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PLANFORBIO

Planning and management tools
for biodiversity in a range of Irish forests

Implementation of an assessment and monitoring programme for Irish and British forests

BIOPLAN

FINAL REPORT

January 2014

Prepared for the Department of Agriculture, Food & the Marine

By

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Table of Contents

ACKNOWLEDGEMENTS ..... **III**
PROJECT TEAM ..... **IV**
EXECUTIVE SUMMARY ..... **V**
INTRODUCTION **9**
Background 9
Forest management for biodiversity 10
Objectives 14
Project structure 14
CHAPTER 1: AFFORESTATION AND BIODIVERSITY **15**
Background 15
Objectives 17
Methodology 17
Results 20
Discussion 26
Conclusions 28
CHAPTER 2: BIODIVERSITY OF FOREST ROADS **29**
Background 29
Objectives 30
Methodology 30
Results 33
Discussion 34
Conclusions 36
CHAPTER 3: LONG-TERM BIODIVERSITY MONITORING **37**
Background 37
Objectives 38
Methodology 38
Results 39
Discussion 40
CHAPTER 4: PLANTED TREE SPECIES AND SPIDER DIVERSITY **41**
Background 41
Objectives 42
Methodology 42
Results 44
Discussion 44
Conclusions 45
CHAPTER 5: FUNGUS GNATS IN CLEAR-FELL AND THINNING DEBRIS **46**
Background 46
Objectives 47
Methodology 47
Results 49
Discussion 50
Conclusions 50
CHAPTER 6: BIODIVERSITY INDICATORS **51**
Background 51
Objectives 52
Methodology 52
Results 54

Discussion 56
Conclusions 58
CHAPTER 7: GROUND-DWELLING INVERTEBRATE DIVERSITY OF MIXED FOREST PLANTATIONS 59
Background 59
Objective 60
Methodology 60
Results 61
Discussion 62
Conclusions 64
CHAPTER 8: BIRD DIVERSITY IN SEMI-NATURAL AND PLANTATION OAK FORESTS 65
Background 65
Objectives 66
Methodology 66
Results 68
Discussion 70
Conclusions 72
CHAPTER 9: INDICATORS OF WOODLAND BIODIVERSITY POTENTIAL 73
Background 73
Objectives 75
Methodology 75
Results 78
Discussion 80
Conclusions 82
CHAPTER 10: FOREST CONNECTIVITY 83
Background 83
Objectives 84
Methods 84
Results 87
Discussion 90
CHAPTER 11: IMPACTS OF LARGE HERBIVORES 93
Background 93
Objectives 94
Methodology 94
Results 96
Discussion 98
Conclusions 100
CHAPTER 12: CLIMATE CHANGE IMPACTS 101
Background 101
Objective 102
Methodology 103
Results 104
Discussion 107
Conclusions 110
CHAPTER 13: BIOPLAN GIS DATABASE 111
Background 111
Output - the BIOPLAN GIS 116
Discussion 124
IMPLICATIONS FOR POLICY AND PRACTICE 125
CONCLUSIONS AND RECOMMENDATIONS 129
BIBLIOGRAPHY 131
FULL LIST OF OUTPUTS FROM THE PROJECT 149

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Executive Summary

The four year BIOPLAN project '*Implementation of an assessment and monitoring programme for biodiversity in Irish and British forests'* was the fourth project in the PLANFORBIO research programme. The main objective of this research was to identify ways in which forest policy and management can safeguard the future of forest biodiversity and associated ecosystem services, and inform environmentally sustainable expansion of Ireland's forests. This project built on the BIOFOREST project '*Biodiversity in Irish plantation forests*' (2000 – 2006) and projects in the PLANFORBIO Programme '*Planning and management tools for biodiversity in a range of Irish forests'*, with a view to exploiting data collected thus far and gathering new data to assess forest biodiversity at both stand and landscape scales. BIOPLAN also aimed to establish links with Forest Research (UK) and establish a framework for future forest biodiversity research in Ireland, which is essential to scientifically underpin sustainable development in the forestry sector.

Ireland has the second lowest forest area in Europe, and it is the government's strategic aim to increase this cover over the coming decades. Ireland has a long history of human impacts on forest cover and, since the early twentieth century, Ireland's forest estate has been expanding mainly through commercial afforestation. For a long period the management of these new forests was solely aimed at sustained timber production. In recent decades, however, international recognition of the need to protect biodiversity and associated ecosystem services, and the contribution that forests can make in this regard, led to the inclusion of an environmental component in Irish forest policy and management plans. As a result of efforts to reduce the negative impacts on biodiversity, Ireland's plantation forest estate, once comprised almost exclusively of non-native conifer tree species now includes one quarter broadleaved forests. Concurrent with evolution in forest management objectives, diversification of the planted forests and expansion of Ireland's forest estate, there has been a transition from almost exclusive state ownership of Ireland's forests to a point now where almost half of Ireland's forests are in private ownership and management.

