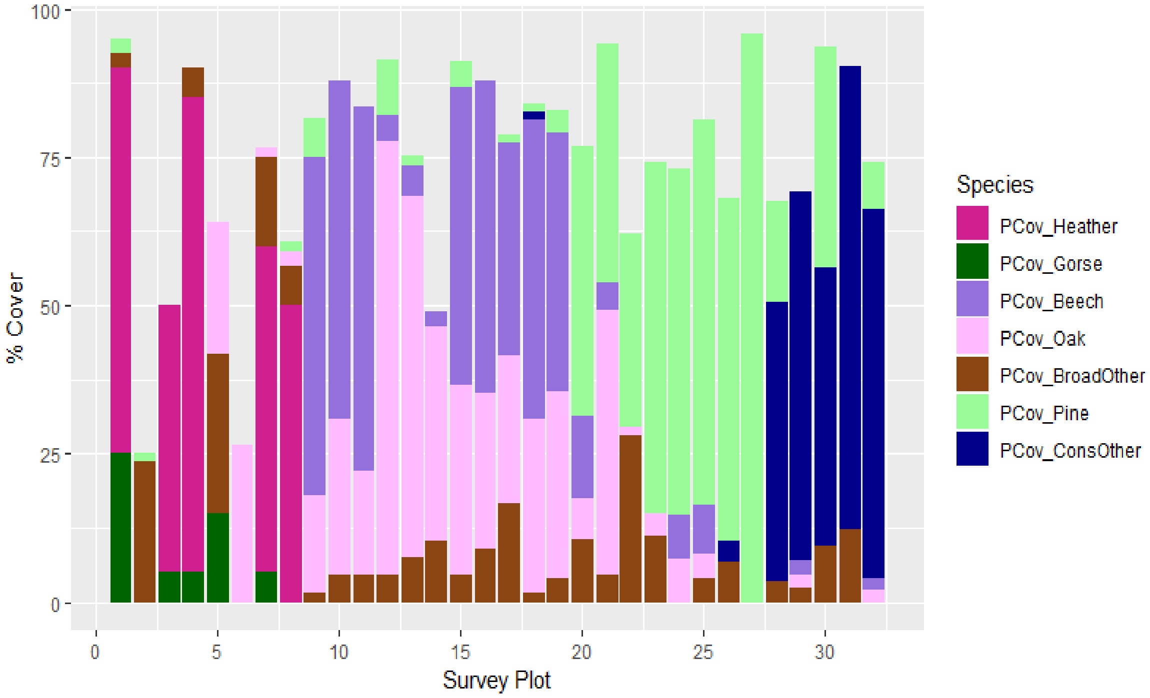


Figure 4.1. The composition of vegetation species in all 32 plots in the New Forest, calculated using the canopy cover metric; PCov_>5m for tree species. Missing land cover not shown in this graph includes: infrastructure (i.e. roads and buildings), water bodies and rough grass. See Table S 4.4 for values (R version 3.4.1).

4.4.2 Vegetation Structure Across the 32 Plots

The LiDAR-derived metrics showed variation in vegetation structure both between and within the habitat classes, highlighting that similar habitat types can contain different structural attributes/profiles (Figure 4.2). As expected, the woodland classes had a higher *Ht_Av, Ht_StDev and PCov_>5m* than the two non-woody classes ('Heathland' and 'Scrubland'), which had a higher *PCov_<0.5m* and *Ht_VDR* (Figure 4.3). Moreover, *Ostorey_Ht* showed that the trees in the 'Heathland' and 'Scrubland' plots were considerably shorter than the trees in the woodland plots (an average of 10 m compared with 16.4 m), either as low growing, successional species (such as silver birch or hawthorn), or young trees seeded from nearby woodlands. Higher *Ht_VDR* in the 'Heathland' and 'Scrubland' plots also reflects a higher proportion of vegetation cover in the lower strata, in this case particularly *PCov_<0.5m*. (This is because a high *Ht_VDR* in a plot which contains at least one tree can only be derived if most of the vegetation is not in the overstorey tree canopy layer, i.e. >5m). It is worth noting that the *PCov_0.5-2m* and *PCov_2-5m* for 'Heathland' and 'Scrubland' habitats were within the range of the six forest habitat types (Figure 4.3). *PCov_2-5m,* and to a lesser extent *PCov_0.5-2m*, varied greatly between the plots across all habitat classes (Figure 4.2) and hence showed no significant difference (see Table S 4.7). *PCov_0.5-2m* and *Ostorey_PenDepth* were highest in the 'Pine' plots; here the low *PCov_2-5m* created increased light availability allowing bracken to persist at 0.5-2 m (Figure 4.2).

Differences between the woodland and non-woodland classes were expected to be large, and proved to be so for all LiDAR variables apart from vegetation heights of 0.5-2 m and 2-5 m. However, variation within the woodland plots was more complex than differences attributable to the distinction between conifer and broadleaved woodlands alone. The woodland classes were structurally indistinguishable from one another in all nine LiDAR-derived variables (with the noted exception of the 'Pine' having a more open canopy compared with 'Beech/Oak' and 'Other Conifers' plots). The woodland plots, thus had overlapping ranges in terms of canopy height, variance and cover, and understorey layering. Therefore, analyses assessing the effects of structure on bird species are more relevant at the plot level rather than at the habitat class level, especially through multi-variate analyses with vegetation composition within the New Forest.

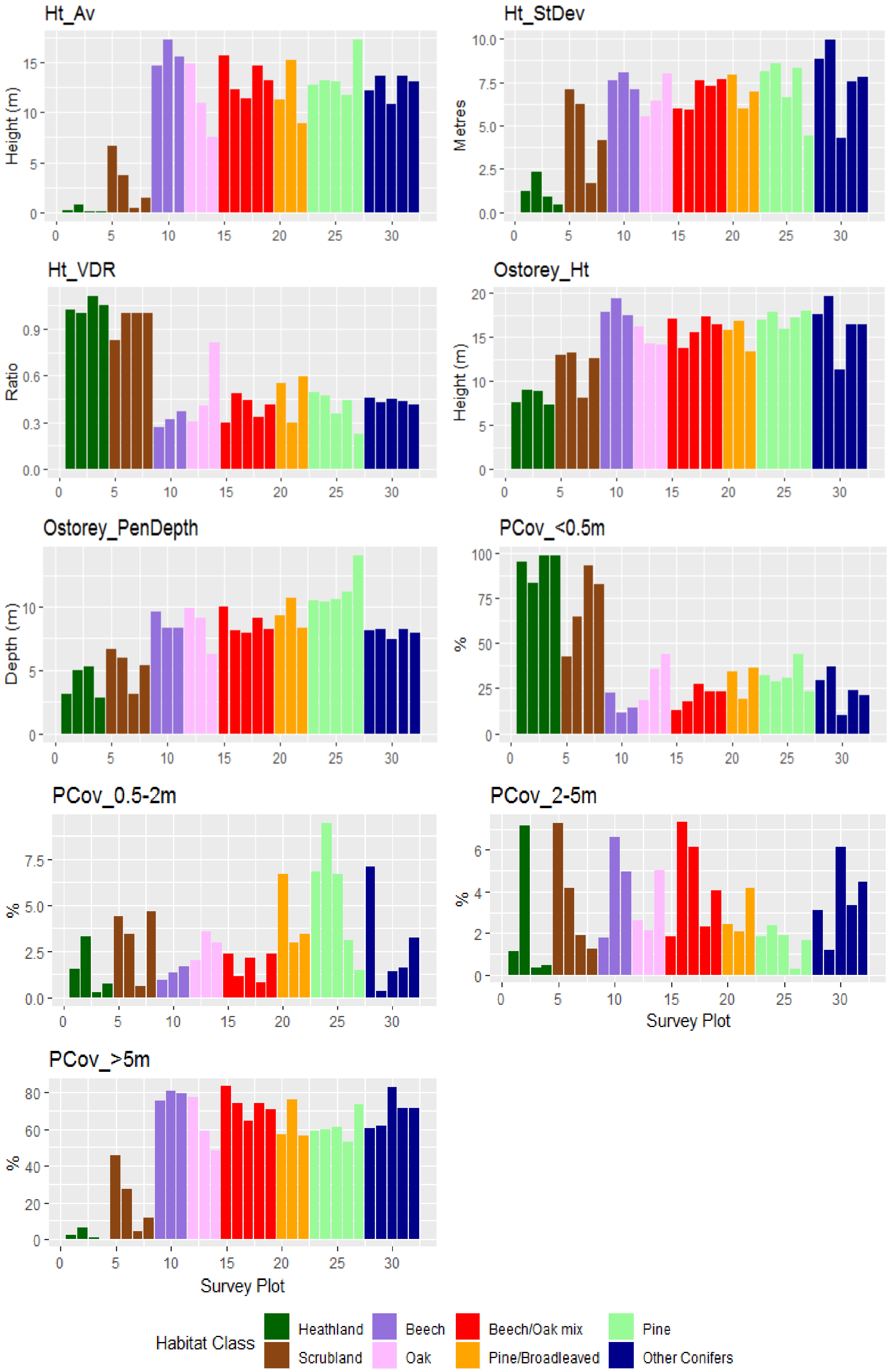


Figure 4.2. LiDAR metrics detailing the vegetation structure in all 32 plots in the New Forest in order of habitat classification. See Table 3.3 for metric derivations (R version 3.4.2).

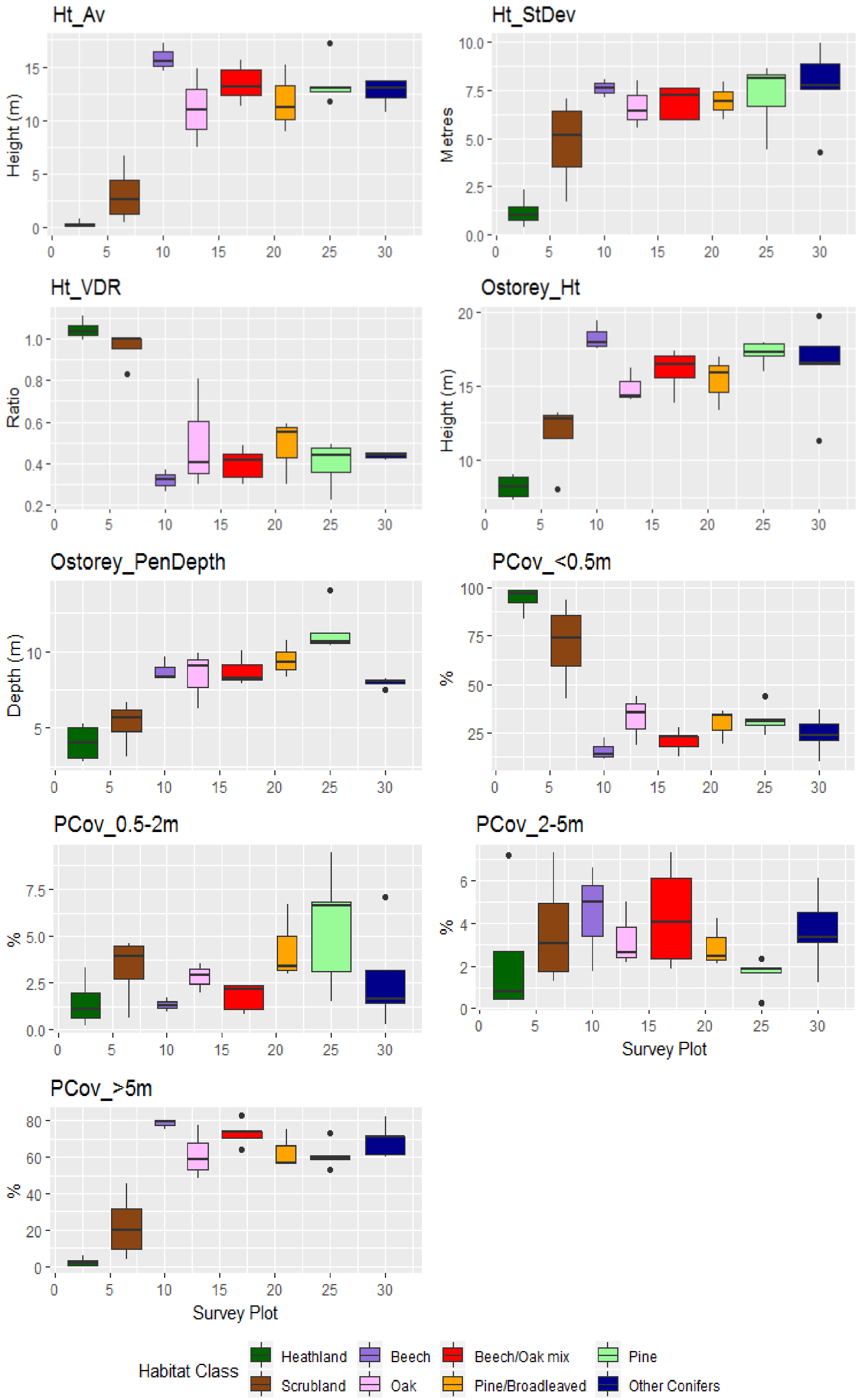


Figure 4.3. The range of the LiDAR metrics detailing the vegetation structural variables in each habitat classification for the New Forest. See Table 3.3 for metric explanation (R version 3.5.2).

4.4.3 Bird Indices

A total of 4,426 birds of 67 species were surveyed in the New Forest from the two surveys over nine and a half weeks from 11th April to 15th June 2016. From these, 58 species comprising 2,617 individuals were selected for the analysis.

The habitats in the New Forest varied in all bird indices both between and within habitat classes (Figure 4.4). 'Heathland' had consistently low *Bird_Density* (mean = 6.7 birds/ha), as did 'Pine' to a lesser extent (mean = 12.0 birds/ha). 'Scrubland' had disproportionately high *Spp_Richness*, *Spp_Diversity* and measures of *Spp_Priority* and *Spp_Rarity* given the overall low *Bird_Density* (mean = 14.1 birds/ha) in comparison with the woodland plots (mean = 19.2 birds/ha; Figure 4.4 and Figure 4.5). 'Other Conifer', on the other hand, had consistently low *Spp_Priority* (mean = 5.0 priority index). 'Beech' had the highest value of *Spp_Decline* (13.0), possibly as a result of the high *Bird_Density* in these plots (mean = 25.1 birds/ha) or indicating an effect of national beech decline. The $Spp\_Priority^{w}$ metric was higher in the broadleaved plots, again possibly due to the high density of birds in this habitat (Figure 4.4).

Overall, 'Heathland', 'Pine' and 'Other Conifer' tended to be lower across all bird metrics, and 'Scrubland' and broadleaved plots tended to be higher (Figure 4.4 and Figure 4.5), supported by the significant differences (at $P < 0.05$) being attributed to a combination of these habitat classes in the post-hoc Tukey test (Table S 4.11). Exceptions to this were that *Bird_Density* was significantly higher in the 'Other Conifer' plots than in the 'Heathland', and *Spp_Priority* was significantly lower in both 'Beech' and 'Pine/Broadleaved' than in the 'Scrubland' (Figure 4.5 and Table S 5.10). More often than not, on average, 'Pine' had lower values of the bird indices than the 'Other Conifer' plots, apart from *Spp_Decline* and *Spp_Priority* which were slightly higher (5 cf. 6 and 4.4 cf. 5.4 respectively; Figure 4.5).

The highest values of *Spp_IRR* were in the 'Other Conifer' and 'Heathland' classes suggesting that these habitats have conservation value in the New Forest (Figure 4.4). However, in spite of this, there was no significant difference in *Spp_IRR* between the habitat classes (Table S 5.10), as a result of the large range of values in the 'Heathland' and 'Other Conifer' habitats, as shown in Figure 4.4 and Figure 4.5. Since *Bird_Density* was extremely low in the 'Heathland', this presumably resulted in the absence of rare species in some of the plots or being present in such low numbers that they were not detected. This further indicates that analyses should be carried out at the plot level to investigate the vegetation characteristics (of structure and composition), driving the high *Spp_IRR* in these plots. Furthermore, community analysis is also required to explore the species contributing to the bird indices.

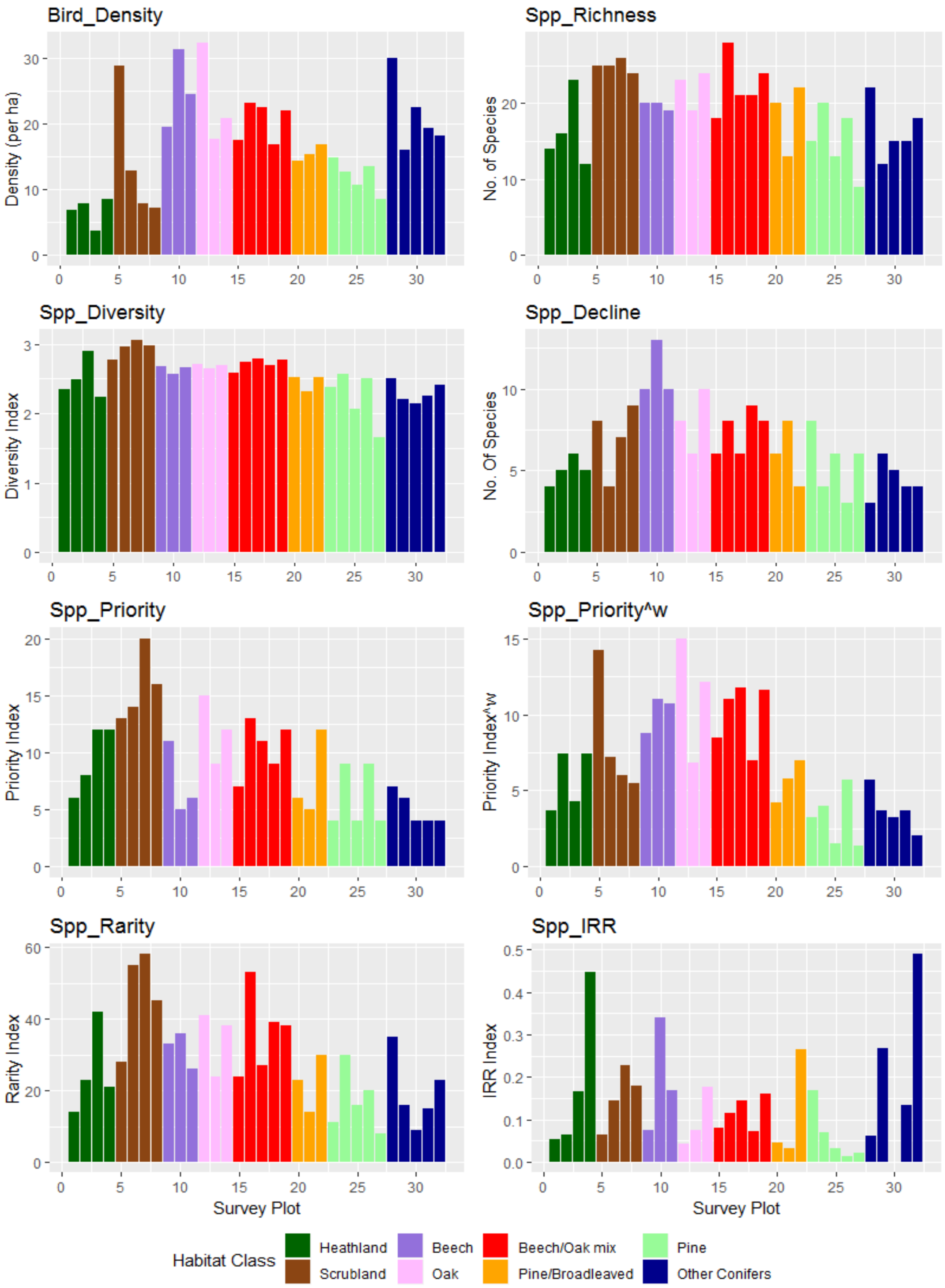


Figure 4.4. The eight calculated bird indices for each of the 32 survey plots in the New Forest. See Section 3.3.6 and Table 3.6 for derivation of bird indices. See Table S 5.9 for values (R version 3.5.2).

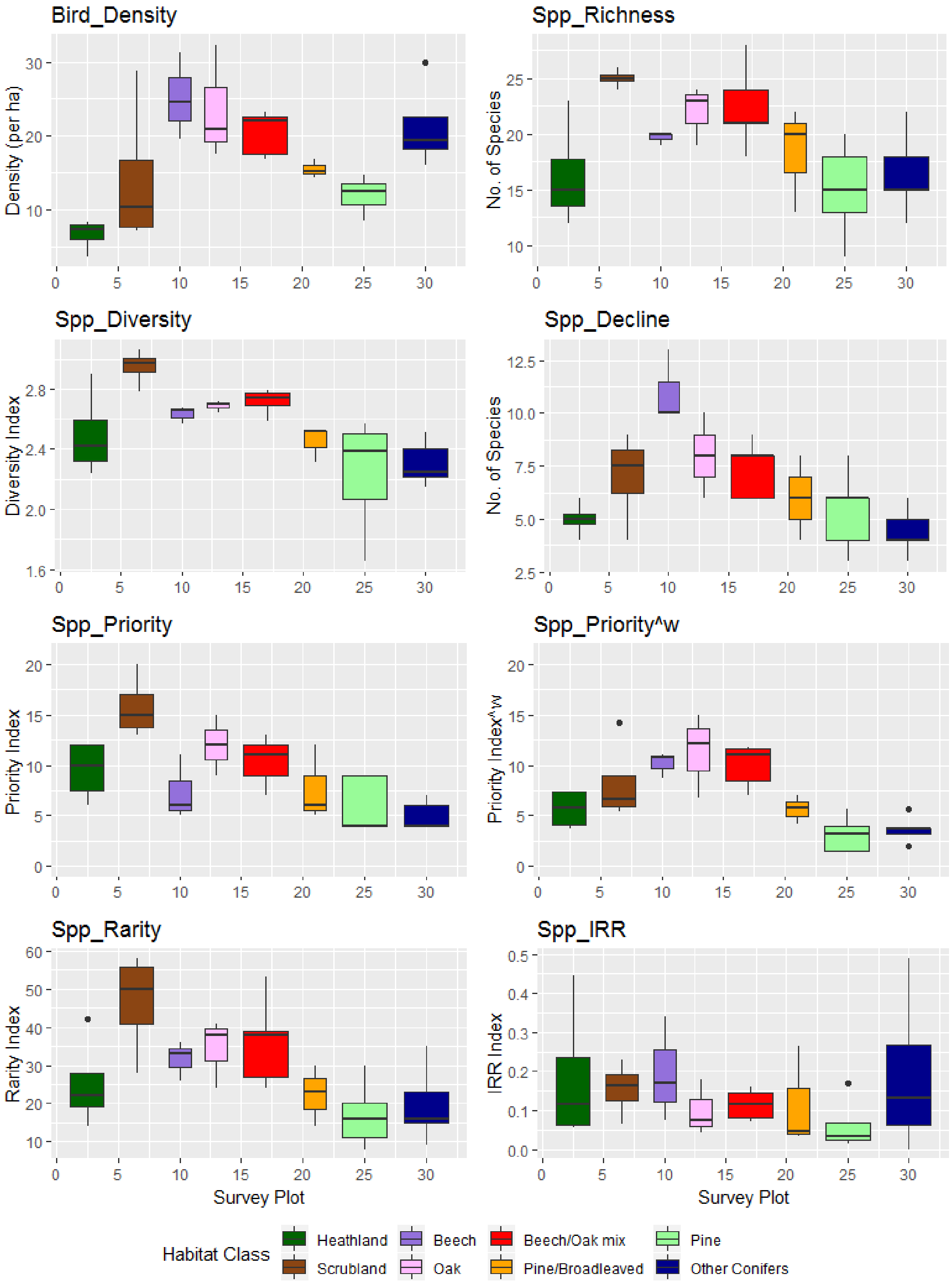


Figure 4.5. Variation in the bird indices between habitat classes in the New Forest. See Section 3.3.6 and Table 3.6 for derivation of bird indices (R version 3.5.2).

4.4.4 Bird-Habitat Relationships

The univariate correlations in Table 4.1 show that most of the bird indices, excluding *Bird_Density*, were significantly negatively correlated (at P <0.05; N = 32) with *PCov_Pine* and/or *PCov_ConsOther* (*r* between -0.35 and -0.63). *Bird_Density, Spp_Decline* and *Spp_Priority*^{w} were positively correlated with *PCov_Beech*, *PCov_Oak* and *Pres_Holly* (*r* between 0.41 and 0.66). As the latter bird index is dependent on *Bird_Density*, this is not surprising. This reflects the fact that *PCov_Beech* and *PCov_Oak* were correlated with *Pres_Holly* (Table S 4.12), as holly predominantly occurred in beech and oak woodlands. *PCov_Beech* was also positively correlated with *Ht_Av* and *Ostorey_Ht*, suggesting that beech woodlands were tall, and thus as *PCov_Beech* increases so would *Ht_Av* (Table S 4.12). Not surprisingly, therefore, *Ht_Av* and *Ostorey_Ht* were also positively correlated with *Bird_Density* (*r* = 0.58 & 0.50; P = 0.001 & 0.003, respectively). Conversely, *Ht_Av* was negatively correlated with *Spp_Diversity* (*r* = -0.36), and both *Ht_Av* and *Ostorey_Ht* were negatively correlated with *Spp_Priority* (*r* = -0.53 & -0.45; P = 0.002 & 0.011, respectively), indicating that tall vegetation does not necessarily result in more bird species as, for example, the vegetation could be beech or pine (both correlated with *Ht_Av*; Table S 4.12).

Ostorey_PenDepth was negatively correlated with *Spp_Diversity* (*r* = -0.44; P = 0.012), *Spp_Priority* (*r* = -0.49; P = 0.005), *Spp_Rarity* (*r* = -0.38; P = 0.032) and *Spp_IRR* (*r* = -0.41; P = 0.019), indicating that an open canopy is detrimental to bird diversity and the abundance of priority and rare species (Table 4.1). *PCov_<0.5m* was negatively correlated with *Bird_Density*, and positively correlated with *Spp_Priority*, with the opposite being true of the *PCov_>5m* (Table 4.1), as a result of *PCov_<0.5m* being negatively correlated with *PCov_>5m* (Table S 4.12). *PCov_Heather* was positively correlated with *Pres_Willow* (and to *PCov_<0.5m*, but negatively correlated with *PCov_>5m*; see Table S 4.12), therefore, as expected these two variables showed a negative relationship with *Bird_Density* (*r* = -0.57 & -0.36; P = 0.001 & 0.041, respectively) and a positive relationship with *Spp_Priority* (*r* = 0.39 for both; P = 0.028 & 0.030, respectively). This suggests that although *Bird_Density* was lower with increased vegetation cover at < 0.5 m (and decreased vegetation cover at >5 m), possibly due to reduced niche availability, *Spp_Priority* favours lower vegetation indicating the importance of open habitats, such as ‘Scrubland’, for conservation priority bird species. This is supported by Figure 4.5 and Table 4.1 which show that *Spp_Priority* was highest in the ‘Scrubland’ with high *PCov_<0.5m* and *PCov_Heather* (Figure 4.1 and Figure 4.2). *Ht_VDR* was also negatively correlated with *Bird_Density* (*r* = -0.53; P = 0.002) (possibly as a result of the high *Ht_VDR* in the ‘Heathland’), but was positively correlated with *Spp_Diversity* (*r* = 0.39; P = 0.025), *Spp_Priority* (*r* = 0.52; P = 0.002) and *Spp_Rarity* (*r* = 0.35; P = 0.049). This indicates the value of varied vegetation structure typical of successional ‘Scrubland’ habitat for increasing bird diversity, priority and rarity. Similarly to *PCov_>5m*, *PCov_2-5m* was positively correlated with *Bird_Density* (*r* = 0.57; P = 0.001), and also with *Spp_Richness* (*r* = 0.36; P = 0.043) and *Spp_Priority*^{w} (*r* = 0.51; P = 0.003), suggesting that shrubby vegetation at 2-5 m, either under the canopy or out in the ‘Scrubland’, increases the number of birds and species present (Table 4.1).

Table 4.1 shows that *Spp_IRR* was only significantly negatively correlated with *Ostorey_PenDepth* and *PCov_Pine* (*r* = -0.41 & -0.39; P = 0.019 & 0.029, respectively). It is worth noting, however, that *Pres_Birch* had a P value of 0.050 (*r* = -0.35) and the *r* values for *PCov_ConsOther* and *PCov_Heather* were positive, albeit non-significant, in the correlation matrix (Table 4.1). This could suggest that the extreme opposing values within these habitats, displayed in Figure 4.4, may be masking any apparent univariate statistical relationships. Further to this, the significant relationships presented in Figure S 4.1 often have low R^2 values, indicating that the fit of the linear regression and simple univariate correlation may not best represent the data. The results also suggest that bird-habitat relationships are more complex than simple univariate correlations, as such, a multi-variate generalised linear modelling approach incorporating both vegetation structure and composition may be more relevant.

Table 4.1. Correlation matrix of the Pearson's r correlation coefficients between the bird indices and habitat structural and composition variables, significant relationships (P <0.05) are in bold. P values are in brackets.

Variables	***Bird_Density***	***Spp_Richness***	***Spp_Diversity***	***Spp_Decline***	***Spp_Priority***	***Spp_Priority***^{w}	***Spp_Rarity***	***Spp_IRR***
PCov_Pine	-0.239 (0.188)	**-0.483 (0.005)**	**-0.629 (0.000)**	-0.260 (0.151)	**-0.446 (0.011)**	**-0.539 (0.001)**	**-0.520 (0.002)**	**-0.387 (0.029)**
PCov_ConsOther	0.203 (0.265)	-0.279 (0.122)	**-0.352 (0.048)**	**-0.385 (0.029)**	**-0.424 (0.016)**	**-0.390 (0.027)**	-0.307 (0.088)	0.231 (0.203)
PCov_Beech	**0.414 (0.018)**	0.247 (0.172)	0.251 (0.166)	**0.603 (0.000)**	-0.028 (0.877)	**0.468 (0.007)**	0.244 (0.177)	0.014 (0.941)
PCov_Oak	**0.506 (0.003)**	0.321 (0.073)	0.334 (0.062)	**0.443 (0.011)**	0.241 (0.185)	**0.656 (0.000)**	0.277 (0.125)	-0.196 (0.282)
PCov_BroadOther	0.124 (0.497)	0.217 (0.233)	0.159 (0.384)	-0.051 (0.783)	0.237 (0.192)	0.306 (0.088)	-0.020 (0.913)	-0.026 (0.886)
PCov_Heather	**-0.566 (0.001)**	-0.038 (0.835)	0.146 (0.424)	-0.098 (0.593)	**0.389 (0.028)**	-0.161 (0.379)	0.161 (0.378)	0.336 (0.060)
PCov_Gorse	-0.194 (0.288)	-0.033 (0.856)	0.033 (0.858)	-0.129 (0.483)	0.102 (0.580)	0.025 (0.891)	-0.093 (0.614)	-0.054 (0.771)
Pres_Bracken	0.063 (0.733)	0.029 (0.874)	-0.080 (0.665)	-0.004 (0.981)	-0.090 (0.623)	0.037 (0.841)	-0.138 (0.451)	-0.003 (0.988)
Pres_Holly	**0.409 (0.020)**	0.226 (0.213)	0.150 (0.412)	**0.410 (0.020)**	0.025 (0.894)	**0.478 (0.006)**	0.113 (0.538)	0.096 (0.601)
Pres_Hawthorn	-0.186 (0.309)	0.100 (0.587)	-0.019 (0.916)	0.042 (0.820)	0.116 (0.528)	-0.019 (0.919)	0.119 (0.517)	0.084 (0.646)
Pres_Birch	0.223 (0.220)	0.038 (0.836)	-0.051 (0.782)	0.224 (0.219)	-0.249 (0.170)	-0.028 (0.878)	-0.185 (0.310)	-0.349 (0.050)
Pres_Willow	**-0.363 (0.041)**	0.187 (0.305)	0.256 (0.157)	-0.075 (0.685)	**0.385 (0.030)**	-0.006 (0.975)	0.223 (0.220)	0.088 (0.634)
Ht_Av	**0.576 (0.001)**	-0.215 (0.238)	**-0.364 (0.041)**	0.263 (0.146)	**-0.530 (0.002)**	0.059 (0.747)	-0.305 (0.090)	-0.186 (0.308)
Ht_StDev	**0.579 (0.001)**	0.094 (0.609)	-0.036 (0.846)	0.132 (0.472)	-0.313 (0.081)	0.121 (0.509)	-0.098 (0.595)	-0.056 (0.762)
Ostorey_Ht	**0.504 (0.003)**	-0.128 (0.484)	-0.225 (0.216)	0.252 (0.165)	**-0.445 (0.011)**	0.031 (0.867)	-0.195 (0.284)	-0.120 (0.515)
Ostorey_PenDepth	0.262 (0.147)	-0.287 (0.112)	**-0.437 (0.012)**	0.066 (0.721)	**-0.485 (0.005)**	-0.143 (0.434)	**-0.381 (0.032)**	**-0.412 (0.019)**
Ht_VDR	**-0.532 (0.002)**	0.263 (0.146)	**0.395 (0.025)**	-0.193 (0.291)	**0.522 (0.002)**	-0.013 (0.946)	**0.351 (0.049)**	0.234 (0.196)
PCov_<0.5m	**-0.701 (0.000)**	0.067 (0.714)	0.267 (0.139)	-0.249 (0.169)	**0.471 (0.006)**	-0.166 (0.362)	0.244 (0.179)	0.209 (0.252)
PCov_0.5-2m	0.005 (0.977)	0.039 (0.833)	-0.041 (0.823)	-0.260 (0.151)	-0.181 (0.322)	-0.232 (0.201)	-0.129 (0.483)	-0.253 (0.163)
PCov_2-5m	**0.567 (0.001)**	**0.360 (0.043)**	0.176 (0.336)	0.240 (0.185)	0.005 (0.976)	**0.509 (0.003)**	0.169 (0.355)	0.048 (0.794)
PCov_>5m	**0.677 (0.000)**	-0.101 (0.582)	-0.286 (0.112)	0.260 (0.151)	**-0.471 (0.007)**	0.151 (0.409)	-0.254 (0.161)	-0.198 (0.278)

4.4.5 Multivariate Bird-Habitat Relationships

4.4.5.1 Density, Species Richness and Diversity

The top model representing *Bird_Density* had an R^2 value of 0.64, and a weight of 0.38, indicating that the model has a 38% chance of being the best, as such, the top set of models were used in model inferencing (Table S 4.14a). *Ht_VDR* and *PCov_2-5m* were the most important variables (with a relative importance of one) and also had the largest estimates and therefore effect sizes (Table 4.2a). The negative estimate of *Ht_VDR* indicated a negative relationship with *Bird_Density*, possibly as a result of the high *Ht_VDR* in the 'Heathland' and 'Scrubland' plots which have lower *Bird_Density*. *Pres_Willow* was also negatively related to *Bird_Density*, but to a lesser extent, possibly another indication that bird density is lower in the 'Scrubland' where there is often more willow (Table 4.2a). The positive relationship of *PCov_2-5m, PCov_Oak*, *PCov_Beech* and *Pres_Holly* to *Bird_Density* suggests that broadleaved woodlands with an understorey vegetation layer of holly result in higher bird abundance (Table 4.2a). *Spp_Richness* was also positively related to *PCov_2-5m* and *PCov_Oak* suggesting that as well as increasing bird abundance, this vegetation layer under an oak canopy also results in more bird species (Table 4.2b). *Spp_Richness* and *Spp_Diversity* had the same top model containing *PCov_Pine* and *PCov_ConsOther* with negative estimates for both (Table S 4.14b & c and Table 4.2b & c). *Spp_Diversity* was also positively related to *Ht_VDR* and *Pres_Willow*, as well as *PCov_Beech*, as a result of the high diversity in the 'Scrubland' plots, whereas the negative relationship with *PCov_<0.5m* is due to the low diversity in the 'Heathland' with the most open ground (Table 4.2c and Table S 4.14c). Overall, 'Scrubland' and broadleaved habitats enhance *Bird_Density, Spp_Richness* and *Spp_Diversity*, and 'Heathland', 'Pine' and 'Other Conifer' habitats are unfavourable for *Bird_Density, Spp_Richness* and *Spp_Diversity*.

4.4.5.2 Declining and Priority Species

The multi-modelling results in Table S 4.14d suggest that there were more declining bird species in tall beech woodlands, indicated by the top model for *Spp_Decline* containing only *PCov_Beech*, which was also the most important variable in the model set with a positive estimate (Table 4.2d), possibly as a result of national beech decline. The model averaged coefficients in Table 4.2d also showed a positive relationship of *Spp_Decline* to *PCov_Oak*. This suggests that declining bird species may also be influenced by oak decline nationally, possibly to a lesser extent than beech, or as a result of oak and beech often being present together in the New Forest. *Spp_Decline* was also negatively related to *PCov_Pine*, *PCov_ConsOther* and *PCov_0.5-2m*, as a result of the higher percentage shrub layer in the pine plots with low *Spp_Decline.*

Spp_Priority was positively related to *Ht_VDR, Pres_Willow* and *PCov_Heather* suggesting that ‘Scrubland’ and ‘Heathland’ are valuable for priority bird species (Table 4.2e). In contrast, *Spp_Priority*^{w} was positively related to *PCov_Oak, PCov_Beech* and *Pres_Holly* suggesting that *Bird_Density* may be skewing the weighted index. However, *Spp_Priority*^{w} was also positively related to *PCov_BroadOther* suggesting a higher abundance of priority bird species in habitats containing other broadleaved species, and also in ‘Scrubland’ (Table 4.2f). This could also suggest that there were numerous lower priority bird species with less of a population decline in the broadleaved habitat classes and fewer higher priority species in the ‘Scrubland’ and ‘Heathland’ classes.

4.4.5.3 Rarity

Similar to *Spp_Richness* and *Spp_Diversity*, the top model for *Spp_Rarity* also only contained *PCov_Pine* and *PCov_ConsOther*, with a negative estimate. The majority of the models in the model set to 6 AIC contained *PCov_Pine* and *PCov_ConsOther* and a combination of other variables with a positive estimate (Table 4.2g and Table S 4.14g). The positive relationship of *Spp_Rarity* to *PCov_Oak* and *PCov_Beech* indicates higher *Spp_Rarity* in the mixed broadleaved habitats, along with *Ht_VDR* and *PCov_<0.5m* represents higher *Spp_Rarity* in the ‘Scrubland’ plots. The model averaged coefficients for *Spp_IRR* in Table 4.2h reveal that the most important variable with the largest effect size and a negative estimate was *PCov_Pine*, followed by *Pres_Birch.* The positive estimates of *PCov_Heather* and *PCov_<0.5m*, indicate a positive relationship with ‘Heathland’ (Table 4.2h). However, when the outlier (plot 30) was removed the model set included *PCov_ConsOther* as the second most important variable with a positive estimate, after *PCov_Pine*, for this updated model set (Table 4.2i and Table S 4.14). Inference and averaging was made on the adjusted model set due to the higher R^2

and did not violate the assumptions of the model (Table S 4.14i). The positive relationship of *PCov_Heather* and *PCov_ConsOther* to *Spp_IRR* in the adjusted model indicates the importance of ‘Heathland’ and ‘Other Conifer’ habitats in the New Forest for rare specialist birds (Table 4.2i). The negative relationship with *PCov_Pine* and *Pres_Birch* suggests a detrimental effect of pine and scrubland on *Spp_IRR* (Table S 4.14i). *Pres_Birch* was not significant (P = 0.05) in the univariate correlation analysis (Table 4.1), but was significant in the model averaging (Table 4.2 and Table S 4.14), supporting the use of multi-variate analysis for ecological data and the importance of alternative habitats in the New Forest.

Models greater than 2AIC are weaker with lower R^2 and weights suggesting that models to 2 AIC be used in any subsequent analysis.

Table 4.2. Relative importance and model averaged coefficients (across the 95% confidence set) of standardised predictors for bird indices in relation to habitat composition and structure, based on the unconditional model average from the top two AICc generalised linear models. See Table 3.1 and Table 3.3 for parameter explanations, and Table 3.6 for bird indices. (Negative coefficients in italics. Rescale, z and c = standardised variables).

a) Bird_Density

Predictor	**N Containing Models**	**Relative Importance**	**Estimate**	**Adjusted SE**	**Lower CI**	**Upper CI**
(Intercept)	-	-	16.854	1.064	14.769	18.939
z.Ht_VDR	5	1	-7.031	1.500	-9.970	-4.092
z.PCov_2-5m	5	1	7.245	2.027	3.271	11.218
z.PCov_Oak	1	0.30	1.082	2.211	-1.660	8.860
c.Pres_Willow	1	0.13	-0.113	0.515	-3.221	1.420
c.Pres_Holly	1	0.10	0.095	0.850	-4.014	5.925
z.PCov_Beech	1	0.09	0.029	0.899	-5.431	6.045

b) Spp_Richness

Predictor	**N Containing Models**	**Relative Importance**	**Estimate**	**Adjusted SE**	**Lower CI**	**Upper CI**
(Intercept)	-	-	19.188	0.774	17.603	20.772
z.PCov_Pine	7	0.98	-4.599	1.659	-7.823	-1.572
z.PCov_Cons Other	4	0.64	-2.038	1.940	-6.255	-0.073
z.PCov_2-5m	4	0.38	0.907	1.573	-1.142	5.912
z.PCov_Oak	3	0.20	0.249	0.973	-2.564	5.035

c) Spp_Diversity

Predictor	**N Containing Models**	**Relative Importance**	**Estimate**	**Adjusted SE**	**Lower CI**	**Upper CI**
(Intercept)	-	-	2.543	0.038	2.469	2.617
z.PCov_ConsOther	6	0.96	-0.242	0.089	-0.399	-0.107
z.PCov_Pine	6	0.96	-0.406	0.112	-0.567	-0.282
z.Ht_VDR	2	0.17	0.028	0.104	-0.237	0.565
z.PCov_Beech	2	0.16	0.007	0.072	-0.299	0.383
z.PCov_Oak	2	0.14	0.010	0.063	-0.223	0.368
c.Pres_Willow	1	0.10	0.002	0.035	-0.186	0.235
z.PCov_<0.5m	1	0.10	-0.001	0.028	-0.184	0.164

d) Spp_Decline

Predictor	**N Containing Models**	**Relative Importance**	**Estimate**	**Adjusted SE**	**Lower CI**	**Upper CI**
(Intercept)	-	-	1.853	0.074	1.709	1.998
z.PCov_Beech	11	0.86	0.300	0.181	0.069	0.631
z.PCov_Oak	10	0.39	0.082	0.139	-0.088	0.507
z.PCov_ConsOther	10	0.40	-0.099	0.169	-0.614	0.123
z.PCov_0.5-2m	7	0.21	-0.022	0.091	-0.447	0.240
z.PCov_Pine	7	0.20	-0.019	0.089	-0.443	0.254

e) Spp_Priority

Predictor	**N Containing Models**	**Relative Importance**	**Estimate**	**Adjusted SE**	**Lower CI**	**Upper CI**
(Intercept)	-	-	9.031	0.555	7.943	10.119
z.PCov_ConsOther	6	1	-3.561	0.969	-5.460	-1.661
z.PCov_Pine	5	0.98	-3.452	1.167	-5.618	-1.425
z.Ht_VDR	2	0.34	0.897	1.503	-0.100	5.435
z.PCov_>5m	1	0.24	-0.565	1.206	-5.018	0.294
c.Pres_Willow	1	0.14	0.315	1.006	-1.059	5.586
z.PCov_Heather	1	0.11	0.174	0.680	-1.099	4.396

f) Spp_Priority^{w}

Predictor	N Containing Models	Relative Importance	Estimate	Adjusted SE	Lower CI	Upper CI
(Intercept)	-	-	1.827	0.064	1.701	1.953
z.PCov_Pine	8	0.98	-0.647	0.187	-0.980	-0.341
z.PCov_Oak	5	0.78	0.384	0.254	0.147	0.831
z.PCov_ConsOther	5	0.58	-0.273	0.261	-0.776	-0.169
z.PCov_BroadOther	3	0.40	0.136	0.189	0.065	0.618
z.PCov_2-5m	2	0.15	0.044	0.118	0.005	0.569
z.PCov_Beech	1	0.02	0.009	0.063	0.131	0.698
c.Pres_Holly	1	0.02	0.003	0.031	-0.130	0.460

g) Spp_Rarity

Predictor	N Containing Models	Relative Importance	Estimate	Adjusted SE	Lower CI	Upper CI
(Intercept)	-	-	28.627	1.858	24.985	32.268
rescale(PCov_Pine)	7	0.90	-8.980	4.439	-16.227	-4.327
rescale(PCov_ConsOther)	6	0.87	-13.446	5.186	-20.317	-9.569
rescale(PCov_Oak)	3	0.20	0.674	3.152	-9.120	15.900
rescale(PCov_<0.5m)	1	0.20	0.085	1.449	-8.158	9.944
rescale(PCov_Beech)	3	0.19	1.166	4.164	-9.208	21.219
rescale(Ht_VDR)	2	0.18	1.629	5.174	-8.720	26.682
rescale(Ht_Av)	2	0.12	-0.536	3.233	-20.766	11.882

h) Spp_IRR with outlier

Predictor	N Containing Models	Relative Importance	Estimate	SE	Lower CI	Upper CI
(Intercept)	-	-	-1.125	0.084	-1.289	-0.962
(phi)	-	-	10.349	2.711	5.034	15.663
rescale(PCov_Pine)	7	0.98	-0.481	0.169	-0.798	-0.181
rescale(Pres_Birch)	3	0.38	-0.163	0.274	-0.993	0.137
rescale(PCov_Heather)	4	0.32	0.060	0.124	-0.117	0.490
rescale(PCov_ConsOther)	3	0.19	-0.011	0.070	-0.354	0.243

i) Spp_IRR (outlier removed)

Predictor	**N Containing Models**	**Relative Importance**	**Estimate**	**SE**	**Lower CI**	**Upper CI**
(Intercept)	-	-	-1.088	0.075	-1.235	-0.942
(phi)	-	-	13.528	3.670	6.335	20.721
rescale(PCov_Pine)	7	0.82	-0.305	0.195	-0.662	-0.078
rescale(PCov_ConsOther)	7	0.45	0.108	0.153	-0.038	0.518
rescale(Pres_Birch)	7	0.45	-0.208	0.291	-0.982	0.055
rescale(PCov_Heather)	7	0.35	0.072	0.132	-0.080	0.496

4.5 Discussion

The New Forest National Park is renowned for its unique habitats predominantly caused by intensive grazing by the verderers' livestock and ponies, and the lowland heath. Some areas of the forest are also in "Inclosure Woodlands" which are excluded from grazing along with a number of ornamental conifer plots, as well as ancient broadleaved woodlands, which vary in grazing intensity. The plots surveyed in the New Forest varied in vegetation composition and structure, and consequently in the bird indices. 'Scrubland' was overall good habitat with the highest values of species richness and diversity, regardless of the low bird density, and also had the highest values of priority and rarity. The importance of 'Scrubland' in the New Forest is reinforced by positive relationships with variables such as height VDR (indicating a variable vegetation structure as is typical of a successional habitat), the presence of willow and percent cover < 0.5 m. By contrast, 'Heathland' is a specialist habitat with relatively low diversity and the lowest bird density of the habitat classes, but has the second highest value of the Index of Relative Rarity (IRR), most likely due to the Dartford Warbler.

'Pine' was overall poor bird habitat with negative relationships of the percent cover of pine with the majority of the bird indices, except bird density. However, whilst the 'Pine' plots had lower bird diversity and density, the 'Other Conifer' plots had a similar bird density to the broadleaved plots but lower bird diversity (although not as low as the 'Pine' plots). Additionally, the 'Other Conifer' habitats, containing species such as Douglas fir, had the highest Index of Relative Rarity value presumably as a consequence of the rare Firecrest.

As expected, the broadleaved plots (‘Oak’ and ‘Beech’) had the highest values of bird density, and bird diversity was higher in broadleaved woodlands than the conifer woodlands, in line with the findings of Adams and Edington (1973). Furthermore, the ‘Beech’ plots also supported the highest number of declining bird species, confirmed by the positive relationship with the percent cover of beech, indicating an effect of national beech decline on local bird population trends.

The conservation of rare and declining bird species requires the landscape to contain a range of habitats, because many such bird species are specialists to a particular habitat. These habitats are often themselves rare which should also be conserved, possibly more so than areas with high diversity. Simply managing habitats for biodiversity (alpha diversity) would neglect rare and declining specialist bird species, indicating that conservation and management should aim to increase gamma diversity over a landscape scale.

4.5.1 High Bird Diversity in Scrubland

The ‘Scrubland’ habitats had not only the highest values of bird diversity but also the highest values of the majority of the bird indices, with the exception of the number of declining species and the Index of Relative Rarity (IRR). Although most of the ‘Scrubland’ and all of the ‘Heathland’ plots had a lower density of birds, this did not equate to the number of bird species in the plots. ‘Scrubland’ had the highest bird species richness, with one exception, and the highest bird diversity of all the habitat plots in the New Forest, in line with Tews et al. (2004). The current study shows that bird diversity is indeed related to habitat diversity in the New Forest, as the ‘Scrubland’ habitat comprised of a number of vegetation types resulting in the high bird diversity and species richness (Wiens and Rotenberry 1981, Poulsen 2002). As well as vegetation composition, vegetation structure also varied within and between the ‘Scrubland’ plots further increasing bird diversity, supporting the findings of MacArthur et al. (1962), MacArthur & MacArthur (1961) and Tews et al. (2004). This result shows that the number of birds is not directly relatable to the number of bird species present, but both are related to the habitat, contradicting the findings of Seoane et al. (2017). As well as bird diversity, ‘Scrubland’ was also beneficial for species richness, priority and rarity, indicating the importance of this habitat for birds in the New Forest, thus management to preserve this habitat is essential.

The relationship of bird species richness to the percent cover of vegetation at 2-5 m demonstrates that complexity and variety in vegetation structure enhance bird species richness, allowing more vegetation species to exist in a successional habitat (such as ‘Scrubland’ and possibly ‘Heathland’) before trees establish (MacArthur et al. 1962, Poulsen 2002). Many studies have suggested that high bird diversity and species richness in scrub environments are a result of the vegetation species and bird habitat preference rather than vegetation structure (e.g. Bazzaz 1975, Estades 1997), whereas, the current study shows that the structure of the habitat is indeed important and is caused by multiple species of vegetation (Wiens and Rotenberry 1981). Seoane et al. (2017) stated that the structural complexity of the vegetation increases the number of niches and results in niche packing, and that this increases bird species richness through the apparent increased bird abundance. However, the current study suggests that vegetation structure is associated with bird species richness, rather than bird abundance. Fuller and Robles (2018) argued that if structural complexity increases bird diversity (MacArthur and MacArthur 1961), then forests of continuous cover would contain the highest diversity. However, the current study shows that bird diversity was highest in the ‘Scrubland’ habitats, supporting both structural complexity and the importance of successional habitats. Furthermore, species priority was also high in the ‘Scrubland’ plots and was positively related to height VDR, percent cover of heather and presence of willow in the multi-model analysis. This suggests that ‘Scrubland’ and also ‘Heathland’ are valuable for conservation priority species as a result of the varied vegetation types and consequently structure, increasing the priority bird species, such as the Willow Warbler.

The bird species present in these habitats which drive the priority index includes the Stonechat (*Saxicola rubicola*), whose population has declined by 41% in England over the ten years to 2015 (Massimino et al. 2017). Stonechats are prevalent in ground vegetation in ‘Scrubland’ and ‘Heathland’ areas. The Spotted Flycatcher has also seen a 41% decline, however, they are more prevalent in shrubby vegetation in open areas of woodland (Svensson et al. 1999, Massimino et al. 2017). A more scrub related species, the Greenfinch, has declined by 58% in England, most likely due to a protozoal disease, *Trichomonas gallinae* that is eradicating finch species, particularly the Greenfinch (Robinson et al. 2010, Massimino et al. 2017). Populations of the Lapwing (*Vanellus vanellus*) and Marsh Tit have seen a 31%

decline, and the Mistle Thrush has seen a slightly lower 29% decline in England in the ten years to 2015, adding to the priority species index in the ‘Scrubland’ plots (Massimino et al. 2017). The ‘Scrubland’ plots also contained the declining Grey Heron (28%, *Ardea cinerea*) and Willow Warbler (15%, Appendix A2) as a result of the wet areas in some of the plots (Massimino et al. 2017).

The percent decline of the Willow Warbler used in this study is a decline in England, but conversely the trend in Scotland has seen a slight increase (11%, 2006-2016) in their numbers (Morrison et al. 2013). The decline in England is possibly due to poorer breeding habitat condition, condition of the wintering grounds and/or the increased cost of migration (Morrison et al. 2013). Willow Warblers have been shown to prefer low mean vegetation height (3.7-5.3 metres) as either early successional or open canopy woodlands (Bellamy et al. 2009). This supports the positive relationship of height VDR and the presence of willow to the priority species index and implies that there may have been a reduction in scrubby areas in England for breeding Willow Warblers most likely due to woodland maturation (Fuller et al. 2005b, Bellamy et al. 2009).

It is worth noting that there are a number of surprising priority birds present within the 1-20% decline band, such as the Blue Tit, Great Tit, Blackbird and Chaffinch (Massimino et al. 2017). These birds are relatively more common than many of the higher priority species, and Blue Tits and Great Tits are generalists and very adaptable to the changing environment, and yet they have shown a decline in the ten years to 2015 (Cresswell and Mccleery 2003, Massimino et al. 2017). Hinsley et al. (2002) implied that the Great Tit might be more susceptible to cold, harsh winters than the Blue Tit, suggesting an effect of climate change on even the most adaptable of species (Massimino et al. 2017).

The ‘Scrubland’ plots also had the highest values of rarity, supported by the positive relationship with height VDR and the percent cover of vegetation < 0.5 m (Table 4.2g) further indicating the importance of the varied structure in the ‘Scrubland’. The high degree of correlation of rarity to bird species richness and diversity (Table S 4.13) resulted in ‘Scrubland’ plots 6 and 7 having the highest bird diversity, species richness and rarity. The multiple habitats that encompassed ‘Scrubland’ in the New Forest include: mature trees, such as oak; successional vegetation, such as silver birch; a woody shrub layer, and wet areas; which contributed to the high bird diversity and species richness and increased the number of rare and conservation priority bird species present in this important habitat. ‘Scrubland’ was at the interface between the woodland and open areas in the New Forest and supported a high diversity of species, including woodland and scrub preferring species, as well as generalist birds. Nevertheless, open habitat specialists and woodland interior specialist species were not supported by this habitat.

Not only did the ‘Scrubland’ plots surveyed in the New Forest have varied structure and vegetation type due to succession, but these plots also contained multiple habitats, including wet areas of bog and pond. This provides habitat for some wetland birds, such as the Mallard (*Anas platyrhynchos*), Grey Heron and Snipe (*Gallinago gallinago*), further adding to and increasing bird diversity and species richness in the ‘Scrubland’ plots. This supports the notion that a diversity of habitats, even on a small scale, will enhance bird diversity and the current study show that increasing alpha and beta diversity is required to maintain gamma diversity over a landscape scale.

4.5.2 Declining Bird Species in Beech Woodlands

Declining bird species were most prevalent in the ‘Beech’ habitat, possibly as a result of beech decline nationally (Jung et al. 2005, Jung 2009). This was indicated by the positive relationship between number of declining bird species and percent cover of beech in the current study. Jung et al. (2005) and Jung (2009) have shown that beech decline has been due to *Phytophthora* diseases which reduce the health of the trees. Fuller et al. (2005b) stated that the defoliation and reduction in beech tree health could benefit some avian species from the increase in defoliating Lepidopteran larval food, increased insect food, increased dead wood benefiting woodpeckers, and increased seed production, providing food for birds, at least in the short term. However, this short-term benefit may simply have expired in the New Forest. Beech tree death results in leaf loss and the lack of tree crown, which usually provides nests and foraging birds with protection from predation, and can result in overall reduction in food availability (Zang 1990, in Fuller et al. 2005b, Hake 1991).

The ‘Beech’ plots also had high bird density, in the most part, but were often characterised by having very little understorey. However, height VDR did not allude to a relationship between bird density and varied vegetation structure, which may be caused by another factor not included in the analysis, such as the level of grazing (Fuller and Gill 2001, Morecroft et al. 2001). The plots with the highest bird density (5, 10, 12 and 28) all had little to no grazing, allowing growth of understorey vegetation and shrub layer. However, Donald et al. (1998) found that grazed and ungrazed stands did not differ in bird abundance or species richness, but did state that grazing significantly affected vegetation structure with smaller broadleaved vegetation able to survive in the ungrazed stands. In contrast, Nelson et al. (2011) found that total exclusion of grazing and seasonal grazing influenced bird abundance, but species richness remained the same. Martin and Possingham (2005) showed that the foraging height preference of bird species indicated susceptibility to grazing and showed a decrease of most bird species. In addition, Donald et al. (1998) showed that migrant birds were more abundant in ungrazed stands, but suggested that their result may have been caused by having fewer species that preferred grazing (such as the Redstart which was present in the current study), as extremely high proportions of migrants were found in grazed woodlands in other studies (Fuller and Crick 1992).

‘Beech’ plot 10 also contained a high proportion of deadwood, which has been shown to increase bird species presence (Cadieux and Drapeau 2017, Ram et al. 2017), and is possibly increasing bird density in this New Forest habitat. Cadieux and Drapeau (2017) showed that old mixed forests were highly important to deadwood bird species and should therefore be a high conservation priority. Variation in vegetation structure in the woodlands would also increase the variety and number of niches available, allowing more birds to inhabit these areas possibly leading to higher bird density (Ram et al. 2017).

Birds which preferentially feed on oak canopies rather than ash, beech or hornbeam were shown not to be in decline in Fuller et al. (2005b). However, these species included Great Tits and Blue Tits which, in the current study, contrary to Fuller et al. (2005b), have been in decline within the ten years to 2015 (Appendix A2). This may suggest that a national decline in oak (Denman et al. 2010), could also be affecting bird species locally, and is supported by the positive relationship of the percent cover of oak with the number of declining species. This could also be a consequence of beech and oak often present together in the broadleaved woodlands, however, Table S 4.12 showed that the percentage of these tree species were not correlated. Denman et al. (2010) showed that oak has been declining over the past century, and furthermore a new disease called Acute Oak Decline (AOD) is causing concern due to rapidly developing symptoms and high tree mortality. AOD occurs in Europe and has been attributed to abiotic stressors weakening the trees and allowing a European bark-boring beetle, *Agrilus biguttatus,* to lay eggs in the bark of oak trees (Denman et al. 2010, Brown et al. 2014). This damage results in patches of dark sticky fluid excreting from cracks in the bark, with necrotic material often near the damage from the beetle larvae (Denman et al. 2010, 2014). It is unclear whether the beetle causes the bacterial infection or if the infection creates opportunity for the beetle to infest the trees (Brown et al. 2014). A number of pathogenic bacteria have been shown to affect oak tree health in Europe including, *Brenneria spp.* and *Gibbsiella quercinecans* (Brady et al. 2010, Denman et al. 2012). Jung (2009) showed that in sandy-loamy to clayey sites *Phytophthora* species, which cause root-rot, are also strongly involved in oak decline.

Emergence of the trichomonosis disease affecting finches in the UK, has also resulted in population declines in the Chaffinch, but to a lesser extent than the Greenfinch (Robinson et al. 2010). Robinson et al. (2010) stated that the UK Chaffinch population declined by 18% in the ten years to 2015, compared with a 58% decline in the Greenfinch. These more common priority species could be the explanation for the weighted priority index being skewed by high bird density in the broadleaved plots. The index weighted the level of conservation priority from the percent decline in the population in England over the ten years with bird density in the plot, which resulted in broadleaved plots of ‘Oak’ and ‘Beech’ containing priority species that were declining less, but which were more abundant. By contrast, the ‘Heathland’ and the majority of the ‘Scrubland’ plots contained fewer higher conservation priority bird species which have suffered more of a decline in the population in England. Furthermore, Inger et al. (2015) reported that common birds were declining rapidly, while less abundant birds were increasing in abundance and biomass. They suggested that declines in common birds were causing the decline in biodiversity and overall abundance and biomass, and that the common birds were of greater importance in terms of ecosystem function and service provision (Inger et al. 2015).

4.5.3 Pine is Poor Bird Habitat

In the New Forest ‘Pine’ was generally poor bird habitat and the percent cover of pine was negatively related with the majority of the bird indices, except bird density. Moreover, the ‘Pine’ plots tended to have a much lower diversity than the ‘Other Conifer’ plots. Wesołowski et al. (2002) stated that conifer woodlands would have the same avifauna as broadleaved woodlands but at a lower bird density. However, the current study shows that the ‘Pine’ plots had a lower bird diversity and density, whilst the ‘Other Conifer’ plots had a similar bird density to the broadleaved plots but lower bird diversity (although not as low as the ‘Pine’ plots).

The study by Du Bus De Warnaffe and Deconchat (2008) showed that bird assemblages were more effected by the dominant tree species (conifer or beech) rather than silviculture practise and the only differences were attributable to the first stages of clear-cutting. The current study showed that vegetation type is consequently important in respect to the negative effect of the percent cover of pine and other conifers to bird species richness and diversity, contradicting the findings of Wesołowski et al. (2002). Adams and Edington (1973) hypothesised that conifer forests would be more impoverished than broadleaved woodlands, in terms of bird diversity, but should contain the same bird abundance, and Donald et al. (1998) also showed no difference in bird abundance or species richness in stands of similar ages, contradicting Wesołowski et al. (2002). Whilst this is the case for the ‘Other Conifer’ habitats, the ‘Pine’ habitats differ in both bird diversity and density from the broadleaved plots, suggesting that the type of conifer is significant to the birds. Moreover, Wesołowski et al. (2002) found that as a consequence of habitat change diversifying the structure in their study pine forests, species richness and abundance increased, thus the varied structure of the ‘Other Conifer’ habitats in the current study could be increasing bird abundance, and species richness to some extent. Paquet et al. (2006) found lower species richness in conifer plantations over 15 years old and noted that this had been shown in Norway spruce plantations by Baguette et al. (1994), as the trees were harvested at around 60 years old, reducing nesting cavities and deadwood (Fuller and Robles 2018). This suggests an effect of woodland age on bird species richness and also abundance (Donald et al. 1998), which could be reflected in the current study. Further to that, Elton (1935), Lack and Venables (1939, in Adams and Edington 1973) and more recently Lindbladh et al. (2019) found that the two woodland types differed in dominant bird species which may be the case in the current study, suggesting a requirement for bird community analysis.

4.5.4 Rare Birds in Heathland and Other Conifer

The Index of Relative Rarity (IRR) was developed by Leroy et al. (2012) to evaluate invertebrate species vulnerability, as they were poorly represented and rarely studied for red lists. Much of the work following Leroy et al. (2012) has been on invertebrate rarity analysis (spiders in Leroy et al. (2013), hoverflies in Miličić et al. (2017)). In the current study, IRR was used to evaluate rare bird species in the New Forest based on UK population estimates. The IRR function calculates a relevant cut-off point of population size from the species present in the landscape, then ranks the rare species (with a population size below this cut-off) exponentially, and common species (with a population size above the cut-off) are given a zero weighting (Leroy et al. 2013). IRR uses both the local bird abundance and the population size weighting applied to each species to calculate a more intuitive index than the rarity index, only accounting for the very rare species, whereas the rarity index ranks most bird species below 1 million breeding pairs/territories. The rarity index could be seen as evaluating the extent of rarity in the birds; however, this index masked the truly rare species with the numerous not so rare bird species. Rarity was high in the 'Scrubland', further supporting the importance of 'Scrubland' to increase bird diversity by providing different habitat niches (Seoane et al. 2017), however, IRR was low. In hindsight the rarity index should have had a lower cut-off point of population size to enhance the value of very rare species in the UK. As such, IRR provided a more accurate and less biased index with rarer bird species having a considerably higher weighting, increasing the value of IRR in the habitat plots in the current study (Leroy et al. 2012).

The multi-model averaging results showed that IRR was positively related to the percent cover of other conifers and heather. The Firecrest, as expected, appeared to be driving the high IRR value in the 'Other Conifer' plots as it had the lowest population size (246 pairs, Eaton et al. 2009), and therefore the highest rarity weighting of 1 in the New Forest (Appendix A2). The negative relationship of IRR to pine indicates a preference for other conifer tree species, particularly Douglas fir. Four out of the five 'Other Conifer' plots and only one 'Pine' plot (22) had the Firecrest present, though the understorey of this 'Pine' plot comprised other conifer species, e.g. western hemlock (Table S 4.2). ‘Pine’ plot 22 also contained holly, suggesting that the Firecrest was utilising the holly (Batten 1973, Clements et al. 2017). ‘Other Conifer’ plot 30 had an IRR value of zero as the Firecrest was absent from this plot which comprised western hemlock and pine. This suggests that Firecrests were either deterred by the pine present in the plot or there was another structural factor influencing their absence, for example stand age or canopy openness. However, this plot was deemed a statistical outlier in the multi-model analysis, and once removed the relationship with percent cover of other conifer species was significant. Although, the Firecrest occurs in plot 31, which was also dominated by western hemlock, pine was not present and the structure differed, confirming the negative effect of pine on the Firecrest. This also suggests that the presence of holly, even in pine dominated plots, provides habitat for the Firecrest and that western hemlock may not be preferential for this bird species, possibly as a result of incorrect leaf morphology (Hanzelka and Reif 2016). The highest density of the Firecrest was in ‘Other Conifer’ plot 32 which consequently had the highest value of IRR. This plot was dominated by coastal redwood and Douglas fir, further supporting a preference for Douglas fir.

Both species of conifer are non-native suggesting that the Firecrest may follow Hanzelka and Reif (2016), who showed that leaf morphology affected the birds more than whether the tree is non-native or native. Douglas fir has been present in Britain since 1827, and although Scots pine is only native to Scotland, it is widely planted for timber in southern England (Woodland Trust 2019c). Batten (1973) found that Firecrests prefer Norway spruce, suggesting that in its absence Douglas fir will be sufficient due to similar leaf morphology. This is supported by the result of ‘Other Conifer’ plot 32 comprising alternative non-native coniferous vegetation, which may be contributing to an increase in the rarest bird in the New Forest. Hanzelka and Reif (2016) also suggested that increasing the heterogeneity of tree species would increase bird diversity in native forests, but a reduction in non-native woodlands would prevent the loss of specialist bird species. Caprio et al. (2009) noted that the introduction of non-native tree species could alter bird community composition and nesting success. The current study shows that the inclusion of non-native conifers has had a positive effect on the Firecrest in the New Forest, and ultimately vegetation species identity is more important to the Firecrest than vegetation structure (Kosicki et al. 2015). Moreover, the effect of the non-native vegetation could be buffered by the mosaic landscape of multiple vegetation types interspersed within the New Forest's large expanse of woodland.

Surprisingly, the Firecrest occurred in a number of the broadleaved plots, 9-11, 14, 16-18, and 'Scrubland' plot 5. The presence of the Firecrest in broadleaved vegetation containing holly in the New Forest was first noted by Batten (1973), indicating habitat suitability of holly in the broadleaved woodland areas in the current study (*pers. comm.* M. Ward 31st January 2016, *pers. comm.* R. Wynn, 25th October 2016, Ward and Wynn 2011, Clements et al. 2017). The population range of the Firecrest has expanded north of the New Forest, and recent population increases, possibly due to climate change, resulted in the species being moved from AMBER listed in the 2009 Birds of Conservation Concern 3 report to GREEN in the 2015 report (Eaton et al. 2009, 2015, Ward and Wynn 2011).

The Dartford Warbler is also considerably rare in the UK with 3200 pairs (Wotton et al. 2009) and only found on lowland heath in southern England, mostly confined to Dorset and Hampshire, at the northern limit of its range (Bibby and Tubbs 1975). However, the most recent Bird Atlas shows that the Dartford Warbler was breeding in north Norfolk (Balmer et al. 2013). In the 19th century, the Dartford Warbler population declined greatly, possibly as a result of habitat loss and fragmentation due to 70% of the lowland heath being replaced by agriculture, forestry and development (Gibbons and Wotton 1996). Since this period of rapid change development has slowed, but the Dartford Warbler is now more vulnerable to degrading habitat as a result of changing land-management practises that allow for scrub encroachment (Gibbons and Wotton 1996), represented by the negative relationship of IRR to the presence of birch in the current study. The current study also showed that IRR was positively related to the percent cover of heather, as a result of the Dartford Warbler being an open heath specialist. Dartford Warblers occupied a range of heather heights and were scarce in heavily grazed plots in the New Forest (Bibby and Tubbs 1975), supported by the negative relationship of IRR to the percent cover of vegetation < 0.5 m in the current study. However, Van den Berg et al. (2001) showed that the Dartford Warbler was positively associated with mature gorse, dry/humid heath and bare soil and less so with young gorse, indicating a more complex relationship with vegetation, and thus supports the inclusion of structural metrics. The 'Heathland' areas in the New Forest did not contain bare soil, therefore the negative relationship with vegetation of < 0.5 m could also be a result of preference for taller mature vegetation. Conversely, Bibby and Tubbs (1975) found that the Dartford Warbler avoided tall stands of gorse and it was the gradual incline from taller ground vegetation to gorse stands that created its optimal habitat, supporting the findings of the current study.

Further work on bird-habitat preference in open heathland should incorporate a detailed study of the heights of the ground vegetation, possibly using remote sensing techniques with a higher resolution, acquired with ground or drone laser equipment, for example. The Dartford Warbler was absent from ‘Heathland’ plot 2, which was boggy heathland, and plot 1, which was adjacent to a pine plantation. This was presumably a consequence of the Dartford Warbler being rarely associated with wet heath and pine (Van den Berg et al. 2001), further supporting the negative relationship of IRR to pine in the current study. Bibby and Tubbs (1975), however, noted that some Dartford Warbler territories in the New Forest were located with pine, and that previous studies found that the young pine trees in amongst the top of the heather were responsible for population increases in the 1930s (e.g. Bond 1939, in Bibby and Tubbs 1975). Although not quantified in the current study, ‘Heathland’ plot 3 contained young pine in amongst the heather where the Dartford Warbler occurred, supporting a tolerance for young pine saplings (*pers. obs.* A. Barnes). Habitat suitability of the Dartford Warbler is also directly affected by fire, including the burning of heather, which alters vegetation height and should be taken into account when deciding heathland management strategies in the New Forest (Regos et al. 2015). This signifies the complexity of bird-habitat relationships, and to conserve multiple bird species, multiple habitat types must be provided and managed efficiently across the landscape.

As well as the breeding habitat, wintering habitat is also vital to population survival. Wintering Dartford Warblers are susceptible to crashes following cold, hard winters (Bibby and Tubbs 1975, Gibbons and Wotton 1996). The Dartford Warbler population was increasing in the UK in the decades before 2005 due to a series of warm winters; however, the increase in more extreme annual climatic events proves difficult for estimating population trends (Fuller et al. 2005b). Gibbons and Wotton (1996) showed that the Dartford Warbler population in southern England increased to around 1600-1670 pairs in 1994, and the most recent population estimate showed an increase to 3200 pairs in 2006 (Wotton et al. 2009). In spite of this, the Dartford Warbler is AMBER listed in the UK and listed as Near Threatened in Europe by The

International Union for Conservation of Nature (IUCN) (Eaton et al. 2015, BirdLife International 2017). Dartford Warbler populations may have increased in the UK as a result of previous milder winters, nevertheless, the total European population is in decline and there is no recent UK population trend data. For the current study, the population trend of the Dartford Warbler was calculated from the average trend of AMBER listed birds as a 1% increase, and was therefore not regarded as a priority species in the current study, most likely resulting in an under-representation. This indicates that conservation of open heathland habitat is vital, not only in southern England but throughout the European range of the Dartford Warbler to thwart any further population declines (Regos et al. 2015).

4.6 Conclusion

These results ultimately reveal that measures of diversity should be combined with measures of declining species and/or priority and rarity to ensure adequate habitat provision. Diversity assumes all species are equal, whereas these measures demonstrate that in order to maximise gamma diversity, these rare and declining species, often in rare habitats, are equally, if not more, important to conserve. Thus, incorporating multiple habitat types into the landscape is essential for conserving and mediating population declines of conservation priority bird species and common species alike. This follows the research by Heikkinen et al. (2004), which stated that the ecological importance of habitats must also be taken into account, in this case for rare and declining bird species. Management efforts in the New Forest should focus on individual niches of multiple specialist bird species by providing particular habitats, such as heathland and soft conifers, whilst maintaining broadleaved and scrubland areas to increase bird diversity.

4.7 Supplementary Material

Table S 4.1. The vegetation cover in the 'Heathland' and 'Scrubland' plots in the New Forest. Information in brackets is % composition of woody vegetation (trees and shrubs). Missing data includes: saplings and infrastructure. Note that % cover vegetation can be more than 100% as shrub can be understorey. See Appendix B2 for variable values.

Plot No.	Habitat Class	%Cover Heather	%Cover Rough Grass	%Cover Gorse	%Cover Wood Vegetation	%Cover Other
1	**Heathland**	65	5	25	5 (pine, birch, willow, hawthorn)	0
2	**Heathland**	0	75	0	25 (willow 70, birch 25, pine 2)	0
3	**Heathland**	45	49	5	1 (silver birch)	0
4	**Heathland**	80	0	5	5 (hawthorn, apple)	10 (bracken)
5	**Scrubland**	0	15	15	48.8 (silver birch 50, oak 45, ash, holly and hawthorn 5)	20 (bracken)
6	**Scrubland**	0	73	0	26.5 (oak 100, hawthorn <1)	0
7	**Scrubland**	55	20	5	16.5 (silver birch 60, willow 30, pine and oak 10)	0
8	**Scrubland**	50	30	0	10.8 (silver birch 60, oak 25, pine 15, hawthorn <5, willow <5)	8 (bracken)

Table S 4.2. Vegetation in the woodland plots in the New Forest as % cover. Data in brackets are % composition. Missing data includes: saplings and infrastructure. Note total % cover >100% (Y=Young).

Plot No.	**Habitat Class**	**%Cover Ground Layer**	**%Cover Understorey**	**%Cover Overstorey**
9	**Beech**	22.2 (grass)	1.8 (Y beech)	81.5 (beech 70, oak 20, pine 8, birch 2)
10	**Beech**	11.5 (leaves/grass)	6.6 (holly)	87.7 (beech 65, oak 30, birch 5)
11	**Beech**	13.9 (bracken)	5.0 (holly)	83.5 (beech 70, oak 20, birch 5)
12	**Oak**	18.2 (leaves/grass)	2.6 (birch)	91.3 (oak 80, pine 10, beech 5, birch 5)
13	**Oak**	35.4 (grass, bracken)	2.2 (holly, Y beech, hawthorn, ash)	75.2 (oak 80, birch 10, beech 7, pine 2)
14	**Oak**	44.0 (bracken)	5.0 (holly, ash, hawthorn)	49.0 (oak 70, birch 20, beech 5)
15	**Beech/ Oak**	12.7 (bracken)	1.9 (holly, western hemlock, Douglas fir, birch)	91.2 (beech 55, oak 35, pine 5, birch 5)
16	**Beech/ Oak**	17.6 (bracken, bramble)	7.3 (Y beech, hawthorn, holly)	87.7 (beech 60, oak 30, birch 10)
17	**Beech/ Oak**	27.5 (bracken, grass, bramble)	6.1 (holly, Y beech)	78.9 (beech 50, oak 35, ash 18, birch 5, pine 2)
18	**Beech/ Oak**	22.9 (bracken, grass, wet grass)	2.3 (holly, hawthorn)	84.0 (beech 60, oak 35, pine, Douglas fir, spruce, western hemlock, rowan & birch 5)
19	**Beech/ Oak**	22.9 (bracken)	4.1 (holly, hawthorn, willow)	82.9 (beech 55, oak 40, pine 5, birch & willow 5)
20	**Pine/ Broad leaved**	34.1 (bracken)	2.5 (Y western red cedar, beech, Y western hemlock, hawthorn)	76.8 (pine 65, beech 20, oak 10, birch 10, ash 5)
21	**Pine/ Broad leaved**	19.2 (bracken, grass)	2.1 (Y pine, western hemlock)	94.1 (oak 50, pine 45, beech 5, birch, sycamore & rowan 5)
22	**Pine/ Broad leaved**	36.1 (bracken)	4.2 (western hemlock, willow, holly)	62.2 (pine 50, birch 30, willow 13, oak 2)
23	**Pine**	32.4 (bracken)	1.9 (oak, birch)	74.1 (pine 80, birch 15, oak 5)
24	**Pine**	28.7 (bracken)	2.4 (Y oak, beech, birch)	72.9 (pine 80, beech 10, oak 10)
25	**Pine**	30.6 (bracken)	1.9 (birch, hawthorn, holly)	81.2 (pine 80, beech 10, oak 5, birch 5)
26	**Pine**	43.8 (bracken)	0.3 (birch, rowan)	68.0 (pine 85, birch 10, larch 5)
27	**Pine**	23.5 (Juncus grass, bracken)	1.7 (birch, hawthorn, beech)	95.7 (pine 100)
28	**Other Conifer**	29.4 (bracken, grass)	3.1 (Y Douglas fir, birch, Y western hemlock)	67.5 (Douglas fir 70, pine 25, birch 5)
29	**Other Conifer**	37.0 (grass)	1.2 (Y beech, Y oak, Y rowan, birch, sycamore, holly, Y Douglas fir)	69.1 (Douglas fir 90, beech, oak & birch 10)
30	**Other Conifer**	9.9 (Juncus grass, grass)	6.1 (birch, sweet chestnut, Y western hemlock)	93.6 (western hemlock 50, pine 40, birch 5, sweet chestnut 5)
31	**Other Conifer**	23.8 (needles)	3.3 (Y western hemlock)	90.4 (western hemlock 75, Douglas fir 20, birch & sweet chestnut 10, oak & larch 5)
32	**Other Conifer**	21.1 (bracken, grass, needles)	4.5 (Y Douglas fir, Y coastal redwood, birch, hawthorn)	74.1 (coastal redwood 50, Douglas fir 30, pine 10, birch 5, oak & beech 5)

Table S 4.3. Presence and absence of other potentially ecologically important vegetation species in each survey plot in the New Forest. (1 = present, 0 = absent). See Table 3.1 for variable explanation.

Plot No.	**Classification**	***Pres_ Bracken***	***Pres_ Holly***	***Pres_ Hawthorn***	***Pres_ Birch***	***Pres_ Willow***
1	**Heathland**	0	0	1	1	1
2	**Heathland**	0	0	0	1	1
3	**Heathland**	0	0	0	1	0
4	**Heathland**	1	0	1	0	0
5	**Scrubland**	1	1	1	1	0
6	**Scrubland**	0	0	1	0	0
7	**Scrubland**	0	0	0	1	1
8	**Scrubland**	1	0	1	1	1
9	**Beech**	0	0	0	1	0
10	**Beech**	0	1	0	1	0
11	**Beech**	1	1	0	1	0
12	**Oak**	0	0	0	1	0
13	**Oak**	1	1	1	1	0
14	**Oak**	1	1	1	1	0
15	**Beech/Oak**	1	1	0	1	0
16	**Beech/Oak**	1	1	1	1	0
17	**Beech/Oak**	1	1	0	1	0
18	**Beech/Oak**	1	1	1	1	0
19	**Beech/Oak**	1	1	1	1	1
20	**Pine/Broadleaved**	1	0	1	1	0
21	**Pine/Broadleaved**	1	0	0	1	0
22	**Pine/Broadleaved**	1	1	0	1	1
23	**Pine**	1	0	0	1	0
24	**Pine**	1	0	0	1	0
25	**Pine**	1	1	1	1	0
26	**Pine**	1	0	0	1	0
27	**Pine**	1	0	1	1	0
28	**Other Conifers**	1	0	0	1	0
29	**Other Conifers**	0	1	0	1	0
30	**Other Conifers**	0	0	0	1	0
31	**Other Conifers**	0	0	0	1	0
32	**Other Conifers**	1	0	1	1	0

Table S 4.4. Percent Cover (PCov) of the vegetation composition variables in each plot in the New Forest used in the analysis. See Table 3.1 for variable explanation.

Plot No.	**Habitat Class**	***PCov_ Pine***	***PCov_ Cons Other***	***PCov_ Beech***	***PCov_ Oak***	***PCov_ Broad Other***	***PCov_ Heather***	***PCov_ Gorse***
1	**Heathland**	2.5	0	0	0	2.5	65	25
2	**Heathland**	1.25	0	0	0	23.75	0	0
3	**Heathland**	0	0	0	0	0	45	5
4	**Heathland**	0	0	0	0	5	80	5
5	**Scrubland**	0	0	0	21.98	26.9	0	15
6	**Scrubland**	0	0	0	26.47	0	0	0
7	**Scrubland**	0	0	0	1.65	14.87	55	5
8	**Scrubland**	1.62	0	0	2.7	6.48	50	0
9	**Beech**	6.5	0	57.1	16.3	1.6	0	0
10	**Beech**	0	0	57	26.3	4.4	0	0
11	**Beech**	0	0	61.5	17.6	4.4	0	0
12	**Oak**	9.1	0	4.6	73	4.6	0	0
13	**Oak**	1.5	0	5.3	60.8	7.6	0	0
14	**Oak**	0	0	2.6	36.1	10.3	0	0
15	**Beech/Oak**	4.6	0	50.1	31.9	4.6	0	0
16	**Beech/Oak**	0	0	52.6	26.3	8.8	0	0
17	**Beech/Oak**	1.4	0	35.9	25.1	16.5	0	0
18	**Beech/Oak**	1.4	1.4	50.4	29.4	1.4	0	0
19	**Beech/Oak**	3.9	0	43.6	31.5	3.9	0	0
20	**Pine/ Broadleaved**	45.4	0	13.96	6.98	10.47	0	0
21	**Pine/ Broadleaved**	40.3	0	4.5	44.8	4.5	0	0
22	**Pine/ Broadleaved**	32.7	0	0	1.3	28.2	0	0
23	**Pine**	59.3	0	0	3.71	11.1	0	0
24	**Pine**	58.3	0	7.3	7.3	0	0	0
25	**Pine**	64.9	0	8.1	4.1	4.1	0	0
26	**Pine**	57.8	3.4	0	0	6.8	0	0
27	**Pine**	95.7	0	0	0	0	0	0
28	**Other Conifers**	16.9	47.2	0	0	3.4	0	0
29	**Other Conifers**	0	62.2	2.3	2.3	2.3	0	0
30	**Other Conifers**	37.4	46.8	0	0	9.4	0	0
31	**Other Conifers**	0	78.1	0	0	12.3	0	0
32	**Other Conifers**	7.8	62.4	1.95	1.95	0	0	0

The results from the ANOVA showed a significant difference (at P <0.05) between the habitat classes for *PCov_Heather* (which occurred between the 'Heathland' class and each woodland class, see Table S 4.6), but not *PCov_Gorse*, which may have been expected to show the same trend, possibly as a consequence of a low amount present (Table S 4.5). It is also notable that there were no statistically significant differences in any of the vegetation composition variables between the ‘Heathland’ and ‘Scrubland’ plots, as a result of the overlap in these habitats (Table S 4.6). As would be expected, the percentage cover of pine, other conifer, oak and beech differ significantly across the eight different habitat classes, reflecting the different composition of each (Table S 4.5). However, *PCov_BroadOther* did not differ significantly across the eight habitat classes, indicating that one or more other broadleaved species (most notably silver birch) was present in many plots across all habitat types (Table S 4.1 and Table S 4.2).

