Tracking the impact of afforestation on bird communities

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Abstract
Studies of the impact of afforestation on biodiversity typically rely on surveys conducted at planted and unplanted sites at a single point in time. This paper reports on surveys of bird diversity in unplanted grassland sites and repeat surveys at the same sites seven years after afforestation and is the first longterm experiment of this type for Ireland. Birds were chosen for this study as they are easily surveyed indicators of biodiversity that respond well to environmental change. Overall, species richness was higher in the young forest plantations than in the pre-planting open grassland sites and this increase was a result in the increase in shrub cover following the cessation of grazing in these sites after afforestation. Some farmland bird species were absent following afforestation, but these were replaced by forest bird species. Wren (*Troglodytes troglodytes*) and warbler (Sylviidae family) species benefited the most from afforestation, though other research suggests that the long-term impact of afforestation on many of these species will be less positive. This study highlights the requirement for long-term investment in biodiversity monitoring following land-use change. Scientific studies of this kind support the integration of biodiversity maintenance with timber production, which is a goal of national and global afforestation policies.

Keywords: *Afforestation, biodiversity, birds, forest management, grassland*.

Introduction

Afforestation refers to the establishment of a forest or stand in an area where the preceding vegetation or land-use was not forest (Helms 1998). Although forest cover in the island of Ireland was once very extensive, deforestation over the past few thousand years has resulted in just 1% of native forest cover remaining by 2008 (Cross 2012). Ambitious afforestation targets have been set by the Irish government since the introduction of the Forestry Act of 1946. Despite not always having met these targets (Malone 1998, COFORD Council 2009), Ireland had the highest afforestation rate in Europe between 1990 and 2007. Nonetheless, Ireland's current forest cover remains well below the European average, at under 12%, with just over 80% of this being plantation forest, compared with an average of 30% forest cover throughout most of the rest of Europe (ITGA 2012). The Irish government currently has a target to achieve a national forest area of 1 million ha (circa 14.5% of land area) by 2030 through further afforestation. In addition to the provision of habitat, forests provide employment and are central to our future "green" economy, as a

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means of increasing carbon capture and as a renewable energy resource (COFORD Council 2009).

The success of afforestation is measured not only by how well it meets economic objectives, but also by how it contributes to social, aesthetic and environmental requirements (Renou and Farrell 2005, Carnus et al. 2006). One of the major issues in developing sustainable forestry in Ireland is ensuring that the industry does not negatively impact on biodiversity. Conservation has a constraining effect on afforestation in Ireland through environmental legislation and efforts to comply with Sustainable Forest Management. To this end, the current Afforestation Grant and Premium Scheme in Ireland incorporates biodiversity considerations, and the Forest Biodiversity Guidelines focus on how to conserve and enhance biodiversity through appropriate planning, conservation and management (An Taisce 2011).

Research has demonstrated that afforestation may have either a positive or negative impact on biodiversity, the magnitude and direction of which is influenced by the land use that preceded forest planting and by local forest management practices and the tree species planted (Hunter 2000, Carnus et al. 2006, Marquiss 2007, Brockerhoff et al. 2008, O'Connell et al. 2012). When planted in areas of low biodiversity value, such as intensively managed agricultural land, plantation forests offer the potential to enhance biodiversity and protect species through the provision of habitat (Freedman 2007, Iremonger et al. 2007, Brockerhoff et al. 2008) for at least a portion of the forest cycle. For many decades, commercial afforestation was confined largely to Ireland's uplands for economic reasons. However, increases in the importance of the uplands for conservation, amenity and wind energy, combined with agricultural reforms and an increased interest in planting diverse mixtures of conifers and broadleaved trees, has meant that afforestation is increasingly taking place at lower elevations, and also on private rather than public lands, where it now competes with food production as a land use.

Ireland supports some internationally significant populations of birds and possesses some notable woodland subspecies such as jay (*Garrulus glandarius hibernicus*) and coal tit (*Periparus ater hibernicus*) (Nairn and O'Halloran 2012). Of 199 bird species assessed in Ireland, 25 were on the Red list, 85 on the Amber list and the remaining 89 on the Green list (Lynas et al. 2007). Red is the highest conservation priority, with species needing urgent action, amber include species of moderate conservation concern, whereas green-listed birds are not threatened.