Against this backdrop the BIOPLAN project set out to respond to the demands of the forestry sector in Ireland. Research was conducted to provide scientific support for the integration of biodiversity conservation into the management of plantation forests, with a focus on silvicultural practices such as site selection, forest road configuration, planted tree species and forestry practices. The development of the Irish forest estate into the future was also investigated, particularly in relation to forest expansion and climate change. Finally, this project set out to establish long-term monitoring options for biodiversity change and evaluation of the effectiveness of biodiversity conservation measures into the future.

The selection of appropriate sites for afforestation is an important issue for sustainable forest management in Ireland, where the forest estate continues to expand at one of the highest rates in Europe. In order to inform recommendations for best-practice afforestation and to investigate trends in biodiversity, long-term monitoring plots were established across the island of Ireland during BIOFOREST. During BIOPLAN an initial resurvey of these sites was undertaken to provide data up to eight years after planting on the diversity and community composition of vascular plants, bryophytes, ground-dwelling spiders, hoverflies and birds. Results show that the changes in biodiversity and community composition following afforestation reflect the shift in habitat from grassland to forest. In general, the number of forest-associated species increases following afforestation, especially in the presence of planted broadleaf trees. Contributing factors to the biodiversity of the newly planted forests included grazing intensity, pre-existing habitat type, tree and sapling cover and ground cover. The findings show that that habitats which support relatively low

biodiversity, such as improved grasslands, should be favoured for afforestation over semi-natural habitats. The selection of sites with low biodiversity value for afforestation offers maximum opportunity for biodiversity enhancement and reduces negative effects associated with forest expansion.

The construction of roads through large, otherwise undisturbed, forests may bring about negative changes in biodiversity by increasing forest fragmentation. However, in fragmented landscapes of plantation forest within which there is little open space, roads may provide the opportunity to enhance biodiversity and their design and management may be a crucial component of sustainable forest management. The potential for forest roads to contribute to plant, invertebrate and bird diversity and the effect of increasing road width on biodiversity early in the forest cycle were investigated. A wealth of biodiversity was associated with forest roads, which acted to increase stand level biodiversity. The findings demonstrate however that, six years in to the forest cycle, no benefit accrued to biodiversity of doubling the standard road width. It is expected that that the benefits of increasing road width will only be realised when the plantation canopy starts to close, in approximately five more years. In order to quantify this impact and the contribution it makes to biodiversity, it will be necessary to continue monitoring of this experiment for several decades.

Because trees are relatively long-lived and processes in forests have long-term cycles, long-term studies play a central role in forest ecology and, to this end, a network of long-term monitoring plots were established during the BIOPLAN project. Baseline plant, invertebrate and bird biodiversity surveys were undertaken at four Sitka spruce and four lodgepole pine plantation forests. All sites used in this study are also part of the Pan-European FutMon long-term monitoring scheme, which will provide added value to data collected.

The selection of tree species in forest plantations is linked to their biodiversity value and the importance of native tree species is increasingly being recognised. The diversity of ground-dwelling spiders was investigated in three conifer plantation types in Ireland: Sitka spruce, lodgepole pine and Scots pine. Results indicate that Scots pine (formerly native in Ireland) had higher species richness than the two North American species, suggesting that planting of species of native provenance offers benefits for biodiversity in plantation forests. Investigation of the mechanisms involved in the observed differences revealed that managing plantations to increase canopy openness and ground vegetation cover and structural diversity increases the availability of microhabitats, which in turn benefits ground-dwelling spider diversity.

The conservation of deadwood-reliant insects is particularly important in managed, plantation forests where disturbance is high and these invertebrates are vulnerable. Intensive forest management can, however, result in landscapes with low quantities of deadwood (woody debris). Very little systematic research on saproxylic biodiversity has been conducted within the forested areas of Ireland to date. This study described the fungus gnat fauna associated with fine woody debris brash and examined the differences in fauna between clear-fell debris and thinning debris. The results clearly show that there are discrete communities of fungus gnats utilizing thinning and clear-fell brash in Sitka spruce plantations in Ireland. The findings show that even a very small area of deadwood habitat can support a wide array of species, which has important implications for forest management where brash mat production should continue and brash should be left *in situ* after both clear-fell and thinning operations while biodiversity value should be incorporated into brash mat creation guidelines. A number of new species to Ireland were recorded in this study, and so further work should be carried out to estimate the full complement of species that are using this resource.