Table S 4.5 also shows that *Pres_Holly* was statistically significantly different (R^2 = 0.44, P = 0.035), and the post-hoc Tukey test in Table S 4.6 showed that the difference occurred between the ‘Heathland’ plots (with no holly present) and the ‘Beech/Oak’ plots (with holly in every plot). All habitat classes had some bracken and silver birch present (Table S 4.3), therefore, unsurprisingly there was no statistically significant difference between the habitat classes. However, statistical significance would have been expected for *Pres_Hawthorn*, as it was absent in the ‘Beech’ plots, and *Pres_Willow*, as it was only present in some ‘Heathland’, ‘Scrubland’ and ‘Pine/Broadleaved’ plots, therefore suggesting that analysis should be carried out at the plot level rather than by habitat classification.

Table S 4.5. The ANOVA results of the vegetation composition variables between habitat classes in the New Forest. Significance at $P<0.05$. R^2 represents the amount of variance explained by the habitat classes. (PCov_ = Percent Cover; Pres_ = Presence).

Variable	**Df**	**Sum Sq**	**Mean Sq**	***f***	**R^2**	**P**
PCov_Pine	7	18642.00	2663.20	29.05	0.89	<0.001
PCov_ConsOther	7	14768.00	2109.70	73.93	0.96	<0.001
PCov_Oak	7	8876.00	1268.00	12.36	0.78	<0.001
PCov_Beech	7	14737.00	2105.30	132.40	0.97	<0.001
PCov_BroadOther	7	344.40	49.20	0.83	0.20	0.571
PCov_Heather	7	9062.00	1294.50	4.86	0.59	0.002
PCov_Gorse	7	311.70	44.53	2.06	0.38	0.088
Pres_Bracken	7	2.94	0.42	2.35	0.41	0.056
Pres_Holly	7	3.37	0.48	2.66	0.44	0.035
Pres_Hawthorn	7	1.59	0.23	0.87	0.20	0.545
Pres_Birch	7	0.38	0.05	0.86	0.20	0.553
Pres_Willow	7	1.41	0.20	1.39	0.29	0.254

Table S 4.6. The results from the post-hoc Tukey test for the significantly different vegetation composition variables in bold (at P<0.05), non-significant variables excluded.

Habitat Pairs/Variable	*PCov_ Pine*	*PCov_ ConsOther*	*PCov_ Oak*	*PCov_ Beech*	*PCov_ Heather*	*Pres_ Holly*
Beech/Oak-Beech	1.000	1.000	0.928	**0.008**	1.000	0.957
Heathland-Beech	1.000	1.000	0.206	**0.000**	**0.016**	0.472
Oak-Beech	1.000	1.000	0.004	**0.000**	1.000	1.000
Other Conifers-Beech	0.817	**0.000**	0.205	**0.000**	1.000	0.800
Pine-Beech	**0.000**	1.000	0.331	**0.000**	1.000	0.800
Pine/Broadleaved-Beech	**0.002**	1.000	1.000	**0.000**	1.000	0.976
Scrubland-Beech	1.000	1.000	0.984	**0.000**	0.439	0.897
Heathland-Beech/Oak	1.000	1.000	**0.006**	**0.000**	**0.005**	**0.033**
Oak-Beech/Oak	1.000	1.000	**0.018**	**0.000**	1.000	0.957
Other Conifers-Beech/Oak	0.700	**0.000**	**0.004**	**0.000**	1.000	0.102
Pine-Beech/Oak	**0.000**	1.000	**0.010**	**0.000**	1.000	0.102
Pine/Broadleaved-Beech/Oak	**0.000**	1.000	0.796	**0.000**	1.000	0.417
Scrubland-Beech/Oak	1.000	1.000	0.333	**0.000**	0.287	0.195
Oak-Heathland	1.000	1.000	**0.000**	0.863	**0.016**	0.472
Other Conifers-Heathland	0.634	**0.000**	1.000	1.000	**0.005**	0.996
Pine-Heathland	**0.000**	1.000	1.000	0.938	**0.005**	0.996
Pine/Broadleaved-Heathland	**0.000**	1.000	0.340	0.490	**0.016**	0.966
Scrubland-Heathland	1.000	1.000	0.599	1.000	0.600	0.989
Other Conifers-Oak	0.901	**0.000**	**0.000**	0.941	1.000	0.800
Pine-Oak	**0.000**	1.000	**0.000**	1.000	1.000	0.800
Pine/Broadleaved-Oak	**0.002**	1.000	**0.002**	0.998	1.000	0.976
Scrubland-Oak	1.000	1.000	**0.000**	0.863	0.439	0.897
Pine-Other Conifers	**0.000**	**0.000**	1.000	0.985	1.000	1.000
Pine/Broadleaved-Other Conifers	**0.014**	**0.000**	0.345	0.613	1.000	1.000
Scrubland-Other Conifers	0.582	**0.000**	0.615	1.000	0.287	1.000
Pine/Broadleaved-Pine	**0.011**	1.000	0.512	0.960	1.000	1.000
Scrubland-Pine	**0.000**	1.000	0.801	0.938	0.287	1.000
Scrubland-Pine/Broadleaved	**0.000**	1.000	0.999	0.490	0.439	1.000

With the exception of *PCov_0.5-2m* and *PCov_2-5m*, all the other structural variables were statistically significantly different (at P <0.05), mainly between the non-woody plots ('Heathland' and 'Scrubland') and the woodland plots (Table S 4.7 and Table S 4.8). However, *Ostorey_PenDepth* also differed significantly between the 'Pine' and 'Beech/Oak' and 'Other Conifer' classes, highlighting that the canopy in the 'Pine' plots was significantly more open. *Ht_StDev* differed only between the 'Heathland' plots and the woodland plots as a result of the low values in the 'Heathland' and also demonstrates that the 'Scrubland' was more variable across the plots (Table S 4.8 and Figure 4.2).

Table S 4.7. The results of the ANOVA for each of the vegetation structural variables as a function of habitat class in the New Forest. Significance at $P < 0.05$. R^2 represents the amount of variance explained by the habitat classes.

Variable	**Df**	**Sum Sq**	**Mean Sq**	***f***	**R^2**	**P**
Ht_Av	7	839.100	119.870	26.160	0.88	<0.001
Ht_StDev	7	134.230	19.175	7.926	0.70	<0.001
Ht_VDR	7	2.120	0.303	24.010	0.88	<0.001
Ostorey_Ht	7	297.800	42.540	12.640	0.79	<0.001
Ostorey_PenDepth	7	159.370	22.767	15.210	0.82	<0.001
PCov_<0.5m	7	21122.000	3017.400	24.520	0.88	<0.001
PCov_0.5-2m	7	63.230	9.033	2.260	0.40	0.064
PCov_2-5m	7	28.960	4.138	0.889	0.21	0.530
PCov_>5m	7	20192.000	2884.600	28.120	0.89	<0.001

Table S 4.8. The results of the post-hoc Tukey test showing the significantly different (at $P < 0.05$) structural variables between habitat class pairs, non-significant variables excluded. Significant P values in bold.

Habitat Pairs/Variable	***Ht_ Av***	***Ht_ StDev***	***Ht_ VDR***	***Ostorey_ Ht***	***Ostorey_ PenDepth***	***PCov_ <0.5m***	***PCov_ >5m***
Beech/Oak-Beech	0.784	0.998	0.979	0.722	1.000	0.999	0.996
Heathland-Beech	**0.000**	**0.000**	**0.000**	**0.000**	**0.001**	**0.000**	**0.000**
Oak-Beech	0.175	0.994	0.487	0.359	1.000	0.601	0.468
Other Conifers-Beech	0.489	1.000	0.829	0.833	0.989	0.964	0.919
Pine-Beech	0.833	1.000	0.977	0.992	0.110	0.522	0.314
Pine/Broadleaved-Beech	0.330	0.999	0.640	0.544	0.996	0.780	0.585
Scrubland-Beech	**0.000**	0.295	**0.000**	**0.002**	**0.021**	**0.000**	**0.000**
Heathland-Beech/Oak	**0.000**	**0.000**	**0.000**	**0.000**	**0.000**	**0.000**	**0.000**
Oak-Beech/Oak	0.814	1.000	0.880	0.986	1.000	0.823	0.749
Other Conifers-Beech/Oak	0.999	0.991	0.999	1.000	0.982	1.000	0.999
Pine-Beech/Oak	1.000	1.000	1.000	0.974	0.039	0.759	0.572
Pine/Broadleaved-Beech/Oak	0.961	1.000	0.961	0.999	0.990	0.946	0.857
Scrubland-Beech/Oak	**0.000**	0.490	**0.000**	**0.031**	**0.007**	**0.000**	**0.000**
Oak-Heathland	**0.000**	**0.003**	**0.000**	**0.002**	**0.002**	**0.000**	**0.000**
Other Conifers-Heathland	**0.000**	**0.000**	**0.000**	**0.000**	**0.001**	**0.000**	**0.000**
Pine-Heathland	**0.000**	**0.000**	**0.000**	**0.000**	**0.000**	**0.000**	**0.000**
Pine/Broadleaved-Heathland	**0.000**	**0.001**	**0.000**	**0.001**	**0.000**	**0.000**	**0.000**
Scrubland-Heathland	0.616	0.058	0.950	0.168	0.828	0.110	0.151
Other Conifers-Oak	0.973	0.982	0.990	0.954	1.000	0.966	0.954
Pine-Oak	0.762	1.000	0.886	0.677	0.055	1.000	1.000
Pine/Broadleaved-Oak	1.000	1.000	1.000	1.000	0.969	1.000	1.000
Scrubland-Oak	**0.001**	0.757	**0.000**	0.350	**0.044**	**0.003**	**0.001**
Pine-Other Conifers	0.997	1.000	0.999	0.995	**0.005**	0.955	0.891
Pine/Broadleaved-Other Conifers	0.999	0.998	0.999	0.995	0.736	0.997	0.986
Scrubland-Other Conifers	**0.000**	0.145	**0.000**	**0.018**	0.051	**0.000**	**0.000**
Pine/Broadleaved-Pine	0.939	1.000	0.964	0.866	0.417	1.000	1.000
Scrubland-Pine	**0.000**	0.313	**0.000**	**0.004**	**0.000**	**0.000**	**0.000**
Scrubland-Pine/Broadleaved	**0.000**	0.605	**0.000**	0.201	**0.004**	**0.001**	**0.000**

Table S 4.9. Bird indices calculated from the bird species present in each plot in the New Forest. For variable explanation see Table 3.6.

Plot No.	**Habitat Class**	***Bird_ Density***	***Spp_ Richness***	***Spp_ Diversity***	***Spp_ Decline***	***Spp_ Priority***	***Spp_ Priority***^{w}	***Spp_ Rarity***	***Spp_ IRR***
1	**Heathland**	6.80	14	2.347	4	6	3.70	14	0.055
2	**Heathland**	7.80	16	2.488	5	8	7.40	23	0.064
3	**Heathland**	3.73	23	2.896	6	12	4.25	42	0.165
4	**Heathland**	8.40	12	2.241	5	12	7.40	21	0.446
5	**Scrubland**	28.75	25	2.781	8	13	14.25	28	0.065
6	**Scrubland**	12.80	25	2.958	4	14	7.20	55	0.146
7	**Scrubland**	7.82	26	3.061	7	20	60.00	58	0.229
8	**Scrubland**	7.18	24	2.986	9	16	5.46	45	0.180
9	**Beech**	19.50	20	2.674	10	11	8.75	33	0.076
10	**Beech**	31.25	20	2.568	13	5	11.00	36	0.340
11	**Beech**	24.50	19	2.657	10	6	10.75	26	0.169
12	**Oak**	32.33	23	2.714	8	15	15.00	41	0.044
13	**Oak**	17.60	19	2.644	6	9	6.80	24	0.076
14	**Oak**	20.86	24	2.702	10	12	12.14	38	0.178
15	**Beech/Oak**	17.50	18	2.584	6	7	8.50	24	0.081
16	**Beech/Oak**	23.17	28	2.739	8	13	11.00	53	0.115
17	**Beech/Oak**	22.50	21	2.792	6	11	11.75	27	0.145
18	**Beech/Oak**	16.83	21	2.688	9	9	7.00	39	0.072
19	**Beech/Oak**	22.00	24	2.776	8	12	11.60	38	0.161
20	**Pine/ Broadleaved**	14.33	20	2.515	6	6	4.17	23	0.046
21	**Pine/ Broadleaved**	15.25	13	2.316	8	5	5.75	14	0.032
22	**Pine/ Broadleaved**	16.83	22	2.524	4	12	7.00	30	0.265
23	**Pine**	14.75	15	2.384	8	4	3.25	11	0.168
24	**Pine**	12.60	20	2.566	4	9	4.00	30	0.069
25	**Pine**	10.67	13	2.067	6	4	1.50	16	0.032
26	**Pine**	13.50	18	2.502	3	9	5.67	20	0.013
27	**Pine**	8.50	9	1.654	6	4	1.33	8	0.023
28	**Other Conifers**	30.00	22	2.514	3	7	5.67	35	0.061
29	**Other Conifers**	16.00	12	2.212	6	6	3.67	16	0.267
30	**Other Conifers**	22.50	15	2.147	5	4	3.25	9	0
31	**Other Conifers**	19.33	15	2.250	4	4	3.67	15	0.133
32	**Other Conifers**	18.20	18	2.404	4	4	2.00	23	0.489

Table S 4.10. ANOVA results of the bird variables between habitat classes in the New Forest. Significance: $P<0.05$. R^2 represents the amount of variance in the index that is explained by the habitat classes.

Variable	Df	Sum Sq	Mean Sq	*f*	R^2	P
Bird_Density	7	1063.200	151.890	5.30	0.61	<0.001
Spp_Richness	7	374.400	53.480	4.03	0.54	0.005
Spp_Diversity	7	1.680	0.240	5.68	0.62	<0.001
Spp_Decline	7	110.400	15.767	5.77	0.63	<0.001
Spp_ Priority	7	358.700	51.240	6.67	0.66	<0.001
Spp_Priority^{w}	7	278.400	39.770	6.75	0.66	<0.001
Spp_Rarity	7	2945.000	420.700	3.97	0.54	0.005
Spp_IRR	7	0.071	0.010	0.66	0.16	0.702

Table S 4.11. The results of the post-hoc Tukey test displaying the habitat class pairs that differ significantly (at $P<0.05$) in terms of each bird index in the New Forest, non-significant variables excluded.

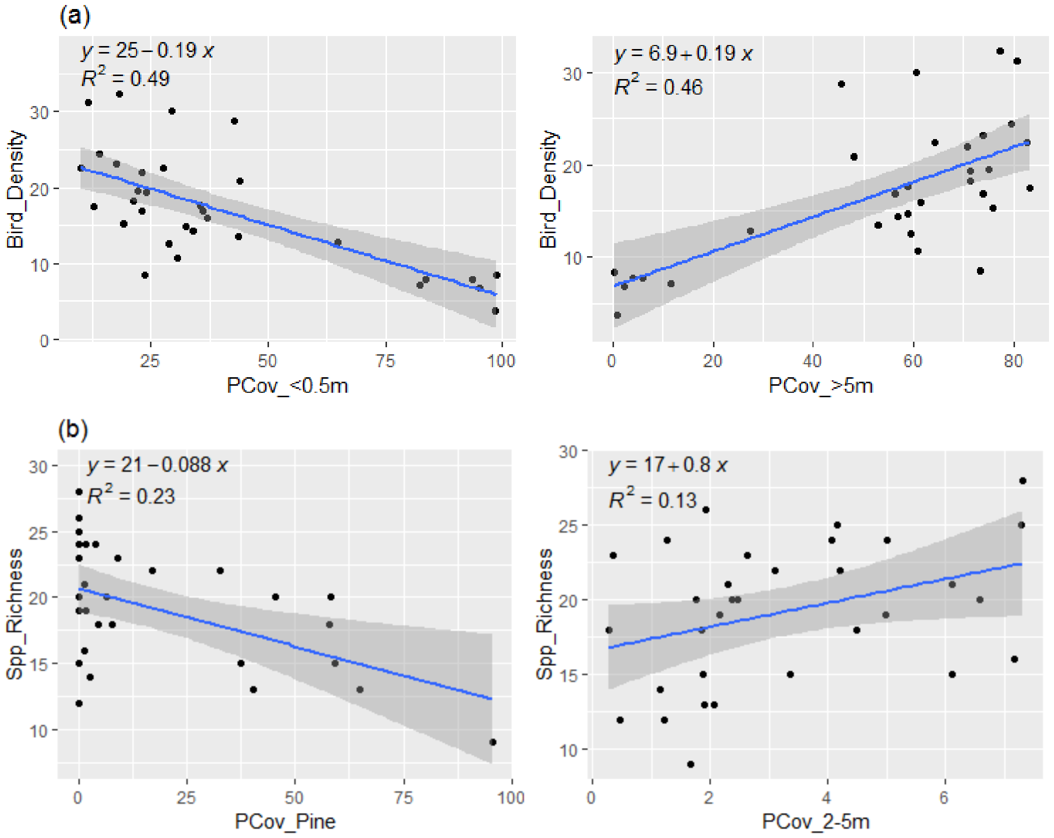
Habitat Pairs/Variables	***Bird_ Density***	***Spp_ Richness***	***Spp_ Diversity***	***Spp_ Decline***	***Spp_ Priority***	***Spp_ Priority***^{w}	***Spp_ Rarity***
Beech/Oak-Beech	0.925	0.965	0.999	0.099	0.792	1.000	0.999
Heathland-Beech	**0.003**	0.915	0.984	**0.002**	0.966	0.279	0.988
Oak-Beech	1.000	0.992	1.000	0.374	0.464	0.999	1.000
Other Conifers-Beech	0.971	0.915	0.395	**0.000**	0.937	**0.022**	0.743
Pine-Beech	**0.046**	0.654	0.185	**0.002**	0.997	**0.012**	0.532
Pine/Broadleaved-Beech	0.386	1.000	0.955	**0.021**	1.000	0.342	0.948
Scrubland-Beech	0.178	0.553	0.504	0.068	**0.011**	0.962	0.572
Heathland-Beech/Oak	**0.016**	0.236	0.737	0.406	1.000	0.194	0.733
Oak-Beech/Oak	0.990	1.000	1.000	1.000	0.992	0.994	1.000
Other Conifers-Beech/Oak	1.000	0.203	0.069	0.124	0.081	**0.008**	0.223
Pine-Beech/Oak	0.251	0.062	**0.021**	0.556	0.238	**0.004**	0.106
Pine/Broadleaved-Beech/Oak	0.904	0.785	0.651	0.936	0.870	0.267	0.598
Scrubland-Beech/Oak	0.660	0.958	0.703	1.000	0.122	0.957	0.804
Oak-Heathland	**0.008**	0.462	0.913	0.297	0.930	0.089	0.928
Other Conifers-Heathland	**0.009**	1.000	0.866	0.999	0.276	0.908	0.993
Pine-Heathland	0.809	0.999	0.580	1.000	0.574	0.769	0.936
Pine/Broadleaved-Heathland	0.414	0.994	1.000	0.992	0.986	1.000	1.000
Scrubland-Heathland	0.521	**0.042**	0.075	0.681	0.065	0.811	0.105
Other Conifers-Oak	0.998	0.440	0.226	0.099	**0.036**	**0.005**	0.527
Pine-Oak	0.103	0.193	0.094	0.412	0.103	**0.002**	0.331
Pine/Broadleaved-Oak	0.589	0.914	0.848	0.809	0.554	0.126	0.836
Scrubland-Oak	0.327	0.955	0.714	0.992	0.643	0.708	0.775
Pine-Other Conifers	0.165	0.998	0.999	0.976	0.999	1.000	1.000
Pine/Broadleaved-Other Conifers	0.817	0.995	0.974	0.880	0.883	0.946	1.000
Scrubland-Other Conifers	0.521	**0.032**	**0.002**	0.312	**0.000**	0.139	**0.013**
Pine/Broadleaved-Pine	0.985	0.907	0.827	1.000	0.990	0.847	0.996
Scrubland-Pine	0.999	**0.008**	**0.001**	0.829	**0.001**	0.076	**0.005**
Scrubland-Pine/Broadleaved	1.000	0.287	0.070	0.992	**0.016**	0.851	0.083

Table S 4.12.Correlation matrix of the habitat variables in the New Forest to one another for the multiple regression (significant relationships are in bold, $P < 0.05$).

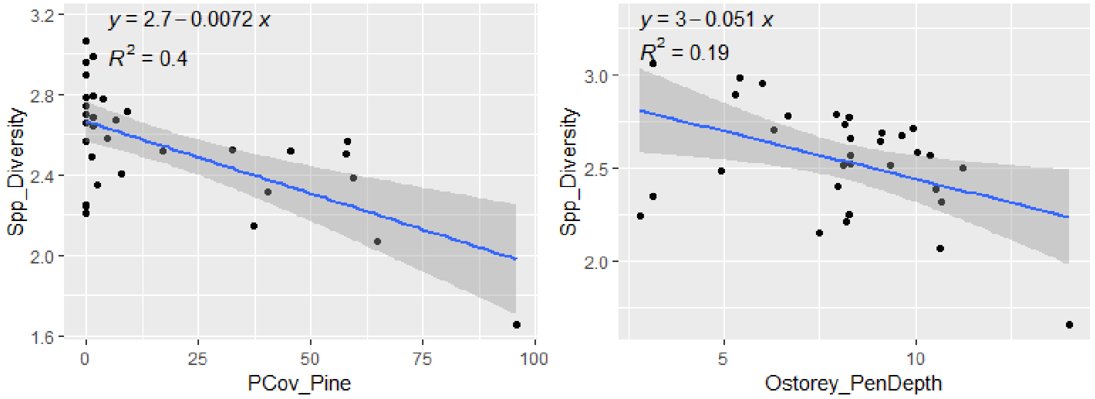
Variables	***Ht_Av***	***Ht_StDev***	***Ostorey_Ht***	***Ostorey_Pen Depth***	***Ht_VDR***	***PCov_<0.5m***	***PCov_0.5-2m***	***PCov_2-5m***	***PCov_>5m***	***PCov_Pine***	***PCov_Cons Other***	***PCov_Beech***	***PCov_Oak***	***PCov_Broad Other***	***PCov_Heather***	***PCov_Gorse***	***Pres_Bracken***	***Pres_Holly***	***Pres_Hawthorn***	***Pres_Birch***	***Pres_Willow***
Ht_Av	1																				
Ht_StDev	**0.732**	1																			
Ostorey_Ht	**0.915**	**0.871**	1																		
Ostorey_PenDepth	**0.864**	**0.614**	**0.821**	1																	
Ht_VDR	**-0.977**	**-0.666**	**-0.858**	**-0.862**	1																
PCov_<0.5m	**-0.948**	**-0.743**	**-0.826**	**-0.776**	**0.927**	1															
PCov_0.5-2m	0.095	**0.392**	0.239	0.293	-0.071	-0.130	1														
PCov_2-5m	0.099	0.194	0.002	-0.114	-0.044	-0.328	-0.025	1													
PCov_>5m	**0.961**	**0.718**	**0.831**	**0.785**	**-0.946**	**-0.994**	0.052	0.261	1												
PCov_Pine	**0.357**	0.168	0.311	**0.684**	**-0.380**	-0.269	**0.487**	-0.290	0.259	1											
PCov_ConsOther	0.204	0.290	0.220	0.005	-0.202	-0.235	-0.083	0.053	0.245	-0.109	1										
PCov_Beech	**0.490**	0.276	**0.408**	0.211	**-0.464**	**-0.488**	-0.289	0.282	**0.505**	-0.291	-0.261	1									
PCov_Oak	0.335	0.199	0.251	0.229	-0.333	**-0.378**	-0.114	0.168	**0.386**	-0.279	-0.332	0.323	1								
PCov_BroadOther	-0.275	-0.045	-0.320	-0.233	0.257	0.116	0.084	**0.509**	-0.167	-0.108	-0.102	-0.196	-0.089	1							
PCov_Heather	**-0.757**	**-0.788**	**-0.748**	**-0.729**	**0.718**	**0.829**	-0.279	**-0.447**	**-0.796**	-0.269	-0.180	-0.277	-0.331	-0.098	1						
PCov_Gorse	**-0.512**	**-0.467**	**-0.548**	**-0.517**	**0.507**	**0.509**	-0.139	-0.077	**-0.507**	-0.212	-0.145	-0.223	-0.170	0.126	**0.563**	1					
Pres_Bracken	0.272	0.327	0.284	**0.379**	-0.269	-0.285	**0.463**	-0.026	0.257	0.341	-0.250	0.111	0.090	0.061	-0.190	-0.208	1				
Pres_Holly	0.320	**0.384**	0.284	0.121	-0.290	**-0.381**	-0.149	**0.391**	**0.374**	-0.282	-0.170	**0.536**	**0.378**	0.222	-0.347	-0.092	0.331	1			
Pres_Hawthorn	-0.211	-0.126	-0.199	-0.148	0.227	0.177	0.064	0.010	-0.189	-0.040	-0.195	-0.064	0.099	-0.150	0.189	0.259	**0.373**	0.168	1		
Pres_Birch	**0.397**	0.296	0.341	0.382	**-0.431**	**-0.394**	0.099	0.115	**0.389**	0.174	0.111	0.170	0.033	0.174	**-0.362**	-0.040	0.085	0.214	-0.293	1	
Pres_Willow	**-0.536**	**-0.422**	**-0.502**	**-0.491**	**0.478**	**0.508**	-0.068	0.013	**-0.519**	-0.192	-0.206	-0.156	-0.240	**0.374**	**0.418**	0.309	-0.158	-0.071	0.061	0.124	1

Table S 4.13. Correlation matrix of the bird indices used in the New Forest analysis.

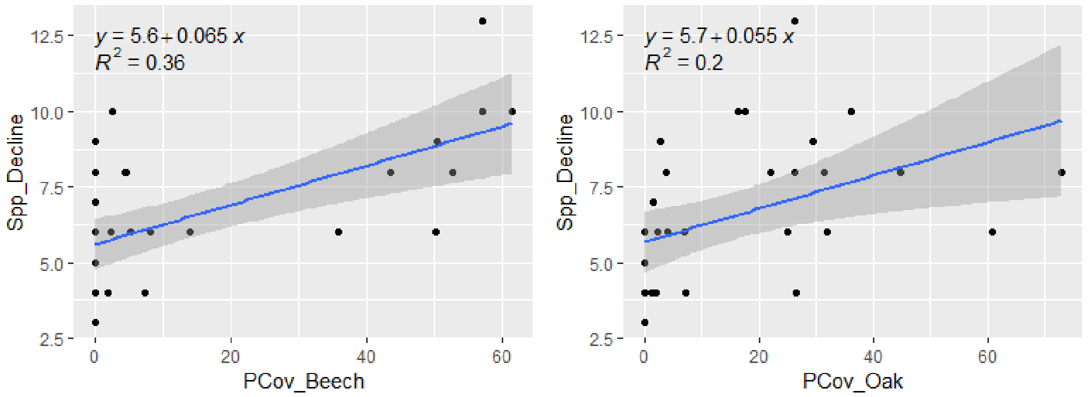
Bird Variables	***Bird_ Density***	***Spp_ Richness***	***Spp_ Diversity***	***Spp_ Priority***	***Spp_ Priority^w***	***Spp_ Decline***	***Spp_ Rarity***	***Spp_ IRR***
Bird_ Density	1							
Spp_ Richness	0.325	1						
Spp_ Diversity	0.136	**0.893**	1					
Spp_ Priority	-0.061	**0.748**	**0.774**	1				
Spp_ Priority^w	**0.640**	**0.599**	**0.567**	**0.519**	1			
Spp_ Decline	**0.359**	0.263	0.293	0.150	**0.530**	1		
Spp_ Rarity	0.114	**0.887**	**0.843**	**0.816**	**0.480**	0.287	1	
Spp_ IRR	-0.027	0.034	0.098	0.115	0.014	0.096	0.167	1



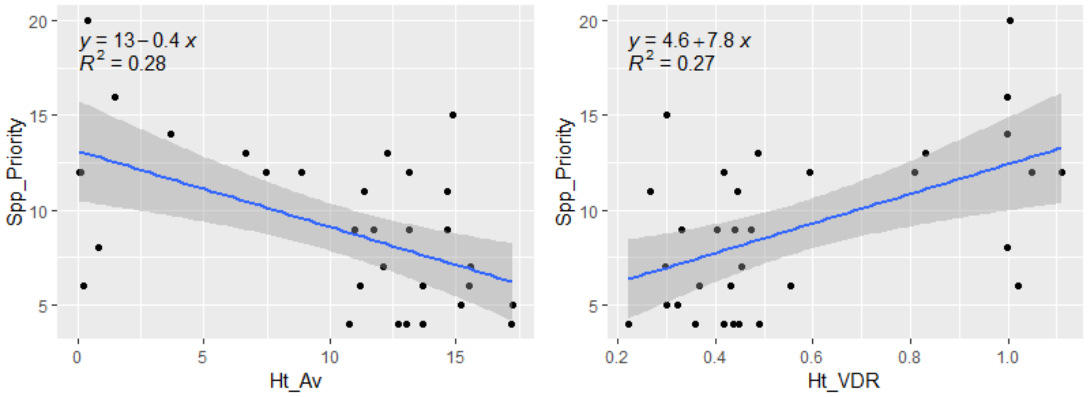
(c)


y = 2.7 − 0.0072 x
R² = 0.4
Spp_Diversity
3.2
2.8
2.4
2.0
1.6
0
25
50
75
100
PCov_Pine
y = 3 − 0.051 x
R² = 0.19
Spp_Diversity
3.0
2.5
2.0
5
10
Ostorey_PenDepth

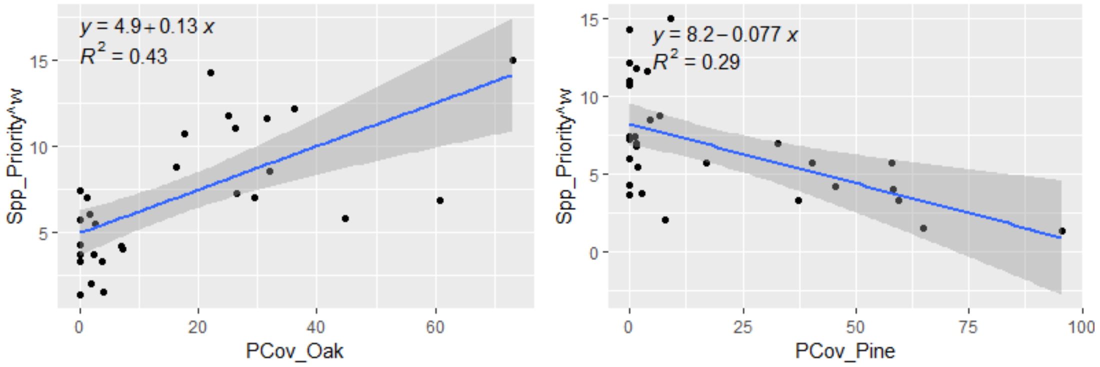
(d)


y = 5.6 + 0.065 x
R² = 0.36
Spp_Decline
12.5
10.0
7.5
5.0
2.5
0
20
40
60
PCov_Beech
y = 5.7 + 0.055 x
R² = 0.2
Spp_Decline
12.5
10.0
7.5
5.0
2.5
0
20
40
60
PCov_Oak

(e)


y = 13 − 0.4 x
R² = 0.28
Spp_Priority
20
15
10
5
0
5
10
15
Ht_Av
y = 4.6 + 7.8 x
R² = 0.27
Spp_Priority
20
15
10
5
0.2
0.4
0.6
0.8
1.0
Ht_VDR

(f)


y = 4.9 + 0.13 x
R² = 0.43
Spp_Priority^w
15
10
5
0
20
40
60
PCov_Oak
y = 8.2 − 0.077 x
R² = 0.29
Spp_Priority^w
15
10
5
0
0
25
50
75
100
PCov_Pine

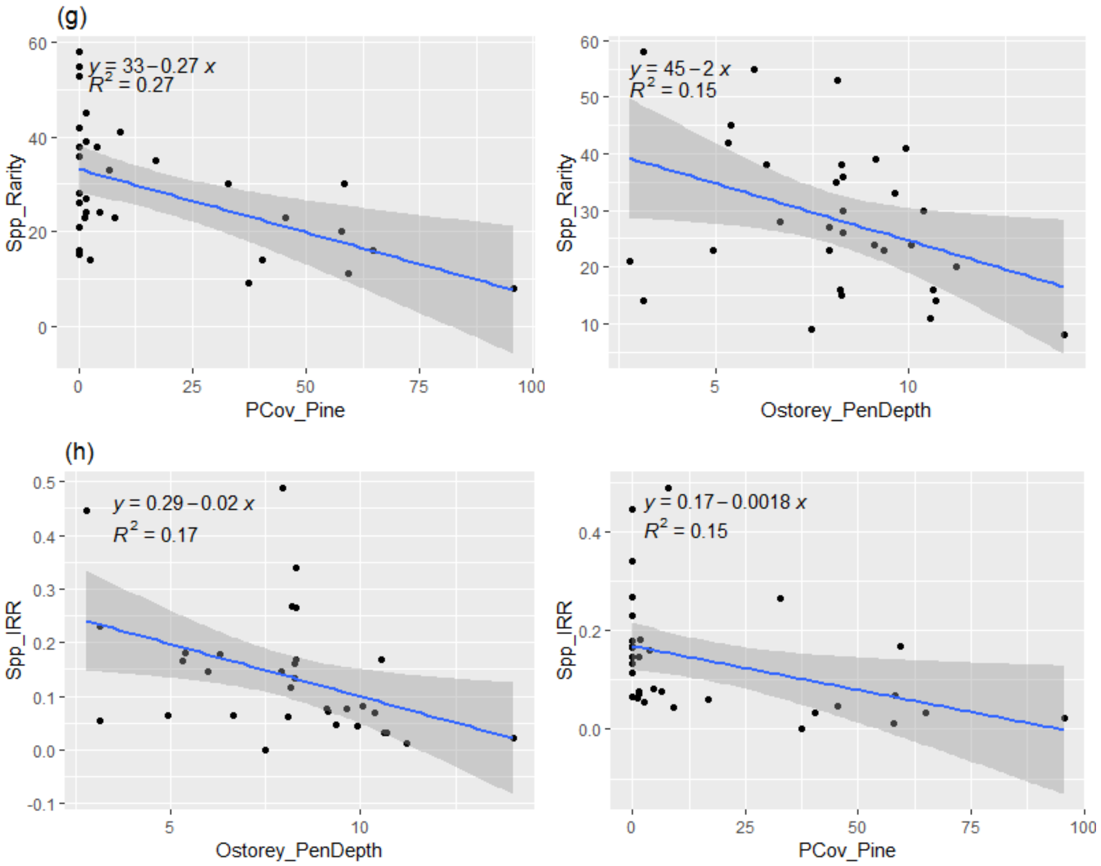


Figure S 4.1. Linear regression graphs for the two highest significant correlation values (Pearson's r) from the correlation matrix in Table 4.1 for each of the bird variables (P<0.05). Confidence intervals are standard error (R version 3.5.2).

Table S 4.14. Model selection tables of the top two AICc ranked models in each candidate set from global models with the appropriate family and link function containing selected and standardised independent variables. ΔAICc = difference in AICc between model and top model. w_i *= Aikaike model weight. acc* w_i *= cumulative model weight. For variable derivation see Table 3.1, Table 3.3 and Table 3.6. *Spp_IRR +0.0001 to "nudge" from 0 to allow model to work. Rescale = standardising variables.*

a) Global Model = Bird_Density ~ Ht_VDR + PCov_2-5m + PCov_Beech + PCov_Oak + Pres_Holly + Pres_Willow, family=inverse.gaussian (link=identity)

Candidate Models	**df**	**logLik**	**AICc**	**ΔAICc**	w_i	***acc*** w_i	**adjR^2**
Ht_VDR + PCov_2-5m	4	-94.32	198.11	0	0.38	0.38	0.64
Ht_VDR + PCov_2-5m + PCov_Oak	5	-93.14	198.59	0.47	0.30	0.68	0.67
Ht_VDR + PCov_2-5m + Pres_Willow	5	-94.01	200.33	2.21	0.13	0.81	0.65
Ht_VDR + PCov_2-5m + Pres_Holly	5	-94.24	200.79	2.68	0.10	0.91	0.65
Ht_VDR + PCov_2-5m + PCov_Beech	5	-94.31	200.93	2.81	0.09	1	0.64

b) Global Model = Spp_Richness ~ PCov_2-5m + PCov_Pine + PCov_ConsOther + PCov_Oak, family=poisson(link=identity)

Candidate Models	**df**	**logLik**	**AICc**	**ΔAICc**	w_i	***acc*** w_i	**Pseudo-R^2**
PCov_Pine + PCov_ConsOther	3	-88.44	183.74	0	0.31	0.31	0.26
PCov_Pine + PCov_ConsOther + PCov_2-5m	4	-87.47	184.42	0.67	0.22	0.53	0.28
PCov_Pine	2	-90.43	185.28	1.53	0.14	0.67	0.07
PCov_Pine + PCov_2-5m	3	-89.61	186.08	2.34	0.10	0.77	0.10
PCov_Pine + PCov_ConsOther + PCov_Oak	4	-88.40	186.28	2.53	0.09	0.86	0.32
PCov_Pine + PCov_Oak	3	-89.88	186.61	2.87	0.07	0.93	0.24
PCov_Pine + PCov_ConsOther + PCov_2-5m	4	-89.20	187.88	4.14	0.04	0.97	0.25
PCov_ConsOther + PCov_Oak	3	-91.15	189.15	5.41	0.02	0.99	0.22

c) Global Model = Spp_Diversity ~ Ht_VDR + PCov_<0.5m + PCov_Pine + PCov_ConsOther + PCov_Oak + PCov_Beech + Pres_Willow, family=Gamma(link=identity)

Candidate Models	**df**	**logLik**	**AICc**	**ΔAICc**	***w_i***	***acc w_i***	**adjR^2**
PCov_ConsOther + PCov_Pine	4	7.09	-4.70	0	0.41	0.41	0.58
PCov_ConsOther + PCov_Pine + Ht_VDR	5	7.31	-2.32	2.38	0.12	0.53	0.59
PCov_ConsOther + PCov_Pine +PCov_Beech	5	7.26	-2.21	2.48	0.12	0.65	0.59
PCov_ConsOther + PCov_Pine + Pres_Willow	5	7.12	-1.93	2.76	0.10	0.75	0.59
PCov_ConsOther + PCov_Pine + PCov_<0.5m	5	7.10	-1.89	2.81	0.10	0.85	0.59
PCov_ConsOther + PCov_Pine + PCov_Oak	5	7.09	-1.88	2.82	0.10	0.95	0.58
Ht_VDR + PCov_Beech + PCov_Oak	5	6.25	-0.19	4.50	0.04	0.99	0.56

d) Global Model = Spp_Decline ~ PCov_0.5-2m + PCov_Pine + PCov_ConsOther + PCov_Oak + PCov_Beech, family=poisson(link=log)

Candidate Models	**df**	**logLik**	**AICc**	**ΔAICc**	***w_i***	***acc w_i***	**Pseudo-R^2**
PCov_Beech	2	-67.51	139.44	0	0.18	0.18	0.34
PCov_Beech + PCov_Oak	3	-66.50	139.85	0.42	0.15	0.33	0.42
PCov_Beech + PCov_ConsOther	3	-66.56	139.99	0.55	0.14	0.47	0.42
PCov_Beech + PCov_ConsOther + PCov_Oak	4	-65.97	141.41	1.97	0.07	0.54	0.46
PCov_Beech + PCov_Pine	3	-67.40	141.66	2.23	0.06	0.60	0.35
PCov_Beech + PCov_0.5-2m	3	-67.41	141.68	2.25	0.06	0.66	0.35
PCov_Beech + PCov_Beech + PCov_Pine	4	-66.30	142.08	2.64	0.05	0.71	0.44
PCov_Beech + PCov_ConsOther + PCov_0.5-2m	4	-66.33	142.14	2.70	0.05	0.76	0.43
PCov_Beech + PCov_Oak + PCov_0.5-2m	4	-66.42	142.32	2.88	0.04	0.80	0.43
PCov_Beech + PCov_Oak + PCov_Pine	4	-66.48	142.45	3.01	0.04	0.84	0.42
PCov_Oak	2	-69.59	143.58	4.15	0.02	0.86	0.19
PCov_Oak + PCov_ConsOther	3	-68.48	143.82	4.38	0.02	0.88	0.27
PCov_ConsOther + PCov_Pine	3	-68.60	144.06	4.62	0.02	0.90	0.26
PCov_ConsOther	2	-69.83	144.07	4.63	0.02	0.92	0.17
PCov_ConsOther + PCov_0.5-2m	3	-68.65	144.15	4.71	0.02	0.94	0.26
PCov_0.5-2m + PCov_Beech + PCov_Pine	4	-67.37	144.22	4.78	0.02	0.96	0.36
PCov_0.5-2m + PCov_ConsOther + PCov_Oak	4	-67.59	144.66	5.22	0.01	0.97	0.34
PCov_0.5-2m + PCov_Oak	3	-68.91	144.68	5.25	0.01	0.98	0.24
PCov_ConsOther + PCov_Oak + PCov_Pine	4	-67.87	145.22	5.79	0.01	0.99	0.32
PCov_Oak + PCov_Pine	3	-69.25	145.36	5.92	0.01	1	0.21

e) Global Model = Spp_Priority ~ Ht_VDR + PCov_>5m + PCov_Pine + PCov_ConsOther + PCov_Heather + Pres_Willow, family=poisson(link=identity)

Candidate Models	**df**	**logLik**	**AICc**	**ΔAICc**	***w_i***	***acc w_i***	**Pseudo-R^2**
PCov_ConsOther + PCov_Pine + Ht_VDR	4	-76.95	163.37	0	0.32	0.32	0.54
PCov_ConsOther + PCov_Pine + PCov_>5m	4	-77.23	163.93	0.56	0.24	0.56	0.53
PCov_ConsOther + PCov_Pine	3	-78.82	164.50	1.13	0.18	0.74	0.48
PCov_ConsOther + PCov_Pine + Pres_Willow	4	-77.77	165.02	1.65	0.14	0.88	0.51
PCov_ConsOther + PCov_Heather + PCov_Pine	4	-78.05	165.58	2.20	0.11	0.99	0.50
PCov_ConsOther + Ht_VDR	3	-81.03	168.92	5.55	0.02	1.01	0.40

f) Global Model = Spp_Priority^w ~ PCov_2-5m + PCov_Beech + PCov_Oak + PCov_BroadOther + PCov_Pine + PCov_ConsOther + Pres_Holly, family=Gamma(link=log)

Candidate Models	**df**	**logLik**	**AICc**	**ΔAICc**	***w_i***	***acc w_i***	**adjR^2**
PCov_Pine + PCov_ConsOther + PCov_Oak	5	-66.68	145.67	0	0.36	0.36	0.68
PCov_Pine + PCov_BroadOther + PCov_Oak	5	-66.75	145.81	0.15	0.34	0.70	0.68
PCov_Pine + PCov_ConsOther + PCov_2-5m	5	-67.84	147.98	2.32	0.11	0.81	0.66
PCov_Pine + PCov_ConsOther	4	-70.24	149.96	4.29	0.04	0.85	0.60
PCov_Pine + PCov_Oak + PCov_2-5m	5	-68.87	150.04	4.38	0.04	0.89	0.63
PCov_Pine + PCov_ConsOther + PCov_BroadOther	5	-68.89	150.08	4.42	0.04	0.93	0.63
PCov_Pine + PCov_Oak	4	-70.82	151.13	5.46	0.02	0.95	0.59
PCov_Oak + PCov_BroadOther + PCov_Beech	5	-69.54	151.39	5.72	0.02	0.97	0.62
PCov_Pine + PCov_ConsOther + Pres_Holly	5	-69.60	151.50	5.83	0.02	0.99	0.62

g) Global Model = glm.nb(Spp_Rarity ~ rescale(Ht_VDR) + rescale(PCov_Pine) + rescale(PCov_ConsOther) + rescale(PCov_Oak) + rescale(PCov_Beech) + rescale(Ht_Av) + rescale(PCov_<0.5m), link=identity (Rescale = standardising variables)

Candidate Model	**df**	**logLik**	**AICc**	**ΔAICc**	***w_i***	***acc w_i***	**Pseudo-R²**
PCov_ConsOther + PCov_Pine	4	-115.64	240.77	0	0.38	0.38	0.48
PCov_ConsOther + PCov_Pine + Ht_VDR	5	-115.51	243.32	2.56	0.11	0.49	0.49
PCov_ConsOther + PCov_Pine + PCov_Oak	5	-115.59	243.48	2.72	0.10	0.59	0.48
PCov_ConsOther + PCov_Pine + PCov_<0.5m	5	-115.62	243.56	2.79	0.10	0.69	0.48
PCov_ConsOther + PCov_Pine + Ht_Av	5	-115.63	243.56	2.80	0.09	0.78	0.48
PCov_ConsOther + PCov_Pine + PCov_Beech	5	-115.64	243.58	2.81	0.09	0.87	0.48
Ht_VDR + PCov_Beech + PCov_Oak	5	-115.87	244.05	3.28	0.07	0.94	0.47
PCov_Pine	3	-119.65	246.15	5.38	0.03	0.97	0.34
Ht_Av + PCov_Beech + PCov_Oak	5	-116.93	246.18	5.41	0.03	1	0.44

h) Global Model = betareg(Spp_IRR*~ rescale(PCov_Pine) + rescale(Pres_Birch) + rescale(PCov_ConsOther) + rescale(PCov_Heather), link="probit") Outlier included.

Candidate Models	**df**	**logLik**	**AICc**	**ΔAICc**	***w_i***	***acc w_i***	**Pseudo-R²**
PCov_Pine	3	36.50	-66.14	0	0.30	0.30	0.23
PCov_Pine + Pres_Birch	4	37.56	-65.63	0.50	0.23	0.53	0.26
PCov_Pine + PCov_Heather	4	37.26	-65.03	1.11	0.17	0.70	0.25
PCov_Pine + PCov_Heather + Pres_Birch	5	38.04	-63.77	2.37	0.09	0.79	0.27
PCov_Pine + PCov_ConsOther	4	36.60	-63.72	2.42	0.09	0.88	0.23
PCov_Pine + PCov_ConsOther + Pres_Birch	5	37.60	-62.90	3.24	0.06	0.94	0.26
PCov_Pine + PCov_ConsOther + PCov_Heather	5	37.28	-62.26	3.88	0.04	0.98	0.25
PCov_Heather	3	33.64	-60.42	5.72	0.02	1	0.08

i) Global Model = betareg (Spp_IRR*~ rescale(PCov_Pine) + rescale(Pres_Birch) + rescale(PCov_ConsOther) + rescale(PCov_Heather), link="probit"). Outlier (plot 30) removed.

Candidate Model	**df**	**logLik**	**AICc**	**ΔAICc**	***w_i***	***acc w_i***	**Pseudo-R^2**
PCov_Pine	3	34.90	-62.92	0	0.17	0.17	0.26
PCov_Pine + Pres_Birch	4	36.13	-62.73	0.19	0.16	0.33	0.32
PCov_Pine + PCov_ConsOther + Pres_Birch	5	37.41	-62.43	0.49	0.13	0.46	0.37
PCov_Pine + PCov_ConsOther	4	35.80	-62.05	0.87	0.11	0.57	0.31
PCov_Pine + PCov_Heather	4	35.65	-61.75	1.17	0.10	0.67	0.30
PCov_Pine + PCov_ConsOther + PCov_Heather	5	37.03	-61.66	1.26	0.09	0.76	0.36
PCov_Pine + PCov_Heather + Pres_Birch	5	36.60	-60.79	2.13	0.06	0.82	0.33
PCov_ConsOther + PCov_Heather	4	34.61	-59.68	3.24	0.03	0.85	0.22
PCov_ConsOther + PCov_Heather + Pres_Birch	5	36.03	-59.67	3.25	0.03	0.88	0.27
PCov_ConsOther + Pres_Birch	4	34.46	-59.38	3.54	0.03	0.91	0.20
Pres_Birch	3	32.73	-58.57	4.35	0.02	0.93	0.10
PCov_Heather	3	32.65	-58.40	4.52	0.02	0.95	0.10
PCov_Heather + Pres_Birch	4	33.73	-57.93	4.99	0.01	0.96	0.14
PCov_ConsOther	3	32.36	-57.84	5.08	0.01	0.97	0.08
(Null)	2	31.10	-57.77	5.15	0.01	0.98	-

5 Bird-Habitat Relationships for Cambridgeshire

5.1 Abstract

Agricultural intensification has caused declines in biodiversity and is still seen today in the continuing decline of farmland birds through habitat loss and fragmentation. Diversity may not be the best measure of habitat health in the agricultural landscape in Cambridgeshire as this suggests all birds are equal, whereas some bird species are suffering greater population declines and are subsequently rare. As expected, woodlands were found to be the most important habitat for birds, with greater bird diversity on the woodland edge and a corresponding exclusion zone in the fields adjacent to the woodland edge. The woodlands were found to be sufficiently interconnected to allow a metapopulation to persist in even the smallest woodlands, regardless of their size. Hedges were shown to not only increase bird diversity in the field habitats, but also support a greater number of declining and rare species, most likely as a result of historic hedge decline. The rarest species, represented by the highest relative rarity (IRR), were found to prefer or be present on isolated patches of relatively tall oak woodland, possibly suggesting an effect of oak/tree decline. However, bird community composition analysis is required to determine these rare and declining bird species.

5.2 Introduction

Agriculture is the dominant land-use for the majority of the UK with c. 72% of lowland Britain consisting of farmland (DEFRA 2018), and Cambridgeshire is no exception. As per Section 2.2.1.1, agricultural intensification began after the Second World War and consisted of multiple changes, such as increasing mechanisation, followed by increased pesticide, herbicide and fertiliser use, and altered timings of processes (Robinson and Sutherland 2002, Batáry et al. 2015). Consequently, this has had a dramatic and devastating effect on wildlife, with farmland birds showing the greatest declines (DEFRA 2019).

Increased agriculture has also caused woodland isolation and fragmentation (Section 2.2.1.4) further affecting wildlife and bird species diversity. This creates a metapopulation of suitable habitat patches with the degree of separation and isolation affecting the connectivity, and thus the ability to interact and disperse to different patches to maintain functional populations in line with Metapopulation Theory (Hanski et al. 1997). Freemark and Merriam (1986) stated that vegetation heterogeneity in forest habitats increased with forest area, in line with the Theory of Island Biogeography (MacArthur and Wilson 1967), with the larger and therefore more heterogeneous forest areas supporting a higher abundance of bird species, mainly interior and resident birds due to the greater area of woodland interior. Conversely, Howell et al. (2000) and Cushman and McGarigal (2003) found that smaller woodlands contained higher species richness, however, this was dominated by generalist and edge preferring bird species. Melin et al. (2018) found that the edges of woodlands, even small woodlands, were more diverse and diversity decreased moving into the woodland interior.

As measures of species richness and diversity can mask changes or effects on more vulnerable bird species, such as those with rare and declining populations (Miller and Cale 2000), this study investigates multiple bird indices. The number of declining species, a measure of conservation priority, rarity and relative rarity, along with bird density, diversity and species richness, are all investigated in relation to vegetation composition and structural variables in multiple habitats over the Cambridgeshire landscape. The aim of this study is to investigate if and by how much these different bird measures vary between the habitats and to identify specific habitat characteristics driving these metrics.

5.3 Methods

A total of 38 transects (length 400 – 1340 m) encompassing six habitat classes (Field_low%_Hedge, Field_med%_Hedge, Field_high%_Hedge, Field By_Wood, Wood Edge and Wood Interior) were established over the Cambridgeshire landscape due to access issues public rights of way were used forming line transects (3.3.1.2, pp. 60 – 61). The vegetation composition along each transect was recorded, either as percentage cover or as presence/absence, for the tree, shrub and field layers using both field and aerial data (3.3.2.2, pp. 63 – 64). The same nine ecologically meaningful LiDAR metrics as for the New Forest analysis were selected (3.3.3.1, pp. 65, Table 3.3) describing the vertical and horizontal vegetation structure in each transect area, along with a metric detailing the proportion of hedge (*P_HedgeLen*) in each transect (3.3.3.2, pp. 66, Table 3.4). Additionally, extra woodland patch variables were created for separate woodland analyses (3.3.3.2, pp. 67, Table 3.5). Breeding bird surveys were carried out (twice) using a transect based method (3.3.4.2, pp. 69), the data for analyses were selected to maximise detected numbers and then distance sampling was applied, to account for distance decay in detection, with bird species grouped by similar levels of detectability (3.3.5.2, pp. 70 – 72). This created a new density estimate from which the eight bird variables were calculated (3.3.6 pp. 73 – 75).

For data analysis, firstly, for vegetation composition and structure and the bird indices, the data were tabulated and graphed for a visual inspection of differences across the transects. This was followed by an ANOVA for all three data sets (3.3.7, pp. 75 – 76), with the transects grouped into the habitat classifications (the results of this are presented in Supplementary Material 5.7). The bird variables were then correlated to the vegetation variables to establish if there were any significant bird-habitat relationships (3.3.7.1, pp. 76). Significantly related variables were then used to populate global models to carry out multi-model inferencing and averaging (3.3.7.2, pp. 76 – 78). Multiple variable models were fitted to explain bird-habitat relationships, as such, the number of models in each candidate set differ depending on the bird index. Model inference and averaging were made on standardised variables in a candidate set of models to six ΔAICc and an accumulated weight ($acc\ w_i$) of approximately one (~100% confidence set) for each bird index following Symonds and Moussalli (2011) and Burnham and Anderson (2002). However, if the top model was more than six AICc different from the next model then inference was made from that top model alone.

5.4 Results

5.4.1 Vegetation Composition Across 38 Transects

Agriculture dominates the Cambridgeshire landscape, therefore the main vegetation in the field transects was crops (Figure 5.1). Across the 22 field transects, cereal crops (*PCov_Cereal*) of mostly wheat and barley, and non-cereal crops (*PCov_CropOther*) most notably oilseed rape and beans, dominated (Table S 5.1).

Fewer transects contained improved grassland (*PCov_ImpGrass*), indicating a predominantly arable landscape. Within the non-cropped area of the field transects the percentage cover of woody vegetation was often < 25%, and the dominant species were blackthorn (*Prunus spinosa*) and hawthorn, along with lower percentages of oak, field maple, bramble and dog rose (*Rosa canina*) (Table S 5.2).

Across the 16 woodland transects, the percentage cover < 2 m consists of a mostly bramble and dog rose shrub vegetation layer (Table S 5.3). The most prevalent species in the understorey layer (defined here as 2-5 m) were hawthorn and blackthorn, and to a lesser extent, field maple, hazel (*Corylus avellana*) and apple spp. (*Malus spp.*). The overstorey (defined as > 5m) species were predominantly ash, oak and field maple, and occasionally elm (Figure 5.1 and Table S 5.3).

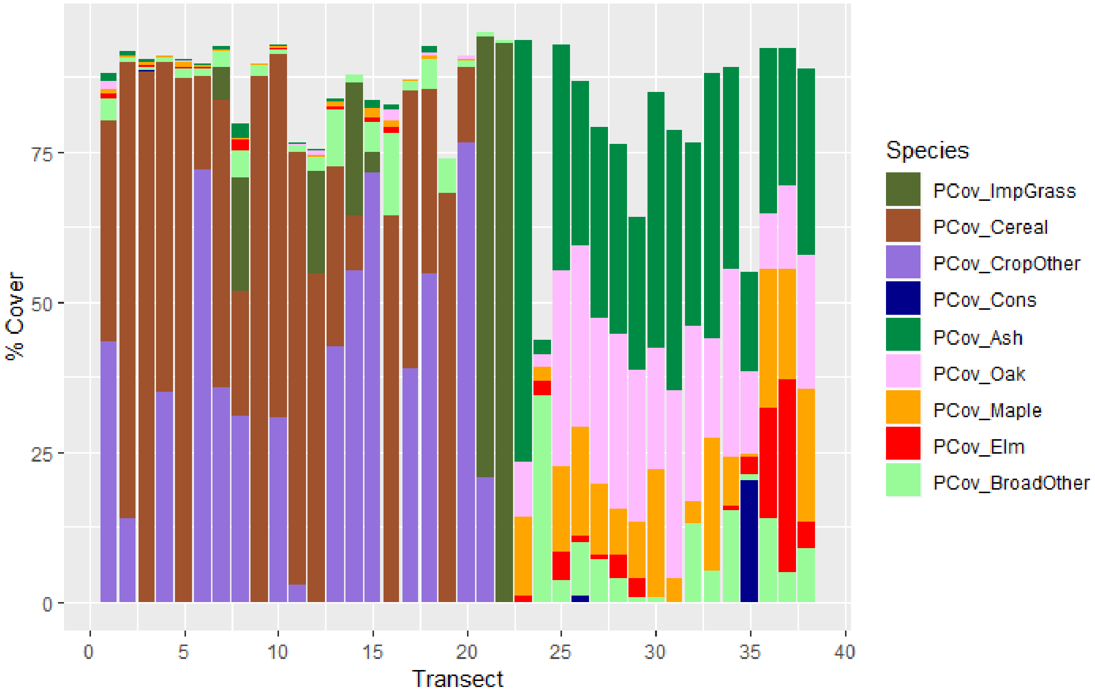


Figure 5.1. The percent cover of all the vegetation types in the 38 transects in Cambridgeshire. Missing land cover: infrastructure (i.e. roads and buildings), water bodies and rough grass/scrub. For variable explanations see Table 3.2 (R version 3.5.2).

5.4.2 Vegetation Structure Across 38 Transects

The LiDAR derived metrics for Cambridgeshire varied between and within habitat classes, emphasizing that similar habitats can contain different structural attributes (Figure 5.2 and Table S 5.7). As expected, the woodland transects had a higher

Ht_Av, Ht_StDev, Ostorey_Ht and *PCov_>5m* than the field transects, which had a higher *Ht_VDR*, *PCov_<0.5m* and *P_HedgeLen* (Figure 5.2 and Figure 5.3). *PCov_0.5-2m* was higher in the field transects overall, however, the only significant difference was between the 'Wood Edge' and 'Field_high%_Hedge' transects (Table S 5.8). *Ostorey_Ht* also, unsurprisingly, showed that vegetation > 5 m in the field transects was consistently lower than in the woodlands (Figure 5.2 and Figure 5.3).

Of the field transects, 'Field_high%_Hedge' had the highest value of *Ostorey_Ht* closely followed by 'Field By_Wood', possibly as a result of encroaching vegetation from adjoining woodland. *PCov_<0.5m* was highest in the 'Field By_Wood' transects, closely followed by 'Field_low%_Hedge' and 'Field_med%_Hedge', indicating that these habitats had more ground vegetation < 0.5 m at slightly different percentages, most likely representing crop (Figure 5.2). However, due to the overlapping range of data in the field transects, as shown in Figure 5.3, the statistical significant difference (at P <0.05) was only between the wood and field transects (Table S 5.8). In contrast, 'Field_high%_Hedge' transect 15 and 'Field By_Wood' transect 18 had a low *PCov_<0.5m* (8.1% and 8.2%, respectively), and a high *PCov_0.5-2m* (82.6% and 84.5%, respectively) suggesting that the majority of the habitat in these transects contained either shrubby vegetation or unharvested crop between 0.5 and 2 m at the time of LiDAR data acquisition (Figure 5.2). The field transects varied in *PCov_0.5-2m* within and between the field habitat classes, and the lower values of *P_HedgeLen* suggest that there may have been vegetation other than hedges present in these transects. *PCov_2-5m* was low in the fields (max. ~10%) and *PCov_>5m* was even less (max. ~9%), however, this indicates that substantial woody vegetation was still present in the fields, albeit at a lower percentage (Figure 5.2). *P_HedgeLen* unsurprisingly increased from 'Field_low%_Hedge' (with 'Field By_Wood' transects having similar *P_HedgeLen*) to 'Field_high%_Hedge' (Figure 5.2). *Ostorey_PenDepth* was higher in the field transects as a result of single trees increasing this metric, whereas the wood transects had less penetrable vegetation possibly as a result of dense understorey vegetation (Figure 5.2 and Figure 5.3). Figure 5.2 also showed variation in vegetation structure between and within the two woodland classes ('Wood Edge' and 'Wood Interior'). However, the results of the ANOVA (Table S 5.7) showed no significant difference between the woodland classes as a result of the overlapping data (Figure 5.3), suggesting that further analysis should be at the transect level, rather than by habitat class. It is worth noting that 'Wood Edge' transect 24 had a higher *PCov_2-5m* than *PCov_>5m* suggesting that the vegetation in this transect was relatively shorter and scrubby, supported by the low *Ht_Av* and *Ostorey_Ht* (Figure 5.2).

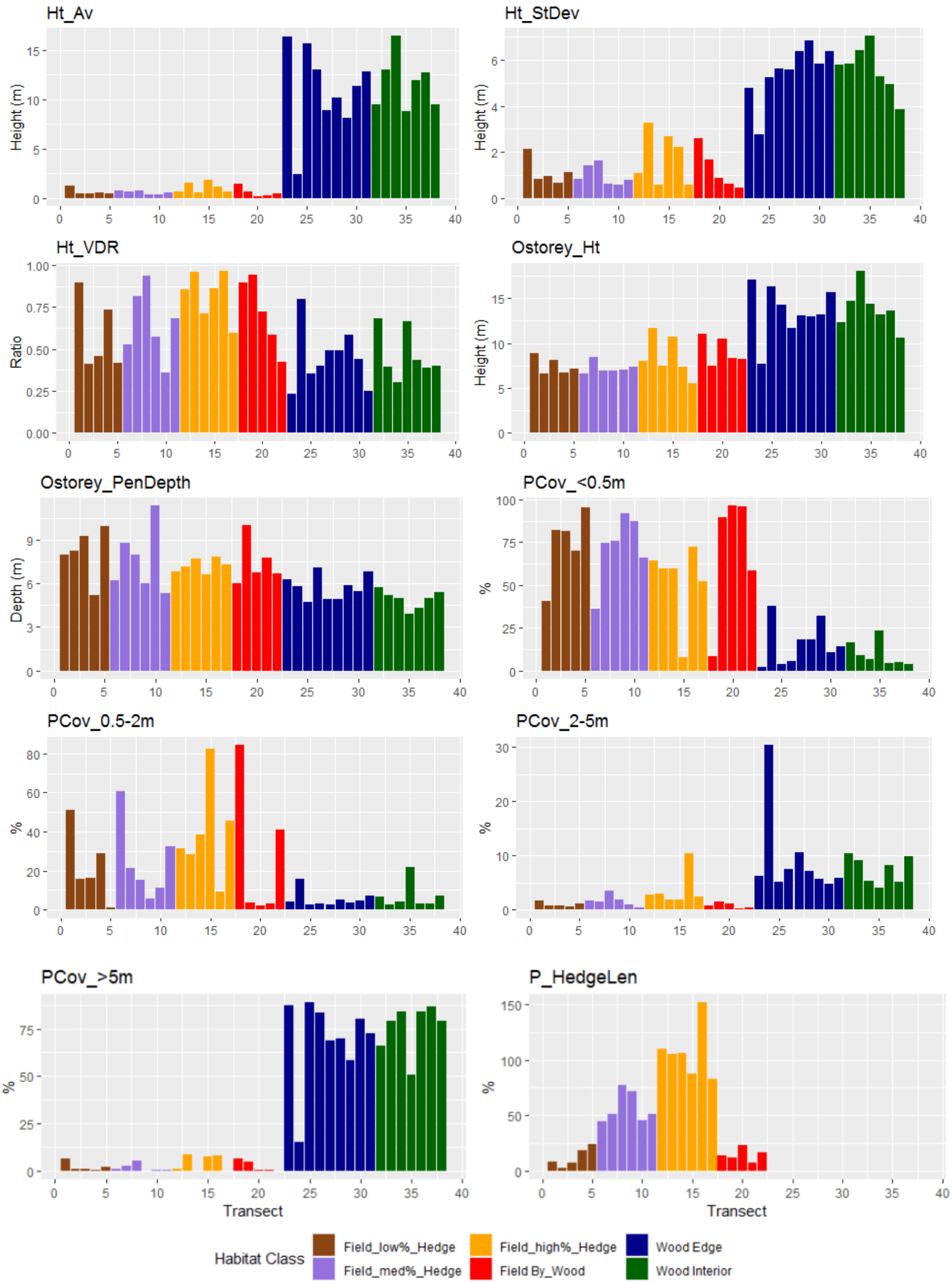


Figure 5.2. Vegetation structure variables for each transect in Cambridgeshire. See Table 3.3 and Table 3.4 for metric derivation (R version 3.5.2).

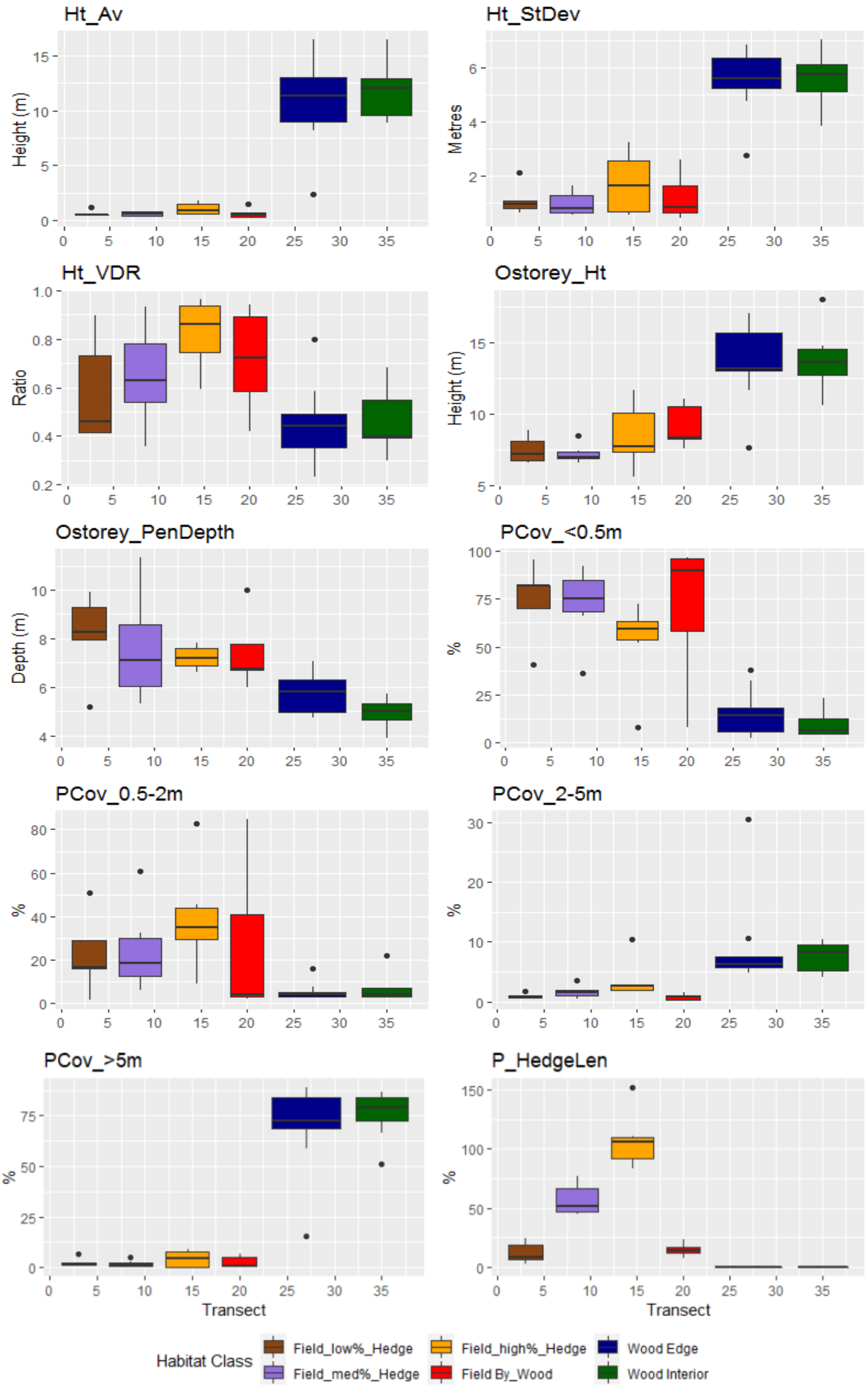


Figure 5.3. The range of the structural variables for each transect class in Cambridgeshire. See Table 3.3 and Table 3.4 for metric derivation (R version 3.5.2).

5.4.3 Extra woodland variables

In order to investigate the effects of Island Biogeography and Metapopulation Theory on the woodlands in Cambridgeshire, extra variables were calculated and added into the separated woodland transect analysis. Figure 5.4 shows that the transects differ within and between the woodland habitat classes, with ‘Wood Edge’ transects 23, 24 and 31 and ‘Wood Interior’ transects 32 and 33 having the highest *Wood.Area* and the highest *Wood.Perimeter* (Table S 5.9), as they were all located in Monks Wood, the largest wood. ‘Wood Interior’ transect 36 had the highest *Wood_P:A* (0.02) and *%_Woods_500m* (173.2%), indicating that this woodland was long and thin (and the smallest; Gamsey Wood), with a high percent of surrounding woodland in the 500 m buffer as a proportion of the surveyed wood area (Figure 5.4). ‘Wood Edge’ transect 26 had the highest value of *%_Woods_1km* (286.7%), indicating the highest percent of surrounding woodlands in a 1 km buffer as a proportion of the surveyed wood area (Figure 5.4). ‘Wood Edge’ transects 25 and 30 and ‘Wood Interior’ transect 34 (all in Aversley Wood) had the lowest values of both *%_Wood_500m* and *%_Wood_1km* demonstrating that Aversley Wood was the most isolated (Figure 5.4). ‘Wood Interior’ transect 34 had the highest value of *Dist_Wood* (966.0 m), closely followed by ‘Wood Edge’ transect 25 (941.1 m), again indicating that the transects in Aversley Wood had the furthest distance to the nearest other woodland (Archer’s Wood). ‘Wood Edge’ transect 26 (Archer’s Wood) had the lowest value of *Dist_Wood* (76.2 m) indicating that this transect had the shortest distance to the nearest woodland (Coppingford Wood). However, the results of the ANOVA (in Table S 5.10) show that there were no statistically significant differences between the two habitat classes for any of the extra woodland variables, presumably as a result of the overlap in data between the two woodland classes (Figure 5.5), and therefore any further analysis should be at the transect level due to the differences between them.

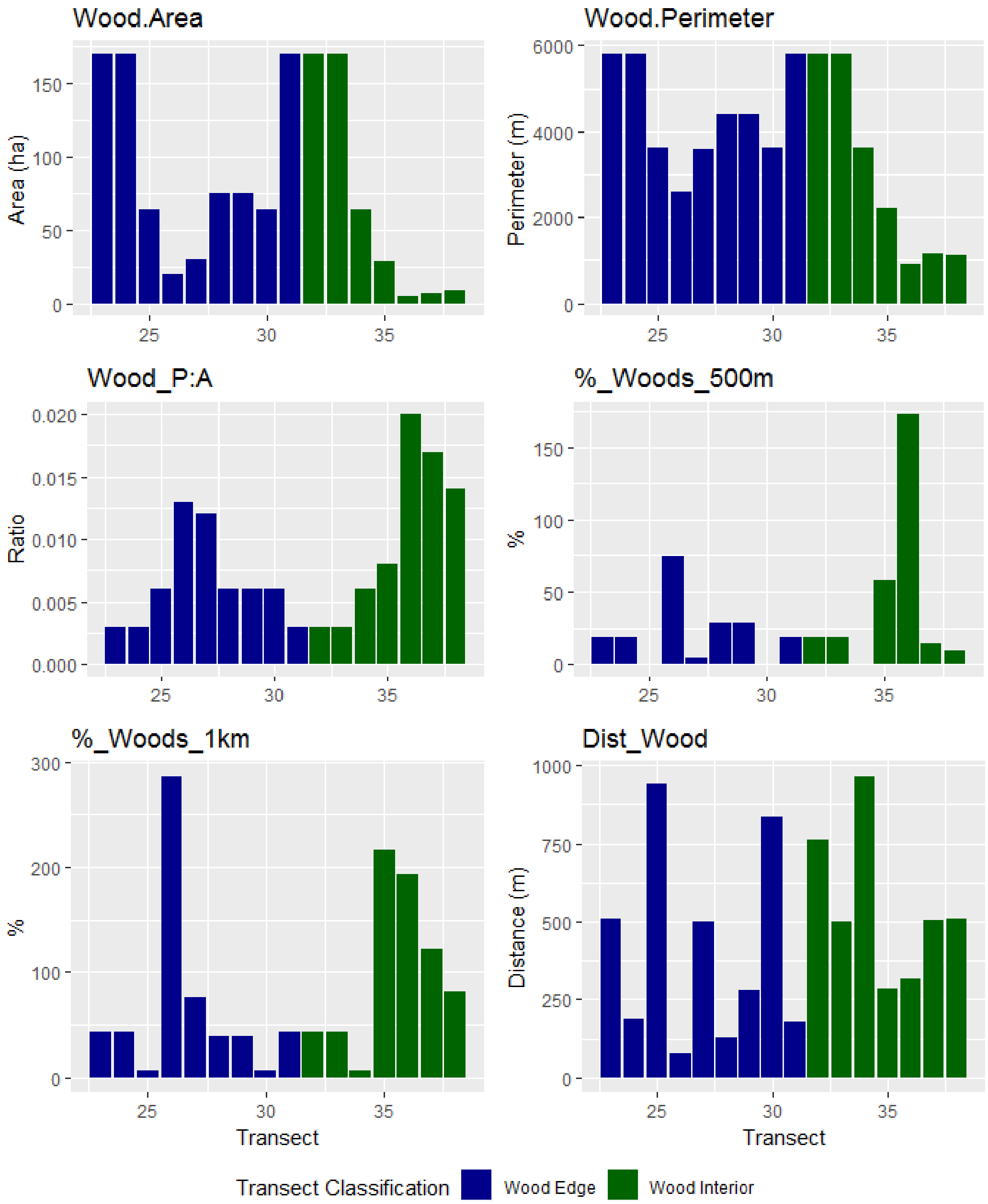


Figure 5.4. Extra woodland variables for the woodland transects in Cambridgeshire. See Table 3.5 for variable explanation (R version 3.5.2).

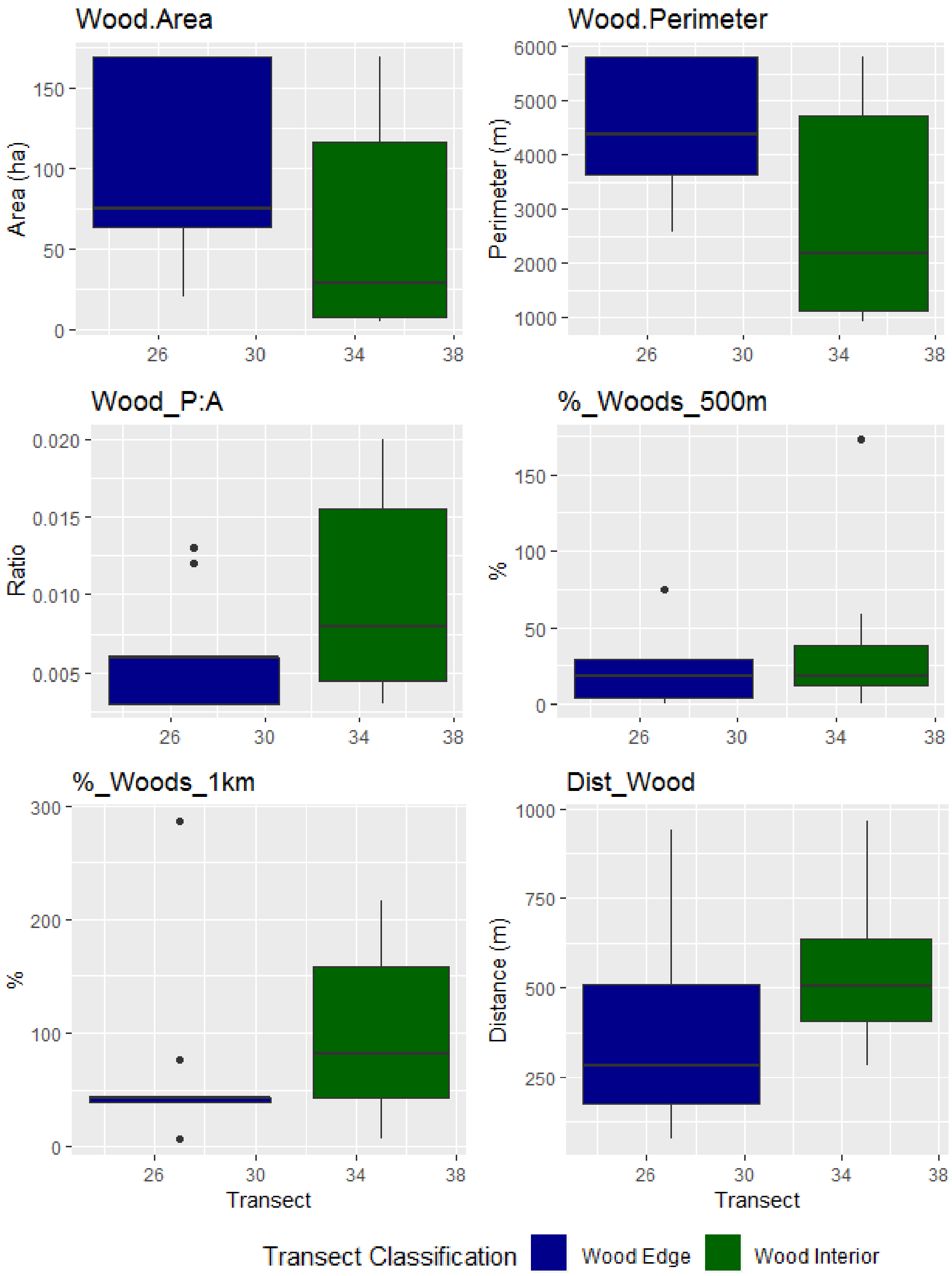


Figure 5.5. The range of the extra woodland variables for the woodland transect classes in Cambridgeshire. See Table 3.5 for metric derivation (R version 3.5.2).

5.4.4 Bird Density Correction

In order to account for distance decay in the surveyed line transects in Cambridgeshire, distance sampling was carried out following Buckland et al. (1993a, 1993b, 1993c) and Miller (2017). This provided a correction factor that could be applied to groups of species (Table 5.1), which was then used to correct the bird density in each survey transect and for each species, where appropriate. Woodland species present in the field transects and densities of hedge dwelling species, such as the Yellowhammer and Common Whitethroat were not corrected because they were more likely to be relatively close to the observer, i.e. in the hedge or other nearby vegetation rather than in the fields, thus minimising error due to distance detectability. The correction was applied to species recorded in the two surveys per transect using distance sampling results from each survey separately as detectability changed with the passing of time, i.e. as a result of foliage emergence and the arrival of breeding migrant birds. The maximum from the two surveys, of this new corrected density, was selected for analysis.

Table 5.1.The calculated group densities from the two bird surveys, the distance sampling density estimate and the ratio of the two surveys as a correction factor for each bird group. Field and woodland birds separated in terms of habitat preference. See Section 3.3.5.2 for species included in the groups.

Wood Survey 1	**Surveyed Density (/ha)**	**Distance Sampling Density (/ha)**	**Ratio**	**Wood Survey 2**	**Surveyed Density (/ha)**	**Distance Sampling Density (/ha)**	**Ratio**
Thrush	1.476	1.728	1.171	**Thrush**	1.585	1.625	1.026
Tits	3.495	6.445	1.844	**Tits**	2.127	3.426	1.611
Doves & Game	2.171	2.527	1.164	**Doves & Game**	3.321	3.839	1.156
Quiet	1.606	3.250	2.023	**Quiet**	2.019	3.791	1.878
Finches	1.476	2.007	1.360	**Finches**	1.411	1.501	1.064
Warblers	2.279	2.971	1.304	**Warblers**	2.062	2.541	1.232
Robin	2.778	3.991	1.436	**Robin**	4.037	5.594	1.386
Woodpeckers & Corvids	1.823	2.000	1.097	**Woodpeckers & Corvids**	1.715	1.831	1.068
Wren	2.952	3.972	1.345	**Wren**	3.408	3.733	1.095
Field Survey 1	**Surveyed Density (/ha)**	**Distance Sampling Density (/ha)**	**Ratio**	**Field Survey 2**	**Surveyed Density (/ha)**	**Distance Sampling Density (/ha)**	**Ratio**
Field Spp	0.999	1.042	1.043	**Field Spp**	0.883	1.118	1.266
Game, Doves & Corvids	0.711	0.765	1.075	**Game, Doves & Corvids**	0.999	1.330	1.331
Hedge Spp	0.860	2.495	2.902	**Hedge Spp**	1.506	3.908	2.595

5.4.5 Bird Indices

The bird indices varied both within and between habitat classes (Figure 5.6), supported by the statistically significant differences in the ANOVA (in Table S 5.13). In general, *Bird-Density*, *Spp_Priority*^{w}, *Spp_IRR*, and to a lesser extent *Spp_Rarity*, were notably lower in field rather than wood transects. ‘Wood Edge’ had consistently high values of the bird indices and ‘Field By_Wood’ was consistently low overall (Figure 5.6). Figure 5.6 and Figure 5.7 showed that the 'Field_high%_Hedge' transects often had comparable bird indices to the woodland transects. Furthermore, *Spp_Decline* was relatively higher in the 'Field_high%_Hedge' transects than all the other habitat classes, suggesting an effect of historic hedge decline (Figure 5.6). However, the statistically significant differences were only between 'Field_high%_Hedge' and both 'Field By_Wood' and 'Field_low%_Hedge', and between 'Field By_Wood' and the other habitat classes (except 'Field_low%_Hedge') indicating a detrimental effect of woodland edge on declining farmland birds (Table S 5.14).

Each habitat class in Cambridgeshire had a relatively high value of *Spp_Priority* in at least one transect ('Field By_Wood' to a lesser extent), with 'Field_high%_Hedge' and 'Wood Edge' having the highest values (Figure 5.6, Table S 5.11 and Table S 5.12). The weighted priority metric mirrors *Bird_Density* suggesting that the number or ranked value alone better represents *Spp_Priority,* as weighting by bird density favours more abundant, but lower priority bird species. Figure 5.6 showed that *Spp_Rarity* was also relatively high in at least one transect in each habitat class in the surveyed Cambridgeshire landscape, with a higher number of rare birds in the woodland. Contrary to this, *Spp_IRR* was highest in 'Wood Interior' transect 34 (0.63) followed by 'Wood Edge' transect 23 (0.54), and relatively low values in the fields (Figure 5.6 and Table S 5.12). Surprisingly, 'Field_low%_Hedge' transect 4 had the highest value of *Spp_IRR* of the field transects (0.23), suggesting the presence of a rare open habitat bird (Figure 5.6).

Overall the 'Wood Edge' transects had higher values of the bird indices than the 'Wood Interior' transects (Figure 5.6 and Figure 5.7). However, this is not apparent in the post-hoc Tukey test (in Table S 5.14), as there were no statistically significant differences (at $P < 0.05$) between the two woodland classes, except for $Spp\_Priority^{w}$, most likely as a result of the overlap (Figure 5.7). Therefore, analyses should be carried out at the transect level to distinguish aspects of the habitats responsible for the bird metrics. The high R^2 values show that for most of the bird indices, the habitat classes explain the variation. However, the lower R^2 values of *Spp_Decline, Spp_Priority* and *Spp_IRR* suggest that other factors may be influencing the variation, such as the habitat characteristics (Table S 5.13).

