

BIRD DIVERSITY OF AFFORESTATION HABITATS IN IRELAND: CURRENT TRENDS AND LIKELY IMPACTS

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ABSTRACT

A programme of afforestation has increased forest cover in Ireland from around 1% to more than 10% in less than a century. This study describes planting trends up to 2005 according to habitat and characterises the bird assemblages of the most commonly afforested habitat types. Most afforestation has taken place on peats, gleys [typical of wet grassland (WG) habitats] and well-drained mineral soils [typical of improved grassland (IG) habitats]. Since 1986, afforestation of gley sites increased dramatically, and afforestation rates on both peats and gleys were more than two times greater than on well-drained mineral soils. Bird diversity was assessed at 27 peatland, WG and IG sites. Bird density and species richness were positively related to shrub and tree cover, which was highest in WG and lowest in peatland sites. Indicator analysis identified several characteristic bird species of WG and peatland, but no characteristic species were identified for IG sites. The ways in which choice of afforestation habitat and subsequent management of plantations could affect bird diversity in Ireland are discussed. Afforestation is likely to benefit the birds most in areas of high-productivity grassland managed at high intensity for grazing and/or cutting, with relatively low levels of shrub cover.

INTRODUCTION

At the beginning of the twentieth century, forest cover in Ireland was <1% (Mitchell 2000). Extensive afforestation over the second half of the twentieth century saw forest cover reach almost 10%, largely through the planting of non-native tree species, such as Sitka spruce *Picea sitchensis*. However, Ireland's forest cover remains among the lowest in Europe European Commission 2011, and the government's aim is to further increase the forest cover to 17% by 2030, continuing one of the fastest ongoing land use changes in Europe. The large-scale conversion of open, agricultural and semi-natural habitats to forest plantations constitutes a major ecological change and can be expected to have a profound influence on the populations of many bird species in Ireland (O'Halloran *et al.* 1998; Walsh *et al.* 2000). The habitats and landscapes in which these new plantations are established determine, to a large extent, the environmental consequences of this change for carbon emissions (Brainard *et al.* 2006) and for biodiversity (van der Horst and Gimona 2005). Legislation and statutory guidelines aim to prevent afforestation from conflicting with conservation interests. The procedures in place may not be sufficient to ensure that all relevant biodiversity concerns are considered during assessment of

afforestation applications (Iremonger *et al.* 2007), although recent changes to assessment procedure (Forest Service 2007a) have increased the likelihood that habitats and species of international conservation importance will be identified.

Most afforestation sites in Ireland are dominated by peatlands (which include bogs and heaths) wet grassland (WG) and improved grassland (IG), or a combination of these habitats (Smith *et al.* 2006). Peatlands in Ireland can be broadly classified as heaths, which occur on peats <0.5m in depth; bogs, which occur on deeper areas of peat; and other categories, which apply to anthropogenically disturbed peat habitats (Fossitt 2000). WG occurs on wet or water-logged soils that are either poorly drained or subject to regular flooding (Fossitt 2000). In an agricultural context, it tends to occupy areas that have not been recently managed at high intensity. In contrast, IG is intensively managed or highly modified agricultural grassland that has been reseeded and/or is regularly fertilised. A variety of factors have influenced the post-war rates of afforestation in different habitats in the UK and Ireland, including land prices, agricultural subsidies, grant-aiding of afforestation and environmental awareness and legislation (Avery and Leslie 1990; IPCC 1999; Warren 2000). However, statistics pertaining to these planting trends have not previously been published.

Walsh *et al.* (2000) predicted that eighteen breeding bird species could potentially be negatively affected by afforestation in Ireland. A recent review (Lynas *et al.* 2007) identified eight of these (curlew, golden plover, lapwing, quail, redshank, red grouse, ring ouzel and twite) as being among the birds of highest conservation concern in Ireland, and a further five (dunlin, merlin, red-throated diver, skylark and snipe) as having an unfavourable conservation status (scientific names of all species mentioned in the text are listed in Appendix 1). However, there is a paucity of published information on the bird assemblages of the most frequently afforested habitats. The Countryside Bird Survey is beginning to generate data on bird populations in the wider countryside (Coombes *et al.* 2006; Crowe *et al.* 2010). However, the resolution of the data in this large-scale survey is low (a single habitat classification is selected to represent each 200m section of transect) and poorly suited to distinguishing subtle differences between the bird communities of different open habitats. Previous studies of birds using these habitats in Ireland have dealt with a limited range of species (e.g. Hutchinson and O'Halloran 1994; Andrews *et al.* 1996; Rees *et al.* 1997), focused on a subset of the structural variation within these habitats (e.g. Lysaght 1989; Moles and Breen 1995; Holt 1996), or examined these habitats as part of a more complex landscape matrix. This makes it difficult to compare the bird assemblages of any of these habitats with those of conifer plantations that would replace them if they were afforested (Wilson *et al.* 2006).

This paper presents an analysis of planting trends on different soil types, which are related to

the three habitats described earlier. The knowledge gap concerning birds of afforestation habitats is addressed by describing the breeding bird assemblages of peatland, WG and IG sites, along with associated hedgerows and scrub. Inter-site differences in bird assemblages are related to variation in habitat structure and to the likely changes that afforestation would entail in these habitats, thus enabling more effective evaluations of the likely ecological impacts, positive or negative, of proposed afforestation projects.

METHODS

AFFORESTATION TRENDS

Data on forest age and soil type of first rotation afforested sites were taken from the National Forest Inventory (NFI), a dataset of randomised systematic samples from plots in 1742 state and private forests representing the entire Irish forest estate (Forest Service 2007b). Forest age at time of survey was used to assign plots to five ten-year planting year classes (covering the period between 1956 and 2005) and a sixth class for forests planted before 1956. Sampling plots were assigned to one of the five soil categories: Peat; Podzol; Gley; Well-drained and Other (see Fig. 1 for a full list of the different NFI soil types comprising these different categories).