In Ireland, and elsewhere, forestry is based mostly on even-aged, single-species, conifer stands. The use of mixed tree species forests in place of pure stands is one of the main methods promoted for maintaining biodiversity in commercial forests. Although supported by both international and national forest policies,

this is a complex issue, which varies according to tree species and across plant and animal groups, and there is insufficient understanding of the biodiversity benefits of mixed forests at present. The effect of a mixed tree species composition was investigated for ground-dwelling spider and beetle species in Scots pine and oak monocultures and Scots pine/oak mixed forests in three geographical regions: Ireland, southern England and eastern England. Spider and beetle species composition was similar across regions and mixed stands did not support higher species richness compared with monocultures. These results indicate that, for ground-dwelling invertebrates, there is no clear advantage of this tree species mix. European, Irish and UK forest management policies currently recommend lower levels of the secondary species in a mix (between 5 and 20%) compared with the levels of mixing considered here (10 - 50%). Therefore further research is required to determine the biodiversity, particularly of canopy-dwelling species, and to investigate the tree species mixes, planting ratios and planting patterns required for mixed stands to benefit forest biodiversity.

To further investigate the effects of planted tree species on biodiversity, the use of native oak in plantation forests was examined in a study comparing the bird diversity in these forests with that of semi-natural oak woodlands with that of highly grazed semi-natural oak woodlands. Bird diversity in plantation oak and semi-natural forests was similar, with no differences in species richness, total bird density, warbler density or density of hole-nesting bird species. However, highly grazed semi-natural oak woods had lower species richness than either of the other two forest types, and a lower density of warblers than oak plantations, which appears to be a result of browsing mediated differences in habitat complexity between the forest types. Results show that bird diversity in woodlands subject to particularly high grazing will be limited, unless ungulate populations are managed to promote the development of a more complex understorey.

Indicators, as surrogate measures of biodiversity, are important tools in sustainable forest management. The need for monitoring and reporting progress of biodiversity enhancement measures under international agreements, and the impossibility of recording all species in forests, has led to significant efforts to develop appropriate indicators. Indicators of biodiversity in plantation forests were developed as part of the BIOFOREST project. In the present study, these provisional indicators for bryophytes, vascular plants, ground-dwelling spiders and birds were tested in Sitka spruce, lodgepole pine, Scots pine, oak and mixed Scots pine/oak plantations, on independent data from both Irish and British sites. Conifer canopy cover was confirmed as a particularly important indicator, due to its influence on below-canopy microclimatic and structural conditions. Early and regular thinning will prevent the closure of the canopy, as well as increasing deadwood volume, which was confirmed as a positive indicator for forest-associated bryophytes. Both proximity to old woodland and stand age were confirmed as positive indicators for forest-associated vascular plants. Stand age was confirmed as a positive indicator for forest-associated spiders and reflects the development of suitable habitat as the plantation matures. This suggests that either maintaining stands beyond commercial maturity or conversion to continuous cover forestry will promote colonisation by native woodland species, particularly those with dispersal limitation. The lack of congruence of biodiversity indicators confirmed across the different species groups tested supports the theory that forest biodiversity indicators are species or taxa specific, and are not readily applicable across different plant or animal groups.

Indicators of biodiversity are an expression of potential rather than actual biodiversity in woodlands, as the existence of suitable habitat does not always lead to the presence of expected organisms. As no single indicator of biodiversity exists, a number of indicators used together is the most realistic way to describe the biodiversity potential of a forest. The aggregation of several indicators into a biodiversity index provides more useful information on forest biodiversity. Indices for assessing the biodiversity potential of native woodlands in Ireland were investigated and used to determine forest condition criteria using the data-set of the National Survey of Native Woodlands. An appropriate index based on the French 'Potential Biodiversity

Index' was developed for use in Ireland and displayed good potential for use in assessing the biodiversity of Irish semi-natural woodlands and further testing is recommended, particularly in plantation forests.

Forest fragmentation is linked to biodiversity loss, and the maintenance of habitat connectivity is promoted as a method of biodiversity conservation in forest management. Little scientific research has been conducted in highly fragmented landscapes such as Ireland, despite the understanding that biodiversity conservation is best addressed at the landscape, rather than the stand scale. The importance of functional connectivity for the biodiversity of spiders and birds was examined in this study using the Probability of Functional Connectivity (PFC), which accounts for both within and between forest patch connectivity. PFC had no significant impact on spider diversity but was negatively related to total bird diversity (including generalist species and those of open habitats) and positively related to forest specialist diversity. Thus forest specialist bird species are negatively affected by habitat fragmentation and decreasing patch size, demonstrating the importance landscape connectivity for the conservation of forest-associated biodiversity.

Wild herbivores, especially deer, have increased their range across Western Europe, including Ireland, in recent decades. The ecological consequences of expanding populations include impacts on: growth and survival of herbs, shrubs and tree species; vegetation, insects, birds and mammals; plant cover and diversity; nutrient and carbon cycles; and a shift in future canopy composition. Surveys of plant biodiversity were undertaken at an existing network of long-term study plots, ranging in size and age, distributed among oak woods in three National Parks in Ireland. The results indicate that grazing significantly alters plant community composition, structure, and tree regeneration. Homogenisation of flora is seen with total herbivore removal and so the maintenance of low grazing is recommended where woodland conservation and regeneration are desired. Large-scale, long-term fencing of oak woodlands should be replaced by large herbivore management programmes to ensure the conservation of diverse native woodland ecosystems.