The surveyed transects in the individual woodlands were combined to allow comparisons between woodlands as a consequence of wood size and transect length. Figure 5.8 showed that total *Spp_Richness* and *Spp_Diversity* differed slightly between the woodlands, with a statistically significant difference between the woodlands (Table S 5.16). Figure 5.8 also showed that Gamsey, the smallest surveyed woodland (4.7 ha), had the highest *Spp_Richness per m of T_Length* and *per ha of Wood.Area* and the highest *Spp_Diversity per m of T_Length* and *per ha of Wood.Area*, closely followed by the second smallest woodland, Raveley (6.7 ha). Monks Wood, the largest woodland (170 ha in the current study), had the lowest values of both *Spp_Richness* and *Spp_Diversity per T_Length* and *per ha Wood.Area*, but had the greatest value of *Spp_Richness* and *Spp_Diversity* overall (Figure 5.8). This suggests that for their size, the smaller woodlands had a greater bird diversity and species richness. Figure S 5.1 showed that woodland size only increased *Spp_Richness* to a point, suggesting that there is a threshold of woodland size to maximise *Spp_Richness* until another factor becomes limiting creating a plateau. Figure S 5.1 indicates that woodlands of ~30 ha size were almost as good (in terms of numbers of species) as woodlands more than double that size, which may contain only one more bird species, depending on the type of woodland. Upton Wood is a similar size to Little Less Wood, but had lower *Spp_Richness*, possibly as a result of the vegetation composition containing a higher percentage of conifer.

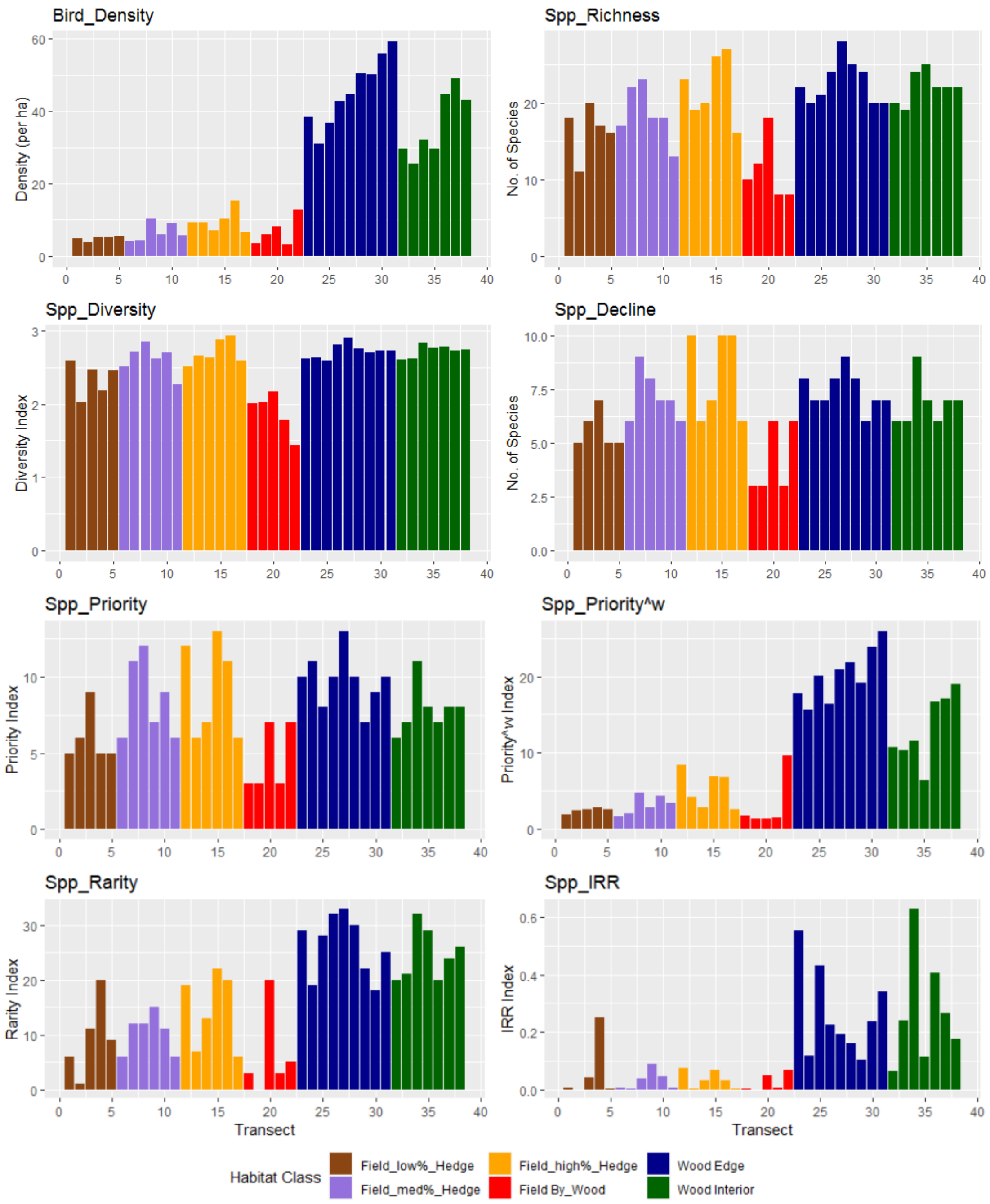


Figure 5.6. The eight calculated Bird Indices for each of the 38 survey transects in Cambridgeshire For calculation of metrics see Table 3.6 and for bird index values see Table S 5.11 and Table S 5.12(R version 3.5.2).

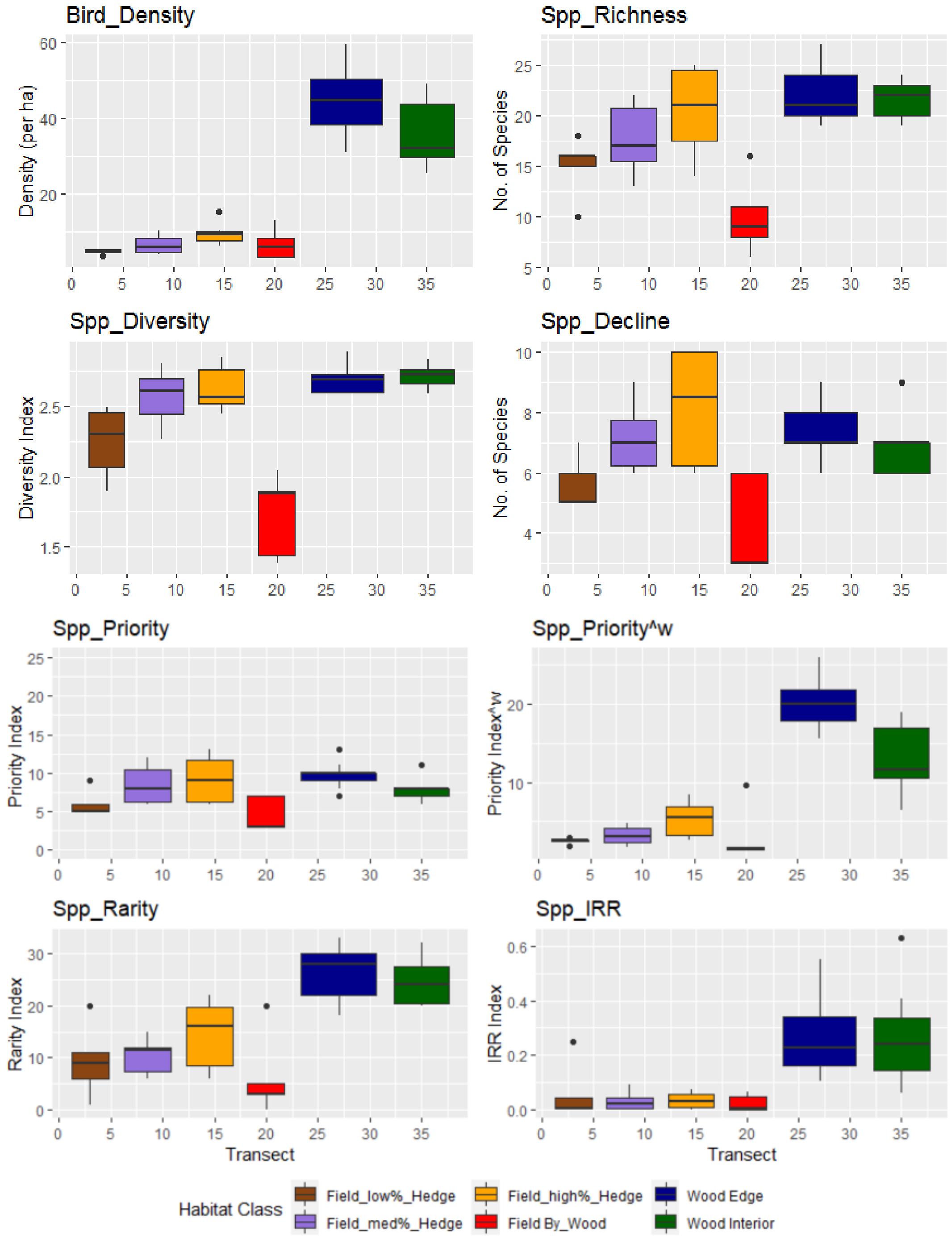


Figure 5.7. Variation in bird variables between habitat classes in Cambridgeshire. For calculation of metrics see Table 3.6 and for bird index values see Table S 5.11 and Table S 5.12 (R version 3.5.2).

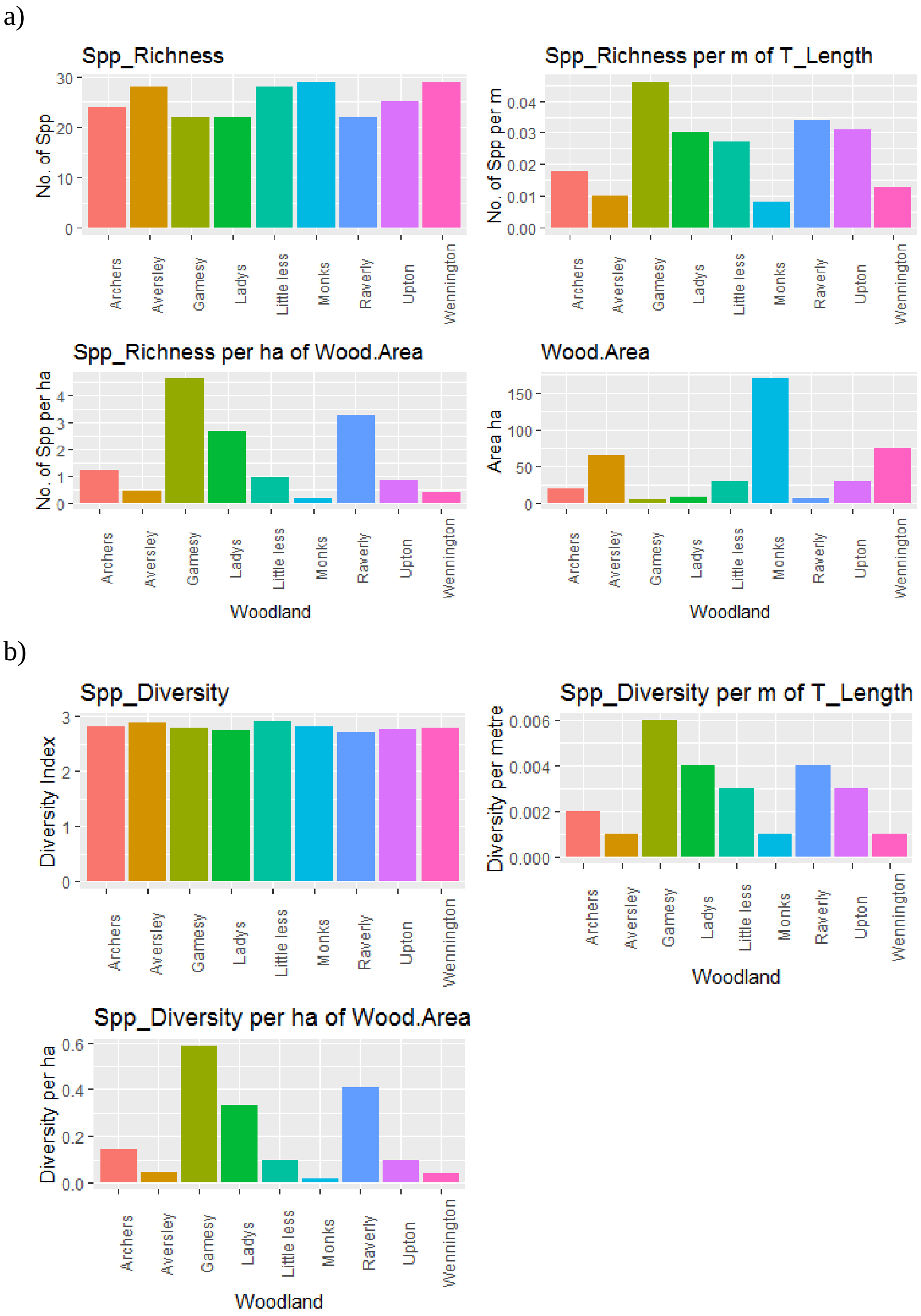


Figure 5.8. From the surveyed transects in each woodland in Cambridgeshire: a) total Spp_Richness, Spp_Richness per metre transect length and per hectare wood area; and b) total Spp_Diversity, Spp_Diversity per metre transect length and per hectare wood area. See Table S 5.15 for values (R version 3.5.2).

5.4.6 Bird-Habitat Relationships

The results of the univariate correlation matrix in Table 5.2 indicate that the majority of the bird indices, with the exception of *Spp_Decline* and *Spp_Priority*, were positively correlated with taller vegetation: *Ht_Av*, *Ostorey_Ht*, *PCov_2-5m* and *PCov_>5m*, and *PCov_Ash*, *PCov_Oak, PCov_Maple* (at P <0.05; N = 38). The correlation also showed that the majority of the bird indices were negatively related with low vegetation, *PCov_<0.5m* and *PCov_0.5-2m*, open canopy (*Ostorey_PenDepth*) and *PCov_ImpGrass*, *PCov_Cereal* and *PCov_CropOther*, indicating a preference for woodland (Table 5.2). *Bird_Density* and *Spp_Priority*^{w} were negatively correlated with *P_HedgeLen* (*r* = -0.52 & -0.46; P = 0.001 & 0.004, respectively) and *Ht_VDR* (*r* = -0.54; P = 0.000 for both), as a result of the higher abundance of birds in the woodland transects where there were no hedges and lower variation in vegetation structure (Table 5.2). Nonetheless, the majority of the bird indices were positively correlated with *PCov_2-5m*, presumably representing vegetation as understorey in the woodlands, or substantial hedges or copses in the fields (Table 5.2).

Spp_Decline was only positively correlated with *P_HedgeLen* (*r* = 0.37; P = 0.024), suggesting that birds with declining populations are more likely to be hedge specialists in the fields and possibly affected by historic hedge removal (Table 5.2). The regression plots in Supplementary Material 5.7 display the top two significant bird-habitat relationships with the highest Pearson's *r* values; moreover, the graphs also show that there was often a considerable split in the data, with the field transects more often than not having lower values than the wood transects. As a result, further analyses need to be carried out to investigate the habitats separately, as the woodland transects may be masking and altering any habitat relationships within the field transects, and vice versa. These results also suggest that the bird-habitat relationships are more complex than simple univariate correlations, and as such, a multi-variate analysis containing both vegetation structure and composition is required to explain them.

Table 5.2. Correlation matrix of Pearson's r coefficient of the bird variables against the habitat variables in Cambridgeshire, significant relationships in bold (at P <0.05), P values in brackets. For variable derivation see Table 3.2, Table 3.3, Table 3.4 and Table 3.6.

Variable	***Bird_Density***	***Spp_Richness***	***Spp_Diversity***	***Spp_Decline***	***Spp_Priority***	***Spp_Priority***^{w}	***Spp_Rarity***	***Spp_IRR***
PCov_Ash	**0.833 (0.000)**	**0.423 (0.008)**	**0.396 (0.014)**	0.231 (0.163)	0.288 (0.080)	**0.818 (0.000)**	**0.710 (0.000)**	**0.803 (0.000)**
PCov_Oak	**0.850 (0.000)**	**0.491 (0.002)**	**0.436 (0.006)**	0.254 (0.124)	0.301 (0.067)	**0.819 (0.000)**	**0.761 (0.000)**	**0.636 (0.000)**
PCov_Maple	**0.768 (0.000)**	**0.379 (0.019)**	**0.388 (0.016)**	0.154 (0.357)	0.192 (0.249)	**0.737 (0.000)**	**0.583 (0.000)**	**0.642 (0.000)**
PCov_Elm	**0.422 (0.008)**	0.217 (0.191)	0.220 (0.185)	0.036 (0.832)	0.030 (0.860)	**0.334 (0.040)**	0.249 (0.132)	0.315 (0.054)
PCov_BroadOther	0.227 (0.171)	0.273 (0.098)	0.298 (0.069)	0.175 (0.294)	0.259 (0.166)	0.229 (0.167)	0.242 (0.144)	0.178 (0.284)
PCov_Cons	0.079 (0.638)	0.194 (0.243)	0.119 (0.476)	0.032 (0.847)	0.011 (0.949)	-0.055 (0.743)	0.223 (0.178)	-0.017 (0.921)
Pres_Thorns	0.142 (0.394)	0.214 (0.197)	0.137 (0.412)	0.071 (0.673)	0.120 (0.473)	0.126 (0.452)	0.184 (0.268)	0.133 (0.426)
Pres_Bramble	-0.009 (0.958)	-0.126 (0.451)	-0.197 (0.235)	-0.079 (0.636)	-0.131 (0.432)	0.026 (0.878)	-0.070 (0.677)	-0.290 (0.077)
Pres_Shrub	0.235 (0.155)	-0.054 (0.747)	-0.117 (0.484)	-0.159 (0.340)	-0.079 (0.637)	0.211 (0.204)	0.073 (0.664)	0.147 (0.377)
PCov_ImpGrass	-0.228 (0.169)	**-0.483 (0.002)**	**-0.628 (0.000)**	-0.189 (0.256)	-0.152 (0.363)	-0.150 (0.370)	**-0.323 (0.048)**	-0.188 (0.259)
PCov_Cereal	**-0.666 (0.000)**	**-0.323 (0.048)**	-0.221 (0.183)	-0.138 (0.409)	-0.249 (0.132)	**-0.638 (0.000)**	**-0.563 (0.000)**	**-0.489 (0.002)**
PCov_CropOther	**-0.555 (0.000)**	-0.179 (0.283)	-0.123 (0.462)	-0.143 (0.393)	-0.177 (0.288)	**-0.561 (0.000)**	**-0.377 (0.020)**	**-0.418 (0.009)**
P_HedgeLen	**-0.516 (0.001)**	0.127 (0.449)	0.169 (0.310)	**0.365 (0.024)**	0.194 (0.244)	**-0.456 (0.004)**	-0.276 (0.093)	**-0.445 (0.005)**
Ht_Av	**0.861 (0.000)**	**0.477 (0.002)**	**0.449 (0.005)**	0.253 (0.125)	0.292 (0.075)	**0.813 (0.000)**	**0.774 (0.000)**	**0.860 (0.000)**
Ht_StDev	**0.877 (0.000)**	**0.574 (0.000)**	**0.526 (0.001)**	0.235 (0.156)	0.299 (0.068)	**0.793 (0.000)**	**0.752 (0.000)**	**0.639 (0.000)**
Ht_VDR	**-0.540 (0.000)**	-0.030 (0.860)	-0.062 (0.713)	-0.066 (0.696)	-0.094 (0.575)	**-0.541 (0.000)**	**-0.398 (0.013)**	**-0.624 (0.000)**
Ostorey_Ht	**0.757(0.000)**	**0.436 (0.006)**	**0.363 (0.025)**	0.212 (0.202)	0.255 (0.122)	**0.699 (0.000)**	**0.726 (0.000)**	**0.794 (0.000)**
Ostorey_PenDepth	**-0.562 (0.000)**	-0.317 (0.053)	-0.210 (0.206)	-0.146 (0.382)	-0.128 (0.443)	**-0.530 (0.001)**	**-0.588 (0.000)**	**-0.526 (0.001)**
PCov_<0.5m	**-0.746 (0.000)**	**-0.451 (0.004)**	**-0.443 (0.005)**	-0.243 (0.141)	-0.283 (0.085)	**-0.729 (0.000)**	**-0.643 (0.000)**	**-0.642 (0.000)**
PCov_0.5-2m	**-0.506 (0.001)**	-0.249 (0.132)	-0.209 (0.208)	-0.090 (0.592)	-0.139 (0.404)	**-0.448 (0.005)**	**-0.426 (0.008)**	**-0.413 (0.010)**
PCov_2-5m	**0.510 (0.001)**	**0.407 (0.011)**	**0.401 (0.013)**	0.264 (0.109)	**0.394 (0.015)**	**0.531 (0.001)**	**0.459 (0.004)**	0.273 (0.097)
PCov_>5m	**0.909 (0.000)**	**0.500 (0.001)**	**0.470 (0.003)**	0.237 (0.152)	0.284 (0.084)	**0.855 (0.000)**	**0.774 (0.000)**	**0.793 (0.000)**

5.4.7 Bird-Habitat Relationships – Field Only Analysis

Bird-habitat relationships for the field transects alone were investigated and presented in the correlation matrix in Table 5.3. Separating the field transects from the wood transects resulted in additional significant univariate relationships, which alluded to more detailed bird-habitat relationships than merely preferring woody vegetation. The correlation for the field only transects showed that *P_HedgeLen* was positively related with all the bird indices, except *Spp_IRR*, indicating the value of hedges as a resource for birds in open agricultural areas. The positive relationship of *Bird_Density, Spp_Richness, Spp_Diversity, Spp_Decline* and *Spp_Priority* with *PCov_2-5m*, suggests that hedges and substantial woody vegetation provides habitat for more birds in the fields (Table 5.3). Consequently, the majority of the bird indices were also positively correlated with the vegetation composition variables. *Bird_Density* was positively correlated with *PCov_Elm* and *PCov_BroadOther* (r = 0.50 & 0.54; P = 0.017 & 0.010, respectively), as was *Spp_Richness* (r = 0.60 & 0.44; P = 0.003 & 0.038, respectively). *Spp_Richness* was also correlated with *PCov_Maple* (r = 0.61; P = 0.003), as was *Spp_Decline* (r = 0.45; P = 0.035; Table 5.3). *Spp_Diversity* and *Spp_Priority* were both positively correlated with *PCov_Ash*, *PCov_Maple* and *PCov_Elm*, indicating that birds benefit from woody vegetation (including trees) in an agricultural landscape (Table 5.3). *Spp_Richness* and *Spp_Diversity* were also negatively correlated with *PCov_ImpGrass* (r = -0.45 & -0.60; P = 0.038 & 0.003, respectively), indicating a detrimental effect on bird diversity and species richness. The positive relationship of *Bird_Density* and *Spp_Richness* with *PCov_BroadOther* also suggests that increasing vegetation diversity increases bird density and species richness.

Spp_Decline was only positively correlated with *PCov_Maple, P_HedgeLen* and *PCov_2-5m*, further indicating the effect of historic hedgerow removal on declining bird species (Table 5.3). Furthermore, *Spp_Rarity* and $Spp\_Priority^{w}$ were both only positively correlated with *P_HedgeLen* (r = 0.53 & 0.50; P = 0.0.12 & 0.019, respectively), suggesting that rare and priority bird species were also hedge specialists. In contrast, *Spp_IRR* had no significant relationships with any of the habitat variables, suggesting a lack of data (as only one field transect had a high *Spp_IRR*), or the result of another undetermined variable being important.

Table 5.3. Correlation matrix of Pearson's r coefficient for the bird indices against the habitat variables in the field only transects in Cambridgeshire, significant relationships in bold (P <0.05), P values in brackets. Variable derivation in Table 3.2, Table 3.3, Table 3.4 and Table 3.6.

Variable	***Bird_Density***	***Spp_Richness***	***Spp_Diversity***	***Spp_Decline***	***Spp_Priority***	***Spp_Priority***^{w}	***Spp_Rarity***	***Spp_IRR***
PCov_Ash	0.235 (0.292)	0.423 (0.050)	**0.434 (0.044)**	0.328 (0.136)	**0.449 (0.036)**	0.198 (0.378)	0.108 (0.633)	-0.158 (0.483)
PCov_Oak	0.355 (0.105)	0.352 (0.108)	0.255 (0.253)	0.219 (0.328)	0.136 (0.547)	0.183 (0.416)	0.259 (0.244)	-0.110 (0.625)
PCov_Maple	0.348 (0.113)	**0.606 (0.003)**	**0.558 (0.007)**	**0.451 (0.035)**	**0.456 (0.033)**	0.305 (0.167)	0.417 (0.054)	-0.018 (0.937)
PCov_Elm	**0.502 (0.017)**	**0.599 (0.003)**	**0.595 (0.003)**	0.415 (0.055)	**0.512 (0.015)**	0.287 (0.195)	0.297 (0.180)	-0.020 (0.928)
PCov_BroadOther	**0.537 (0.010)**	**0.444 (0.038)**	0.413 (0.056)	0.261 (0.240)	0.213 (0.341)	0.244 (0.273)	0.162 (0.470)	-0.191 (0.396)
PCov_Cons	0.064 (0.778)	0.339 (0.122)	0.216 (0.335)	0.312 (0.157)	0.395 (0.069)	0.144 (0.522)	0.284 (0.200)	0.100 (0.656)
Pres_Thorns	0.094 (0.676)	0.180 (0.424)	0.083 (0.713)	0.044 (0.846)	0.091 (0.687)	0.015 (0.948)	0.144 (0.523)	0.129 (0.566)
Pres_Bramble	-0.063 (0.78)	-0.180 (0.424)	-0.267 (0.229)	-0.079 (0.727)	-0.244 (0.273)	0.002 (0.992)	-0.031 (0.891)	0.038 (0.866)
Pres_Shrub	0.028 (0.901)	-0.234 (0.294)	-0.280 (0.208)	-0.245 (0.271)	-0.195 (0.385)	-0.033 (0.886)	-0.243 (0.275)	-0.121 (0.591)
PCov_ImpGrass	0.220 (0.326)	**-0.445 (0.038)**	**-0.599 (0.003)**	-0.146 (0.516)	-0.080 (0.723)	0.408 (0.060)	-0.214 (0.338)	0.032 (0.888)
PCov_Cereal	-0.197 (0.379)	0.058 (0.799)	0.172 (0.445)	0.029 (0.899)	-0.046 (0.839)	-0.178 (0.428)	-0.043 (0.850)	0.039 (0.863)
PCov_CropOther	-0.150 (0.506)	0.181 (0.420)	0.212 (0.343)	-0.017 (0.941)	0.010 (0.964)	-0.290 (0.190)	0.185 (0.409)	-0.022 (0.924)
P_HedgeLen	**0.639 (0.001)**	**0.713 (0.000)**	**0.685 (0.000)**	**0.715 (0.000)**	**0.598 (0.003)**	**0.496 (0.019)**	**0.526 (0.012)**	0.003 (0.988)
Ht_Av	0.255 (0.252)	0.367 (0.093)	0.382 (0.080)	0.209 (0.351)	0.206 (0.357)	0.240 (0.282)	0.097 (0.668)	-0.144 (0.522)
Ht_StDev	0.227 (0.311)	0.342 (0.119)	0.348 (0.113)	0.110 (0.625)	0.138 (0.540)	0.101 (0.656)	0.047 (0.834)	-0.250 (0.261)
Ht_VDR	0.252 (0.258)	**0.430 (0.046)**	0.337 (0.125)	0.177 (0.432)	0.190 (0.398)	0.076 (0.737)	0.244 (0.273)	-0.021 (0.928)
Ostorey_Ht	0.102 (0.653)	0.060 (0.792)	-0.062 (0.784)	-0.043 (0.850)	0.010 (0.963)	0.062 (0.785)	0.116 (0.609)	-0.134 (0.552)
Ostorey_PenDepth	0.013 (0.955)	0.069 (0.761)	0.169 (0.452)	-0.015 (0.947)	0.079 (0.725)	-0.136 (0.546)	-0.227 (0.309)	-0.381 (0.080)
PCov_<0.5m	-0.028 (0.901)	-0.116 (0.606)	-0.137 (0.544)	-0.102 (0.652)	-0.089 (0.695)	-0.205 (0.361)	-0.003 (0.989)	0.056 (0.806)
PCov_0.5-2m	-0.068 (0.764)	0.023 (0.920)	0.049 (0.828)	0.043 (0.849)	0.033 (0.883)	0.163 (0.468)	-0.036 (0.874)	-0.022 (0.923)
PCov_2-5m	**0.662 (0.001)**	**0.620 (0.002)**	**0.549 (0.008)**	**0.537 (0.010)**	**0.445 (0.038)**	0.364 (0.096)	0.394 (0.070)	-0.067 (0.768)
PCov_>5m	0.329 (0.135)	0.369 (0.091)	0.374 (0.086)	0.136 (0.547)	0.167 (0.456)	0.148 (0.510)	0.042 (0.855)	-0.248 (0.267)

5.4.8 Bird-Habitat Relationships – Woodland Only Analysis

The correlation matrix for the woodland only analysis in Table 5.4 displayed significant relationships for only four bird indices with the habitat variables (at $P < 0.05$). Firstly, *Spp_Richness* and *Spp_Diversity* were negatively correlated with *Wood.Area* (r = -0.66 and -0.69; P = 0.005, 0.003, respectively), and *Spp_Diversity* was also negatively correlated with *Wood.Perimeter* (r = -0.56; P = 0.025) and positively correlated with *Wood_P:A* (r = 0.58; P = 0.019). This suggests that smaller woodlands contained more bird species, possibly as a result of an edge effect over the entire woodland, i.e. small woods are essentially all edge. The non-significant relationships with *%_Woods_500m* and *%_Woods_1km* could also suggest that the woodlands were sufficiently interconnected such that the size of the woodland was immaterial (Table 5.4).

Spp_Priority^{w} was negatively correlated with *PCov_Cons* (r = -0.55; P = 0.029), suggesting a detrimental effect of conifers on the abundance of priority bird species (Table 5.4). The remaining significant relationships in Table 5.4 were with *Spp_IRR*, which was positively correlated with *PCov_Ash*, *Ht_Av*, *Ostorey_Ht* and *PCov_>5m* (r = 0.57, 0.81, 0.75 and 0.59; P = 0.020, 0.000, 0.001, 0.016, respectively), and negatively correlated with *Pres_Bramble*, *Ht_VDR* and *PCov_<0.5m* (r = -0.53, -0.77 and -0.64; P = 0.035, 0.000, 0.008, respectively). This details that in the study woodlands, rare bird species prefer tall, dense ash woodlands with uniform structure and no shrub vegetation layer. The relationship of *Spp_IRR* to *PCov_Ash* could also suggest an effect of ash dieback in the Cambridgeshire woodlands. The absence of significant bird-habitat relationships in the woodland transects, could also suggest that the variables of vegetation composition and structure were not influencing variation in the bird indices. This may suggest that they contained the same bird assemblages, other indeterminate environmental factors were affecting the birds, or the small woodland sample size (due to the nature of the landscape and accessibility issues) prevented significant habitat relationships being detected.

It should be noted that the field variables (*P_HedgeLen*, *PCov_CropOther* etc.) were excluded from this analysis as they are not present in the woodlands. The NA for both *Pres_Thorns* and *Pres_Shrub* in Table 5.4 results from presence in all the woodland transects (Table S 5.4).

Table 5.4. Correlation matrix of Pearson's r coefficient for the bird variables against the habitat variables with extra wood variables in the woodland transects only in Cambridgeshire, significant relationships in bold (at P <0.05), P values in brackets. For variable derivation see Table 3.2, Table 3.3, Table 3.4, Table 3.5 and Table 3.6.

Variables	***Bird_Density***	***Spp_Richness***	***Spp_Diversity***	***Spp_Decline***	***Spp_Priority***	***Spp_Priority***^{w}	***Spp_Rarity***	***Spp_IRR***
PCov_Ash	0.166 (0.539)	-0.176 (0.515)	-0.220 (0.413)	0.164 (0.545)	0.022 (0.935)	0.286 (0.283)	0.174 (0.519)	**0.574 (0.020)**
PCov_Oak	0.251 (0.348)	0.254 (0.343)	0.261 (0.329)	0.297 (0.264)	0.102 (0.706)	0.283 (0.289)	0.459 (0.074)	0.058 (0.831)
PCov_Maple	0.205 (0.447)	-0.177 (0.512)	0.065 (0.810)	-0.152 (0.574)	-0.242 (0.367)	0.209 (0.438)	-0.164 (0.545)	0.217 (0.420)
PCov_Elm	0.222 (0.409)	-0.010 (0.972)	0.085 (0.754)	-0.212 (0.431)	-0.279 (0.295)	0.010 (0.971)	-0.175 (0.517)	0.081 (0.765)
PCov_BroadOther	-0.441 (0.087)	-0.169 (0.531)	-0.037 (0.891)	0.015 (0.955)	0.208 (0.440)	-0.283 (0.288)	-0.248 (0.355)	-0.102 (0.706)
PCov_Cons	-0.314 (0.0236)	0.295 (0.268)	0.142 (0.599)	-0.041 (0.880)	-0.127 (0.640)	**-0.545 (0.029)**	0.200 (0.458)	-0.250 (0.351)
Pres_Thorns	NA	NA	NA	NA	NA	NA	NA	NA
Pres_Bramble	0.308 (0.246)	0.091 (0.799)	0.069 (0.799)	-0.038 (0.889)	0.139 (0.608)	0.328 (0.214)	0.029 (0.914)	**-0.530 (0.035)**
Pres_Shrub	NA	NA	NA	NA	NA	NA	NA	NA
Ht_Av	0.050 (0.855)	-0.070 (0.798)	-0.009 (0.975)	0.259 (0.332)	-0.046 (0.865)	0.056 (0.837)	0.386 (0.140)	**0.812 (0.000)**
Ht_StDev	0.173 (0.521)	0.350 (0.184)	0.304 (0.252)	0.024 (0.930)	-0.165 (0.541)	-0.104 (0.702)	0.263 (0.326)	0.018 (0.948)
Ht_VDR	-0.346 (0.189)	0.058 (0.831)	-0.117 (0.666)	-0.332 (0.209)	-0.159 (0.557)	-0.388 (0.138)	-0.380 (0.146)	**-0.773 (0.000)**
Ostorey_Ht	-0.011 (0.968)	0.029 (0.916)	0.005 (0.985)	0.256 (0.338)	-0.026 (0.923)	-0.072 (0.792)	0.431 (0.095)	**0.747 (0.001)**
Ostorey_PenDepth	0.293 (0.270)	-0.275 (0.302)	-0.194 (0.471)	0.067 (0.805)	0.203 (0.451)	0.374 (0.153)	-0.017 (0.951)	0.003 (0.992)
PCov_<0.5m	-0.080 (0.769)	0.139 (0.607)	-0.052 (0.849)	-0.167 (0.537)	0.129 (0.634)	-0.092 (0.733)	-0.261 (0.330)	**-0.638 (0.008)**
PCov_0.5-2m	-0.350 (0.184)	0.001 (0.996)	-0.083 (0.761)	-0.118 (0.662)	0.011 (0.967)	-0.434 (0.093)	-0.093 (0.733)	-0.401 (0.124)
PCov_2-5m	-0.329 (0.214)	-0.228 (0.395)	-0.216 (0.421)	-0.065 (0.811)	0.275 (0.303)	-0.089 (0.743)	-0.335 (0.205)	-0.329 (0.213)
PCov_>5m	0.256 (0.339)	-0.004 (0.989)	0.126 (0.643)	0.151 (0.576)	-0.169 (0.532)	0.207 (0.441)	0.288 (0.280)	**0.591 (0.016)**
Wood.Area	-0.273 (0.307)	**-0.664 (0.005)**	**-0.692 (0.003)**	-0.053 (0.846)	-0.214 (0.427)	-0.008 (0.978)	-0.337 (0.203)	0.012 (0.536)
Wood.Perimeter	-0.198 (0.462)	-0.449 (0.081)	**-0.559 (0.025)**	0.043 (0.875)	-0.041 (0.881)	0.154 (0.569)	-0.174 (0.518)	0.012 (0.676)
Wood_P:A	0.260 (0.331)	0.423 (0.103)	**0.579 (0.019)**	0.054 (0.843)	0.016 (0.952)	-0.080 (0.767)	0.108 (0.691)	-0.027 (0.811)
%_Woods_500m	0.028 (0.918)	0.139 (0.608)	0.256 (0.339)	-0.175 (0.517)	-0.314 (0.236)	-0.291 (0.274)	-0.143 (0.597)	0.011 (0.768)
%_Woods_1km	-0.045 (0.868)	0.376 (0.152)	0.420 (0.106)	-0.315 (0.234)	-0.021 (0.939)	-0.090 (0.740)	0.232 (0.387)	0.029 (0.476)
Dist_Wood	-0.251 (0.349)	-0.139 (0.607)	-0.089 (0.744)	-0.114 (0.674)	0.092 (0.735)	-0.131 (0.629)	-0.046 (0.865)	-0.187 (0.740)

5.4.9 Multivariate Bird-Habitat Relationships

5.4.9.1 Density, Species Richness and Diversity

Bird_Density had six top models with an accumulated weight of 1.01 from which inference was made (Table S 5.19a). *PCov_2-5m* was the most important variable included in all four models (Table 5.5a). *PCov_Ash* had the highest estimate, and therefore effect size, followed by *PCov_>5m*, *PCov_2-5m*, *PCov_Oak* and *PCov_Elm* all with positive estimates, and thus positive relationships with *Bird_Density* indicating higher bird density in the woodlands (Table 5.5a). The most important variable for both *Spp_Richness* and *Spp_Diversity* was *PCov_ImpGrass* which had a negative estimate in both instances (Table 5.5b & c). Both bird indices were also positively related to *Ht_StDev, PCov_2-5m, PCov_Oak, PCov_Ash* and *PCov_Maple*. This demonstrates that *Spp_Diversity* and *Spp_Richness* were higher in the woodlands, especially with a varied canopy height, and lower in the fields (Table 5.5b & c). *Spp_Diversity* was also positively related to *PCov_BroadOther* as a result of tree species diversity increasing bird diversity. However, the confidence intervals for *PCov_2-5m* contained zero (Table 5.5), which may indicate a weak relationship and possibly suggests that the hedges in the field habitat, possibly at this height range, were perhaps contributing to this relationship.

As a result, the habitats (field and wood) were separated for further analysis and, as expected, showed a positive relationship of *Spp_Richness* and *Spp_Diversity* to *P_HedgeLen* in the field only analysis (Table 5.6b & c). Moreover, all of the bird indices, except *Spp_IRR*, had a positive relationship with *P_HedgeLen* in the field habitat, highlighting the importance of hedges in an agricultural environment for birds (Table 5.6). As well as being positively related to *P_HedgeLen*, both *Spp_Richness* and *Spp_Diversity* were again negatively related to *PCov_ImpGrass* suggesting a detrimental effect of improved grass in an agricultural environment (Table 5.6b & c).

In the woodland analysis, there were 27 models in the candidate set for *Bird_Density*, with the null model as the forth top model and the remainder having weak model weights and very low R^2 values (Table S 5.21a). Consequently, the confidence intervals for all the predictor variables contained zero (Table 5.7a), suggesting that inference on these variables should be made with caution, and that the top two AICc models may be more appropriate. Nonetheless, the negative relationship with *PCov_BroadOther* and *PCov_Cons* indicates that woodlands containing more broadleaved species and conifer were more likely to have lower *Bird_Density*. However, the high *Bird_Density* in 'Wood Interior' plots 36-38 is unexplained, especially as 'Wood Interior' transect 36 had a high *PCov_BroadOther*. The negative relationship with *Wood.Area* and *Dist_Wood* (Table 5.7a) may be a consequence of the largest wood, Monks Wood, having a greater area than nearby woodlands and a lower density due to the greater amount of interior space available. The wood only analysis showed no support for the best model to contain any of the selected variables for *Spp_Richness* (Table S 5.21b), suggesting that the variation of *Spp_Richness* in the woodlands was caused by factors not included in this analysis. Additionally, the woodland only analysis, showed that five out of the eight bird indices contained the Null model, often as the best model, in the candidate set (Table S 5.21), indicating that there were no distinguishing features of the habitat creating the differences in the bird indices. However, Figure 5.6 clearly exhibited differences in the bird indices between and within the woodland transect classes. Nonetheless, the extra woodland variables included in the analysis, showed that *Wood.Area* was negatively related to *Bird_Density, Spp_Richness* and *Spp_Diversity*, suggesting that smaller woods have higher values of these variables (Table 5.7). As mentioned above, this apparently counterintuitive result could be attributable to the structure of the smaller woodlands as essentially being all edge habitat, contrasting with the more open habitat characteristic of the interiors of larger woods. Figure S 5.1 supports this, as the larger woodlands overall had higher *Spp_Richness* when the woodland transects were combined. However, increasing woodland size only increases *Spp_Richness* to a point, then levels off (Figure S 5.1) suggesting that another limiting factor, perhaps most likely the size of the regional species pool, exists after this point.

5.4.9.2 Declining and Priority Species

In the landscape level multi-model analysis, the top models for *Spp_Decline* had correlated variables of *PCov_Ash* and *PCov_Oak* with *P_HedgeLen*. When the correlated variables were removed the best model contained *P_HedgeLen* and *PCov_<0.5m,* followed by the Null model, suggesting that there may be conflicting factors leading to *Spp_Decline* in the two habitats separately; hedge in the field and ash/oak in the woodland (Table S 5.19d). When analysed separately the results showed that the bird species in this landscape were suffering more from a decline in hedgerows than any given aspect of the woodland. This was demonstrated by the Null model being the best model for the wood transects alone (Table S 5.21d), and *P_HedgeLen* was the only variable in the top model and was the most important variable for the field only analysis (Table S 5.20d). The positive relationship of *P_HedgeLen* with *Spp_Decline* in the fields, suggests that historic hedgerow removal for agricultural intensification in the Cambridgeshire landscape may still be contributing to bird population declines (Table 5.6d). *Spp_Decline* was also positively related to *PCov_Cons*, although to a lesser extent, suggesting a conifer specialist may also exist in the agricultural habitat. The positive relationships of *PCov_2-5m*, *PCov_Maple* and *PCov_Elm* suggest that declining species are also supported in shrubby trees of elm and field maple. However, the confidence intervals contain zero, so these variables should be interpreted with caution (Table 5.6d). Although the top woodland model for *Spp_Decline* was the null model, the variables of *Ht_VDR*, *%_Woods_500m* and *PCov_Oak* in the remaining models may indicate that smaller, isolated oak woodlands also support or contain declining bird species.

In the landscape level study, *Spp_Priority* had a top model with only *PCov_2-5m* with a positive estimate. The top six models in the candidate set all contained *PCov_2-5m* and positive estimates of *PCov_Oak*, *PCov_Ash*, *Ht_Av* and *Ht_StDev* in turn, and the sixth model had a negative estimate of *PCov_<0.5m*, suggesting that priority bird populations were in woodlands with a varied height (Table 5.5e). The rest of the models contained the latter variables singularly and all had low model weights, very low Pseudo-R^2 and similar variable importance, suggesting that the variables are interchangeable, and therefore *PCov_2-5m* was the influencing factor (Table S 5.19e and Table 5.5e). Furthermore, the low Pseudo-R^2 values indicate a poor fit for all the models (Table S 5.19e), which suggests conflicting factors may exist in the wood and field habitats for priority species. Once again the Null model was the top model for the woodland analysis (Table S 5.21e), although the negative relationship of both *Spp_Priority* and *Spp_Decline* to *%_Woods_500m* may indicate that there are more birds with declining populations present in isolated woodlands (Table 5.7d & e). Similar to *Spp_Decline*, the field habitat analysis showed a positive relationship of *Spp_Priority* with *P_HedgeLen*, *PCov_2-5m*, *PCov_Maple*, *PCov_Elm* and *PCov_Cons* (Table 5.6e) further indicating the importance of hedges in an agricultural environment and also suggesting priority species in fields with conifer. However, again the confidence intervals of the latter four variables including *PCov_Cons* contained zero, indicating that they may not have an effect (Table 5.6d & e), but could be the result of one transect with conifer present.

Spp_Priority^{w} was weighted by *Bird_Density*, and as a result, showed similar relationships in the landscape scale analysis, indicating a preference for woodlands with understorey (Table 5.5a & f). The field transect analysis had three top models with positive estimates to both *P_HedgeLen*, *PCov_ImpGrass* and *PCov_Maple* (Table 5.6f), suggesting that there were numerous priority species, possibly with less of a population decline, in the hedges with field maple and in improved grassland. In the woodland analysis *Spp_Priority*^{w} was positively related to *Pres_Bramble*, *PCov_Oak* and *Ostorey_PenDepth*, and negatively related to *PCov_Cons*, *PCov_BroadOther* and *%_Woods_1km* similar to the *Bird_Density* results (Table 5.7a & f). However, *PCov_Cons* was more important in the *Spp_Priority*^{w} analysis, suggesting that there were considerably fewer priority species in woodlands with conifer (Table 5.7f).

5.4.9.3 Rarity

Spp_Rarity was positively related to *PCov_Oak* and *PCov_2-5m* and negatively related to *PCov_ImpGrass* in the landscape level analysis (Table 5.5g). This suggests that there were more rare bird species in the woodlands, particularly with oak, possibly with an understorey layer or in hedges in the fields (Table 5.5g). Once separated, it is evident that *Spp_Rarity* was positively related to *P_HedgeLen*, *PCov_Cons*, *PCov_Maple* and *PCov_2-5m* in the field only analysis, indicating that rare birds were also in the conifer as well as the hedges, although again, the confidence intervals for the latter three contain zero, therefore they may not affect *Spp_Rarity* (Table 5.6g). The woodland analysis showed a positive relationship with *PCov_Oak* and *Ostorey_Ht*, and a negative relationship with *Wood.Area* (Table 5.7g). This suggests that rare bird species are more likely to be in smaller, oak woodlands with a taller overstorey, however, all three variables should be taken with caution due to the confidence intervals containing zero (Table 5.7g).

The landscape level analysis in Table S 5.19h showed that seven top models to six ΔAICc represented *Spp_IRR* with a Pseudo-R^2 of around 0.70 for all, indicating a good model fit. The model coefficients were positive estimates of *Ht_Av*, *PCov_2-*

5m and *PCov_Elm* and a negative estimate of *PCov_0.5-2m* suggesting that several rare bird species were present in the woodland habitats rather than the agricultural areas, possibly with an understorey vegetation layer (Table 5.5h). This was supported by the separated results, as the top model for *Spp_IRR* in the field transects analysis was the Null model, and the remaining models in the candidate set had a low Pseudo-R^2 (max = 0.17), indicating that none of the habitat variables suitably explained the variation in *Spp_IRR* in the fields (Table S 5.20h). The coefficients in Table 5.6h showed spurious results with *PCov_Cons* and *Pres_Thorns* as the only positive coefficients, however, none of the variables may have an effect on *Spp_IRR* due to all the confidence intervals containing zero. Furthermore, 'Field_low%_Hedge' transect 4 with the highest *Spp_IRR* of the field transects does not contain conifer, but does contain hawthorn and blackthorn, although so do many of the field transects. The negative relationship with *Ostorey_PenDepth* is possibly as a result of the high *Spp_IRR* and low *Ostorey_PenDepth* in 'Field_low%_Hedge' transect 4, yet again, this should be inferred with caution. As expected, the results in Table S 5.21h showed six top models to explain *Spp_IRR* in the wood transects with a substantially higher Pseudo-R^2 of 0.76 for the top model, possibly as a result of several rare bird species in the woodlands. *Ht_Av* had a positive estimate, and *Ht_VDR* and *Pres_Bramble* were negative (Table 5.7h), suggesting a preference and a specialisation to mature, tall woodland with little structural variation and no shrubby ground layer, possibly as wood interior rather than wood edge. Less reliably, there was a positive relationship to *PCov_<0.5m* and a negative relationship to *PCov_Cons*, possibly indicating that rare species require open ground vegetation in woodlands without conifer.

Table 5.5. Relative importance and model averaged coefficients (across the 95% confidence set) of standardised predictors for bird indices in relation to habitat composition and structure at the landscape scale, based on the unconditional model average from the top six AICc generalised linear models (Table S 5.19). See Table 3.2, Table 3.3 and Table 3.4 for parameter explanations, and Table 3.6 for bird indices. (Negative coefficients in italics; z. and rescale = standardised parameters).

a) Bird_Density

Predictor	**N Containing Models**	**Relative Importance**	**Estimate**	**Adjusted SE**	**Lower CI**	**Upper CI**
(Intercept)	-	-	21.711	1.762	18.257	25.165
z.PCov_2-5m	6	1	7.401	3.489	0.564	14.239
z.PCov_Elm	3	0.52	4.401	6.545	-5.226	21.994
z.PCov_>5m	2	0.40	12.051	15.173	21.408	39.504
z.PCov_Ash	2	0.39	12.679	16.260	21.760	43.427
z.PCov_Oak	2	0.22	6.593	12.828	20.155	41.087

b) Spp_Richness

Predictor	**N Containing Models**	**Relative Importance**	**Estimate**	**Adjusted SE**	**Lower CI**	**Upper CI**
(Intercept)	-	-	19.421	0.741	17.969	20.873
z.PCov_ImpGrass	9	0.95	-3.691	1.456	-6.235	-1.571
z.Ht_StDev	4	0.57	2.518	2.557	0.953	7.865
z.PCov_2-5m	6	0.43	1.105	1.798	-1.250	6.359
z.PCov_Oak	2	0.22	0.773	1.678	0.191	6.975
z.PCov_Ash	2	0.09	0.268	0.993	-0.550	6.308
z.PCov_Maple	2	0.06	0.139	0.703	-1.110	5.834

c) Spp_Diversity

Predictor	**N Containing Models**	**Relative Importance**	**Estimate**	**Adjusted SE**	**Lower CI**	**Upper CI**
(Intercept)	-	-	2.541	0.042	2.459	2.624
z.PCov_ImpGrass	18	1	-0.389	0.064	-0.515	-0.263
z.Ht_StDev	3	0.36	0.075	0.116	0.018	0.399
z.PCov_2-5m	6	0.38	0.061	0.102	-0.051	0.373
z.PCov_BroadOther	6	0.21	0.022	0.061	-0.086	0.292
z.PCov_Oak	3	0.17	0.029	0.076	-0.021	0.362
z.PCov_<0.5m	3	0.13	-0.020	0.062	-0.341	0.032
z.PCov_Ash	3	0.11	0.017	0.058	-0.044	0.343
z.PCov_Maple	3	0.10	0.015	0.054	-0.053	0.339

d) Spp_Decline

Predictor	N Containing Models	Relative Importance	Estimate	Adjusted SE	Lower CI	Upper CI
(Intercept)	-	-	6.737	0.436	5.883	7.591
z.P_HedgeLen	4	0.62	1.062	1.175	-0.347	3.780
z.PCov_<0.5m	4	0.46	-0.653	0.991	-3.420	0.605
z.PCov_2-5m	4	0.37	0.427	0.852	-0.922	3.231

e) Spp_Priority

Predictor	N Containing Models	Relative Importance	Estimate	Adjusted SE	Lower CI	Upper CI
(Intercept)	-	-	7.947	0.474	7.019	8.875
z.PCov_2-5m	6	1	2.163	1.223	-0.235	4.561
z.PCov_Oak	1	0.14	0.130	0.526	-1.259	3.087
z.PCov_Ash	1	0.14	0.127	0.520	-1.261	3.060
z.Ht_Av	1	0.14	0.123	0.512	-1.267	3.030
z.Ht_StDev	1	0.13	0.103	0.483	-1.388	2.995
z.PCov_<0.5m	1	0.13	-0.098	0.459	-2.851	1.319

f) Spp_Priority^{w}

Predictor	N Containing Models	Relative Importance	Estimate	Adjusted SE	Lower CI	Upper CI
(Intercept)	-	-	9.309	1.101	7.151	11.467
z.PCov_2-5m	5	0.94	3.568	2.686	-1.318	8.911
z.PCov_>5m	3	0.53	6.202	6.303	5.743	17.855
z.PCov_Oak	3	0.47	5.984	6.712	6.012	19.216
z.PCov_Elm	2	0.22	0.333	1.955	-6.266	9.359
z.PCov_ 0.5-2m	1	0.18	0.269	0.678	-0.228	3.162

g) Spp_Rarity

Predictor	N Containing Models	Relative Importance	Estimate	Adjusted SE	Lower CI	Upper CI
(Intercept)	-	-	16.868	1.411	14.101	19.634
rescale(PCov_Oak)	4	1	14.482	4.114	6.418	22.546
rescale(PCov_2-5m)	2	0.59	3.539	4.181	-1.486	13.560
rescale(PCov_ ImpGrass)	2	0.51	-1.325	1.561	-4.962	-0.271

h) Spp_IRR

Predictor	N Containing Models	Relative Importance	Estimate	SE	Lower CI	Upper CI
(Intercept)	-	-	-0.838	0.054	-0.943	-0.733
(phi)	-	-	16.072	4.103	8.031	24.114
rescale(Ht_Av)	7	1	1.029	0.108	0.817	1.242
rescale(PCov_2-5m)	3	0.27	0.025	0.066	-0.100	0.286
rescale(PCov_Elm)	3	0.21	0.007	0.052	-0.182	0.246
rescale(PCov_0.5-2m)	3	0.21	-0.004	0.045	-0.209	0.175

Table 5.6. Relative importance and model averaged coefficients (across the 95% confidence set) of standardised predictors for bird indices in relation to habitat composition and structure of the field-only transects, based on the unconditional model average from the top six AICc generalised linear models (Table S 5.20). See Table 3.2, Table 3.3, Table 3.4 and Table 3.5 for parameter explanations, and Table 3.6 for bird indices. (Negative coefficients in italics).

a) Bird_Density

Predictor	**N Containing Models**	**Relative Importance**	**Estimate**	**Adjusted SE**	**Lower CI**	**Upper CI**
(Intercept)	-	-	7.029	0.736	5.586	8.472
z.P_HedgeLen	5	0.73	2.605	2.199	0.024	7.076
z.PCov_Elm	3	0.17	0.231	1.019	-2.836	5.599
z.PCov_2-5m	5	0.28	0.789	2.180	-3.760	9.384
z.PCov_Maple	2	0.12	-0.056	0.489	-3.116	2.169
z.PCov_BroadOther	2	0.11	0.046	0.730	-3.790	4.617
z.PCov_Oak	1	0.05	-0.101	0.542	-4.797	0.758

b) Spp_Richness

Predictor	**N Containing Models**	**Relative Importance**	**Estimate**	**Adjusted SE**	**Lower CI**	**Upper CI**
(Intercept)	-	-	17.273	0.946	15.419	19.127
z.P_HedgeLen	6	0.90	6.057	2.878	2.373	11.023
z.PCov_ImpGrass	3	0.40	-1.538	2.116	-6.792	-0.973
z.PCov_Maple	2	0.33	1.614	2.610	0.517	9.147
z.PCov_Elm	2	0.19	0.893	2.121	0.031	9.404
z.PCov_2-5m	3	0.09	0.505	1.873	-0.988	11.916
z.PCov_BroadOther	1	0.02	0.006	0.334	-4.603	5.245

c) Spp_Diversity

Predictor	**Estimate**	**SE**	**t**	**P**	**Lower CI**	**Upper CI**
(Intercept)	2.410	0.047	51.194	<0.001	2.320	2.505
z.P_HedgeLen	0.476	0.104	4.588	<0.001	0.272	0.686
z.PCov_ImpGrass	-0.399	0.071	-5.650	<0.001	-0.534	-0.249

d) Spp_Decline

Predictor	**N Containing Models**	**Relative Importance**	**Estimate**	**Adjusted SE**	**Lower CI**	**Upper CI**
(Intercept)	-	-	6.409	0.575	5.281	7.537
z.P_HedgeLen	5	0.76	2.258	1.738	0.351	5.629
z.PCov_Cons	3	0.23	0.334	0.878	-1.146	4.019
z.PCov_2-5m	5	0.22	0.338	1.178	-2.598	5.690
z.PCov_Maple	3	0.16	0.190	0.695	-1.490	3.813
z.PCov_Elm	3	0.14	0.122	0.618	-1.952	3.754

e) Spp_Priority

Predictor	**N Containing Models**	**Relative Importance**	**Estimate**	**Adjusted SE**	**Lower CI**	**Upper CI**
(Intercept)	-	-	7.227	0.612	6.029	8.426
z.P_HedgeLen	5	0.86	3.170	1.826	0.869	6.463
z.PCov_Cons	3	0.49	1.226	1.631	-0.437	5.435
z.PCov_Elm	3	0.18	0.386	1.061	-0.915	5.244
z.PCov_2-5m	3	0.13	0.264	1.126	-2.821	6.957
z.PCov_Maple	1	0.10	0.166	0.665	-1.117	4.371

f) Spp_Priority^{w}

Predictor	**N Containing Models**	**Relative Importance**	**Estimate**	**Adjusted SE**	**Lower CI**	**Upper CI**
(Intercept)	-	-	1.193	0.105	0.987	1.399
z.P_HedgeLen	2	0.95	0.704	0.276	0.286	1.194
z.PCov_ImpGrass	2	0.86	0.440	0.277	0.062	0.962
z.PCov_CropOther	1	0.14	-0.074	0.209	-1.051	-0.010
z.PCov_Maple	1	0.05	0.030	0.144	0.094	1.132

g) Spp_Rarity

Predictor	**N Containing Models**	**Relative Importance**	**Estimate**	**Adjusted SE**	**Lower CI**	**Upper CI**
(Intercept)	-	-	3.176	0.232	2.722	3.631
rescale(P_HedgeLen)	4	0.66	0.758	0.693	0.105	2.187
rescale(PCov_Cons)	4	0.26	0.136	0.361	-0.539	1.601
rescale(PCov_Maple)	4	0.24	0.154	0.387	-0.452	1.733
rescale(PCov_2-5m)	4	0.20	0.078	0.426	-1.352	2.139

h) Spp_IRR

Predictor	**N Containing Models**	**Relative Importance**	**Estimate**	**SE**	**Lower CI**	**Upper CI**
(Intercept)	-	-	-1.196	0.087	-1.366	-1.026
(phi)	-	-	13.570	5.185	3.408	23.732
rescale(PCov_Cons)	4	0.35	0.066	0.116	-0.059	0.434
rescale(Ostorey_PenDepth)	3	0.31	-0.057	0.106	-0.408	0.042
rescale(PCov_>5m)	3	0.19	-0.024	0.070	-0.351	0.100
rescale(Pres_Thorns)	4	0.18	0.024	0.132	-0.425	0.693

*Table 5.7. Relative importance and model averaged coefficients (across the 95% confidence set) of standardised predictors for bird indices in relation to habitat composition and structure of the woodland-only transects, based on the unconditional model average from the top six AICc generalised linear models (**Table S 5.21**). See Table 3.2, Table 3.3, Table 3.4 and Table 3.5 for parameter explanations, and Table 3.6 for bird indices (Negative coefficients in italics).*

a) Bird_Density

Predictor	**N Containing Models**	**Relative Importance**	**Estimate**	**Adjusted SE**	**Lower CI**	**Upper CI**
(Intercept)	-	-	41.477	2.573	36.433	46.521
z.PCov_BroadOther	8	0.32	-2.688	4.574	-16.633	0.086
z.PCov_Cons	6	0.32	-1.518	3.515	-15.410	0.598
z.Wood.Area	6	0.29	-3.743	7.115	-27.610	1.771
z.Ostorey_PenDepth	5	0.28	3.724	7.280	-1.667	28.702
z.Ht_VDR	8	0.17	-1.272	3.605	-18.252	3.078
z.Dist_Wood	5	0.11	-0.625	2.512	-16.321	4.593
c.Pres_Bramble	4	0.11	0.763	2.853	-3.872	17.930
z.PCov_Elm	4	0.07	0.389	2.271	-7.677	18.501

b) Spp_Richness

Predictor	**N Containing Models**	**Relative Importance**	**Estimate**	**Adjusted SE**	**Lower CI**	**Upper CI**
(Intercept)	-	-	22.375	1.293	19.840	24.910
z.Wood.Area	4	0.29	-0.806	1.925	-8.089	2.500
z.Ht_StDev	4	0.20	0.316	1.356	-3.673	6.767
z.PCov_Cons	4	0.19	0.230	1.358	-4.511	6.937
z.Ostorey_PenDepth	4	0.19	-0.181	1.279	-6.522	4.583

c) Spp_Diversity

Predictor	**N Containing Models**	**Relative Importance**	**Estimate**	**Adjusted SE**	**Lower CI**	**Upper CI**
(Intercept)	-	-	2.720	0.019	2.682	2.758
z.Wood.Area	6	0.82	-0.097	0.058	-0.197	-0.041
z.Ht_StDev	2	0.21	0.010	0.027	-0.030	0.129
z.Wood_P.A	3	0.18	0.020	0.045	0.023	0.191
z.PCov_Oak	2	0.15	0.006	0.022	-0.039	0.122
z.Ostorey_PenDepth	1	0.08	0.002	0.014	-0.064	0.115
z.Ht_VDR	1	0.07	-0.001	0.011	-0.090	0.064
z.PCov_Ash	1	0.06	0.000	0.010	-0.083	0.079

d) Spp_Decline

Predictor	**N Containing Models**	**Relative Importance**	**Estimate**	**Adjusted SE**	**Lower CI**	**Upper CI**
(Intercept)	-	-	1.972	0.102	1.772	2.171
z.Ht_VDR	3	0.21	-0.018	0.105	-0.512	0.336
z.%_Woods_500m	3	0.21	-0.018	0.108	-0.527	0.355
z.PCov_Oak	3	0.20	0.015	0.103	-0.351	0.501

e) Spp_Priority

Predictor	**N Containing Models**	**Relative Importance**	**Estimate**	**Adjusted SE**	**Lower CI**	**Upper CI**
(Intercept)	-	-	2.189	0.091	2.010	2.368
z.%_Woods_500m	3	0.23	-0.029	0.113	-0.534	0.280
z.PCov_Elm	3	0.22	-0.027	0.110	-0.524	0.287
z.PCov_2-5m	3	0.22	0.023	0.094	-0.241	0.449

f) Spp_Priority^{w}

Predictor	**N Containing Models**	**Relative Importance**	**Estimate**	**Adjusted SE**	**Lower CI**	**Upper CI**
(Intercept)	-	-	17.072	1.153	14.812	19.331
z.PCov_Cons	7	1.00	-5.917	1.303	-8.470	-3.363
c.Pres_Bramble	1	0.38	1.872	2.785	0.427	9.456
z.PCov_BroadOther	1	0.18	-0.591	1.510	-7.123	0.677
z.Ht_VDR	1	0.07	-0.164	0.900	-7.514	2.478
z.PCov_Oak	1	0.05	0.099	0.715	-3.078	6.818
z.Ostorey_PenDepth	1	0.05	0.068	0.713	-4.401	7.377
z.%_Woods_1km	1	0.04	-0.032	0.589	-6.362	4.779

g) Spp_Rarity

Predictor	**N Containing Models**	**Relative Importance**	**Estimate**	**Adjusted SE**	**Lower CI**	**Upper CI**
(Intercept)	-	-	25.500	1.383	22.790	28.210
z.PCov_Oak	3	0.41	1.730	2.781	-1.413	9.893
z.Ostorey_Ht	3	0.40	1.580	2.659	-1.651	9.614
z.Wood.Area	3	0.31	-1.038	2.213	-8.908	2.228

h) Spp_IRR

Predictor	**N Containing Models**	**Relative Importance**	**Estimate**	**SE**	**Lower CI**	**Upper CI**
(Intercept)	-	-	-0.313	0.060	-0.431	-0.196
(phi)	-	-	28.941	11.325	6.744	51.137
rescale(Ht_ VDR)	2	0.63	-0.471	0.377	-0.991	-0.516
rescale(Pres_ Bramble)	2	0.54	-0.176	0.196	-0.618	-0.033
rescale(Ht_Av)	4	0.37	0.312	0.414	0.543	1.125
rescale(PCov_ Cons)	1	0.04	-0.003	0.030	-0.312	0.172
rescale(PCov_ <0.5m)	1	0.04	0.003	0.048	-0.390	0.521

5.5 Discussion

The Cambridgeshire landscape consists of two contrasting land uses; agricultural areas, which dominate, and interspersed woodlands of various sizes. As expected the woodlands, particularly the edges (in line with Melin et al. (2018)), had higher values for the majority of the bird indices, with the exception of the number of declining and priority species which were higher in the ‘Field_high%_Hedge’ transects, as a consequence of historic hedgerow removal (Robinson and Sutherland 2002). These results show that hedges in the agricultural landscape not only increased bird density, diversity and species richness, supporting the findings of O’Connell et al. (2015) and Fuller et al. (2005b), but also supported declining, priority and rare bird species. Furthermore, declining and priority bird species existed in both the field and wood habitats, suggesting detrimental effects in both habitats and possibly an interaction, as exhibited in the ‘Field By_Wood’ transects. However, IRR in the field habitat was not related to the proportion of hedge but rather loosely on the percentage of conifers. However, the weak and possibly spurious relationship suggests there are other reasons, or simply stochasticity, for the occurrence of the rare Yellow Wagtail and Raven.

Woodlands were, nonetheless, shown to be extremely important for all the bird indices over the Cambridgeshire landscape. Fragmentation has provided woodland edge habitat, increasing overall gamma bird diversity and also rarity in some instances. The Null models and weak relationships with the vegetation structural and composition variables in the woodland only analysis suggests that there are other factors influencing the bird indices. However, the positive relationships with oak and height of the vegetation suggest that declines in oak, and possibly ash, are affecting declining, conservation priority and rare woodland bird species in Cambridgeshire. Variables were included to investigate the effects of fragmentation and isolation on the bird indices. The woodland only analysis demonstrates that the interconnectedness of the fragmented woodlands is more important for the bird indices than the vegetation composition and structural variables, regardless of woodland size. Monks Wood, the largest wood (~170 ha), is most likely acting as a source population for the sufficiently interconnected smaller woods to allow populations to persist.

Although the woodlands support higher values of the majority of the bird indices, the rarity index indicates that specialist species also exist in the field habitats with some species (e.g. the Yellowhammer) reliant on hedges. Other species, such as the Skylark, also depend on open areas of suitable habitat, highlighting the importance of a heterogeneous landscape in terms of both vegetation structure and composition. This further shows that simply managing habitats for biodiversity (alpha diversity) would neglect rare and declining specialist bird species, therefore gamma diversity over the landscape should be conserved and managed. Furthermore, the cropped, as well as the uncropped areas providing semi-natural habitat, must be managed appropriately and heterogeneously for the species present and to maximise gamma diversity.

5.5.1 Connected Woodlands increase Bird Indices

As expected, the current study showed that the woodlands, particularly their edges, had higher bird density than the fields, in line with Heikkinen et al. (2004). The edge affect results from a greater number of resident, passage and open country birds using the edges of woodlands, thus increasing species richness and diversity (Paquet et al. 2006, Šálek et al. 2015, Terraube et al. 2016, Melin et al. 2018). Whittingham et al. (2009) found a strong positive influence of woodland edge with the territory distribution of all ten of their study bird species. Birds (and other taxa) were found by Šálek et al. (2015) to respond to variables in buffers of 300-500 m of woodland/agricultural edges, whereas the current study only went to only included 40 m of woodland edge, and still displayed higher species richness and diversity. Habitat diversity, edge density, including shrub density, and land cover classes (along with landscape management and heterogeneity) were shown to increase species richness (Šálek et al. 2015). This was reflected in the positive relationship of bird diversity to a more open canopy (as represented by overstorey penetration depth) in the current study, possibly a result of the greater variation in vegetation height at the woodland edge. This created a higher value of this metric as the taller trees create a greater distance between the canopy and the ground, and therefore facilitate greater laser pulse penetration. The high values of species priority weighted by bird density ($Spp\_Priority^{w}$) in the 'Wood Edge' transects is in line with Paquet et al. (2006) who also showed that the "conservation value index" which was calculated from the frequency of occurrence, conservation status in Europe or southern Belgium and species richness, was higher in edge habitats and open areas in the forest. However, the current study shows that this metric is overshadowed by bird density, and therefore conservation status or priority of the bird species should be used.

Melin et al. (2018) found that an edge effect still occurred in small woods, with bird diversity and abundance decreasing into the centre. The same small woods, in the current study, also had increased bird species richness and diversity as a result of the larger edge to area ratios, and the subsequent survey routes sampled a larger proportion of the woodland, which contained a greater proportion of edge (Bellamy et al. 1996). Consequently, bird species diversity was negatively correlated with the length of the woodland perimeter in the univariate analysis, and positively related to the ratio of woodland perimeter to area in both the univariate and multi-model analysis. However, Kleijn et al. (2011) demonstrated that source-sink dynamics will lead to an inflated abundance and species richness on low quality habitat fragments, overestimating its value and underestimating the value of larger, higher quality habitats.

Bird density, diversity and species richness were negatively related to wood area in the current study, contradicting the findings of Freemark and Merriam (1986), Robbins et al. (1989) and Bennett et al. (2004), and more recently Gardner et al.

(2019). Furthermore, rarity was also negatively related to wood area, implying that more individuals, including rare species, were present in the smaller woods in Cambridgeshire, perhaps as a result of territories being smaller or in closer proximity because of limited space, and greater vegetation density, following Šálek et al. (2015). Similarly, Mayer and Cameron (2003) found that guild species richness was related to both size and number of forest patches, whereas diversity was only related to size of forest patch. Howell et al. (2000) also presented similar findings, but suggested that the fragmented forests were more likely to contain generalists and edge species, and the larger, less fragmented forests contained interior species and long-distance migrants. Hinsley et al. (1996) also showed that bird species were associated differently to woodland area, for example the Wren and the Blackbird preferred small woodlands, whereas the Chiffchaff preferred larger woodlands.

Bellamy et al. (1996) separated edge and woodland interior bird species and noted a relationship of woodland species to woodland area, whereas, edge species were related to perimeter length, indicating different bird species assemblages. Moreover, Melin et al. (2018) showed that vegetation density was highest at the woodland edge and was also related to bird diversity. It should also be noted that larger woods will contain some species with larger minimum area requirements that seldom, if ever, occur in small woods (Hinsley et al. 1996b)

Species richness and diversity per hectare of woodland was actually highest in the smallest wood, Gamsey, further indicating higher species richness and diversity per unit area, supporting the findings by Melin et al. (2018). The species richness-area curve showed an increase in species richness to a point where it plateaued, signifying that either all the species or habitat types in the landscape have been encountered or there are other limiting factors (Scheiner et al. 2000). This therefore suggests that any woodland, regardless of size, is beneficial to increase bird density, species richness and diversity in an agricultural environment. The edge effects between forests and open areas may also provide valuable ecosystem services for functional guilds in fragmented landscapes (Terraube et al. 2016). Nonetheless, it is worth noting that the smallest woodland in the current study was 4.7 ha, therefore recommendations of woodland size are only substantiated for woodlands above this.

In the woodland only analysis species decline and species priority were negatively related to the percentage of woodland in the surrounding 500 m buffer (as a proportion of woodland size), indicating an effect of woodland fragmentation and isolation on declining bird species. This result was driven by the isolation of Aversley Wood (transects 25, 30 and 34), the furthest distance from the other woodlands, suggesting that the declining and priority species are self-sustaining in this woodland. However, the extra woodland variables included to analyse Metapopulation and Island Biogeography Theories only include woodlands in the study area (Figure 3.6). Species decline was also negatively related to height VDR, and positively related to the percent cover of oak in the woodland only analysis, suggesting that oak decline may be affecting declining bird populations. However, this result does not elude to the size of the woodlands, and should be interpreted with caution as confidence intervals of all the variables contain zero. Consequently, species rarity in the woodland analysis was also positively related to the percent cover of oak, suggesting that these declining bird species are also rare. Furthermore, the positive relationship with overstorey height and negative relationship with wood area suggests that the fragmented, smaller oak woodland with tall canopies sustain rare bird species. Thus, connections between natural and ancient woodlands must be maintained and managed across the landscape to conserve rare bird species (Haslem and Bennett 2011).

Opdam et al. (1985) also showed that patch size and isolation affect bird communities in patchy environments more than vegetation composition and structure of woodland patches. The proximity of the nearest larger wood provides a random assortment of bird species to the smaller patches, increasing in number of species as patch size increases (Opdam et al. 1985). Therefore, larger blocks of woodland or a threshold of habitat cover must be maintained to continue to support certain species with large area requirements (Broughton et al. 2013), along with a range of other sizes of fragmented woodlands to support adaptable, generalist species (Howell et al. 2000). Conversely, Robles and Ciudad (2012) showed that the occupancy by their study species (the Middle Spotted Woodpecker, *Leiopicus medius*) was determined by habitat quality rather than patch size, and extinction was also based on population size rather than patch size.

Estrada et al. (1993) found that patches of vegetation throughout the agricultural area provided biotic connectivity between isolated forest fragments and foraging areas. This suggests that Monks Wood, the largest wood in the current study, could be providing a source population for the sufficiently interconnected woodlands and be mediating any effect of woodland size in line with Metapopulation Theory (Hanski et al. 1997). For example, the relatively rare woodland interior specialist, the Marsh Tit, was present in almost every woodland transect, regardless of size and proximity to the edge. This is unexpected as Broughton et al. (2012) found that Marsh Tit occupation was lower within 50 m from the woodland edge. However, minimum territory size for Marsh Tits in the Cambridgeshire landscape has been estimated at 4-6 ha (Broughton and Hinsley 2015) and thus the study woods, when including edge habitat, were large enough to support at least one territory. The interconnectedness of the surrounding woodlands could also be causing a higher density than expected (Kleijn et al. 2011), or the birds could be foraging or passing through the woodland edge. On the other hand, the result could be skewed due to the bird densities being inflated by the distance sampling density estimates, calculated from the 'Wood Interior' transects, artificially increasing bird densities due to detectability, rather than the true surveyed density (Newson et al. 2008).

The woodland fragments should therefore be maintained in various sizes and in close enough proximity in order to enhance biodiversity and regenerate the ecosystem with direct benefits to agriculture (Estrada et al. 1993, Graham and Blake 2001). Haslem and Bennett (2011) reported that 65% of the regional species in their survey areas in Australia were in agricultural areas, but wooded areas increased the number of bird species in an agricultural mosaic, highlighting the importance of semi-natural, woody vegetation in agricultural areas for birds. However, it is worth noting that these disturbed woodlands may suffer from lower reproductive output and increased energy costs for parents, for example, due to a lack of certain resources and/or greater predation pressure (Robinson et al. 1995, Graham and Blake 2001). This has been shown to result in reduced Marsh Tit longevity compared to their interior counterparts (Hinsley et al. 2009b).

Agricultural intensification has impacted all measures of the bird indices in this and most agricultural landscapes, with higher bird index values in the semi-natural vegetation (Rodewald and Yahner 2001, Robinson and Sutherland 2002, Fuller et al. 2005b), supported by the positive relationship of the percentage of vegetation at 2-5 m with all the indices. However, the edge affect from the woodlands also reflects outward into the fields, creating poor bird habitat, as reflected in all of the bird indices. This affect is worsened when the adjacent fields comprise improved grass, reflected in the negative relationship with bird species richness and diversity in the field-only analysis, and also to rarity in the landscape analysis, possibly as a result of the over-management of these areas and their uses. The grass areas either contained livestock (horses and cows) or were possibly being grown for silage, all of which have detrimental effects on ground nesting birds from trampling, grazing or cutting, probably more so than any other crop (Vickery et al. 2001). Nelson et al. (2011) stated that a balance must therefore be achieved between avifauna conservation and agricultural practises, such as cattle grazing. Conversely, the species priority index weighted by bird density (*Spp_Priority*^{w}) in the fields was positively related to improved grass, suggesting that a greater number of lower priority bird species were present in the improved grass. This was possibly as a result of the large number of corvids, such as Rooks, in 'Field By_Wood' transect 22 feeding on the grass outside Aversely Wood, probably attracted by horse dung (Barnett et al. 2004). The presence of the corvid predators is likely reducing the number of other bird species in the fields next to woodlands, thereby creating an exclusion zone (Donald et al. 2001b, Sheldon et al. 2004).

5.5.2 Hedges are Important Bird Habitat

In the current study, the greatest number of declining bird species occurred in the field transects with the highest proportion of hedge length, possibly as a result of increased specialisation of bird communities in open-habitats (Terraube et al. 2016). Consequently, in the landscape analysis the proportion of hedge length was the most important variable with a positive relationship with the species decline index. Once separated, the field only analysis showed that all of the bird indices, except IRR, were positively related to the proportion of hedge length. This demonstrates that hedges, often as the only semi-natural vegetation, are not only vital to increase bird diversity in the agricultural landscape (in line with Paquet et al. 2006, O'Connell et al. 2015), but also provide habitat for declining and rare bird species.

Hedges are a vital resource for a number of farmland birds, and the continued decline of these species, such as the Yellowhammer (Whittingham et al. 2009), suggests that farmland bird populations are still suffering the effects of historic hedgerow removal (Cornulier et al. 2011). The continued decline may be due to a delayed reaction and subsequent lack of improvement in habitat, ultimately leading to negative feedback and extinction (Kleijn et al. 2011). For example, Yellowhammers prefer mosaic habitats with some trees, but avoid densely forested areas (supporting the absence in the ‘Field By_Wood’ transects), and nest in field boundaries either on the ground (in ditches or grassy margins) or in hedges or bushes (Stoate et al. 1998, Hagemeijer and Blair 1997 in Bradbury et al. 2000). As a result, the removal of hedgerows or lack of hedge management, as well as filling or clearing ditches, planting right up to the field edge and intensively managed grassland, have had detrimental effects on the breeding productivity of the Yellowhammer in southern England (Bradbury et al. 2000).

Research by Cornulier et al. (2011) suggested that there were two historical periods of major hedge decline, 1955-1970 for field drainage schemes and 1983-1994 for creating larger fields, both occurring later than previously suggested (Robinson and Sutherland 2002). Cornulier et al. (2011) stated that habitat loss may not directly or immediately affect bird demography, but results in breeding birds in the population becoming non-breeders. This results in a decrease in population size and recruitment without affecting adult survival, as shown with the Yellowhammer. Cornulier et al. (2011) also showed that the population density index of the Yellowhammer was strongly correlated with hedgerow length but with a time-lag of 2-3 years.

Hedges increase heterogeneity of agricultural areas which in turn increases species richness and beta-diversity by providing habitat for early-successional bird communities (Paquet et al. 2006). Gregory and Baillie (1998) noted that for some bird species, 50% of the population they recorded was in agricultural land, indicating that agricultural areas increase beta diversity, supporting the results of the current study. Additionally, Lack (1988, in Hinsley and Bellamy 2019) found that there were 1.7 times more birds at hedge intersections which provide more habitat per unit area, often with bushes and trees. Predominantly generalist bird species, such as the Wren, Robin, Blackbird, Blue Tit and Great Tit, were also more frequently found at the hedge intersections and had territories associated with hedges (Whittingham et al. 2009). In the current study, bird species richness was also positively related to the percent cover of field maple and elm in the field only analysis, suggesting that increasing these species, and therefore vegetation diversity in the hedges, would increase vegetation species richness and diversity (Wiens and Rotenberry 1981, Poulsen 2002).

Specialist bird species exist in both habitats in the current study (field and wood); but there is considerable overlap of bird species with declining populations of what were traditionally considered to be woodland species, such as the Garden Warbler and Mistle Thrush, now present in both habitats, possibly due to the increased hedge vegetation or proximity to woodland. However, generalist species could be having a detrimental effect on specialist species in both habitats by increasing competition (Hinsley et al. 2009b). Hinsley et al. (2009b) found that generalist bird species, as well as specialists, suffer in terms of reproductive output and parental energy costs in a fragmented landscape of managed secondary habitat, such as small woods, parks and gardens. Consequently, generalists, such as the Blackbird, Blue Tit, Chaffinch, Green Woodpecker and Great Tit were in decline in the current study (Appendix A2).

Originally forest clearance and agricultural intensification provided new opportunities for a number of bird species, including the Green Woodpecker and declining open habitat preferring species, such as the Skylark (Wilson et al. 2009, in Hinsley and Gillings 2012). However, subsequent changes in management practises, including changing to autumn sowing, cropping up to wood edges and the application of pesticides reducing food availability, certainly contributed to bird population declines. This indicates the importance of maintaining heterogeneous vegetative habitat patches and increasing habitat complexity over an agricultural landscape to benefit declining bird species (Terraube et al. 2016). As well as natural/semi-natural woodlands, the field type, crop heterogeneity and lower growing semi-natural vegetation must be managed and maintained in order to protect these iconic farmland birds. Whilst increasing habitat heterogeneity in intensively managed farmland would be beneficial, any increase in habitat heterogeneity into low-intensity farmland could be detrimental to specialist priority species, which may reduce and fragment their habitat and should be avoided (Batáry et al. 2011).

The number of declining and priority bird species was higher in the field transects, but declining species also occurred in the woodland, suggesting interaction between the two habitats. This demonstrates that semi-natural areas in an agricultural landscape, such as small woods, hedges and unimproved grassland, are crucial to preserve certain bird species. Historic changes in agricultural practises are still affecting bird populations, many of which can provide ecological services such as pest control and seed dispersal and thus should be maintained not only for wildlife health, but also to improve both agricultural production and the environment in general. Therefore, management strategies over an agricultural landscape should incorporate increasing the proportion and diversity of hedges and other semi-natural habitat such as small woods, whilst also maintaining open habitats for birds and other taxa. Variation in agricultural practises, even at small scales, are also recommended to increase the spatial and temporal heterogeneity of farmland management and land-use.

5.5.3 Relative Rarity

Often as a result of long term population declines, a number of bird species that occur in the Cambridgeshire landscape are rare, such as the Spotted Flycatcher and the Turtle Dove with only 33,000 and 14,000 territories in the UK, respectively (Musgrove et al. 2013). The rarest bird species in the Cambridgeshire landscape was the Common Raven (Appendix A2) with only 7000 breeding pairs in the UK (Musgrove et al. 2013).

Conversely to rarity, IRR was highest in ‘Wood Interior’ transect 34 (0.63), however, this was closely followed by ‘Wood Edge’ transect 23 (0.54). This consequently resulted in the significant positive relationship with the average height of the vegetation (trees) and also the coverage of the 2-5 m vegetation height layer, most likely as understorey vegetation. The positive relationship to the percent cover of elm may also indicate a greater number of rare bird species in the smaller woods with more elm, and could indicate an effect of fragmentation also on this index. The woodland analysis also demonstrates a positive relationship to the average height of the vegetation, and along with the negative relationship to height VDR and the presence of bramble suggests that rarer species prefer closed, taller canopy woodlands. The negative relationship to the percentage of conifer in the woodland analysis, suggests a detrimental effect on rare bird species, however, the confidence intervals of this variable contain zero and should therefore be treated with caution, most likely due to the low percentage of conifer in the landscape. In this agricultural landscape the woodlands are extremely important habitat for rare bird species, and the fragmentation has provided habitat for birds in woodland edges, increasing bird diversity. It should be noted that the transects with the highest value of rarity and IRR did not contain the Raven, therefore a number of other rare bird species are contributing to the value of the indices. These could include the woodland interior specialist the Marsh Tit, more open woodland preferring Spotted Flycatchers (Svensson et al. 1999), or even less rare species, such as the Bullfinch (Appendix A2). However, this is not apparent as the metric encompasses multiple bird species, therefore further analyses is required to investigate bird community composition.

As Broughton and Hinsley (2015) stated that Marsh Tits prefer structural complexity, this would be expected to be explained by height VDR in the current study. However, this metric was greater where there was more ground or lower vegetation with a single tall tree, therefore more uniform vegetation at any height would result in a lower value of height VDR. In addition, ‘Wood Edge’ transect 24 had a high value of height VDR and a low IRR value, contributing to the negative relationship. The positive relationship with the percent cover of vegetation in the 2-5 m layer in the landscape level analysis may better represent the species preference for understorey vegetation, increasing woodland structural complexity for the Marsh Tit, in line with Broughton and Hinsley (2015).