SITE SELECTION

Twenty-seven unplanted sites were selected in three broad habitat types; eight in peatland, ten in WG and nine in IG. The peatland study sites were

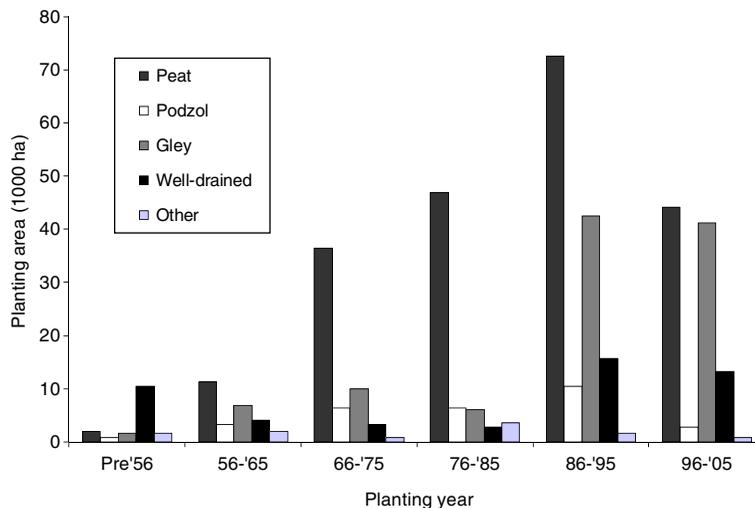


Fig. 1—Afforestation in Ireland by soil category and planting year. The Peat category includes basin and blanket peat soils of > 30cm depth, as well as cutaway peat; the Well-drained category comprises brown earth and brown and grey/brown podzolic soils; and soils in the Other category are sand, lithosol, regosol and rendzina. The data are restricted to extant, first rotation forests and do not include forests harvested at the time of data collection.

predominantly upland and lowland blanket bog (PB2 and PB3 in Fossitt 2000), but some also included areas of wet heath (HH3 in Fossitt 2000) in a matrix dominated by blanket bog. All were situated predominantly on shallow to deep peat. WG study sites comprised grass- and sedge-dominated plant communities in enclosed fields, on a range of soils from base-rich limestone-derived gley to acidic podzols derived from siliceous rock, corresponding to GS4 in Fossitt (2000). IG sites were more intensively farmed, well-drained areas, broadly corresponding to the Fossitt (2000) category of GA1. IG is typically maintained by regular re-seeding and/or fertilising, and is often found in large, contiguous areas comprising several smaller original fields whose boundaries have been removed to increase the proportion of field to field margin.

To ensure that study sites were characteristic of open habitats used for afforestation, 24 of the 27 sites were matched with nearby conifer plantations planted five years previously. Information taken from ground vegetation, aerial photographs taken just before afforestation and interviews with landowners and forest managers were used to ensure that the habitats of these plantations were closely matched with the study sites. The remaining three study sites were afforested in the year following this study. Each of the three study habitats has a clumped distribution within Ireland, peatland habitats being concentrated in the north and the west, WGs occurring most commonly in the midlands and in the west and improved agricultural land being most abundant in the east of the country (Bourke *et al.* 2007). As a consequence, although the geographical areas represented by the different habitat groups overlapped, there were differences between them (Fig. 2). An analysis was therefore

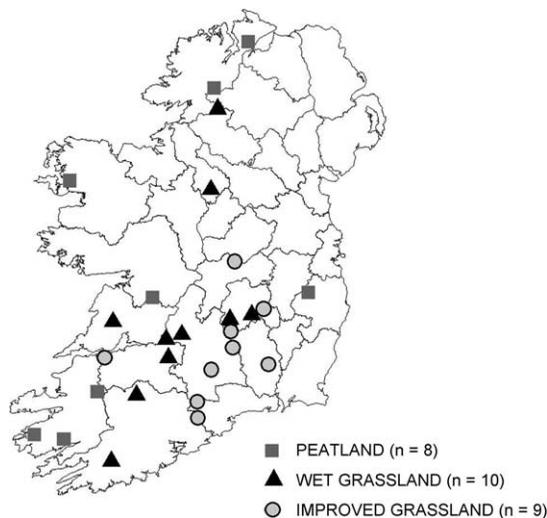


Fig. 2—Locations of the 27 study sites, coded by habitat type.

carried to determine whether, and to what extent, the influence of habitat type was separate from that of geographical location (see below). The boundaries of most study sites were determined by the extent of the habitat type they represented. However, some sites were situated in blocks of habitat too large to be surveyed in their entirety and therefore were delineated arbitrarily according to field boundaries or other geographical features. Especially in grassland sites, ownership was also a factor, determining which areas fieldworkers had permission to access.

BIRD SURVEYS

Each site was surveyed for birds using a mapping methodology similar to that described by Brown and Shepherd (1993). Sites were walked, ensuring all areas were approached to within 50m. For areas with shrub or tree cover, this minimum distance was reduced to 20m. The species, behaviour and position of all birds seen or heard were recorded on a 1:4000 map. All birds detected within 15m of the site boundaries were included in analyses. The common and scientific names of all species found in the survey are given in Appendix 1.

Bird data were collected from each site over two visits, one in May/early June and one in June/early July, in the summers of 2002 and 2004. Bird surveys took between 1 and 3 hours and were conducted between 0700 and 1800 hours. Effects of time of day on bird detectability were minimised by excluding periods in the early morning and evening when birds are known to be particularly active (Thomas 1999), which are therefore not comparable with other times of the day (Patterson *et al.* 1995; Sim *et al.* 2005), and by approaching all parts of each site to within a sufficiently small distance to maximise detection of non-singing individuals. In addition, visit times were similar for each of the three habitat groups. Bird surveys were not conducted in heavy or persistent rain, or in strong winds (greater than Beaufort scale 4). Clusters of birds of the same species were recorded as having a maximum number of two individuals, to reduce the influence of fledged family parties on density estimates. Flying birds of the following species and groups were excluded from analyses because their presence could not be assumed to indicate an association with the habitat at their location: grey heron, gulls, pigeons, common swift, wagtails, swallows and martins, corvids, starling and finches with the exception of bullfinch.