Forests take a relatively long time to come to maturity, and during the next 50-100 years there will be significant changes in the world's climate, which will impact on the ecology and productivity of forests, and existing frameworks for biodiversity protection will need to be appropriately adapted. To evaluate the potential impacts of climate change on the distribution of woodland species in Ireland, a number of species distribution modelling techniques were used to predict the change in distribution of 104 plant, bird and butterfly species in response to a number of climate change scenarios. Under a scenario of unlimited dispersal almost 50% of species tested were projected to decline in range in response to future climate change. Under a scenario of limited long-range dispersal, declines in range were predicted for all 104 species. These findings have implications for forest management in Ireland where adaptive management will be required to prevent climate biodiversity loss, particularly in planted forests and for rare species.

A custom built database was created to provide a single repository for all data collected during the BIOPLAN project to facilitate data access, further analysis, visualisation, and exploration. This is particularly important as forest biodiversity science seeks to address questions covering large temporal and spatial scales. Primary forest biodiversity data is expensive and time consuming to collect and so the availability of archived biodiversity data can make a significant contribution to the future of this discipline and create opportunities for more integrative forest biodiversity research and further analysis of data in additional ways.

From these research findings 30 specific recommendations are made for policy and practice in the Irish forestry sector to mitigate the effects of forest management on biodiversity. These recommendations are underpinned by sound scientific research to ensure that their implementation will help to deliver on Ireland's commitment to sustainable forest management.

Introduction

Background

Although once almost completely covered in woodland, the island of Ireland suffered extensive deforestation over the past few thousand years (Mitchell, 2000) and today just 85,000 ha, or 1.2%, of native forest cover remains (Cross, 2012). In response to this, significant afforestation targets have been set by the Irish government since the introduction of the Forestry Act of 1946 (Malone, 2008) and Ireland has had the highest afforestation rate in Europe since 1990 (FAO, 2007). Nonetheless, Ireland's forest cover remains at just under 10.5%, with just almost 90% of this being plantation forest, compared with 30% forest cover throughout most of the rest of Europe. Conifer forest is the dominant forest type in Ireland representing 74.2% of all forests, while broadleaved forests represent just 25.8% (Forest Service, 2013). The Irish government aims to increase forest cover to 1 million ha (*circa* 14.5% of land area) by 2030 through further afforestation (COFORD Council, 2009) supported by policy incentives (McCarthy *et al.*, 2003).

Forests provide employment and are central to our future green economy as a carbon fixing and renewable energy resource (COFORD Council, 2009; Department of Agriculture Food & the Marine, 2010), and the success of afforestation is measured not only by how well it meets economic objectives, but by how it contributes to environmental requirements including biodiversity conservation (Renou and Farrell, 2005; Carnus *et al.*, 2006). Species conservation is necessary for the provision of many ecosystem functions and services which are central to future land-use policies and are dependent on a wide range of species (Balvanera *et al.*, 2006; Viglizzo *et al.*, 2012; Bastian, 2013). Historically forest management was focussed almost exclusively on timber production, but global awareness of the need for sustainable forest management led to significant advances in the maintenance and enhancement of biodiversity in Irish forests. Across Europe, policy processes such as the negotiations on a European Forest Convention and the EU Forest Strategy target the integration of biodiversity conservation into management of commercial forests in order to provide suitable habitats for forest-dwelling species (Kraus and Krumm, 2013). Changes in forest management practices in recent decades reflect the growing importance of ecological considerations (Barbati *et al.*; Lindenmayer *et al.*, 2006; Klenner *et al.*, 2009; Johansson *et al.*, 2013). It is recognised that changes in design and management of forest plantations can significantly improve their value for biodiversity, particularly where conditions mimic the structure of natural forests (Gardner, 2010; Hartmann *et al.*, 2010). Furthermore, forest management and related biodiversity and ecosystem services are related to forest ownership (Schaich and Plieninger, 2013). The significant expansion proposed for Ireland's forest estate, and move from public to private ownership, offers the ideal opportunity to plan and manage new forests to ensure long-term compliance with national obligations under environmental legislation including the EU birds and habitats directives and sustainable forest management.