In the field only analysis of the current study, species decline, priority, rarity and IRR were all positively related to the percent cover of conifer in the agricultural habitat, and although this parameter may not be wholly contributing to these bird indices, it may indicate similar findings to Heikkinen et al. (2004). They showed a positive relationship of conifer cover and semi-natural grassland with the number of agricultural bird pairs and a negative relationship with cultivated field cover. This suggests that semi-natural areas and even conifers can enhance and support bird populations in an agricultural landscape, providing there is enough suitable habitat, such as hedges, in close proximity to sustain the populations.

The highest value of IRR in the field habitats was in ‘Field_low%_Hedge’ transect 4 which contained the rarest bird species, the Raven, and also the Yellow Wagtail (Appendix A2). The multi-model analysis for the field transects showed that the top model was the Null model, indicating that the available variables did not explain the variation in IRR in the field transects. The remaining candidate set of top models were very weak with unreliable variable estimates. Nonetheless, the negative relationship with overstorey penetration depth (or canopy openness), was most likely a result of the low value in ‘Field_low%_Hedge’ transect 4 and the possible cause of the presence of the Raven and Yellow Wagtail. Further to that, the occurrence of these species driving the higher IRR may be stochastic, and therefore purely by chance, or due to other unexplored factors.

5.6 Conclusion

As a result of agricultural intensification, the bird populations in the Cambridgeshire study landscape have adapted to form a metapopulation in the variously sized woodlands surrounding one larger woodland, Monks Wood. These smaller woodlands not only contain a higher diversity of birds, especially at the edges, but also support rare and declining bird species. Equally in the fields, the hedges provide vital habitat for birds not only increasing bird diversity and species richness, but also supporting a large number of declining farmland birds, presumably in relation to historic hedge decline. This study shows that semi-natural vegetation is vital to support biodiversity, declining, conservation priority and rare species in this agricultural landscape. However, these metrics comprise multiple bird species, therefore habitats could have similar values but contain different bird assemblages particularly in the contrasting habitats. Thus, bird community analysis is required to determine the species driving these indices in the various habitats.

5.7 Supplementary Material

Table S 5.1. The percent cover of the three crop categories in the field transects (percent cover of crops/improved grass in woodland is zero).

Transect No.	Habitat Class	% Cover Improved Grass	% Cover Cereal	% Cover Other Crop
1	**Field_low%_Hedge**	0	37	43
2	**Field_low%_Hedge**	0	76	14
3	**Field_low%_Hedge**	0	88	0
4	**Field_low%_Hedge**	0	55	35
5	**Field_low%_Hedge**	0	87	0
6	**Field_med%_Hedge**	0	16	72
7	**Field_med%_Hedge**	6	48	36
8	**Field_med%_Hedge**	19	21	31
9	**Field_med%_Hedge**	0	88	0
10	**Field_med%_Hedge**	0	61	31
11	**Field_med%_Hedge**	0	72	3
12	**Field_high%_Hedge**	17	55	0
13	**Field_high%_Hedge**	0	30	43
14	**Field_high%_Hedge**	22	9	55
15	**Field_high%_Hedge**	4	0	71
16	**Field_high%_Hedge**	0	64	0
17	**Field_high%_Hedge**	0	46	39
18	**Field By_Wood**	0	31	55
19	**Field By_Wood**	0	68	0
20	**Field By_Wood**	0	13	76
21	**Field By_Wood**	74	0	21
22	**Field By_Wood**	93	0	0

Table S 5.2. The percent cover of woody non-arable vegetation below and above 2 m and percent composition of woody vegetation species present in the field transects as hedges, copses or single trees. For species in brackets the % composition is unavailable. Infrastructure (i.e. buildings and roads) not included.

T_ No.	**Habitat Class**	**% Cover <2m**	**% Cover >2m**	**% Composition Vegetation**
1	**Field_low%_Hedge**	11.8	8.2	ash 15, oak 15, hawthorn 15, blackthorn 15, field maple 10, elm 10, sycamore 5, elder 5, cherry 5, dog rose 5 (bramble, buckthorn)
2	**Field_low%_Hedge**	7.8	2.2	ash 40, blackthorn 24, dog rose 16, field maple 15, hawthorn 5, lime 5, dogwood 1 (bramble)
3	**Field_low%_Hedge**	9.9	2.1	field maple 20, ash 20, hawthorn 20, elm 15, oak 10, pine 5, elder 5, lime 3, dog rose 1, willow 1
4	**Field_low%_Hedge**	8.8	1.2	blackthorn 20, hawthorn 20, field maple 20, willow 20, elm 10, ash and oak 5, elder 5 (bramble)
5	**Field_low%_Hedge**	9.7	3.3	blackthorn 20, hawthorn 20, field maple 20, ash 10, oak 10, elm 10, elder 5, willow 2, cherry 1, horse chestnut 1 (dog rose)
6	**Field_med%_Hedge**	9.2	2.8	dog rose 25, blackthorn 20, hawthorn 20, field maple 10, ash 10, elm 10, oak 2, elder 1, lime 1, dogwood 1 (bramble, apple)
7	**Field_med%_Hedge**	6.9	4.1	blackthorn 60, hawthorn 20, ash 10, field maple 5, oak 1, elder 1, dog rose 1 (bramble)
8	**Field_med%_Hedge**	20.2	8.8	ash 25, blackthorn 20, hawthorn 20, elm 20, elder 10, field maple 5 (dog rose, dogwood)
9	**Field_med%_Hedge**	9.8	2.2	blackthorn 40, hawthorn 40, field maple 5, elder 5, elm 5, willow 3, ash 1, dog rose 1 (bramble, dogwood)
10	**Field_med%_Hedge**	7.5	1.5	blackthorn 25, hawthorn 25, field maple 23, elm 15, ash 10, sycamore 2 (dogwood, dog rose)
11	**Field_med%_Hedge**	23.7	1.3	blackthorn 25, hawthorn 25, ash 15, oak 10, sycamore 10, field maple 5, elm, horse chestnut and aspen 5 (bramble, dog rose, dogwood)
12	**Field_high%_Hedge**	23.8	4.2	blackthorn 30, hawthorn 20, oak 15, field maple 10, ash 10, dog rose 7, willow 5, cherry 1, elm 1, dogwood 1 (bramble)
13	**Field_high%_Hedge**	15.1	11.9	willow 24, blackthorn 20, hawthorn 10, sycamore 10, field maple 5, elder 5, buckthorn/dogwood 10, dog rose 5, ash 5, elm 5, oak 1 (bramble)
14	**Field_high%_Hedge**	12.1	1.9	blackthorn 35, hawthorn 35, dog rose 20, field maple 5, ash 3, elder 1, willow 1
15	**Field_high%_Hedge**	15.7	9.3	blackthorn 25, hawthorn 15, field maple 15, ash 15, elm 10, cherry 5, hazel 5, dogwood 5, willow 1, oak 1, elder 1, apple 1, western red cedar 1 (bramble, dog rose)
16	**Field_high%_Hedge**	17.5	18.5	blackthorn 50, oak 10, elm 6, hornbeam 6, field maple 5, ash 5, hawthorn 5, elder 5, sycamore 5, willow 1, wild service 1, dogwood 1 (dog rose, bramble)
17	**Field_high%_Hedge**	12.5	2.5	blackthorn 40, dog rose 30, hawthorn 20, field maple 5, elder 5 (bramble, dogwood)
18	**Field By_Wood**	7.7	7.3	blackthorn 30, hawthorn 30, ash 14, oak 10, dog rose 10, field maple 7 (bramble)
19	**Field By_Wood**	25.6	6.4	blackthorn 35, hawthorn 35, elder 10, dog rose 10 (bramble)
20	**Field By_Wood**	9.3	1.7	blackthorn 30, hawthorn 30, oak 27, field maple 12 (bramble)
21	**Field By_Wood**	5.2	0.8	blackthorn 30, hawthorn 30, dog rose 10, willow 10, elder 5 (bramble)
22	**Field By_Wood**	6.4	0.6	blackthorn 30, hawthorn 30, dog rose 10, elder 5, dogwood 5 (bramble)

Table S 5.3.Vegetation composition in the woodland transects of Cambridgeshire at the different height layers. (% composition unavailable for species under the canopy due to difficulties in estimations).

T_ No.	Habitat Class	%Cover Ground/Shrub Layer (<2m)	%Cover Understorey Layer (2-5m)	%Cover Overstorey (>5m)
23	**Wood Edge**	6.3 (dog rose)	6.2 (hawthorn, blackthorn, hazel, apple, maple, wild service)	87.5 (ash 75, field maple 14, oak 10, elm 1)
24	**Wood Edge**	54.0 (dog rose, bramble)	30.4 (blackthorn 30, hawthorn 30, buckthorn 5, hazel 2, wild service 1, dogwood 1, elder 1)	15.6 (oak 5, ash 5, field maple 5, elm 5, birch 5)
25	**Wood Edge**	6.1 (dog rose, brambles)	5.1 (blackthorn, hawthorn, hazel, field maple, apple)	88.8 (ash 40, oak 35, field maple 15, elm 5, wild service 4, apple 1)
26	**Wood Edge**	8.7 (guelder rose, dog rose, brambles)	7.5 (hawthorn, blackthorn, hazel, field maple, apple)	83.8 (oak 33, ash 30, field maple 20, wild service 10, apple 5, elm 1, western hemlock 1)
27	**Wood Edge**	20.9 (bramble, dog rose)	10.6 (blackthorn, hawthorn, field maple, dogwood, elder, hazel, apple)	68.5 (ash 40, oak 35, field maple 15, cherry 1, sycamore 1, aspen 1, willow 1, elm 1)
28	**Wood Edge**	23.2 (bramble, dog rose)	7.2 (blackthorn, hawthorn, field maple, cherry, elm, dogwood)	69.7 (ash 41, oak 38, field maple 10, cherry 5)
29	**Wood Edge**	35.8 (bramble, dog rose)	5.7 (blackthorn, hawthorn, elder, field maple, apple, ash, elm, willow, hazel)	58.5 (ash 40, oak 39, field maple 15, elm 5, horse chestnut 1)
30	**Wood Edge**	15.1 (bramble, dog rose)	4.8 (elm, hazel, hawthorn, blackthorn, field maple, elder, dogwood)	80.1 (ash 50, field maple 25, oak 24, wild service 1)
31	**Wood Edge**	21.4 (bramble, dog rose)	5.9 (hawthorn, Y ash, hazel, apple, field maple, willow)	72.6 (ash 55, oak 40, field maple 5)
32	**Wood Interior**	23.5 (dog rose, grass)	10.3 (hawthorn, blackthorn, hazel, field maple, aspen, apple, wild service	66.2 (ash 40, oak 38, aspen 10, field maple 5, silver birch 5, hazel 1)
33	**Wood Interior**	11.9 (bramble, dog rose)	9.2 (hazel, hawthorn, blackthorn, willow)	78.9 (ash 50, field maple 25, oak 19, silver 5, willow 1)
34	**Wood Interior**	10.7 (grass)	5.4 (hazel, hawthorn, dogwood, field maple, blackthorn, cherry, elm, ash)	83.9 (ash 38, oak 35, field maple 9, aspen 5, wild service 5, beech 2, hazel 2, willow 1, elm 1)
35	**Wood Interior**	45.1 (bramble)	4.0 (hazel, blackthorn, Y ash, hawthorn, dogwood, elm, field maple)	50.9 (ash 30, oak 25, pine 25, elm 5, western red cedar 5, field maple 1, silver birch 1, willow 1, western hemlock 1, Norway spruce, Douglas fir)
36	**Wood Interior**	7.6 (dog rose)	8.1 (blackthorn 5, hawthorn, apple, field maple, elder, hazel, dogwood)	84.2 (ash 30, field maple 25, elm 20, oak 10, hazel 5, wild service 5)
37	**Wood Interior**	8.1 (dog rose, bramble)	5.1 (blackthorn 4, hawthorn, dogwood, elm, elder, ash, oak, hazel)	86.8 (elm 35, ash 25, field maple 20, oak 15, apple 1)
38	**Wood Interior**	11.3 (bramble)	9.8 (blackthorn 10, elder, hawthorn, apple)	78.9 (ash 35, oak 25, field maple 25, elm 5)

Table S 5.4. Presence/absence data for each transect in Cambridgeshire (1 = present, 0 = absent). See Table 3.2 for variable explanation.

T_No.	Habitat Class	*Pres_Thorns*	*Pres_Bramble*	*Pres_Shrub*
1	**Field_low%_Hedge**	1	1	1
2	**Field_low%_Hedge**	1	1	1
3	**Field_low%_Hedge**	1	0	1
4	**Field_low%_Hedge**	1	1	1
5	**Field_low%_Hedge**	1	0	1
6	**Field_med%_Hedge**	1	1	0
7	**Field_med%_Hedge**	1	1	1
8	**Field_med%_Hedge**	1	0	1
9	**Field_med%_Hedge**	1	1	0
10	**Field_med%_Hedge**	1	0	1
11	**Field_med%_Hedge**	1	1	1
12	**Field_high%_Hedge**	1	1	1
13	**Field_high%_Hedge**	1	1	1
14	**Field_high%_Hedge**	1	1	1
15	**Field_high%_Hedge**	1	1	0
16	**Field_high%_Hedge**	1	1	1
17	**Field_high%_Hedge**	1	1	1
18	**Field By_Wood**	1	1	1
19	**Field By_Wood**	1	1	1
20	**Field By_Wood**	1	1	1
21	**Field By_Wood**	1	1	1
22	**Field By_Wood**	1	1	1
23	**Wood Edge**	1	0	1
24	**Wood Edge**	1	1	1
25	**Wood Edge**	1	1	1
26	**Wood Edge**	1	1	1
27	**Wood Edge**	1	1	1
28	**Wood Edge**	1	1	1
29	**Wood Edge**	1	1	1
30	**Wood Edge**	1	1	1
31	**Wood Edge**	1	1	1
32	**Wood Interior**	1	0	1
33	**Wood Interior**	1	1	1
34	**Wood Interior**	1	0	1
35	**Wood Interior**	1	1	1
36	**Wood Interior**	1	0	1
37	**Wood Interior**	1	1	1
38	**Wood Interior**	1	1	1

The ANOVA of the vegetation composition variables in Table S 5.5, showed that the percentage cover of ash, oak, field maple, improved grass, cereal and other crops showed a statistically significant difference (at P <0.05) between habitat classes. The percentage cover of elm, other broadleaved species, conifers, and the presence of thorns (hawthorn, blackthorn and buckthorn (*Frangula alnus*)), bramble and shrub were not significantly different (Table S 5.5). For the percentage cover of ash, oak and field maple, significant difference occurred between all field and wood habitat classes, but not within any field or wood habitat classes as shown in Table S 5.6. A similar pattern occurred for *PCov_Cereal,* except there was a significant difference between 'Field By_Wood' and 'Field_low%_Hedge', but not with either woodland class. This is because the 'Field By_Wood' transects, with one exception, were low in *PCov_Cereal*, whilst the 'Field_low%_Hedge' transects were in larger arable fields with a high *PCov_Cereal* (Table S 5.1). The statistically significant difference for *PCov_ImpGrass* occurred between the 'Field By_Wood' transects with the 'Field_low%_Hedge' and both the woodland classes (Table S 5.6), because of the high *PCov_ImpGrass* in two of the 'Field By_Wood' transects and a 0% cover in all 'Field_low%_Hedge' transects (Table S 5.1). Finally, the only statistically significant difference for *PCov_CropOther* occurred between 'Field_high%_Hedge' and 'Wood Edge'. It is also worth noting that the P value was 0.055 in the ANOVA output between 'Wood Interior' and 'Field_high%_Hedge', even though *PCov_CropOther* was zero for all the woodland transects (Table S 5.6).

Table S 5.5. Results of the ANOVA of the vegetation composition variables between habitat classes in Cambridgeshire. Significance P<0.05. R^2 *represents the amount of variance explained by the habitat classes.*

Variable	**Df**	**Sum Sq**	**Mean Sq**	***f***	**R^2**	**P**
PCov_Ash	5	9556.000	1911.300	19.940	0.76	<0.001
PCov_Oak	5	4240.000	848.100	20.490	0.76	<0.001
PCov_Maple	5	1122.000	224.400	8.383	0.57	<0.001
PCov_Elm	5	348.700	69.730	2.376	0.27	0.061
PCov_BroadOther	5	277.400	55.480	1.372	0.18	0.261
PCov_Cons	5	46.960	9.392	0.848	0.12	0.526
Pres_Thorns	5	0.140	0.028	1.078	0.14	0.391
Pres_Bramble	5	1.179	0.236	1.469	0.19	0.227
Pres_Shrub	5	0.596	0.119	1.762	0.22	0.149
PCov_ImpGrass	5	4521.200	904.230	3.113	0.33	0.021
PCov_Cereal	5	24069.000	4814.000	12.250	0.66	<0.001
PCov_CropOther	5	8227.900	1645.590	3.778	0.37	0.008

Table S 5.6. The significantly different habitat pairs (at P<0.05) responsible for the significant differences in the ANOVA (Table S 5.5) from the post-hoc Tukey test of the vegetation composition variables in bold, non-significant variables excluded.

Habitat Pairs/Variables	***PCov _ Ash***	***PCov _Oak***	***PCov_ Maple***	***PCov_ ImpGrass***	***PCov_ Cereal***	***PCov_ CropOther***
Field_high%_Hedge-Field By_Wood	1.000	1.000	1.000	0.141	0.919	0.999
Field_low%_Hedge-Field By_Wood	1.000	1.000	1.000	**0.043**	**0.010**	0.943
Field_med%_Hedge-Field By_Wood	1.000	1.000	1.000	0.077	0.196	1.000
Wood Edge-Field By_Wood	**0.000**	**0.000**	**0.006**	**0.016**	0.355	0.124
Wood Int-Field By_Wood	**0.000**	**0.000**	**0.009**	**0.024**	0.408	0.159
Field_low%_Hedge-Field_high%_Hedge	1.000	1.000	1.000	0.982	0.070	0.790
Field_med%_Hedge-Field_high%_Hedge	1.000	1.000	1.000	1.000	0.694	0.996
Wood Edge-Field_high%_Hedge	**0.000**	**0.000**	**0.005**	0.968	**0.028**	**0.037**
Wood Int-Field_high%_Hedge	**0.000**	**0.000**	**0.008**	0.974	**0.043**	0.055
Field_med%_Hedge-Field_low%_Hedge	1.000	1.000	1.000	0.999	0.671	0.963
Wood Edge-Field_low%_Hedge	**0.000**	**0.000**	**0.008**	1.000	**0.000**	0.614
Wood Int-Field_low%_Hedge	**0.000**	**0.000**	**0.012**	1.000	**0.000**	0.660
Wood Edge-Field_med%_Hedge	**0.000**	**0.000**	**0.004**	0.997	**0.000**	0.123
Wood Int-Field_med%_Hedge	**0.000**	**0.000**	**0.006**	0.998	**0.001**	0.162
Wood Int-Wood Edge	0.901	0.857	1.000	1.000	1.000	1.000

Table S 5.7. ANOVA results for the variance in the vegetation structural variables based on the habitat classification in Cambridgeshire. Significance: $P < 0.05$. R^2 represents the amount of variance in the index explained by the habitat classes.

Variable	Df	Sum Sq	Mean Sq	*f*	R^2	P
Ht_Av	5	1041.30	208.30	34.73	0.84	<0.001
Ht_StDev	5	169.98	34.00	36.20	0.85	<0.001
Ht_VDR	5	0.71	0.14	4.24	0.40	0.005
Ostorey_Ht	5	306.80	61.36	14.17	0.69	<0.001
Ostorey_PenDepth	5	48.80	9.76	5.44	0.46	0.001
PCov_<0.5m	5	28168.00	5634.00	13.34	0.68	<0.001
PCov_0.5-2m	5	5733.00	1146.60	3.05	0.32	0.023
PCov_2-5m	5	446.10	89.22	4.55	0.42	0.003
PCov_>5m	5	44752.00	8950.00	55.05	0.90	<0.001
P_HedgeLen	5	57217.00	11443.00	84.27	0.93	<0.001

Table S 5.8. Results of the post-hoc Tukey test showing the significantly different habitat pairs responsible for the significant differences in the ANOVA (Table S 5.7) for vegetation structural metrics in bold; non-significant variables excluded (at P <0.05).

Habitat Pairs/ Variables	***Ht_Av***	***Ht_StDev***	***Ht_VDR***	***Ostorey_Ht***	***Ostorey_Pen Depth***	***PCov_<0.5m***	***PCov_0.5-2m***	***PCov_2-5m***	***PCov_>5m***	***P_Hedge Len***
Field_high%_Hedge-Field By_Wood	1.000	0.962	0.909	0.994	1.000	0.741	0.894	0.882	1.000	**0.000**
Field_low%_Hedge-Field By_Wood	1.000	1.000	0.877	0.813	0.965	0.999	0.999	1.000	1.000	0.999
Field_med%_Hedge-Field By_Wood	1.000	0.997	0.992	0.653	1.000	1.000	1.000	0.999	1.000	**0.000**
Wood Edge-Field By_Wood	**0.000**	**0.000**	0.133	**0.007**	0.251	0.001	0.368	**0.019**	**0.000**	0.231
Wood Int-Field By_Wood	**0.000**	**0.000**	0.228	**0.006**	**0.034**	**0.000**	0.505	0.137	**0.000**	0.278
Field_low%_Hedge-Field_high%_Hedge	1.000	0.915	0.280	0.972	0.877	0.534	0.715	0.912	1.000	**0.000**
Field_med%_Hedge - Field_high%_Hedge	0.999	0.759	0.563	0.904	0.997	0.581	0.768	0.966	0.999	**0.000**
Wood Edge-Field_high%_Hedge	**0.000**	**0.000**	**0.006**	**0.001**	0.331	**0.021**	**0.025**	0.192	**0.000**	**0.000**
Wood Int-Field_high%_Hedge	**0.000**	**0.000**	**0.015**	**0.001**	**0.045**	**0.009**	0.053	0.661	**0.000**	**0.000**
Field_med%_Hedge-Field_low%_Hedge	1.000	1.000	0.992	1.000	0.986	1.000	1.000	1.000	1.000	**0.000**
Wood Edge-Field_low%_Hedge	**0.000**	**0.000**	0.768	**0.000**	**0.038**	**0.000**	0.604	**0.023**	**0.000**	0.414
Wood Int-Field_low%_Hedge	**0.000**	**0.000**	0.878	**0.000**	**0.004**	**0.000**	0.735	0.161	**0.000**	0.468
Wood Edge-Field_med%_Hedge	**0.000**	**0.000**	0.329	**0.000**	0.130	**0.000**	0.437	**0.029**	**0.000**	**0.000**
Wood Int-Field_med%_Hedge	**0.000**	**0.000**	0.489	**0.000**	**0.013**	**0.000**	0.588	0.209	**0.000**	**0.000**
Wood Int-Wood Edge	0.991	1.000	1.000	1.000	0.817	0.992	1.000	0.961	0.924	1.000

Table S 5.9. The extra woodland variables for each of the woodland transects in Cambridgeshire. See Table 3.5 for metric derivation.

T_ No.	**Habitat Class**	***Wood. Area* (ha)**	***Wood. Perimeter* (m)**	***Wood_ P:A* (m)**	***%_Woods _500m***	***%_Woods _1km***	***Dist_ Wood* (m)**
23	**Wood Edge**	170.00	5819	0.003	18.66	43.07	508.17
24	**Wood Edge**	170.00	5819	0.003	18.66	43.07	188.69
25	**Wood Edge**	64.07	3631	0.006	0.00	6.72	941.21
26	**Wood Edge**	19.93	2585	0.013	74.78	286.65	76.18
27	**Wood Edge**	29.79	3571	0.012	4.24	76.76	498.92
28	**Wood Edge**	74.70	4392	0.006	29.12	39.62	129.59
29	**Wood Edge**	74.70	4392	0.006	29.12	39.62	280.69
30	**Wood Edge**	64.07	3631	0.006	0.00	6.72	834.18
31	**Wood Edge**	170.00	5819	0.003	18.66	43.07	176.48
32	**Wood Interior**	170.00	5819	0.003	18.66	43.07	759.84
33	**Wood Interior**	170.00	5819	0.003	18.66	43.07	498.29
34	**Wood Interior**	64.07	3631	0.006	0.00	6.72	966.00
35	**Wood Interior**	29.06	2195	0.008	58.16	217.02	282.71
36	**Wood Interior**	4.72	927	0.020	173.15	193.49	317.42
37	**Wood Interior**	6.70	1136	0.017	14.44	122.52	505.90
38	**Wood Interior**	8.21	1123	0.014	9.68	81.62	511.31

Table S 5.10. The ANOVA results for the variance in the extra woodland variables based on the woodland habitat classes. Significance at P <0.05.

Variable	**Df**	**Sum Sq**	**Mean Sq**	***f***	**R^2**	**P**
Wood.Area	1	3164	3164	0.702	0.05	0.416
Wood.Perimeter	1	8353619	8353619	2.967	0.17	0.107
Wood_P:A	1	0.0001	0.0001	1.918	0.12	0.188
%_Woods_500m	1	1631	1631	0.867	0.06	0.367
%_Woods_1km	1	5114	5114	0.736	0.05	0.405
Dist_Wood	1	82776	82776	1.018	0.07	0.330

Table S 5.11. The bird indices calculated from the corrected bird densities for the field survey areas (transects) and the classification in Cambridgeshire. See Table 3.6 for variable explanation.

T_ No.	**Habitat Class**	***Bird_ Density***	***Spp_ Richness***	***Spp_ Diversity***	***Spp_ Decline***	***Spp_ Priority***	***Spp_ Priority^w***	***Spp_ Rarity***	***Spp_ IRR***
1	**Field_low% _Hedge**	4.782	18	2.594	5	5	1.880	6	0.005
2	**Field_low% _Hedge**	3.774	11	2.025	6	6	2.486	1	0.000
3	**Field_low% _Hedge**	5.286	20	2.473	7	9	2.627	11	0.044
4	**Field_low% _Hedge**	5.044	17	2.189	5	5	2.889	20	0.227
5	**Field_low% _Hedge**	5.397	16	2.457	5	5	2.580	9	0.003
6	**Field_med% _Hedge**	4.073	17	2.514	6	6	1.660	6	0.004
7	**Field_med% _Hedge**	4.414	22	2.714	9	11	2.123	12	0.003
8	**Field_med% _Hedge**	10.326	23	2.854	8	12	4.739	12	0.038
9	**Field_med% _Hedge**	6.106	18	2.625	7	7	2.806	15	0.092
10	**Field_med% _Hedge**	8.909	18	2.697	7	9	4.406	11	0.047
11	**Field_med% _Hedge**	5.722	13	2.268	6	6	3.442	6	0.005
12	**Field_high% _Hedge**	9.256	23	2.504	10	12	8.477	19	0.072
13	**Field_high% _Hedge**	9.166	19	2.659	6	6	4.230	7	0.001
14	**Field_high% _Hedge**	7.040	20	2.634	7	7	2.862	13	0.031
15	**Field_high% _Hedge**	10.353	26	2.874	10	13	6.898	22	0.067
16	**Field_high% _Hedge**	15.444	27	2.933	10	11	6.842	20	0.031
17	**Field_high% _Hedge**	6.407	16	2.586	6	6	2.633	6	0.002
18	**Field By_Wood**	3.444	10	2.006	3	3	1.744	3	0.002
19	**Field By_Wood**	6.069	12	2.015	3	3	1.371	0	0.000
20	**Field By_Wood**	8.168	18	2.165	6	7	1.358	20	0.050
21	**Field By_Wood**	3.369	8	1.783	3	3	1.451	3	0.005
22	**Field By_Wood**	12.843	8	1.441	6	7	9.673	5	0.070

Table S 5.12. The bird indices calculated from the corrected bird densities for the woodland survey areas (transects) and the classification in Cambridgeshire. See Table 3.6 for variable explanation.

T_ No.	**Habitat Class**	***Bird_ Density***	***Spp_ Richness***	***Spp_ Diversity***	***Spp_ Decline***	***Spp_ Priority***	***Spp_ Priority^{w}***	***Spp_ Rarity***	***Spp_ IRR***
23	**Wood Edge**	38.493	22	2.613	8	10	17.797	29	0.542
24	**Wood Edge**	30.989	20	2.632	7	11	15.619	19	0.125
25	**Wood Edge**	36.746	21	2.598	7	8	20.063	28	0.443
26	**Wood Edge**	42.746	24	2.804	8	10	16.392	32	0.231
27	**Wood Edge**	44.824	28	2.907	9	13	20.854	33	0.188
28	**Wood Edge**	50.344	25	2.758	8	10	21.776	30	0.161
29	**Wood Edge**	50.067	24	2.698	6	7	19.105	22	0.103
30	**Wood Edge**	56.024	20	2.722	7	9	23.828	18	0.237
31	**Wood Edge**	59.291	20	2.722	7	10	25.933	25	0.338
32	**Wood Interior**	29.624	20	2.608	6	6	10.696	20	0.060
33	**Wood Interior**	25.516	19	2.619	6	7	10.333	21	0.244
34	**Wood Interior**	32.021	24	2.836	9	11	11.568	32	0.632
35	**Wood Interior**	29.477	25	2.764	7	8	6.452	29	0.130
36	**Wood Interior**	44.687	22	2.782	6	7	16.733	20	0.394
37	**Wood Interior**	49.066	22	2.724	7	8	17.090	24	0.260
38	**Wood Interior**	42.960	22	2.735	7	8	19.008	26	0.184

Table S 5.13. Results of the ANOVA showing the variation in each of the bird indices between the habitat classes in Cambridgeshire. Significance: P <0.05. R^2 represents the amount of variance in the index that is explained by the habitat classes.

Variables	**Df**	**Sum Sq**	**Mean Sq**	***f***	**R^2**	**P**
Bird_Density	5	11347.00	2269.30	54.20	0.89	<0.001
Spp_Richness	5	627.00	125.39	11.51	0.64	<0.001
Spp_Diversity	5	4.08	0.82	24.13	0.79	<0.001
Spp_Decline	5	56.62	11.32	6.62	0.51	<0.001
Spp_Priority	5	116.00	23.19	4.82	0.43	0.002
Spp_Priority^{w}	5	1903.40	380.70	39.74	0.86	<0.001
Spp_Rarity	5	2340.00	467.90	13.30	0.68	<0.001
Spp_IRR	5	0.49	0.10	6.70	0.51	<0.001

Table S 5.14. The results of the post-hoc Tukey test displaying the significantly different habitat class pairs (in bold) in terms of each bird index in Cambridgeshire; non-significant variables are excluded (at P <0.05).

Habitat Classes/ Bird Index	***Bird_ Density***	***Spp_ Richness***	***Spp_ Diversity***	***Spp_ Decline***	***Spp_ Priority***	***Spp_ Priority***^{w}	***Spp_ Rarity***	***Spp_ IRR***
Field_high% _Hedge – Field By_Wood	0.978	**0.000**	**0.000**	**0.000**	**0.019**	0.845	0.219	1.000
Field_low% _Hedge – Field By_Wood	0.997	0.188	**0.001**	0.546	0.911	1.000	0.955	0.997
Field_med% _Hedge – Field By_Wood	1.000	**0.007**	**0.000**	0.009	0.062	1.000	0.856	1.000
Wood Edge – Field By_Wood	**0.000**	**0.000**	**0.000**	**0.001**	**0.002**	**0.000**	**0.000**	**0.015**
Wood Int – Field By_Wood	**0.000**	**0.000**	**0.000**	**0.017**	0.144	**0.000**	**0.000**	**0.017**
Field_low% _Hedge – Field_high% _Hedge	0.827	0.092	**0.019**	**0.030**	0.192	0.660	0.715	0.999
Field_med% _Hedge – Field_high% _Hedge	0.964	0.675	0.994	0.770	0.995	0.838	0.826	1.000
Wood Edge – Field_high% _Hedge	**0.000**	0.953	0.974	0.898	0.995	**0.000**	**0.008**	**0.013**
Wood Int – Field_high% _Hedge	**0.000**	0.991	0.955	0.480	0.888	**0.001**	**0.047**	**0.016**
Field_med% _Hedge – Field_low% _Hedge	0.998	0.764	0.063	0.376	0.431	0.999	1.000	0.999
Wood Edge – Field_low% _Hedge	**0.000**	**0.007**	**0.001**	0.146	**0.044**	**0.000**	**0.000**	0.053
Wood Int – Field_low% _Hedge	**0.000**	**0.021**	**0.002**	0.578	0.699	**0.000**	**0.002**	0.058
Wood Edge – Field_med% _Hedge	**0.000**	0.157	0.763	0.998	0.876	**0.000**	**0.000**	**0.011**
Wood Int– Field_med%_ Hedge	**0.000**	0.300	0.722	0.998	0.995	**0.000**	**0.002**	**0.014**
Wood Int– Wood Edge	0.074	1.000	1.000	0.946	0.518	**0.001**	0.993	1.000

Table S 5.15. Spp_Richness and Spp_Diversity of the combined (if more than one) woodland transects, total, per m transect length and per hectare of woodland area. Variable explanations in Table 3.5.

Woodland	**Area (ha)**	**T_ Length**	***Total Spp_ Richness***	***Spp_ Richness/ T_Length***	***Spp_ Richness per ha***	***Total Spp_ Diversity***	***Spp_ Diversity/ T_Length***	***Spp_ Diversity per ha***
Archers	19.9	1298	24	0.018	1.204	2.804	0.002	0.141
Aversley	64.1	2768	28	0.010	0.437	2.874	0.001	0.045
Gamsey	4.7	475	22	0.046	4.659	2.782	0.006	0.589
Lady's	8.2	730	22	0.030	2.678	2.735	0.004	0.333
Little Less	29.8	1024	28	0.027	0.940	2.907	0.003	0.098
Monks	170.0	3640	29	0.008	0.171	2.803	0.001	0.016
Raveley	6.7	647	22	0.034	3.281	2.724	0.004	0.406
Upton	29.1	811	25	0.031	0.860	2.764	0.003	0.095
Wennington	74.7	2155	29	0.013	0.388	2.783	0.001	0.037

Table S 5.16. One sample t-test results showing significant differences at P<0.05, between the woodland Spp_Richness and Spp_Diversity metrics.

Variable	**t**	**Df**	**P**	**Mean**	**Lower CI**	**Upper CI**
Total Spp_Richness	24.73	8	<0.001	25.444	23.072	27.817
Spp_Richness/T_Length	5.70	8	0.001	0.024	0.014	0.034
Spp_Richness per ha	3.13	8	0.014	1.624	0.429	2.819
Total Spp_Diversity	139.88	8	<0.001	2.797	2.751	2.843
Spp_Diversity/T_Length	4.86	8	0.001	0.003	0.001	0.004
Spp_Diversity per ha	2.93	8	0.019	0.196	0.042	0.349

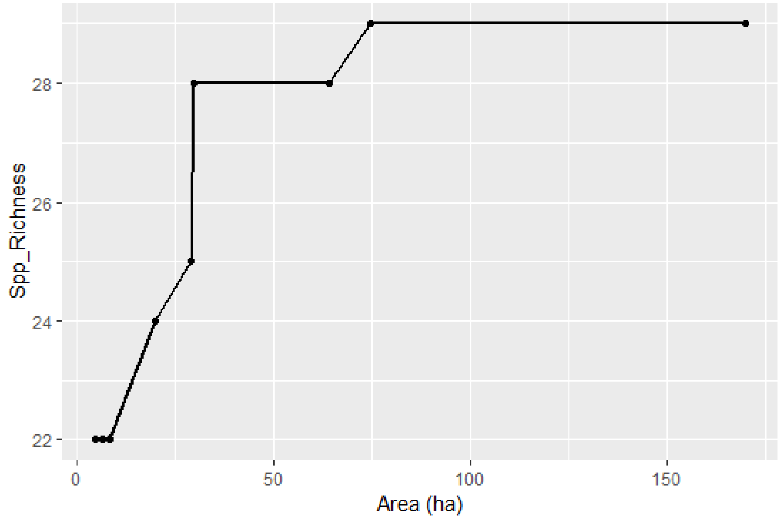
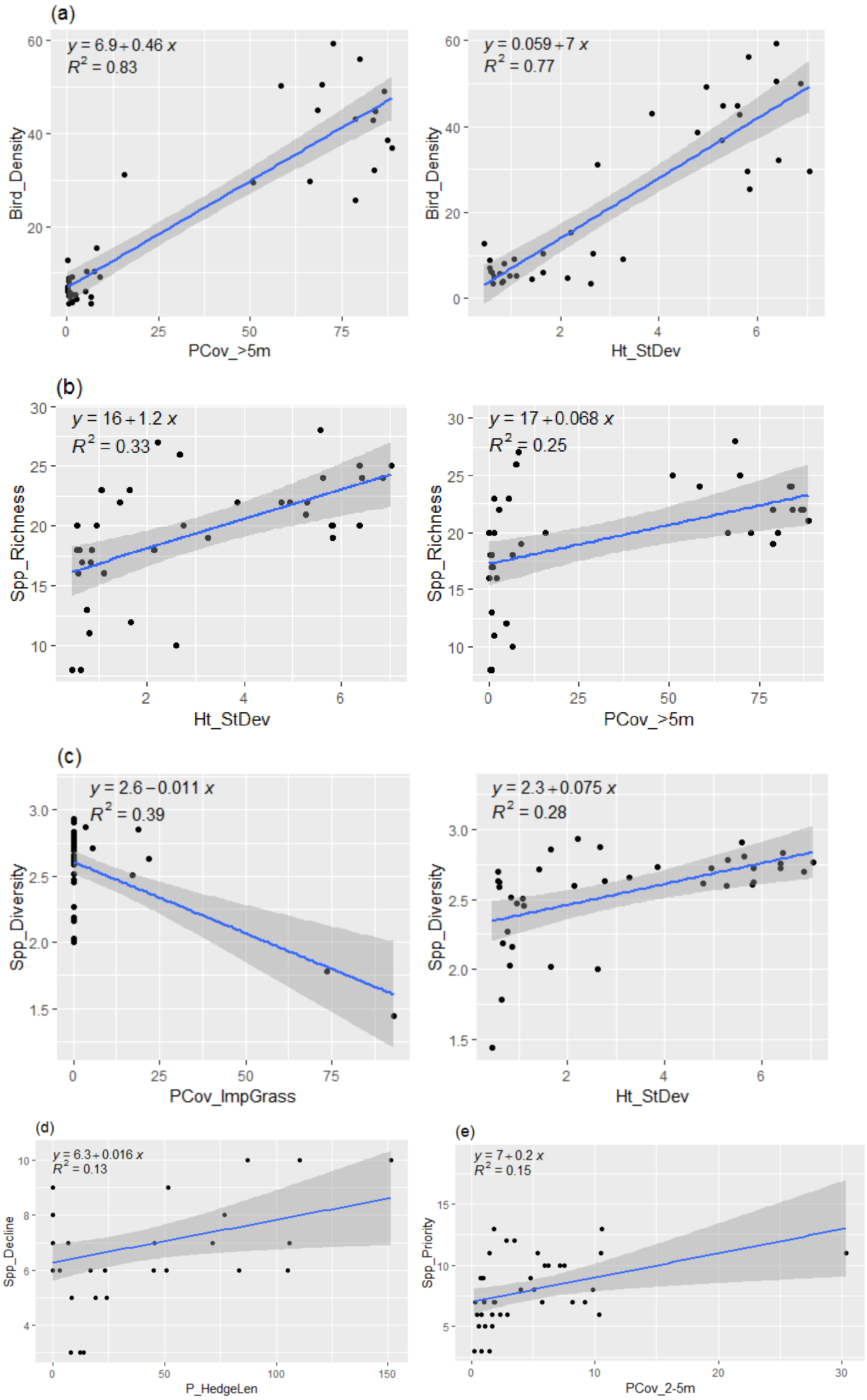


Figure S 5.1. Spp_Richness plotted against woodland area in hectares (R version 3.5.2).


(a)
y = 6.9 + 0.46 x
R² = 0.83
Bird_Density
60
40
20
0
25
50
75
PCov_>5m
y = 0.059 + 7 x
R² = 0.77
Bird_Density
60
40
20
0
2
4
6
Ht_StDev
(b)
y = 16 + 1.2 x
R² = 0.33
Spp_Richness
30
25
20
15
10
2
4
6
Ht_StDev
y = 17 + 0.068 x
R² = 0.25
Spp_Richness
30
25
20
15
10
0
25
50
75
PCov_>5m
(c)
y = 2.6 − 0.011 x
R² = 0.39
Spp_Diversity
3.0
2.5
2.0
1.5
0
25
50
75
PCov_ImpGrass
y = 2.3 + 0.075 x
R² = 0.28
Spp_Diversity
3.0
2.5
2.0
1.5
2
4
6
Ht_StDev
(d)
y = 6.3 + 0.016 x
R² = 0.13
Spp_Decline
10
8
6
4
0
50
100
150
P_HedgeLen
(e)
y = 7 + 0.2 x
R² = 0.15
Spp_Priority
15
10
5
0
10
20
30
PCov_2-5m

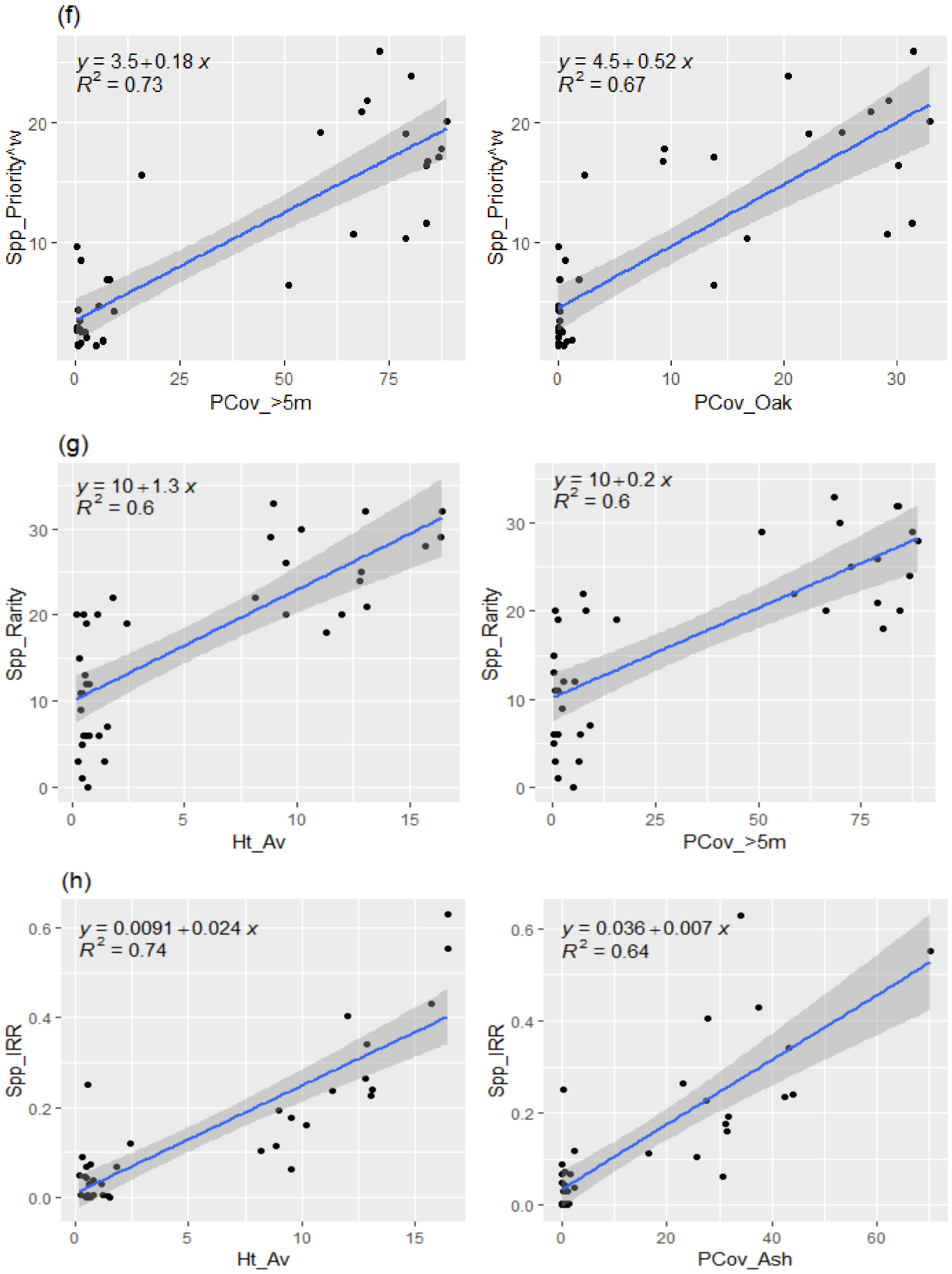
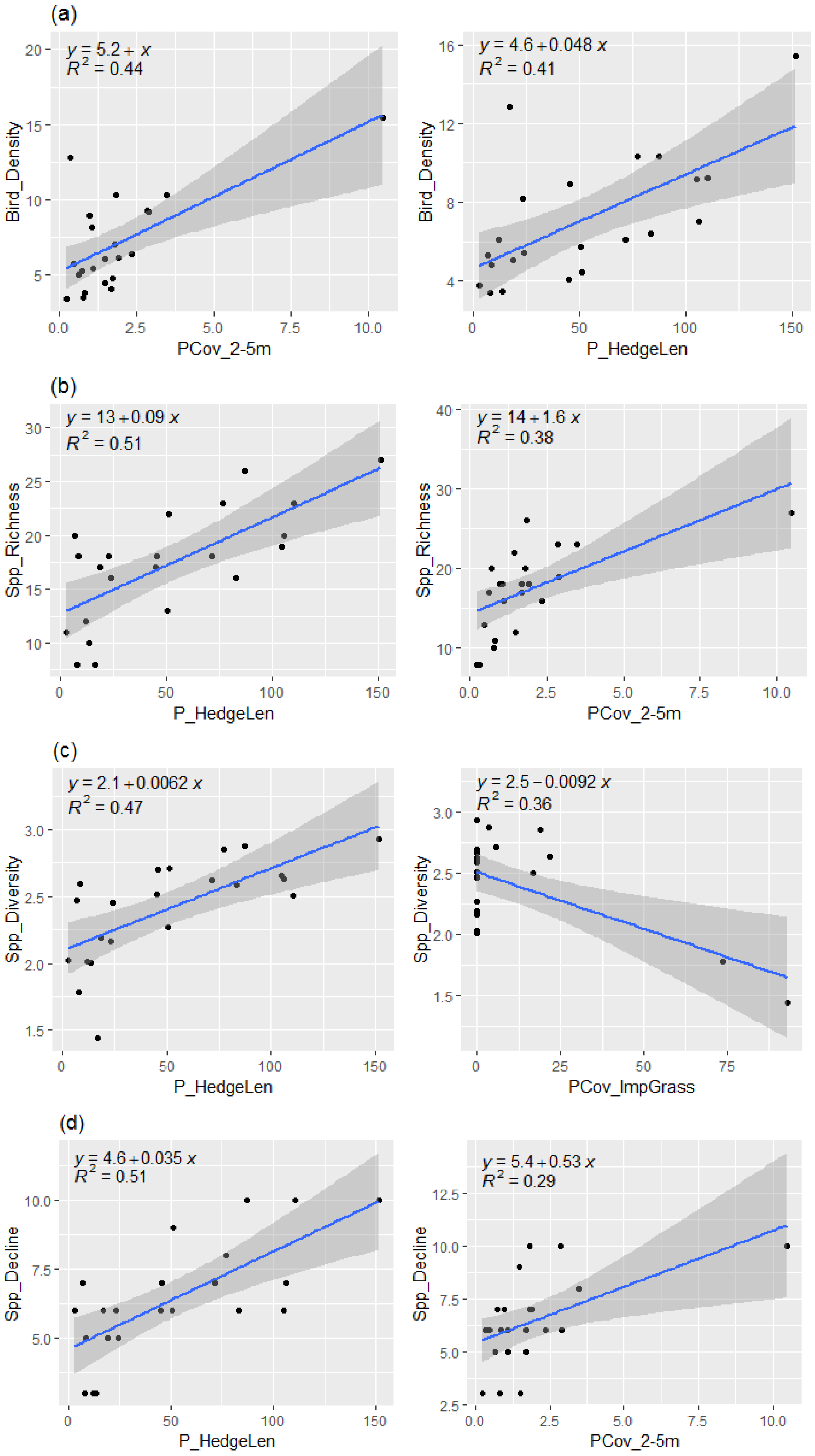


Figure S 5.2. Linear regression graphs for the top two significant variables with the highest Pearson's r for each bird index (a) – (h) in the landscape correlation matrix in Table 5.2 (P <0.05). Confidence intervals are shown as the standard error. For variable derivation see Table 3.2, Table 3.3, Table 3.4 and Table 3.6 (R version 3.5.2).


(a)
y = 5.2 + x
R2 = 0.44
Bird_Density
PCov_2-5m
y = 4.6 + 0.048 x
R2 = 0.41
Bird_Density
P_HedgeLen
(b)
y = 13 + 0.09 x
R2 = 0.51
Spp_Richness
P_HedgeLen
y = 14 + 1.6 x
R2 = 0.38
Spp_Richness
PCov_2-5m
(c)
y = 2.1 + 0.0062 x
R2 = 0.47
Spp_Diversity
P_HedgeLen
y = 2.5 − 0.0092 x
R2 = 0.36
Spp_Diversity
PCov_ImpGrass
(d)
y = 4.6 + 0.035 x
R2 = 0.51
Spp_Decline
P_HedgeLen
y = 5.4 + 0.53 x
R2 = 0.29
Spp_Decline
PCov_2-5m

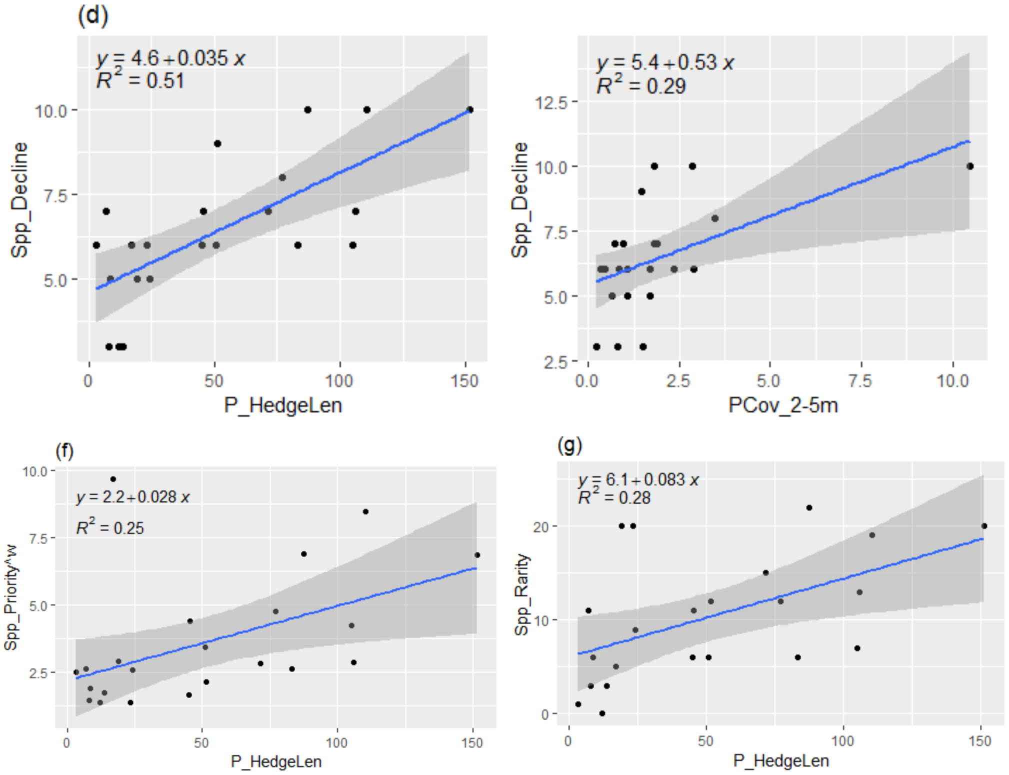


Figure S 5.3. Linear regression graphs for the top two significant variables with the highest Pearson's r for each bird index (a) – (g) in the field correlation matrix in Table 5.3, Spp_IRR excluded as there were no significant correlated variables (P <0.05). Confidence intervals are shown as the standard error. For variable derivation see Table 3.2, Table 3.3, Table 3.4 and Table 3.6 (R version 3.5.2).

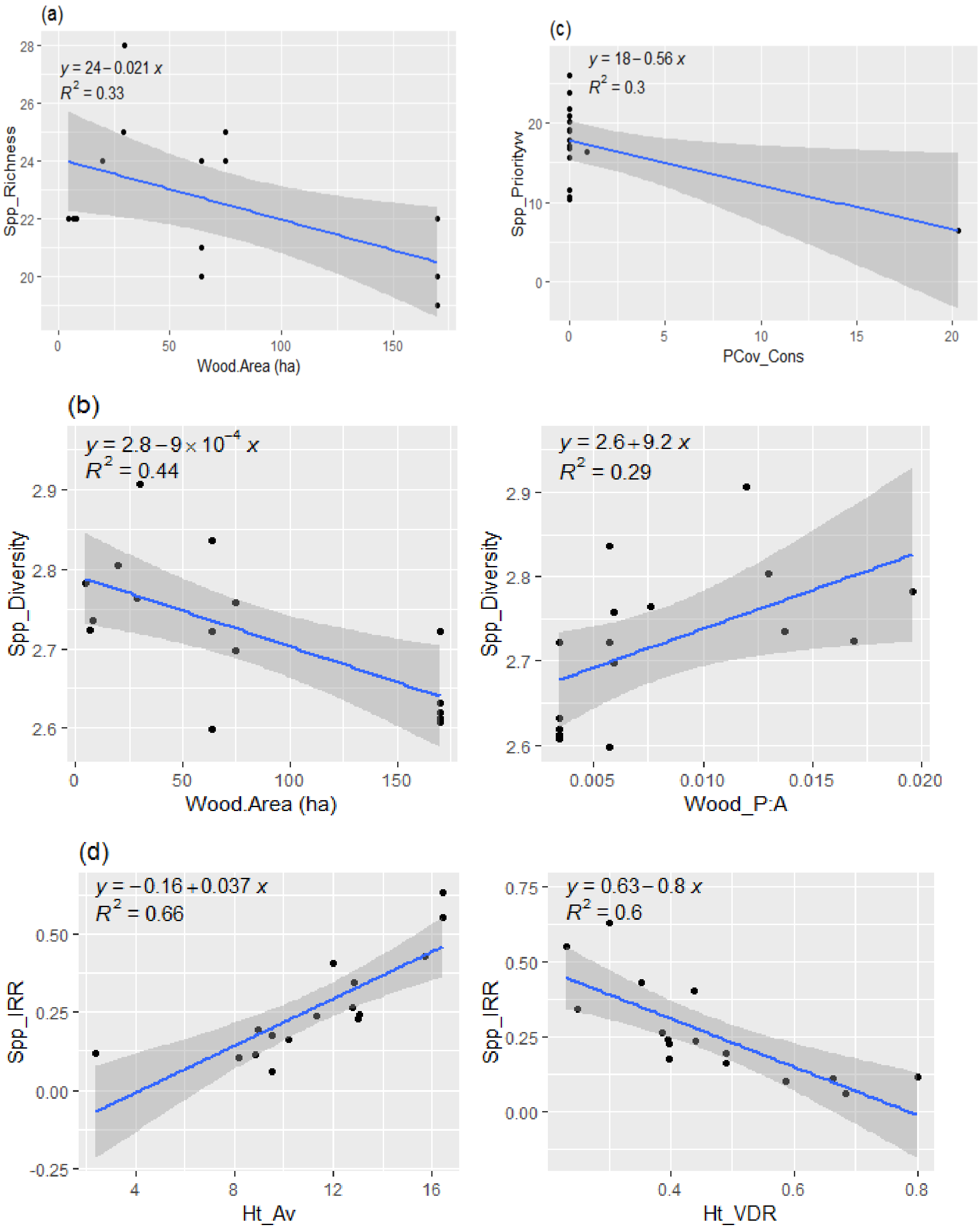


Figure S 5.4. Regression graphs for the significant relationships from the woodland only correlation matrix in Table 5.4 (P <0.05). Confidence intervals are shown as the standard error. For variable derivation see Table 3.2, Table 3.3, Table 3.4, Table 3.5 and Table 3.6 (R version 3.5.2).

Table S 5.17. Correlation matrix of all the habitat variables used in the in Cambridgeshire study.

Variables	**PCov_ Ash**	**PCov_ Oak**	**PCov_ Maple**	**PCov_ Elm**	**PCov_ Broad Other**	**PCov_ Cons**	**Pres_ Thorns**	**Pres_ Bramble**	**Pres_ Shrub**	**PCov_ Imp Grass**	**PCov_ Cereal**	**PCov_ Crop Other**	**P_ Hedge Len**	**Ht_ Av**	**Ht_ StDev**	**Ht_ VDR**	**Ostorey _Ht**	**Ostorey _Pen Depth**	**PCov_ <0.5m**	**PCov_ 0.5-2m**	**PCov_ 2-5m**
PCov_Ash	1																				
PCov_Oak	**0.798**	1																			
PCov_Maple	**0.714**	**0.643**	1																		
PCov_Elm	0.206	0.162	**0.557**	1																	
PCov_Broad Other	0.054	0.162	0.191	0.163	1																
PCov_Cons	0.027	0.073	-0.089	0.014	-0.095	1															
Pres_ Thorns	0.124	0.123	0.112	0.059	0.105	0.029	1														
Pres_ Bramble	-0.192	-0.031	-0.089	-0.063	-0.098	0.088	-0.085	1													
Pres_Shrub	0.215	0.222	0.179	0.087	0.103	0.048	-0.048	-0.151	1												
PCov_Imp Grass	-0.242	-0.244	-0.219	-0.111	-0.188	-0.056	0.053	0.102	0.076	1											
PCov_ Cereal	**-0.606**	**-0.602**	**-0.526**	-0.266	-0.246	-0.139	-0.248	-0.113	-0.084	-0.184	1										
PCov_Crop Other	**-0.499**	**-0.506**	**-0.430**	-0.213	-0.220	-0.116	0.092	0.187	**-0.381**	-0.004	0.019	1									
P_Hedge Len	**-0.537**	**-0.535**	**-0.461**	-0.215	-0.050	-0.125	-0.087	0.132	-0.280	0.052	0.394	**0.384**	1								
Ht_Av	**0.940**	**0.865**	**0.769**	**0.376**	0.171	0.114	0.134	-0.169	0.216	-0.266	**-0.652**	**-0.515**	**-0.557**	1							
Ht_StDev	**0.825**	**0.877**	**0.644**	0.298	0.240	0.292	0.165	-0.056	0.217	**-0.330**	**-0.668**	**-0.477**	**-0.514**	**0.892**	1						
Ht_VDR	**-0.640**	**-0.535**	**-0.528**	-0.224	0.197	0.046	-0.067	0.281	-0.081	-0.010	0.267	**0.445**	**0.580**	**-0.634**	**-0.384**	1					
Ostorey_Ht	**0.858**	**0.794**	**0.610**	0.276	0.106	0.199	0.145	-0.122	0.201	-0.218	**-0.673**	**-0.351**	**-0.511**	**0.925**	**0.889**	**-0.490**	1				
Ostorey_Pen Depth	**-0.510**	**-0.524**	**-0.482**	**-0.340**	-0.219	-0.271	0.134	-0.250	0.068	0.118	**0.605**	0.222	0.317	**-0.590**	**-0.610**	0.195	**-0.564**	1			
PCov_<0.5m	**-0.756**	**-0.713**	**-0.690**	-0.353	-0.266	-0.112	-0.110	-0.033	-0.011	0.269	**0.738**	0.213	**0.422**	**-0.823**	**-0.805**	**0.371**	**-0.778**	**0.692**	1		
PCov_0.5-2m	**-0.482**	**-0.478**	**-0.434**	-0.216	-0.135	0.016	-0.101	0.273	**-0.415**	0.106	0.083	**0.676**	**0.370**	**-0.477**	**-0.394**	**0.462**	**-0.344**	0.039	-0.045	1	
PCov_2-5m	**0.342**	**0.364**	**0.329**	0.151	**0.855**	-0.017	0.129	0.010	0.156	-0.225	**-0.406**	**-0.391**	-0.210	**0.362**	**0.430**	-0.035	0.233	**-0.351**	**-0.406**	**-0.305**	1
PCov_>5m	**0.927**	**0.883**	**0.841**	**0.429**	0.198	0.095	0.141	-0.134	0.235	-0.276	**-0.664**	**-0.541**	**-0.575**	**0.982**	**0.907**	**-0.609**	**0.882**	**-0.604**	**-0.827**	**-0.510**	**0.406**

Table S 5.18. Correlation matrix of the bird variables in Cambridgeshire.

Bird Variables	***Bird_ Density***	***Spp_ Richness***	***Spp_ Diversity***	***Spp_ Decline***	***Spp_ Priority***	***Spp_ Priority^w***	***Spp_ Rarity***	***Spp_ IRR***
Bird_ Density	1							
Spp_ Richness	**0.537**	1						
Spp_ Diversity	**0.485**	**0.899**	1					
Spp_ Decline	0.288	**0.796**	**0.669**	1				
Spp_ Priority	**0.387**	**0.777**	**0.648**	**0.937**	1			
Spp_ Priority^w	**0.965**	**0.502**	**0.431**	**0.350**	**0.460**	1		
Spp_ Rarity	**0.758**	**0.804**	**0.636**	**0.610**	**0.631**	**0.730**	1	
Spp_IRR	**0.664**	**0.389**	**0.331**	0.290	**0.330**	**0.662**	**0.713**	1

Table S 5.19. Model selection tables of the top six AICc ranked models in each candidate set from global models with the appropriate family and link function containing selected independent variables. ΔAICc = difference in AICc between model and top model. w_i = Aikaike model weight. acc w_i = cumulative model weight. For variable derivation see *Table 3.2, Table 3.3, Table 3.4 and Table 3.6. *Spp_IRR was +0.0001 to nudge values from 0 (Thomas et al. 2017).*

a) Global Model = Bird_Density ~ PCov_Elm + PCov_Ash + PCov_Oak + PCov_2-5m + PCov_>5m, family= Gamma(link = identity)

Candidate Models	**df**	**logLik**	**AICc**	**ΔAICc**	w_i	**acc** w_i	**adjR^2**
PCov_2-5m + PCov_>5m	4	-113.32	235.86	0	0.31	0.31	0.88
PCov_2-5m + PCov_Ash + PCov_Elm	5	-112.24	236.35	0.50	0.24	0.55	0.89
PCov_2-5m + PCov_Elm + PCov_Oak	5	-112.44	236.75	0.89	0.20	0.75	0.89
PCov_2-5m + PCov_Ash	4	-114.05	237.32	1.46	0.15	0.90	0.88
PCov_2-5m + PCov_>5m + PCov_Elm	5	-113.25	238.38	2.53	0.09	0.99	0.88
PCov_2-5m + PCov_Oak	4	-116.16	241.53	5.68	0.02	1.01	0.86

b) Global Model = Spp_Richness ~ PCov_Ash + PCov_Oak + PCov_Maple + PCov_BroadOther + PCov_ImpGrass + PCov_Cereal + Ht_Av + Ht_StDev + PCov_2-5m + PCov_>5m, family= Poisson (link= identity)

Candidate Models	**df**	**logLik**	**AICc**	**ΔAICc**	w_i	**acc** w_i	**Pseudo-R^2**
PCov_ImpGrass + Ht_StDev	3	-105.34	217.39	0	0.34	0.34	0.45
PCov_ImpGrass + Ht_StDev + PCov_2-5m	4	-104.74	218.69	1.30	0.18	0.52	0.47
PCov_ImpGrass + PCov_Oak	3	-106.36	219.42	2.03	0.12	0.64	0.42
PCov_ImpGrass + PCov_Oak + PCov_2-5m	4	-105.39	219.99	2.60	0.09	0.73	0.45
PCov_ImpGrass + PCov_2-5m	3	-107.06	220.82	3.43	0.06	0.79	0.40
PCov_ImpGrass + PCov_2-5m + PCov_Ash	4	-106.07	221.35	3.96	0.05	0.84	0.43
PCov_ImpGrass + PCov_Ash	3	-107.33	221.38	3.99	0.05	0.89	0.39
PCov_ImpGrass + PCov_Maple + PCov_2-5m	4	-106.47	222.15	4.76	0.03	0.92	0.42
Ht_StDev	2	-108.92	222.19	4.80	0.03	0.95	0.34
PCov_ImpGrass + PCov_Maple	3	-107.86	222.42	5.03	0.03	0.98	0.38
Ht_StDev + PCov_2-5m	3	-108.02	222.74	5.35	0.02	1	0.37

c) Global Model = Spp_Diversity ~ PCov_Ash + PCov_Oak + PCov_Maple + PCov_BroadOther + PCov_ImpGrass + Ht_StDev+ PCov_<0.5m + PCov_2-5m, family=Gamma(link=identity)

Candidate Models	**df**	**logLik**	**AICc**	**ΔAICc**	w_i	**acc** w_i	**adjR^2**
PCov_ImpGrass + Ht_StDev	4	0.64	7.93	0	0.18	0.18	0.55
PCov_ImpGrass + Ht_StDev + PCov_2-5m136	5	1.47	8.93	1.00	0.11	0.29	0.57
PCov_ImpGrass + Ht_StDev + PCov_BroadOther	5	1.13	9.61	1.68	0.08	0.37	0.56
PCov_ImpGrass + PCov_2-5m	4	-0.22	9.64	1.71	0.08	0.45	0.53
PCov_ImpGrass + PCov_Oak	4	-0.27	9.74	1.81	0.07	0.52	0.52
PCov_ImpGrass + PCov_Oak + PCov_2-5m	5	0.92	10.04	2.11	0.06	0.58	0.55
PCov_ImpGrass + PCov_<0.5m	4	-0.50	10.21	2.28	0.06	0.64	0.52
PCov_ImpGrass + PCov_<0.5m+ PCov_2-5m	5	0.68	10.52	2.59	0.05	0.69	0.55
PCov_ImpGrass + PCov_Ash + PCov_2-5m	5	0.55	10.78	2.85	0.04	0.73	0.54
PCov_ImpGrass + PCov_Ash	4	-0.82	10.85	2.92	0.04	0.77	0.51
PCov_ImpGrass + PCov_Maple + PCov_2-5m	5	0.50	10.88	2.95	0.04	0.81	0.54
PCov_ImpGrass + PCov_Maple	4	-0.83	10.88	2.95	0.04	0.85	0.51
PCov_ImpGrass + PCov_BroadOther + PCov_Oak	5	0.47	10.93	3.00	0.04	0.89	0.54
PCov_ImpGrass + PCov_BroadOther + PCov_Ash	5	0.19	11.49	3.56	0.03	0.92	0.54
PCov_ImpGrass	3	-2.55	11.80	3.87	0.03	0.95	0.46
PCov_ImpGrass + PCov_BroadOther + PCov_<0.5m	5	0.02	11.83	3.89	0.03	0.98	0.53
PCov_ImpGrass + PCov_BroadOther + PCov_Maple	5	-0.15	12.17	4.24	0.02	1.00	0.53
PCov_ImpGrass + PCov_BroadOther	4	-1.58	12.37	4.44	0.02	1.02	0.49

d) Global Model = Spp_Decline~ P_HedgeLen + PCov_2-5m + PCov_<0.5m, family=poisson(link=identity)

Candidate Models	**df**	**logLik**	**AICc**	**ΔAICc**	***w_i***	**acc *w_i***	**Pseudo-R^2**
P_HedgeLen + PCov_<0.5m	3	-76.96	160.62	0	0.24	0.24	0.25
(Null)	1	-79.68	161.47	0.85	0.16	0.40	-
P_HedgeLen	2	-78.69	161.72	1.10	0.14	0.54	0.11
P_HedgeLen + PCov_2-5m	3	-77.54	161.78	1.16	0.14	0.68	0.25
PCov_2-5m	2	-79.03	162.41	1.79	0.10	0.78	0.07
P_HedgeLen + PCov_<0.5m + PCov_2-5m	4	-76.62	162.45	1.84	0.10	0.88	0.35
PCov_<0.5m	2	-79.17	162.69	2.07	0.09	0.97	0.06
PCov_<0.5m + PCov_2-5m	3	-78.86	164.44	3.82	0.04	1.01	0.09

e) Global Model = Spp_Priority~ PCov_Ash + PCov_2-5m + PCov_Oak + Ht_StDev + Ht_Av + PCov_<0.5m, family=poisson(link=identity)

Candidate Models	**df**	**logLik**	**AICc**	**ΔAICc**	***w_i***	**acc *w_i***	**Pseudo-R^2**
PCov_2-5m	2	-88.46	181.27	0	0.21	0.21	0.16
PCov_2-5m + PCov_Oak	3	-88.09	182.88	1.61	0.10	0.31	0.18
PCov_2-5m + PCov_Ash	3	-88.09	182.89	1.62	0.09	0.40	0.18
PCov_2-5m + Ht_Av	3	-88.10	182.91	1.65	0.09	0.49	0.18
PCov_2-5m + Ht_StDev	3	-88.19	183.08	1.81	0.09	0.58	0.17
PCov_2-5m + PCov_<0.5m	3	-88.19	183.09	1.82	0.09	0.67	0.17
Ht_StDev	2	-89.67	183.69	2.42	0.06	0.73	0.09
PCov_Oak	2	-89.73	183.79	2.53	0.06	0.79	0.08
Ht_Av	2	-89.78	183.91	2.64	0.06	0.85	0.08
PCov_<0.5m	2	-89.81	183.97	2.70	0.06	0.91	0.08
PCov_Ash	2	-89.83	184.00	2.74	0.05	0.96	0.08
(Null)	1	-91.22	184.55	3.29	0.04	1	-

f) Global Model = Spp_Priority^{w} ~ PCov_Elm + PCov_Oak + Ht_StDev + PCov_0.5-2m + PCov_2-5m + PCov_>5m, family=Gamma

Candidate Models	**df**	**logLik**	**AICc**	**ΔAICc**	w_i	**acc** w_i	**adjR^2**
PCov_2-5m + PCov_>5m	4	-96.01	201.23	0	0.37	0.37	0.74
PCov_2-5m + PCov_0.5-2m + PCov_Oak	5	-95.37	202.62	1.39	0.18	0.55	0.75
PCov_2-5m + PCov_Oak	4	-96.76	202.73	1.50	0.17	0.72	0.73
PCov_2-5m + PCov_Oak + PCov_Elm	5	-95.82	203.51	2.28	0.12	0.84	0.75
PCov_2-5m + PCov_Elm + PCov_>5m	5	-96.00	203.88	2.65	0.10	0.94	0.74
PCov_>5m	3	-99.07	204.85	3.62	0.06	1	0.70

g) Global Model = Spp_Rarity ~ rescale(PCov_Oak) + rescale(PCov_Maple) + rescale(PCov_ImpGrass) + rescale(PCov_CropOther) + rescale(Ht_VDR) + rescale(PCov_<0.5m) + rescale(PCov_2-5m), link="identity"

Candidate Models	**df**	**logLik**	**AICc**	**ΔAICc**	w_i	**acc** w_i	**Pseudo-R^2**
PCov_Oak + PCov_2-5m	4	-128.93	267.07	0	0.31	0.31	0.44
PCov_Oak + PCov_2-5m + PCov_ImpGrass	5	-127.74	267.35	0.28	0.27	0.58	0.47
PCov_Oak + PCov_ImpGrass	4	-129.22	267.66	0.59	0.23	0.81	0.43
PCov_Oak	3	-130.74	268.18	1.11	0.18	0.99	0.39

h) Global Model = Spp_IRR* ~ rescale(PCov_2-5m) + rescale(PCov_Elm) + rescale(Ht_Av) + rescale(PCov_0.5-2m), link="loglog"

Candidate Models	**df**	**logLik**	**AICc**	**ΔAICc**	w_i	**acc** w_i	**Pseudo-R^2**
Ht_Av	3	71.33	-135.96	0	0.44	0.44	0.69
Ht_Av + PCov_2-5m	4	71.66	-134.12	1.84	0.17	0.61	0.70
Ht_Av + PCov_Elm	4	71.38	-133.54	2.42	0.13	0.74	0.69
Ht_Av + PCov_0.5-2m	4	71.35	-133.50	2.46	0.13	0.87	0.69
Ht_Av + PCov_Elm + PCov_2-5m	5	71.71	-131.54	4.42	0.05	0.92	0.70
Ht_Av + PCov_2-5m + PCov_0.5-2m	5	71.67	-131.46	4.50	0.05	0.97	0.70
Ht_Av + PCov_0.5-2m + PCov_Elm	5	71.40	-130.92	5.04	0.04	1.01	0.69

*Table S 5.20. Model selection tables for field transects in Cambridgeshire. The top six AICc ranked in each candidate set from global models with the appropriate family and link function containing selected and standardised independent variables. ΔAICc = difference in AICc between model and top model. w_i = Aikaike model weight. acc w_i = cumulative model weight. For variable derivation see Table 3.2, Table 3.3, Table 3.4 and Table 3.6. *Spp_IRR with +0.0001 to nudge the values away from 0 (Thomas et al. 2017).*

a) Global Model = Bird_Density ~ PCov_Oak + PCov_Elm + PCov_Maple + P_HedgeLen + PCov_>5m, family=inverse.gaussian(link=identity)

Candidate Models	**df**	**logLik**	**AICc**	**ΔAICc**	w_i	**acc** w_i	**adjR^2**
P_HedgeLen	3	-48.28	103.90	0	0.37	0.37	0.34
P_HedgeLen + PCov_Elm	4	-48.10	106.56	2.66	0.10	0.47	0.35
PCov_2-5m	3	-49.62	106.58	2.68	0.10	0.57	0.25
P_HedgeLen + PCov_Maple	4	-48.11	106.58	2.68	0.10	0.67	0.35
P_HedgeLen + PCov_2-5m	4	-48.26	106.86	2.96	0.08	0.75	0.34
P_HedgeLen + PCov_BroadOther	4	-48.27	106.89	2.98	0.08	0.83	0.34
PCov_2-5m + PCov_Oak	4	-48.78	107.91	4.01	0.05	0.88	0.31
PCov_Elm	3	-50.48	108.30	4.40	0.04	0.92	0.19
PCov_BroadOther	3	-50.85	109.03	5.13	0.03	0.95	0.17
PCov_2-5m + PCov_Elm	4	-49.37	109.09	5.19	0.03	0.98	0.27
PCov_2-5m + PCov_Maple	4	-49.62	109.60	5.70	0.02	1	0.25

b) Global Model = Spp_Richness ~ PCov_Elm + PCov_Maple + PCov_BroadOther + PCov_ImpGrass + P_HedgeLen + PCov_2-5m, family=poisson(link=identity)

Candidate Models	**df**	**logLik**	**AICc**	**ΔAICc**	***w_i***	**acc *w_i***	**Pseudo- R^2**
P_HedgeLen + PCov_ImpGrass	3	-57.68	122.69	0	0.33	0.33	0.64
P_HedgeLen + PCov_Maple	3	-57.75	122.84	0.14	0.31	0.64	0.63
P_HedgeLen + PCov_Elm	3	-58.39	124.12	1.43	0.16	0.80	0.60
P_HedgeLen	2	-60.63	125.89	3.20	0.07	0.87	0.47
PCov_ImpGrass + PCov_2-5m	3	-59.81	126.95	4.26	0.04	0.91	0.52
PCov_ImpGrass + PCov_Elm	3	-60.13	127.59	4.90	0.03	0.94	0.50
PCov_2-5m + PCov_Maple	3	-60.13	127.60	4.91	0.03	0.97	0.50
P_HedgeLen + PCov_2-5m	3	-60.25	127.83	5.14	0.03	1.00	0.49
P_HedgeLen + PCov_BroadOther	3	-60.62	128.57	5.88	0.02	1.02	0.47

c) Global Model = Spp_Diversity~ PCov_Ash + PCov_Elm + PCov_Maple + PCov_BroadOther + PCov_ImpGrass + P_HedgeLen + Ht_Av + PCov_2-5m, family= Gamma (link=identity).