HABITAT AND ENVIRONMENTAL DATA

The position of all hedges, and the shape, size and position of areas of non-hedge shrub and tree cover at each site were recorded on copies of six-inch

(scale 1:4000) Ordnance Survey (OS) maps published in the 1930s. Percentage values for canopy cover were estimated and assigned to the following categories: tree-lines (linear tree-features with a width of one tree), semi-natural woodland (two trees or more in width and greater than 2m in height), shrub cover (2m or less in height), pre-thicket forest plantation and closed canopy forest plantation. Areas occupied by other non-pasture habitat types, such as farmyards and gardens, were also mapped. Hedge sections were allocated scores from 1 to 4 for height (1 = < 1m; 2 = 1–2m; 3 = 2–4m; 4 = > 4m), width (1 = < 1m; 2 = 1–2m; 3 = 2–3m; 4 = > 3m) and vegetation beneath hedge canopy (1 = bare, 2 = less dense than canopy, 3 = similar to canopy, 4 = of similar density to and wider than canopy). These scores were summed and used to classify hedges as small (maximum combined score of 8), medium or large (a minimum combined score of 10).

Information on management of livestock and cutting of grass for hay and silage was available for eighteen of the study sites. Stocking rates were calculated by multiplying the number of livestock units per hectare (1 unit = one cow, one horse or ten sheep) by the proportion of the year during which the site was grazed and ranged from 0 to 2.1. The intensity of silage/hay harvesting was calculated as the number of cuts taken per year and ranged from 0 to 2. These two variables were added together to give an index of forage yield. The apparent intensity of grazing and hay/silage cutting on the site was also estimated according to an index from 0 to 3 (0, none apparent; 1, little effect on vegetation structure; 2, considerably reduced the height of some but not all grazed species and 3, severe grazing or cutting) in three 10m × 10m plots within each site. Average height of field layer vegetation was also measured at each of these plots. Elevation of each plot was taken from OS Discovery series maps. These measurements were averaged to give site measures of elevation, grazing/cutting intensity and sward height for all sites.

DATA PREPARATION AND ANALYSIS

All bird mapping data, hedge survey and point counts were entered onto digitised maps using ArcView GIS 3.2. This program was used to calculate the lengths of hedges and areas of non-hedge features, and to assign birds recorded during mapping surveys to areas of open land, areas within 10m of hedges, and areas of other (non-hedge) tree or shrub cover. Hedges were represented in the GIS by one-dimensional vectors. To allow for the width of the hedge itself (commonly up to 4m), birds recorded as being within 12m of a hedge

vector were defined as being within 10m of a hedge.

A site size-independent measure of species richness was derived using species richness accumulation curves (calculated as the statistic Mao Tau) plotted against cumulative number of individuals recorded (N) for each site, using EstimateS (Colwell 2006). The units of the x -axes were rescaled by dividing them by the density of individuals (N/Area), effectively converting them to area. From these graphs, species richness values were standardised to the point on each accumulation curve equivalent to the size of the smallest site (6.2ha).

Canonical Correspondence Analysis (CCA) with variance partitioning was carried out using the *vegan* package in R (Borcard *et al.* 1992) to assess whether and to what extent the variation in bird assemblages attributable to site type was confounded with geographical location. Following Legendre (1990), the matrix of two-dimensional coordinates x and y was completed by adding all terms of the cubic surface regression: $(x + y) \times (x + y) \times (x + y)$, to account for non-linear patch and gap elements of spatial variation in bird assemblages. Global non-metric multidimensional scaling analysis (NMS), cluster analysis and indicator analysis were carried out using PC-Ord (McCune 2006) to better understand the relationships between bird assemblages and habitat variables. All other statistical analyses were carried out using SPSS 12.01 (SPSS 2003). Bird species densities were calculated as the maximum number of individuals recorded in a site during a visit, divided by the site area.

NMS was used to summarise site-level information on bird assemblages (Legendre and Legendre 1998). Ordination analyses were undertaken on bird density and habitat data across all sites, using Sørensen (or Bray and Curtis) distance measures. The parameter setup used for the NMS analyses was as follows: six initial axes; 50 runs with real data; stability criterion 0.0005; twenty iterations to evaluate stability; maximum number of iterations 500; initial step down of 0.2; random starting coordinates and 50 randomised runs for Monte Carlo testing. All species occurring in < 5% of the sites in a particular ordination were excluded from analyses.

Flexible-beta cluster analysis ($\beta = 0.25$) was undertaken using PC-Ord (McCune and Mefford 1997) to identify clusters of sites with similar species assemblages. Indicator species analysis, as described by Dufrene and Legendre (1997), was used to identify the species that typified bird assemblages in these clusters. An indicator value I was calculated for each species in a given group of sites g as the

product of the relative mean density d of the species in the group (compared to the mean density of the species across all groups), and the proportion of sites n in the group where that species occurred:

$$I_{ag} = \frac{d_{ag}}{d_a} \times \frac{n_{ag}}{n_g}$$

A species was considered typical of a group when I was larger than 0.25, and the result of the Monte Carlo test (1000 runs) was significant.

The relationships between ordination axes, bird species richness and density and environmental variables were assessed using non-parametric correlation (Kendall's τ_b). Differences between three or more groups were tested using ANOVA, and pairwise comparisons carried using Dunnett's T3 statistic, which does not assume homogeneity of variance between groups. Parametric statistical analyses were carried out where data conformed to the assumptions of the test. Re-analysis using a non-parametric test (Kruskal–Wallis) yielded similar results to all ANOVA results presented here.

RESULTS

AFFORESTATION TRENDS

Peat was the largest of the five afforested soil categories, both overall (accounting for 52% of all afforestation) and in every ten-year period after 1956 (Fig. 1). Until 1955, the majority of afforestation was on Well-drained soils. The proportion of planting on Peat increased between 1956 (12%) and 1985 (71%), and subsequently declined to 43% between 1996 and 2005. Between 1956 and 1985, the proportions of planting on Gley (25–9%), Podzol (16–6%) and Well-drained soils (15% to 4%) all decreased, principally due to the proportional increase in the area of afforestation on Peat. After 1986, the proportion of afforestation on Gley increased substantially, reaching a maximum of 40% between 1996 and 2005. The proportion of planting on Other soils has remained low, varying between 1% and 10%.

BIRD ASSEMBLAGES OF OPEN HABITATS

A total of 46 bird species was recorded. The number of species recorded at a single site ranged from 5 to 28, and the estimated total density of birds at a site ranged from 0.3 to 10.7 birds per hectare. According to CCA ordination, habitat type and geographic location together accounted for 31.5% of the variation in bird species assemblage. The proportion of variation that was accounted for by habitat type alone was 20%, while that accounted for by spatial location alone was consider-

ably smaller (7%) with a further 5% of this variation attributable to both variables.