The expansion of Ireland's forest estate during the early twentieth century was undertaken to replace woodland cover and meet high timber demand. Due to the lack of native Irish conifer species and limited range of native broadleaf species with commercial potential, much of the forests planted comprised fast-growing non-native conifer tree species in even-aged, single species stands. These forests are typically poor for biodiversity as they support largely generalist plant and animal species (Hartley, 2002; du Bus de Warnaffe and Lebrun, 2004; Stephens and Wagner, 2007; Bremer and Farley, 2010). However, small

changes in forest management practise can significantly increase their biodiversity value (Carnus *et al.*, 2006; Hartmann *et al.*, 2010; Paillet *et al.*, 2010). Some flora and fauna are more affected by forestry than others and, while some are negatively affected by plantation forests, some species are favoured by intensive forest management (Avery and Leslie, 1990; Peterken, 2001; Paillet *et al.*, 2010). The biodiversity of planted forests is influenced by pre- and post-afforestation management practices (Hunter, 2000; Carnus *et al.*, 2006; Marquiss, 2007; Brockerhoff *et al.*, 2008). The value of planted forests for biodiversity is also related to the biodiversity of the land-use they replace and the wider-scale landscape biodiversity, particularly where native forests are rare. In countries such as Ireland which have an extensively modified and intensively managed agricultural landscapes, and native forests have become rare, plantation forestry can benefit landscape biodiversity, particularly when appropriately managed (Hartley, 2002; Brockerhoff *et al.*, 2005; Brockerhoff *et al.*, 2008; Fuller *et al.*, 2008).

A number of initiatives support forest management for biodiversity in Ireland, including the National Biodiversity Plan, Forest Biodiversity Guidelines, the Native Woodland Scheme, and the Forestry Environment Protection Scheme. The Afforestation Grant and Premium Scheme in Ireland incorporates biodiversity considerations (An Taisce, 2011). However, Ireland continues to see declines in the conservation status of important protected habitats and species (NPWS, 2008). Forest management and policy for biodiversity conservation must be underpinned by evidence-based research and scientific study. In this regard the role of forests in supporting and enhancing biodiversity is of interest to scientists, forest managers and policy makers alike and is essential to inform appropriate planning and for compliance with EU Directives and international legislation. Areas of interest at present, in an Irish context, are our rare native woodlands, impacts of afforestation and forest management practices such as preceding land use, choice of tree species, open space and grazing on biodiversity, climate change impacts and the use of indicators for biodiversity assessment.

Forest management for biodiversity

Afforestation, or the establishment of forests on previously unforested land, is proceeding in Ireland at one of the highest rates in Europe and this expansion of the forest sector must be managed in a manner that is environmentally sustainable and to internationally recognised standards (Department of Agriculture Food & the Marine, 2010). The selection of sites for afforestation is guided by relevant Irish and European legal and regulatory frameworks and international protocols, and compliance with Ireland's forest biodiversity guidelines (Forest Service, 2000b) is mandatory for site selection and for grant aid. Afforestation leads to changes in ecosystem structure in relation to shading, micro-climate, nutrient cycling and habitat availability which, in turn, impacts on biodiversity. 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; Marquiss, 2007; Brockerhoff *et al.*, 2008; O'Connell *et al.*, 2012). For many decades, commercial afforestation was confined largely to Ireland's uplands for economic reasons. However, increases in the importance of our 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 altitudes. Afforestation is increasingly taking pace on private rather than public lands, where it now competes with food production as a land use, and is also controlled in many environmentally sensitive areas in Ireland (Upton *et al.*, 2014).

Therefore, studies of the impacts of afforestation on biodiversity are critical to inform continued expansion of Ireland's forest estate, particularly in relation to the selection of appropriate sites for afforestation.

The construction of roads through large, otherwise, undisturbed forests, may bring about negative changes in biodiversity by increasing forest fragmentation (Avon *et al.*, 2013; Johansson *et al.*, 2013). However, in fragmented landscapes of plantation forest within which there is little open space, roads may provide the opportunity to enhance biodiversity (Warren and Fuller, 1993; Delgado *et al.*, 2007; Smith *et al.*, 2007) and the design and management of forest roads is crucial for sustainable forest management (Lindenmayer *et al.*, 2006). The inclusion of open spaces, including forest roads, is an objective of forest management for biodiversity conservation in Ireland (Forest Service, 2000b). Assessment of the potential for forest roads and corridors to contribute to biodiversity conservation objectives is important in the development of forest policy where the Forest Biodiversity Guidelines currently stipulate that between five and ten per cent of forest area within plantations larger than 10 hectares must be retained as open space (Forest Service, 2000b).