Forests are important habitats for birds, and although Irish forests possess fewer bird species, and in particular woodland specialists, than found in the rest of Europe (Fuller et al. 2007), they support a large number of bird species, including several of conservation concern (Lynas et al. 2007, Nairn and O'Halloran 2012).

Bird species are protected at European level under the EU Birds Directive (209/147/EC) and at national level under the Wildlife Act 1976 and The Wildlife (Amendment) Acts 2000–2010. Afforestation impacts on biodiversity through a diverse, interacting array of mechanisms such as changes to food and breeding resources, dispersal opportunities and physiological tolerance limits (Gardner 2010). Birds are useful indicators of biodiversity as they are typically positioned near the tops of food chains, are responsive to environmental change, their populations are readily surveyed and their biology is well understood (Furness and Greenwood 1993). Therefore, the impact of afforestation on the biodiversity of bird communities is of interest to forest managers and policy makers alike.

Despite political and scientific interest in the impact of afforestation on biodiversity, few direct studies have been conducted to date. Studies of the influence of afforestation have typically been indirect in nature, examining biodiversity in forest plots at different stages of the afforestation cycle (chronosequences) at different locations (Moss et al. 1979, Newton and Moss 1981, Pithon et al. 2005, Wilson et al. 2006, Freedman 2007, Marquiss 2007). Such studies have aimed to detect temporal trends from studies of different aged sites and are conducted due to necessity or convenience (Pickett 1989). However, they cannot overcome the potential confounding effect of biotic and abiotic differences in replicate sites. The impact of afforestation at five forest sites in Ireland that were established as long-term monitoring plots during the BIOFOREST project was assessed in this study (Iremonger et al. 2007) by direct comparison of bird communities at the sites before and after planting.

Materials and methods

Study sites

Five grassland sites, all designated for afforestation were selected for use in this study. These sites comprised agriculturally improved and wet grassland habitats (GA1 and GS4 in Fossitt 2000), which are the main habitat types currently undergoing afforestation in Ireland . The sites were located at Garyandrew (Co. Tipperary), Kilbraugh (Co. Kilkenny), Coolsnaghtig (Co. Cork), Donaghmore (Co. Laois) and Mullanmeen Under (Co. Fermanagh). These sites were planted in 2003 with an intimate mixture of conifer and broadleaf trees with the exception of Mullanmeen Under which was only planted with Sitka spruce (*Picea sitchensis* Bong.). Sites were afforested according to the Republic of Ireland Forest Service Guidelines and UK Forest Commission Guidelines, both of which require a minimum of 10% of the area to be retained as open space for biodiversity conservation. The tree species planted and other site details are outlined in Table 1.

Bird diversity surveys

Bird surveys were conducted in 2002, before the sites were planted, and in 2010, seven years after planting. Surveys were carried out using point transects (Bibby et al. 2000), during which birds were counted for 10 minute periods from fixed positions. Point counts were of 10 minute duration, during which time all birds seen and heard within 50 m of the observer were recorded and their distances from the observer estimated subjectively. A minimum of six point counts were located at each study site. Points were placed at a minimum of 100 m apart and positioned to incorporate internal and external field boundaries. The same point locations were surveyed before and after afforestation at each site. Bird surveys were conducted only during good weather conditions and not in heavy or persistent rain, or in winds

Table 1: *Details of the location, size, grassland type and tree species planted at each of the five study sites that were sampled pre-afforestation and seven years post-afforestation.*