At the seventeen sites for which stocking densities and hay/silage harvesting information were available, forage yield was negatively correlated with sward height ($\tau_b = 0.39$, $n = 17$, $P = 0.034$) but not significantly correlated with grazing/cutting intensity ($\tau_b = 0.37$, $n = 17$, $P = 0.054$). However, for the thirteen grassland sites for which this information was known, forage yield was negatively correlated with sward height ($\tau_b = 0.68$, $n = 13$, $P = 0.002$) and positively correlated with grazing/cutting intensity ($\tau_b = 0.47$, $n = 13$, $P = 0.039$).

NMS ordination of bird densities at the site level generated two axes, which together accounted for 94% of the variation in species densities between sites (Fig. 3). Axis 1, which accounted for 66% of the variation in the original data, was strongly and positively correlated with bird species richness ($\tau_b = 0.39$, $n = 27$, $P = 0.005$) but not with total abundance of birds ($\tau_b = 0.16$, $n = 27$, $P = 0.25$). Peatland sites had lower Axis 1 values than grassland sites (Fig. 3). Axis 1 values were also positively correlated with soil pH ($\tau_b = 0.44$, $n = 27$, $P = 0.001$) and proportional area of hedge ($\tau_b = 0.32$, $n = 27$, $P = 0.021$), and negatively correlated with proportional area of open land ($\tau_b = -0.33$, $n = 27$, $P = 0.015$), but not significantly correlated with elevation ($\tau_b = 0.01$, $n = 27$, $P = 0.95$), forage yield ($\tau_b = 0.19$, $n = 17$, $P = 0.30$), grazing/cutting intensity ($\tau_b = -0.02$, $n = 27$, $P = 0.88$) or sward height ($\tau_b = 0.08$, $n = 27$, $P = 0.57$).

Axis 2, which accounted for 28% of the variation in the original data, was strongly correlated with species richness ($\tau_b = -0.54$, $n = 27$, $P < 0.001$) and total bird abundance ($\tau_b = 0.85$, $n = 27$, $P < 0.001$). IG sites had higher Axis 2 values than WG sites, although there was some overlap between these groups (Fig. 3). Axis 2 values among all site types were positively correlated with proportional area of open land and negatively correlated with proportional areas of hedge ($\tau_b = 0.47$, $n = 27$, $P = 0.001$) and of non-hedge tree/shrub cover ($\tau_b = 0.52$, $n = 27$, $P < 0.001$) and sward height ($\tau_b = 0.48$, $n = 27$, $P = 0.001$), but not significantly correlated with elevation ($\tau_b = 0.01$, $n = 27$, $P = 0.92$), soil pH ($\tau_b = 0.08$, $n = 27$, $P = 0.56$), forage yield ($\tau_b = 0.28$, $n = 17$, $P = 0.13$), grazing ($\tau_b = 0.19$, $n = 27$, $P = 0.18$). However, among grassland sites, Axis 2 values were positively correlated with both forage yield ($\tau_b = 0.77$, $n = 13$, $P < 0.001$) and grazing intensity ($\tau_b = 0.42$, $n = 20$, $P = 0.014$).

SITE CLUSTERS

Cluster analysis of bird community data separated sites into five groups (Fig. 3). Four of the clusters

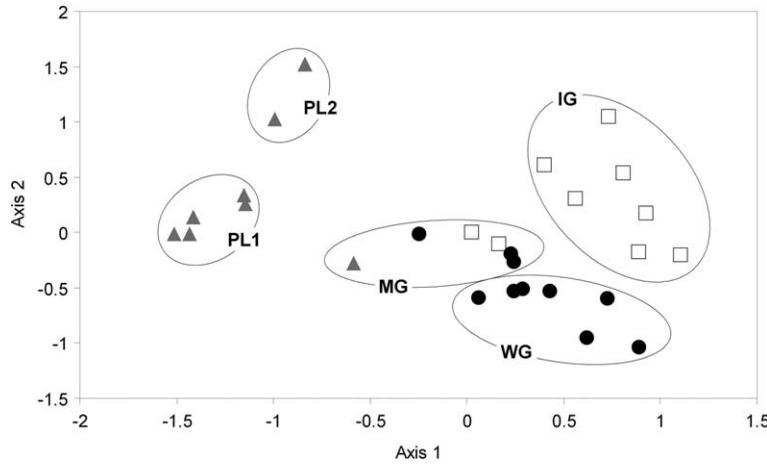


Fig. 3—NMS ordination of bird density data from mapping surveys of 27 unplanted sites. Final stress for two-dimensional solution = 9.046, final instability = 0.00001 (from 50 Monte Carlo runs). Sites are coded by habitat as follows: □ = improved grassland, ▲ = peatland, and ● = wet grassland. Clusters are coded by cluster name as follows: IG, Improved grassland cluster; MG, Mixed grassland cluster; PL1 and 2, Peatland clusters 1 and 2; WG, Wet grassland cluster.

comprised just one habitat type: peatland clusters 1 and 2 (PL1 and PL2), the WG cluster and the IG cluster. A fifth group, the mixed grassland cluster (MG), was dominated by a mix of WG and IG sites. MG sites all had higher Axis 2 values, were more heavily grazed and had lower shrub and tree cover than WG sites, but were of a more upland, less intensively agricultural nature than IG sites. The only peatland site in MG cluster was the most agriculturally improved of all the peatland sites, comprising wholly enclosed land, and situated entirely on shallow (<0.5m) peat.

There was substantial variation in the mean site size between the different clusters (Table 2), with WG sites being significantly smaller than IG sites ($F_{4,22} = 3.70, P = 0.02$). The proportions of cover

types in sites also differed between clusters (Fig. 4). WG sites had the lowest proportion of open land, with approximately two thirds of the site under hedge cover or other tree and shrub cover. MG and IG sites had a progressively larger proportion of open land, a smaller area of land within 10m of a hedge and almost no other tree and shrub cover. PL sites had no hedges and negligible other tree and shrub cover and therefore were entirely open land. WG sites had higher swards than other clusters and lower grazing/cutting index and forage yield than the other grassland clusters (Table 2).