The fact that the dominant trees in woodland can live for several centuries means that woodland processes occur over extremely long time scales (Peterken, 2000; Fukami and Wardle, 2005). Even in forest plantations managed by clear-felling, rotation lengths can range from several decades to over a century, depending on the species planted, and the planting of successive rotations means that similarly long time scales are involved (Peng, 2000; Kula and Giunalay, 2012). There are two methods used to study such long-term processes: space-for-time substitution (often referred to as chronosequences) and long-term monitoring (Bakker *et al.*, 2002). In space-for-time substitution, a temporal trend is inferred from studying sites of different ages, under the assumption that spatial variation and temporal variation are equivalent (Pickett, 1989). With long-term monitoring, a site is continuously studied using a formal monitoring system, giving the ability to directly relate change to time (Bakker *et al.*, 2002). While space-for-time substitution is a useful way of exploring changes that exceed the lifespan of investigators or their projects (Walker *et al.*, 2010), a number of studies have highlighted its shortcomings (Pickett, 1989; Feldpausch *et al.*, 2007; Johnson and Miyanishi, 2008). This is mainly due to the particular importance of site history, which makes space-for-time substitution useful only for identifying general trends or generating hypotheses (Pickett, 1989). This means that long-term monitoring studies are invaluable (Strayer *et al.*, 1986; Kratz *et al.*, 2003; Magurran *et al.*, 2010; Silvertown *et al.*, 2010); however, despite this being widely recognised, such studies are relatively rare (Strayer *et al.*, 1986). The longer a study is, the more valuable it becomes, allowing for the detection of cumulative or slow-acting impacts (Silvertown *et al.*, 2010). However, although detecting these impacts is an important first step, to fully understand their causes, long-term experiments are needed (Bakker *et al.*, 1996; Silvertown *et al.*, 2010). This ability to attribute change to specific causes is essential for the understanding of future change (Silvertown *et al.*, 2010). The results of such experiments and of long-term studies in general, can provide important insights to inform policy makers and practitioners.

The selection and management of canopy tree species in production forests is intrinsically linked to their biodiversity value, and is an important consideration in forest expansion policy and planning. Given the prevalence of introduced tree species used in plantation forests in Ireland, the potential for native tree species to provide both timber and forest biodiversity is of interest. The most commonly planted tree species is Sitka spruce (*Picea sitchensis*) at 52.5%, followed by 'other pines', principally lodgepole pine (*Pinus contorta*), which make up 9.7%, with Norway spruce (*Picea abies*) and Scots pine (*Pinus sylvestris*) comprising 4.2% and 1.3% of planting, respectively (Forest Service, 2013). Of these tree species, Scots pine

is the only one with an Irish connection, having been a major component of Irish native woodlands which virtually disappeared from Ireland during the first millennium (Roche *et al.*, 2009) and now considered 'semi-native' by the forestry industry in Ireland (Forest Service, 2000b).

Thinning and pruning are commonly employed in Irish plantations in order to prevent excess competition between trees and to promote growth in trees selected for retention until final harvest. The vast majority of commercial plantations in Ireland are clear-felled. The deadwood or "logging residues" produced by clear-fell and thinning operations is usually deemed unacceptable for conventional timber processing and so is left in situ after operations have ceased. Many parts of Scandinavia have been using these residues as a biofuel for decades to counteract rising fossil fuel prices while curbing greenhouse gas emissions (Lundborg, 1998). Even though this practice has been to the detriment of native saproxylic organisms, it is gaining momentum in other E.U. states. These piles of woody debris, known as brash piles, are understudied habitats of potential importance for saproxylic Diptera (true-flies), particularly fungus gnats.

Monitoring forest biodiversity is crucial for sustainable forest management, which in turn is critical for the conservation of forests and forest biodiversity (Gardner, 2010). Since a complete assessment of biodiversity is rarely possible, other than at very small scales, indicators as surrogate measures of biodiversity are important tools, and have received considerable attention over the past decade (Humphrey and Watts, 2004; Niemi and McDonald, 2004; Gardner, 2010). Indicators are attributes of a habitat whose values, when measured, are indicative of environmental conditions that extend beyond their own measurement. The indicator concept is based on the principle that easily measured features that affect or derive from variation in biodiversity can be used as an index of biodiversity (Landres *et al.*, 1988; Ferris and Humphrey, 1999). Three types of indicators can be identified - compositional (e.g. species), structural (e.g. physiognomy of forest stands and associated habitats) and functional (e.g. processes such as nutrient cycling) (Ferris and Humphrey, 1999; Larsson, 2001). A large amount of work has been undertaken in developing indicators at national and international scales, such as the pan-European indicators for sustainable forest management (MCPFE, 2003). However, there is also a need for indicators that can be used by forest managers in the assessment of biodiversity at the stand scale (Ferris and Humphrey, 1999). In order to be of practical use to forest managers, these indicators need to be easy to assess, repeatable, cost-effective and ecologically meaningful (Ferris and Humphrey, 1999), and the indicator selection process is potentially one of the most important components of forest biodiversity monitoring (Ferris and Humphrey, 1999; Gardner, 2010).