Site	Location	Area (ha)	Grassland type	Tree species planted
Garyandrew, Co. Tipperary	7° 50' 29.3"W 52° 28' 10.5"N	17.0	Improved	Ash (*Fraxinus excelsior* L.), sycamore (*Acer pseudoplantus* L.)
Kilbraugh, Co. Kilkenny	7° 33' 35.3"W 52° 38' 35.3"N	20.7	Improved/ wet	Japanese larch (*Larix kaempferi Lam.*), Sitka spruce (*Picea sitchensis* (Bong.) Carr), alder (Alnus glutinosa L.)
Coolsnaghtig, Co. Cork	9° 8' 35.4"W 51° 45' 0.5"N	6.2	Wet	Ash, oak (*Quercus petraea* (Matt.) Liebl.), Spanish chestnut (*Castanea sativa* Mill.), European larch (*Larix decidual* Mill.), Japanese larch, Sitka spruce, alder
Donaghmore, Co. Laois	7° 36' 14.9"W 52° 52' 5.6"N	11.9	Wet	Japanese larch, Sitka spruce, alder, ash, sycamore
Mullanmeen, Co. Fermanagh	7° 45' 0.3"W 54° 35' 14."N	29.5	Wet	Sitka spruce

greater than Beaufort scale 4, as the detectability of many bird species is severely reduced in persistent rain or in windy conditions (Bibby et al. 2000).

In each of the two sampling years, each point was surveyed twice: once early in the breeding season (April–May) and once later (May–June). Each site was visited twice to decrease the risk of missing either early or late breeding species, which might be absent or relatively undetectable during one of the visits. Counts were carried out between two hours after sunrise and two hours before sunset, to avoid sampling around dawn and dusk when bird singing activity typically peaks, which could confound comparisons between sites. Timing of visits was varied such that all points received one visit in the morning and one in the afternoon in each sampling year. Individuals that were detected in flight were excluded from analysis as their presence does not necessarily indicate a breeding association with the habitat over which they are flying.

Habitat recording

The results of previous Irish research has identified that the shrub cover and tree cover components of habitat vegetation have strong influences on bird communities (Wilson et al. 2006, Sweeney et al. 2010a). Therefore, the percentage cover of deciduous trees, conifer trees and of shrubs were recorded within a radius of 50 m at each sampling point in each year. Shrub cover was defined as all woody vegetation from 0.5 to 2.0 m in height, not including the commercial tree crop.

Data analysis

The data collected during point counts were used to derive estimates of bird density (numbers of individual birds ha^{-1}) and species richness (cumulative number of species) at each survey point. Distance software was used to derive species densities from field observations (Buckland et al. 2001). This was necessary because the detectability of birds differed between species and habitats. Estimation of densities using Distance is a standard method in ornithology that allows more reliable comparisons of numbers across different habitats and species. The conservation value of each site was estimated as the sum of the density of each species recorded at a site divided by the estimated national abundance of that species (Crowe et al. 2011).

Differences in the conservation value, species richness and total bird density in addition to the recorded habitat variables, shrub cover, conifer tree cover and deciduous tree cover, between pre- and post-afforestation states at each individual sampling point were assessed using paired t-tests. Prior to performing all t-tests, normality and homogeneity of variance were tested using Kolmogorov-Smirnov and Levenes tests, respectively. Where assumptions of normality and /or homogeneity of variance were not met, differences between paired samples were tested using Wilcoxon's Signed Rank Tests . All univariate tests were conducted in PASW Statistic 18.

Generalised linear models using a Poisson distribution were used to identify those habitat variables related to species richness and conservation value at each sampling point . Variables in each analysis included conifer tree cover, deciduous tree cover and shrub cover. To adjust for site effects, site was included as an explanatory factor in all models. The overall impact of afforestation was included in the analyses, with treatment (before and after afforestation) included as a categorical variable. To determine the most important explanatory variables in the GLMs, applied model averaging was applied using the dredge function in the R library MuMIn .

Non-metric multi-dimensional scaling (NMDS) analysis was carried out to determine between-site patterns in the bird community assemblages using PC-ORD (version 6; MjM Software, Gleneden Beach, Oregon, USA) using Sørensen distance measures. All species which were recorded in <5% of the point count locations were excluded from the analyses.