Indicator analysis of the clusters identified eight species (blackbird, blue tit, coal tit, goldcrest, magpie, robin, wren and willow warbler) as being typical of WG sites, and meadow

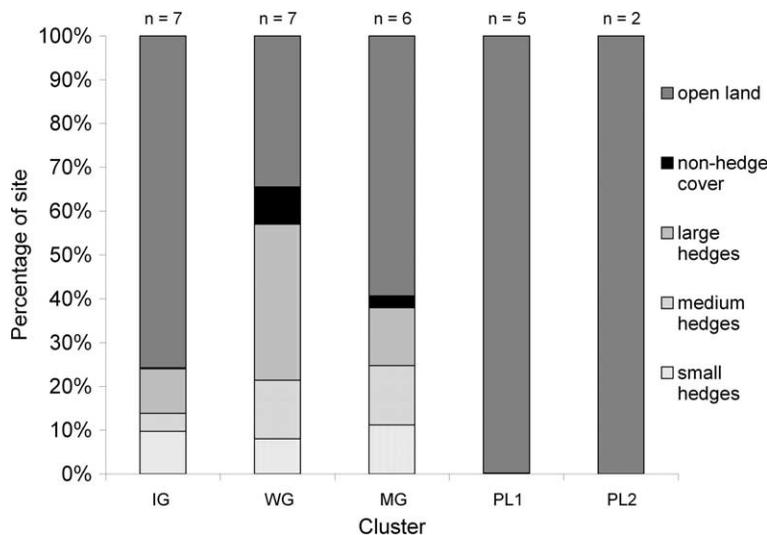


Fig. 4—Proportions of cover types in the five clusters. The values for hedge cover shown are the proportions of sites in each cluster within 10m of each of the three hedge categories. Cluster codes are explained in the text.

Table 1—Indicator species and their indicator values (IndVal) for clusters from bird mapping survey data. Cluster codes are explained in text. Only species with a maximum IndVal of 25 or more are included. The max IndVal is indicated in bold, and the *P*-values reported are the chance of obtaining such a high IndVal at random. The last two indicators in the table, both for IG cluster, are the absence of skylark and snipe, rather than presence or abundance.

<i>Species</i>	<i>IG (n = 7)</i>	<i>WG (n = 7)</i>	<i>MG (n = 6)</i>	<i>PL1 (n = 5)</i>	<i>PL2 (n = 2)</i>	<i>P value</i>
<i>Presence of . . .</i>						
Blackbird	17	56	26	0	0	0.002
Blue Tit	26	65	6	0	0	0.001
Coal Tit	2	62	24	1	0	0.001
Goldcrest	14	64	16	1	0	0.006
Magpie	8	52	3	0	0	0.009
Robin	17	52	30	0	1	0.049
Wren	19	56	23	1	0	0.003
Willow Warbler	3	71	21	0	1	0.002
Meadow Pipit	2	11	31	45	9	0.021
Skylark	0	2	10	54	23	0.021
<i>Absence of . . .</i>						
Skylark	39	27	6	0	0	0.002
Snipe	32	17	14	1	8	0.007

pipit and skylark as being typical of PL1 sites (Table 1). Although no species were identified as being typical of the sites in the other clusters, the absence of skylark and snipe was characteristic of IG sites.

Site species richness was much higher in grassland than in PL sites and was significantly higher in WG sites than in IG sites ($F_{4,22} = 25.0$, $P < 0.001$; Table 2). Total bird abundance was significantly higher in the MG and WG sites than in the other clusters and was highest in WG sites ($F_{4,22} = 23.9$, $P < 0.001$; Table 2). Only the sites in the grassland clusters were on enclosed land and therefore had hedges. In these sites, bird densities within 10m of hedges descended from about ten birds per hectare for WG sites to just over four birds per hectare for IG sites ($F_{2,17} = 4.8$, $P = 0.023$; Table 2). Densities of birds in other tree and shrub cover descended progressively from 40 birds per hectare in IG sites to just under five birds per hectare in PL2 sites ($F_{3,17} = 3.3$, $P = 0.046$; Table 2). However, as only MG and WG sites had more than 1% other tree and shrub cover, the potential for this variable to influence the bird assemblages in other site clusters was limited. Densities of birds in open land were not as high as in the vicinity of hedges or other types of cover (Table 2), being less than one bird per hectare in IG and PL2 sites, and between three and five times this density in the other clusters ($F_{4,22} = 3.75$, $P < 0.018$).

SPECIES RICHNESS AND BIRD DENSITY

Across all sites, bird species richness and density were correlated with shrub and tree cover (Table 3). Bird species richness and density were positively correlated with area of hedges, shrub cover and semi-natural woodland. Bird species richness was also positively correlated with area of treelines. Neither bird species richness nor bird density was correlated with young or mature plantation forest cover. Among grassland sites, bird density was negatively correlated with forage yield ($\tau_b = 0.62$, $n = 13$, $P = 0.004$) and grazing/cutting intensity ($\tau_b = 0.44$, $n = 20$, $P = 0.010$) and positively correlated with sward height ($\tau_b = 0.52$, $n = 20$, $P = 0.002$). Bird species richness was not significantly correlated with either forage yield ($\tau_b = -0.39$, $n = 13$, $P = 0.077$) or grazing intensity ($\tau_b = 0.33$, $n = 20$, $P = 0.055$) but was positively correlated with sward height ($\tau_b = 0.45$, $n = 20$, $P = 0.008$). The proportional area of open habitat in grassland sites was positively correlated with grazing/cutting intensity ($\tau_b = 0.44$, $n = 20$, $P = 0.010$).

DISCUSSION

AFFORESTATION TRENDS

The relationship between soil and habitat type is moderated by a number of factors such as climate,

Table 2—Mean values for size of site (ha), stocking density (in livestock units ha⁻¹), silage and hay cutting frequency (cuts year⁻¹), forage yield index, grazing/cutting index, sward height (m), bird species richness (R), total bird density, bird density in hedges, bird density in areas of non-hedge tree and shrub cover and bird density in open areas, in the five groups of sites according to cluster analysis of bird species assemblage data. Habitat codes are explained in text. Bird densities are expressed as the mean number of birds detected per visit per hectare. All means are given ± standard error.