Candidate Model	**df**	**logLik**	**AICc**	**ΔAICc**	**adjR^2**	**Multiple R^2**
P_HedgeLen + PCov_ImpGrass	-	-	1.89	0	0.64	0.67

d) Global Model = Spp_Decline ~ PCov_Maple + PCov_Elm + PCov_Cons + P_HedgeLen + PCov_2-5m, family=poisson(link = identity)

Candidate Models	**df**	**logLik**	**AICc**	**ΔAICc**	w_i	**acc** w_i	**Pseudo- R^2**
P_HedgeLen	2	-44.10	92.82	0	0.30	0.30	0.49
P_HedgeLen + PCov_Cons	3	-43.28	93.89	1.06	0.18	0.48	0.60
P_HedgeLen + PCov_Maple	3	-43.77	94.88	2.06	0.11	0.59	0.53
P_HedgeLen + PCov_Elm	3	-43.95	95.23	2.41	0.09	0.68	0.51
P_HedgeLen + PCov_2-5m	3	-44.09	95.52	2.70	0.08	0.76	0.49
PCov_2-5m	2	-45.69	96.02	3.19	0.06	0.82	0.27
PCov_2-5m + PCov_Cons	3	-44.78	96.90	4.08	0.04	0.86	0.39
PCov_Maple	2	-46.31	97.26	4.44	0.03	0.89	0.19
(Null)	1	-47.67	97.55	4.73	0.03	0.92	-
PCov_Elm	2	-46.51	97.64	4.82	0.03	0.95	0.16
PCov_2-5m + PCov_Maple	3	-45.33	97.99	5.17	0.02	0.97	0.32
PCov_2-5m + PCov_Elm	3	-45.58	98.49	5.67	0.02	0.99	0.29
PCov_Cons	2	-47.03	98.69	5.87	0.02	1.01	0.09

e) Global Model = Spp_Priority~ PCov_Maple + PCov_Elm + P_HedgeLen + PCov_Cons + PCov_2-5m, family=poisson(link = log)

Candidate Models	**df**	**logLik**	**AICc**	**ΔAICc**	w_i	**acc** w_i	**Pseudo- R^2**
P_HedgeLen + PCov_Cons	3	-47.49	102.31	0	0.41	0.41	0.53
P_HedgeLen	2	-49.65	103.93	1.63	0.18	0.59	0.36
P_HedgeLen + PCov_Elm	3	-48.69	104.71	2.41	0.12	0.71	0.44
P_HedgeLen + PCov_Maple	3	-48.88	105.09	2.78	0.10	0.81	0.42
PCov_2-5m + PCov_Cons	3	-49.43	106.20	3.89	0.06	0.87	0.27
P_HedgeLen + PCov_2-5m	3	-49.65	106.63	4.32	0.05	0.92	0.36
PCov_Elm	2	-51.34	107.32	5.01	0.03	0.95	0.23
PCov_2-5m	2	-51.78	108.19	5.88	0.02	0.97	0.20
PCov_Cons + PCov_Elm	3	-50.44	108.20	5.90	0.02	0.99	0.30

f) Global Model = Spp_Priority^{w} ~ PCov_Maple + PCov_Elm + PCov_CropOther + PCov_ImpGrass + P_HedgeLen + PCov_2-5m, family= inverse.gaussian (link = log)

Candidate Model	**df**	**logLik**	**AICc**	**ΔAICc**	w_i	**acc** w_i	**adj** R^2
P_HedgeLen + PCov_ImpGrass	4	-35.35	81.06	0	0.81	0.81	0.51
P_HedgeLen + PCov_CropOther	4	-37.11	84.58	3.52	0.14	0.95	0.42
PCov_ImpGrass + PCov_Maple	4	-38.17	86.70	5.64	0.05	1	0.36

g) Global Model = Spp_Rarity ~ rescale(PCov_Maple) + rescale(PCov_Oak) + rescale(PCov_2-5m) + rescale(P_HedgeLen) + rescale(PCov_Cons) + rescale(PCov_Elm), link=”sqrt”

Candidate Models	**df**	**logLik**	**AICc**	**ΔAICc**	w_i	**acc** w_i	**Pseudo-**R^2
P_HedgeLen	3	-69.19	145.71	0	0.31	0.31	0.19
P_HedgeLen + PCov_Cons	4	-68.28	146.91	1.20	0.17	0.48	0.25
P_HedgeLen + PCov_Maple	4	-68.65	147.65	1.94	0.12	0.60	0.23
PCov_Maple	3	-70.52	148.38	2.68	0.08	0.68	0.10
(Null)	2	-71.89	148.42	2.71	0.08	0.76	-
P_HedgeLen + PCov_2-5m	4	-69.11	148.58	2.87	0.07	0.83	0.20
PCov_2-5m	3	-70.75	148.84	3.13	0.06	0.89	0.09
PCov_Cons	3	-71.32	149.97	4.26	0.04	0.93	0.04
PCov_Cons + PCov_2-5m	4	-69.87	150.08	4.38	0.03	0.96	0.15
PCov_Maple + PCov_2-5m	4	-70.10	150.56	4.85	0.03	0.99	0.13
PCov_Cons + PCov_Maple	4	-70.50	151.36	5.65	0.02	1.01	0.10

h) Global Model = Spp_IRR* ~ rescale(Pres_Thorns) + rescale(Ostorey_PenDepth) + rescale(PCov_>5m) + rescale(PCov_Cons), link="loglog"

Candidate Models	**df**	**logLik**	**AICc**	**ΔAICc**	**w_i**	**acc w_i**	**Pseudo-R^2**
(Null)	2	55.80	-106.97	0	0.25	0.25	-
Ostorey_PenDepth	3	56.69	-106.05	0.91	0.16	0.41	0.10
PCov_Cons	3	56.67	-106.02	0.95	0.15	0.56	0.06
PCov_>5m	3	56.29	-105.24	1.73	0.10	0.66	0.06
Ostorey_PenDepth + PCov_Cons	4	57.76	-105.17	1.79	0.10	0.76	0.17
Pres_Thorns	3	55.84	-104.34	2.62	0.07	0.83	0.01
PCov_>5m + PCov_Cons	4	57.28	-104.21	2.76	0.06	0.89	0.14
Ostorey_PenDepth + Pres_Thorns	4	57.14	-103.93	3.03	0.05	0.94	0.12
PCov_Cons + Pres_Thorns	4	56.70	-103.04	3.92	0.03	0.97	0.06
PCov_>5m + Pres_Thorns	4	56.40	-102.44	4.52	0.03	1	0.09

Table S 5.21. Model selection tables for wood transects in Cambridgeshire – top six AICc ranked models in each candidate set from global models containing selected independent variables. ΔAICc = difference in AICc between model and top model. w_i = Aikaike model weight. acc w_i = cumulative model weight. For variable derivation see Table 3.2, Table 3.3, Table 3.4, Table 3.5 and Table 3.6.

a) Global Model = Bird_Density ~ PCov_BroadOther + Pres_Bramble + PCov_Cons + Ostorey_PenDepth + PCov_Oak + Ht_VDR + Wood.Area + Dist_Wood, family=Gamma(link = identity)

Candidate Models	**df**	**logLik**	**AICc**	**ΔAICc**	**w_i**	**acc w_i**	**adjR^2**
Ostorey_PenDepth + Wood.Area	4	-54.78	121.19	0	0.19	0.19	0.42
PCov_BroadOther + PCov_Cons	4	-55.52	122.69	1.49	0.09	0.28	0.37
PCov_BroadOther	3	-57.41	122.82	1.63	0.09	0.37	0.20
(Null)	2	-59.16	123.24	2.05	0.07	0.44	-
Ht_VDR	3	-57.94	123.87	2.68	0.05	0.49	0.14
PCov_Cons	3	-58.16	124.32	3.13	0.04	0.53	0.12
PCov_BroadOther + Dist_Wood	4	-56.46	124.56	3.37	0.04	0.57	0.29

Pres_Bramble	3	-58.34	124.68	3.49	0.03	0.60	0.10
Ostorey_PenDepth	3	-58.47	124.94	3.75	0.03	0.63	0.08
Wood.Area	3	-58.52	125.04	3.85	0.03	0.66	0.08
Pres_Bramble + Ht_VDR	4	-56.71	125.05	3.85	0.03	0.69	0.26
Ostorey_PenDepth + PCov_BroadOther	4	-56.77	125.17	3.98	0.03	0.72	0.26
PCov_Oak	3	-58.60	125.19	4.00	0.03	0.76	0.07
Dist_Wood	3	-58.66	125.32	4.13	0.02	0.78	0.06
PCov_BroadOther + Pres_Bramble	4	-56.87	125.38	4.18	0.02	0.80	0.25
Pres_Bramble + PCov_Cons	4	-56.89	125.42	4.22	0.02	0.82	0.25
PCov_BroadOther + Wood.Area	4	-56.92	125.48	4.28	0.02	0.84	0.24
PCov_Cons + Wood.Area	4	-56.98	125.59	4.40	0.02	0.86	0.24
Dist_Wood + Ht_VDR	4	-56.99	125.61	4.42	0.02	0.88	0.24
Ht_VDR + PCov_BroadOther	4	-57.25	126.13	4.94	0.02	0.90	0.21
Wood.Area + Ht_VDR	4	-57.26	126.16	4.97	0.02	0.92	0.21
Dist_Wood + PCov_Cons	4	-57.27	126.18	4.99	0.02	0.94	0.21
PCov_BroadOther + PCov_Oak	4	-57.40	126.44	5.25	0.01	0.95	0.20
PCov_Cons + Ht_VDR	4	-57.45	126.55	5.35	0.01	0.96	0.19
Ostorey_PenDepth + Ht_VDR	4	-57.57	126.77	5.58	0.01	0.97	0.18
PCov_Oak + Ht_VDR	4	-57.75	127.13	5.94	0.01	0.98	0.16
Dist_Wood + Wood.Area	4	-57.77	127.18	5.99	0.01	0.99	0.16

b) Global Model = Spp_Richness ~ PCov_Cons + Ht_StDev + Ostorey_PenDepth + Wood.Area, family=poisson(link = identity)

Candidate Models	**df**	**logLik**	**AICc**	**ΔAICc**	w_i	**acc** w_i	**Pseudo-R^2**
(Null)	1	-41.55	85.39	0	0.32	0.32	-
Wood.Area	2	-40.88	86.67	1.28	0.17	0.49	0.34
Ht_StDev	2	-41.31	87.54	2.14	0.11	0.60	0.1
PCov_Cons	2	-41.39	87.69	2.30	0.10	0.70	0.09
Ostorey_PenDepth	2	-41.41	87.74	2.34	0.10	0.80	0.08
Wood.Area + Ht_StDev	3	-40.73	89.46	4.07	0.04	0.84	0.42
Wood.Area + PCov_Cons	3	-40.82	89.64	4.25	0.04	0.88	0.37
Wood.Area + Ostorey_PenDepth	3	-40.88	89.75	4.36	0.04	0.92	0.34
Ht_StDev + Ostorey_PenDepth	3	-41.21	90.43	5.04	0.03	0.95	0.17
Ht_StDev + PCov_Cons	3	-41.25	90.49	5.10	0.03	0.98	0.16
PCov_Cons + Ostorey_PenDepth	3	-41.34	90.67	5.28	0.02	1	0.11

c) Global Model = Spp_Diversity~PCov_Oak + PCov_Ash + Wood.Area + Ht_StDev + Ht_VDR + Ostorey_PenDepth + Wood_P:A, family=inverse.gaussian(link=identity)

Candidate Models	**df**	**logLik**	**AICc**	**ΔAICc**	w_i	**acc** w_i	**adjR^2**
Wood.Area	3	21.27	-34.54	0	0.38	0.38	0.39
Wood.Area + Ht_StDev	4	22.06	-32.48	2.06	0.13	0.51	0.40
Wood.Area + PCov_Oak	4	21.81	-31.98	2.56	0.1	0.61	0.39
Wood.Area + Ostorey_PenDepth	4	21.5	-31.37	3.17	0.08	0.69	0.35
Ht_StDev + Wood_P.A	4	21.45	-31.27	3.27	0.07	0.76	0.35
Wood.Area +Ht_VDR	4	21.35	-31.06	3.48	0.07	0.83	0.34
Wood_P.A	3	19.49	-30.98	3.56	0.06	0.89	0.24
Wood.Area +PCov_Ash	4	21.27	-30.91	3.63	0.06	0.95	0.34
Wood_P.A + PCov_Oak	4	20.97	-30.31	4.23	0.05	1	0.33

d) Global Model = Spp_Decline ~ PCov_Oak + Ht_VDR + %_Woods_500m, family=poisson(link=log)

Candidate Models	**df**	**logLik**	**AICc**	**ΔAICc**	w_i	**acc** w_i	**Pseudo-** R^2
(Null)	1	-31.59	65.47	0	0.48	0.48	-
Ht_VDR	2	-31.48	67.88	2.41	0.14	0.62	0.11
%_Woods_500m	2	-31.49	67.90	2.43	0.14	0.76	0.10
PCov_Oak	2	-31.50	67.93	2.46	0.14	0.90	0.09
Ht_VDR + %_Woods_500m	3	-31.40	70.79	5.32	0.03	0.93	0.20
Ht_VDR + PCov_Oak	3	-31.44	70.87	5.40	0.03	0.96	0.16
%_Woods_500m + PCov_Oak	3	-31.45	70.89	5.42	0.03	0.99	0.15

e) Global Model = Spp_Priority ~ PCov_Elm + PCov_2-5m + %_Woods_500m, family=poisson(link=log)

Candidate Models	**df**	**logLik**	**AICc**	**ΔAICc**	w_i	**acc** w_i	**Pseudo-** R^2
(Null)	1	-35.14	72.57	0	0.44	0.44	-
%_Woods_500m	2	-34.88	74.68	2.11	0.15	0.59	0.09
PCov_Elm	2	-34.90	74.72	2.15	0.15	0.74	0.08
PCov_2-5m	2	-34.93	74.78	2.21	0.15	0.89	0.07
%_Woods_500m + PCov_2-5m	3	-34.69	77.37	4.80	0.04	0.93	0.16
PCov_2-5m + PCov_Elm	3	-34.74	77.48	4.90	0.04	0.97	0.14
%_Woods_500m + PCov_Elm	3	-34.76	77.53	4.96	0.04	1.01	0.13

f) Global Model = Spp_Priority^w ~PCov_Oak + PCov_Cons + PCov_BroadOther + Pres_Bramble + Ht_VDR + Ostorey_PenDepth + %_Woods_1km, family=Gamma(link = identity)

Candidate Models	**df**	**logLik**	**AICc**	**ΔAICc**	w_i	**acc** w_i	**adj**R^2
PCov_Cons + Pres_Bramble	4	-42.88	97.40	0	0.38	0.38	0.57
PCov_Cons	3	-45.18	98.36	0.96	0.23	0.61	0.43
PCov_Cons + PCov_BroadOther	4	-43.61	98.85	1.45	0.18	0.79	0.53
PCov_Cons + Ht_VDR	4	-44.64	100.92	3.52	0.07	0.86	0.46
PCov_Cons + PCov_Oak	4	-44.85	101.34	3.94	0.05	0.91	0.45
PCov_Cons + Ostorey_PenDepth	4	-45.00	101.64	4.23	0.05	0.96	0.44
PCov_Cons + %_Woods_1km	4	-45.13	101.90	4.50	0.04	1	0.43

g) Global Model = Spp_Rarity ~ PCov_Oak + Ostorey_Ht + PCov_2-5m + Wood.Area, family=poisson(link=identity)

Candidate Models	**df**	**logLik**	**AICc**	**ΔAICc**	***w_i***	**acc *w_i***	**adjR^2**
PCov_Oak	2	-46.56	98.04	0	0.23	0.23	0.18
Ostorey_Ht	2	-46.76	98.44	0.40	0.19	0.42	0.16
(Null)	1	-48.16	98.60	0.56	0.17	0.59	-
Ostorey_Ht +Wood.Area	3	-45.73	99.46	1.42	0.11	0.70	0.26
Wood.Area	2	-47.27	99.46	1.42	0.11	0.81	0.11
Ostorey_Ht + PCov_Oak	3	-45.93	99.87	1.83	0.09	0.90	0.24
Wood.Area + PCov_Oak	3	-46.04	100.08	2.04	0.08	0.98	0.23

h) Global Model = Spp_IRR ~ rescale(PCov_Cons)+ rescale(Pres_Bramble)+ rescale(Ht_Av)+ rescale(Ht_VDR)+ rescale(PCov_<0.5m), link="loglog")

Candidate Models	**df**	**logLik**	**AICc**	**ΔAICc**	***w_i***	**acc *w_i***	**Pseudo-R^2**
Ht_VDR + Pres_Bramble	4	19.84	-28.04	0	0.49	0.49	0.76
Ht_Av	3	17.29	-26.58	1.45	0.24	0.73	0.66
Ht_VDR	3	16.75	-25.50	2.54	0.14	0.87	0.63
Ht_Av + Pres_Bramble	4	17.63	-23.62	4.41	0.05	0.92	0.69
Ht_Av + PCov_Cons	4	17.45	-23.25	4.78	0.04	0.96	0.67
Ht_Av + PCov_<0.5m	4	17.33	-23.02	5.01	0.04	1	0.66

6 Bird Community Analysis – New Forest

6.1 Abstract

Bird community composition has often been analysed using diversity measures or by selecting particular species. Multi-Dimensional Scaling (MDS) is a useful tool to graphically display the similarities and dissimilarities in communities in 2D space. MDS was used to investigate bird community composition in each of the habitat survey plots and, by analysing the relationships with vegetation composition and structure, understand the drivers of bird community composition and the species contributing to the bird indices in Section 4. As expected, the woodland and non-woodland habitats were positioned separately on the MDS, with the greatest dissimilarities between and within the non-woodland habitats (‘Heathland’ and ‘Scrubland’). These differences were attributed to three groups/guilds of bird species, namely wetland, scrubland and open specialists. The woodland habitats were more similar in bird community composition, however, once separated they grouped largely into their assigned habitat classes, with the bird communities in both the ‘Other Conifer’ and the ‘Pine’ plots separate, and therefore dissimilar, from one another and from the broadleaved habitats. The coniferous bird species, such as the Firecrest, were more associated with the ‘Other Conifer’ habitats, rather than the ‘Pine’, supporting the high IRR in this habitat (Section 4). However, the Common Crossbill was strongly associated with the ‘Pine’ and the rare Wood Warbler with the mixed ‘Pine/Broadleaved’ plots, suggesting that elimination of pine should be avoided. Ultimately, the result showed that similar habitats can differ in bird community composition as a result of vegetation composition, structure and other factors, such as wetness, presence of dead wood and surrounding habitat.

6.2 Introduction

Bird community analyses in the past have almost always used a measure of bird species richness or diversity (e.g. Willson 1974, Laiolo 2002), or individual species or guild analysis (e.g. Rodewald and Yahner 2001) to investigate bird-habitat relationships at various landscape scales (e.g. Seoane et al. 2017), and demonstrate that birds respond most to structural complexity of the vegetation and the distribution of important habitat (Heikkinen et al. 2004). However, this masks any specialist bird species and communities within habitats that are often sensitive to landscape change (Miller and Cale 2000, Radford et al. 2005). Bird community analyses have taken different forms, for example, Willson and Comet (1996) used a measure of proportional similarity (following Holmes and Pitelka (1968)) which combined species occurrence and proportional abundance to investigate bird communities in multiple habitats over multiple years. Principal Components Analysis (PCA) and Redundancy Analysis have also been used (Neumann et al. 2016). Laiolo (2005) used PCA to summarise environmental variables and relate these, using generalised linear models, to bird diversity and abundance. However, Minchin (1987) showed that Multidimensional Scaling (MDS) was the most robust and effective method. More recently studies, such as Lindbladh et al. (2019), have used MDS to compare the bird communities in various vegetation types, before drastic changes are made. However, these studies investigate bird community in similar broad habitat classifications (i.e. woodland or farmland), rather than over complex, multi-habitat landscapes. Although bird communities are well regarded to differ between broad habitat classifications, this has not explicitly been quantified.

The current study uses MDS to compare and contrast the bird community composition between and within the multiple habitats over the New Forest landscape. The species associated with the survey plots driving bird community composition, and contributing to the bird indices in Section 4 were assessed. Measures of vegetation composition and structure were related to the ordination to investigate influences on the differences in bird community composition.

6.3 Methods

The bird survey data collected (following 3.3.4.1, pp. 68 and 3.3.5.1, pp. 69 – 70) were used to create MDS plots which displayed the similarities in bird species composition between the habitats in the New Forest. Following Oksanen (2015) the *metaMDS* function used the Bray-Curtis dissimilarity and plotted in 2D. This was carried out for all 32 plots at the landscape level with a stress value of 0.079 (stress less than 0.2 is considered good; Kruskal 1964, Wickelmaier 2008), and for the 24 woodland plots (stress = 0.167), with MDS plots of the bird species responsible for the ordination included (3.3.8, pp. 78 – 79).

To identify which particular structural or compositional aspects of each habitat type were most strongly driving the bird species assemblages associated with each plot, the MDS plots were reproduced with the vegetation composition variables (4.4.1, pp. 82 – 83) and the LiDAR-derived vegetation structural metrics (4.4.2, pp. 83 – 86) presented, using the *envfit* function in R version 3.5.1/3.5.2 (3.3.8, pp. 78).

It is worth noting that in the MDS plots the axis values do not represent any relative measure or value, the MDS simply displays the habitats by the similarities and dissimilarities of bird communities; therefore it is the distance, rather than the axis values, that should be interpreted (Borgatti 1997).

6.4 Results

6.4.1 Multidimensional Scaling (Landscape Scale)

Multidimensional Scaling (MDS) in Figure 6.1a graphically displays the survey plots in the New Forest based on the similarities and dissimilarities in bird community composition. The ordination was based solely on bird density and composition in each plot and therefore the apparent clustering of plots into habitat types indicates bird-habitat selection preferences (Figure 6.1a). A second MDS plot in Figure 6.1b displays the bird species of the New Forest arranged in ordination space related to the survey plots, and facilitates the interpretation of the bird species responsible for the bird-habitat trends.

The ‘Heathland’ and ‘Scrubland’ plots in Figure 6.1a, were positioned further apart from the woodland plots in the ordination, indicating greater dissimilarity in bird community composition, to one another as well as to the woodland plots. The woodland plots were positioned closer together and therefore were more likely to be similar in bird community composition. The ‘Heathland’ and ‘Scrubland’ plots stretch out to the left of the graph demonstrating a woodland–non-woodland gradient along the horizontal axis, with ‘Heathland’ plots 1, 4 and 3 becoming progressively less wooded and ‘Scrubland’ plots 6, 8 and 7 becoming progressively more wet and less wooded (Figure 6.1a). The woodland plots display a conifer–broadleaved gradient along the vertical axis, with the mixed ‘Pine/Broadleaved’ plots in the centre indicating intermediate bird community of both coniferous and broadleaved woodland (Figure 6.1a). The dashed black lines in Figure 6.1a represent the gradient boundaries running through zero on the graph separating conifer from broadleaved plots (horizontal line) and woodland from non-woodland plots (vertical line).

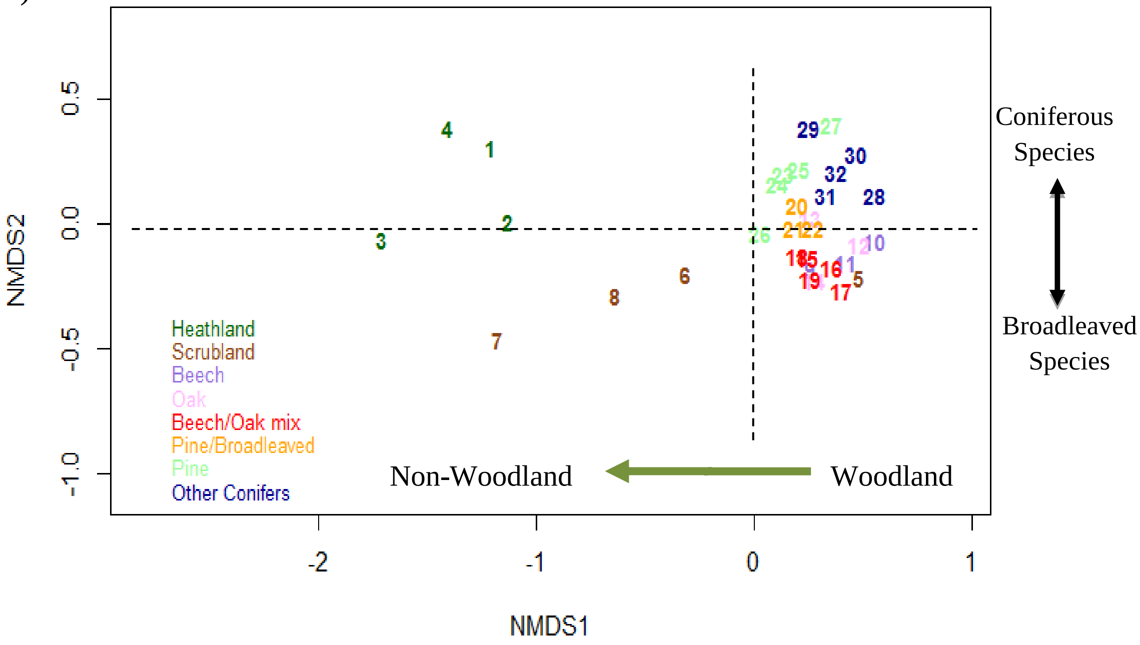
The ‘Heathland’ and ‘Scrubland’ plots also followed the vertical broadleaved–conifer gradient; ‘Heathland’ plot 1 contained more coniferous species (e.g. pine) and ‘Scrubland’ plot 7 contained more deciduous species (e.g. silver birch). The position of ‘Scrubland’ plot 7 suggests that this plot had the least amount of tree cover of the ‘Scrubland’ plots, but a similar amount of tree cover as ‘Heathland’ plots 1 and 2 (Figure 6.1a). ‘Heathland’ plots 2 and 3 sit on or near the gradient boundary between conifer and broadleaved, possibly suggesting that these plots may have contained an equal (and low) cover of broadleaved and conifer trees, influencing the birds present, but these plots were most influenced by the presence of bird species preferring open areas (Figure 6.1a).

‘Oak’ plot 13, ‘Pine’ plot 26 and ‘Scrubland’ plot 5 were unusually positioned in the woodland cluster of the MDS in Figure 6.1a, apparently on the ‘wrong side’ of the gradient boundary lines (i.e. their bird species assemblage and habitat classification do not entirely match). ‘Oak’ plot 13 was situated just above the horizontal boundary in the conifer section amidst the mixed ‘Pine/Broadleaved’ plots. ‘Pine’ plot 26 was situated just below the horizontal boundary line in the broadleaved section, suggesting the presence of some broadleaved or mixed woodland preferring bird species (Figure 6.1a). ‘Scrubland’ plot 5 was positioned with the broadleaved plots, close to ‘Beech/Oak mix’ plot 17 and ‘Beech’ plot 11, indicating more broadleaved habitat preferring bird species.

Overall, Figure 6.1a thus shows that there were distinct bird assemblages associated with ‘Heathland’, ‘Scrubland’ and the woodland habitats (thus reiterating the concept of guilds). This also indicates considerable variation across the bird communities of all three habitat types, presumably dependant on numerous factors of vegetation structure and composition.

Figure 6.1b displays the bird species responsible for the ordination of the survey sites in 2D space. Generally, the birds were positioned where they might be expected given the separation into woodland and non-woodland habitat (vertical dashed line) and conifer and broadleaved woodland (horizontal dashed line). However, there are a few exceptions. For example, the Redstart and Mistle Thrush were located in the ‘Scrubland’ section, possibly as a result of higher detectability in the ‘Scrubland’ near to woodland edges and, in the case of the Mistle Thrush, foraging on open ground. The Long-Tailed Tit and Wren were positioned in the MDS plot unusually in the conifer section, and were possibly either passing through or utilising the lower vegetation layers for nesting. The open habitat birds in the ‘Heathland’ and ‘Scrubland’ plots, were separated into three distinct clusters: typical heathland species, such as the Dartford Warbler and Curlew (*Numenius arquata*), in the top left furthest corner away from the woodland; wetland species, such as the Grey Heron and Little Egret (*Egretta garzetta*) in the bottom left corner, again furthest away from the woodland; and scrubby species, such as the Bullfinch and Lesser Whitethroat (*Sylvia curruca*) near the woodland boundary line on the MDS plot (Figure 6.1b).

a)



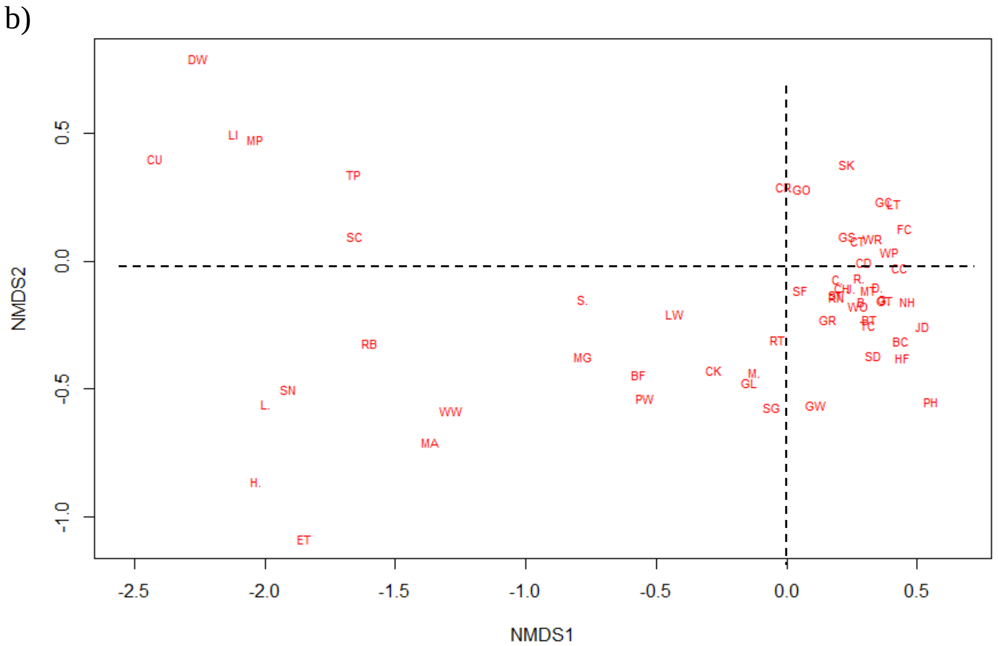


Figure 6.1. Multidimensional Scaling (MDS) plots of a) the 32 survey plots in the New Forest annotated to highlight gradients and relationships, and b) the bird species in 2D space responsible for the ordination of the plots. See Appendix A1 for bird species codes (R version 3.5.1).

6.4.2 Multidimensional Scaling with Environmental Factors

6.4.2.1 Landscape Scale with Vegetation Composition

Figure 6.2 displays the significantly correlated vegetation composition data from Table 6.1 to the ordination (at $P < 0.05$). The arrows point in the direction of the most rapid change (gradient) of the variable and the length of the arrows are proportional to the correlation between the ordination and environmental variables (R^2), and therefore the strength of the gradient (Oksanen 2015). The significant environmental variables in Figure 6.2 confirm the habitat gradients described in Section 6.4.1. Table 6.1 showed that *PCov_Heather* was the most important factor related to the ordination of the plots ($R^2 = 0.70$; $P = 0.001$), indicating that the 'Heathland' plots were positioned first and there was a gradient of increasing *PCov_Heather* (Figure 6.2). *PCov_Gorse* was less related to the positioning of the 'Heathland' plots ($R^2 = 0.21$; $P = 0.041$), although still significant, possibly indicating an increasing amount of gorse in 'Heathland' plots 1 and 4. Figure 6.2 showed that *Pres_Willow* was the distinguishing factor separating 'Scrubland' plots 6, 7 and 8, from the other plots ($R^2 = 0.28$; $P = 0.008$). *PCov_Beech* and *PCov_Oak* were positioned virtually on top of one another pointing down and in the direction of the broadleaved plots (Figure 6.2), with similar significance and relationship to the ordination ($R^2 = 0.28$ and 0.31; $P = 0.013$ and 0.011, respectively), suggesting that these variables are interchangeable. *Pres_Holly* was also significantly related to the position of the broadleaved plots at a slight angle to *PCov_Beech* and *PCov_Oak,* ($R^2 = 0.26$; $P = 0.008$), suggesting that holly was present in some of the broadleaved woodlands (Figure 6.2). *PCov_Pine* and *PCov_ConsOther* were also almost in the same position pointing to the top of the graph in the direction of the conifer plots (Figure 6.2). *PCov_Pine* had a slightly stronger correlation to the ordination than *PCov_ConsOther* ($R^2 = 0.29$ and 0.25; $P = 0.010$ and 0.021, respectively), possibly as a result of fewer ‘Other Conifer’ plots (Table 6.1). The opposing direction of the conifer arrows to the broadleaved arrows in Figure 6.2 highlights the vertical conifer–broadleaved gradient, and the length of the arrows signifies the strength of the gradient. Subsequently, further investigation of the woodland plots alone may be needed to explain the ordination and bird communities in the woodland plots.

*Table 6.1. The envfit output for the vegetation composition variables in all the survey plots in the New Forest. The stars represent the level of significance: ‘***’ <0.001, ‘**’ <0.01, ‘*’ <0.05, ‘.’ <0.1. This was carried out with 999 permutations. (NMDS = Non-Metric Multidimensional Scaling).*

Variable	NMDS1	NMDS2	R^2	P	Significance
PCov_Pine	0.171	0.985	0.289	0.010	**
PCov_ConsOther	0.209	0.978	0.247	0.021	*
PCov_Beech	0.291	-0.957	0.279	0.013	*
PCov_Oak	0.305	-0.952	0.306	0.011	*
PCov_BroadOther	0.016	-1.000	0.076	0.300	
PCov_Heather	-0.984	0.178	0.703	0.001	***
PCov_Gorse	-0.825	0.565	0.206	0.041	*
Pres_Bracken	0.992	-0.126	0.117	0.157	
Pres_Holly	0.470	-0.883	0.259	0.010	**
Pres_Hawthorn	-0.385	0.923	0.007	0.910	
Pres_Birch	0.747	-0.665	0.141	0.080	.
Pres_Willow	-0.558	-0.830	0.281	0.008	**

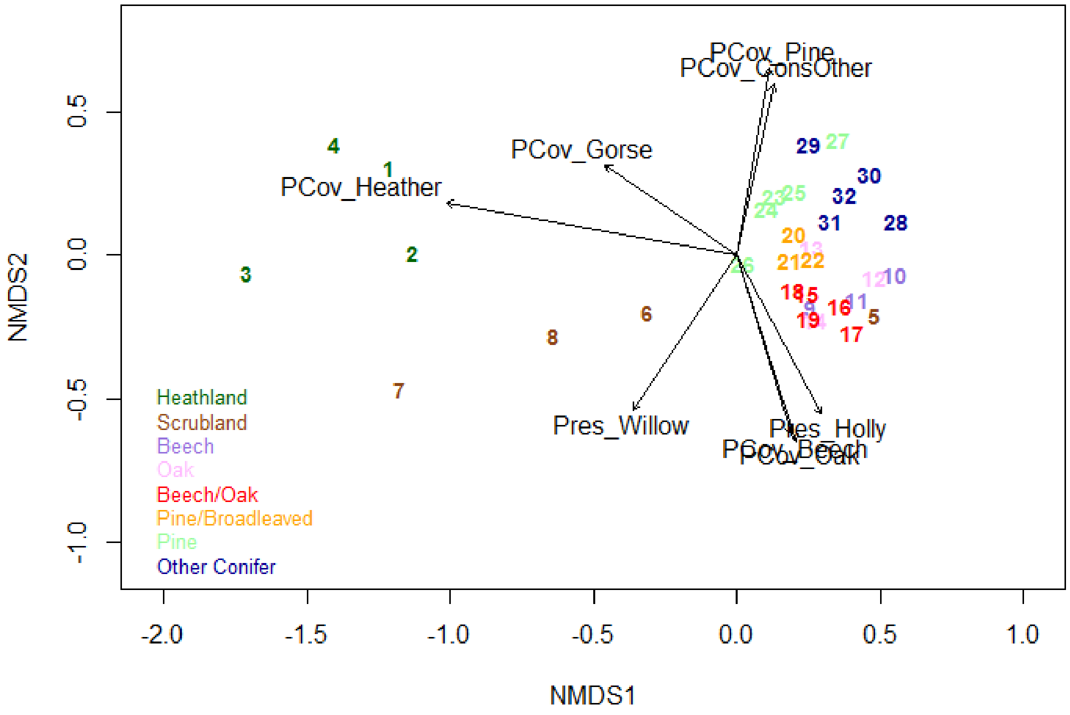


Figure 6.2. MDS of all the survey plots with the envfit function displaying the significant (at P <0.05) vegetation composition variables as environmental factors to explain the ordination. See Table 6.1 for P values (R version 3.5.2).

6.4.2.2 Landscape Scale with Vegetation Structure

Figure 6.3 displays the significant vegetation structural metrics correlated with the ordination of the survey plots (at P <0.05). The most important structural metric was *PCov_<0.5m* ($R^2 = 0.89$; P = 0.001), further indicating that the ordination positioned the ‘Heathland’ and ‘Scrubland’ plots first with increasing *PCov_<0.5m* (Table 6.2 and Figure 6.3). *Ht_VDR* was also correlated with the position of the ‘Heathland’ and ‘Scrubland’ plots, indicating that these habitats have a greater disparity between their mean and maximum height per plot ($R^2 = 0.67$; P = 0.001). *PCov_>5m*, *Ht_Av* and to a lesser degree *Ostorey_Ht* ($R^2 = 0.85$, 0.77 and 0.55, respectively; P = 0.001 for all) were significantly correlated with the woodland plots, depicting tall vegetation (Table 6.2 and Figure 6.3). ‘Other Conifer’ plot 28 was directly associated with both *Ostorey_Ht* and *PCov_>5m* suggesting that this plot had the tallest trees of the woodland plots (Figure 6.3). The majority of the ‘Pine’ and some of the ‘Other Conifer’ plots (29, 30 and 32) were separated from the other woodland plots by being significantly related with *Ostorey_PenDepth* ($R^2 = 0.72$; P = 0.001). This suggests that these conifer plots had a more open canopy, possibly as a result of silviculture or the absence of understorey vegetation.

Ht_StDev was significantly correlated with the ordination ($R^2 = 0.70$; $P = 0.001$), indicating that, in the most part, the broadleaved woodlands and the ‘Pine/Broadleaved’ plots differed from the conifer woodlands as having a varied structure (Table 6.2 and Figure 6.3).

PCov_2-5m was less strongly significantly correlated with the ordination ($R^2 = 0.23$; $P = 0.02$) and positioned to the left of and pointing towards the broadleaved gradient (Figure 6.3). Moreover, the factor was positioned between the ‘Scrubland’ and broadleaved plots suggesting an increasing gradient in *PCov_2*-5m, either as understorey in the broadleaved plots or as the main vegetation layer in the ‘Scrubland’ plots. ‘Pine’ plot 26 was directly on the *PCov_2-5m* line suggesting that this plot had a higher value than the other ‘Pine’ plots, and therefore possibly more understorey (Figure 6.3). The environmental factor lines in Figure 6.3 also converged above ‘Pine’ plot 26, suggesting either this plot was associated with all the factors, or none, causing the plot to be relatively dissimilar to the other woodland plots.

PCov_>5m was directly opposite *PCov_<0.5m*, displaying the opposing relationship and gradients separating the ‘Heathland’ and ‘Scrubland’ plots from the woodland plots (Figure 6.3). *Ht_VDR* was almost directly opposite *Ht_Av* (as taller mean height) and *Ostorey_PenDepth*, because high canopy closure and limited understorey tended to result in a lower height VDR (i.e. a higher median which is likely to be closer to the 95th height percentile). Further analysis should investigate the woodlands separately from the ‘Heathland’ and ‘Scrubland’ plots, in order to explain bird community differences between and within the woodland classes. Notably, *PCov_0.5-2m* was the only non-significant variable ($P = 0.34$), possibly suggesting that there was a similar percentage of vegetation at this height range in the majority of the survey plots.

*Table 6.2. The envfit output for the vegetation structural metrics in all the survey plots in the New Forest. The stars represent the level of significance: '***' <0.001, '**' <0.01, '*' <0.05, '.' <0.1. This was carried out with 999 permutations. (NMDS = Non-Metric Multidimensional Scaling).*

Variable	NMDS1	NMDS2	R^2	P	Significance
Ht_Av	0.907	0.422	0.771	0.001	***
Ht_StDev	0.992	-0.125	0.701	0.001	***
Ht_VDR	0.971	0.241	0.673	0.001	***
Ostorey_Ht	0.768	0.640	0.554	0.001	***
Ostorey_PenDepth	-0.869	-0.495	0.724	0.001	***
PCov_<0.5m	-0.991	-0.137	0.890	0.001	***
PCov_0.5-2m	0.508	0.861	0.073	0.338	
PCov_2-5m	0.406	-0.914	0.233	0.024	*
PCov_>5m	0.984	0.180	0.854	0.001	***

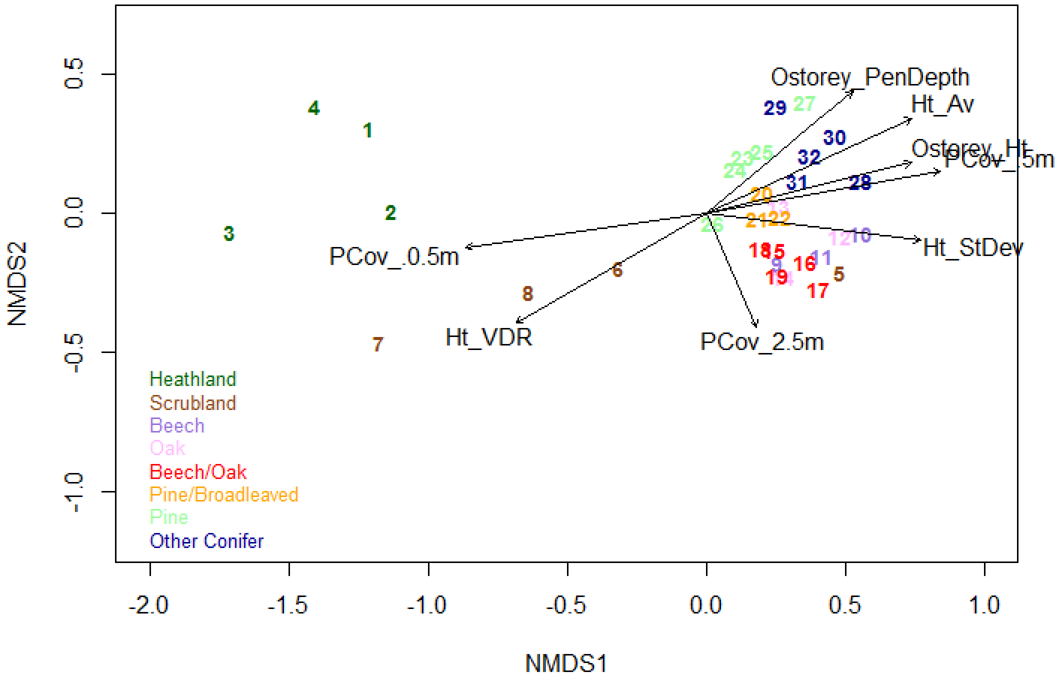


Figure 6.3. The envfit of the significant vegetation structure data (at P <0.05) displayed on the MDS for all the survey plots in the New Forest. See Table 6.2 for P values. PCov_.0.5m = PCov_<0.5m, PCov_2.5m= PCov_2-5m, PCov_.5m = PCov_>5m (R version 3.5.2).

6.4.3 Bird Community at the Local Scale: Woodland Habitats

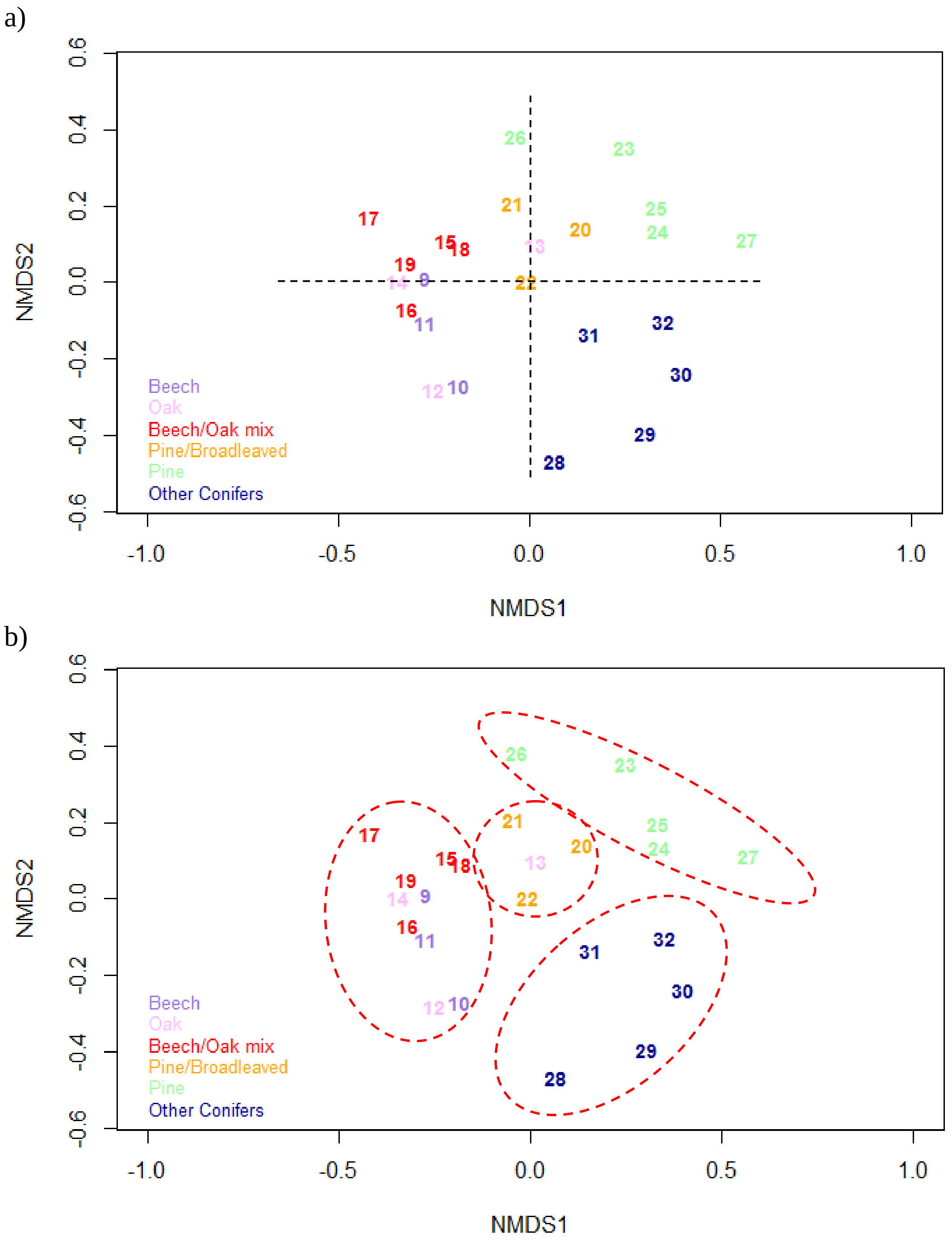
In order to further investigate the differences between woodland bird community composition in the New Forest, an MDS was plotted for the woodland habitat classes alone (Figure 6.4). As with Figure 6.1a, the dashed black lines in Figure 6.4a represent the gradient boundaries running through zero on the MDS graph. Here the vertical dashed line separates conifer from broadleaved plots, but with

‘Pine’ plot 26 and ‘Pine/Broadleaved’ plot 21 falling just to the left of the line, demonstrating the presence of some birds associated with broadleaved habitats in those two plots. In addition, ‘Oak’ plot 13 sits just to the right of the dashed line, suggesting the presence of bird species associated with pine or mixed habitat in this plot (Figure 6.4a). The horizontal dashed line clearly separates the ‘Pine’ and ‘Pine/Broadleaved’ plots from the ‘Other Conifer’ plots, and also divides the various ‘Beech’, ‘Oak’ and ‘Beech/Oak’ plots (Figure 6.4a). Further examination is required to identify whether there is some specific structural measure that can account for this separation. ‘Pine/Broadleaved’ plot 22 sits right at the intersection of the two dashed lines, suggesting that this plot was the most intermediary in terms of bird species (Figure 6.4a).

It could be argued that the different habitat types (with ‘Beech’, ‘Oak’ and ‘Beech/Oak’ combined) form distinct clusters on the MDS plot as annotated in Figure 6.4b. ‘Oak’ plot 13 was the exception which was located in the ‘Pine/Broadleaved’ cluster (Figure 6.4b). The ‘Pine/Broadleaved’ mixed plots were, as expected, in the centre with plot 22 in the middle of the ordination, indicating intermediate broadleaved/coniferous bird community (Figure 6.4b). The two coniferous woodland classes (‘Pine’ and ‘Other Conifers’) formed separate clusters, and were also relatively spread out, indicating dissimilarities between and within these habitat classes (Figure 6.4b). ‘Pine’ plot 26 and ‘Other Conifer’ plot 28 were particularly separated from their cluster centre. ‘Oak’, ‘Beech’ and ‘Beech/Oak’ mixed plots were mostly clustered together on the left of the graph (Figure 6.4b). ‘Beech’ plot 10 along with ‘Oak’ plot 12, and ‘Beech/Oak’ plot 17 were positioned further away from the main broadleaved cluster, suggesting that these plots contained slightly different bird assemblages.

The bird species responsible for the ordination of the woodland plots are displayed in Figure 6.4c, and for the most part, the bird species associated with broadleaved woodlands were on the left of the graph, and coniferous woodland preferring bird species were on the right. Typically broadleaved preferring bird species, such as the Great Spotted Woodpecker, Long-Tailed Tit, Wren and Woodpigeon were the exception on the conifer side, and Carrion Crow was also more associated with conifer woodland, possibly as a result of higher detectability in open conifer or for nesting sites (Figure 6.4c). The central position of the Wren, Woodpigeon and Robin possibly suggests that they were more abundant in the mixed

‘Pine/Broadleaved’ habitats or present in multiple habitat types. Chiffchaffs and Spotted Flycatchers are predominantly broadleaved preferring bird species, but were located on the boundary line, possibly indicating their presence in mixed woodlands or all habitat types (Figure 6.4c). The ‘Other Conifer’ plots were associated with different coniferous habitat preferring bird species than the ‘Pine’ plots, e.g. the Goldcrest, Firecrest, Siskin (*Carduelis spinus*) and Goldfinch, possibly creating a vertical gradient from the dissimilarities between these two habitat classes (Figure 6.4c). Unsurprisingly, the Crossbill, a pine specialist, was associated with the ‘Pine’ plots.



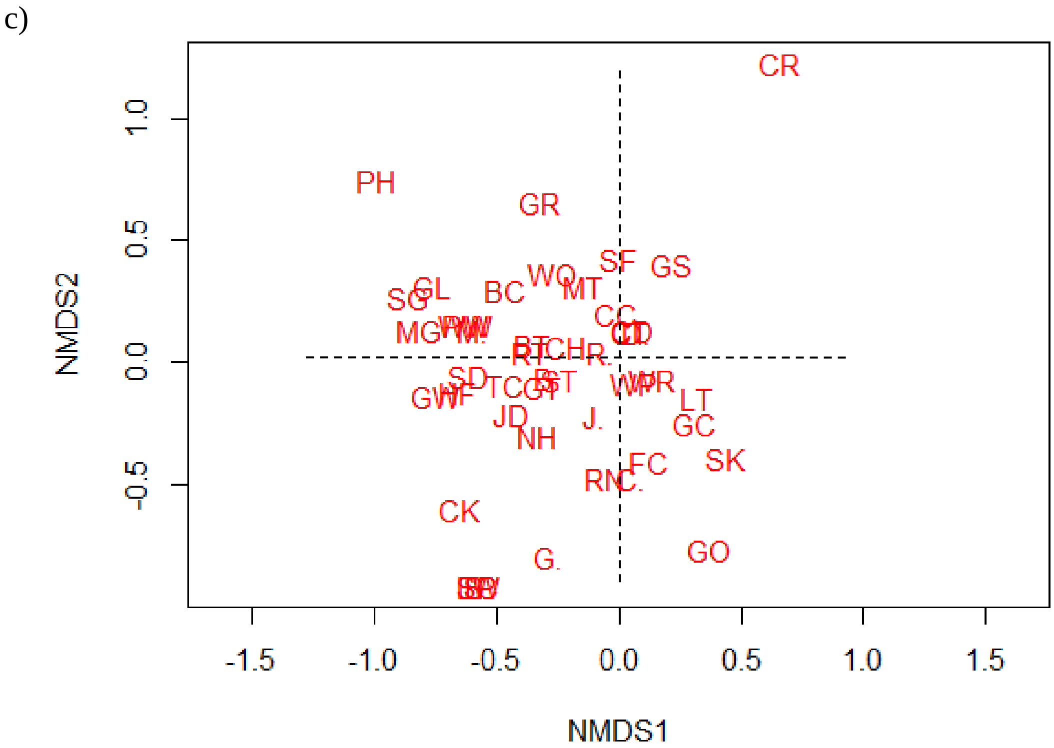


Figure 6.4. a) The MDS of the 24 woodland survey plots in the New Forest, b) annotated manually (not derived statistically) with dashed red ovals to indicate the plot groupings on the ordination, and c) the bird species responsible for the ordination. See Appendix A1 for bird species codes (R version 3.5.2).

6.4.4 Multidimensional Scaling at the Local Scale (Woodland) with Environmental Variables: Vegetation Composition and Structure

The significant (at $P < 0.05$) vegetation composition variables from Table 6.3 are displayed as environmental factors on the woodland ordination in Figure 6.5. The vegetation composition variables converge on ‘Pine/Broadleaved’ plot 22, suggesting that either this plot contains the full suite of bird species or was an intermediate habitat. As expected, the vegetation variables were clearly correlated with the respective habitat classifications. None of the variables were directly opposite from one another suggesting that they were not necessarily mutually exclusive (Figure 6.5). *PCov_Pine* was the most important factor with the highest correlation coefficient ($R^2 = 0.71$; $P = 0.001$) indicating that the ‘Pine’ plots were positioned first on the ordination. *Pres_Bracken* was significantly correlated with ‘Oak’ plot 13, ‘Pine/Broadleaved’ plot 21 and ‘Pine’ plot 26 ($R^2 = 0.35$; $P = 0.011$), resulting in different bird species assemblages (Figure 6.5). *Pres_Holly* was also significantly related with the broadleaved plots ($R^2 = 0.24$; $P = 0.047$), and in a similar position to *PCov_Oak* and *PCov_Beech* (Figure 6.5), again, indicating the presence of holly in oak and beech woodlands. The similar position of *PCov_Oak, PCov_Beech* and *Pres_Holly* also suggests that they were not mutually exclusive in the broadleaved plots and, therefore another factor must be causing the dissimilarities in the broadleaved bird communities. It is also worth noting that *PCov_BroadOther, Pres_Hawthorn, Pres_Birch and Pres_Willow,* were not significantly correlated with the ordination of the woodland plots (Table 6.3), suggesting that these species were present in the majority of plots, or were not a contributing factor.

Figure 6.6 showed that *Ostorey_PenDepth* was the most important structural variable which was significantly correlated with the ordination of the 'Pine' plots, including the 'Pine/Broadleaved' mixed plots ($R^2 = 0.44$; $P = 0.002$). This suggests that the bird community in these plots was influenced by an open canopy, as is typical of pine plantations. *PCov_2-5m* was almost directly opposite *Ostorey_PenDepth,* as the significant factor correlated with the ordination of the broadleaved habitats ($R^2 = 0.27$; $P = 0.031$), and was directly associated with 'Beech' plot 10 and 'Oak' plot 12 (Figure 6.6). This suggests that the broadleaved plots had a higher percent cover of vegetation at the 2-5 m understorey layer. This will prevent penetration of the laser pulses to the ground and therefore creates lower *Ostorey_PenDepth,* hence the opposing positions on the MDS. The non-significance of the other structural variables in Table 6.4 indicates that the woodland habitat classes were not distinct from one another in height; therefore these vegetation structural metrics did not affect bird community composition.

*Table 6.3. The envfit output for the significant vegetation species in the woodland plots in the New Forest (P <0.05), the stars represent the level of significance: '***' <0.001, '**' <0.01, '*' <0.05, '.' <0.1. This was carried out the 999 permutations. (PCov_ = % cover; NMDS = Non-Metric Multidimensional Scaling).*

Variable	NMDS1	NMDS2	R^2	P	Significance
PCov_Pine	0.716	0.698	0.710	0.001	***
PCov_ConsOther	0.508	-0.861	0.540	0.002	**
PCov_Beech	-0.999	-0.049	0.445	0.005	**
PCov_Oak	-1.000	-0.019	0.367	0.008	**
PCov_BroadOther	-0.733	0.680	0.056	0.562	
Pres_Bracken	-0.063	0.998	0.352	0.011	*
Pres_Holly	-0.993	-0.119	0.243	0.047	*
Pres_Hawthorn	0.231	0.973	0.049	0.577	
Pres_Birch	0	0	0	1	
Pres_Willow	-0.960	0.280	0.034	0.731	

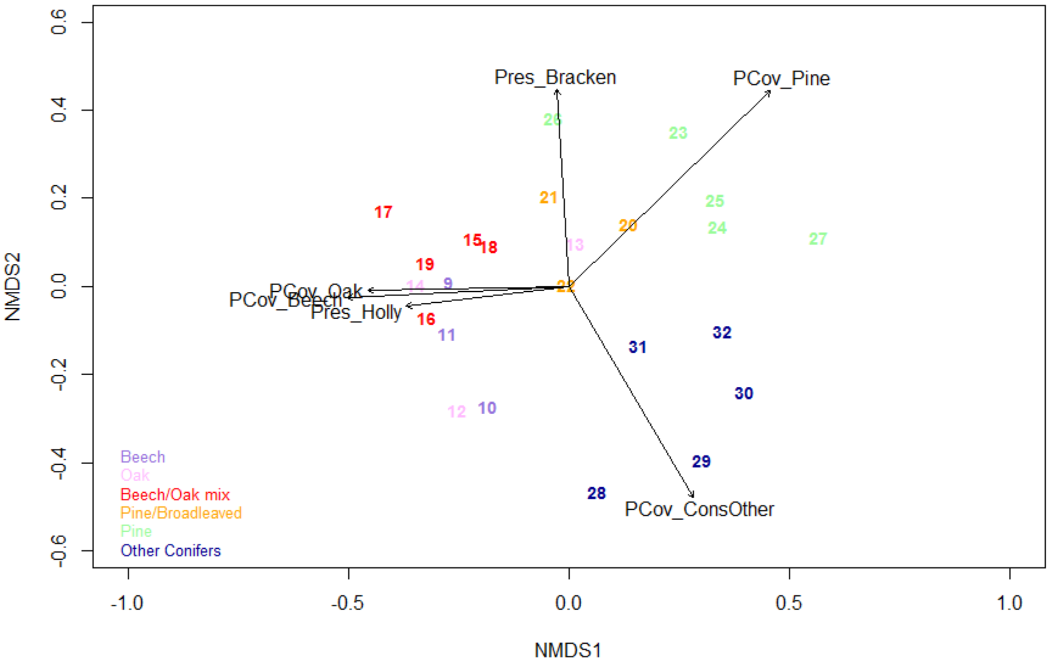


Figure 6.5. MDS of the woodland plots with the significant vegetation composition variables (at P <0.05) as factors in the envfit. See Table 6.3 for the P values (R version 3.5.1).

Table 6.4. The envfit output for the significant vegetation structural metrics in the woodland plots in the New Forest (P <0.05), the stars represent the level of significance: ‘***’ <0.001, ‘**’ <0.01, ‘*’ <0.05, ‘.’ <0.1. This was carried out the 999 permutations. (NMDS = Non-Metric Multidimensional Scaling).

Variable	NMDS1	NMDS2	R^2	P	Significance
Ht_Av	0.404	-0.915	0.014	0.851	
Ht_StDev	-0.515	-0.857	0.016	0.857	
Ostorey_Ht	0.494	-0.869	0.010	0.886	
Ostorey_PenDepth	0.521	0.854	0.440	0.002	**
Ht_VDR	-0.957	0.289	0.010	0.919	
PCov_<0.5m	0.278	0.961	0.123	0.248	
PCov_0.5-2m	0.690	0.724	0.228	0.057	.
PCov_2-5m	-0.596	-0.803	0.266	0.031	*
PCov_>5m	-0.322	-0.947	0.119	0.268	

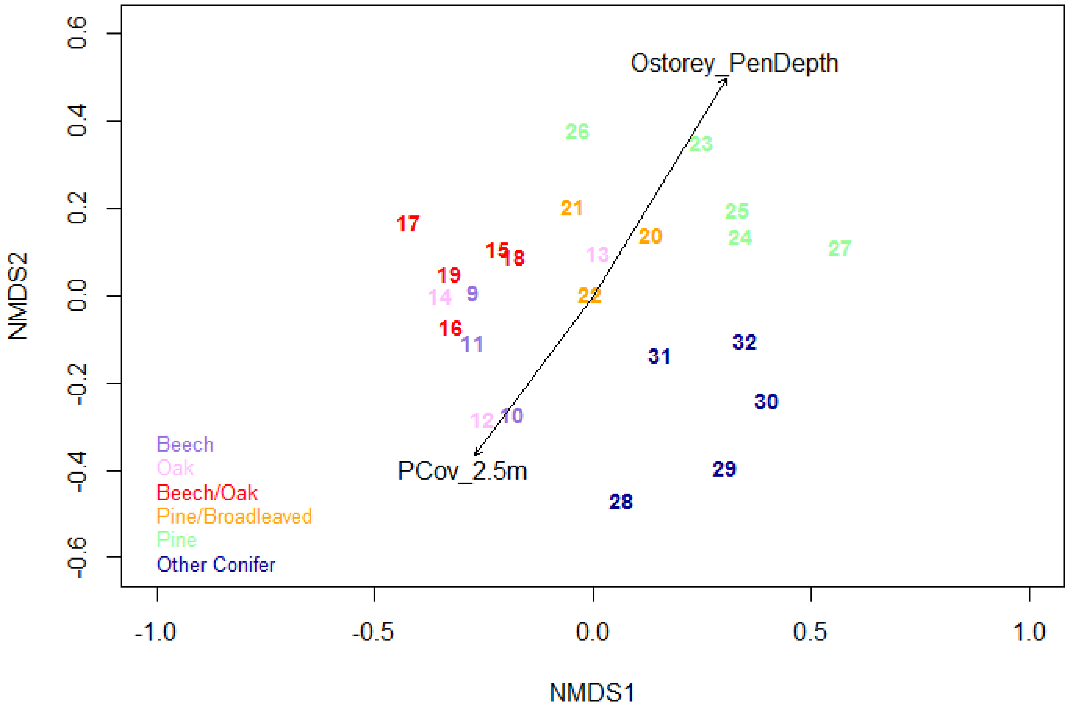


Figure 6.6. The MDS of the woodland plots with the significant vegetation structural metrics (at P <0.05) as envfit factors. See Table 6.4 for P values (PCov_2.5m = PCov_2-5m; R version 3.5.2).

6.5 Discussion

Multidimensional scaling of the habitats in the New Forest National Park showed that bird community composition differed between and within the habitat classes, demonstrating that similar habitats comprised different bird species assemblages.

The ordination was solely based on the bird species densities in each survey plot, which oriented the survey plots into the respective habitat classes creating a woodland–non-woodland horizontal gradient and a conifer–broadleaved vertical gradient, indicating an influence of bird habitat preference (Donald et al. 1998, Buchanan et al. 2016). The larger distances between the non-woodland habitats (‘Heathland’ and ‘Scrubland’) on the landscape MDS (in Figure 6.1a) indicated distinct dissimilarities in bird communities within these habitat types, and also with the woodland habitats, in line with Wilson (1974). The ordination of the bird species revealed that the non-woodland habitats were separated by three clusters of predominantly open area specialist species, wetland species and more scrub preferring species.

The woodland plots were closer together than the non-woodland plots on the landscape MDS, indicating some similarities, although, the separation of the broadleaved from the coniferous plots indicated dissimilarities in bird community composition. The distances of the plots on the MDS also varied within and between the woodland habitat classes, suggesting that some woodland survey plots were more similar in bird community composition than others. Moreover, the relevant vegetation composition and structural variables were shown to be correlated with the landscape ordination of the survey plots. As expected, they highlighted gradients of taller, woodland vegetation towards the woodland plots, and increasing percent open ground and ground vegetation towards the open habitats. An exception in the landscape analysis was ‘Scrubland’ plot 5 which was positioned with the broadleaved habitats, suggesting a similar bird community because it contained woodland preferring species such as Nuthatches and Blackcaps.

The woodland plots were analysed separately and clustered into the assigned habitat classes on the ordination (in Figure 6.4). The broadleaved habitats grouped together separate from both the ‘Pine’ and ‘Other Conifer’ plots, which were also in two separate clusters on the woodland MDS, indicating differences in bird community composition. Amongst all woodland plots, the main exception was ‘Oak’ plot 13 which was located with the ‘Pine/Broadleaved’ plots, suggesting the presence of conifer preferring bird species, although this plot contained very little conifer. The distances between the woodland plots on the ordination indicated subtle differences in bird community based on vegetation composition and structure to some extent. The variables showed associations of the habitat classes with the relevant vegetation species. Moreover, the presence of bracken, and also structural variables of canopy openness (overstorey penetration depth) and the percent cover of vegetation at 2-5 m highlighted the importance of woodland understorey vegetation to bird community composition.

6.5.1 Bird Community Non-Woodland (Landscape Scale)

The obvious separation of the ‘Heathland’ from the ‘Scrubland’ plots on the landscape MDS indicates dissimilarities in bird community composition of these two non-woodland habitats. The bird species occurring in the non-woodland plots arranged into three distinct clusters of predominantly heathland birds, birds associated with wetland (including waterfowl and waders), and scrubland birds, resulting in the subsequent positioning of the survey plots with gradients in habitat properties and ecological factors. The environmental factors analysed in the current study showed that the ordination was fitted to explain the ‘Heathland’ plots first, as the highest correlation coefficients (R^2) were for the percent cover of heather and the percent cover of vegetation < 0.5 m. It is also worth noting that the percent cover of vegetation < 0.5 m had the highest coefficient overall, closely followed by the percent cover of vegetation > 5 m, observably separating the habitat classes and creating the woodland–non-woodland gradient.

‘Heathland’ plot 3 was positioned furthest away from the woodland plots indicating the greatest dissimilarity in bird species composition, resulting from the gradient of increasing ground vegetation (from ‘Heathland’ plot 2 to 3), and therefore very little, if any, woody vegetation (supported by the strong relationship with the percent cover of vegetation < 0.5 m). As a result, ‘Heathland’ plot 3 was the only non-woodland plot to contain Skylark, along with the presence of the Curlew and Lapwing, suggesting that this plot was far enough away from substantial woody vegetation which would have caused an ‘edge effect’ deterring all three bird species (Donald et al. 2001b, Sheldon et al. 2004, Brown et al. 2014). The unusual positioning of the Skylark on the MDS was due to the species occurring in ‘Heathland’ plot 3, but also in ‘Oak’ plot 12, pulling it towards the woodland plots. It is unusual for Skylarks to occur in an area surrounded by woodland as they tend to avoid high boundaries (Donald et al. 2001b, Vickery and Arlettaz 2012), but this may suggest that the felled area beside ‘Oak’ plot 12 was large enough to accommodate this species. The Dartford Warbler was also present in ‘Heathland’ plot 3 and 4, possibly as a result of this species being an open habitat specialist (Moore 1962, Bibby and Tubbs 1975, Van den Berg et al. 2001). However, Bibby and Tubbs (1975) noted that in periods of population growth, the Dartford Warbler may have spread into other areas; for example, in dense and scattered pine in Spain and France, and nesting in trees in Suffolk (Venables 1937).

Wetland bird species, such as the Grey Heron, Snipe and Reed Bunting, also occurred in 'Heathland' plot 3, presumably due to the wet grassy area to the west of this plot, positioning it lower than the other 'Heathland' plots and just under the horizontal boundary line. However, this plot was still positioned close to the boundary line, possibly due to the presence of a number of coniferous bird species, such as the Goldcrest, Siskin and, to some extent, the Goldfinch, as a result of conifer trees nearby, demonstrating an effect of surrounding habitat on the bird community (Bergen et al. 2007, Neumann et al. 2016).

Notably, in the current study the differing positions of these 'Heathland' plots indicate observed differences in bird species composition in apparently similar habitats. 'Heathland' plot 4 contained the least wetland bird species (and the presence of the Curlew and Dartford Warbler) which shifted the plot upward on the MDS, the furthest distance from the wetter plots. The Curlew is regarded as a wetland species, however, in the current study they occurred in relatively dry, open heath, possibly as a result of a reduction in suitable breeding habitat through land drainage (Henderson et al. 2002). The Curlew was also relatively rare in the current study, and declined by 14% in England over the ten years to 2015 (Brown et al. 2015, Massimino et al. 2017). However, the trend from 1995-2013 showed that the population had decreased by 32% in England and 55% in Scotland to now only 66,000 breeding pairs in the UK (Brown et al. 2015). This decline in the Eurasian Curlew has been of concern to many ornithologists in recent years (e.g. Franks et al. 2017, Pearce-Higgins et al. 2017). The current trajectory of the Curlew population is likely to result in extreme rarity in the UK and, therefore it should be considered amongst the UK's highest conservation priority bird species with many organisations keen to aid the population's recovery (Brown et al. 2015). The decline is a consequence of a number of factors, but of most relevance here is low fledgling success caused by increased predation and trampling by grazing livestock (Franks et al. 2017), attributed to overgrazing in the New Forest. A reduction in breeding habitat through agricultural practises has also affected the Curlew, as has the increased afforestation and fragmentation of woodlands providing an ‘edge’ effect, often increasing predator numbers and vulnerability to predation, up to one kilometre from the woodland (Roodbergen et al. 2012, Brown et al. 2015, Franks et al. 2017). As a result of this, the preferred open habitat of the Curlew is diminishing and landscape management strategies should include wide areas of open, undisturbed grassland and heathland with exclusions of grazers in order for the species to recover.

‘Heathland’ plot 4 also had the highest percent cover of heather, which resulted in the strong correlation with this environmental factor, indicating the gradient in the percentage of heather from ‘Heathland’ plot 1 to plot 4, which possibly created sufficient habitat cover for the Curlew and the Dartford Warbler. Moreover, ‘Heathland’ plot 1 contained coniferous bird species (such as the Goldcrest, Goldfinch and Siskin), most likely as a result of the adjacent conifer plantation on the longest edge of this plot, positioning it further towards the woodland plots on the ordination. This also demonstrates that the coniferous gradient also exists between the ‘Heathland’ plots on the MDS, supported by the environmental factors of percent cover of pine and other conifer species not pointing directly into the woodland plots, thus not exclusively related to the woodland. ‘Heathland’ plot 1 contained the highest percentage of gorse, however, the presence of substantial pine cover could have deterred the Dartford Warbler, further affecting the position of the plot on the ordination (Bibby and Tubbs 1975, Van den Berg et al. 2001). ‘Heathland’ plot 2 did not contain heather or coniferous bird species, but had a number of wetland, open heath and scrubby birds. This possibly resulted in a more intermediate habitat and hence positioned the plot on top of the coniferous–broadleaved boundary line on the MDS, and also demonstrates a vertical wetland–heathland gradient. ‘Heathland’ plot 2 also had a high percent cover of broadleaved tree species, which most likely increased the presence of scrubby habitat preferring birds more representative of an intermediate heathland/scrubland habitat. Nonetheless, this plot was located in open heath in the New Forest, and therefore classified as ‘Heathland’.

The ‘Scrubland’ plots contained wetland and scrubby habitat preferring bird species, and also more generalist woodland bird species compared with the ‘Heathland’ plots. This was presumably due to the ecotonal habitat transitioning from the adjacent woodland in the early stages of succession, increasing the presence of scrubby bird species, such as Willow Warblers. The significant relationship of the height Vertical Distribution Ratio (VDR) with the ordination of the 'Scrubland' plots indicates a strong gradient from plots 6 to 8 and 7; however, they all had a maximum score of 1 (as for all three plots the median vegetation height was 0.01 m). Nevertheless, this supports a greater variation in vegetation layers of this successional habitat, with areas of open ground, shrubby vegetation and the occasional tree, creating a higher value of height VDR than the woodland areas. The gradient from 'Scrubland' plots 6 to 8 and 7 was demonstrated by the relationship with the percent cover of ground (increasing from 64.9% to 82.5% to 93.5%). The 'Scrubland' habitats also tended to be wetter with a gradient of increasing abundance of wetland bird species, as well as a decrease in substantial woody vegetation.

Unfortunately, the wetland areas were not quantified in the current study, but the presence of willow species was a significant factor in the direction of the 'Scrubland' plots, symbolising an increase in wetness along the 'Scrubland' gradient. 'Scrubland' plot 7, therefore, was particularly wet with a high number of wetland bird species, such as the Little Egret, Grey Heron, Lapwing and Snipe, further adding to and increasing bird diversity and species richness (Section 4.5.1). The position of the Lapwing, Grey Heron, Willow Warbler and Snipe supports the findings in Section 4.5.1 that these species were also contributing to the high priority index in the 'Scrubland' plots, however, the Stonechat was more associated with the Heathland habitats. Furthermore, the positioning of the Mallard, Reed Bunting and possibly the Stonechat, supports the contribution to the high rarity index in 'Scrubland' plots 6 and 7 (Section 4.5.1). The Cuckoo (*Cuculus canorus*) and Common Crossbill were also present in plot 6, whilst the Snipe and Little Egret were present in plot 7 further increasing the rarity index. 'Scrubland' plot 7 also had the least woody vegetation and was the only 'Scrubland' plot to contain the Meadow Pipit (*Anthus pratensis,* similar to the 'Heathland' plots), contributing to the greater distance (and therefore greater dissimilarity) from the woodland survey plots. 'Scrubland' plot 7 was also the furthest, in geographical terms, from the woodland, but contained substantial woody vegetation which classified it as 'Scrubland' rather than 'Heathland'. The Common Crossbill was located on the vertical boundary line of the MDS as a they were present in 'Scrubland' plot 6, as well as in the 'Pine' and 'Pine/Broadleaved' plots, in line with Fuller et al. (2005a) who noted that the

Common Crossbill frequented areas of scrub, neutral grass and some bracken land covers, possibly to forage; however, pine and conifer woodlands are their preferred habitat. This altered the positioning of plot 6 upwards and more towards the woodlands; however, scrubby bird species dominated this plot with more woodland species and fewer wetland bird species.

Buchanan et al. (2016) carried out a similar study to the current one, relating vegetation factors, including tree basal area, shrub density and tree species diversity to bird populations, and showed that specialist bird species declined over time with increasing basal area and oak dominance, and decreasing tree and shrub diversity. This is shown in the unusual positioning of ‘Scrubland’ plot 5 with the broadleaved plots, as a result of the larger oak trees in this plot increasing the average height of the trees, in this case, and altering bird community composition. ‘Scrubland’ plot 5 graduated from open scrub with bracken and silver birch to large, mature oak trees to the north of the plot in a relatively small area of only four hectares. Not surprisingly, ‘Scrubland’ plot 5 had fewer scrubby bird species, such as Stonechats, and contained more woodland species, such as the Nuthatch, Chiffchaff, Blackcap, Dunnock, Jackdaw, Woodpigeon and Firecrest, creating a woodland, rather than ‘Scrubland’, bird community. Buchanan et al. (2016) also found that specialist bird species occurred in scrubland and more open, successional habitats and that in the long-term, bird populations were affected strongly by basal area of trees (therefore stand age/maturity), supporting the importance of the scrubland habitats in the current study. Buchanan et al. (2016) also found that floristic diversity influenced bird population diversity and that bird species were affected by different vegetation variables, which were shown to increase bird diversity in the ‘Scrubland’ habitats of the New Forest (Section 4.5.1). Therefore, landscape-scale conservation efforts should be implemented to protect the various bird communities (Buchanan et al. 2016), by maintaining a heterogeneous landscape over the New Forest.

6.5.2 Bird Community in the Woodland Plots

The woodland plots were positioned much closer together on the landscape MDS in Figure 6.1, indicating similarities in bird species composition, however, a vertical conifer–broadleaved gradient exists (with mixed ‘Pine/Broadleaved’ in the middle), supported by the significant relationships of the respective vegetation composition variables. The woodland plots were also separated from the non-woodland plots by structural variables representing substantial vegetation, such as average height, indicating similar characteristics in the woodlands. Moreover, clustering within the conifer and broadleaved habitats suggests differences in bird community between the habitat classes, and the small distances between plots within the habitat classes, also indicates slight dissimilarities as plots with the same bird community would have a distance of zero.

‘Oak’ plot 13, and ‘Pine’ plot 26, to some extent, were unusually positioned on the landscape MDS. ‘Oak’ plot 13 was surprisingly positioned in the middle of the ‘Pine/Broadleaved’ plots, due to the presence of coniferous bird species, such as the Siskin, Goldcrest and Coal Tit. However, ‘Oak’ plot 13 had a markedly low percent cover of pine, therefore the presence of coniferous bird species and the positioning of the plot could be as a result of the adjacent conifer woodland (Bergen et al. 2007, Neumann et al. 2016), or these species adaptation to deciduous woodland (e.g. Broughton et al. 2019). ‘Pine’ plot 26 was positioned on the MDS where the boundary lines converge (at the 0, 0 position), suggesting an intermediate habitat, possibly occurring as a result of the high density of Chaffinches and low density of Wrens.

The relationships with the environmental factors indicate an influence of canopy openness (overstorey penetration depth) on the bird communities of some of the conifer habitats. By contrast, variation in tree height influenced the bird communities in the broadleaved habitats, due to the variable vegetation structure and understorey layer, contrasted with the more uniform height in a pine plantation. The positioning of the percent cover of vegetation in the 2-5 m layer indicated that the most rapid change in the variable was in the direction of the broadleaved plots. However, as the environmental factor was not directly over the broadleaved plots, but located between the ‘Scrubland’ and the broadleaved plots, this suggests that both habitats had a gradient in vegetation at this height range. The strength of this relationship was weak, represented by the length of the arrow in Figure 6.3 and the lower R^2 in Table 6.2, and therefore the woodland plots were analysed alone to investigate the differences in woodland bird community composition (Figure 6.4).

6.5.2.1 Conifer Bird Community

The woodland only MDS arranged the ‘Pine’ and ‘Other Conifer’ plots into two distinct groups on the same side of the ordination (Figure 6.4a & b), indicating differences in bird community composition between the two conifer habitat classes. This is in contrast with findings of Lindbladh et al. (2019) who found that Norway spruce and pine plantations in Sweden had overlapping yet distinct bird communities, although the ‘Other Conifer’ plots in the current study did not contain Norway spruce, possibly resulting in the greater dissimilarities. Furthermore, distinct dissimilarities in bird community composition within both the ‘Pine’ and ‘Other Conifer’ classes were represented by the relatively large distances between the plots on the woodland ordination.

The Firecrest was the main bird species separating the ‘Other Conifer’ plots from the ‘Pine’ plots on the woodland ordination (Figure 6.4c). The greater distances, and therefore dissimilarities in bird community composition, between the ‘Other Conifer’ plots were probably because of the different coniferous vegetation (Figure 6.4a). Other coniferous preferring bird species, such as the Goldcrest, Siskin and Goldfinch, were also located with the ‘Other Conifer’ plots, indicating higher densities, and therefore habitat preference, possibly highlighting the detrimental effect of pine on birds (Section 4.5.3). Unexpectedly, species such as the Long-Tailed Tit, Woodpigeon and Wren were associated with and present in higher densities in the ‘Other Conifer’ plots. Lindbladh et al. (2019) found that the Wren and the Woodpigeon were unique to Norway spruce plantations, which are similar in leaf morphology to Douglas fir (present in the current study); however, the Long-tailed Tit may be foraging around the edge of the woodland, passing through, or could be collecting spiders’ silk for their nests (McGowan et al. 2004). Furthermore, the Wren may benefit from reduced competition and an increase in nesting sites in the lower vegetation in more open conifer plots, due to its adaptability to multiple habitats (Armstrong and Whitehouse 1977). This indicates that the ‘Other Conifer’ plots also provided habitat for other bird species, increasing bird diversity, possibly as a result of continued management. This highlights the importance of the ‘Other Conifer’ habitats in the New Forest for supporting rare specialist coniferous species, such as the Firecrest, but also supporting a greater bird diversity than the ‘Pine’ habitats (see Section 4.4.3). Additionally, ‘Other Conifer’ plot 28 contained a number of broadleaved preferring bird species (e.g. Treecreeper and Nuthatch), positioning it nearer the broadleaved plots on the woodland MDS, as a consequence of the adjacent scrubby broadleaved vegetation, further indicating an influence of surrounding habitat on bird community composition (Bergen et al. 2007, Neumann et al. 2016).

The Coal Tit and Common Crossbill are typical conifer specialists located in the ‘Pine’ section of the MDS (Fuller et al. 2005a), however, unusually the Dunnock, Collared Dove and Great Spotted Woodpecker were also located in the ‘Pine’ section (Figure 6.4c). The Dunnock may prefer the lower vegetation cover in the open pine plots, and Collared Doves regularly nest in conifers as they provide good cover and support for their nests. The association of the Great Spotted Woodpecker, thought to be a predominantly broadleaved species (Donald et al. 1998), may be as a result of increased observer effort due to higher detectability (easier to observe in an open pine plantation than in a dense or complex broadleaved woodland). This is supported by the relationship of the environmental factor depicting the openness of the canopy with the ‘Pine’ plots. Also, food items may be more readily available and easier to access in the bark or cones of pine trees (Myczko and Benkman 2011). However, the Great Spotted Woodpecker has a broad habitat range, and therefore presence in the conifer plots is less unusual. Cadieux and Drapeau (2017) showed that mixed woodlands in Canada had a higher number of deadwood bird species, and Summers (2004) showed that the Great Spotted Woodpecker excavated and nested in snags (dead standing wood) in Scots pine in Scotland. However, the diameter at breast height (DBH) of the snags with nesting holes were relatively large (median = 51 cm, Summers 2004). The DBH of the trees in the current study in southern England were not measured, but the ‘Pine’ plots were often relatively young plantations (*pers. obs.* A. Barnes), suggesting that the Great Spotted Woodpecker may have only utilised the ‘Pine’ plots for feeding, rather than nesting.

The location of the Common Crossbill on the woodland MDS was a result of ‘Pine’ plot 23 having the highest density, which suggests a preference for open pine in line with Fuller et al. (2005a), and movement around the landscape following ripe cone availability. This suggests that although pine is poor in terms of the bird indices (Section 4.5.3), this habitat class varied in bird community composition and provided habitat for the relatively rare Common Crossbill. The environmental variable for the percent cover of pine was directly related to ‘Pine/Broadleaved’ plot 20, increasing to ‘Pine’ plots 23-25 and with the highest percentage in ‘Pine’ plot 27. However, ‘Pine’ plot 27 was not positioned on the pine environmental factor line, suggesting that differences in bird community were not necessarily as a result of the amount of pine present, but partly as a result of the absence of Coal Tits.

‘Pine’ plot 26, ‘Pine/Broadleaved’ plot 21 and ‘Oak’ plot 13 were significantly related to the presence of bracken, possibly increasing the number of scrubby birds, such as the Greenfinch. Notably, this variable was not significant in the multi-model analysis in Section 4.4.5, indicating that bird community composition is influenced by alternative variables to measures of bird diversity. ‘Pine’ plot 26 was relatively dissimilar to the other ‘Pine’ plots, possibly as a result of being bordered by heathland on two sides. The silver birch on the edge of the plot, along with the presence of bracken, presumably increased the number of broadleaved indicator bird species, such as the Chaffinch and scrubby species, such as the Greenfinch, most likely temporarily encroaching into the plot, possibly for feeding.
‘Pine/Broadleaved’ plot 21 also contained a number of deciduous bird species, driving the relationship with the presence of bracken, however, this may also be due to this mixed plot being surrounded by broadleaved woodland except on one side. The ordination of the plots also suggests that the bird species in ‘Oak’ plot 13 were correlated with the presence of bracken, possibly altering the species composition to that of a more open canopied woodland habitat, similar to pine.

‘Pine/Broadleaved’ plot 22, positioned in the centre of the woodland MDS, was predominantly pine with various other vegetation species, including silver birch at the interface between the pine and the open, maintained grass (caravan and camping site). Willow, young western hemlock, bracken and bramble were also present in the plot, providing host to a variety of bird species, including the Chiffchaff, Greenfinch, as well as coniferous preferring bird species and was the only pine plot to contain the rare Wood Warbler and Firecrest. However, as the Firecrest tends to avoid pine, this was presumably as a result of the other conifer species (western hemlock) in this plot. Wood Warblers are influenced more by vegetation structure, suggesting that occurrence in this plot was because of the age, and therefore structure of the vegetation. This plot may have previously been a plantation, presumably abandoned as a result of the close proximity to the caravan site. The pine trees were more mature and the plot less managed allowing other vegetation species to persist (*pers. obs*. A. Barnes), and subsequently more bird species in line with Baguette et al. (1994), Paquet et al. (2006) and Fuller and Robles (2018). Consequently, this plot also had the highest abundance of Chaffinches out of all 32 plots in the New Forest, or possibly as a result of anthropogenic interaction, either from deliberate feeding or from discarded food. ‘Pine/Broadleaved’ plot 22 represents a highly mixed intermediate habitat and hence it was positioned in the middle of the ordination and at the intersection of all the environmental variables (Donald et al. 1998).

The UK population of Wood Warblers is only 6500 singing males due to significant population declines in lowland Britain of 33% between 1980 and 2009, and a further 8% decline from 2005 to 2015 (Musgrove et al. 2013, Massimino et al. 2017). Although the decline is slowing, population numbers are critically low, hence the Wood Warbler is RED listed in the UK (Eaton et al. 2015). Huber et al. (2016, 2017) showed that the Wood Warbler prefers broadleaved forests on fairly steep areas with nutrient poor soils and a homogeneous structure for breeding. Fuller (2001) also noted that the Wood Warbler prefers heavily grazed open woodlands. This provides an explanation for the presence of the Wood Warbler in ‘Pine/Broadleaved’ mixed plot 22 as it is predominantly mixed woodland with an understorey of other conifer species (western hemlock), in a heavily grazed area on a relatively undulating slope, and explains the higher IRR in this plot (Section 4.4.3). The Wood Warbler was also present in ‘Beech/Oak’ plot 15, and although it has not been exposed to heavy grazing, there is a lack of understorey and shrubby vegetation under the beech canopy. This suggests that the Wood Warbler possibly benefits from grazing or is otherwise unaffected. However, the extremely low abundance of the Wood Warbler recorded in the current study of the New Forest is indicative of species rarity, and therefore lacks enough evidence to infer habitat preference. A recent study by Buchanan et al. (2018) suggested that the wintering grounds of the Wood Warbler in Africa was not the key driver of the population decline, as they actually prefer deteriorated woodland with less woodland cover on their wintering grounds. This suggests that the decline in the Wood Warbler is more than likely as a result of poor breeding habitat in the UK rather than over wintering habitat, therefore more research must be carried out to conserve this species.

The structural variable overstorey penetration depth, detailing the openness of the canopy, was positioned directly opposite the percent cover of vegetation in the 2-5 m layer on the woodland MDS (Figure 6.6). This was most likely a consequence of the understorey vegetation layer at 2-5 m preventing the laser from penetrating further in the broadleaved woodlands, therefore creating a lower overstorey penetration depth and also indicates canopy openness. Generally, there was no understorey vegetation in the open ‘Pine’ plots, whereas, this was not the case in some of the ‘Other Conifer’ habitats which often had a relatively open canopy overall increasing the percent cover of understorey, therefore positioning the plots perpendicular to these two structural variables on the woodland MDS.

The non-significance of the other structural variables indicates that the woodlands did not differ in tree height or that the bird assemblage was not affected by any differences in the height or variation in tree height, but more by canopy openness and vegetation composition. This supports the findings by Swift et al. (2017) who noted a weak relationship of avian richness to vegetation structure and found that canopy composition was the most important variable influencing total richness. Flade (1994, in Wesołowski et al. 2018) showed that in temperate forests in Germany, pine woodlands were particularly low in bird diversity, however, the addition of Norway spruce increased species richness and resulted in a different assemblage of bird species in line with the findings of the current study. Lindbladh et al. (2019) on the other hand, found that conversion of pine to Norway spruce would increase homogeneity over the landscape, but including coniferous reserves increases overall gamma diversity. This supports the importance of the ‘Other Conifer’ plots in the New Forest providing habitat for different bird species assemblages and increasing landscape (gamma) diversity.

6.5.2.2 Broadleaved Bird Community

The ‘Oak’, ‘Beech’ and ‘Beech/Oak’ mixed plots were, on the whole, grouped together in a broadleaved cluster on the woodland MDS indicating similar bird species composition, with the exception of ‘Oak’ plot 13 which was again positioned in the middle of the ‘Pine/Broadleaved’ plots. ‘Oak’ plot 13 had a high density of Coal Tits and Siskins, but also contained twice as many Goldcrests as ‘Pine/Broadleaved’ plot 21, even though the plot contained very little pine. Goldcrests were not uncommon in the broadleaved plots, but were more often at higher densities in the conifer plots, hence a greater association to the ‘Other Conifer’ plots. This was presumably a result of the conifer in the adjacent woodland, demonstrating that the survey plots were not exclusive of their surroundings and birds were not restricted to the survey plots (Bergen et al. 2007, Neumann et al. 2016).

Although the majority of the broadleaved plots were close together on the MDS, ‘Beech’ plot 10 and ‘Oak’ plot 12 were together a distance from the rest of the broadleaved plots and close to the horizontal conifer–broadleaved boundary line. This was presumably because ‘Oak’ plot 12 contained some pine and ‘Beech’ plot 10 was bounded on one side with conifer, resulting in the highest density of Goldcrests in the broadleaved plots. Broadleaved plots 10 and 12 were also significantly and directly related to the percent cover of vegetation at 2-5 m (Figure 6.6), indicating an increase in understorey vegetation at this stratum, as a result of ‘Oak’ plot 12 being in an “Inclosure” woodland excluded from high levels of grazing (Tubbs 2001), and thus increased niche availability. Bellamy et al. (1998) showed that Nuthatches were related to shrubby hazel understorey. Whilst hazel was absent from the vegetation survey in the current study in the New Forest, the shrubby vegetation at 2-5 m may have resulted in the highest density of Nuthatches in ‘Oak’ plot 12, or this may also be a result of increased breeding success in oak woodlands (Nilsson 1976). ‘Beech’ plot 10 also had the highest density of Jackdaws which was closely associated to this plot on the MDS (Figure 6.4), possibly due to the greater amount of deadwood (*pers. obs.* A. Barnes) creating more nesting sites (Cadieux and Drapeau 2017). This indicates that there are multiple factors, other than the variables of vegetation composition and structure, used in the current study, influencing bird species community composition.