Results

Thirty one bird species were recorded during the current study. Grey wagtail (*Motacilla cinerea)*, jackdaw (*Corvus monedula*), skylark (*Alauda arvensis*) and spotted flycatcher (*Muscicapa striata*) were all recorded at grassland habitats before planting, but not at afforested sites. Blackcap (*Sylvia atricapilla*), grasshopper warbler (*Locustella naevia*), goldfinch (*Carduelis carduelis*), hooded crow (*Corvus*

Table 2: *Densities (individuals ha^{-1} ± SE) of birds recorded at sites one year before planting and seven years post-planting.*

	Pre-planting		Post-planting	
Common species				
Chaffinch (*Fringilla coelebs*)	1.45	(0.48)	1.33	(0.51)
Robin (*Erithacus rubecula*)	1.45	(0.38)	0.80	(0.28)
Wren (*Troglodytes troglodytes*)	0.62	(0.25)	1.79	(0.40)
Blackbird (*Turdus merula*)	0.92	(0.24)	0.95	(0.27)
Warblers				
Willow warbler (*Phylloscopus trochilus*)	0.41	(0.17)	2.29	(0.50)
Chiffchaff (*Phylloscopus collybita*)	0.21	(0.16)	0.81	(0.35)
Grasshopper warbler (*Locustella naevia*)	0.0	-	0.04	(0.04)
Blackcap (*Sylvia atricapilla*)	0.0	-	0.52	(0.20)
Hole nesters				
Coal tit (*Parus ater*)	0.51	(0.23)	0.6	(0.32)
Blue tit (*Parus caeruleus*)	0.62	(0.32)	0.53	(0.22)
Great tit (*Parus major*)	0.06	(0.06)	0.35	(0.16)
All other species	4.43	(1.39)	2.70	(0.54)
Total Bird Density	12.75	(2.75)	12.93	(1.82)

cornix) and lesser redpoll (*Carduelis cabaret*) were all recorded at afforested sites, but not at pre-afforested grassland sites. The most common bird species at pre-planting sites were chaffinch (*Fringilla coelebs*), robin (*Erithacus rubecula*), goldcrest (*Regulus regulus*) and blackbird (*Turdus merula*). Densities of chaffinches, robins and blackbirds remained relatively stable at sites following afforestation and the densities of wren (*Troglodytes troglodytes*) and warbler (Sylviidae) species increased (Table 2). A number of Amber listed species were recorded including spotted flycatcher and skylark at pre-planting grassland sites, grasshopper warbler at post-planted sites and kestrel (*Falco tinnunculus*) and swallow (*Hirundo rustica*) which occurred at these sites both before and after planting.

There was no significant difference in the density of birds ($z = -1.7$, $p = 0.09$) or the conservation value ($z = -1.52$, $p = 0.13$) of survey points before or after afforestation (Figure 1). However, there was a significant increase in bird species richness following afforestation ($z = 3.44$, $p = 0.001$) (Figure 1).

There was considerably more variation between individual bird communities in pre-afforestation grasslands sites in comparison to the post-afforestation bird assemblages at these same sampling locations. This is indicated by the wider spacing of the pre-afforestation sites in the NMS ordination space compared to the post afforestation sites, which occupy a relatively tight cluster. This indicates that bird communities in these sites become more homogenous following afforestation (Figure 2).

There were considerable differences in the habitat variables between pre- and post-afforestation (Figure 3). Shrub cover ($t = -5.9$, df = 31, $p < 0.001$), conifer tree

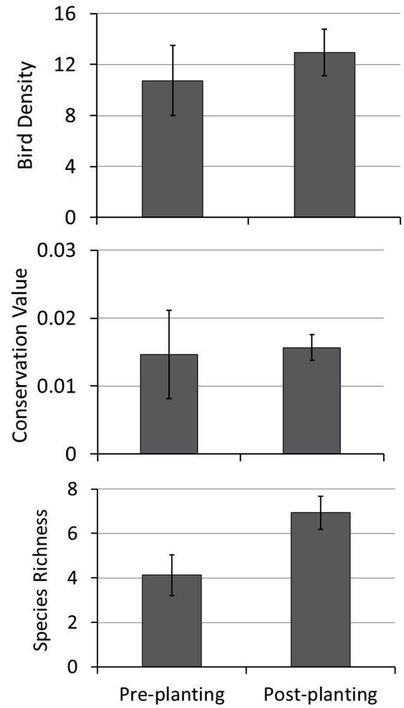

Figure 1: *Mean density of birds, number of bird species (species richness) and site conservation value in the study sites before and after afforestation.*

cover (z=3.74, df =31, p < 0.001) and deciduous tree cover (z=3.55, df = 31, p < 0.001) all increased significantly in the seven years following afforestation (Figure 3).