	<i>IG</i>	<i>WG</i>	<i>MG</i>	<i>PL1</i>	<i>PL2</i>
Area (ha)	20.5 ± 4.8	11.0 ± 0.9	19.0 ± 6.0	22.0 ± 3.4	43.0 ± 20.0
Elevation (m)	135 ± 24	107 ± 15	163 ± 40	115 ± 38	168 ± 68
Stocking density	1.3 ± 0.6	0.7 ± 0.1	1.1 ± 0.3	0.7 ± 0.3	0.4
Silage/hay cuts	0.7 ± 0.7	0.3 ± 0.2	0.4 ± 0.3	0	0
Forage yield	2.0 ± 0.1	1.0 ± 0.1	1.5 ± 0.2	0.7 ± 0.3	0.4
Grazing/cutting index	2.0 ± 0.4	0.7 ± 0.4	2.1 ± 0.3	2.2 ± 0.1	0
Sward height	0.26 ± 0.02	0.76 ± 0.06	0.42 ± 0.11	0.31 ± 0.02	0.42 ± 0.02
Standardised R	11.5 ± 1.0	18.1 ± 1.1	13.3 ± 1.4	4.8 ± 0.6	2.7 ± 0.2
Total density	2.2 ± 0.5	9.5 ± 1.3	4.8 ± 0.3	3.2 ± 0.3	0.8 ± 0.3
Hedge density	4.6 ± 1.2	10.4 ± 1.6	7.1 ± 1.2	—	—
Tree/shrub density	38.1 ± 14.3	34.5 ± 8.3	5.8 ± 3.5	2.5 ± 2.2	—
Open density	0.8 ± 0.1	4.9 ± 1.3	3.5 ± 0.9	3.2 ± 0.3	0.8 ± 0.3

altitude and management. Nonetheless, the habitat types inferred from the soil data are broad enough to encompass substantial ecological variation, while still being closely linked with the main soil types that have been afforested over the past 50 years (Fossitt 2000). Between 2001 and 2005, research teams based in Trinity College Dublin and University College Cork groundtruthed more than 100 afforestation sites in Ireland as part of the site selection process for several biodiversity studies (Wilson *et al.* 2006; Smith *et al.* 2008). The majority of plantations encountered during these site visits were planted on peatland, WG and IG sites, confirming the inferences between soil and habitat type drawn in this study.

The proportion of state afforestation on peat has decreased slightly over the past 30 years, but in absolute terms, the area of peat planted with trees

has increased over this time. During this time, the Forest Service published guidelines aimed at discouraging planting on deep peat, for largely economic reasons (Forest Service 2000). Since this time there has been an increasing awareness in the forestry industry that afforestation on peat should be discouraged for other reasons including the stability of carbon stocks held in organic soil (Byrne and Milne 2006) and the biodiversity of peatland habitats (Coillte 2007), as well as compliance with EU directives. Over the same time period, afforestation on gleys has substantially increased in both relative and absolute terms. These soils, which are typical of WG habitats (Fossitt 2000), account for a higher proportion of planting over the past fifteen years than at any previous time.

The Forest Service requires that National Parks and Wildlife Service must be consulted before

Table 3—Kendall’s τ_b correlations between bird species richness (R) and site densities and six types of tree and shrub cover.

<i>Variable</i>	<i>R</i>			<i>Density</i>	
	<i>N</i>	<i>Kendall’s τ_b</i>	<i>P</i>	<i>Kendall’s τ_b</i>	<i>P</i>
Total hedge	27	0.64	< 0.001	0.24	0.084
Shrub	27	0.38	0.009	0.51	0.001
Treeline	27	0.38	0.01	0.31	0.046
Woodland	27	0.37	0.02	0.42	0.005
Mature plantation	27	-0.10	0.5	-0.12	0.5
Young plantation	26	-0.07	0.65	-0.04	0.8

afforestation of areas that have been designated as National Heritage Areas, Special Areas for Conservation or Special Protection Areas. Although some important peatland and grassland areas in Ireland have been designated, many remain undesignated (Dwyer 2000), and therefore have no formal legal protection from land-use changes. Outside designated areas, sites where habitats or species listed in the Annexes of EU Habitats Directive 92/43/EEC are identified should now be referred to the Forest Service ecologist (Forest Service 2007a). As the requirement for referral applies only to internationally protected habitats and species, sites of local or regional biodiversity importance may still be overlooked.

BIRD ASSEMBLAGES OF OPEN HABITATS

The six commonest species found in this study were meadow pipit, wren, robin, blackbird, willow warbler and chaffinch. Because one of the criteria used to inform site selection was that study sites should be typical of afforestation sites, the sites used in this study were at higher altitude and in less intensively agricultural landscapes than the sites used by previous studies of farmland birds in Ireland (Lysaght 1989; Moles and Breen 1995; Holt 1996). Nevertheless, four of the five commonest species found in these previous studies were also among the six commonest species in this study (wren, robin, blackbird, dunnock and chaffinch). The relative abundance of meadow pipit and willow warbler in this study reflects the wider availability of scrub and less agriculturally intensive nature of the study sites compared with lowland agricultural areas.

The densities attained by birds in areas of tree and shrub cover, and especially in hedges, were far higher than in areas of open pasture (Table 2). Most of the bird species encountered in this survey were not specialists of open habitat. Moreover, forest specialists typically occur at higher densities in habitats with high levels of tree and shrub cover than peatland and grassland specialists do in open habitats (Lack 1933; Moss *et al.* 1979). However, as open areas accounted for more than 50% of sites in all clusters apart from WG, their contribution to the carrying capacity of sites for birds is important. For instance, although the density of birds in open areas of PL1 sites was lower than in areas of hedge or other cover in IG sites, it was more than three times higher than the density of birds in the open habitat of IG sites. Cover of open land was sufficiently dominant in both of these clusters (Fig. 4) that PL1 sites had higher overall bird densities. If, after they are afforested, open areas experience even a small increase in the carrying capacity for birds, this could result in a large overall increase in the numbers of birds supported by afforestation sites. This is especially true for sites with relatively low areas of

hedge or other types of shrub and tree cover, and sites with particularly low densities of birds in areas of pasture. Berg (2002a) found that ‘residual habitats’, such as shrubby areas and natural grassland, could have a disproportionate influence on the bird assemblages of farmland-dominated landscapes, despite occupying only about 10% of the land area.