The occurrence of the Cuckoo in ‘Oak’ plot 12, further indicates that this plot contained the relevant scrubby or shrubby woody vegetation for the Cuckoo host species, for example the Dunnock, contributing to the positioning of the plot on the MDS (Douglas et al. 2010). ‘Oak’ plot 12 was adjacent to a relatively recently felled plot that was in the primary stage of succession, which most likely resulted in the presence of the Tree Pipit (*Anthus trivialis*), Stonechat and Whitethroat in this woodland plot. This again demonstrates the effect of surrounding habitat and the wider landscape on bird community composition (Paquet et al. 2006, Bergen et al. 2007, Neumann et al. 2016). Paquet et al. (2006) noted in an agricultural context that clearcutting or felling in a forest created important, temporary, successional habitat that was occupied by shelter specific bird assemblages, rather than an intermediary of birds from the woodland and agricultural land. Thus, ‘Oak’ plot 12 was the only woodland plot to contain the Tree Pipit, supporting a benefit of selective felling in forested areas in the New Forest (Paquet et al. 2006).

‘Beech/Oak’ mixed plot 17 was also relatively dissimilar to the other broadleaved plots, but in the opposite direction to plots 10 and 12, and further away from the vertical conifer gradient on the MDS (Figure 6.4). This was a result of fewer conifer preferring bird species, and was the only woodland plot without conifer specialist Goldcrest, which was consequently positioned directly opposite this plot on the ordination. ‘Beech/Oak’ plot 17 also had the highest density of Blackcaps, possibly as a result of being surrounded by broadleaved copses. The environmental factors of percent cover of both oak and beech were almost in the same position on the woodland MDS, reinforcing that these two broadleaved tree species were not mutually exclusive in the New Forest, but often occurred together and equally explained the ordination of the broadleaved plots. This suggests that the bird communities may be robust to declines in one or the other broadleaved tree species, but not both. However, the spread of the plots on the MDS suggests that the environmental factors may have been present in differing quantities or other factors may be influencing the bird communities to create dissimilarities between the broadleaved plots.

The presence of holly was also positioned next to the percent cover of oak and beech on the woodland MDS (Figure 6.5), as a result of it being most often found in these woodlands. Moreover, the arrow for the presence of holly was pointing lower down than the beech and oak factors, indicating that broadleaved plots 16, 11, and possibly even 12 and 10, might have had more bird species, such as the Goldcrest, associated with holly. The Firecrest was also present in all except four of the broadleaved woodland plots, presumably as a result of the population range expansion utilising holly in the broadleaved woodlands of the New Forest (Batten 1973, Ward and Wynn 2011, Clements et al. 2017). However, this was not apparent in the MDS as differences in the broadleaved plots were attributable to other bird species and differences in bird community composition overall, but does provide an explanation for the significant relationship of holly with the broadleaved plots. Nevertheless, the woodland MDS shows that occurrence of the Hawfinch in ‘Beech/Oak’ plot 16 also resulted in the highest rarity index of the woodland plots (Section 4.4.3). The Hawfinch, the second rarest species recorded in the current New Forest study, was also contributing to the IRR value in broadleaved plots 10, 11, and 16, and also in ‘Beech/Oak’ plot 19 (Section 4.4.3). For the UK as a whole, 800 pairs were recorded in in 2011 (Clements 2013, Appendix A2), with the New Forest lying in the western part of the species’ range (Kirby et al. 2015). The IUCN Red List stated that the Hawfinch was increasing globally, however, the UK population is RED listed due to population declines between the mid-1980s and 2003-04 (Eaton et al. 2009, 2015, BirdLife International 2016). The scarcity of the Hawfinch in the UK, due to these rapid declines over the past 25 years, makes it difficult to determine annual population numbers (Hewson et al. 2007, Clements 2013, Kirby et al. 2015). Furthermore, contrary to the IUCN Red List Statement, Kirby et al. (2015) stated that the short-term European trend from 1990-2011 actually showed a decline of 30% of the Hawfinch population. This suggests that conservation efforts should be stepped up Europe-wide and most particularly in the UK to halt further declines. Ward (2014) found that in the north and east of the current study site in the New Forest, Hawfinches preferred Norway spruce and Douglas fir for roosting. Conversely, for breeding, Hawfinches prefer greater woodland cover in deciduous woodlands (Kirby et al. 2015), thus the greater association with the four broadleaved plots in the current study.

Overall, this analysis indicates dissimilarities in bird community composition between the broadleaved habitats in the New Forest. The results of the current study suggest that a range of broadleaved woodlands varying in vegetation structure and species composition is required to encompass multiple bird communities, also with differing levels of grazing by maintaining exclusion of ungulates, to allow understorey growth to maximise niche availability.

6.6 Conclusion

MDS graphically demonstrates the similarities and dissimilarities in bird community composition, and the bird species responsible for the ordination, based on bird habitat preferences. In the non-woodland habitats, three guilds; wetland, scrubby and open habitat preferring birds, were responsible for the bird community composition. The woodland habitats were more similar in community composition on the landscape MDS, but once separated showed differing community composition as a consequence of vegetation composition. The birds responsible for the ordination were also shown to support their contribution to the bird indices, such as the Firecrest in the ‘Other Conifer’ plots and the Dartford Warbler in the ‘Heathland’ driving IRR. Furthermore, although pine was deemed poor bird habitat in terms of the bird indices in Section 4, the bird community differed from the other habitats. Pine/broadleaved mixed habitats supported rare and declining species such as the Wood Warbler, and thus should not be fully eradicated. This analysis indicates that bird-habitat relationships should not only encompass vegetation structure and composition, but also the ecological importance of habitats for bird communities and other factors, such as wetness, the amount of deadwood and the surrounding habitat. Management of multiple variations of these habitat classes are required to support multiple and varying bird communities. Furthermore, the bird community analysis highlighted additional variables to those found to be important for the bird indices in Sections 4.4.4 and 4.4.5, indicating that multiple measures of bird occurrence and status should be evaluated over the landscape prior to the implementation of any management strategies.

7 Bird Community Analysis – Cambridgeshire

7.1 Abstract

Bird community composition has often been analysed using diversity measures or by selecting particular species. Multi-Dimensional Scaling (MDS) is a useful tool to graphically display the similarities and dissimilarities in community on 2D space. Unsurprisingly, the current study found distinct dissimilarities in the bird communities between the field and woodland habitats in Cambridgeshire. The fields adjacent to woodlands also often contained dissimilar community composition to the other field habitats, and were associated to the crop variables. The scrubby woodland habitat was particularly dissimilar to the other woodlands (which were clustered), and supported the globally threatened Turtle Dove. Once separated, the woodland habitats were positioned not by edge or interior classification, as would be expected, but by their structural characteristics. Furthermore, the smallest woodlands were not intermediary in bird community. Notably, similar habitats with similar diversity measures in the same woodland were also shown to differ in bird community composition.

7.2 Introduction

Bird communities are often studied using a measure of diversity or take different forms (see Section 6.2). The habitats in the Cambridgeshire landscape were shown in Section 0 to differ in bird diversity as well as the other bird indices. However, habitats with similar bird diversity could differ in bird community composition as a result of a number of factors, such as vegetation structure and composition, prey/food availability or even inter/intraspecific competition. The current study uses Multi-Dimensional Scaling (MDS) to investigate bird community composition in the habitats over the agricultural landscape in Cambridgeshire and the bird species responsible for the differences. The habitat compositional and structural variables were also analysed to understand the drivers creating any differences in bird community composition between habitats, and habitats with similar measures of the bird indices.

7.3 Methods

The bird abundance data collected (following 3.3.4.2, pp. 69 and 3.3.5.2, pp. 70 – 72) was used to create MDS plots which displayed the similarities in bird species composition between the habitats in Cambridgeshire. Following Oksanen (2015) the Bray-Curtis dissimilarity was used and the *metaMDS* function submitted the data to Wisconsin double standardisation to fit the data to the MDS with the lowest stress (stress less than 0.2 is considered good) and was plotted in 2D (Kruskal 1964, Wickelmaier 2008). This was carried out at the landscape level for all 38 transects (stress = 0.123), and at the local level for the 22 field transects (stress = 0.157), and the 16 woodland transects (stress = 0.155), with MDS plots of the bird species responsible for each ordination included (3.3.8, pp. 78 – 79).

To identify which particular structural or compositional aspects of each habitat were most strongly driving the bird species assemblages associated with each, the MDS plots were reproduced with the LiDAR-derived vegetation structural metrics (5.4.2, pp. 133 – 137) and vegetation composition variables (5.4.1, pp. 132 – 134) presented using the *envfit* function in R version 3.5.1/3.5.2 (3.3.8, pp. 78).

It is worth noting that in the MDS plots the axis values do not represent any relative measure or value, the MDS simply displays the habitats by the similarities and dissimilarities of bird communities; therefore it is the distance, rather than the axis values, that should be interpreted (Borgatti 1997).

7.4 Results

7.4.1 Multidimensional Scaling (Landscape Scale)

Multidimensional Scaling (MDS) in Figure 7.1 graphically displays the survey transects in the Cambridgeshire landscape based on the similarities and dissimilarities in bird community composition. The ordination was based solely on bird density and composition in each transect and therefore displays bird-habitat selection preferences (Figure 7.1a). The second MDS plot in Figure 7.1b displays the bird species in the Cambridgeshire landscape arranged in ordination space related to the survey transects, and facilitates the interpretation of the bird species responsible for the bird-habitat trends.

The transects were arranged on the MDS with a woodland–non-woodland gradient along the horizontal axis, with the woodland transects clustered close together on the right hand side of the graph (a distance from the field transects), indicating similar woodland bird community composition. ‘Wood Edge’ transect 24 was slightly separate from the rest of the woodland transects, possibly as a result of the scrubby vegetation, indicated by the occurrence of the Turtle Dove, Willow Warbler and Bullfinch (Figure 7.1b). A number of species were illegible on the MDS in Figure 7.1b, which therefore suggests that further investigation on the woodland transects separately is required.

On the field transect side of the MDS, the transects did not show a gradient of decreasing woody vegetation (in this case represented by hedges), as one would have expected given the horizontal woodland–non-woodland gradient. The field transects with various *P_HedgeLen* were arranged together with no distinctive clusters, suggesting that there may be other factors (such as vegetation structure or composition) resulting in the variations in bird community within the field habitat classes (Figure 7.1). ‘Field_high%_Hedge’ transect 16 was positioned in the centre of the MDS in Figure 7.1a, close to the woodland boundary line, most likely as a result of the high *P_HedgeLen* (151.5%) increasing the number of broadleaved woodland bird species present (e.g. the Blackbird). The occurrence of certain bird species creates dissimilarities between the field transects, for example ‘Field_high%_Hedge’ transect 12 and ‘Field_med%_Hedge’ transect 7 were the only transects to have the Cuckoo and Pied Wagtail (*Motacilla alba*), respectively (Figure 7.1b). The Dunnock was positioned in the centre of the MDS as a result of being present in almost all of the transects (31/38 transects), utilising both woodland and field habitats in Cambridgeshire (Figure 7.1b).

The ‘Field By_Wood’ transects were positioned a greater distance from one another, with transects 19, 20 and 22 furthest from the other field transects indicating greater dissimilarities in bird community, possibly as a result of the close proximity of woodland (Figure 7.1a). ‘Field By_Wood’ transect 19 was located at the bottom of the MDS near the vertical boundary line, possibly as a result of the high density of the Carrion Crow (Figure 7.1b). On the opposite side of the MDS, ‘Field By_Wood’ transect 22 was most likely located as a result of the occurrence and high density of the Rook (*Corvus frugilegus;* 6.76 birds/ha), possibly as a result of the high *PCov_ImpGrass* (93.1%).

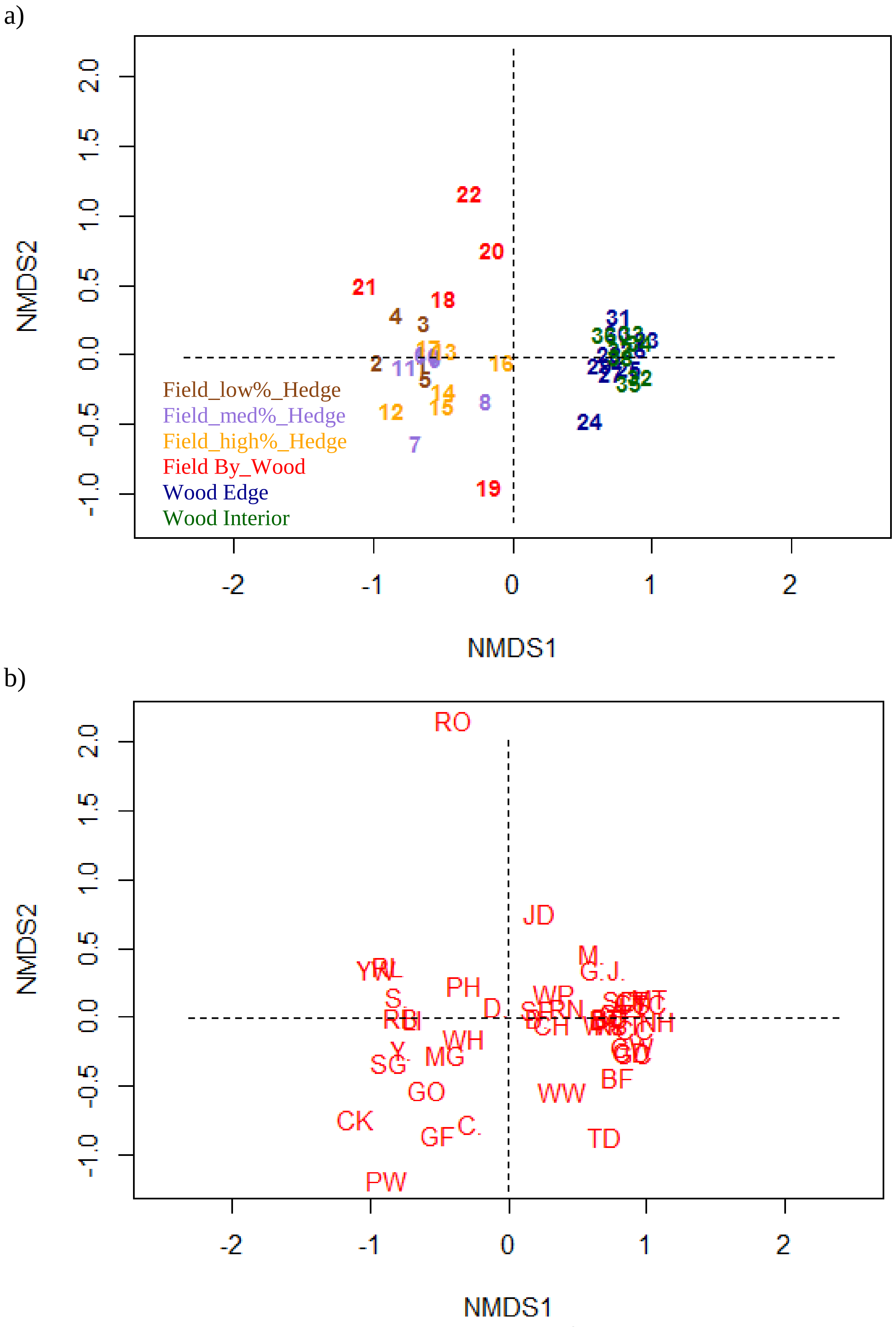


Figure 7.1. Multidimensional Scaling (MDS) plots of a) the 38 survey transects in Cambridgeshire annotated with dashed lines through 0,0, and b) the bird species in 2D space responsible for the ordination of the transects. See Appendix A1 for bird species codes (R version 3.5.2).

7.4.2 Multidimensional Scaling with Environmental Factors

7.4.2.1 Landscape Scale with Vegetation Composition

Figure 7.2 displays the significantly correlated vegetation composition variables from Table 7.1 to the ordination (at $P < 0.05$). All three crop vegetation variables were significantly related to the ordination in different directions towards the field transects, represented by three separate arrows, and signifies the greatest gradient in the factor (Figure 7.2). *PCov_ImpGrass* was significantly ($R^2 = 0.30$; $P = 0.005$) and directly related to 'Field By_Wood' transects 20 and 22; *PCov_CropOther* was significantly ($R^2 = 0.29$; $P = 0.003$) and directly related to 'Field_low%_Hedge' transects 3 and 4 (less so to 'Field By_Wood' transects 18 and 21; Figure 7.2). *PCov_Cereal* was pointing further down on the MDS and significantly related ($R^2 = 0.54$; $P = 0.001$) to 'Field_high%_Hedge' transects 14, 15 and 12 and 'Field_med%_Hedge' transect 7 (Figure 7.2). This indicates that the bird communities in the field transects varied by crop type.

PCov_Ash and *PCov_Oak* were significantly related with the highest R^2 (0.74, 0.71 respectively) in the direction of the woodland transects, indicating that the woodland transects were fitted first on the MDS (Table 7.1 and Figure 7.2). *PCov_Ash* and *PCov_Oak* were slightly separate on the MDS, possibly indicating minor differences in the bird communities in the woodlands (Figure 7.2); therefore further analysis may be required to investigate the woodland transects separately. *PCov_Maple* was also significantly related to the ordination in the direction of the woodland transects ($R^2 = 0.57$; $P = 0.001$) in almost the same position as *PCov_Ash,* but not as strongly (indicated by the lower R^2 and length of the arrow), suggesting that these two variables were somewhat interchangeable and similarly influenced woodland bird community (Figure 7.2). *PCov_BroadOther* was also significantly related ($R^2 = 0.21$; $P = 0.023$) to the ordination, but in the direction of 'Wood Edge' transect 24 (Figure 7.2), presumably as a result of the scrubby vegetation, such as silver birch, in this transect.

It is worth noting that the non-significance of *PCov_Elm, PCov_Cons, Pres_Thorns, Pres_Bramble* and *Pres_Shrub* (Table 7.1) to the ordination, indicates either the vegetation was present in too low quantities to show any effect on bird community, or present in the majority of transects, therefore not contributing to any variation in bird species assemblage.

*Table 7.1. The envfit output for the vegetation composition for all transects in Cambridgeshire. The stars represent the level of significance: '***' <0.001, '**' <0.01, '*' <0.05, '.' <0.1. This was carried out with 999 permutations.*

Variable	NMDS1	NMDS2	R^2	P	Significance
PCov_Ash	0.978	0.211	0.744	0.001	***
PCov_Oak	0.997	0.074	0.714	0.001	***
PCov_Maple	0.975	0.224	0.566	0.001	***
PCov_Elm	0.998	0.066	0.119	0.081	.
PCov_BroadOther	0.571	-0.821	0.209	0.023	*
PCov_Cons	0.755	-0.656	0.050	0.317	
Pres_Thorns	0.919	0.395	0.035	0.579	
Pres_Bramble	-0.978	0.208	0.017	0.718	
Pres_Shrub	0.766	0.643	0.070	0.295	
PCov_ImpGrass	-0.289	0.957	0.304	0.005	**
PCov_Cereal	-0.842	-0.540	0.539	0.001	***
PCov_CropOther	-0.954	0.298	0.292	0.003	**

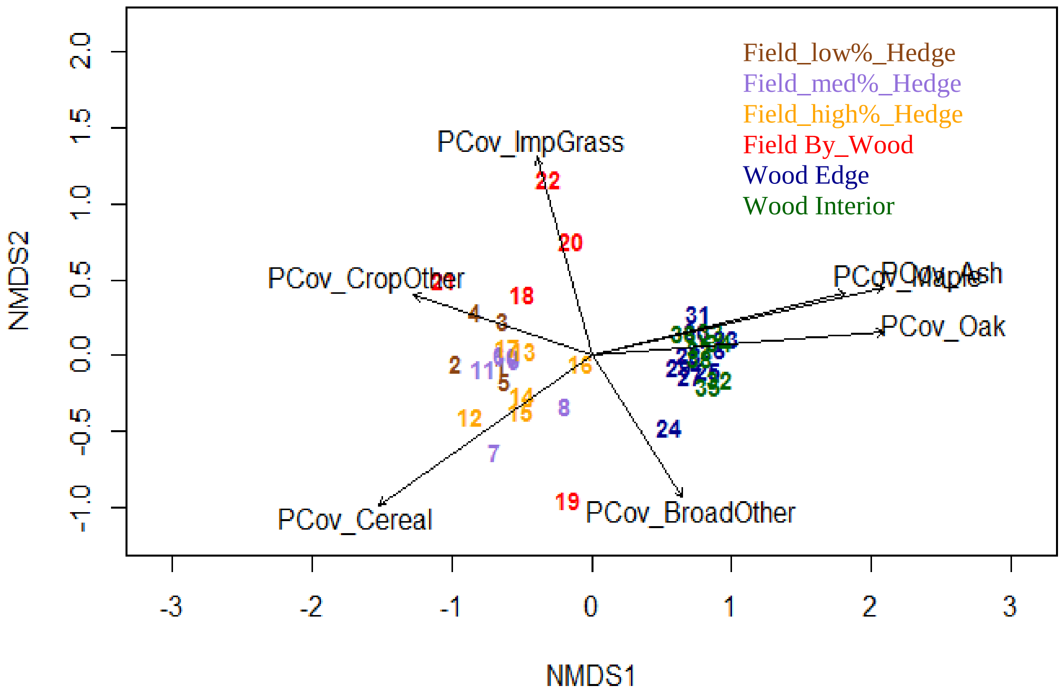


Figure 7.2. MDS of all the survey transects with the envfit function displaying significant (at P <0.05) correlated vegetation composition as environmental factors to explain the ordination. See Table 7.1 for the P values (R version 3.5.1).

7.4.2.2 Landscape Scale with Vegetation Structure

All the vegetation structural variables were significantly related to the ordination at $P < 0.05$ (Table 7.2). Expectedly, *PCov_<0.5m*, representing lower or ground vegetation, was significantly related ($R^2 = 0.60$; $P = 0.001$) to the ordination in the direction of the field transects (Figure 7.3). The location of the arrow in Figure 7.3 shows the gradient of the most rapid change in this variable from 'Field_med%_Hedge' transect 11 to 'Field_low%_Hedge' transect 2 (66.3% and 81.9% ground, respectively). *P_HedgeLen* and *Ostorey_PenDepth* were significantly related to the ordination in a similar position on the MDS ($R^2 = 0.34$, 0.39 respectively; $P = 0.001$ for both), suggesting that they are interchangeable (Figure 7.3). This is possibly due to transects with higher *P_HedgeLen* having a greater chance of containing a tree, therefore increasing the height of the vegetation in open habitats, creating greater penetration depth to the laser pulses. The gradient in *PCov_0.5-2m* from 'Field_high%_Hedge' transects 13 to 17 was weaker ($R^2 = 0.27$; $P = 0.003$) represented by the length of the arrow, which may suggest that other transects also varied in this metric (Figure 7.3). The position on the MDS of the structural variable *Ht_VDR* in Figure 7.3, indicates a gradient in the ratio and suggests that 'Field By_Wood' transect 19 had greater variation in vegetation structure, possibly as a result of the adjacent woodland.

Ht_Av, *PCov_>5m* and *Ostorey_Ht* in Figure 7.3 were directly opposite *PCov_<0.5m* and significantly related to the ordination ($R^2 = 0.83$, 0.86, 0.71, respectively, $P = 0.001$ for all). These variables depict tall vegetation and were unsurprisingly in the direction of the woodland transects. The high R^2 values also indicate that the woodland transects were plotted first on the ordination (Table 7.2 and Figure 7.3). *Ht_Av* and *PCov_>5m* were in a similar position on the MDS, and are therefore interchangeable, as high values for both are indicators of woodland (Figure 7.3). *Ostorey_Ht* and *Ht_StDev* were positioned slightly apart, suggesting that slight structural differences in the woodland transects create dissimilarities in the bird community, indicating that analysis of the woodland transects alone is required (Table 7.2 and Figure 7.3). *PCov_2-5m* was directly related to 'Wood Edge' transect 24 ($R^2 = 0.40$; $P = 0.001$), indicating a high percent of vegetation in the 2-5 m layer and an additional indication of the scrubby vegetation in this transect influencing the bird community, and therefore the position on the MDS (Figure 7.3).

*Table 7.2. The envfit output for the vegetation structural metrics for all transects in Cambridgeshire. The stars represent the level of significance: '***' <0.001, '**' <0.01, '*' <0.05, '.' <0.1. This was carried out with 999 permutations.*

Variable	NMDS1	NMDS2	R^2	P	Significance
P_HedgeLen	-0.754	-0.657	0.338	0.001	***
Ht_Av	0.989	0.146	0.830	0.001	***
Ht_StDev	0.996	-0.087	0.825	0.001	***
Ht_VDR	-0.523	-0.852	0.367	0.001	***
Ostorey_Ht	0.940	0.342	0.711	0.001	***
Ostorey_PenDepth	-0.819	-0.573	0.387	0.001	***
PCov_<0.5m	-0.994	-0.105	0.601	0.001	***
PCov_0.5-2m	-0.992	0.125	0.266	0.003	**
PCov_2-5m	0.794	-0.608	0.397	0.001	***
PCov_>5m	0.994	0.109	0.864	0.001	***

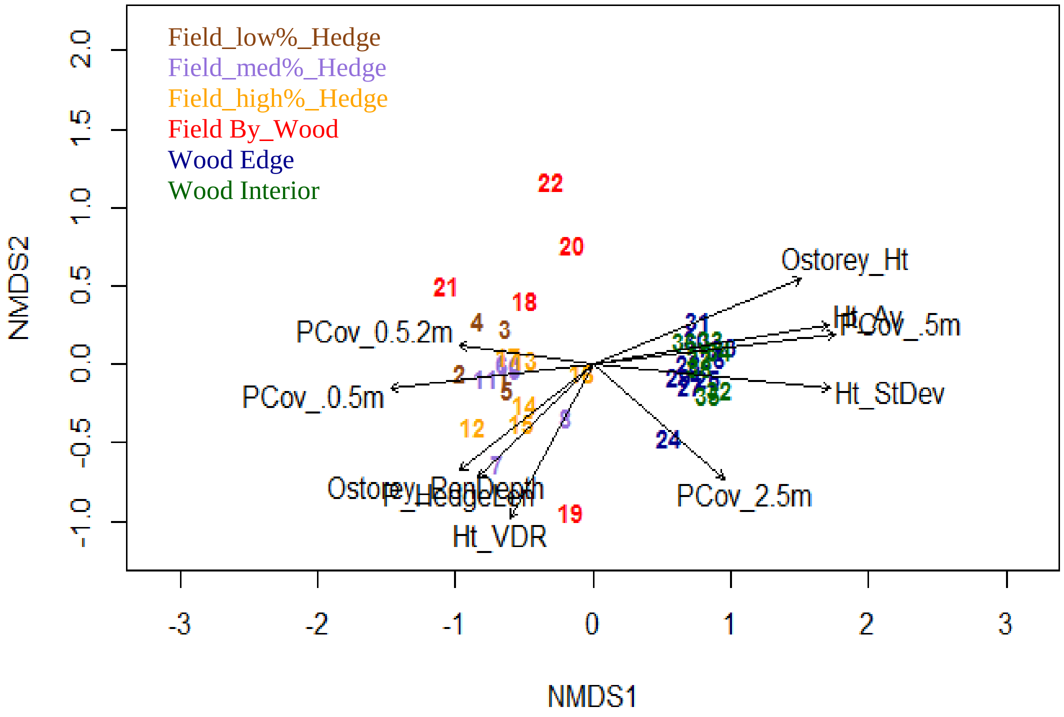


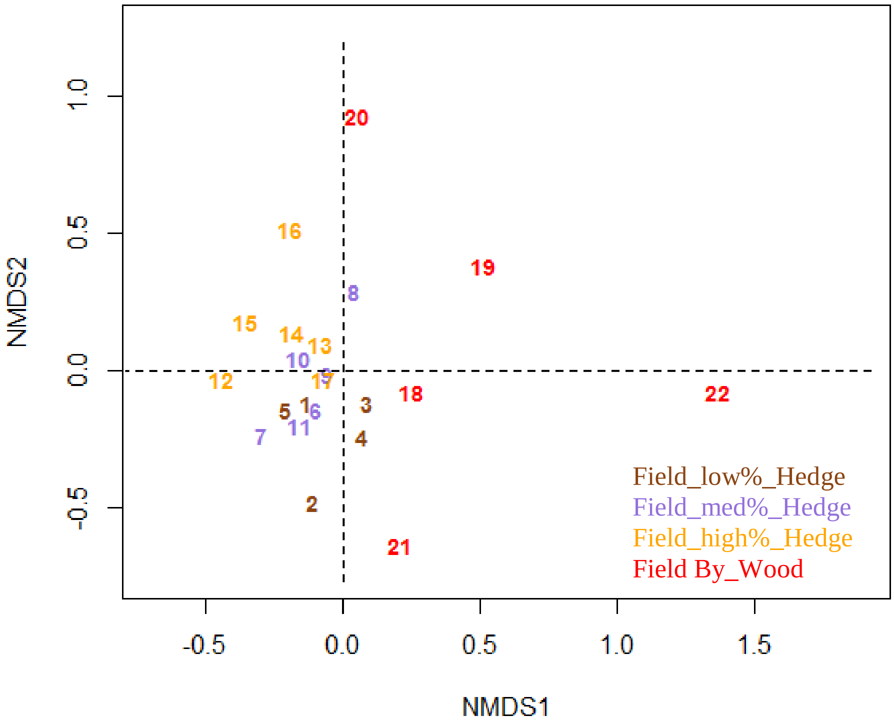
Figure 7.3. MDS of all the survey transects with the envfit function displaying the significant (at P <0.05) vegetation structural metrics as environmental factors relating to the ordination. See Table 7.2 for the P values (R version 3.5.2).

7.4.3 Bird Community Analysis at a Local Scale: Fields

In order to investigate differences in bird community in the agricultural habitats, the field transects were separated from the woodland transects on the ordination. Figure 7.4a showed that the 'Field By_Wood' transects were very dissimilar in bird community composition to each other and dissimilar to the majority of the other field transects. ‘Field By_Wood’ transect 18 was possibly the only exception as it was relatively close to the other field transects, indicating similarities in bird community to the ‘Field_low%_Hedge’ transects, but still, relatively dissimilar to the remainder of the ‘Field By_Wood’ transects. ‘Field By_Wood’ transect 22 was at the greatest distance from any of the other field transects indicating greater dissimilarity, possibly due to the high densities of Rook and Jackdaw shown in Figure 7.4b (2.68 and 6.76 birds/ha respectively).

The field transects appeared to group losely into the habitat classes demonstrating a vertical gradient with some overlap in Figure 7.4a. ‘Field_high%_Hedge’ transects were mostly in the top left corner, ‘Field_med%_Hedge’ in the bottom left corner and ‘Field_low%_Hedge’ in the bottom half of the MDS. The bird species, for example the Garden Warbler and Great Spotted Woodpecker, responsible for the ordination shown in Figure 7.4b, shows that the majority of the scrubby and woodland specific bird species were located as expected with the ‘Field_high%_Hedge’ transects. The species that prefer more open habitat, such as the Skylark, were located in the bottom left of the MDS with the field transects with a lower proportion of hedge length (Figure 7.4b). The Green Woodpecker, Jackdaw and Rook were located in the ‘Field By_Wood’ section of the MDS, presumably as a result of the requirement of both habitats for these species.

a)



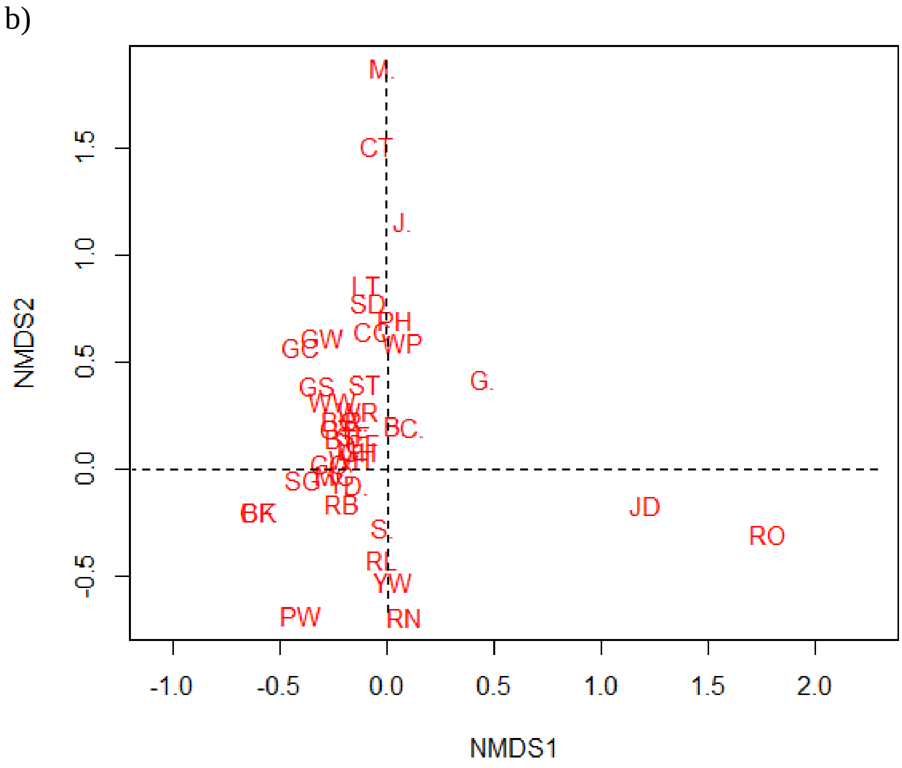


Figure 7.4. a) MDS of the field transects in Cambridgeshire and b) the bird species responsible for the ordination. See Appendix A1 for bird species codes (R version 3.5.2).

7.4.4 Multidimensional Scaling at a Local Scale (Field) with Environmental Factors: Vegetation Composition and Structure

The only significantly related vegetation composition variable to the separated field transect MDS was *PCov_ImpGrass* ($R^2 = 0.52$; $P = 0.01$), in the direction of the 'Field By_Wood' transects as shown in Figure 7.5. This suggests that the dissimilarities in bird community in these transects may be as a result of the high percentage of improved grass, possibly increasing the detectability or presence of corvids (e.g. Jackdaw and Rook in Figure 7.4b). As the other vegetation composition variables were not significantly related (at $P < 0.05$) to the field only ordination (Table 7.3), this suggests that differences in bird community composition between the transects may be influenced by other factors, such as vegetation structure.

Table 7.4 and Figure 7.6 show that the only significantly related (at $P < 0.05$) vegetation structural variable to the field MDS was *P_HedgeLen* ($R^2 = 0.36$;

P = 0.010). This variable was located, as expected, in the direction of the ‘Field_high%_Hedge’ transects, highlighting that the variation in the bird communities was related somewhat to the proportion of hedge length in the transects (Figure 7.6). Moreover, the non-significance of the other variables may be due to the percentage of woody vegetation in the fields being too low to be significant to the ordination, or possibly too similar across the transect classes (Table 7.4). This suggests that other indeterminate factors may be responsible for the dissimilarities in bird community composition or may merely be as a result of stochasticity.

*Table 7.3***.** *The envfit output for the vegetation composition for the field transects in Cambridgeshire. The stars represent the level of significance: ‘***’ <0.001, ‘**’ <0.01, ‘*’ <0.05, ‘.’ <0.1. This was carried out with 999 permutations.*

Variable	NMDS1	NMDS2	R^2	P	Significance
PCov_Ash	-0.877	0.481	0.073	0.449	
PCov_Oak	-0.491	0.871	0.143	0.198	
PCov_Maple	-0.885	0.466	0.236	0.071	.
PCov_Elm	-0.508	0.862	0.148	0.189	
PCov_BroadOther	-0.258	0.966	0.215	0.100	.
PCov_Cons	-0.976	0.217	0.012	0.851	
Pres_Thorns	0.529	0.848	0.027	0.552	
Pres_Bramble	0.947	-0.322	0.007	0.944	
Pres_Shrub	0.998	-0.060	0.036	0.696	
PCov_ImpGrass	0.909	-0.417	0.523	0.015	*
PCov_Cereal	-0.866	-0.500	0.094	0.413	
PCov_CropOther	-0.627	0.779	0.119	0.303	

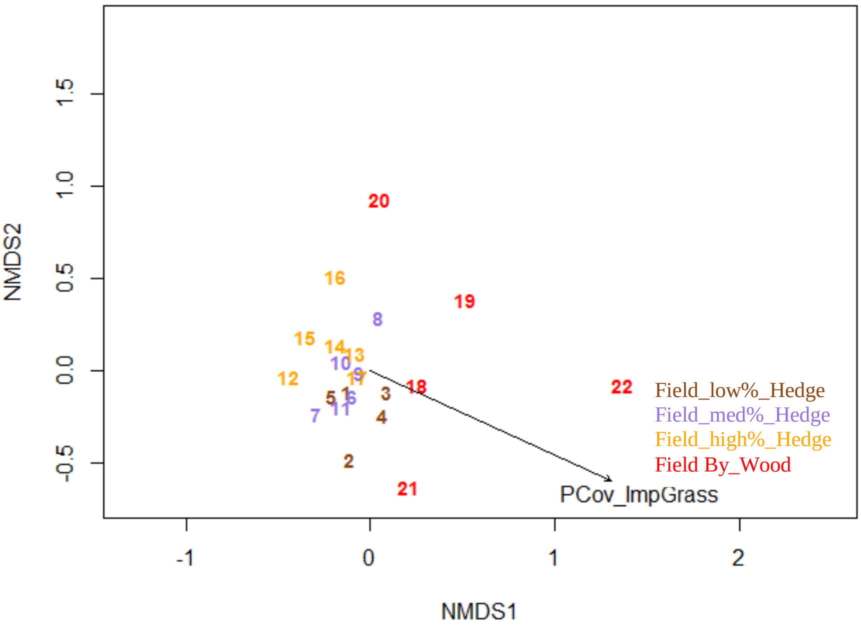


Figure 7.5. MDS of the field transects with the envfit function displaying the significant vegetation composition metrics (at P <0.05) as environmental factors to explain the ordination. See Table 7.3 for the P values (R version 3.5.1).

*Table 7.4. The envfit output for the vegetation structure metrics for the field transects in Cambridgeshire. The stars represent the level of significance: '***' <0.001, '**' <0.01, '*' <0.05, '.' <0.1. This was carried out with 999 permutations.*

Variable	NMDS1	NMDS2	r2	Pr(>r)	Significance
P_HedgeLen	-0.726	0.687	0.355	0.010	**
Ht_Av	-0.737	0.676	0.065	0.499	
Ht_StDev	-0.517	0.856	0.105	0.331	
Ht_VDR	-0.386	0.922	0.228	0.074	.
Ostorey_Ht	0.148	0.989	0.077	0.472	
Ostorey_PenDepth	-0.732	0.681	0.006	0.949	
PCov_<0.5m	0.686	0.727	0.006	0.947	
PCov_0.5-2m	-0.112	-0.994	0.017	0.846	
PCov_2-5m	-0.468	0.883	0.282	0.084	.
PCov_>5m	-0.373	0.928	0.113	0.313	

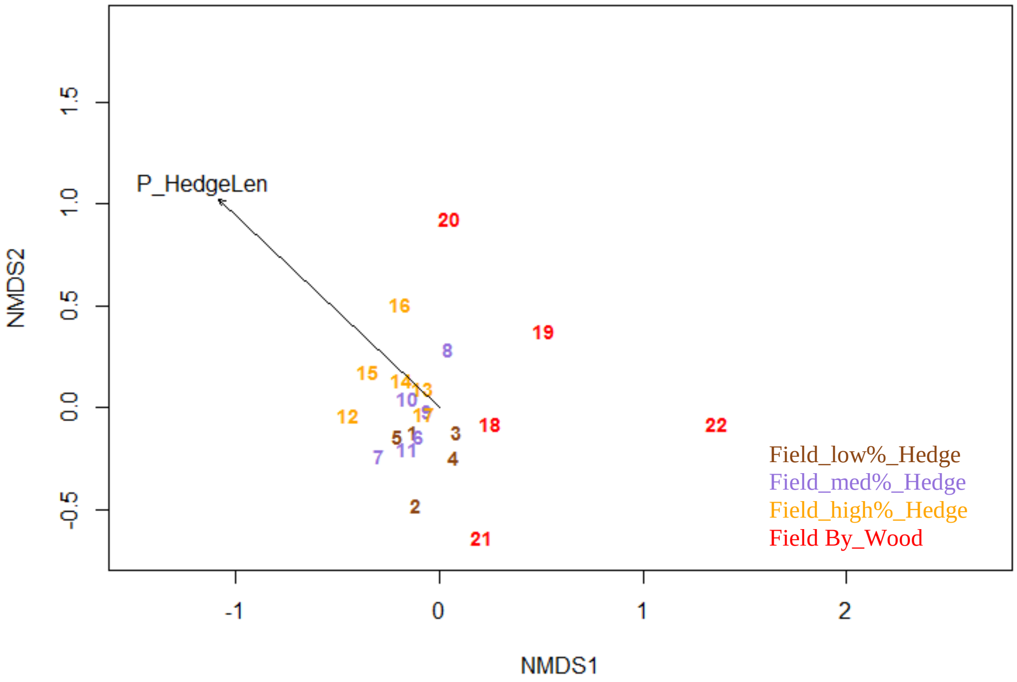


Figure 7.6. MDS of the field transects with the envfit function displaying the significant vegetation structure metrics (at P <0.05) as environmental factors to explain the ordination. See Table 7.4 for the P values (R version 3.5.1).

7.4.5 Bird Community Analysis at a Local Scale: Woodland

Separated, the woodland transects were still relatively clustered together on the MDS in Figure 7.7a, with no obvious gradient or separation between the ‘Wood Edge’ and ‘Wood Interior’ transects, as might have been expected. ‘Wood Edge’ transect 24 was the exception which was positioned relatively far away from the other woodland plots on the MDS (Figure 7.7a), indicating high dissimilarity in bird community composition. This was probably as a result of this transect containing scrubby habitat, indicated by the position of scrubby bird species, such as the Willow Warbler and Turtle Dove, in the MDS in Figure 7.7b. Moreover, dissimilarities in bird communities were indicated by the distances between the woodland transects on the MDS in Figure 7.7a, aside from ‘Wood Edge’ transect 28 and ‘Wood Interior’ transect 38 which almost had a distance of zero, and therefore similar bird community composition.

Figure 7.7b displays the bird species responsible for the ordination of the woodland survey transects with the similarly occurring bird species clustered in the centre of the MDS, and the dissimilarities represented by separated bird species on the MDS in Figure 7.7b. For example, the Goldcrest was related to ‘Wood Interior’ transect 35 as a result of the conifer trees present in this plot, and ‘Wood Edge’ transect 31 was associated with the Spotted Flycatcher, suggesting an open woodland. The Raven was located at the bottom of the MDS, indicating higher abundance in ‘Wood Edge’ transect 25, possibly as a consequence of a high abundance in the adjacent fields. The Marsh Tit was positioned near to ‘Wood Edge’ transect 30 and ‘Wood Interior’ transect 36 as a result of the high density of the species, however, ‘Wood Edge’ transect 23 had the highest density of Marsh Tits. A number of transects were contiguous within the same woodlands, but were relatively dissimilar in bird community composition, noted most obviously in ‘Wood Edge’ transects 25 and 30 (Figure 7.7a). Figure 7.7b also shows that there were a number of bird species clustered around ‘Wood Edge’ transect 25 responsible for the positioning on the MDS, such as the Jackdaw, Collared Dove, Goldfinch, Magpie and Carrion Crow.

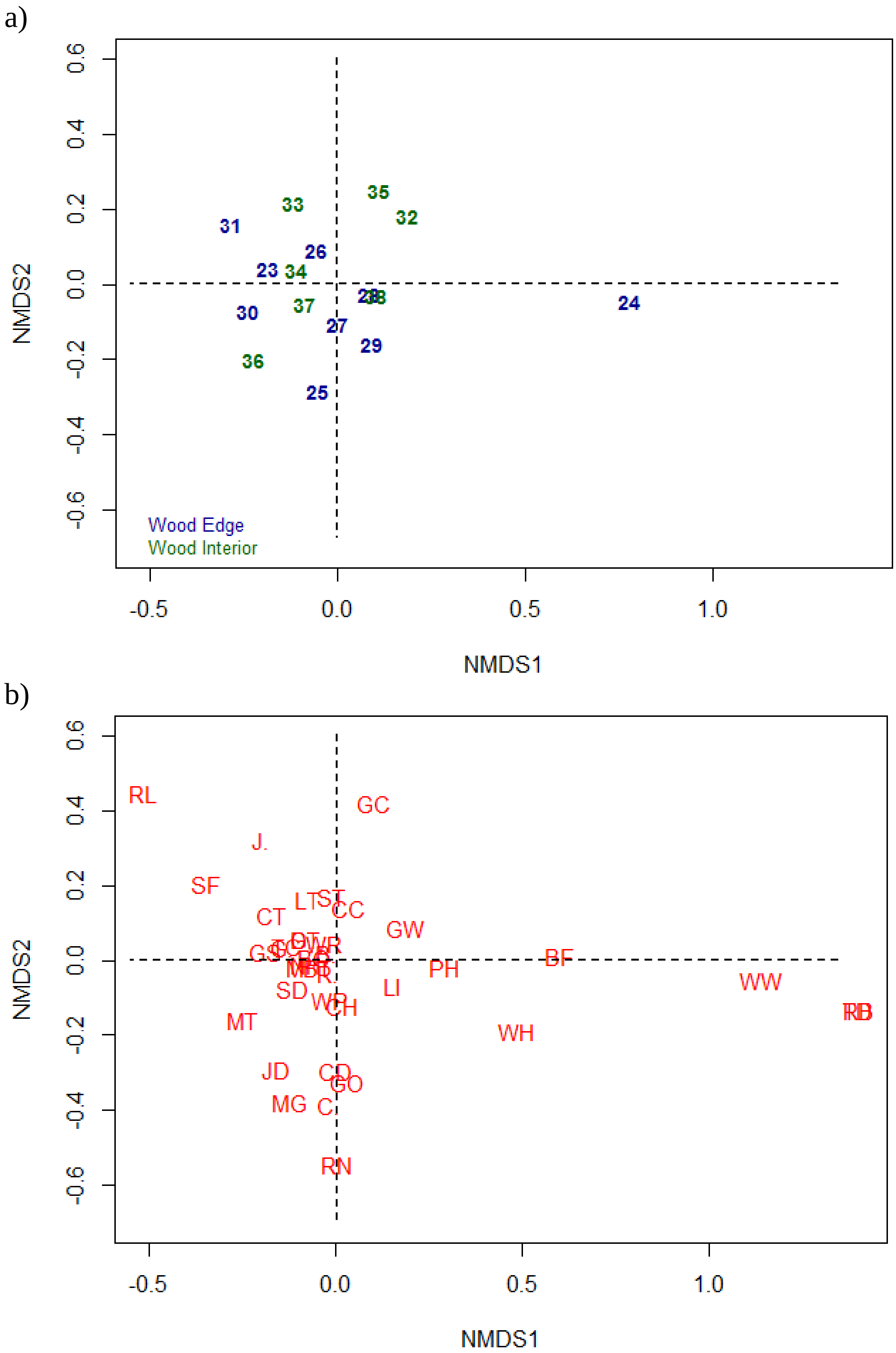


Figure 7.7. a) MDS of the woodland only transects in Cambridgeshire and b) the bird species responsible for the ordination. See Appendix A1 for bird species codes (R version 3.5.2).

7.4.6 Multidimensional Scaling at a Local Scale (Woodland) with Environmental Factors: Vegetation Composition and Structure

Table 7.5 and Figure 7.8 showed that *PCov_Ash*, *PCov_Maple* and *PCov_BroadOther* were significantly related (at $P < 0.05$) to the ordination of the woodland transects in opposite directions. *PCov_BroadOther* had the highest R^2 (0.54; $P = 0.029$) and was in the direction of 'Wood Edge' transect 24 and directly related to 'Wood Edge' 28 and 'Wood Interior' transect 38 (Figure 7.8), indicating the presence of more scrubby vegetation, such as silver birch. *PCov_Ash* ($R^2 = 0.51$; $P = 0.015$) was directly opposite *PCov_BroadOther*, indicating opposing relationships and that the two variables were mutually exclusive (Figure 7.8). *PCov_Ash* was directly related to 'Wood Interior' transect 34 and 'Wood Edge' transect 23 as a result of the largest gradient (33.9% and 70.3% ash, respectively). In an almost perpendicular position, *PCov_Maple* was significantly related ($R^2 = 0.37$; $P = 0.038$) to the woodland ordination increasing from 'Wood Interior' transects 37 to 36 (Figure 7.8), possibly as a result of the smaller woods having a higher percentage of maple than the other woodland transects.

The non-significance of the remainder of the vegetation composition variables in Table 7.5, indicates either similar percent cover in each of the woodland transects or no influence on bird community composition.

Figure 7.9 displays the vegetation structural variables that were significantly related to the woodland transect ordination from Table 7.6, and indicates a horizontal gradient of woodlands with a higher percent of lower stature vegetation on the right and woodlands with taller vegetation on the left of the MDS. *PCov_>5m* had the highest R^2 (0.77; $P = 0.002$) and was directly related to 'Wood Edge' transect 30 indicating that this transect was positioned first on the MDS (Table 7.6 and Figure 7.9). *PCov_2-5m*, *Ht_VDR* and *PCov_<0.5m* were directly opposite *PCov_>5m* and *Ht_Av*, pointing in the direction of 'Wood Edge' transect 24, which indicates the dissimilarities in woodland structure and the gradient of lower stature vegetation. *Ostorey_Ht* was almost directly opposite, further supporting the gradient and was directly associated with 'Wood Interior' transect 34 ($R^2 = 0.57$; $P = 0.007$), and possibly related to 'Wood Edge' transect 31, indicating a higher overstorey height (Figure 7.9). The significant relationship with *PCov_0.5-2m* ($R^2 = 0.53$; $P = 0.018$)

indicates a higher percent of vegetation at this level in ‘Wood Interior’ transects 32 and 35 creating the dissimilarity in bird community composition.

The non-significance of *Ostorey_PenDepth* and *Ht_StDev* indicates that the woodland transects were similar in canopy openness and in variation of tree height, or these variables did not influence bird community (Table 7.6). None of the extra wood variables (e.g. *Wood.Area* etc.) were significantly related to the MDS, suggesting that the size of woodland and percent of surrounding woodland were irrelevant to the bird communities as assessed here (Table 7.6).

*Table 7.5. The envfit output for the vegetation composition metrics for the woodland transects in Cambridgeshire. The stars represent the level of significance: ‘***’ <0.001, ‘**’ <0.01, ‘*’ <0.05, ‘.’ <0.1. This was carried out with 999 permutations.*

Variable	NMDS1	NMDS2	R^2	P	Significance
PCov_Ash	-0.979	0.202	0.505	0.015	*
PCov_Oak	-1.000	-0.031	0.107	0.486	
PCov_Maple	-0.651	-0.759	0.365	0.038	*
PCov_Elm	-0.229	-0.974	0.141	0.326	
PCov_BroadOther	0.980	-0.201	0.543	0.029	*
PCov_Cons	0.155	0.988	0.212	0.150	
Pres_Thorns	0.000	0.000	0.000	1	
Pres_Bramble	0.904	-0.427	0.042	0.746	
Pres_Shrub	0.000	0.000	0.000	1	

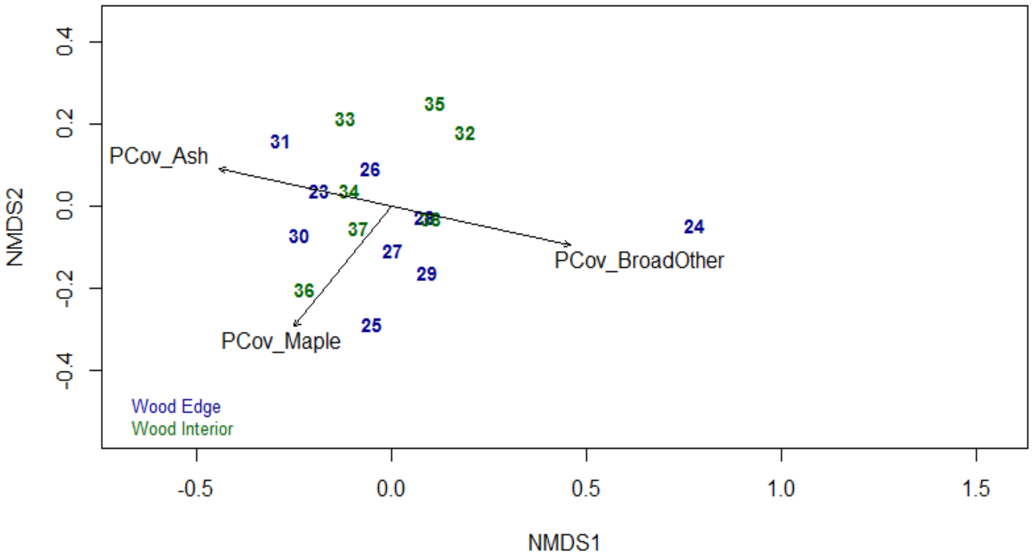


Figure 7.8 MDS of the woodland transects with the envfit function displaying the significant (at P<0.05) vegetation composition metrics as environmental factors to explain the ordination. See Table 7.5 for the P values (R version 3.5.1).

*Table 7.6. The envfit output for the vegetation structure metrics for the woodland transects in Cambridgeshire. The stars represent the level of significance: '***' <0.001, '**' <0.01, '*' <0.05, '.' <0.1. This was carried out with 999 permutations.*

Variable	NMDS1	NMDS2	R^2	P	Significance
Ht_Av	-1.000	0.001	0.655	0.002	**
Ht_StDev	-0.744	0.668	0.342	0.059	.
Ht_VDR	0.990	0.139	0.679	0.001	***
Ostorey_Ht	-0.941	0.338	0.571	0.007	**
Ostorey_PenDepth	-0.071	0.997	0.050	0.677	
PCov_<0.5m	0.990	0.138	0.546	0.008	**
PCov_0.5-2m	0.630	0.776	0.526	0.018	*
PCov_2-5m	0.996	-0.091	0.705	0.012	*
PCov_>5m	-0.964	-0.265	0.774	0.002	**
Wood.Area	0.296	0.955	0.221	0.186	
Wood.Perimeter	0.363	0.932	0.157	0.350	
Wood_P:A	-0.338	-0.941	0.183	0.285	
P_Woods_500m	-0.672	-0.740	0.034	0.780	
P_Woods_1km	-0.163	0.987	0.045	0.721	
Dist_Wood	-0.592	-0.806	0.104	0.478	

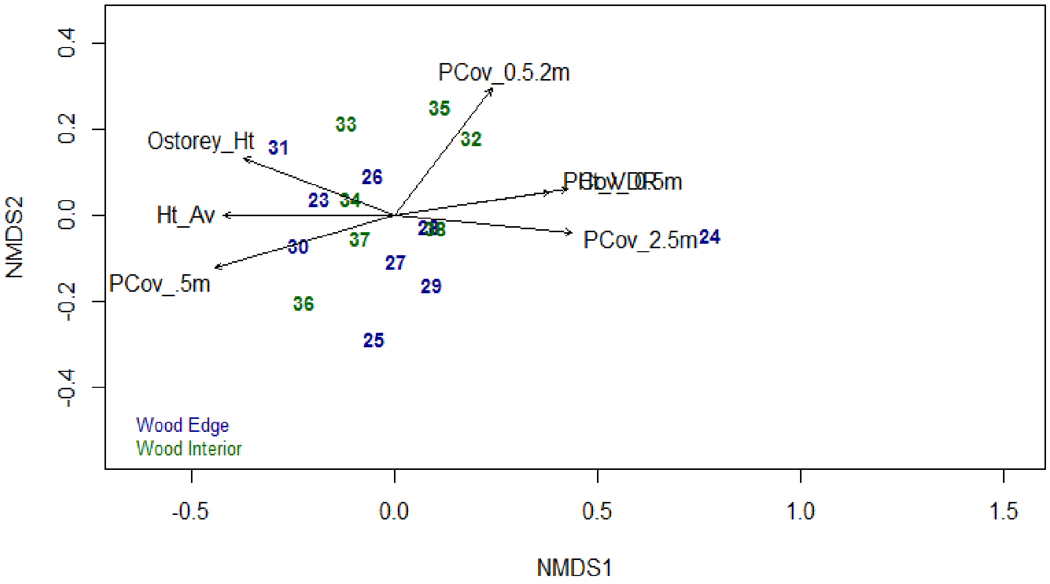


Figure 7.9. MDS of the woodland transects with the envfit function displaying the significant (P<0.05) vegetation structural metrics as environmental factors to explain the ordination. See Table 7.6 for the P values (R version 3.5.1).

7.5 Discussion

Multidimensional scaling of the Cambridgeshire survey transects exhibited differences both between and within the habitat classes in this agricultural landscape. As would be expected, a woodland–non-woodland horizontal gradient separated the clustered woodland transects from the more dissimilar field transects in terms of bird community composition. This was supported by the relationships with the structural variables in Figure 7.3; with variables depicting taller vegetation (such as average height) significantly related in the direction of the woodland transects, and lower stature vegetation variables (such as the percent cover < 0.5 m) significantly related to the field transects. Although these two habitats appeared not to display any further obvious gradients on the ordination, once separated the field and woodland analysis provided more dissimilarities in bird community composition.

The majority of the ‘Field By_Wood’ transects were at a greater distance from each other and from the other field transects on both the landscape and field MDS, indicating greater dissimilarity in bird community composition. The inclusion of the environmental factors to the landscape MDS demonstrated relationships of the field transects with the crop vegetation variables and lower stature vegetation. The separated field MDS particularly highlights that the percent cover of improved grass appears to be driving the bird community in the ‘Field By_Wood’ transects. The bird species ordination shows that the main difference in bird community in these transects was as a result of the presence of corvids, namely the Rook, Jackdaw, Raven, and possibly the Jay.

The other field transects, classified by the proportion of hedge length to transect length, were intermingled in a large cluster on the left of the MDS, with some more dissimilar than others, and unsurprisingly, related to the proportion of hedge length. It should be noted that only one structural and one vegetation composition variable were significantly related to the ordination of the field transects, possibly indicating a weak relationship when analysed alone or there are other factors affecting the bird community in the fields.

The woodland transects were tightly clustered on the opposite side of the landscape MDS, with the exception of ‘Wood Edge’ transect 24 as a result of the scrubby vegetation providing habitat for scrub preferring bird species, such as the Turtle

Dove, Willow Warbler and Bullfinch. The MDS for the woodland transects alone, on the other hand, showed opposing associations with different vegetation composition and consequently structural variables, creating variation in bird community composition. Furthermore, the woodland MDS also demonstrates that variation even occurs in transects within the same woodland (Figure 7.7). Furthermore, the non-significance of the connectivity variables (such as wood area, or distance to nearest wood) supports the notion in Section 5.4.7 that the woodland communities in Cambridgeshire were sufficiently interconnected for a metapopulation to persist.

7.5.1 The Exclusion Zone in Fields Adjacent to Woodlands

The majority of the ‘Field By_Wood’ transects, with the exception of transect 19, were positioned above the rest of the field transects on the landscape MDS (Figure 7.1). ‘Field By_Wood’ transects 20, 22 and 19 were also located near the vertical woodland boundary line on the MDS, suggesting the presence of woodland bird species encroaching from the adjacent woodlands. However, once separated the field only MDS showed that the woodland bird species were more associated with the ‘Field_high%_Hedge’ transects, thus the ‘Field By_Wood’ transects contained different bird assemblages. The relatively close positioning of ‘Field By_Wood’ transects 18 and 21, with ‘Field_low%_Hedge’ transects 3 and 4, on both the landscape and field MDS, may suggest that the lack of hedge in these habitats creates some similarities in bird community. However, all the ‘Field By_Wood’ transects had a low percentage of hedge length indicating that another reason is responsible for the dissimilarities. It is worth noting that similarly occurring bird species across the field transect classes were positioned close to the vertical boundary line on the MDS.

The percent cover of improved grass was directly associated with ‘Field By_Wood’ transects 20 and 22 on the landscape MDS as the largest gradient in this variable, and on the field MDS it was directly between transects 21 and 22 as a result of the high percent cover of improved grass in these transects (93.1% and 73.6% respectively). The main influence on the positioning of ‘Field By_Wood’ transect 22 was the high density of Rook (6.8 birds/ha) as indicated on both the landscape and field MDS; in fact, ‘Field By_Wood’ transect 22 was the only transect to contain the Rook. Furthermore, the field analysis also shows that the Jackdaw is also associated to ‘Field By_Wood’ transect 22, the Jay to transect 20 and the Raven to transect 21. ‘Field By_Wood’ transect 19 was positioned separately from the others on the landscape MDS and associated with the high density of the Carrion Crow, however, this was not apparent in the field MDS.

The presence of these corvid species suggests that they have a preference for the fields adjacent to woodlands (Andrén 1995, Fuller et al. 2005b) and are most likely utilising the improved grass for feeding (Barnett et al. 2004). On the other hand, the high abundance of corvids in the ‘Field By_Wood’ transects could also result from greater detectability than in the woodlands. Nonetheless, the presence of these corvid predators has resulted in the absence of woodland birds in fields adjacent to woodlands through predator avoidance, as suggested by Vickery and Arlettaz (2012), and a similar argument almost certainly applies to open ground nesting birds (Suhonen et al. 1994, Donald et al. 2001b). This more than likely created the ‘exclusion zone’ as indicated by the poor values of the bird indices in Section 5.4.5. Furthermore, improved grass is also deficient in insects and weed seeds, depending on the level of intensive management, reducing food availability for birds (Estrada et al. 1993, Wilson et al. 1999), therefore sympathetic changes in farming will help to conserve species (Gregory and Baillie 1998).

The current study mostly supports the findings of Andren (1992) that in their study site in Sweden, the Magpie and Hooded Crow (*Corvus cornix*) were associated with agricultural areas, and the Jay and Raven were associated with larger woodlands, as shown in the landscape MDS (Figure 7.1). The presence of Jay in ‘Field_low%_Hedge’ transect 3 was the exception, and was most likely utilising the small copse in this transect. However, Stevens et al. (2008) stated that since the reduction in persecution, the Jay has become more generalist in habitat preference and is more abundant in fragmented woodlands.

The location of the Jackdaw on the woodland side of the landscape MDS was as a result of ‘Wood Interior’ transects 36 and 37 having the highest densities. However, Andren (1992) noted that the Jackdaw was predominantly agricultural and mainly preyed on nests in agricultural land, whereas the current study suggests that the Jackdaw possibly utilises both fields and smaller woodlands in fragmented agricultural landscapes for both nesting and feeding (Andren 1992). Dunning et al. (1992) referred to this as landscape complementation, where the close proximity of the small woodlands to the fields complements the resources available to the Jackdaw in each. Moreover, the high densities in these smaller woodlands, as well as the edges of larger woodlands, may indicate an edge-preference by Jackdaws.

The similarities in some bird species contrasted with differences in others, for example, the Green Woodpecker was present in both ‘Field By_Wood’ transects 20 and 22 and was therefore positioned between them on the field MDS (Figure 7.4), near the location of ‘Field By_Wood’ transect 19, despite not being recorded in this transect. This consequently causes difficulties in interpreting the MDS and may suggest that the dissimilarities in bird community composition were greater and more complex than can be represented in 2D space (Sturrock and Rocha 2000). However, increasing the number of dimensions leads to problems in both presenting and understanding the data (Borgatti 1997). Borgatti (1997) stated that there are discrepancies over the tolerable stress of the MDS output, but less than 0.1 is excellent, whereas more than 0.15 is unacceptable. However, this is subjective as Kruskal (1964) and Wickelmaier (2008) found that stress under 0.2 is acceptable and over 0.2 is poor. The stress in the current study was 0.157 for the field transect MDS and was regarded as acceptable, but may be causing issues.

The Green Woodpecker is characteristically a woodland species that forages for ants on cultivated ground in fields and also clearings in conifer plantations (Rolstad et al. 2000). Fuller et al. (2005a) further classified the Green Woodpecker as a grassland species associated with hedgerows and copses in pastoral grassland. Nilsson et al. (1992) noted that declines of the Green Woodpecker population in Sweden were due to changes in agricultural practises. Dorresteijn et al. (2013) showed in their study site, that Green Woodpeckers are more commonly found in ‘wood pasture’ than in forests, however, this habitat is rare in Europe, suggesting alternative habitat use (Fuller and Robles 2018). The Green Woodpecker could have benefitted from agricultural intensification originally, with the increase in fragmented woodlands next to cultivated areas (Rolstad et al. 2000, Dorresteijn et al. 2013). Additionally, the Green Woodpecker may not have been encountered in suitable habitat in the current study as a result of the low population density.

7.5.2 Field Community Composition

The field transects appeared to cluster randomly on the landscape MDS with little distinction based on the classification. However, analysis with the environmental variables in Figure 7.2 showed that the field transects were separated on the MDS by the vegetation composition variables representing crop type, indicating an influence on bird community composition, in line with Siriwardena et al. (2000) and Vickery and Arlettaz (2012). Vickery and Arlettaz (2012) found that a heterogeneous landscape in terms of both cropped and semi-natural habitat, and also variation in crop types and farming practises, would enhance breeding success and foraging ability in farmland birds. For example, Reed Buntings are traditionally associated with wet areas, therefore changes in agricultural practises must be adopted, such as increasing wet areas, providing weed rich winter stubble and conservation headlands (Peach et al. 1999). However, Gruar et al. (2006) showed that Reed Buntings utilise oil seed rape in place of wet areas, rape supported more weed seeds and invertebrate food in the breeding season than cereals or set-aside. Furthermore, Whittingham et al. (2009) found that all ten species in their study were strongly associated with oil seed rape, particularly Reed Buntings, Dunnocks and Yellowhammers. This relationship was not completely apparent in the current study as the highest density of Reed Buntings was in ‘Field_med%_Hedge’ transect 10 which was not directly associated with the other crops variable, but did contain oil seed rape. Furthermore, Reed Buntings were present in almost all of the field transects, further supporting Gruar et al. (2006) by utilising dry farmland in Cambridgeshire. The Linnet was positioned on the horizontal boundary line on the landscape MDS in the same position as Reed Bunting (Figure 7.1b), indicating that both bird species were present in the majority of the field transects, and could also be exploiting the invertebrate rich oil seed rape (Whittingham et al. 2009). Eybert et al. (1995) showed that the Linnet avoided cereal and potato crops, but selected rape, meadows and fallow, and, in their study area, also avoided hedgerows, due to predator avoidance, preferring to nest in scrub. As the transects included multiple bird species and the other crops variable incorporated both beans and oil seed rape, this may be masking any single species bird-habitat relationships in the current study.

The percent cover of other crops was directly associated with ‘Field_low%_Hedge’ transects 3 and 4, indicating the largest gradient in this variable (Figure 7.2). However, the low R^2 value (0.29) and the fact that the highest percent of other crops was actually in ‘Field By_Wood’ transect 20, suggests that the closeness of the woods may be more influential on the bird community than the presence of other crops. The percent cover of vegetation at the 0.5-2 m layer was also closely related to the same area on the MDS (Figure 7.3), representing the gradient from 'Field_high%_Hedge' transect 13 to 17, most likely as hedge vegetation.

The structural variables overstorey penetration depth and the proportion of hedge length were in a similar position on the landscape MDS (Figure 7.3), as a result of the hedge vegetation in the open habitat containing trees which allowed greater laser penetration. Unsurprisingly, these metrics were also significantly related to 'Field_high%_Hedge' transects 14, 15, 12 and 'Field_med%_Hedge' transect 7, indicating a relationship with hedge specialist bird species on the MDS, such as the Whitethroat and Yellowhammer. Furthermore, the position of the declining Cuckoo near 'Field_high%_Hedge' transect 12 supports the high species decline index in this transect (Section 5.5.2) as a result of the high proportion of hedge, overstorey penetration depth and possibly the percentage of cereal. The Goldfinch was positioned on the field transect side of the landscape MDS as it was present in a number of field transects, and the highest density of Goldfinch was in 'Field_high%_Hedge' transect 15. This indicates that fields are the preferred habitat of the Goldfinch in the Cambridgeshire landscape, and that the species was particularly associated with hedges.

'Field_high%_Hedge' transect 16 was positioned just off the centre of the landscape MDS (Figure 7.1) on the field side as a result of the highest proportion of hedge length (151.5%). This increased the density of woodland birds positioning it more towards the woodlands but on the field side, indicating that the bird community in this transect was predominantly farmland birds. The similar, almost central position of the Dunnock on the MDS would suggest a higher density in 'Field_high%_Hedge' transect 16, however, the highest density was in 'Wood Edge' transect 30 and the central positioning was as a result of the species occurring in almost every transect in the Cambridgeshire survey area.

Once separated the field MDS showed that transects with varying amounts of hedge were separated somewhat by the proportion of hedge length with some overlap, indicating an effect on bird community composition. This is supported by the significant relationship of the proportion of hedge length to the ordination as the only significant vegetation structural variable (Figure 7.6). The horizontal boundary line separates the transects containing woodland bird species on the top of the MDS, from the transects with fewer woodland bird species and more scrubby, open species below. The majority of the woodland specific bird species, such as the Garden Warbler and Great Spotted Woodpecker, were located with the 'Field_high%_Hedge' transects on the MDS (Figure 7.4b), presumably due to a higher percentage of hedge or woody vegetation. The species that prefer more open habitat, such as the Skylark and Yellow Wagtail, were located in the bottom left of the MDS near the field transects with a lower the proportion of hedge length, most likely avoiding high boundaries (Donald et al. 2001b). Skylarks have a preference for vegetation at 0.55 m for breeding habitat (Donald et al. 2001b), and are also affected by crop type, particularly autumn sown crops, supporting the position of the percent cover of vegetation at 0.5-2 m and other crops on the landscape MDS (Bradbury et al. 2005). However, for foraging, both the Skylark, and the Yellowhammer, prefer shorter patches of vegetation, possibly resulting in the position of the Yellowhammer near the < 0.5 m structural variable in the current study (Odderskær et al. 1997, Douglas et al. 2009).

The positioning of the Yellow Wagtail on the landscape (and field MDS) supports the presence of this rare species in transect 4 (Section 5.5.3). Yellow Wagtail territories are associated with fields which flood in the winter providing wet areas of short grass and bare ground in the summer, whereas nesting sites are associated with longer areas of grass for nest cover (Bradbury and Bradter 2004), possibly contributing to the relationship with vegetation at 0.5-2 m. Bradbury and Bradter (2004) suggested that increasing wet areas in fields will increase both nesting and feeding sites for the Yellow Wagtail. However, they also stated that increasing wet areas may be detrimental to waders by reducing soil invertebrates, and suggested that a variety of grass heights be maintained and grazing be kept at low densities in the breeding season, to avoid the trampling of nests. Conversely, Gilroy et al. (2008) found that available bare ground, and also crop height, field boundary habitats and soil organic content, had little effect on Yellow Wagtail territory abundance, but was significantly influenced by soil penetrability and crop type, possibly contributing to the relationship with the percentage of other crop. Soil penetrability, as well as organic content, affected the abundance of aerial insects, suggesting that the Yellow Wagtail selected breeding sites based on soil penetrability to maximise nestling prey abundance. They implied that agricultural methods and soil degradation have contributed to soil compaction, thus, combining the findings of both studies would suggest that increasing wet areas in fields may combat both soil degradation and penetrability, and therefore negate the decline of the Yellow Wagtail.

Vickery and Arlettaz (2012) also stated that the Yellow Wagtail forages in field margins due to reduced prey in the crop as a consequence of pesticide use. Prey density is often higher in taller crops, however, they are less accessible and predator detection is lower (Odderskær et al. 1997), indicating that heterogeneity in crop structure, as well as in grass margins, is vital for farmland birds (Vickery and Arlettaz 2012), following Optimal Foraging Theory (Charnov 1976). Unfortunately, due to the LiDAR being collected in a previous year to the bird data in the current study, the crop structure and type will differ, and therefore is not directly interpretable. However, this result suggests that there are still consistencies in vegetation characteristics of the field margins, due to similar management for example, suggesting that for species such as the Yellowhammer it is the semi-natural vegetation that is more important.

Surprisingly, the field transect ordination only had two significant relationships with the environmental factors; the proportion of hedge length and percent cover of improved grass. The non-significance of the other vegetation variables suggests similar percentages of vegetation species and structure across the field transects, or that the differences in these variables were insignificant when analysed alone. This therefore, suggests that other variables were responsible for the dissimilarities in bird community composition displayed on the MDS or were due to chance. However, the significance of the proportion of hedge length does emphasize the value of hedge habitat in an agricultural setting.

7.5.3 Woodland Scrub Community

‘Wood Edge’ transect 24 was positioned a greater distance from the other woodland transects on both the landscape and the woodland only MDS. The bird species such as the Turtle Dove and Willow Warbler, and environmental variables, such as the percent cover of other broadleaved species and vegetation at 2-5m, signify lower, scrubby, broadleaved vegetation, such as silver birch, in this habitat, most likely increasing the number of available niches (Seoane et al. 2017). This supports the findings by Miller et al. (2003) who showed that understorey vegetation and herbaceous ground cover were strong determinants of bird community. The percent cover of other broadleaved vegetation was not directly associated with ‘Wood Edge’ transect 24 on either MDS; suggesting that the bird community in other transects is also influenced by this variable. Height VDR and the percent cover of vegetation < 0.5 m was also significantly associated in the direction of ‘Wood Edge’ transect 24 (Figure 7.9). This indicates that ‘Wood Edge’ transect 24 had more ground vegetation and fewer trees present, suggesting it was further from the woodland edge, which created a higher ratio (the tallest of the woodland transects), resulting in greater dissimilarity to the other woodland habitats.

The three species, Turtle Dove, Willow Warbler and Bullfinch, associated with this transect as a result of the scrubby vegetation, are all in decline; the Turtle Dove being globally threatened. Moreover, ‘Wood Edge’ transect 24 was the only transect to contain the Turtle Dove in the Cambridgeshire landscape and was the only woodland plot to contain Reed Buntings, resulting in a distinctly dissimilar bird community. However, the bird indices in Section 5 do not place much importance on this habitat, presumably because the index values are dominated by numerous species with less of a decline. Turtle Dove territories favour established scrub and hedgerows over 4 m tall, with positive effects of standing water, bare ground and fallow, and were negatively impacted by grazing (Dunn and Morris 2012). The Turtle Dove is solely granivorous and has switched from wild weed seeds to spilt cereal grain, animal feed and stored grain, possibly leading to a lower reproductive output exacerbating the decline (Browne and Aebischer 2003, Dunn and Morris 2012). Dunn and Morris (2012) concluded that maintaining scrub patches with seed-rich habitat for foraging close by is required to retain Turtle Dove territories. Degrading breeding habitat with the reduction in early successional habitat in England and the removal and reduced quality of hedges and woodland understorey vegetation has mostly likely led to the declines in the Willow Warbler and Bullfinch (Siriwardena et al. 2001, Proffitt et al. 2004, Morrison et al. 2013).

7.5.4 Woodland Community Composition

The woodland transects were tightly clustered on the landscape MDS, however, small distances between them indicated some dissimilarity as habitats with exactly the same bird species assemblage would have a distance of zero. Similarly, the woodland bird species on the landscape MDS in Figure 7.1b indicate differences in bird community as they were arranged in three clusters with a number of single bird species positioned separately on the woodland side of the MDS. The main cluster contained a number of predominantly woodland bird species (illegible on the plot), presumably common bird species associated with the majority of the woodlands. The cluster towards the vertical boundary line on the MDS contains the Wood Pigeon, Raven, Spotted Flycatcher, Chaffinch and Dunnock, presumably as a result of these species also being present in some of the field transects. The third cluster contains the Mistle Thrush, Green Woodpecker and Jay, indicating that the bird communities in the woodland transects may differ in these less common species, as a result of lower densities. However, the separated woodland MDS in Figure 7.7 does not show the same clustering of species, but demonstrates dissimilarities in bird community composition between and within the two woodland classes. It would be expected that following Melin et al. (2018) the smaller woodlands should be intermediary between the wood edge and wood interior. Whilst this might be the case for bird diversity and abundance (Section 0), the current study shows that the bird community does not follow this pattern.

The environmental variables on the landscape MDS also point in slightly different directions (Figure 7.2 and Figure 7.3), suggesting that bird assemblages may differ between ash and oak habitats and also vary with vegetation height. This is also displayed on the woodland only MDS in Figure 7.9 as the vegetation structural variables, such as average height, overstorey height and percent cover above 5 m, separate the woodland transects with taller vegetation from the transects with higher percent of lower stature vegetation (e.g. the percent cover vegetation < 0.5 m and 2-5 m layers, and a higher height VDR). Additionally, the vegetation composition variables also showed an almost horizontal gradient of higher percent cover of ash on the upper left and a higher percent cover of other broadleaved species on the lower right of the MDS in Figure 7.8, further indicating lower stature often succesional vegetation, such as silver birch. The almost perpendicular relationship of the percent cover of field maple indicates an increasing gradient from ‘Wood Interior’ transect 37 to 36 (18.4 cf. 23.1%). Moreover, these transects were the smallest woodlands surveyed in the Cambridgeshire landscape and had higher percentages of field maple or elm and lower percentages of ash and oak than the transects in the larger woodlands, suggesting that variation in vegetation composition could have created the dissimilarities in bird community in line with Adams and Matthews (2019). Furthermore, ‘Wood Interior’ transect 38 (Lady’s Wood) was in almost the same position as ‘Wood Edge’ transect 28 (Wennington

Wood) on the MDS, indicating virtually identical bird species composition, supporting the findings in Section 5.4.7. However, dissimlarities in bird species composition still exist, hence the transects did not have a zero distance. The Goldcrest was associated with 'Wood Interior' transect 35 which had the highest percentage of conifer (Figure 7.7b). However, there was no significant relationship with the percent cover of conifer on the MDS, presumably as a result of only two wood transects containing conifer.

Unexpectedly, a number of transects located in the same woodlands differed in terms of bird community composition, suggesting that variations in vegetation structure and/or composition affect bird community. However, the results may also be stochatic, or possibly as a result of a population density effect. For example, 'Wood Interior' transects 33 and 32 were both in Monks Wood, but were in different positions on the MDS. 'Wood Interior' transect 32 was related with the percentage of vegetation at 0.5-2 m on the MDS indicating a higher percentage of shrubby vegetation, and in fact also had a higher percent cover of vegetation less than 0.5 m than transect 33, resulting in more scrubby bird species, such as the Bullfinch and Willow Warbler. Moreover, 'Wood Interior' transects 32 and 33 showed similar diversity metrics in Section 5.4.5 (rarity and IRR being the exception), demonstrating that apparently similar habitats in the same woodland can have similar bird diversity and species richness, but differ in bird community composition. Furthermore, 'Wood Interior' transect 33 had a higher percent cover of vegetation taller than 5 m adding to the dissimilarity between 'Wood Interior' transects 33 and 32.

'Wood Edge' transects 23 and 31 were both in Monks Wood but the different values of overstorey height and percent cover of ground, created a more scrubby habitat in transect 31, which possibly increased the density of Spotted Flycatchers, resulting in the dissimilarities in bird community. It is worth noting that the Spotted Flycatcher is also positioned close to the Jay on the woodland MDS. The Jay is the main predator of Spotted Flycatcher nests in southern England, and is also found in greater abundance in fragmented woodlands over agricultural land, increasing predation pressure (Andren 1992, Stevens et al. 2008).

The result of the current study demonstrates that similar habitats with similar measures of bird diversity and species richness, even in the same location or woodland, can be dissimilar in bird community composition. This therefore supports the argument that diversity measures should not be taken alone when deciding management practises, as the identity of the species present is also important.

The non-significance of the extra woodland variables (based on patch area and isolation) suggests that the woodlands in the Cambridgeshire landscape were sufficiently interconnected to allow movement, possibly resulting in variations of similar bird communities (Opdam et al. 1985, Hanski et al. 1997, Lopes et al. 2016). Opdam et al. (1985) stated that smaller woodlands in a fragmented landscape were made up of random selections from the larger woodlands. This also suggests that any differences in bird community in the woodlands in Cambridgeshire were therefore expected to result from smaller scale woodland rather than landscape variables as shown in Sallabanks et al. (2006). However, both landscape and stand variables must be consulted with regards to conservation measures in order to prevent local extinction due to isolation or poor habitat (Hanski et al. 1997, Sallabanks et al. 2006). The non-significance of overstorey penetration depth in the wood only MDS would also suggest that the woodland transects had similar canopy openness overall, or this was insignificant to bird community composition. Section 5.4.9.1 showed a positive relationship of overstorey penetration to bird density, whereas the bird community composition was not significantly affected by canopy openness. This reiterates the importance of combining multiple measures of bird diversity, conservation priority and rarity as well as bird community composition for any conservation management strategies.

7.6 Conclusion

Unsurprisingly, the greatest dissimilarities on the landscape MDS were between the woodland and field habitats, however, differences in bird community were also shown to exist within each habitat class. The field MDS showed dissimilarities of the ‘Field By_Wood’ transects from the other field transects which were more similar, and any dissimilarities may not be easily interpretable due to the smaller distances between the transects on the MDS. Further analysis may be needed if the bird community in the fields is analysed alone, such as proximity of woodland or copse, land-use or urban areas, the climate or weather, and geographic location. The lack of significant variables also suggests that the landscape bird community analysis may be more reliable (and also has a lower stress value of 0.12), indicating that the entire landscape should be taken into account before any management prescription can be implemented in an agricultural area (i.e. Section 7 and 7.5.1). The ordination in the MDS may also be struggling to fit the relationships in the 2D space possibly leading to unreliable results (Borgatti 1997). Moreover, Borgatti (1997) showed that all MDS with non-zero stress will have some degree of distortion, (i.e. not be perfect) and the larger distances will be more accurate than the smaller distances.

The woodland MDS showed that whilst the smaller woodlands are intermediary between the wood edge and wood interior in terms of bird diversity and abundance, they did not follow this pattern in bird community. Moreover, woodland area did not significantly influence bird community composition, suggesting that the woodlands in the Cambridgeshire landscape are sufficiently connected. This community analysis shows that in this agricultural landscape transects of seemingly similar habitat with similar measures of bird diversity and species richness, even in the same location or woodland, can vary in bird community composition, and vice versa. The bird community composition of the habitats is thus affected by factors including vegetation composition and structure, which differ from the factors that influence the bird measures in Section 5.5. Diversity measures should, therefore not be taken alone when deciding management practises, as the identity of the species present is also important. Furthermore, this analysis also shows that individual bird species have alternative habitat preferences in different landscapes. This was most obvious with the Goldfinch which was associated more with farmland and particularly hedges in the Cambridgeshire landscape, whereas they were associated with conifer in the New Forest analysis (Section 6.4).

8 Effects of Landscape Change on Bird Indices and Communities

8.1 Abstract

The British landscape has changed dramatically over the last century mainly through agricultural intensification and increasing urbanisation, resulting in very little natural land left. Some natural land has been protected to maintain natural biodiversity and populations, however, this may also be managed and affected by anthropogenic disturbances, for example grazing of livestock and coppicing. The following study utilises the results from the previous chapters to predict effects of further change on the bird indices in the habitats of the two contrasting study landscapes, the New Forest and Cambridgeshire. The bird species affected by these changes were also interpreted from the MDS results and management recommendations presented as an accumulation of the results. In the New Forest, decreasing scrubby vegetation (at 2-5 m) was predicted to more than halve bird density. Reducing the percent cover of pine to 0% in the New Forest woodland plots would double species richness and the priority index, and greatly increase bird diversity and rarity, but by contrast decreasing the percent cover of beech would decrease the number of declining species supported. In Cambridgeshire it was the percent hedge length that supported more declining species over the landscape, and decreasing percent hedge length would halve bird density, species richness, diversity, priority, and the numbers of declining species and rare species in agricultural areas. Increasing the percent cover of improved grass from 0 to 100% in the Cambridgeshire transects was predicted to approximately halve species richness and diversity. A decrease of 40 to 0% cover of oak was predicted to reduce the rarity index by a third, whilst a reduction in average height of vegetation (i.e. trees) was predicted to decrease the Index of Relative Rarity, suggesting that woodlands with taller trees in Cambridgeshire support more rare bird species.