Plantation forests in Ireland and the UK are typically coniferous monocultures, often consisting of exotic species. These plantation forests are generally thought to support lower biodiversity than broadleaf woodlands, due to homogeneous habitat provision (Lust *et al.*, 1998; Fuller *et al.*, 2008). However, there is growing interest in the use of mixed tree species plantations to safeguard biodiversity (Gamfeldt *et al.*, 2013; Kraus and Krumm, 2013). Planting two or more tree species can result in an increased diversity of microhabitat types and related food resources, which may therefore enhance forest biodiversity (Lust *et al.*, 1998; Ishii *et al.*, 2004). In response to this, mixed stands now receive considerable attention in forest policy and management plans and planting of this forest type has increased across Europe, Ireland and the UK (Spence *et al.*, 1997; Forest Service, 2000b; Spiecker, 2003). However, recent studies have demonstrated that mixed species plantations may not always support higher species diversity than monocultures (Cavard *et al*., 2011; Oxbrough *et al*., 2012). This necessitates the investigation of mixed forest plantations at local scales to determine their usefulness for biodiversity conservation and to directly inform forest management.

The national forest estate in Ireland covers 10.5% of land area or 731,650 hectares (Forest Service, 2013). However, the average size of individual forests is just 10 hectares (IFFPA, 2011). Fragmentation of forested landscapes is related to a reduction in biodiversity associated with habitat loss and reduced patch size (Fahrig, 2003; Echeverría *et al.*, 2007; St-Laurent *et al.*, 2009). Even though the current rate of forest expansion in Ireland is among the highest in Europe (Wilson *et al.*, 2012), the increase in the amount of forest area may not be sufficient to ensure the persistence of species diversity if dispersal means (forest connectivity) are insufficient (Nikolakaki, 2004). Forest connectivity in the overall landscape is important for species level biodiversity and is a key element in mitigating the negative effects of forest fragmentation, particularly in highly fragmented landscapes, such as found in Ireland (Zuidema *et al.*, 1996). Modern forest management is more focussed on landscape scale processes than traditional stand scale management (Saura *et al.*, 2011b), and management of landscape connectivity affords us the opportunity to enhance biodiversity as prescribed by international agreements and legislation for biodiversity conservation. Understanding the effects of forest connectivity on biodiversity is essential for the formulation of appropriate forest management plans (Zuidema *et al.*, 1996) as these effects are related to the degree of fragmentation in the landscape and the overall area of forest cover in the landscape (Martensen et al., 2008).

The distribution ranges of large wild herbivores, such as deer, have been increasing globally in recent decades and they are now considered to be overabundant in temperate woodlands (e.g. Côté *et al.*, 2004; Apollonio *et al.*, 2010). The ecological impacts of this overabundance include effects on the growth and survival of many herbs, shrubs, and tree species; modification of patterns of vegetation dynamics; impacts, at cascading levels, on insects, birds and other mammals; reducing plant cover and diversity; changing of nutrient and carbon cycles; and redirection of succession to shift future canopy composition (Côté *et al.*, 2004). The intensity of grazing can have a critical impact on forest biodiversity in an ecosystem (Fuller and Gill, 2001; Côté *et al.*, 2004) which has a knock-on effect on biotic and abiotic components and ecosystem functioning (Pollard and Cooke, 1994; McShea and Rappole, 2000; Rooney and Waller, 2003; Côté *et al.*, 2004; Allombert *et al.*, 2005; Bugalho *et al.*, 2011). Impacts should not, however, be seen with simplistically negative connotation, as many impacts associated with large herbivores play vital roles in woodland survival, such as importance for tree regeneration (e.g. Vera, 2000), maintaining vegetation heterogeneity in the habitat (Kelly, 2000; Perrin *et al.*, 2011), and maintaining habitats suitable for a wide range of taxa (McShea and Rappole, 2000; Stewart, 2001; Oxbrough *et al.*, 2005; Díaz, 2006). Semi-natural woodlands are globally important ecosystems and are being impacted through a range of anthropogenic activities which induce vegetation changes, including climate change, invasive species, fire, logging, agriculture and overgrazing. Investigation of the vegetation community data across several decades will allow for increased understanding of the functioning of the woodland ecosystem. Forest policy in Ireland aims to promote the continued regeneration of native woodlands while maintaining and improving a range of woodland biodiversity and ecosystem functions. In the case of plantation forests, biodiversity can be optimised through the management of large herbivores. Management methods should include the identification of current large herbivore impacts, assessment of aims for increasing woodland biodiversity, and alteration or maintenance of large herbivore populations through culling and fencing.

The development of forest management practices and policies which account for climate change impacts is essential to maintaining the stability and sustainability of the forest sector in Ireland (Department of Agriculture Food & the Marine, 2010). Climate change affects forest ecosystems through effects on tree growth, carbon sequestration, species composition and species interactions which potentially threaten the

services currently provided by our forests. Predicted impacts of climate change on biodiversity in temperate oceanic regions such as Ireland include a decline in cold-tolerant species and shifts in the geographic range of many plant and animal species (Lindner *et al.*, 2010). Forest management and policies must take the impacts of climate change into account as forests will have to adapt to changes in climate and increased risk of extreme weather events (Lindner *et al.*, 2010). Adaptive management strategies are required for sustainable management of forests under future climate change in order to ensure the persistence of many species and related ecosystem services (Heller and Zavaleta, 2009). There is an urgent and on-going requirement for scientific research to address knowledge gaps of responses and adaptive capacity of our forests to guide adaptive management (Heller and Zavaleta, 2009; Mori *et al.*, 2013).