Results from the general linear modelling showed that the number of bird species recorded was positively related to afforestation and the amount of shrub cover within the habitat. There was no impact of site, conifer tree cover or

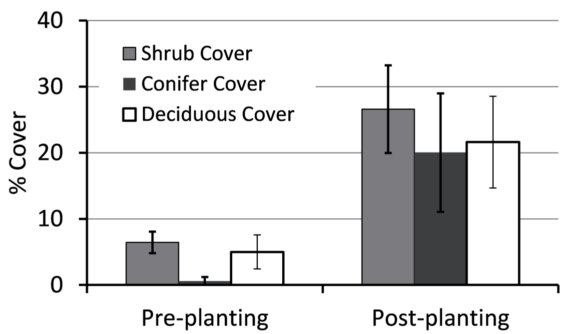

Figure 2: *Mean percentage of shrub cover and conifer and deciduous tree cover recorded one year pre-afforestation and 7 years post-afforestation.*

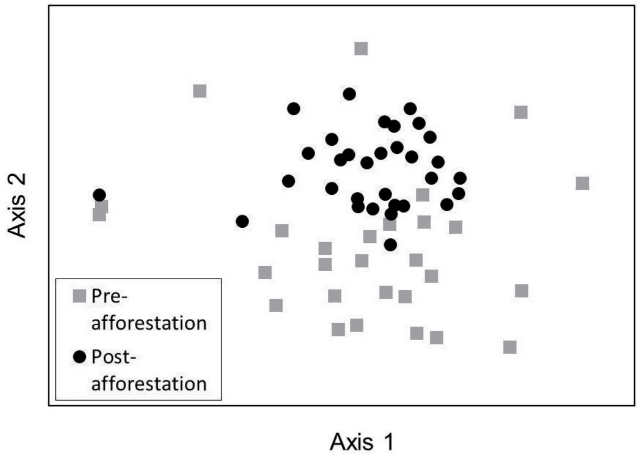

Figure 3: *NMS ordination of bird community data of the study sites pre- and post-afforestation (■ = Pre-afforestation; ● = Pre-afforestation). Variation accounted for by axis 1 = 44.2% and by axis 2 = 17.7%. Final stress for two-dimensional solution = 18.02; final instability = 0.00049 from 50 Monte Carlo simulation runs).*

deciduous tree cover on bird species richness (Table 3). None of the habitat variables measured had a significant impact on the computed conservation value metric of the study sites.

Discussion

Increasing national and global afforestation, combined with increasing constraints from conservation priorities means that now, more than ever, there is a need for data on the effects of afforestation on biological diversity. This is particularly true for birds, since almost all species are protected under the Wildlife Act (Wildlife Act 1976).

Studies of the effect of afforestation on biodiversity typically use "space-for-time substitution", which involves the extrapolation of a temporal trend from a series of different-aged samples from a single point in time to derive information on the possible effects of afforestation (Pickett 1989). This method is particularly useful in forest biodiversity research, where forest rotations are lengthy when compared to the typical duration of a research project. However, it is not always possible to elucidate temporal trends from such a chronosequence approach, due to results being confounded by abiotic and/or biological differences between sites. The value of long term tracking of biodiversity in recently established forest sites, such as in this study, is being increasingly recognised (Gardner 2010). The current study of bird diversity in Irish forests is the first analysis of the long-term impacts of afforestation on biodiversity in Ireland. Of the four bird species that were recorded only at unplanted sites, the grey wagtail, skylark and jackdaw are birds of open habitats. However, the amber listed spotted flycatcher is associated with both farmland and forest habitats (Nairn and O'Halloran, 2012) and its absence from afforested sites is noteworthy. The observed increase in density of birds post-afforestation was due primarily to increases in the numbers of a few key groups, particularly wren and warbler species. All of these birds commonly nest on or near the ground in herbaceous vegetation or low-lying brambles (Gibbons et al. 1993) and the decrease in grazing pressure as a result of afforestation (Marquiss 2007) most likely facilitated their increase. Willow warblers inhabit both woodland and farmland habitats (Nairn and O'Halloran 2012), but although they were relatively rare before planting, they were the most abundant species in the young forest plantations. Woodland species such as blackcap, grasshopper warbler, goldfinch and lesser redpoll all benefited from afforestation of the grassland sites and were found exclusively post-planting. Long-term benefits for these species cannot be inferred from the results of the current study as the bird communities of plantation forests change as the trees mature (Wilson et al. 2006, Sweeney et al. 2010a), and the current study is confined to the effect of afforestation during the early stages of the forest cycle. Subsequent surveys of this site, as the planted forests mature, will be required to extend our understanding of the effects of afforestation to later forest growth stages.