8.2 Introduction

Over the past century the landscape has changed dramatically as a result of anthropogenic behaviour, with advances in technology causing agricultural intensification, and with increasing urbanisation (Fuller et al. 1995). The European landscape has been managed by humans for approximately the last 10,000 years through the cultivation of crops, grazing of livestock for the provision of food and the felling of trees for timber or coppicing for poles (Batáry et al. 2015). Consequently, there is very little natural land left and much of the countryside in Europe is artificial with the open landscape being maintained by farm animals and agriculture, rather than indigenous grazers and natural disturbances (Batáry et al. 2015). Agricultural land covers approximately 38% of the Earth's terrestrial surface (FAO 2014 in O'Connell et al. 2015): 12% (1.53 billion ha) is cropland and 26% (3.38 billion ha) is pasture land (Foley et al. 2011). Around 90% of forest cover in Britain is plantation (Donald et al. 1997, Hartley 2002), and most European woodlands have been altered at some stage. In Britain, 43% of all woodland cover was coniferous in 2010 (Forestry Commission 2013).

Up until the Second World War, wildlife often benefited from agriculture (e.g. the Skylark; Bradbury et al. 2005), but since then rapid changes, particularly in agricultural practises and urbanisation, have significantly altered the landscape, ultimately affecting indigenous avifauna (Robinson and Sutherland 2002, Hayhow et al. 2015, 2017). Authorities have tried to implement measures to mediate the effects of landscape change, such as Agri-environment Schemes (AES) and protected areas, but often to no avail (see Section 2.2 for more information).

This study investigates scenarios of land-use change using the significant variables from the multi-model analysis for the two study areas, the New Forest and Cambridgeshire, predicting the outcome for bird diversity and composition. This could be used to inform the relevant authorities and land users on management strategies, and for example, to predict the effects of implementing the changes set out in the New Forest Management Plan (New Forest National Park Authority 2010, updated in the Partnership Plan: Natural England et al. 2015) or potential changes influenced by current affairs in agriculture on the Cambridgeshire landscape (e.g. the effects of "Brexit" on agricultural and other market factors).

8.3 Methods

The multi-model averages in Section 4.4.5 for the New Forest and Section 5.4.9 for Cambridgeshire were used to evaluate a land-use change by altering selected significant vegetation composition or structural variables, and examining the effects on the bird indices following Thomas et al. (2017). The most important variable derived from the multi-model results was used as this provided the most reliable result.

A ‘new data frame’ was defined by the user, which contained a continuous sequence of values for the selected independent variable to be plotted, along with the mean values of the other variables in the model set to 2 AIC, as this was deemed the best approximating model set (Thomas et al. 2017). Models greater than 2 AIC were weaker. The new data frame was close to the limits of the actual data to avoid uncertainty and spurious predictions. The predict function in the *stats* package in R (R Core Team 2018) was run using the model average and the new data frame to predict values of the chosen bird index with increasing values of the chosen habitat variable. The actual surveyed data for the chosen bird index was plotted against the selected independent variable. The prediction line was then plotted on top of the actual data to show any effect of changing the variable on the bird index, along with the standard error and the 95% confidence intervals.

In addition, the community level results from the Multidimensional Scaling (MDS) in Section 6 for the New Forest and Section 7 for Cambridgeshire, or directly from the data if this was not possible, were used to speculate on the likely species and community assemblage changes associated with predicted shifts in the bird indices. Management recommendations were then interpreted from the prediction results for each landscape.

8.3.1 New Forest Scenarios

The New Forest Management Plan (New Forest National Park Authority 2010) stated that it is vital to remove vegetation that will cause scrubbing up in the heathland. Scrub removal was simulated by decreasing scrubby/shrubby vegetation (*PCov_2-5m*) and varying the vertical profile of the vegetation (*Ht_VDR*) in the survey plots, and predicting the effect on *Bird_Density*. These variables were chosen to represent this scenario as they displayed a significant relationship in the multi-model analysis in Table 4.2a and Table S 4.14a.

The New Forest Management Plan (New Forest National Park Authority 2010) also stated that they intended to increase the proportion of broadleaved trees in the New Forest. The pine removal scenario explores the effects on the bird indices of a change in the percent coverage of pine in individual plots in the New Forest. Pine removal or reduction in the survey plots, would result from felling the pine and allowing the plots to re-seed as broadleaved woodland, or alternatively, replanting plots with a high percentage of pine would have the opposite effect on the bird indices.

Beech is declining across the UK, and the results of Section 4.5.2 showed that more bird species with declining populations in England were supported in the survey plots with high *PCov_Beech* in the New Forest (Figure 4.4). Therefore, the beech decline scenario simulates the effect of change in the percent cover of beech (*PCov_Beech*) on the number of declining bird species (*Spp_Decline*).

8.3.2 Cambridgeshire Scenarios

As a result of agricultural intensification particularly in the second half of the 20th Century, hedgerows were removed to give rise to larger fields for the manoeuvrability of larger machinery; this (amongst other factors) resulted in the national decline of a number of farmland bird populations. The scenario of hedge decline predicts the effect on the bird indices that were significantly related to the percent of hedge length (*P_HedgeLen*).

Agricultural intensification also resulted in changes to more intensively managed lowland grassland, which is depleted of insect prey and regularly cut. In the current economic climate there is uncertainty as to how the UK’s agricultural policies will change after leaving the EU. Increases in diversification of farming as a result of reduced subsidies may increase recreational practises, such as horse grazing and pony trekking which would increase the percent cover of improved grassland. On the other hand, a reduction in cattle grazing due to export limitations or climate change concerns may result in a reduction of improved grass in favour of more arable production. In the landscape level analysis in Section 5.4.9, *PCov_ImpGrass* was significantly related to both *Spp_Richness* and *Spp_Diversity* (Table 5.2 and Table 5.5), and was therefore used to predict the effect of changes in this vegetation on bird species richness and diversity in the landscape.

Oak decline (as with beech decline) has also been reported nationally, which is probably having negative effects on UK bird populations. *Spp_Rarity* was shown to be significantly related to *PCov_Oak* (Table 5.5), suggesting that rare birds are located and supported in the transects with a high percent cover of oak (i.e. the woodland transects). Change in oak was used to predict an effect on bird species rarity.

Ht_Av is significantly related to *Spp_IRR* (Index of Relative Rarity) in the landscape analysis in Table 5.5 and Table S 5.19. This indicates that there are rare birds (with smaller national population sizes) located in the transects in Cambridgeshire with high average height of vegetation (i.e. more trees). Tree loss through selective felling, or natural gaps created by tree death, such as from oak decline or ash dieback, would reduce the average height over the transect and influence *Spp_IRR* in that transect. The scenario of tree loss (and therefore reduced average height) was used to predict the effect on IRR.

8.4 Results

8.4.1 Scrub Removal – New Forest

It can be predicted from Figure 8.1a that reducing *PCov_2-5m* (representing scrubby vegetation) from 10% to 0% within a plot (or compartment) would more than halve *Bird_Density* (28 to ~12 birds/ha). However, Figure 8.1b predicts that *Bird_Density* will decrease with an increase in *Ht_VDR*, which was most likely influenced by the high *Ht_VDR* in the 'Heathland' and 'Scrubland' habitats which had low *Bird_Density* (see Figure 4.4 in Section 4.4.3). The range of values of *Bird_Density* with medium *Ht_VDR,* in Figure 8.1b, indicates that the relationship is more complex, and that it is not just scrub, per se, that will increase *Bird_Density,* but scrub development as part of a varied vegetation profile. A medium *Ht_VDR* is caused by a smaller median vegetation height but a high maximum height in a plot, i.e. from increased *PCov_2-5m* with an overstorey, which would result in more birds, and most likely more bird species. It should be noted that the other bird indices with relationships to *Ht_VDR* and *PCov_2-5m*, such as *Spp_Richness*, were not strong enough to be represented or make reliable predictions from (Table 4.2a and Table S 4.14a).

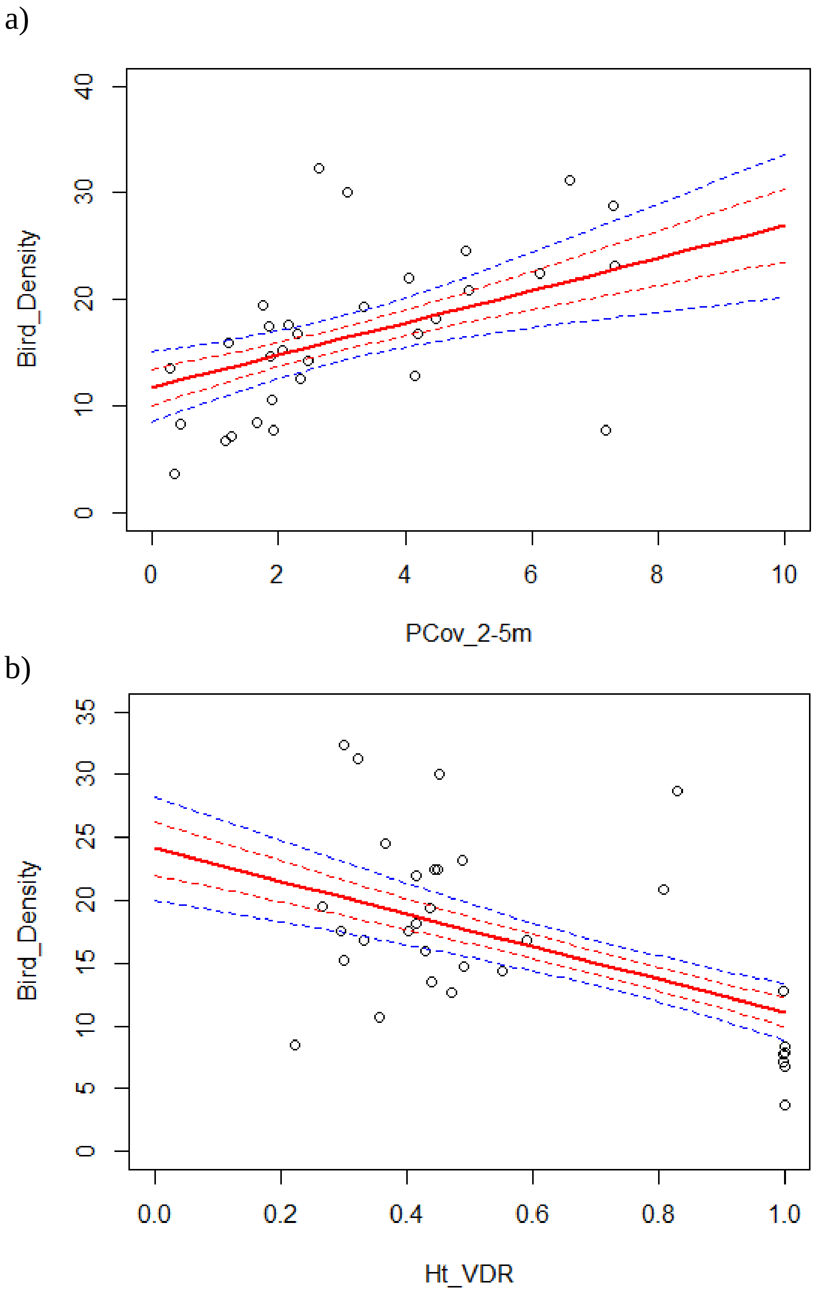


Figure 8.1. The prediction of Bird_Density with increasing a) PCov_2-5m and b) Ht_VDR, with the mean of the other variables in the model remaining constant, represented by the solid red line. The open circles represent the actual data points; red dashed lines are the standard error and the blue dashed lines are 95% confidence intervals (R version 3.5.2).

8.4.2 Pine Removal – New Forest

It can be predicted from Figure 8.2 that decreasing *PCov_Pine* would significantly increase *Spp_Richness*, *Spp_Diversity*, *Spp_Priority* and *Spp_Rarity* at the plot level. In fact, increasing *PCov_Pine* to 100% in a plot would almost halve

Spp_Richness (from 21 to 11 species), and would reduce *Spp_Diversity,* from 2.68 diversity at 0% pine to 1.86 diversity at 100% pine (Figure 8.2b). An increase in *PCov_Pine* would likely reduce the numbers of broadleaved woodland preferring bird species, such as the Redstart, at the plot (or compartment) level. Increasing *PCov_Pine* to 50% in the survey plots would reduce *Spp_Richness* by ~5 species (16 species at 50% pine), and would also reduce *Spp_Diversity* to 2.27. This possibly suggests a threshold of no more than 50% pine may be permissible to maintain *Spp_Richness* and *Spp_Diversity* in a plot, although, less is preferable and would increase both indices.

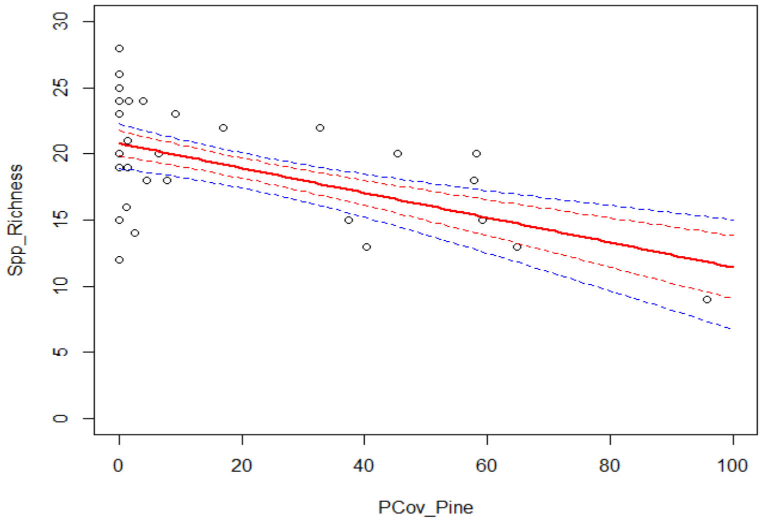
From Figure 8.2c it can be predicted that decreasing *PCov_Pine* in a plot from 78% to 0% would double *Spp_Priority,* whilst an increase to 100% *PCov_Pine* would reduce *Spp_Priority* to almost a third (0% pine = 10.6, 78% pine = 5.3, and 100% pine= 3.7 priority index). The prediction inferred from Figure 8.2c suggests that a tolerable threshold of ~30% *PCov_Pine* in a plot (or compartment) would be required to maintain support for *Spp_Priority*. Furthermore, the range of *Spp_Priority* values at 0% *PCov_Pine* (3.7 to 14.3 priority) indicates that other variables were influencing the values in these plots.

It can be predicted from Figure 8.2d that decreasing *PCov_Pine* to 50% in a plot would more than double *Spp_Rarity* (19.4), and a decrease to 0% pine would double *Spp_Rarity* again (33.6). Conversely, increasing *PCov_Pine* to 100% would drastically reduce *Spp_Rarity* to only 4.5. This indicates that pine is extremely detrimental to rare bird species in the New Forest, and suggests that a much lower threshold of *PCov_Pine* be recommended within compartments to maintain rare, broadleaved preferring bird species, such as the Mistle Thrush.

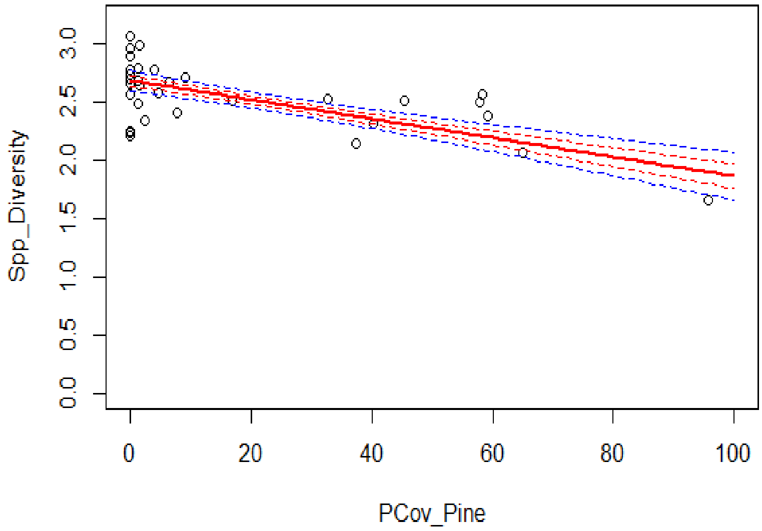
Figure 8.2e shows that *Spp_IRR* is also predicted to increase with decreasing *PCov_Pine.* However, the standard error and confidence intervals could not be calculated by the predict function for the beta regression. Moreover, the shallow slope of the line indicates uncertainty in the prediction (Figure 8.2e). This suggests that alternative factors are influencing *Spp_IRR* in the other habitats, but nonetheless demonstrates that *PCov_Pine* is detrimental to *Spp_IRR* and rare bird species.

The results indicate that higher percent cover of pine in the plots is detrimental to all of the bird indices except *Bird_Density,* most likely as a result of the dominating effect of the non-woodland plots. This suggests that converting the pine plots back to broadleaved species would benefit and improve the bird metrics, showing support for the New Forest Management Plan. This would increase the number of woodland bird species and alter the bird community to that of the broadleaved plots as displayed on the MDS in Figure 6.1 and Figure 6.4. Conversely, an increase in pine over the plot would not only reduce the bird indices (becoming more deficient in bird species), but would also alter the bird community composition to be consistent with 'Pine' plot 27. This would mean that priority species such as the Mistle Thrush, and rare species such as the Hawfinch would be absent, and populations of Chaffinches and Blackbirds would decline locally. However, the total eradication of pine is discouraged as the Wood Warbler occurred in 'Pine/Broadleaved' plot 22 and 'Beech/Oak' plot 16 (with 32% and 4% pine respectively), and pine specialists such as the Common Crossbill (notably associated with the 'Pine' plots in the MDS in Figure 6.4c) would be lost.

a)



b)



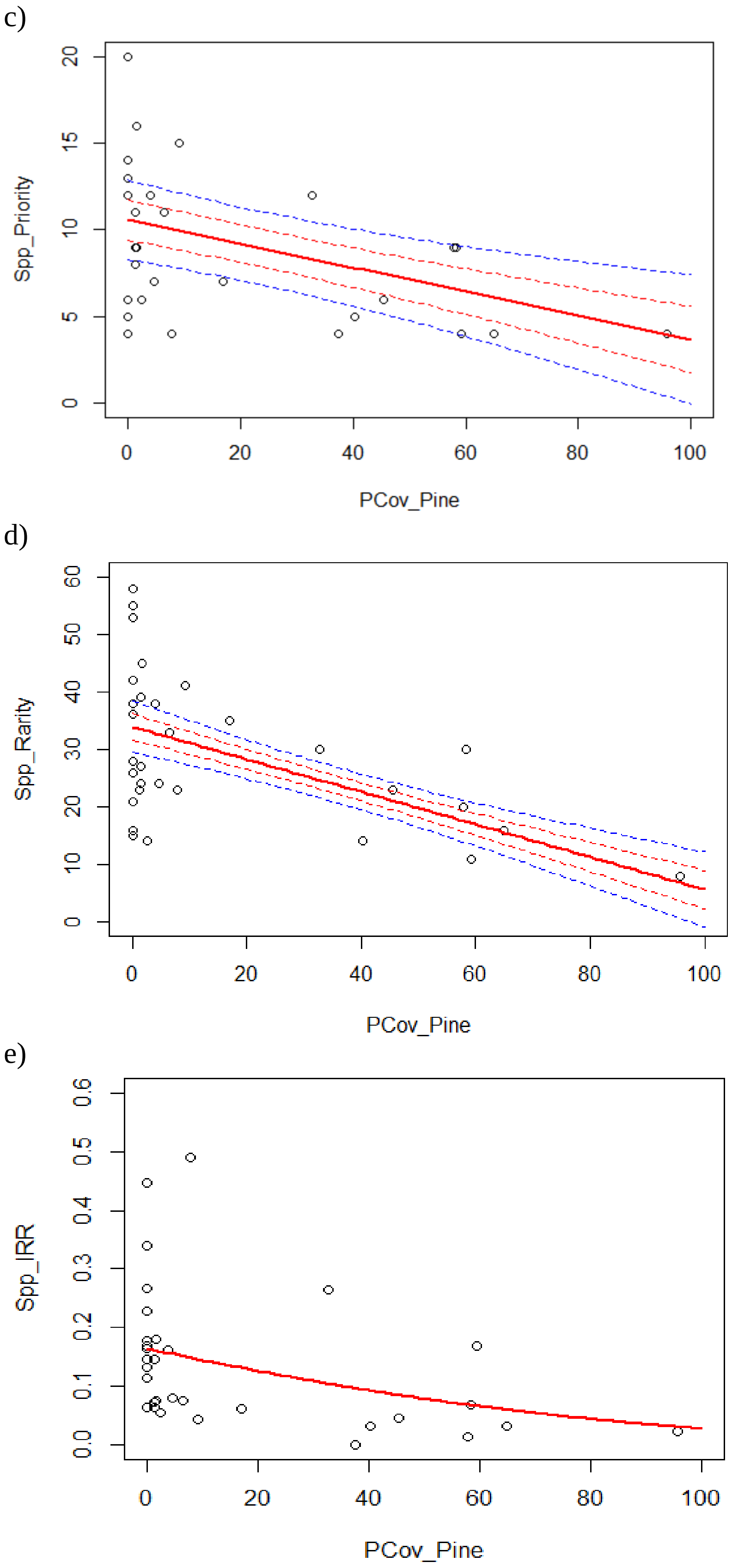


Figure 8.2. Predicting the effect of increasing PCov_Pine on a) Spp_Richness, b) Spp_Diversity, c) Spp_Priority, d) Spp_Rarity and e) Spp_IRR (without confidence intervals and with plot 30 included Table 4.2h). Actual data as circles, SE red dashed lines, 95% CI blue dashed lines (R version 3.5.2).

8.4.3 Beech Decline – New Forest

From the data presented in Figure 8.3 it can be predicted that *Spp_Decline* (i.e. the number of declining bird species supported) would decrease with decrease in *PCov_Beech*, from ~9 declining species at 70% beech in the plot, to < 6 declining species at 0% beech. This suggests that national scale trends of beech decline, and its associated impacts on bird species decline, can be detected at the local level within the New Forest.

The larger confidence intervals at the higher percentages of *PCov_Beech* and the shallow slope of the prediction, suggest that other factors are also influencing *Spp_Decline* in the survey plots, such as *PCov_Oak* and *Ht_Av*, included in the multi-models (Table 4.2d). This could also be a result of *PCov_Oak* and *PCov_Beech* being present together in the majority of the broadleaved plots (Figure 4.1 and Table S 4.2). The range of values at 0% *PCov_Beech* (3 to 9 declining species) also suggests that other factors are affecting *Spp_Decline* in the conifer and/or non-woodland plots (i.e. 'Heathland' and 'Scrubland'). Unfortunately, the other relationships of the habitat variables to *Spp_Decline* in the survey plots in the New Forest were not strong enough to make reliable predictions from. Nevertheless, a threshold of at least 50-60% *PCov_Beech* should be maintained in the broadleaved plots across the New Forest landscape to support certain declining bird species that depend on beech.

The MDS analysis in Section 6 shows that the Hawfinch, Chaffinch and Great Tit were associated with a high percent cover of beech. However, the Chaffinch and the Great Tit occurred in plots with various percentages of beech, which suggests that these generalist species are buffered from beech decline by utilising other tree species, such as oak. In contrast, the Hawfinch was only present in plots with a high percent cover of beech: 'Beech' plots 10 and 11, and 'Beech/Oak' plots 16 and 19, with 57%, 62%, 53% and 44% cover beech, respectively. This suggests that the Hawfinch has a greater dependence on beech, at least in the New Forest, and therefore is more likely to be negatively influenced by beech decline. The Hawfinch was not recorded in all the plots with a high percent cover of beech, probably as a result of not being observed due to their rarity and mobility, but this also indicates that other factors could influence habitat preference, such as woodland cover (Kirby et al. 2015).

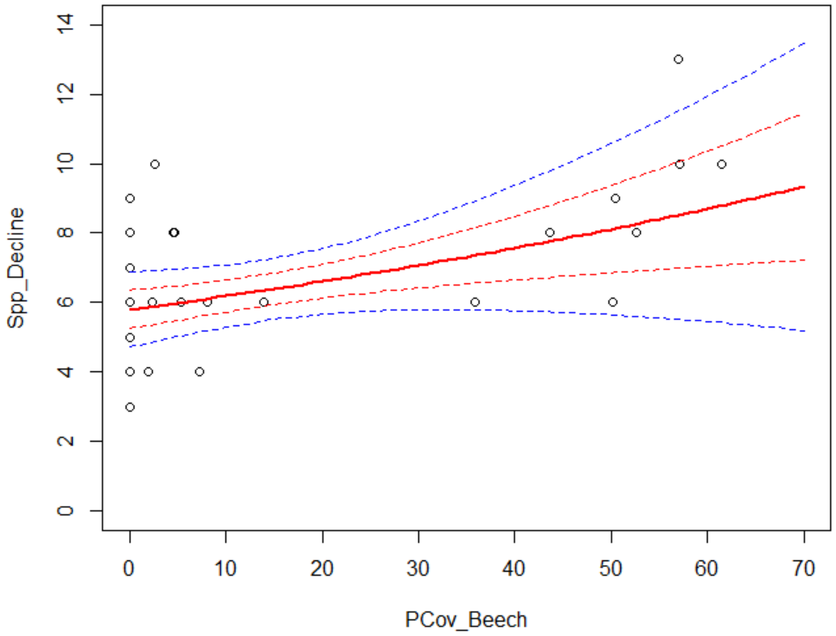


Figure 8.3. Predicting the effect of increasing PCov_Beech on Spp_Decline with the mean of the other variables in the model remaining constant, represented by the solid red line. Red dashed lines are the standard error and the blue dashed lines are 95% confidence intervals, circles are actual data (R version 3.5.2).

8.4.4 Hedge Decline – Cambridgeshire

In the landscape-level predictions (i.e. including both wood and field transects together), *P_HedgeLen* was only significantly related to *Spp_Decline*, indicating that the transects with higher proportions of hedge length supported more bird species with declining populations in this agricultural landscape. However, there was uncertainty in the confidence intervals at the higher values of *P_HedgeLen* and a shallow slope of the line, most likely as a result of the woodland transects having 0% *P_HedgeLen* and reasonably high *Spp_Decline* (see Table S 5.12). Therefore, the field transects were analysed alone, to consider the effects of decreasing *P_HedgeLen* on the bird indices in the farmed area (Table 5.6 and Table S 5.20).

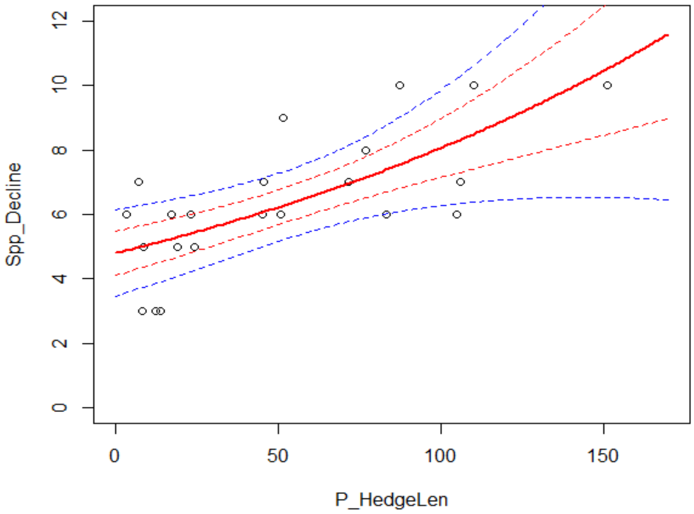
It can be predicted from Figure 8.4a that decreasing *P_HedgeLen* would significantly decrease the number of declining bird species (*Spp_Decline*) supported in the farmland transects. Furthermore, the field-only transect predictions in Figure 8.4 show that decreasing *P_HedgeLen* would also decrease *Bird_Density*, *Spp_Richness*, *Spp_Diversity*, *Spp_Priority* and *Spp_Rarity*, presumably decreasing generalist woodland birds, as well as declining and rare farmland birds. For the majority of the bird indices in the field transects, increasing *P_HedgeLen* to 100%

(i.e. the full length of a field transect) would double or more than double the value of the bird index, highlighting the great importance of hedgerows for birds in an agricultural landscape.

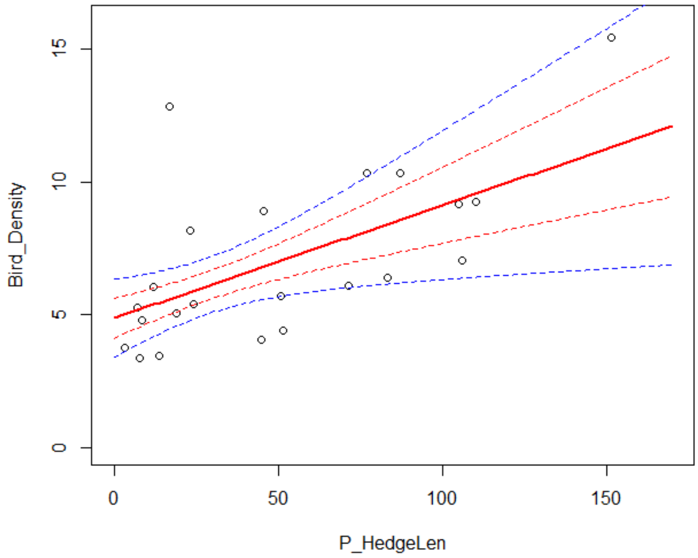
At the landscape-level the most suitable habitat variable to assess the role of hedgerows and other shrubby vegetation is *PCov_2-5m*. Figure 8.5 shows that *Bird_Density* and *Spp_Priority* would be predicted to more than halve with a decrease of *PCov_2-5m* from 40% and 35%, respectively, to 0%, in the transects over the landscape. However, the prediction has greater uncertainty at higher values of *PCov_2-5m*, represented by the large confidence intervals. This is a consequence of the single data point, 'Wood Edge' transect 24, having high *PCov_2-5m*, but lower *Bird_Density* and *Spp_Priority* (Figure 8.5). This suggests that other variables included in the model set, such as *PCov_Oak* and *PCov_Ash* (Section 5.4.9) could be increasing *Bird_Density* and *Spp_Priority*, in conjunction with the understorey in the woodlands. In contrast, the field transect-only analysis in Figure 8.4, predicted that *P_HedgeLen* would increase *Bird_Density* and *Spp_Priority* rather than *PCov_2-5m*, most likely as a result of *P_HedgeLen* being a more accurate measure of the presence of hedge habitat in the transects, i.e. this includes hedges less than 2 m tall.

From the MDS in Figure 7.1b it can be predicted that the hedge specific birds affected by hedge decline would be the Whitethroat, Goldfinch, Yellowhammer, Linnet and Greenfinch. Furthermore, the MDS indicates that other species such as the Magpie, Starling, Cuckoo, Carrion Crow and Pied Wagtail would also be affected by a decline in *P_HedgeLen* (Figure 7.1b and Figure 7.3).

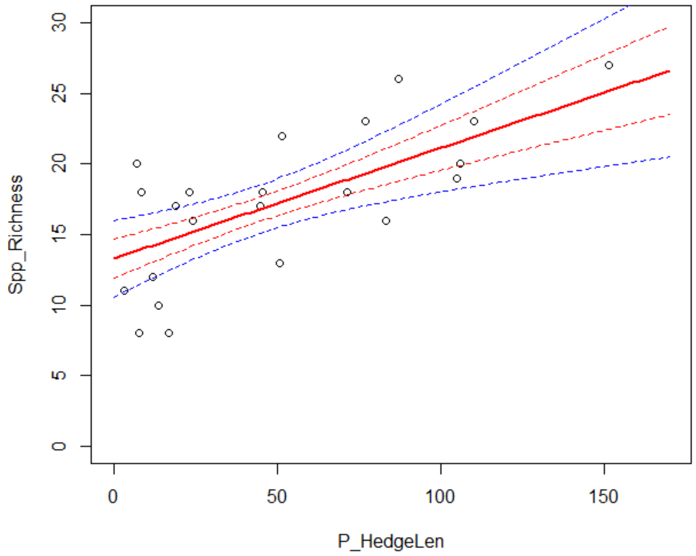
a)



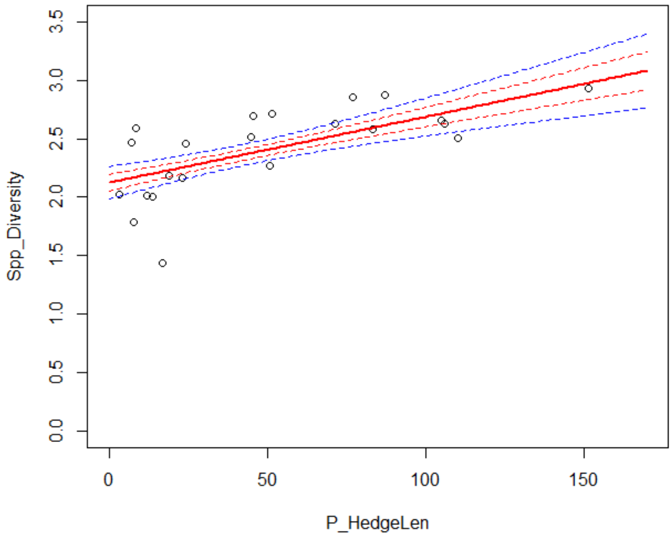
b)


Bird_Density
0
5
10
15
0
50
100
150
P_HedgeLen

c)


Spp_Richness
0
5
10
15
20
25
30
0
50
100
150
P_HedgeLen

d)


Spp_Diversity
0.0
0.5
1.0
1.5
2.0
2.5
3.0
3.5
0
50
100
150
P_HedgeLen

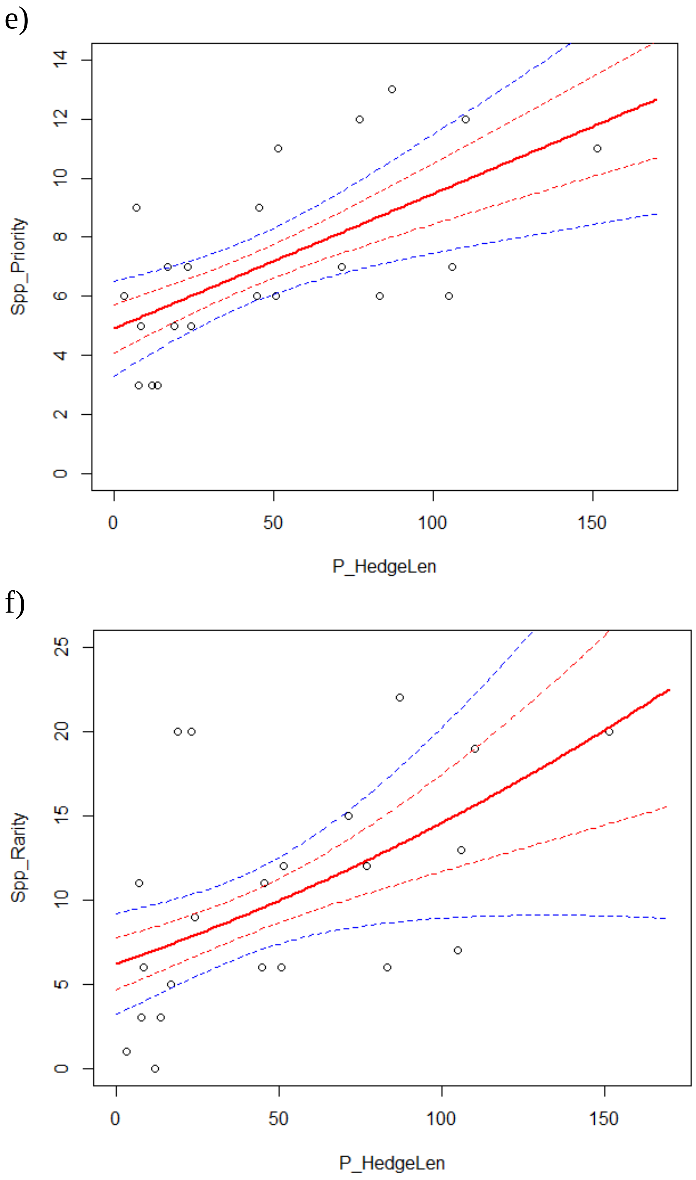


Figure 8.4. The prediction of increasing P_HedgeLen on a) Spp_Decline, b) Bird_Density, c) Spp_Richness, d) Spp_Diversity e), Spp_Priority and f) Spp_Rarity in the field only transects. The open circles represent the actual data points, solid red line is the prediction (with the mean of the other variables in the model remaining constant), red dashed lines are the standard error and blue dashed lines are 95% confidence intervals. Note, connecting hedges contribute to P_HedgeLen, in addition to hedgerow along the line of the transect, see Section 3.3.3.2 (R version 3.5.2).

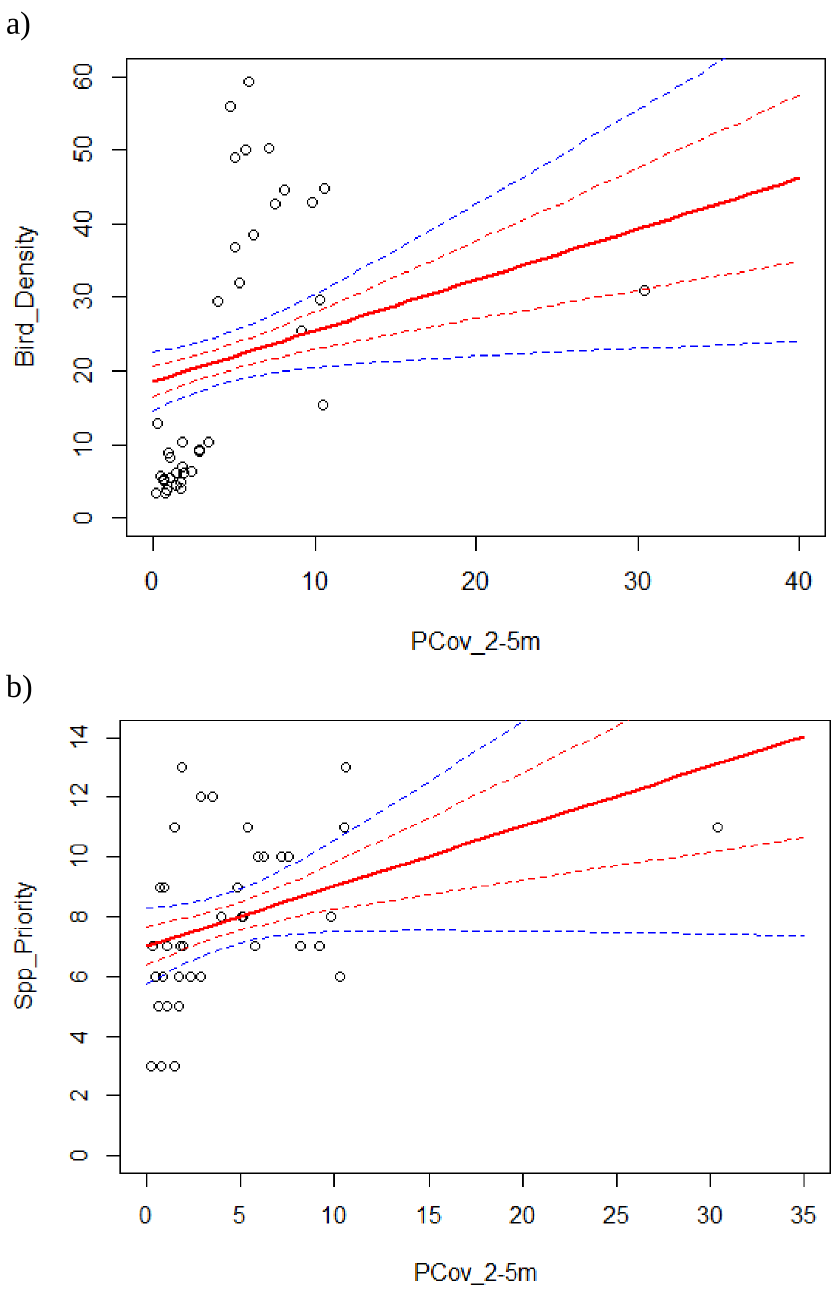


Figure 8.5. The effect of PCov_2-5m on a) Bird_Density and b) Spp_Priority in the landscape, with the mean of the other variables in the model remaining constant. Red solid line is the prediction, red dashed lines are the standard error and the blue dashed lines are the 95% confidence intervals (R version 3.5.2).

8.4.5 Spread of Improved Grass – Cambridgeshire

From the data presented in Figure 8.6 it can be predicted that increasing *PCov_ImpGrass* in the transects would decrease both *Spp_Richness* and *Spp_Diversity*. The prediction shows that increasing *PCov_ImpGrass* from 0% to 100% would halve *Spp_Richness* and almost halve *Spp_Diversity* (Figure 8.6).

Figure 8.6 also shows that increasing *PCov_ImpGrass* to 50% would significantly reduce *Spp_Richness* and *Spp_Diversity* (to 14 species and 2.15 diversity, respectively). This indicates that improved grass is detrimental to birds, most likely as a result of the low insect prey density, causing a reduction in the numbers of foraging birds in these transects.

The transects with high *PCov_ImpGrass* were 'Field By_Wood' transects 21 and 22 (73.6 and 93.1%, respectively), with the apparent 'exclusion zone' contributing to the significant reduction in *Spp_Richness* and *Spp_Diversity* in the fields adjacent to woodlands. The MDS plots in Figure 7.1 and Figure 7.4 show that the bird species associated with these two transects were the corvid species, i.e. the Rook, Raven and Jackdaw, and no woodland bird species were associated with these transects. This supports the effect of high predator numbers in fields adjacent to woodlands creating 'exclusion zones' and reducing numbers of woodland birds. Furthermore, it is worth noting that the Yellowhammer did not occur in any of the 'Field By_Wood' transects, and the Whitethroat was absent from the two transects with the highest percent cover of improved grass. However, this could also be due to the lack of hedges in these transects.

The low number of data points with high *PCov_ImpGrass* has resulted in larger confidence intervals on the *Spp_Richness* prediction shown in Figure 8.6a. However, the confidence intervals for *Spp_Diversity* in Figure 8.6b are narrower, possibly as a result of the smaller range of *Spp_Diversity* values. It is worth noting that only *Spp_Richness* and *Spp_Diversity* were significantly related to *PCov_ImpGrass* in the multi-models in Section 5.4.9, and therefore no other bird indices can be predicted associated with increasing *PCov_ImpGrass*.

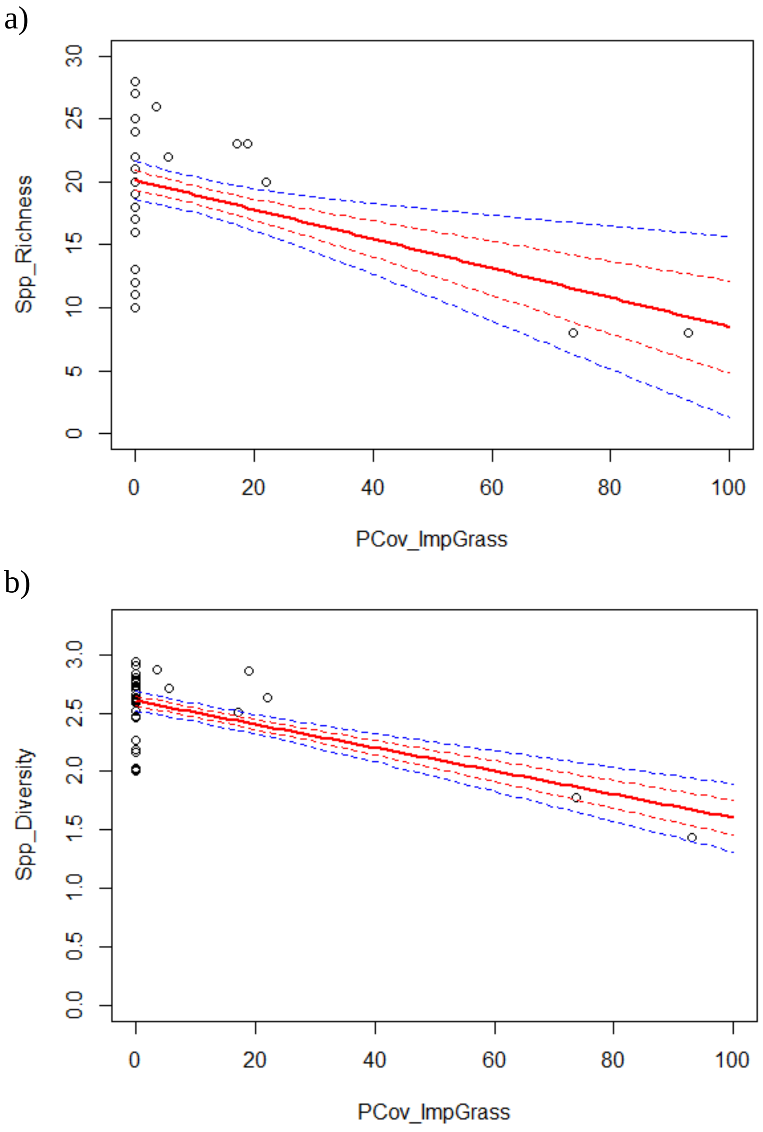


Figure 8.6. The prediction of the effect of PCov_ImpGrass on a) Spp_Richness b) Spp_Diversity over the landscape, with the mean of the other variables in the model remaining constant. Red solid line is the prediction, red dashed lines are the standard error and the blue dashed lines are the 95% confidence intervals (R version 3.5.2).

8.4.6 Oak Decline – Cambridgeshire

It can be predicted from Figure 8.7 that decreasing *PCov_Oak* from 40% to 0% would reduce *Spp_Rarity* to more than a third (i.e. the number of rare species that could be supported) in the transects (from a rarity index of 35 at 40% oak to 11 at 0%). The prediction also indicates that a reduction from 40% to only 20% *PCov_Oak* in the transects would reduce *Spp_Rarity* by almost half (~23 rarity), and a further reduction to 10% *PCov_Oak* would halve *Spp_Rarity* (~17). This indicates that oak decline nationally has the potential to affect the national population size of certain bird species, and thereby increase their rarity. This national-scale trend would play out at the local level in the Cambridgeshire landscape. *PCov_Oak* was also relatively rare in the Cambridgeshire landscape as the highest percentage cover is only 32.9% in any woodland transect (Table S 5.3). The larger confidence intervals at the greater values of *PCov_Oak*, suggest that other variables may also influence *Spp_Rarity* in the transects, such as *PCov_2-5m*, present in the full model set (Table S 5.19g), which may also indicate an effect of understorey on rare birds.

A major decrease in oak, such as from tree death or felling, would result in a bird community similar to 'Wood Edge' transect 24 which has 2.9% oak and a low rarity value – a number of rare species, including the Raven, Marsh Tit and Spotted Flycatcher were absent from this transect. However, 'Wood Edge' transect 24 was the only transect in the Cambridgeshire study landscape that contained the Turtle Dove, which is rarer than both the Spotted Flycatcher and the Marsh Tit, and has also declined dramatically (by 87%) in the ten years leading up to 2015 (Massimino et al. 2017). 'Wood Edge' transect 24 also had the highest density of Bullfinch which was rated as reasonably rare in the current study. This is most likely contributing to the relationship of species rarity with the percent cover of vegetation at 2-5 m, but more importantly further supports the need for a heterogeneous landscape for habitat-specific birds.

It is difficult to ascertain from the MDS results in Section 7.4.1 which rare bird species were related to a higher percentage of oak, as the woodland birds were positioned close together (and on top of the woodland transects) and cannot not be easily differentiated (Figure 7.1). However, the data show that the transects with a higher percentage cover of oak contained a number of rare bird species, including the Marsh Tit, Raven and Spotted Flycatcher, that are likely to be affected by a decline in oak. Presumably as a result of their rarity, Ravens only occurred in three transects in the current study, one of which contained the Marsh Tit, almost certainly contributing to the relationship of species rarity with the percent cover of oak.

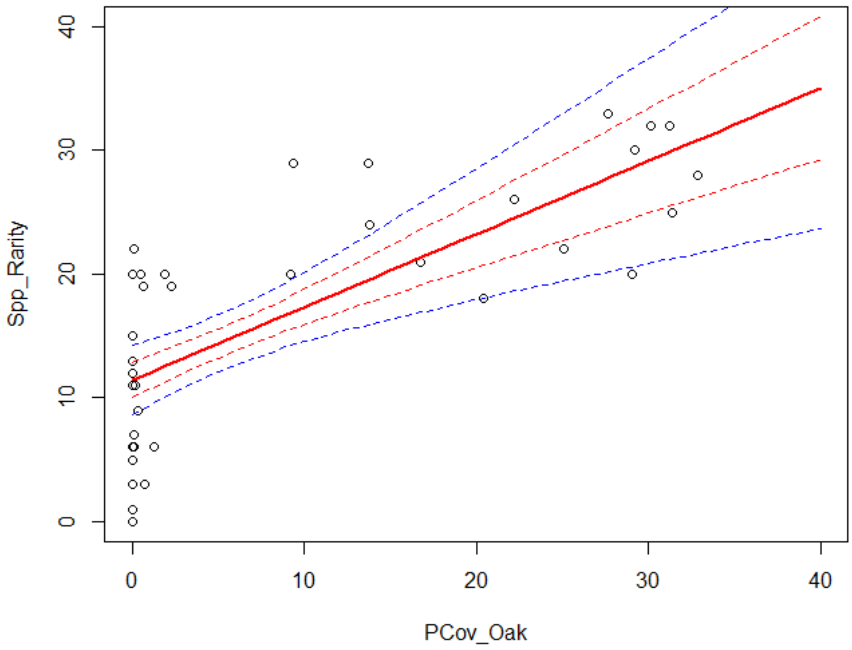


Figure 8.7. Predicting the effect of increasing PCov_Oak on Spp_Rarity in the landscape, with the mean of the other variables in the model remaining constant. Open circles are the actual data, solid red line is the prediction, red dashed lines are standard error and the blue dashed lines are 95% confidence intervals (R version 3.5.2).

8.4.7 Tree Loss Due to Changed Woodland Management or Tree Disease – Cambridgeshire

Spp_IRR shown in Figure 8.8, is predicted to significantly decrease with decreasing *Ht_Av*, suggesting that transects with taller trees in Cambridgeshire will support more rare birds, and natural tree growth and gap filling will increase *Spp_IRR*. Note that as beta regression was used in the analysis, the function could not calculate the confidence intervals and standard error for *Spp_IRR* (Figure 8.8). Decreasing the *Ht_Av* of a transect by only 5 m, from 15 m to 10 m, is predicted to decrease *Spp_IRR* by almost half (from 0.38 to 0.22), presumably caused by the loss of tall trees, such as ash and oak, indicating that a loss of only a few tall trees in a transect could have dramatic effects on *Spp_IRR*. Furthermore, reducing *Ht_Av* to 0 m, as a result of felling the whole area of a transect, would dramatically reduce *Spp_IRR* to 0.03, indicating the importance of maintaining taller oak and ash trees (mainly in the woodland transects) over the agricultural landscape for rare bird species. This may also advocate ‘continuous cover’ felling of shorter broadleaved trees, if required. However, these conclusions are only directly applicable to the Cambridgeshire landscape included in this study; in other contexts, different rare species will be influenced by other factors. For example, clear fell areas in conifer plantations can be important for species, such as the Tree Pipit and Nightjar.

The MDS plots in Figure 7.1b and Figure 7.7b do not indicate any specific bird species that would be affected by changes in average tree height, due to the clustering of the woodland species. However, the transect with the highest average tree height and highest species rarity contained the rare Raven and Marsh Tit, indicating that they would be influenced by an alteration in average height of the vegetation and, perhaps in the case of the Marsh Tit, could be lost locally.

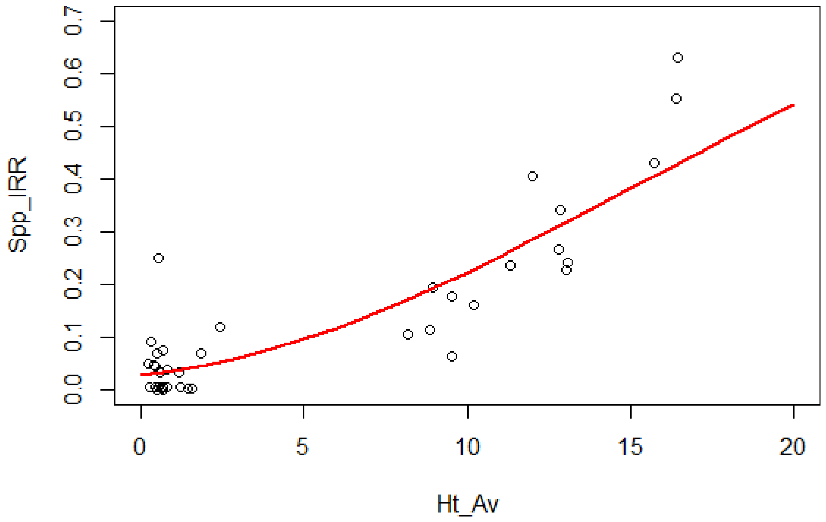


Figure 8.8. Predicting the effect of increasing Ht_Av on Spp_IRR, from the multi-model set, with the mean of the other variables in the model remaining constant. Note: standard error and confidence intervals could not be calculated from a beta regression model (R version 3.5.2).

8.5 Discussion

The significant relationships from the multi-model analysis in Section 4.4.5 were used to investigate the scenarios of scrub and pine removal, and of beech decline in the New Forest on the relevant bird indices. Scrubby vegetation removal at the 2-5 m height level was predicted to decrease bird density. However, due to the complexity of the variable height VDR, the results suggest a varied vegetation profile would increase bird density, and also species richness and diversity. A decrease in pine over the plot was predicted to significantly increase bird species richness, diversity, the numbers of priority and rare species and IRR at the plot level.

A threshold could be implemented to prevent a reduction in the bird indices over a plot, however, maintaining a low percentage of pine (~20%) would be advisable to provide habitat for rare conifer preferring bird species. The number of declining species supported was predicted to decrease with decreasing percent cover of beech, indicating that a higher number of bird species with declining populations were located in the beech woodlands in the New Forest. This could be attributed to national beech decline being detected in the plot level bird populations.

For Cambridgeshire, scenarios of hedge and oak decline, the spread of improved grass, and tree loss as a result of woodland management changes or disease, were investigated using the significant multi-model results from Section 5.4.9. The prediction showed that decreasing percent hedge length would also decrease bird density, species richness, diversity, the numbers of declining, priority and rare species in the fields, decreasing generalist woodland birds, as well as declining and rare farmland birds. An increase in the percent cover of improved grass was predicted to decrease both species richness and diversity, considered to be a result of low prey density reducing the numbers of foraging birds, and also the presence of corvid predators creating an exclusion zone in fields adjacent to woodlands. Decreasing the percent cover of oak over the transect was predicted to decrease the rarity index. This is a possible consequence of national oak decline, which has the potential to affect the national population sizes of certain bird species, such as the Marsh Tit and Spotted Flycatcher. This national-scale trend was reflected in the local populations in the Cambridgeshire landscape. IRR was predicted to decrease with decreasing average height of vegetation over the transect, indicating that rare birds (with smaller population sizes), such as the Raven and Marsh Tit, were located in the woodlands with a higher average height. A reduction of average height from 15 m to 10 m is predicted to decrease IRR by almost half, and could result from, for example, oak decline, ash dieback or selective felling.

8.5.1 Scrub Removal – New Forest

The New Forest Act (1949), the New Forest National Park Management Plan (New Forest National Park Authority 2010), and Smith and Burke (2010) all stated that removal and prevention of scrubbing up of the heathland, by grazing, cutting and burning vegetation, is essential to maintain condition of the heathland in the New Forest. The New Forest SAC designation was implemented partly for the protection of the heathland and for the conservation of three heathland specialist bird species: the Dartford Warbler, Nightjar and Woodlark (*Lullula arborea*), as well as species from other taxa and flora (Anon. 1949). Only the Dartford Warbler was recorded in the current study, as a result of the nocturnal behaviour of the Nightjar and the rarity of the Woodlark (Wright et al. 2007, Woodward et al. 2018).

The two variables selected to represent scrubland vegetation (height VDR and percent cover of vegetation at 2-5 m) had opposing effects on bird density as a result of the low density in the majority of the 'Scrubland' and 'Heathland' plots (Figure 4.4). However, one of the 'Scrubland' plots (5) had reasonably high bird density, high height VDR and high percentage cover of vegetation in the 2-5 m height range. Furthermore, the high percentage of vegetation above 5 m would indicate that in hindsight, this habitat may have been better classified as wood pasture, along with a number of heavily grazed, open, broadleaved woodland plots. This suggests that increasing the percentage of vegetation in the 2-5 m layer and taller vegetation (overstorey), would increase bird density, and is more likely to be understorey vegetation or scrub development increasing the vertical profile of the vegetation (i.e. height VDR). This is supported by Sallabanks et al. (2006), albeit in a conifer woodland, who showed that bird abundance varies as a result of understorey, canopy cover and also the size of the tree trunks. Moreover, the relationship with height VDR was more complex than the linear regression suggests, as the survey plots with medium height VDR varied in bird density. This suggests that in order to increase bird density, the scrubby areas would benefit from graduating down from taller trees, creating an ecotonal habitat varying in structure across the whole vegetation height profile, similar to pasture woodlands in the New Forest.

Overall, the 'Scrubland' habitats had the highest bird species richness and diversity in the current study (in line with the findings by Ralph 1985, and Tews et al. 2004), and also had the highest value of species priority (Figure 4.4). This indicates that allowing some 'Heathland' to scrub up would increase the number of bird species including priority bird species (Section 4.4.3). Unfortunately, the relationship between species richness and the percent cover of vegetation at 2-5 m was not included in the multi-models and so could not be predicted (Table 4.2). Additionally, the relationship of height VDR with the priority species index was also not reliable enough to predict with confidence, most likely as a result of the dominating effect of the conifers. Similarly, the relationships of the priority index with the presence of willow, depicting the scrub habitat, and the percent cover of heather, in the heathland, were not reliable enough to form predictions. However, the results of the current study still demonstrate that the ‘Scrubland’ and ‘Heathland’ habitats support priority bird species, such as the Dartford Warbler in the heathland (Section 4.5.4 and 6.5.1), although limitations in the data (i.e. too few Dartford Warblers, and no heathland specialists Nightjar and Woodlark recorded) prevented any reliable predictions.

Literature on ‘young-growth’ habitats was reviewed by Fuller (2012) who stated that declines in associated bird species in America motivated research of this habitat. However, there was comparatively little similar research in Europe, but there are concerns over population declines of scrubby (young-growth) bird species in Britain, such as the Willow Warbler (Fuller et al. 2007, Fuller 2012a, Morrison et al. 2013). Furthermore, in the current study the scrubby variables (percent cover of vegetation at 2-5m and height VDR) were not included in the model set for bird diversity, presumably as a result of the dominant negative effect of the conifers, and so could also not be used for prediction. Nonetheless, the results of the current study show that bird density, species richness, bird diversity, species priority and the number of bird species with nationally declining populations are related to both vegetation structure and composition (Section 4.4.5). Therefore, the ‘Scrubland’, pasture woodland and enclosed woodland plots will provide habitat for different bird assemblages, provided they vary in structure and composition.

As previously mentioned in Section 4.5.4, the Dartford Warbler is a lowland heath specialist confined to southern England, and therefore the species and its habitat in the New Forest are protected. However, this conflicts with maximising overall biodiversity, which according to the current study, was highest in the ‘Scrubland’ suggesting that allowing the heathland to scrub-up could increase bird diversity. Gibbons and Wotton (1996) showed that the Dartford Warbler is more vulnerable to scrub encroachment after periods of rapid land development for agriculture and forestry has slowed. Regos et al. (2015) showed that the Dartford Warbler was also affected by fire regimes, therefore management involving fire, such as heathland burning to prevent scrub encroachment, must be evaluated before incorporation into conservation plans. In addition, Paquet et al. (2006) showed that selective felling provided temporary ecotonal habitat for species, such as the Tree Pipit, and Dorresteijn et al. (2013) found that wood pasture was beneficial for the Green

Woodpecker, both results consistent with a beneficial effect of increasing scrub in the New Forest. Thus, the current research demonstrates that bird diversity (or biodiversity) should not be used in isolation to infer the health of a habitat or to inform management strategies. To maximise landscape bird diversity (gamma diversity) the multiple bird measures and bird community analysis are in agreement with the review by Vickery and Arlettaz (2012) favouring heterogeneous landscapes. The management plans for the New Forest could therefore allow scrubbing up of some heathland whilst also maintaining large areas of heathland for rare and declining heath-specific bird species. This indicates that more research must be carried out in order to evaluate the extent of both habitats required to support their respective populations.

The results of the current study, therefore, supports both: (i) the protection of the heathland habitat in the New Forest, to conserve rare species, such as the Dartford Warbler (Section 4.5.4); and (ii) the provision for some areas to scrub up, possibly at the expense of woodland, which would increase vegetation diversity, and subsequently, bird diversity and support more priority species in the New Forest.

8.5.2 Pine Removal – New Forest

The Management Plan for the New Forest stated that they would “increase the proportion of broad-leaved woodland in the Inclosures...reinstating or creating new areas of native woodland”, and therefore remove or reduce the amount of conifers, including pine in the woodland (New Forest National Park Authority, 2010, p. 29). The result (in Figure 8.2) predicted that decreasing the percent cover of pine would increase bird species richness and diversity, and support more conservation priority and rare bird species, and therefore be beneficial for birds. Bird density was the only bird index with the percent cover of pine absent in the multi-models, presumably as a result of the dominant effect of the non-woodland plots. The Management Plan also suggested utilising the New Forest’s reputation to sustainably manage hardwood (i.e. broadleaved tree species) privately and locally, most likely as a substitute for conifer (New Forest National Park Authority 2010).

A decrease to 50% cover of pine over a plot would increase species richness and support a greater number of priority species, and may be sustainable. However, bird diversity and rarity would drastically reduce (consistent with Adams and Edington 1973), and rare species, such as the Hawfinch, would most likely be lost from the plot. Mixed broadleaved and pine woodland plots would buffer the negative effects of pine and would also increase bird diversity, somewhat, by creating intermediate values of species richness and diversity as shown in Figure 4.4. Populations of coniferous woodland-preferring species, such as the Goldcrest, Siskin, Common Crossbill (Calladine et al. 2018), and to a lesser extent, Coal Tit, as well as dead wood species (Cadieux and Drapeau 2017), will also be sustained by maintaining some conifer plots, and thus also maintain or increase overall gamma bird diversity.

The ‘Pine’ plots surveyed in the New Forest were relatively young plantations. These plantations are often felled before reaching maturity and thus lack certain microhabitats such as holes in dead wood (Baguette et al. 1994, Fuller et al. 2007, Fuller and Robles 2018). Many bird species are associated with either mature woodlands or early successional habitats, with few preferring these intermediate stages of growth, resulting in low bird density and diversity in the pine plantation plots (Paquet et al. 2006, Fuller et al. 2007, Begehold et al. 2015).

Plantation management will also influence the birds, for example, clear felling will benefit early successional species (Costello et al. 2000, Gram et al. 2003), and birds, such as the Tree Pipit (Paquet et al. 2006). In contrast, bird species preferring mature, complex and dense vegetation structures, such as the Wood Warbler, would benefit from continuous cover felling (Calladine et al. 2015, Huber et al. 2016, 2017), but this will affect bird species that avoid gaps (Fuller and Robles 2018). However, in the US, Costello et al. (2000) found higher species richness in the clear cut areas compared with the selected/continuous cover felling woodlands, which contained the same species as mature woodlands. Gram et al. (2003) showed that bird species were affected differently by the silviculture practises in the US, but agreed that clearcutting provides habitat for successional species. In contrast, the study in Belgium by Du Bus De Warnaffe and Deconchat (2008), showed that in coniferous plantations continuous cover did not increase bird diversity in comparison to clear-felling. This suggests that heterogeneity may be required, in terms of woodland composition and management practises in plots across the landscape in order to increase bird diversity (Hartley 2002, Gram et al. 2003, Begehold et al. 2015).

The rare Firecrest was shown in the current study to avoid pine, with a preference for the ‘Other Conifer’ habitats and also utilised holly in the broadleaved habitats in the New Forest (Batten 1973, Clements et al. 2017). Furthermore, a recent study by McNab et al. (2019) showed that the Common Crossbill has adapted to feed preferentially on non-native conifers (such as Sitka spruce) in Scotland. In the current study, the highest values of IRR (Index of Relative Rarity) were in ‘Heathland’ and ‘Other Conifer’ plots (Figure 4.4), but unfortunately due to the small sample size of only one plot from each habitat having a high value, the relationship, and prediction, was not reliable, and therefore not included. The ‘Other Conifer’ plots were also higher in species richness and diversity than the pine plots (Figure 4.4), indicating that out of the non-native conifers in the New Forest the other soft conifers were preferable to pine. This suggests that providing and maintaining smaller patches of other conifer species, such as Douglas fir, will conserve rare bird species whilst maintaining bird diversity. However, it is worth noting that some of the ‘Pine’ plots had slightly higher values of species decline and species priority than the ‘Other Conifer’ plots (Figure 4.4), further advocating the inclusion of some pine species, perhaps as mixed pine/broadleaved plots. Additionally, management practises may also differ between the pine and other conifer plantations resulting in the different bird diversity and composition (Fuller and Robles 2018). It may be the case that continued silviculture of clear-felling occurred in the surveyed ‘Pine’ plots resulting in even stand-aged plots (*pers. obs.* A. Barnes), and temporary successional habitats (Costello et al. 2000, Gram et al. 2003, Paquet et al. 2006). Conversely, selected or continuous cover felling possibly occurred in some of the ‘Other Conifer’ plots, indicated by the large trees with felled patches in the survey plots (*pers. obs.* A. Barnes), creating variety in vegetation species composition and stand age from re-growth (Calladine et al. 2015, 2018).

The results of the current study indicate that retaining pine (and also other conifer species) in some plots will ultimately increase alpha diversity in those plots and gamma bird diversity over the landscape, as well as providing habitat for rare specialist bird species, such as the Firecrest. The intermediate even stand-age pine surveyed in this study is poor bird habitat, while clear-cutting sections provides successional habitat for rare species. Therefore, management practises should also be varied over the landscape to benefit multiple bird species.

8.5.3 Beech Decline – New Forest

Beech decline has been documented in the UK for a number of years (Jung 2009), and therefore may be influencing bird population numbers in England. The number of bird species with population declines that can be supported in the survey plots was predicted to increase with increasing percent cover of beech. This indicates that beech loss could be having a negative effect on bird populations in England, which is reflected by the high number of bird species with declining populations supported in the beech plots in the New Forest.

The New Forest National Park Management Plan stated that the National Park Authority intended to increase the proportion of broadleaved woodland in the Inclosures (enclosed woodland), and also increase effort to restore and connect fragmented pasture woodland, as well as areas of coppice (New Forest National Park Authority 2010). The prediction results indicate that the percent cover of beech should be maintained and increased in the woodland plots, either as native broadleaved woodlands or, as the management plan suggests, for hardwood timber. The results of the current study may also indicate that certain bird species prefer dense beech canopy, and thus that beech decline, and also silviculture, could disrupt the continuous canopy cover affecting these species (Begehold et al. 2015). Begehold et al. (2015) found that late and decaying beech woodland stages, with a high amount of deadwood was preferred by the majority of abundant bird species, and a considerable number avoid the younger phases. However, the results of the current study suggest that sustainably managing beech for timber would create variation in canopy openness and tree age, increase understorey vegetation cover (2-5 m), and consequently could have the positive effect of supporting more bird species and different bird communities (Begehold et al. 2015). Begehold et al. (2015) also showed that bird species have differing habitat preferences in the beech woodlands. For example, creating gaps would benefit the Wren, younger stages are preferred by the Wood Warbler, whereas the Hawfinch prefers the late, mature woodland stage (Begehold et al. 2015). Therefore, a mosaic of structural diversity in beech woodlands would provide habitat for a greater diversity of bird species, which conventionally managed beech woodlands lack (Begehold et al. 2015, Wesołowski et al. 2018). Furthermore, protecting insectivorous bird species will also provide top-down control of invertebrate herbivory, favourable for the growth of beech and oak seedlings and saplings (Dekeukeleire et al. 2019).

The results of the current study indicate that the Hawfinch is dependent on beech in the New Forest. In addition, Kirby et al. (2015) stated that declines in the Hawfinch may have also been attributed to Dutch elm disease in the past, as well as factors beyond the woodland, such as food availability, due to them travelling further for resources. Furthermore, as the bird indices in the current study encompass multiple bird species, it is difficult to evaluate individual relationships. However, the results suggest that a decline in the percent cover of beech would clearly affect the Hawfinch population (and possibly populations of Great Tits and Chaffinches although these two species are likely to be more affected by declines in oak, (Whittingham et al. 2001)) in the survey plots in the New Forest. It is worth noting that a high number of declining species were also supported in an ‘Oak’ plot, possibly suggesting that Acute Oak Decline (AOD) could also be effecting bird population declines in England, reflected in the birds in the New Forest (Section 4.5.2, e.g. Denman et al. 2010, Denman et al. 2012). Unfortunately, due to the small sample size predictions were not able to be made, but does suggest that the New Forest could be acting as a refuge for oak dependant bird species, as the presence of AOD has not yet been recorded widely in the New Forest (Forest Research 2019).

8.5.4 Hedge Decline – Cambridgeshire

In recent decades, agricultural intensification has altered the landscape over Europe from a mosaic of habitats to more of a uniform configuration, which has resulted in farmland biodiversity declines, including birds (Donald et al. 2001b, Benton et al. 2003, Wilson et al. 2005, Vickery and Arlettaz 2012). Farmland now has more bird species with declining populations than any other habitat type (Tucker and Evans 1997, Wilson et al. 2005). Historic hedge removal as part of agricultural intensification has had a particularly detrimental effect on bird abundance and diversity (Robinson and Sutherland 2002). This is supported by the multi-model results in the current study, which showed that the number of declining species that can be supported in the Cambridgeshire transects was positively related to the proportion of linear features (e.g. field boundaries) that are hedge (Section 5.4.9). However, there was uncertainty in the relationship, most likely as a result of the dominating influence of the woodlands. Therefore, a reliable prediction could not be made, but suggests that hedges in the field transects support more declining bird species than the woodland transects, consistent with the findings of Tucker and Evans (1997).

The woodlands had no hedges (i.e. linear features, as defined in Section 3.3.3.2), therefore indicating that other factors were influencing the number of declining species present in the woodland transects, such as understorey vegetation at the 2-5 m height layer (Table 5.5d and Table S 5.19 d). Section 5.5.2 also shows that the bird species with declining populations differed between the woodland and the field habitats, for example, the Marsh Tit was present in the woodlands, whereas the Yellowhammer was declining in the field habitats. This demonstrates the mosaic nature of the landscape and the importance of maintaining habitat quality in both the farmland and woodland elements, as well as overall habitat heterogeneity.

Consequently, the field transects were analysed alone, and decreasing the proportion of hedge length was predicted to decrease the number of species with declining populations in England that can be supported in the transects (Fuller et al. 1995, Siriwardena et al. 1998, Donald et al. 2001a). Furthermore, decreasing the proportion of hedge length would also significantly decrease bird density, species richness, diversity, and the number of conservation priority and rare species over the field transects. This confirms that hedges are vital in an agricultural landscape to support and maintain bird diversity and richness, as well as rare and declining bird species. The review by Hinsley and Bellamy (2019) supports the findings of the current study, and found that only seven species can be termed as hedge specific, including the Yellowhammer, Greenfinch, Goldfinch, Linnet and Whitethroat. They also showed that heterogeneity and diversity of hedges, in terms of structure (volume, height, width), compositional vegetation species and surrounding habitat must be maintained in order to accommodate multiple bird species with differing preferences, and therefore increase overall gamma bird diversity (Sauerbrei et al. 2017, Hinsley and Bellamy 2019).

The results of the current study suggest that a reasonable management measure would be to border every field by a hedge along the full length of the field (*P_HedgeLen* = 100%), which would increase the number of bird species present. Bird density, diversity, species richness, and the number of birds with declining populations supported should increase, plus mediating any further bird declines. However, this may be too simplistic, as the study by Sauerbrei et al. (2017)

demonstrated an influence of hedge structural diversity on three groups of bird species. Two of the groups, one containing the Yellowhammer, and the other the Blackbird, preferred hedges with trees (Sauerbrei et al. 2017). This is supported by Morgan and O'Connor (1980) who showed that hedgerows were important nesting habitat for the declining Yellowhammer in farmland, and hedges containing trees were preferred as perches for singing. Moreover, the Yellowhammer group preferred long and broad hedges, whereas the Blackbird group preferred tall hedges (Sauerbrei et al. 2017). The third group containing the Whitethroat, however, preferred smaller, broken-up hedges with gaps and lower vegetation density (Sauerbrei et al. 2017). Furthermore, all three groups were shown to be affected by a reduction in the biomass of hedges, indicating that alternative habitat must be provided if hedges are to be removed or reduced, for example by excessive cutting, and to support heterogeneity in the hedgerows (Sauerbrei et al. 2017).

Undoubtedly, increasing woodland area or the number of woodland patches in the Cambridgeshire landscape would increase bird diversity and species richness to a greater degree than increasing the number or length of hedgerows (Figure 5.6). However, an excessive increase in woodland area would cause further population declines of farmland bird species which depend on hedges, such as the Yellowhammer, and open habitat species, such as the Skylark, that avoid high boundaries (Suhonen et al. 1994, Donald et al. 2001b). Furthermore, Pickett and Siriwardena (2011) showed that increasing farmland heterogeneity increases 66-75% of individual bird species abundance, and promotes biodiversity by increasing the number of single habitats that birds prefer. However, they also showed that certain, often threatened and declining bird species, avoid areas with high heterogeneity, such as the Turtle Dove, Skylark, Linnet, Yellow Wagtail and Reed Bunting. Pickett and Siriwardena (2011) concluded that promoting heterogeneity of the farmed area may increase bird species density at the expense of overall national diversity, as threatened species often require specific management needs. This further supports the findings of the current study, that in order to prevent further bird declines, bird diversity and the numbers of rare and declining bird species, must be managed appropriately in order to maintain overall landscape bird diversity.

Increasing the percent cover of lower vegetation at 2-5 m over the landscape was also predicted to increase both bird density and the number of conservation priority bird species that can be supported in the transects. The relatively large confidence intervals indicate uncertainty in the prediction due to there being only one scrubby habitat included in the Cambridgeshire study (‘Wood Edge’ transect 24), suggesting that more data are required for further research. Nevertheless, in the New Forest analysis in Section 4.5.1 of the current study, the areas with scrubby successional vegetation exhibited high bird diversity and a greater number of conservation priority bird species (Figure 4.4). Therefore, this suggests that increasing scrubby vegetation at 2-5 m in the field transects would increase the number of bird species present, as well as increasing bird density, as per the prediction taken from Figure 8.5. However, the uncertainty in the prediction of species of conservation priority was more complex, as both woodland and field transects had high values of the index, almost certainly for different reasons (Figure 5.6). Nonetheless, it can be predicted from the analysis of the field transects that increasing the proportion of hedge length would increase the number of conservation priority species supported in these transects. It is worth noting that hedge height can range below 2 m and possibly beyond 5 m, therefore the percent cover at 2-5 m would not encompass all woody vegetation in the fields, and hence was not significant in the field-only analysis.

The current study has also shown that as well as the declining bird species, bird community composition also differs between the woodland and non-woodland habitats (Figure 7.1). Therefore, the bird indices and individuals in the community should be evaluated simultaneously to inform management strategies across the landscape, and hence provide habitat (e.g. hedges) for declining and/or rare bird species, as well as increasing bird diversity. This would involve increasing the heterogeneity of semi-natural/woody habitat and hedge diversity (in terms of height, width, volume etc.), over the agricultural landscape (Sauerbrei et al. 2017).

8.5.5 Spread of Improved Grass – Cambridgeshire

In the second half of the 20th Century lowland grassland management changed and became more intensive through the increased use of fertilisers, switching to silage and increased livestock densities (Vickery et al. 2001). In the current economic climate with the threat of the UK leaving the European Union (EU), there are a number of uncertainties with regards to the UK’s agricultural policies. This could result in diversification due to a reduction in profitability and losses of EU subsidies, or increased cereal crop production rather than an increase in livestock pasture due to export limitations (Ares 2019). One such diversification could be an increase in horse grazing for recreational purposes and pony trekking. This could contribute to an increase in the coverage of, at least partially, improved grassland (pasture), as surveyed in the current study which was predicted to decrease bird species richness and diversity and was overall poor bird habitat. Climate change could also result in the production of different crop types in a warming Britain, and potentially a reduction in beef farming, and therefore pasture, to reduce greenhouse gas emissions (e.g. Subak 1999).

The grass (and crop) composition and structure in the fields influences bird species, as well as the land-use management (Wilson et al. 2005, Vickery and Arlettaz 2012). Improved grass is fast-growing, has high sward density, often of uniform height, and low invertebrate prey abundance, therefore creating difficulties in foraging and predator detection or avoidance for birds (Wilson et al. 1999, 2005, Hopkins 2000). Faster growing grassland for silage is regularly cut early in the breeding season destroying nests, and regular cutting also prevents seed heads forming reducing seed availability for granivorous birds (Vickery et al. 2001). Improved grass used as pasture also increases nest failure from grazing and trampling, although Vickery et al. (2001) suggested that moderate grazing may increase sward diversity and structure and restore biodiversity. Unsurprisingly, from the data in Figure 8.6 it can be predicted that increasing the percent cover of improved grass would decrease both bird species richness and diversity in the transects, as a result of the unsuitability of improved grass for both foraging and nesting (Wilson et al. 1999, Vickery et al. 2001). The rarity index was also negatively related to the percent cover of improved grass, presumably as a result of reducing the number of bird species and consequently rare bird species, however, this relationship was not strong enough to make reliable predictions (Table S 5.19). It is worth noting, that the negative relationship of bird species richness with percent cover of cereal was also not reliable enough to form a prediction, but suggests that cereal had less of an impact, possibly as a result of higher prey density or the availability of cereal grain (Wilson et al. 1999).

In the current study there were only two transects with high percent coverage of improved grass, which created greater uncertainty represented by the larger confidence intervals in the species richness prediction. However, the bird diversity prediction had narrower confidence intervals, as a result of the smaller range of values. Furthermore, both transects with the highest percent coverage of improved grass were ‘Field By_Wood’ transects, thus the adjacent woodland created an exclusion zone and edge effect in the field transect, probably caused by predator avoidance, contributing to the lower number of bird species (Terraube et al. 2016). At the time of the survey, only one field in one transect was being utilised as pasture containing cattle. However, there were signs of horses which could have caused higher invertebrate prey numbers attracted to any dung, and therefore increased the abundance of corvids (Barnett et al. 2004, Section 7.5.1). Hence, the presence of corvids in the ‘Field By_Wood’ transects (with high percentages of improved grass), potentially also caused a decrease in bird species richness and diversity as a result of other bird species avoiding these predators (Suhonen et al. 1994). Consequently, the MDS in Figure 7.1 and Figure 7.4 show that these transects had reduced numbers of woodland preferring bird species as a result of both high predator density creating an exclusion zone and potential low prey density (Wilson et al. 1999, Donald et al. 2001b, Sheldon et al. 2004).

If the improved grass was used as pasture, then it would be expected that hedges bordering the fields would be maintained, more so than in cereal fields, to enclose livestock (Sparks et al. 1996, Hinsley and Bellamy 2019), which would in turn increase bird diversity (Section 5.5.2). Hinsley and Bellamy (2019) stated that ‘stock-proof’ hedges were dense and wide, providing foraging and nesting habitat. However, grazing can reduce the hedge vegetation creating gaps, and in the 1950s and 1980s hedges were removed for larger machinery and to reduce pests, such as rabbits and Woodpigeons (Hinsley and Bellamy 2019). This was not necessarily the case in the current study as the field transects varied in percent cover of improved grass and proportion of hedge length. Coincidently, the two transects with high percent cover of improved grass had lower proportions of hedge length, possibly as a result of the adoption of fences to replace hedges, thereby further decreasing bird diversity, or as a result of only part of the field being surveyed. Furthermore, Hinsley and Bellamy (2019) noted that intensive grassland management may reduce the positive effects of hedges in pastoral landscapes. In contrast, Wilson et al. (1999) concluded that in tillage-dominated farmland, intensively managed grassland could actually provide an important food source for birds, if managed appropriately. The review by Wilson et al. (2005) showed that structure and composition of the farmed area effects the phenology of bird species differently, therefore there is not one prescription that will benefit all species. Moreover, Perkins et al. (2000) noted that low-intensity cattle grazing over the autumn and winter would provide the heterogeneous sward structure, bare earth and seeding plants required by many bird species, which would maximise the range of bird species able to use improved grass throughout the winter.

Wilson et al. (1999) indicated the complexity of the system and further research should be carried out to evaluate the effects of improved grass on bird diversity. Invertebrate species used to feed young birds are lost, but other invertebrate species may thrive in high density vegetation of managed grassland (Wilson et al. 1999). The result of the current study suggests that less intensive grassland may be more beneficial, however, the edge effect of nearby woodland could negate any benefit. Moreover, semi-natural vegetation areas, such as hedges, grass margins and drainage ditches, must also be increased to improve bird species richness and diversity, while simultaneously providing ecosystem services, such as crop pollination (Whittingham 2011, Batáry et al. 2015), flood defence and, in a more global context, to combat desertification in a warming climate (e.g. Nagendra et al. 2013).

8.5.6 Oak Decline – Cambridgeshire

A national decline in oak as a result of Acute Oak Decline (AOD) has been reported in the UK (Denman et al. 2014) and could be having negative effects on UK bird populations (see Section 4.5.2 for more information on the symptoms and causes of AOD). It can be predicted from the results in Figure 8.7 that an increase in the percent cover of oak in the transects over the Cambridgeshire landscape would support more rare bird species. The highest percent cover of oak in any of the transects in the Cambridgeshire study was 32.8%, this is low when compared with the highest percent cover of ash (70%), indicating an overall lower abundance of oak in the landscape. Reducing cover of oak in the transects from 40% to 20% would almost halve the rarity index and thus suggests the potential for a national decline in oak to have a pronounced effect on rare birds in Cambridgeshire.

AOD has been reported nearby the Cambridgeshire study landscape (Forest Research 2019) and has been recorded in Monks Wood, but symptoms are dependent on environmental conditions (*pers. comm.* R. K. Broughton, 13th December 2019). Mitchell et al. (2019) compiled a case study on the potential effects of oak decline on Monks Wood, and stated that 7 bird species would be affected. The suggested management plan consists of maintaining the contribution of oak at a minimum of 70% (Mitchell et al. 2019), whereas the maximum percent cover of oak in the woodland transects in the current study is only 33% (highest percent composition estimated in the field was 40% oak). This recommendation was for the whole of Monks Wood, but implies that the current percentage of oak in the woodlands in the Cambridgeshire landscape may be too low to prevent deterioration of habitat quality and to provide habitat for oak-associated species. Mitchell et al. (2019) also recommended replanting oak and protecting the saplings from deer grazing to rejuvenate the woodland. Planting small leaved lime in the gaps and alder, in wetter areas, was also advised as this could also be beneficial to oak-associated species (Mitchell et al. 2019).