Of the clusters with hedges (WG, IG and MG), the density of birds in and around hedges was highest in WG sites and lowest in the IG sites (Table 2), probably due to the greater size and structural complexity of hedges in WG sites (Fig. 4). Differences in the carrying capacity of hedge habitat between these clusters were further increased by the total amount of hedge habitat available—over twice as much in WG as in IG sites, with the area of hedge habitat in MG sites being intermediate. The density of birds in non-hedge tree and shrub cover was higher in the IG cluster than in any other cluster, and nearly ten times as high as the density of birds within 10m of hedges in the same cluster. However, the effect of this cover type on overall bird abundances in IG sites was negligible, due to the very small proportion of the area of these sites that it occupied (Fig. 4). These differences in vegetation structure between site types may be partly due to the observed differences in the levels of grazing and cutting among these sites (Table 1). Grazing and cutting of hay and silage could have a direct impact on the establishment and subsequent growth of trees and shrubs in grassland areas. Alternatively, hedgerows and other areas of non-crop land might be kept smaller in intensively managed pastures to maximise the area of grassland under production. In some situations, scrub may be cleared from agricultural land to maximise the area qualifying for governmental farming grant payments (BirdWatch Ireland 2010).

It is likely that the most proximate limit on densities of birds in areas of intensive pasture, such as were typified by IG sites, is a lack of suitable cover for generalist and woodland birds. Although a more detailed study would be required to determine the precise nature of the relationship between grazing and cutting on tree and shrub development, the data in this study suggest that availability of suitable habitat for many species can be limited by intensive grassland management. If these pastures were managed in a way that allowed the dimensions of hedges to increase, or small patches of shrubs and trees to develop, this could have a positive effect on the numbers of woodland and generalist birds in this type of farmland. The effect of afforestation would likely be a similar (if more dramatic) one in the long term, with additional short-term benefits to bird communities of open pasture during the tree establishment phase, resulting from the cessation of grazing by livestock and harvesting for hay or silage.

All of the species identified in Table 1 as being typical of the WG cluster are relatively common in Irish agricultural landscapes (Copland 2010; Crowe *et al.* 2010) and occur in a wide variety of habitats, although they are particularly associated with habitats that have an element of tree cover (Fuller 1982). With the exception of blue tit, all these species are also abundant in Irish conifer plantations (Wilson *et al.* 2006). Elements of tree cover such as patches of woodland and large hedges, which serve to increase the suitability of sites to a wide range of generalist and woodland bird species, may have the opposite effect on open habitat specialists such as meadow pipit and skylark. Sparks *et al.* (1996) caution that while large hedges are beneficial for many species (especially those typical of woodland habitats), their presence may be detrimental to open habitat species such as skylark. While a decrease in grazing levels would have a positive impact on the biodiversity of many upland areas (Bleasdale and Sheehy Skeffington 1995; MacGowan and Doyle 1996), some grazing is required to maintain many peatland and grassland habitats.

IMPACT OF AFFORESTATION ON BIRDS

The predominant change in a bird community that results from establishment of a forest is from one typical of an open habitat to that of a forest. This change has been documented by recent studies of bird assemblages over the commercial forest cycle (Wilson *et al.* 2006; Sweeney *et al.* 2010). Whether the impact of a plantation on birds is positive or negative depends largely on the bird assemblage of the open habitat that it replaces. Peatland sites tend to have low bird diversity but are occupied by habitat specialists. Among the birds recorded on peatland sites in this study were several upland specialists such as hen harrier, red grouse, stonechat and whinchat. Species recorded breeding in the less intensively farmed grassland sites included curlew, grasshopper warbler, reed bunting, whitethroat and willow warbler. These birds are dependent on ground or shrub layer vegetation that is typically absent or scarce during later stages of the forest cycle. Species such as meadow pipits and skylarks, though present in other habitats, reach far higher densities in upland and semi-natural grassland habitats (Chamberlain and Gregory 1999; Vanhinsbergh and Chamberlain 2001) and are absent from closed canopy plantations (Wilson *et al.* 2006). Studies in the UK (Wallace and Good 1995), Europe (Ellenberg 1988; Andres and Ojeda 2002) and North America (Lachance *et al.* 2005) show that afforestation of peatlands and other areas typified by a low intensity of management tends to diminish biodiversity at local and regional scales. Not all areas of peatland and rough pasture support high densities or diversity of specialist birds of open

and shrub-rich habitats. However, the bird assemblage of any peatland or WG site proposed for afforestation should be examined to ensure that important open habitat species are not negatively affected. This could be done directly, by surveying the birds present on the site, or indirectly, by assessing the presence and prevalence of indicators of bird diversity (Smith *et al.* 2008).

IG sites in this study were typically stocked at an average density of 1.3LU ha⁻¹ and were dominated by areas of open habitat, with few areas of scrub outside of the hedgerows. However, the results of this study show that open habitat specialists are scarce or absent from these high-productivity grasslands. This is supported by the fact that absence of skylark and of snipe, which are among the most widespread specialists of open habitats (though both are on the Irish Amber List due to recent declines in population size), were indicators for IG sites (Table 1). In the long term, birds in these sites will potentially benefit from afforestation. Many of the bird species that inhabit the hedges in these sites are capable of occupying the forests that replace them. Python *et al.* (2005) found no evidence of a negative impact of up to 30% cover of conifer plantations on bird diversity in typical Irish lowland farmland landscapes. Most of the common species that inhabit these landscapes are generalists, capable of attaining high densities in conifer plantations (Wilson *et al.* 2006). Compared to intensively managed farmland, young forests could provide more suitable habitat for species requiring high levels of shrub cover, because the vegetation they require is not restricted to the field boundaries. However, both abundance and diversity of birds tend to decline in plantation forests following canopy closure, due to shading out and dieback of all lower vegetation layers (Patterson *et al.* 1995; Marion and Frochet 2001; Wilson *et al.* 2006). The value of plantations for many farmland birds can further be improved by maintaining sufficient open space around existing hedges so that they will not be shaded out as the plantation matures, ensuring temporal continuity of dense shrub cover through the commercial cycle and increasing the presence of native broadleaved trees.

The recent dominance of gleys among afforested mineral soils indicates that WG sites are being afforested more commonly than IG sites. This is confirmed by a recent study of forestry trends, which found that, in areas of grassland, the least productive fields are the most likely to be afforested (Kearney 2001). The results of this study indicate that, compared to IG sites, WGs farmed at lower intensity typically have high levels of shrub and tree cover, and support diverse assemblages of birds, on which the overall impact of afforestation will be negative.