In some situations afforestation is reported to be related to a loss of bird species (Newton 2004, Marquiss 2007). However, plantation forests can, in some contexts, offer opportunities for biodiversity conservation, particularly when forests replace biodiverse-poor land uses such as is the case for intensive agriculture (Hunter 2000, Carnus et al. 2006, Brockerhoff et al. 2008). In this study, a positive impact of afforestation on bird communities was demonstrated at the grassland sites through increased species richness. Surprisingly, no impact of site on the species richness or conservation value following afforestation was found. One might expect a more beneficial impact of afforestation at the improved grassland site (Garyandrew) as it had low habitat complexity pre-afforestation, relative to the wet grassland sites.

The positive impact of afforestation at this early stage of the commercial forest cycle is mediated through changes in vegetation structure. Pre-planting, improved grassland sites had little shrub or tree cover and were therefore lacking in vegetation structure, particularly in comparison to wet grassland sites. Bird diversity in Ireland is positively related to habitat complexity in plantation forests (Wilson et al. 2006, Sweeney et al. 2010a). Following afforestation, the development of shrub cover significantly enhanced the habitat complexity of the sites, making them attractive to a wider range of species than the homogenous short sward grassland it replaced. Pre-thicket forests therefore provide suitable habitat for species characteristic of semi-natural open habitats, and have the potential to increase biodiversity in intensively managed agricultural landscapes.

Table 3: *Summary output of the model-averaged parameter estimates from generalised linear models of bird species richness recorded at the study sites.*

Parameter	Estimate	Adjusted SE	z value	Significance (*p*)[a]
Intercept	4.50	1.420	3.18	0.0015
Afforestation	2.80	0.800	3.49	0.0005
Site	−0.50	0.811	0.62	0.5300
Shrub cover	0.05	0.024	2.28	0.0200

[a] Significant at 0.05 level.

However, although there was a significant increase in bird species richness following afforestation, ordination of the bird communities revealed a contraction of diversity between bird communities following afforestation as demonstrated by the restriction in ordination space occupied by the bird communities of planted sites. This resulted in the bird communities at sites becoming much more similar to each other post-planting than they were before planting. This is an important distinction as afforestation, up to the stage assessed, resulted in a loss of bird assemblage beta diversity (variation in communities between sites) .

The observed changes to bird communities of sites following afforestation may have been in part due to fluctuations in bird populations at a national scale over the seven years between sampling dates. However, as the observed changes were consistent with those anticipated from previous studies (e.g. Sweeney at al. 2010b, Wilson et al. 2010), it appears that the overwhelming factor affecting bird communities in this study was the impact of afforestation.

As these plantation forests mature, considerable changes in the bird communities may occur . Based on this past research, biodiversity of birds in Irish plantation forests is largely dependent on habitat complexity and in particular, the retention of shrub cover. Future monitoring of these sites is required to determine the impact of the maturing tree crop on the shrub cover within these sites and the resulting effects on bird communities.

Conclusions

The combined priorities of biodiversity and wood production are critical issues to be resolved for future afforestation in Ireland, and can be achieved through targeted management practices. Long-term investigations such as this study are essential to inform policy in order to protect biodiversity over the plantation cycle.

Afforestation of grassland areas is likely to increase over the next decade or so. Tree planting in these areas should be staged to occur gradually over an extended period of time so that the resulting forest cover is uneven in age and structure, thus maximising biodiversity. Establishment of new forests in this way will help to ensure continued availability of pre-thicket forest habitats, which may be disproportionately important for bird diversity in some agricultural landscapes.

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