In the New Forest analysis in Section 8.5.3, AOD was implied to be influencing the birds, whereas the results for the Cambridgeshire landscape showed a direct influence of oak supporting rare bird species, which could be at risk if oak is not maintained (Denman et al. 2010). Caprio et al. (2009) found that specialist bird guilds were influenced by oak biomass and core area, and thus retention of oak is paramount to maintain specialist bird species. The current study also showed that, as is often the case, the rare species were typically specialist bird species, such as the Marsh Tit and Spotted Flycatcher, rather than adaptable generalists (e.g. Fuller et al. 2005b). Moreover, the large confidence intervals in Figure 8.7 suggest uncertainty at higher percentages of oak, indicating that other variables possibly influenced the rarity index (probably in the woodlands), such as the percent cover of understorey vegetation at 2-5 m, present in the model set (Table S 5.19).

The Marsh Tit has been shown to select habitats with a higher proportion of oak in the winter (Broughton et al. 2014b). In the current study, the Marsh Tit occurred in the majority of the woodland transects despite varying amounts of oak, reflecting its use of a range of habitats in the breeding season (Broughton et al. 2012a, Broughton and Hinsley 2015). Caprio et al. (2009) reported that generalist bird species were influenced more by oak core area in the winter, whereas the specialist guilds required oak year round. Broughton et al. (2012a) demonstrated that the Marsh Tit is a woodland interior specialist species and selects breeding habitat with structural complexity and canopy cover, rather than the proportion of a particular tree species. The Spotted Flycatcher also occurred in transects with high percentages of oak in the current study, and has been shown to prefer open woodlands (Cramp et al. 1993, in

Freeman and Crick 2003). However, they also occurred in field transects in the current study, but at lower densities following Cramp et al. (1993, in Freeman and Crick 2003), possibly contributing to the relationship of rarity with scrubby vegetation (at 2-5m). By contrast, the transect with the lowest percentage cover of oak contained the rare and declining Turtle Dove. Pickett and Siriwardena (2011) found that the Turtle Dove avoided farmland with high heterogeneity, suggesting that the preferred areas of scrub for the Turtle Dove should possibly be accompanied by homogeneous cropland of a type favourable for foraging. In the current study, the Turtle Dove occurred in scrubby vegetation bounded on one side by a field and by scrubby, successional vegetation on the other, indicating that at least for Turtle Doves, heterogeneity of semi-natural vegetation in the form of scrubbier areas in the farmland could be beneficial.

This therefore, indicates that multiple bird indices should be evaluated and incorporated into management strategies. Prescriptions for specific bird species within communities should include, but not be limited to, an increase in heterogeneity of semi-natural habitat over the landscape. In order to provide for rare species, a strategy of increasing and maintaining oak woodlands, such as those detailed in Denman et al. (2010) and Mitchell et al. (2019), and also incorporating scrubby habitats is recommended.

8.5.7 Tree Loss due to Tree Disease or Changes in Woodland Management – Cambridgeshire

Decreasing average height of the vegetation was predicted to significantly decrease IRR (Figure 8.8), as a result of the higher relative rarity of birds in the woodlands in Cambridgeshire. The transects with the highest values of IRR had an average height of the vegetation above 15 m, suggesting that the rare bird species preferred tall vegetation, presumably of oak or ash. Tree death creating natural gaps as a result of oak decline (Denman et al. 2014) or ash dieback (Pautasso et al. 2013), or removal of trees by selective felling would reduce the average height of the vegetation over the transect, and influence IRR in these transects. A decrease in average height of only 5 m (from 15 m to 10 m) is predicted to almost halve IRR, indicating that a loss of only a few tall trees could have dramatic effects on IRR locally in the transect, and further indicates the importance of the woodlands in this fragmented agricultural landscape. Yet again, this result does not reflect the relationship with the rare and declining Turtle Dove, as the percent cover of vegetation at 2-5 m was not reliable enough to form a prediction.

In the landscape MDS, average height was positioned almost in the same place as the percent cover of oak (Figure 7.2 and Figure 7.3), whereas on the woodland only MDS (Figure 7.8 and Figure 7.9) average height was almost in the same position as the percent cover of ash, and oak was not significant. Transects with the highest average height contained higher percentages of ash than oak, which was consistent with the tendency for the taller trees in the study woods to be ash, but the lower percent cover of oak in the landscape may have also contributed. This suggests that over the landscape oak was influencing the relative rarity of birds, whereas in the woodlands the ash was contributing more to relative rarity.

Ash dieback first appeared in Poland in the 1990s and in 2012 it was reported in the UK, with recent reports of it in Monks Wood in the Cambridgeshire study site (Pautasso et al. 2013, Mitchell et al. 2014b, *pers. obs.* R. K. Broughton). The results of the current study suggest that ash dieback, as well as oak decline, may affect rare bird species by reducing the numbers of large mature trees, which is reflected locally in the IRR being greater in transects with taller ash (or oak). Furthermore, this result suggests that rare bird species may be able to cope with the loss of either ash or oak, but not both. Mitchell et al. (2014b) recommended oak as one of the 22 possible alternatives to ash, as oak supports all of the bird species associated with ash and 69% of the species from other taxa. This implies that a similar suite of rare bird species would be affected by declines in both ash and oak, supporting the results of the current study. An increase in the prevalence of the diseases effecting oak and ash would result in tree loss, and therefore a reduction in average height, due to death or the removal of infected trees, subsequently reducing IRR and habitat for rare bird species. This indicates that preservation of mature, i.e. taller, trees should be attempted if at all possible, and that clear-felling of mature woodland could have a significant impact on the habitat required by rare species in the Cambridgeshire landscape. It also reinforces the importance of preserving fragmented woodland in intensive agricultural landscapes.

It was difficult to decipher from the MDS analysis the specific bird species that would be affected by changes in average height, due to the clustering of the woodland species on both the landscape and woodland only MDS (Figure 7.1b and

Figure 7.7b). The data show that the transect with the highest average height and highest species rarity contained the Raven and Marsh Tit, indicating that they would be influenced by an alteration in average height of the vegetation. The report by Mitchell et al. (2014a) supports the results from the current study, as they highlighted that rare bird species, such as the Marsh Tit, would be affected by ash dieback (which would reduce the average height of the woodland transect area). They also indicated that the Spotted Flycatcher and Bullfinch would be affected by ash dieback, however, the results of the current study suggest that these species are more likely to be affected by oak decline (Mitchell et al. 2014a). Furthermore, as there are similarities in the bird species occupying both oak and ash, it is conceivable that all of these rare species would be affected by a reduction in oak and/or ash, with the reduction in overall average height of the vegetation that this would entail.

Thus, if possible, measures to prevent further spread of these diseases should be promoted, along with extra effort in maintaining and replanting oak and ash, either as new woodland fragments or re-growth in the decimated woodlands. However, further research is required to test the likelihood of replanted ash and oak surviving, or whether different tree species should be considered. Moreover, Mitchell et al. (2014b) showed that mixtures of tree species could replace diseased ash; they found that 11 tree species, including sycamore, hazel and birch spp., can support 84% of species associated with ash, whereas replacing ash with oak and beech will support 74% of species (Mitchell et al. 2014a). However, they also stated that a mixture of tree species may not be more beneficial than replacing ash with oak, when specifically considering bird species (Mitchell et al. 2014b).

8.6 Recommendations

8.6.1 New Forest Management

The results of the current study recommend a number of additional actions to those stated in the New Forest Management Plan (New Forest National Park Authority 2010), such as including multiple beneficial habitats to increase habitat heterogeneity and overall landscape (gamma) bird diversity. Certain areas of heathland adjacent to the woodland should be allowed to scrub up, which would increase bird diversity, whilst maintaining open heathland areas for rare species, such as the Dartford Warbler. Further research should evaluate the amount of heathland required to maintain these rare and declining species and how much scrub is needed to increase bird species diversity and conservation priority. An edge of a few metres might be sufficient and could be created at the expense of forest rather than heathland, and have a graduated profile from the taller, woodland trees to shorter scrub.

Grazing can continue at acceptable levels to maintain the pasture woodlands, however, rotations are advised to allow growth of the understorey and also the replacement of overstorey trees in the event of diseases, and other factors, such as severe weather, removing the canopy species. Livestock numbers should also be controlled at a predetermined level and exclusions put in place in open areas over the breeding season to avoid excessive grazing and trampling of ground nesting birds (Nelson et al. 2011). Certain woodlands should remain enclosed, as this supports slightly different bird communities and increases bird density.

Creating heterogeneity in the broadleaved woodlands by managing tree age, canopy cover, vegetation species composition, coppice and silviculture practises, should provide habitats for a number of bird species and communities (Begehold et al. 2015). Pure pine plantations should be discouraged and instead minimal amounts of pine should be included in mixed broadleaved-coniferous woodlands to maintain pine specialist bird species, such as the Common Crossbill, and also rarer species, such as the Wood Warbler. Softwood conifer species (such as Douglas fir) can be maintained as small patches differing in silviculture practises in order to maintain rare bird species, such as the Firecrest, and continue to have relatively high bird diversity. Increasing habitat diversity, including woodland structure and composition, and also open areas of heath, scrubland and wetland, will increase habitat heterogeneity over the landscape which will therefore, increase overall (gamma) bird diversity across the New Forest. This follows the findings of Wilcove et al. (1986) who stated that regional protection of areas should encompass multiple habitat types in order to attain variation and preserve diversity over the landscape.

8.6.2 Cambridgeshire Management

The results of the current study show that multiple bird measures, i.e. density, diversity, the number of birds with declining populations, rare birds and bird community composition, should ideally be taken into account before any management strategies can be implemented. Thus, the recommendation from the current study is that overall heterogeneity of both farmed and unfarmed areas be improved and maintained over the agricultural landscape in Cambridgeshire to increase habitat provision for a range of species (Heikkinen et al. 2004, Whittingham 2011, Vickery and Arlettaz 2012).

In the farmed area this should include reducing the amount of improved grass, creating variation in cropping and in field vegetation heights through grazing or cutting. Changes to farmland management, such as reducing autumn tillage to increase over-winter food supplies in stubbles should also aid the population recovery of bird species, including the Lapwing. Semi-natural uncropped areas should be increased and managed by increasing the proportion of hedge length along boundaries, whilst maintaining more natural open areas for specific species, such as the Skylark. Hedge diversity in terms of structure, (height and width) and vegetation composition should also be increased across the landscape to provide habitat for multiple bird species, but again providing that open areas are also maintained. Ultimately, continuing to implement the most effective prescriptions of both AES in the non-cropped areas and AEM (Agri-management Schemes) over the agricultural landscape will benefit bird populations, enhance biodiversity and ecosystem services, such as pollination and bio-control (Whittingham 2011).

The connectivity of woodlands by hedges must be maintained, whilst increasing the proportion of both oak and ash in the landscape (if possible) which will provide habitat for birds in general and rare species in particular. This can be achieved by either increasing the number of woodlands and/or managing and monitoring the condition of the oak and ash trees over the landscape (e.g. Mitchell et al. 2014a, 2019), with further research required into the extent of the declines. The reports by Mitchell et al. (2014a, 2019) suggested replacing the ash and oak with tree species that will provide for the associated species. They suggested replacing ash with oak as the best alternative, despite the potential threats to oak from AOD and other diseases. However, since ash dieback is already prevalent in Monks Wood, and in the surrounding woodlands, following the management plan set out by Mitchell et al. (2019) appears to be the current best option, i.e. to maintain oak and associated species in the woodlands in Cambridgeshire. The average tree height should also be maintained at the higher values associated with rare bird species. However, this will be difficult, therefore long term management should provide alternatives, such as replanting before infected trees are lost and increasing woodland area. Rapid and timely replacement of potentially millions of ash trees is also important in terms of climate change mitigation.

Scrubby vegetation areas should also be increased to enhance bird diversity and provide habitat for declining and rare species, such as the Turtle Dove. Although not obvious in the current study, the results also indicate that AES implementation has created a network of hedges over the landscape, which is sufficient to maintain bird populations in the smaller, isolated woodlands (represented by negative relationships of bird indices with woodland area in Section 5.4.8 and 5.4.9). This suggests that birds are benefiting from AES schemes over the Cambridgeshire landscape. However, improvements to increase habitat quality must be made to multiple habitats, particularly the inclusion of grass and wild flower margins (Batáry et al. 2015), to conserve multiple bird species over the agricultural landscape. Therefore, increasing overall habitat heterogeneity over the landscape is vital to stop further declines of farmland birds. This is in line with Margules and Pressey (2000) who agreed that whole landscapes must be managed for conservation purposes, as reserves alone are only the starting point for nature conservation. Reserves should represent the biodiversity in each landscape, and not be skewed towards specific species or habitats, but rather encapsulate them all and these areas should be separated from processes that threaten diversity (Margules and Pressey 2000). The reserves in the Cambridgeshire landscape are a population source and should be maintained in line with managing the semi-natural habitat over the landscape. Heterogeneity in terms of habitat types (i.e. crop types, grassland, hedgerow diversity) should be maintained within landscapes, with AES implemented at a landscape scale, and also between landscapes with differing management (AEM) as well as landscape types.

8.7 Conclusions

For various reasons the two contrasting landscapes in the New Forest and agricultural Cambridgeshire could still be subject to landscape change, whether as a consequence of human or natural intervention. In both landscapes, vegetation at the 2-5 m height level, as scrub in the New Forest or hedges in Cambridgeshire, are managed and any reduction in the amount available would have a great effect on, albeit, different bird indices and community composition. The number of declining bird species supported in the New Forest and bird rarity in Cambridgeshire would suffer from a loss or reduction in the dominant tree species (beech and oak/average height, respectively), as a result of tree disease or deforestation. Alternatively, the replacement of pine in the New Forest and improved grass in Cambridgeshire would benefit bird species and increase species richness and diversity in both landscapes (and also priority, rarity and IRR in the New Forest). Management recommendations were thus provided for both landscapes with reference to the previous analysis. Overall, management should be aimed at providing habitat for not only the greatest number of species but particularly those with smaller and declining population sizes, by maintaining and promoting landscape heterogeneity and therefore gamma diversity.

9 Comparison of the New Forest and Cambridgeshire

In this section, the results from Sections 4-7 are used to compare and contrast the New Forest landscape with the Cambridgeshire landscape, in terms of habitat types and composition, and the consequent effects on the bird species indices and community composition. The landscapes were chosen as they represent two major and contrasting land-uses and are representative of typical landscapes in southern England. Similar bird species exist in both landscapes, but have different relationships to the habitats and vegetation characteristics. The bird indices were also shown to respond to alternative habitats present in each landscape.

9.1 Landscape Comparison

Landscapes in the New Forest and Cambridgeshire were chosen for this study as they have contrasting land-use management and habitat composition and are typical landscapes of southern England. The New Forest is a National Park on the south coast of England, with managed broadleaved and coniferous forests, and open heath. The contrasting Cambridgeshire landscape is typical English farmland, of mainly arable crops and improved grass with pockets of fragmented, predominantly broadleaved, managed and unmanaged woodlands. The New Forest contains somewhat more ‘natural’ habitat; although the enclosed (“Inclosure”) woodlands are interspersed with managed conifer plantations, the heathland is managed through heather and gorse burning, and there is intensive grazing in the unenclosed (“Open”) forest. The Cambridgeshire landscape is intensively managed agricultural land, which has reduced the amount of natural and semi-natural habitat. The remnants of an ancient woodland are now confined to a number of small woodland patches inter-connected with hedgerows throughout the landscape. The study of these two landscapes allows for comparisons and contrasts of the habitats to be evaluated and related to the bird indices present. The landscapes also differ in terms of land sparing and sharing; with the protected areas of woodland and heathland of the New Forest in one location mostly separate from intensive management, whereas the Cambridgeshire landscape is typical of land sharing with beneficial/semi-natural habitat (woodlands) interspersed throughout the farmed area providing refuges from intensive agriculture (Phalan et al. 2011). The effects of the management practises on the birds in each landscape can also be compared and evaluated.

Vegetation composition varies greatly between the open habitats of the two landscapes. The majority of the open space in the New Forest study site is lowland heath comprising of heather, gorse and grass (heavily grazed), often with encroaching successional vegetation, such as silver birch, forming scrubland. The intensive grazing of the open heathland in the New Forest has enabled rare flora to survive by reducing competition from other species, creating a rare and protected habitat termed *Molinia* meadows (Tubbs 2001, Putman 2010, JNCC 2019). The New Forest Act (1949) was created to reduce and prevent the scrubbing up of the heathland by cutting and burning the vegetation not controlled by grazing, such as gorse, pine and bracken, in order to maintain the condition of the heathland (Smith and Burke 2010). The heather is also cut for fire breaks and was historically baled for the road industry, but this is now being implemented in mire restoration (Smith and Burke 2010). By contrast, the open habitat in Cambridgeshire is agricultural land of predominantly arable crops, such as oil seed rape, winter wheat, beans, and areas of improved grassland. The agricultural landscape contains limited natural or semi-natural habitat which is often hedges (of various dimensions and quality), small copses, single trees, or grass field margins used as public rights of way. The intensification of agriculture has posed numerous threats and challenges to avifauna, continuing to put pressure on already struggling farmland bird populations (e.g. Robinson and Sutherland 2002).

The woodlands in Cambridgeshire are small and fragmented throughout the agricultural landscape, typical of land sharing; the largest was Monks Wood which measured ~170 ha in the current study, and the smallest woodland was Gamsey Wood which only covered ~5 ha. In contrast, the New Forest has a vast forested area extending beyond the survey area of the current study, and covers ~20,000-40,000 ha (Tubbs 2001), representing land sparing. The woodlands in the two landscapes vary in vegetation composition. Whilst oak is present in both areas, the New Forest is predominantly oak and beech woodland, with a number of pine and soft conifer plantations. However, Cambridgeshire is predominantly oak, ash and field maple woodland, with some elm and other broadleaved species in various quantities, and occasionally conifer.

The structure and composition of the understorey also differs between the two landscapes. The New Forest often has a distinct lack of lower vegetation in the unenclosed forest as a result of intensive grazing and dense beech canopy, except for the presence of bracken in more open canopied woodlands (Putman 2010). Grazing in the unenclosed forest prevents growth of typical shrub species, such as hazel and birch, and creates an unusually high abundance of holly (Putman 2010). However, the defensive mechanisms of the holly do not exclude it completely from grazing, as it often lacks foliage on the lower section creating a unique structure more akin to a tree than a holly bush (*pers. obs.* A. Barnes). Furthermore, intensive grazing in the unenclosed forest has also prevented new growth of canopy tree species, therefore resulting in an odd age structure of large trees grown when grazing is suppressed (Putman 2010). By contrast, understorey in the Cambridgeshire woodlands is comprised of a variety of plant species creating a relatively dense, multifaceted understorey, predominantly hazel, hawthorn and blackthorn, but also contains species, such as wild service and elm. Cambridgeshire also contains many more shrubby species on the woodland floor, such as honeysuckle, dog rose and bramble, which are present in the open areas too, and frequently in hedgerows. In the current study, variation in vegetation between the two landscapes resulted in differing vegetation composition variables for the analysis, to capture the dominant vegetation in each site.

The vegetation structural metrics included in the bird-habitat analysis were relatively similar for the two landscapes. However, the proportion of hedge length to transect length (*P_HedgeLen*) was added to the Cambridgeshire analysis as a measure of linear woody features in this agricultural landscape. Moreover, extra variables were calculated for the woodland analysis in Cambridgeshire to consider the effects of fragmentation and isolation on the birds, following the theory of Island Biogeography (MacArthur and Wilson 1967). The height threshold used to separate understorey and overstorey vegetation was kept the same (5 m) in both landscapes, despite the fact that previous studies measured different thresholds in the two landscapes. Evans (2018) calculated that the overstorey height was above 5 m for the New Forest, whereas 8 m was suggested in Monks Wood in Cambridgeshire (Hill and Broughton 2009). This may have been a consequence of the understorey vegetation in Cambridgeshire, such as hawthorn and blackthorn, being reasonably tall as trees rather than shrubs, causing difficulties in the discrepancy between overstorey and understorey vegetation (*pers. obs.* A. Barnes).

As a result of the configuration of the two landscapes, the breeding bird survey method implemented in the two landscapes also differed, from plot-based sampling in the New Forest to line transect sampling in Cambridgeshire. The New Forest was conveniently compartmentalised into small habitat blocks or plots of ~5-15 ha (larger in open heathland) bounded by paths, fences or drainage ditches, often adjacent to opposing habitats, for example blocks of conifer plantations were often interspersed within the expanse of enclosed broadleaved woodland. Access issues in Cambridgeshire resulted in the survey being carried out as line transects confined to public rights of way which consisted mainly of footpaths and tracks along field edges or through woodlands. Line transect sampling was adhered to for the woodland surveys in Cambridgeshire, in order to maintain consistency in sampling method. The survey routes in the smaller woodlands followed the footpaths, which often covered the majority of the woodland, but were still treated as line transects as the whole woodland was not necessarily covered in the survey. Moreover, due to access issues privately owned woodlands could only be surveyed along the woodland edge, if public rights of way were available. Consequently, distance sampling was carried out on the line transect surveys to account for distance decay in detectability, therefore estimates of bird density were calculated to incorporate the unrecorded birds further from the observer and these density estimates were used in the analysis.

The contrasting landscapes allow comparisons to be made and possible management strategies to be inferred from each other. However, caution should be taken when referring to other landscapes, as bird species differ as a result of latitude and altitude as well has habitat type (Wesołowski and Fuller 2012).

9.2 Bird Density, Species Richness and Diversity

As a consequence of the variation in habitat composition, unsurprisingly the values of the bird indices varied between the contrasting landscapes. The highest bird density was in the Cambridgeshire landscape, in ‘Wood Edge’ transect 31 (59.3 birds/ha) which was almost double the highest density in the New Forest (32.3 birds/ha in ‘Oak’ plot 12), possibly as a result of the high bird density often found at the woodland edge (Terraube et al. 2016, Melin et al. 2018). Alternatively, the distance sampling may have artificially increased bird density in an already high-density habitat. The lowest bird density was also in the Cambridgeshire landscape in 'Field By_Wood' transect 21 (3.4 birds/ha), probably as a result of the exclusion zone adjacent to the woodlands, and was slightly lower than the lowest density in the New Forest, in 'Heathland' plot 3 (3.7 birds/ha).

The majority of the 'Scrubland' plots in the New Forest also had low bird density, however, both the second highest species richness and the highest diversity was in 'Scrubland' plot 7 in the New Forest (26 bird species and 3.06 diversity index, respectively). The highest species richness in the New Forest was in 'Beech/Oak' plot 16 with a value of 28 bird species, the same as the highest value of species richness in Cambridgeshire (in 'Wood Edge' transect 27). This indicates that the edge of fragmented woodlands provide habitat for a similar number of species as the protected broadleaved woodlands of the New Forest, further supporting Terraube et al. (2016), and a positive effect of land sharing. Bird species richness and diversity was also relatively high in 'Field_high%_Hedge' transect 16 in Cambridgeshire (27 bird species and 2.93 diversity index, respectively). This was comparable to the 'Scrubland' habitat in the New Forest, as a result of the high proportion of hedge length increasing the amount of scrubby vegetation in 'Field_high%_Hedge' transect 16, and demonstrates the importance of semi-natural vegetation (land sharing) for birds in an agricultural landscape (Tews et al. 2004). Bird diversity was slightly lower in the Cambridgeshire landscape compared with the New Forest, most likely as a result of the detrimental effect of farmland and a negative effect of land sharing. However, the current study demonstrates that a high proportion of hedge in an agricultural landscape may sustain considerable bird diversity, similar to protected broadleaved woodlands in the New Forest, although the bird species contributing to these metrics are more likely to vary.

The lowest measures of species richness and diversity were in the Cambridgeshire landscape in 'Field By_Wood' transects 22 (8 bird species and 1.44 diversity index, respectively) and 21 (8 bird species and 1.78 diversity index, respectively), and were similar to the lowest values in 'Pine' plot 27 in the New Forest (9 bird species and 1.65 diversity index). As a result, Section 4.4.5.1 and 5.4.9.1 of the current study showed that species richness and bird diversity were negatively related to the percent cover of crop type in Cambridgeshire, and the percent cover of pine and other conifer in the New Forest, demonstrating that these habitats were equally poor for bird diversity and species richness. Furthermore, average species richness and diversity in the ‘Field By_Wood’ transects in Cambridgeshire were lower than the averages for the ‘Pine’ plots in the New Forest (average species richness = 11.2 cf. 16.2 bird species, and average diversity = 1.9 cf. 2.3 index, respectively). This indicates that fields adjacent to woodlands were depleted of bird species, and were overall poor bird habitat, another negative effect of land sharing. This may also be due to woodland birds remaining in the woodland, possibly as a result of increased predator abundance at the woodland edge (Suhonen et al. 1994), whereas the ‘Pine’ habitats in the New Forest contained birds encroaching from the surrounding woodlands.

9.3 Species Decline, Priority and Rarity

The bird indices of species decline, priority and rarity were higher in the New Forest than in the Cambridgeshire landscape, suggesting that land sparing supports more rare and declining bird species. Variation in habitats (and geographic location) of the two landscapes also resulted in different declining and rare bird species. The highest value of species decline in the New Forest was in ‘Beech’ plot 10 containing 13 declining species, whilst in Cambridgeshire ‘Field_high%_Hedge’ transects 12, 15 and 16 all had 10 declining species, only 3 less than the New Forest. This possibly suggests equal severity of beech decline in the New Forest and hedgerow decline in Cambridgeshire, most likely as a result of the continued loss of woodland and semi-natural vegetation (Vickery et al. 2004, Jung et al. 2005, Jung 2009).

As previously mentioned, the declining species also differed between the two landscapes; the New Forest contained populations of Tree Pipits, Stonechats, Lapwing and Grey Heron, whereas in Cambridgeshire Turtle Doves and hedge specialist the Yellowhammer were the main declining bird species. The Skylark is another declining species and whilst present in both landscapes, it occurred in 22 transects in Cambridgeshire, whereas it was only present in two plots in the New Forest, presumably indicating a preference for open farmland habitat, therefore providing support for a heterogeneous landscape (Suhonen et al. 1994, Donald et al. 2001b, Vickery and Arlettaz 2012).

The highest species priority was in ‘Scrubland’ plot 7 in the New Forest (20 priority index), and was almost double the highest species priority in Cambridgeshire in ‘Wood Edge’ transect 27 and ‘Field_high%_Hedge’ transect 15 (13 priority index for both). This demonstrates that ‘Scrubland’ in the New Forest had a greater number of bird species with higher percentages of population declines than in the Cambridgeshire landscape, and further suggests that land sparing supports more declining bird species. For example, both landscapes contained Spotted Flycatchers which declined by 41% in the ten years to 2015, however, Stonechats, which were only present in the New Forest, also declined by 41% in the ten years to 2015, increasing the value of the priority metric (Massimino et al. 2017).

The lowest values of species decline and species priority were in the ‘Field By_Wood’ transects in Cambridgeshire (3 for both), and the ‘Pine’ and ‘Other Conifer’ plots in the New Forest (*Spp_Decline* = 3; *Spp_Priority* = 4, in both habitats). This is presumably a consequence of low bird density and diversity in these habitats, resulting in a reduced number of declining bird species, indicating overall poor bird habitat.

The highest value of species rarity was also in ‘Scrubland’ plot 7 in the New Forest (58 rarity index), and was almost double the highest rarity value in ‘Wood Edge’ transect 27 in Cambridgeshire (33 rarity index). Rare habitats in the New Forest have created a unique landscape, which has most likely resulted in an increased number of declining and rare bird species. Consequently, the majority of the New Forest is a SSSI for rare flora and fauna, including Dartford Warblers (and other species not recorded in the current study). However, the Firecrest has a lower population size than the Dartford Warbler in the UK, and is not part of the SSSI designation (Natural England 2019b). This is presumably because the Firecrest is GREEN listed in the UK due to recent increases in population numbers, and is of Least Concern in Europe (Eaton et al. 2015, BirdLife International 2019). The Firecrest northernmost range is in southern England, and therefore are rare in the rest of the UK as a result of their range rather than habitat quality (Batten 1973, Balmer et al. 2013, Clements et al. 2017). Nonetheless, in order to retain populations of Firecrests in the New Forest, suitable habitat must be maintained and protected along with relevant management strategies for Dartford Warblers (Section 4.5.4).

The rarity rating was created based on the UK population size for each species and was used in the analysis of both landscapes for comparison (Section 3.3.6). The Raven had the lowest population size in Cambridgeshire and was given a value of 6, whereas the rarest bird in the New Forest was the Firecrest which had a much lower population size and was assigned a value of 8 (Section 3.3.6 and Appendix A2). However, this possibly suggests that an alternative ranking system should have been implemented to emphasise rarer birds in the Cambridgeshire landscape. Fortunately, the Index of Relative Rarity (IRR) achieves this by only weighting birds present in the landscape based on national population size, local abundance and relevant population size cut-off (Leroy et al. 2012, Leroy 2016). The highest value of IRR in the New Forest was in ‘Other Conifer’ plot 32 (0.49), closely followed by ‘Heathland’ plot 4 (0.45), irrelevant of the low bird density in these habitats. However, Cambridgeshire had the highest value of IRR overall; 0.64 in ‘Wood Interior’ transect 34 and 0.54 in ‘Wood Edge’ transect 23, suggesting that there could be a greater number of rare bird species in the Cambridgeshire landscape than in the New Forest. This could be a result of habitat loss and degradation and an increased number of rare birds occupying smaller areas of fragmented woodland (land sharing). On the other hand, this may actually be as a result of the population size cut-off in Cambridgeshire being more than double the New Forest cut-off (177,500 cf. 68,000 breeding pairs/territories, respectively). This therefore assigned higher rarity weights to birds with larger population sizes, producing a higher proportion of birds classed as rare in Cambridgeshire. As indicated, the species with the highest IRR weighting also differed between the landscapes. The Raven (7000 breeding pairs in the UK), Turtle Dove (14,000 territories) and Yellow Wagtail (15,000 territories) had the highest weightings in Cambridgeshire (Musgrove et al. 2013, Appendix A2). In contrast, the Firecrest, with only 246 breeding pairs in the UK, and the Dartford Warbler, with 3200 breeding pairs, were the rarest species in the New Forest and the rarest species overall (Eaton et al. 2009, Wotton et al. 2009, Appendix A2). As previously mentioned, Firecrests are rare as a result of the small range in the UK and Dartford Warblers are a lowland heath specialist (Bibby and Tubbs 1975, Van den Berg et al. 2001, Balmer et al. 2013). Ravens were present in both habitats, but had a higher weighting in Cambridgeshire due to the higher population size cut-off (Appendix A2). Nonetheless, these results continue to support the requirement for heterogeneous landscapes, regardless of land sharing or sparing, in order to increase IRR and provide and maintain specialist habitat for rare and/or declining birds.

9.4 Bird Species and Community Composition

As a result of the variation in habitats between the two landscapes, along with geographical separation, there was also disparity in the occurrence of certain bird species, and therefore alternative bird species assemblages. For example, the New Forest contained the Redstart, Hawfinch, Crossbill, Tree Pipit and the previously mentioned Firecrest, which were absent in Cambridgeshire. By contrast, Turtle Doves and Yellow Wagtails were present in Cambridgeshire but absent in the New Forest, possibly indicating another advocate for the scrubbing up of some of the heathland in the New Forest to provide habitat for the declining Turtle Dove. The occurrences of these species are in line with the Bird Atlas (Balmer et al. 2013), however, other species, such as Yellowhammers, were only recorded in the Cambridgeshire landscape, but the Bird Atlas indicates that they occur in the New Forest. This disparity may have resulted from the habitats surveyed in the New Forest excluding farmland, and therefore any hedges preferred by the Yellowhammer.

As explained by Hinsley and Gillings (2012) classifying bird species as woodland or farmland birds may be too broad as bird-habitat associations are scale and temporally dependent. In the current study, the Goldfinch displayed alternative habitat associations in the two study areas as a result of landscape composition. In the New Forest, the Goldfinch was predominantly associated with coniferous woodland, whereas in the Cambridgeshire landscape the Goldfinch occurred in agricultural areas and was particularly associated with hedges. The Goldfinch has most likely varied its habitat use as a result of habitat and landscape change, adapting to an alternative habitat in the absence of their preferred habitat (Hinsley and Gillings 2012), as a result of the shortage of conifers in Cambridgeshire and its long history of arable agriculture. Moreover, as with a number of traditionally woodland birds, the Goldfinch may utilise farmland as a result of the diminishing and degrading woodland habitat associated with agricultural intensification, or may be more of a generalist than a coniferous woodland species (Fuller et al. 1995, 2005a).

Spatial variation in bird communities was reviewed by Wesołowski and Fuller (2012) and found that birds adapt to varying conditions throughout their geographic range and bird-habitat relationships are continuously evolving. Often single relationships within a particular habitat were taken to represent the species' preference. However, in the current study this has been shown not to be the case for the Goldfinch in the two landscapes in Britain, and indicates landscape specific habitat preferences (Wesołowski and Fuller 2012). In the current study, the Common Crossbill was also found to be present in two different habitats in the New Forest landscape, conifer and 'Scrubland', possibly suggesting landscape complementation or a preference for multiple habitats. The Crossbill is specific to conifers (Benkman 1993), therefore presence in the 'Scrubland' may have been coincidental and as a result of a transient family group, or suggests that they are, in fact, not entirely limited to conifer woodlands. Furthermore, Fuller (2002, in Wesołowski and Fuller 2012), showed that the Redstart, Bullfinch and Coal Tit depend on conifers in the Bialowieza Forest in Poland, whereas, in Britain these species were present in a wider range of habitats often preferring deciduous woodlands. This is supported by the current study, and further demonstrates geographic variation in bird-habitat preference. Hinsley and Gillings (2012) also stated that understanding species habitat preferences is important for understanding bird ecology, which needs to be defined at a smaller scale and may also change from year to year. This indicates that the birds present in each landscape need to be evaluated to inform management strategies before any drastic changes are made.

Birds may also need multiple habitats for different purposes and require alternative habitats in close proximity (Hinsley and Gillings 2012), which is regarded as landscape complementation, as previously mentioned (Dunning et al. 1992). For example, in the current study Jackdaws were present in both woodland and field transects in Cambridgeshire, presumably indicating that they nest in the woodland, as they are hole nesters (Röell and Bossema 1982), and feed in the fields (Barnett et al. 2004). The increased bird diversity and species richness in the 'Scrubland' habitat in the New Forest, may have resulted from landscape complementation as it is the intermediary habitat between the woodland expanse and the open heathland. Selected felling has also been shown to benefit species, such as the Tree Pipit, by temporarily creating open spaces in forested areas, increasing habitat heterogeneity (Paquet et al. 2006).

The contrasting habitat composition of the open habitats in the two landscapes also resulted in the occurrence of alternative bird species. The Yellowhammer and Whitethroat, for example, were present in Cambridgeshire, but were absent from the New Forest survey, most likely as a result of their preference for farmland and hedges (Bradbury et al. 2000). On the other hand, the Dartford Warbler, Stonechat and Curlew did not occur in Cambridgeshire, most likely as a result of an absence of preferred habitat (Balmer et al. 2013). The Dartford Warbler is limited by habitat range, is specific to lowland heath, preferring heather and gorse with young pine saplings, and is therefore confined to specific habitats in southern England, such as 'Heathland' in the New Forest (Bibby and Tubbs 1975, Van den Berg et al. 2001, Balmer et al. 2013). The absence of Curlew from the Cambridgeshire landscape is probably a result of the reduction in wet habitat, most likely from land drainage schemes, and avoidance of dominating 'edge effects' from surrounding fragmented woodlands (Brown et al. 2015). The Curlew is a declining species and regarded as top conservation priority in the UK (Brown et al. 2015, Massimino et al. 2017). Reduced breeding success as a result of trampling, grazing and predator abundance, has resulted in the decline of the UK's Curlew population, explaining the low density in the New Forest (Franks et al. 2017), and highlighting the importance of providing and maintaining habitat for this and other declining bird species.

The bird community analysis using Multidimensional Scaling (MDS) in Section 0 and 7 clearly shows distinct dissimilarities of the woodland and non-woodland preferring bird species communities in both the New Forest and Cambridgeshire landscapes. The open, non-woodland habitats appeared to be more dissimilar in the New Forest than in the Cambridgeshire landscape, however, this may be due to the smaller sample size of open habitats in the New Forest creating larger distances between habitats on the MDS plots. Nonetheless, the open habitats in the New Forest ('Heathland' and 'Scrubland') were vastly different in terms of bird community composition, whereas the bird communities in the field habitats in Cambridgeshire, with the exception of the 'Field By_Wood' transects, were more similar to one another with no distinct pattern. The 'Field By_Wood' transects in the Cambridgeshire landscape, on the other hand, showed comparable amounts of dissimilarity from the woodland areas, as the New Forest open habitats, but on the opposite axis.

The woodland plots in the New Forest were clustered on the MDS in Figure 6.4 into the relevant habitat classes, whereas the woodland transects in Cambridgeshire were more similar to each other in bird community, most likely as a result of similar vegetation composition (Figure 7.7). The similarity in vegetation composition of the fragmented woodlands is a consequence of the woodlands being remnants of an ancient forest that once covered the whole landscape (Steele and Welch 1973), resulting in similar suites of bird species most likely sourced from the largest wood, Monks Wood (Section 5.5.1). There was only one transect in Cambridgeshire with slightly more conifers than the others (‘Wood Interior’ transect 35), however, this only created slight variation in bird community, but was still similar to ‘Wood Interior’ transect 32 in Monks Wood with no conifers, indicating an alternative reason for the slight dissimilarity (Section 7.5.4).

9.5 Summary

The two study landscapes are only 260 km (160 miles) from one another in the south of England, but are completely different landscapes with alternative habitat composition and configuration, and subsequently bird species indices and bird community composition. The more natural habitats in the New Forest are overall better for birds than the habitats in the agricultural landscape of Cambridgeshire. However, the current study does not propose that the Cambridgeshire landscape should be converted into a landscape similar to the New Forest, as there are species specific to this landscape and region of the UK that also need to be conserved. This therefore demonstrates that individual landscapes have specific management needs for the species present, with scope to improve the habitat to encourage other bird species and increase overall bird diversity.

Equally, the New Forest and Cambridgeshire landscapes represent model landscapes which can be used in further analysis on the effect of landscape change on birds. Moreover, it is also worth noting that bird-habitat relationships fluctuate within species as a result of other factors, such as age, weather, season and climate change (Fuller 2012b), further supporting the requirement of a heterogeneous landscape to provide multiple habitats for multiple species and safeguard against the impact of climate change.

10 Conclusion

Diversity indices are often used as the standard measure to evaluate the health of habitats. However, a key result of the current study shows that diversity should not solely be used as this favours more common species, while rare and declining bird species are often specialists requiring alternative and sometimes unusual habitat. Bird density, species richness and diversity differed between and within habitat classes in each of the two landscapes (Research Question 1.1). Specific habitat characteristics that maximise diversity etc. (Research Question 1.2), and were related to rare and declining bird species (Research Question 1.3), are highlighted in the following sections for each landscape. The results of the current study showed that the highly diverse habitats differed from those that support declining bird species, (particularly in Cambridgeshire) and rare bird species, (particularly in the New Forest) answering Research Question 1.4.

Diversity encompasses numerous common species and yet birds suffering with population declines will continue to decline, and rare birds will become rarer, unless their habitat niches are maintained. Consequently, this supports a recommendation of maintaining habitat heterogeneity over the landscape, which is vital to provide these specific habitats. In order to maximise and enhance overall bird diversity (gamma diversity) heterogeneity should be at the landscape scale, both within and between different landscapes.

The presence of various bird species indicates habitat health, based on the assumption of the presence of the vegetation required for nesting and foraging, and can therefore indicate the health of other taxa, including invertebrates and flora. Hannah et al. (2017) showed that priority bird species, which were determined as endangered and of conservation priority by federal agencies, can be used to indicate the health of the habitat. They used two bird species to show that the occurrence of those species in longleaf pine habitat were correlated with bird species richness, and therefore the two species could be used as indicators to aid with forest restoration (Hannah et al. 2017). In the current study, the priority bird species index was evaluated alongside the rarity index, because a species could show a high percentage of decline, seemingly suggesting that the species is of high priority, but remain relatively common. By contrast, if the population size of a rare bird species was to decrease, the effect would be more significant than the equivalent decrease in a large population. Therefore, the Index of Relative Rarity (IRR) highlights the important rare bird species of conservation concern. The rarity indices evaluate UK population sizes and place importance on the national (UK) population size, whereas species priority was based on population trends in England. Therefore, the results of the current study recommend managing the landscape for multiple rare bird species, as well as those suffering population declines, as a priority in order to enhance bird population numbers, particularly for rare bird species (with smaller population sizes) that are also exhibiting population declines. Furthermore, the state of bird species must be investigated in a wider context, as a species may not be in decline in England or the UK, but may be declining in Europe, for example, or declining in England but not in Scotland, and must be investigated and managed accordingly.

The current study utilises Light Detection and Ranging (LiDAR) data collected from an aircraft flown over the study areas. LiDAR allows measures of the full three dimensional structure of vegetation to be collected more easily and efficiently at larger scales over whole landscapes than traditional field based measures (Clawges et al. 2008, Hill et al. 2014, Coops et al. 2016). Reviews by Tews et al. (2004) and Newton et al. (2009), showed that the main focus of studies using LiDAR has been on woodlands, whereas the current study successfully utilised LiDAR to detail vegetation structure for both woodland and open, non-woodland areas (heathland and agricultural land). For example, the measures of average vegetation height and percent cover of ground vegetation less than 0.5 m over the survey plots/transects detailed the openness of both the wooded and heathland/agricultural habitats. The current study shows that in line with the early studies by MacArthur and MacArthur (1961), bird diversity is related to structural components of the habitat. Rhodes et al. (2015) showed that using remote sensing data in combination with field data from ground-based habitat surveys increases the accuracy of predicting diversity and/or habitat suitability. Moreover, the current study also shows that vegetation composition is equally important for bird-habitat relationships, as demonstrated by the relationships with the bird indices. However, habitat selection can also be species specific as certain birds rely on particular vegetation types (e.g. the Crossbill and conifers), whereas structure is more important for other bird species, such as the Wood Warbler (Huber et al. 2016, 2017). This also varies at different times of the year, for example, Broughton et al. (2012a) showed that the Marsh Tit selects specific vegetation structure in the spring, but selected a greater proportion of oak in their wintering habitat (Broughton et al. 2014a).

The bird-habitat relationships were supplemented through the bird community composition analysis (using MDS) for the two landscapes, which detailed dissimilarities between and within the habitat classes (Research Question 2.1). This showed that bird communities differed between similar habitat classes in the two landscapes, indicating that subtle differences in vegetation structure or composition resulted in discrepancies in the suites of bird species present. This was also used to interpret the bird species and communities affected through changes in selected habitat variables in the prediction analysis. Scenarios were selected based on the habitat variables significantly related to the bird indices included in the multi-model analysis in Sections 4.4.5 and 5.4.94.4.5. Overall, the results of the prediction analysis support heterogeneity in the landscape.

In answer to Research Question 4.3, the combination of best habitats for the New Forest should include; scrubland to increase bird diversity whilst maintaining heathland for the rare Dartford Warbler, continuing to maintain enclosed and unenclosed beech woodlands to support declining bird species, and should also contain conifer species, such as Douglas fir, for the rare Firecrest. The Cambridgeshire landscape, on the other hand, should contain a variety of hedgerows in terms of structure and composition to provide habitat for declining hedge specialists, such as the Yellowhammer, whilst maintaining open areas of unimproved grassland for species such as the Skylark. Additionally, oak and ash woodlands of various sizes (min. 4.7 ha) should also be maintained across the landscape, sufficiently connected with a network of hedgerows to support rare bird populations (Research Question 4.3). Specific conclusions and recommendations are detailed in the following sections for the two contrasting landscapes.

10.1 The New Forest

The results of the current study showed that each habitat class in the New Forest was beneficial to a particular bird index, with the exception of the ‘Pine’ plots, as these were generally poor bird habitat. This was demonstrated in the negative relationships of the percent cover of pine with the majority of the bird indices. The broadleaved woodlands had the highest bird density, which was positively related to the percent cover of oak and understorey (vegetation at 2-5 m), whereas the ‘Scrubland’, although low in bird density, had the highest bird diversity. The univariate analysis showed that the habitat characteristics that maximised species richness and diversity (Research Question 1.2) included a varied vegetation profile, i.e. scrub. An increase in the variation of the vegetation profile was also predicted to increase bird density, and subsequently increase bird diversity, and therefore provide habitat for scrubby bird species, such as the Willow Warbler (Research Question 4.2). This prediction was complex but reliable, but there were no scrubby variables significantly related to bird diversity to accurately predict an effect (Research Question 4.1).

The plots with higher percentage cover of beech were shown to support the highest number of bird species with declining populations (Research Question 1.3), indicating an effect of national beech decline, reflected in the local bird populations. Increasing the percent cover of beech accurately predicted an increase in the number of declining bird species supported in this habitat (Research Question 4.1 and 4.2). In contrast, the two alternative habitats with relatively low bird density, diversity and species richness (‘Heathland’ and ‘Other Conifer’), supported higher numbers of rare bird species (Research Question 1.4), as a result of their specificity to these habitats. This was supported by the positive relationship with the percent cover of heather and other conifers in the multi-model analysis (Research Question 1.3). This was shown on the MDS to be because of the occurrence of the rare Dartford Warbler and the Firecrest respectively; however, accurate predictions were not feasible due to the unreliability of the prediction (Research Question 4.1). In answer to Research Question 1.4, the habitat classes that supported declining and rare bird species differed from highly diverse habitats in the New Forest.

The current study has also shown that each habitat in the New Forest contained a conservation priority bird species, indicating the best or most accommodating plots for these species, whether it be a bird species suffering population declines (*Spp_Decline*, *Spp_Priority*) or a bird species with a low national population size (*Spp_Rarity*, *Spp_IRR*). The priority index in the current study was significantly correlated with bird species richness and diversity, but to a lesser extent than the rarity index (Appendix B5). This suggests that it is possible to manage the landscape for species diversity whilst also managing for conservation priority species. However, due to the calculation of the metrics this may lead to managing more abundant priority species rather than extremely rare species. IRR was not correlated with bird species richness or diversity, indicating that rare birds cannot be managed similarly to increasing bird diversity. Maintaining specialised habitats in the landscape is thus required for the rarest species in the UK, and should be a conservation priority.

The bird community analysis demonstrated differences between and within the habitat classes (Research Question 2.1), with the relevant vegetation composition variables influencing the bird species composition, and therefore, the positioning of the survey plots on the MDS (Research Question 2.2). The vegetation structural variables were also related to the relevant habitat classes, with the variable depicting ground vegetation (<0.5 m) associated with the open habitats, and the variables representing taller vegetation (e.g. *PCov_>5m* and *Ht_Av*) associated with the woodland plots. The percent cover of the vegetation layer at 2-5 m was positioned between the 'Scrubland' and woodland plots, indicating that these habitats contained vegetation in this height range - as understorey in the case of woodlands (Research Question 2.2). The woodland-only MDS showed that the relevant vegetation composition variables were also associated with the habitat classes; additionally, the presence of bracken was also associated with the 'Pine' plots (Research Question 2.2). The structural variables indicated that the 'Pine' plots were more open (*Ostorey_PenDepth*), and the percent cover of vegetation in the 2-5 m layer was associated with the broadleaved plots, indicating more understorey (Research Question 2.2).

The current study recommends that the New Forest National Park Authority should aim to maintain and protect the heathland and ground nesting birds by controlling the intensity of grazing by reducing the amount of livestock and wild roaming species. Exclusions around ground nesting birds or in suitable nesting habitat during the breeding season, or seasonal grazing, should be implemented, in order to prevent trampling and allow ground vegetation to recover, and subsequently provide foraging and nesting habitat for certain bird species (Nelson et al. 2011). At present, the Forestry Commission displays posters around the New Forest to inform dog walkers of the risk that their pets pose to ground nesting birds, however, keeping dogs on leads is not always adhered to. Nonetheless, some bird species prefer grazed habitats, including the Wood Warbler and the Redstart, which were relatively common in the New Forest. Other bird species, such as the Willow Warbler, Blackcap and Garden Warbler, prefer non-grazed scrubby habitats (Donald et al. 1998). This suggests that the New Forest can continue to allow grazing, but less intensively, selectively and seasonally.

Since ‘Scrubland’ was shown to be of great importance to many of the bird indices in the current study, increasing and allowing some areas to ‘scrub up’ with differing levels of successional woodland would encompass multiple bird niches thereby increasing bird diversity (Research Question 4.4). The current study also suggests that birds would benefit from the scrubby areas graduating from taller woodland edges. However, open ‘Heathland’ areas must also be maintained at a predetermined sufficient area to conserve and protect species, such as the rare Dartford Warbler (Donald et al. 1998). Management should also evaluate the effect of pine (trees and saplings), and fire regimes on the Dartford Warbler in order to fully understand and protect the species. Landscape heterogeneity is paramount, and this should include open wet areas providing another specific habitat that will increase overall gamma bird diversity (Research Question 4.3).

Although the broadleaved woodlands were overall better bird habitat, heterogeneous woodland habitats should also be maintained, and could include ornamental conifers. In the current study, the individual homogeneous pine plantations with no understorey and no shrub level vegetation supported very few birds, therefore the amount and size of the plantations should be reduced. Instead, smaller managed pine plots or mixed pine/broadleaved habitats would have less of a negative impact on the bird indices, whilst still providing habitat for conifer specific bird species (Research Question 4.4). Studies in Europe, such as Flade (1994 in Wesołowski et al. 2018), showed that mixed conifer-broadleaved habitats often increased breeding bird species richness and abundance, however, Donald et al. (1998) did not find a difference in western Britain. This possibly suggests maintaining some separation of the broadleaved and conifer habitats, in order to maintain overall gamma bird diversity over the New Forest landscape. Furthermore, the management of the silvicultural practises in the New Forest would benefit from being varied across the landscape, as the current study has shown that both clear-felling and continuous cover felling benefit different bird species.

The practice of silviculture often prevents the maturation of trees, which inhibits ecosystem establishment, and such plantations, depending on the stage of development, can be deficient in bird species. This suggests that the ages of tree species in the New Forest should also be varied, as the young-growth stage and the mature-growth stage have both been shown to accommodate many more bird species than intermediate-growth stages. The current study also demonstrates that plantations of other conifers, such as Douglas fir, in the New Forest constitute a resource for wildlife by supporting rare bird species, such as the Firecrest. This further advocates the implementation of heterogeneity across the landscape to increase overall gamma bird diversity.

10.2 Cambridgeshire

In the Cambridgeshire landscape, the woodlands were overall better bird habitat than the fields, with particularly high values for the majority of the bird indices in the woodland edge transects, as a result of the edge effect. In answer to Research Question 1.2, the woodland variables, such as the percent cover of oak, ash and understorey (2-5 m), were shown to be positively related to bird density, species richness and diversity; therefore, the woodland habitats in this agricultural landscape maximise these indices.

By contrast, the field transects adjacent to the woodlands were generally poor bird habitat (Research Question 1.1) and this was considered to be because of the potential for a high abundance of predators creating an exclusion zone. However, the fields with a high proportion of hedge, although low in bird density, supported more bird species with declining populations (Research Question 1.3), presumably related to historic hedgerow decline influencing bird populations. Moreover, it answers Research Question 1.4, that in agricultural Cambridgeshire bird species with declining populations are supported in a different habitat to the diverse woodland.

Furthermore, when analysed separately, the hedges in the field transects were significantly related to the majority of the bird indices, highlighting their importance (Research Question 1.3). The prediction analysis indicates that increasing the proportion of hedge length to the full length of a transect would increase the majority of the bird indices (Research Question 4.1 and 4.2). This should also be accompanied by increasing hedge diversity in terms of width, height and composition in order to improve the agricultural landscape (Research Question 4.4) by providing heterogeneity in hedge habitat for multiple bird species (Sauerbrei et al. 2017, Hinsley and Bellamy 2019).

AES were not recorded or evaluated in the current study, however, attempts have been made to enhance the hedgerows in certain areas of the Cambridgeshire landscape, for example by planting bird food strips and game cover. Studies have also shown the importance of flower margins, and other types of margins, for birds (Hinsley and Bellamy 2019), which should also be implemented and increased alongside the grassy footpaths and other boundaries within the landscape to improve ecologically poor agricultural land (Research Question 4.4).

The crop type, and also farmland management, was shown to affect the bird indices with the spread of improved grassland proving the most detrimental. Increasing the percent cover of improved grass was accurately predicted to reduce bird species richness and diversity (Research Question 4.1 and 4.2).

In answer to Research Question 4.4 management of the farmland, as well as crop type/land-use, should also be varied over the landscape to improve heterogeneity and provide the multitude of habitats required by multiple bird species. Practises to consider include grazing intensity, timing of tillage and crop sowing, and pesticide use. Grazing, if implemented correctly, can actually aid the foraging abilities of some bird species. Along with maintaining wildflower margins, hedgerows and grass margins will provide food as well as nesting areas for birds and enhance ecosystem services, including pollination and biological pest control. There was also very little, if any, notable set-aside/fallow land over the study area which would also provide habitat for open country bird species, such as the Lapwing (Research Question 4.4).

Woodland area was negatively related to bird diversity and species richness, probably because the interior areas of larger woods tend to have fewer birds/species because vegetation density is low compared to woodland edges, and especially those of small woods. The current study suggests that the largest wood, Monks Wood, is a source providing the smaller woodlands with bird species that might not otherwise be present or persist in small woodlands (e.g. the Marsh Tit), with the hedgerow network providing connectivity between the woods. Furthermore, the prediction analysis shows that these woodland reserves and smaller woods must be conserved and managed to maintain woodland bird populations, and to mitigate the effects of declines in both ash and oak, which will reduce habitat suitability (Research Question 4.1 and 4.2). This will in turn conserve the number of rare birds that are supported in more mature woodlands (Research Question 4.4).

The bird community analysis in section 7 showed that the habitat classes in the Cambridgeshire landscape differed in bird community, with greater dissimilarities between the field transects than the woodland transects (Research Question 2.1). The vegetation composition variables showed that the crop type (e.g. *PCov_Cereal*) separated the field transects from one another and from the woodland transects, with the relevant woodland variables (i.e. taller vegetation) associated with the woodland transects (Research Question 2.2). The separated field-only MDS showed that the bird community composition in the field transects was influenced by only two significant habitat variables: the percent cover of improved grass and the proportion of hedge length. The vegetation composition (particularly the percent cover of ash, other broadleaved species, and field maple), and the vegetation structural variables (separating scrubby woodlands from taller woodlands), were responsible for the ordination and dissimilarities in woodland only bird community composition (Research Question 2.2).

10.3 Comparison

Comparisons were outlined between the New Forest and Cambridgeshire landscapes in the discussion Section 11.1. Both landscapes contained woodland and non-woodland areas, which differed in vegetation composition and structure. Consequently, the bird indices and specific bird species composition also differed between the two landscapes (Research Question 3.1). Nevertheless, the open areas in both landscapes (heathland/scrubland and agricultural land) had lower bird density than the corresponding woodlands, although the ‘Scrubland’ habitats in the New Forest had the highest bird diversity of any habitat. Overall, species decline, priority and rarity were higher in the New Forest than in the Cambridgeshire habitats. IRR, on the other hand, was higher in the woodland habitats in Cambridgeshire than the ‘Heathland’ and ‘Other Conifers’ habitats in the New Forest, but this could be an effect of the higher population size cut-off.

Furthermore, the bird species responsible for the values of the metrics differed between the two landscapes (Research Question 3.3). For example, the New Forest contained the Redstart, Tree Pipit, and rare species, such as the Firecrest, Hawfinch, Common Crossbill and Dartford Warbler, which were absent in the Cambridgeshire study area. By contrast, the Rook, Whitethroat, the declining Turtle Dove, and farmland species such as the rare Yellow Wagtail and the Yellowhammer, were present in Cambridgeshire but absent in the New Forest (Research Question 3.3).

A common theme between these two landscapes (Research Question 3.2) is that, overall, the woodlands were more beneficial for birds, particularly the broadleaved woodlands in the New Forest and the woodland edge habitats (also broadleaved) in Cambridgeshire. However, the specific or unusual habitats, such as the ‘Heathland’ and ‘Other Conifers’ in the New Forest, and the hedges in Cambridgeshire, supported more declining or rare bird species than the diverse woodland habitats (Research Question 1.4 and 3.2).

10.4 Limitations and Reflections of the Study

As a result of the difficulties of scale in landscape studies, most have concentrated on individual habitats or bird species. The current study used a range of bird indices to evaluate habitat health and bird-habitat relationships, and the scope of the current study created measures that encapsulated the majority of the bird species present, eliminating the need to study every single species. Bird density, species richness and diversity were used to measure the overall complement of bird species present. The numbers of birds with declining populations, of conservation priority and rarity were also measured in order to include bird species that require significantly greater attention and to evaluate the reliability of the diversity metrics. As the measures covered multiple bird species, considerable interpretation of the results was required and the bird community composition analysis, using Multidimensional Scaling (MDS) aided this interpretation. However, the results of certain species were often masked and the metrics favoured more common and generalist bird species, particularly the measures based on bird density and diversity. The bird species included in the analysis also had different habitat relationships which posed difficulties when interpreting the results and relating them to individual species.

The current study consisted of a plot-based analysis with the variables averaged over the plot area, however, another method could have analysed the individual registrations of the birds and the habitat variables in a predetermined area surrounding their locations. This latter approach may have been less reliable as the individual may not have been utilising that area, but only passing through. However, the methods used recorded the activity of the birds, thus only birds utilising this area (e.g. for feeding or nesting, alarm calling/singing) were included in the analysis. Territory mapping (e.g. Broughton et al. 2012a) collates bird registrations from multiple visits over a period of time to determine if an area is a breeding territory of a particular species or individual and not simply used by chance or sporadically. However, due to time constraints, the current study had bird observation data for only two visits in one year at each study site. Furthermore, the location of a bird in the survey area could not necessarily be located exactly as often the observation was only audible. Therefore, the average habitat variables over the survey area and the maximum bird count (density) of the repeated surveys were used in the current analysis. This is also the reason that raptors and birds flying over were not included in the analysis because it could not be guaranteed that the individual birds were utilising the habitat patch/area due to their large ranges. In addition, some bird species known to occur in the survey areas were not encountered by the surveyor. For example, the Lesser Spotted Woodpecker (*Dryobates minor*) has been recorded in the New Forest, but unless targeted in surveys is extremely illusive and difficult to record. Other uncommon, quiet and/or inconspicuous bird species may have occurred in the survey areas and were not recorded, however, this would have been the same in all surveys as they were carried out by the same person, maintaining consistency.

The small sample size (i.e. number of habitat plots/transects) most likely constrained the statistical reliability of the relationships, while the data in Figure 4.4 and Figure 5.6 show quite obvious relationships. However, due to time constraints, 32-38 habitat areas (plots/transects) were surveyed in each landscape. If time allows, it is suggested that more survey areas for each habitat class be surveyed, and possibly include more habitat classes in order to gain a better overall representation of the landscape. Sample size was also constrained by one observer carrying out and repeating the surveys over the breeding season (April-June). Only one observer was used in order to reduce bias in both identification and distance, although observer effort and bias of multiple observers could have been accounted for in the modelling. Increasing the number of survey areas would provide for more reliable statistical analysis, however, it was suggested that ~400 survey areas would be required to gain complete statistical reliability and this was unachievable (*pers. comm.* R. Stafford, 16th December 2016). The analysis would also benefit from having an even number of plots and transects of each habitat class. In the New Forest, this imbalance was caused by re-classifying the survey areas once the breeding bird surveys had commenced, due to the unreliability of the tree species map. The vegetation surveys were required to be outside the breeding season, and were therefore completed after the bird surveys due to commencing the research study in January. This inhibited extra habitat plots/transects from being surveyed in the same year because at that point the habitat classification was unknown. In hindsight, the transects in Cambridgeshire should have had a greater distance between them following Bibby et al. (1992). However, this arose due to the transects that were contiguous with the same woodland being cut in half to prevent the transect being too long.

The LiDAR data acquired for this project provided measures of vegetation structure over a landscape scale, which has been shown to be more time and cost effective than field-based analyses. However, as the LiDAR data were acquired prior to commencing the study there are discrepancies in the vegetation structure information. This was shown to be irrelevant for woodlands as unless felled the vegetation structure is relatively stable (Broughton et al. 2012a, 2012b). However, this mis-match caused discrepancies in evaluating bird-habitat relationships in the crop vegetation, as this would almost certainly have changed from the time the LiDAR was flown. Furthermore, the LiDAR in Cambridgeshire was flown in September, whereas the bird data were collected in the spring (April-June). In both time periods, there may not have been much crop present, as it would either be growing or have been harvested, possibly negating the relationships with the crop vegetation in this study, and therefore reducing the mis-match of timing.

The Cambridgeshire analysis contained a measure of linear vegetation features, to incorporate hedges into the analysis, using the LiDAR. The habitat variables depicting the percent cover of vegetation at the various height intervals were also used to capture the woody vegetation in the farmed areas (hedges or trees). However, the height intervals were not found to be significant as a result of the woody vegetation height range overlapping several intervals. The proportion of hedge length was found to be significant in the analysis, particularly in the field only analysis, and for the bird indices this was sufficient for effective analysis. However, Sauerbrei et al. (2017) showed that bird species also respond to various hedge structural diversity measures, which were not included in the current study, but should be incorporated into management plans to provide habitat for a range of bird species and enhance overall landscape bird diversity. Furthermore, future research on individual bird species should include more detailed hedge metrics. However, for the current analysis, as the bird indices included multiple bird species this would most likely not have been significant. Another LiDAR variable, the height Vertical Distribution Ratio (VDR) was used in the current study to depict the structural variation in the vegetation profile. However, the interpretation of this variable is complex, therefore in hindsight a metric detailing Foliage Height Diversity (FHD) would have been more appropriate to allow comparison of the structural diversity of habitats and interpret the relationships with the bird indices (MacArthur and MacArthur 1961).

The extra woodland variables *%_Woods_500m* and *%_Woods_1km* calculated the area of other woodlands in the surrounding 500 m and 1 km, respectively, as a proportion of the survey woodland area. There were few significant relationships with these variables, possibly suggesting that the cover of woodland as a percentage of the buffer area may have provided more useful results. However, the variables used may indicate that larger woodlands, such as Monks Wood, may not require a high proportion of surrounding woodland to sustain its bird populations.

The Cambridgeshire analysis also included Distance Sampling of the data collected over the survey transects to account for distance decay of detectability and consequent failure to record birds further from the observer. However, this may have artificially increased the bird densities in already high density areas, thereby altering the habitat relationships. This suggests that in this instance a relevant cut-off/truncation of the survey width to that with a reliable probability of detection (i.e. 70-90%) may have been a more suitable option considering the low number of registrations for each bird species. This resulted in the birds being grouped according to detectability, which may have wrongly influenced the corrected density, if rarer birds were grouped with more common birds, for example.

Avian populations have been in decline in the UK since the 1970s through agricultural intensification, industrial development and changing land-management (Batáry et al. 2015). Farmland birds have suffered the greatest declines and the scope of the current study may warrant further longer-term analysis into the state of the UK's birds (Hayhow et al. 2015, 2017). The current study was conducted as a snapshot in time with only one year of bird data at each study site. Therefore, it was sensible to make use of the ten year bird trends from 2005 to 2015, as it was more relevant than the bird trends over 40+ years. However, the ten year trend used in this study does not necessarily capture or portray the true state of bird populations, as a number of species were shown to be in decline when overall the population has been stable or increasing (such as Green Woodpecker). Other studies have carried out longer-term analysis on particular species and the BTO, JNCC and the RSPB continue to monitor population trends in the UK (Massimino et al. 2017). Nonetheless, the current system of acting when a species population is critically low may need to be revised, in order to mediate population declines before they are irreversible and while species are still numerous enough to be studied effectively.

Other variables, such as climate, weather, timing, movement, etc. will also be affecting the occurrences of birds in the habitats. However, these data were not available for the current study and by repeating the surveys and only surveying in ideal weather conditions a number of these factors were accounted for, in order to focus on the relationships with habitat structure and composition. Birds alter their habitat preferences in changing conditions and many can adapt at least to some extent. However, it is the habitat specialist species that are less likely to adapt and are therefore vulnerable to decline, and to the subsequent effects of small population sizes (inbreeding, stochastic extinction), necessitating intervention to conserve their habitats and futures. These species may also be affected by interspecific competition in the habitat. However, due to the scope of the current study focussing on bird-habitat relationships, the small sample size and the difficulties in assessing interspecific competition, particularly at a landscape scale, this was not analysed.

10.5 Future Research

The bird surveys in the current study were carried out in the breeding season, so the bird-habitat relationships are in terms of breeding habitat preference. Fledging success and the condition of breeding habitats have been shown to be vital for many bird species' survival (Roodbergen et al. 2012, Brown et al. 2015). Studies have also shown that a number of species have alternative habitat preferences in the winter or migrate to over-wintering grounds in different continents (Morrison et al.

2013, Broughton et al. 2014a, Vickery et al. 2014, Buchanan et al. 2018). Therefore, increasing breeding habitat quality will not necessarily improve populations of birds affected by winter conditions (over wintering survival), or on migration (Fuller and Robles 2018). This indicates that the habitats used in different seasons in the UK should also be reviewed when making drastic conservation management decisions, and habitat resources required throughout the year should be provided at a landscape-scale (Fuller and Robles 2018). As for migrant birds, more research is being carried out on the condition of the over-wintering grounds, for instance in sub-Saharan Africa, in order to investigate causes of population declines (e.g. Morrison et al. 2013). However, the research is bird species specific suggesting that IRR could also be calculated on the wintering sites and bird community analyses should also be carried out to gain further insight into bird-habitat relationships at different times of the year.

The current study shows the value of scrubland habitat in the New Forest for biodiversity, and suggests that allowing some of the heathland to scrub up would be beneficial to a number of bird species. However, further research needs to be carried out to establish the amount of open heathland required by the rare Dartford Warbler, and how much scrub is needed to increase bird species diversity and the numbers of priority species.

Further work on bird-habitat preference in open heathland should also incorporate a detailed study of the heights of the ground vegetation, possibly using remote sensing techniques with a higher resolution, acquired with ground or drone laser equipment, for example. Furthermore, management of conifer in the New Forest should also be monitored to evaluate whether smaller plantation plots or mixed woodland plots are more beneficial at maintaining bird diversity. Further analysis of the bird community in the open habitats (heathland/fields) may be required, considering factors such as proximity of woodland or copse, surrounding land-use or urban areas, the climate or weather, and geographic location, before any drastic changes are implemented. The positive results of the New Forest in the current study could also be used to guide management in other areas, however, caution is advised as the results are landscape specific.

The theories of Island Biogeography and of Metapopulations have been mentioned in the current study and further work is required in order to evaluate the effects of the fragmentation and connectivity of the woodlands on the birds before any drastic changes to the woodlands are implemented. The Cambridgeshire landscape could also be used to influence management strategies in other arable landscapes seeking to enhance the connectivity of woodland habitats. However, more work needs to be carried out over the woodlands as a whole in order to evaluate the minimum area required to sustain viable bird populations (e.g. as in Bellamy et al. 1996, Hinsley et al. 1996a). Research into the effect of crop type on bird community composition, and also implementation of more scrubby areas, with particular interest in the occurrence of the Turtle Dove, should also be carried out in the agricultural landscape in Cambridgeshire.

Care must also be taken when creating new habitat and should consist of native vegetation that can sustain natural populations throughout the food chain. Studies on invertebrate prey are also required as the current study infers good habitat, by implying that the relevant prey is available to the birds. However, this is possibly limiting in some habitats and could be the reason for lower bird densities in certain habitats. Further work on the effects of competition (inter- and intra-specific) and predation may be required as this will also be influencing the birds present in certain habitats and may explain some discrepancies (Fuller 2012c).

As previously mentioned, no obvious signs of AES were observed in the current study, therefore, further work is needed to evaluate the effectiveness of AES in the context of the Cambridgeshire landscape. This could be carried out by observations or tagging birds and recording movements over the landscape (Siriwardena et al. 2006), for example, but this would require multiple observers and a considerable amount of time and resources. AES and AEM are practical incentives to improve and enhance agricultural areas for birds, which should continue to be implemented concentrating on the most effective prescriptions rather than the most easily applied. For example, Walker et al. (2018) noted that the AES must also provide habitat in the winter and should be able to withstand winter weather. Furthermore, research needs to be carried out into the extent of the area of habitat required for either each bird species or to allow multiple habitats to coexist without detriment to one another.

Regardless of the way it is implemented, habitat for as many species as possible must be maintained over the landscape in order to prevent further bird population declines. Heterogeneity is important as what may be suitable of one species or taxa may be detrimental to another, for example, clearing the forest floor in the New Forest for flowering plants and butterflies would reduce nesting vegetation for bird species, such as the Wren, and cover for small mammals. Further research attention is particularly needed on declining farmland birds with ways of implementing a variety of suitable habitat in and around the farmed area, and increasing heterogeneity between fields, not only in terms of crop type but also management. Farmers and land owners need to work together with scientists to apply the most suitable strategies for the species present in the landscapes concerned. The economic costs to such measures and also individual landscapes should be analysed, but healthy bird populations will also provide valuable ecosystem services such as biological pest control and seed dispersal and enhance the aesthetic and recreational value of the landscape. Any strategies implemented must also be monitored for a substantial period afterwards, in order to evaluate the effectiveness of any scheme/practise. Furthermore, the listing of priority (RED, AMBER or GREEN) may need to be evaluated further as birds with declining populations, such as the Linnet, may have decreased from a high population size to a lower, but still relatively large, population, whereas rarer species should be monitored consistently as any declines could wipe out the population in the UK.

10.6 Concluding Remarks

This study is novel as it used an integrated approach with remotely sensed LiDAR data and field acquired vegetation and bird data to investigate bird-habitat relationships at a landscape scale, over two contrasting landscapes to allow for a comparison. The current study used a range of bird indices to evaluate habitat health and bird-habitat relationships, and created measures that encapsulated the majority of the bird species present, eliminating the need to study every single species.

The two landscapes consisted of alternative habitats and therefore bird species, but analysis of each landscape has the same concluding results. The first key result proves that biodiversity should not be used alone to represent habitats, as this masks habitat specific specialists that are more often rare and/or in decline. Habitats can also have the same diversity, or any other index value, but contain a different suite of bird species. Therefore, these multiple measures of the bird indices and bird community composition must all be taken into account before, during, and after any management strategies are implemented. The second key result is that in order to provide specialist habitat and also increase diversity, multiple habitats must be maintained across the landscape. Furthermore, heterogeneity must also be maintained between landscapes, in order to provide a variety of habitats to sustain numerous populations of birds and consequently other taxa. Immediate specific management strategies should be implemented for species with low and declining UK populations, with possible broader implications, for example across Europe, and further work should aim to incorporate a range of bird indices over the landscape.

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Appendix A

Appendix A1. The two letter bird species code and common names for BBS/CBC surveys provided by the BTO.

BTO SPECIES CODES

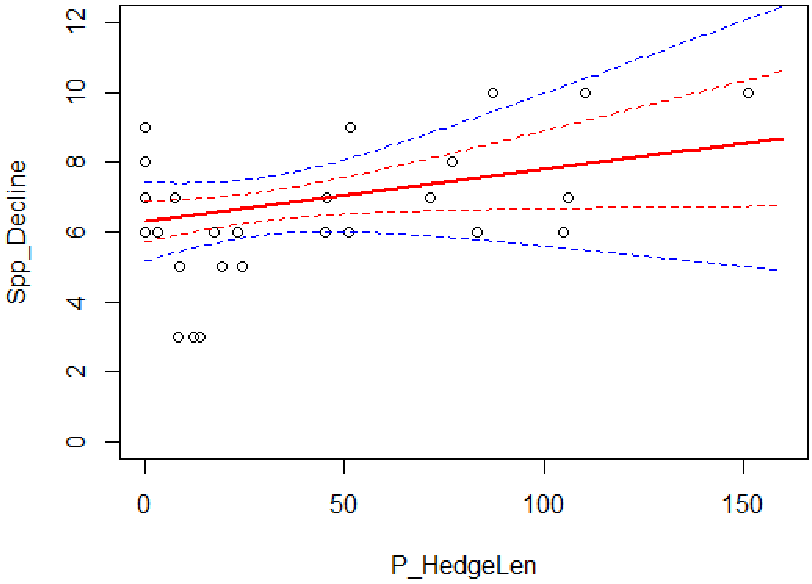
AC	Arctic Skua
AE	Arctic Tern
AV	Avocet
BO	Barn Owl
BY	Barnacle Goose
BA	Bar-tailed Godwit
BR	Bearded Tit
BS	Berwick's Swan
BI	Bittern
BK	Black Grouse
TY	Black Guillemot
BX	Black Redstart
BJ	Black Tern
B.	Blackbird
BC	Blackcap
BH	Black-headed Gull
BN	Black-necked Grebe
BW	Black-tailed Godwit
BV	Black-throated Diver
BT	Blue Tit
BU	Bluethroat
BL	Brambling
BG	Brent Goose
BF	Bullfinch
BZ	Buzzard
CG	Canada Goose
CP	Capercaillie
C.	Carrion Crow
CW	Cetti's Warbler
CH	Chaffinch
CC	Chiffchaff
CF	Chough
CL	Cirl Bunting
CT	Coal Tit
CD	Collared Dove
CM	Common Gull
CS	Common Sandpiper
CX	Common Scoter
CN	Common Tern
CO	Coot
CA	Cormorant
CB	Corn Bunting
CE	Corncrake
CI	Crested Tit
CR	Crossbill (Common)
CK	Cuckoo
CU	Curlew
DW	Dartford Warbler
DI	Dipper
DO	Dotterel
DN	Dunlin
D.	Dunnock
EG	Egyptian Goose
E.	Eider
FP	Feral Pigeon
ZL	Feral/hybrid goose
ZF	Feral/hybrid mallard type
FF	Fieldfare
FC	Firecrest
F.	Fulmar
GA	Gadwall
GX	Gannet
GW	Garden Warbler
GY	Garganey
GC	Goldcrest
EA	Golden Eagle
OL	Golden Oriole
GF	Golden Pheasant
GP	Golden Plover
GN	Goldeneye
GO	Goldfinch
GD	Goosander
GI	Goshawk
GH	Grasshopper Warbler
GB	Great Black-backed Gull
GG	Great Crested Grebe
ND	Great Northern Diver
NX	Great Skua
GS	Great Spotted Woodpecker
GT	Great Tit
GE	Green Sandpiper
G.	Green Woodpecker
GR	Greenfinch
GK	Greenshank
H.	Grey Heron
P.	Grey Partridge
GV	Grey Plover
GL	Grey Wagtail
GJ	Greylag Goose
GU	Guillemot
FW	Guineafowl (Helmeted)
HF	Hawfinch
HH	Hen Harrier
HG	Herring Gull
HY	Hobby
HZ	Honey Buzzard
HC	Hooded Crow
HP	Hoopoe
HM	House Martin
HS	House Sparrow
JD	Jackdaw
J.	Jay
K.	Kestrel
KF	Kingfisher
KI	Kittiwake
KN	Knot
LM	Lady Amherst's Pheasant
LA	Lapland Bunting
L.	Lapwing
TL	Leach's Petrel
LB	Lesser Black-backed Gull
LS	Lesser Spotted Woodpecker
LW	Lesser Whitethroat
LI	Linnet
ET	Little Egret
LG	Little Grebe
LU	Little Gull
LO	Little Owl
LP	Little Ringed Plover
AF	Little Tern
LE	Long-eared Owl
LT	Long-tailed Tit
MG	Magpie
MA	Mallard
MN	Mandarin Duck
MX	Manx Shearwater
MR	Marsh Harrier
MT	Marsh Tit
MW	Marsh Warbler
MP	Meadow Pipit
MU	Mediterranean Gull
ML	Merlin
M.	Mistle Thrush
MO	Montagu's Harrier
MH	Moorhen
MS	Mute Swan
N.	Nightingale
NJ	Nightjar
NH	Nuthatch
OP	Osprey
OC	Oystercatcher
PX	Peafowl/Peacock
PE	Peregrine
PH	Pheasant
PF	Pied Flycatcher
PW	Pied Wagtail
PG	Pink-footed Goose
PT	Pintail
PO	Pochard
PM	Ptarmigan
PU	Puffin
PS	Purple Sandpiper
Q.	Quail
RN	Raven
RA	Razorbill
RG	Red Grouse
KT	Red Kite
ED	Red-backed Shrike
RM	Red-breasted Merganser
RQ	Red-crested Pochard
FV	Red-footed Falcon
RL	Red-legged Partridge
NK	Red-necked Phalarope
LR	Redpoll (Lesser)
RK	Redshank
RT	Redstart
RH	Red-throated Diver
RE	Redwing
RB	Reed Bunting
RW	Reed Warbler
RZ	Ring Ouzel
RP	Ringed Plover
RI	Ring-necked Parakeet
R.	Robin
DV	Rock Dove (not feral)
RC	Rock Pipit
RO	Rook
RS	Roseate Tern
RY	Ruddy Duck
RU	Ruff
SM	Sand Martin
SS	Sanderling
TE	Sandwich Tern
VI	Savi's Warbler
SQ	Scarlet Rosefinch
SP	Scaup
CY	Scottish Crossbill
SW	Sedge Warbler
NS	Serin
SA	Shag
SU	Shelduck
SX	Shorelark
SE	Short-eared Owl
SV	Shoveler
SK	Siskin
S.	Skylark
SZ	Slavonian Grebe
SN	Snipe
SB	Snow Bunting
ST	Song Thrush
SH	Sparrowhawk
AK	Spotted Crake
SF	Spotted Flycatcher
DR	Spotted Redshank
SG	Starling
SD	Stock Dove
SC	Stonechat
TN	Stone-curlew
TM	Storm Petrel
SL	Swallow
SI	Swift
TO	Tawny Owl
T.	Teal
TK	Temminck's Stint
TP	Tree Pipit
TS	Tree Sparrow
TC	Treecreeper
TU	Tufted Duck
TT	Turnstone
TD	Turtle Dove
TW	Twite
WA	Water Rail
W.	Wheatear
WM	Whimbrel
WC	Whinchat
WG	White-fronted Goose
WH	Whitethroat
WS	Whooper Swan
WN	Wigeon
WT	Willow Tit
WW	Willow Warbler
OD	Wood Sandpiper
WO	Wood Warbler
WK	Woodcock
WL	Woodlark
WP	Woodpigeon
WR	Wren
WY	Wryneck
YW	Yellow Wagtail
Y.	Yellowhammer

Appendix A2. Population trends in England, UK population sizes, and the priority, rarity and IRR weighting for each bird species.

Bird Species	Population Change 2005-2015 (%)	Priority Weighting (0-5)	Population Number	Rarity Weighting (0-8)	IRR Weighting New Forest	IRR Weighting Cambs
Blackbird	-1	1	4.9m pairs	0	0	0
Blackcap	55	0	1.1m terr	0	0	0
Blue Tit	-9	1	3.4m terr	0	0	0
Bullfinch	20	0	190 000 terr	3	0	0.038
Carrion Crow	7	0	1m terr	0	0	0
Chaffinch	-18	1	5.8m terr	0	0	0
Chiffchaff	52	0	1.1m terr	0	0	0
Coal Tit	6	0	680 000 terr	1	0	0
Collared Dove	-23	2	980 000 pairs	1	0	0
Common Crossbill	10	0	39 000 pairs	5	0.229	NA
Cuckoo	-40	2	15000 pairs	5	0.614	0.907
Curlew	-14	1	66 000 pairs	4	0.057	NA
Dartford Warbler	1	0	3200 pairs	6	0.913	NA
Dunnock	2	0	2.3m terr	0	0	0
Firecrest	8	0	246.6 total pairs	8	1	NA
Garden Warbler	-11	1	170 000 terr	3	0	0.06
Goldcrest	7	0	520 000 terr	1	0	0
Goldfinch	76	0	1.2m pairs	0	0	0
Great Spotted Woodpecker	4	0	140 000 pairs	3	0	0.115
Great Tit	-6	1	2.5m terr	0	0	0
Green Woodpecker	-8	1	52 000 pairs	4	0.122	0.547
Greenfinch	-58	3	1.7m pairs	0	0	0
Grey Heron	-28	2	12 000 pairs	5	0.683	NA
Grey Wagtail	-24	2	35 000 pairs	5	0.275	NA
Hawfinch	-19	1	800 pairs	7	0.983	NA
Jackdaw	34	0	1.3m pairs	0	0	0
Jay	5	0	170 000 terr	3	0	0.06
Lapwing	-31	2	130 000 pairs	3	0.001	NA
Lesser Whitethroat	26	0	74 000 terr	4	0.036	NA
Linnet	5	0	410 000 terr	2	0	0
Little Egret	58	0	700 pairs	7	0.986	NA
Long-Tailed Tit	9	0	330 000 terr	2	0	0.001
Magpie	2	0	550 000 terr	1	0	0

Bird Species (contd.)	Population Change 2005-2015 (%)	Priority Weighting (0-5)	Population Number	Rarity Weighting (0-8)	IRR Weighting New Forest	IRR Weighting Cambs
Mallard	0	0	100 000 pairs	4	0.006	NA
Marsh Tit	-31	2	41 000 terr	4	0.209	0.642
Meadow Pipit	0	0	1.9m pairs	0	0	NA
Mistle Thrush	-29	2	160 000 terr	3	0	0.075
Nuthatch	29	0	220 000 terr	3	0	0.018
Pheasant	-4	1	2.2m females	0	0	NA
Pied Wagtail	-11	1	460 000 pairs	2	0	0
Raven	24	0	7000 pairs	6	0.809	1
Red-Legged Partridge	-18	1	82 000 terr	4		NA
Redstart	48	0	100 000 pairs	4	0.006	NA
Reed Bunting	18	0	230 000 terr	3	0	0.014
Robin	8	0	6m terr	0	0	0
Rook	-11	1	990 000 pairs	1	NA	0
Siskin	102	0	410 000 pairs	2	0	NA
Skylark	-11	1	1.4m terr	0	0	0
Snipe	2	0	76 000 pairs	4	0.032	NA
Song Thrush	0	0	1.1m terr	0	0	0
Spotted Flycatcher	-41	3	33 000 terr	5	0.3	0.717
Starling	-38	2	1.8m Pairs	0	0	0
Stock Dove	16	0	260 000 terr	2	0	0.006
Stonechat	-41	3	56 000 pairs	4	0.099	NA
Tree Pipit	-19	1	88 000 pairs	4	0.015	NA
Treecreeper	10	0	180 000 terr	3	0	0.048
Turtle Dove	-87	5	14 000 terr	5	NA	0.918
Whitethroat	15	0	1.1m terr	0	NA	0
Willow Warbler	-15	1	2.2m terr	0	0	0
Wood Warbler	-8	1	6500 males	6	0.822	NA
Woodpigeon	6	0	5.3m pairs	0	0	0
Wren	11	0	7.7m terr	0	0	0
Yellow Wagtail	0	0	15 000 terr	5	NA	0.907
Yellowhammer	-12	1	700 000 terr	1	NA	0

Appendix B



Appendix B1. Predicting the effect of increasing P_HedgeLen on Spp_Decline over the landscape. The open circles represent the actual data points, the solid red line is the prediction (with the mean of the other variables in the model remaining constant), the red dashed lines are the standard error and the blue dashed lines are 95% confidence intervals (R version 3.5.2).