Even if such areas remain unplanted, their continued value for open habitat birds is likely to be contingent on grazing levels. Recent and ongoing changes to Europe's Common Agricultural Policy, including the decoupling of stocking rates from agricultural payments, as well as the requirement for cross-compliance with conservation directives in Natura 2000 (Visser *et al.* 2007), are expected to result in reduced grazing levels in many overgrazed areas, with a positive impact on bird diversity (Waterhouse *et al.* 2006). However, in some marginal areas of farmland, grazing pressure may fall below levels needed to suppress development of tree and shrub vegetation layers. In such areas, succession will typically result in the development of a closed canopy, and the disappearance of most low-lying shrubs and ground vegetation, with loss of habitat for farmland bird species (MacDonald *et al.* 2000; Scozzafava and De Sanctis 2006). Berg (2002b) suggests that areas of abandoned semi-natural grassland might be maintained by periodical grazing. If unplanted areas in forested landscapes continue to be grazed at low intensity, this might allow the persistence of low-lying shrubs and associated ground-nesting bird species.

CONCLUSIONS

Afforestation of open habitats of high conservation value will generally have a negative impact on local or regional breeding bird diversity. In Ireland, such habitats include grasslands that are farmed at low intensity, particularly those with relatively high levels of shrub and tree cover, and any areas of peatland or grassland that support open habitat specialists. Alternative land uses to afforestation should be considered for sites that support suites of bird species that are characteristic of open habitats, especially species such as curlew, golden plover, hen harrier, merlin, lapwing, red grouse and whinchat, which are of national (Lynas *et al.* 2007) or international (BirdLife International 2004) conservation interest. However, the future of the bird assemblages in many open farmland habitats is likely to depend on the persistence of low-intensity grazing and grass-harvesting regimes (Shrub 2003).

The bird communities of many shrub-poor and over-grazed grasslands, as well as some degraded peatland areas, are sufficiently impoverished that their replacement with the bird assemblages supported over the course of a forest cycle may, in many cases, constitute an increase in the avian diversity of these areas. This is especially likely to be the case if the management of these plantations takes birds into account. In particular, sufficient open space should be left around existing areas of shrub and tree cover to enable their persistence

during later stages of the forest cycle. Increasing the rate at which agriculturally improved grasslands are afforested will support the government's strategic aim to increase forest cover, improve the ability of intensively farmed areas to act as carbon sinks and relieve the pressure to plant on more biodiverse habitats.

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APPENDIX 1

List of birds found during study

Blackbird (<i>Turdus merula</i>)	Long-eared owl (<i>Asio otus</i>)
Blackcap (<i>Sylvia atricapilla</i>)	Long-tailed tit (<i>Aegithalos caedatus</i>)
Black-headed gull (<i>Larus ribundus</i>)	Magpie (<i>Pica pica</i>)
Blue tit (<i>Cyanistes caeruleus</i>)	Mallard (<i>Anas platyrhynchos</i>)
Bullfinch (<i>Pyrrhula pyrrhula</i>)	Meadow pipit (<i>Anthus pratensis</i>)
Chaffinch (<i>Fringilla coelebs</i>)	Mistle thrush (<i>Turdus viscivorus</i>)
Chiffchaff (<i>Phylloscopus collybita</i>)	Pheasant (<i>Phasianus colchicus</i>)
Coal tit (<i>Periparus ater</i>)	Pied wagtail (<i>Motacilla alba</i>)
Collared dove (<i>Streptopelia decaocto</i>)	Raven (<i>Corvus corax</i>)
Cuckoo (<i>Cuculus canorus</i>)	Red grouse (<i>Lagopus lagopus</i>)
Curlew (<i>Numenius arquata</i>)	Reed bunting (<i>Emberiza schoeniclus</i>)
Dipper (<i>Cinclus cinclus</i>)	Robin (<i>Erithacus rubetra</i>)
Dunnock (<i>Prunella modularis</i>)	Rook (<i>Corvus frugilegus</i>)
Feral pigeon (<i>Columba livia</i>)	Sedge warbler (<i>Acrocephalus schoenobaenus</i>)
Goldcrest (<i>Regulus regulus</i>)	Siskin (<i>Carduelis spinus</i>)
Goldfinch (<i>Carduelis carduelis</i>)	Skylark (<i>Alauda arvensis</i>)
Grasshopper warbler (<i>Locustella naevia</i>)	Snipe (<i>Gallinago gallinago</i>)
Great tit (<i>Parus major</i>)	Song thrush (<i>Turdus philomelos</i>)
Greenfinch (<i>Carduelis chloris</i>)	Sparrowhawk (<i>Accipiter nisus</i>)
Grey heron (<i>Ardea cinerea</i>)	Spotted flycatcher (<i>Muscicapa striata</i>)
Grey wagtail (<i>Motacilla cinerea</i>)	Starling (<i>Sturnus vulgaris</i>)
Hen harrier (<i>Circus cyaneus</i>)	Stock dove (<i>Columba oenas</i>)
Hooded crow (<i>Corvus corone</i>)	Stonechat (<i>Saxicola torquata</i>)
House martin (<i>Delichon urbicum</i>)	Swallow (<i>Hirundo rustica</i>)
House sparrow (<i>Passer domesticus</i>)	Swift (<i>Apus apus</i>)
Jackdaw (<i>Corvus monedula</i>)	Wheatear (<i>Oenanthe oenanthe</i>)
Kestrel (<i>Falco tinnunculus</i>)	Whinchat (<i>Saxicola rubetra</i>)
Lapwing (<i>Vanellus vanellus</i>)	Whitethroat (<i>Sylvia communis</i>)
Lesser redpoll (<i>Carduelis cabaret</i>)	Willow warbler (<i>Phylloscopus trochilus</i>)
Linnet (<i>Carduelis cannabina</i>)	Wood pigeon (<i>Columba palumbus</i>)

Other species mentioned in text

Golden Plover (<i>Pluvialis apricaria</i>)	Red-throated Diver (<i>Gavia stellata</i>)
Merlin (<i>Falco columbarius</i>)	Ring Ouzel (<i>Turdus torquata</i>)
Quail (<i>Coturnix coturnix</i>)	Twite (<i>Carduelis flavirostris</i>)
	White-fronted Goose (<i>Anser albifrons</i>)