Objectives

The objectives of this project, as outlined in the project proposal were to:

1. Determine a set of tested indicators of forest biodiversity for the variety of forest types in Ireland: the tool kits for biodiversity assessment.
2. Produce practice protocols for forest managers and practitioners based on the outputs of the FORESTBIO, RHODO and HEN HARRIER projects.
3. Determine forest condition criteria for Irish Native Woodland Scheme forests.
4. Support a close collaboration for forest biodiversity research between Britain and Ireland that will lead to synergistic information exchange.
5. Predict the future species composition of the variety of Irish forests that will prevail under different climate scenarios.
6. Continue the monitoring of permanent plots established during the BIOFOREST project.

Project structure

This project was funded by COFORD for a period of 4 years from 2010 to 2013 as part of the PLANFORBIO Research Programme. The research was divided into a number of Work Packages, each of which addressed a specific aim of the project as shown in the table below.

Work Package No.	Work Package Title
1	Long-term monitoring
2	Testing biodiversity indicators
3	Indicators of woodland biodiversity potential
4	Forest connectivity
5	Impacts of large herbivores
6	Climate change impacts

The findings of the research are presented in this report as a series of chapters, each of which deals with a topic of particular interest to biodiversity conservation and sustainable forest management in Ireland.

Chapter 1

Afforestation and biodiversity

Sandra Irwin, Anke Dietzsch, Linda Coote, Conor T. Graham, Lauren Fuller, Mark W. Wilson, Daniel L. Kelly, Fraser J.G. Mitchell, Thomas C. Kelly and John O'Halloran
Work Package 1A

In landscapes where the area of plantation forest is increasing the impact of this expansion on biodiversity is an important consideration in the selection of sites for afforestation. The current study was designed as a follow-on to the Irish BIOFOREST forest biodiversity research project (2000-2006), to provide a direct investigation of the effects on biodiversity of forest plantations established on agriculturally improved grasslands, wet grasslands and peatlands. During the first seven years following planting, afforestation had a positive effect on plant, invertebrate and bird biodiversity in highly managed improved grassland sites, due to forest-associated species moving into the sites as a result of the increase in shrub cover. The effects of afforestation on biodiversity varied according to preceding land use. Plantations established in landscapes dominated by intensively managed agriculture were found to increase landscape scale biodiversity. However, afforestation of natural or semi-natural areas was more likely to have a negative impact on biodiversity conservation. Scientific studies of this kind support the integration of biodiversity maintenance with timber production, which is a goal of national and global afforestation policies and this research highlights the requirement for long-term investment in biodiversity monitoring following land use change.

Background

In Ireland commercial afforestation is taking place at a faster pace than in any other EU country and commercial plantations represent the majority of the forested area. This expansion of forest area must be managed in a manner that is ecologically sustainable and without negative environmental impact (Department of Agriculture Food & the Marine, 2010). Ireland's forest estate is dominated by exotic conifer species such as Sitka spruce, planted on former grassland or peatland (Forest Service, 2007). These plantation forests can have major impacts on the biota that existed in the previously unforested habitats (Mitchell, 2000; Lindenmayer *et al.*, 2003) and many studies report on the impacts of commercial afforestation on the biodiversity of pastures and grasslands (see Felton *et al.*, 2010 for review). The majority of these studies, however, come from the Asia-Pacific region and are of limited relevance to the situation in Europe. Most studies on the effects of afforestation on biodiversity in the temperate region of the Northern Hemisphere have focused on a single taxonomic group such as birds (Baguette *et al.*, 1994; Pithon *et al.*, 2005), invertebrates (Butterfield, 1997) or plants (Rusina *et al.*, 2011). While some studies have used a multiple taxon approach, these are not based on long-term monitoring (Lachance *et al.*, 2005; Paillet *et al.*, 2010). The only study that investigated changes in plant and bird diversity in a long-term monitored conifer plantation was limited to a single study site (Sykes *et al.*, 1989), making generalisations difficult to derive. Furthermore, existing scientific studies tend to treat all pasture land as one homogenous group and fail to take into account biodiversity variation among different grassland pre-afforestation habitats – variation which may have a major bearing on subsequent biodiversity change. While grasslands managed at low

intensity can have high biodiversity, this does not always hold true for intensively managed grasslands. This important difference has largely been ignored by previous studies, and data on the effects of afforestation in different European habitats are limited, making site selection for afforestation an area requiring further exploration. Resolution of the conflict between biodiversity conservation and forest expansion must be based on scientific evidence. In this regard, empirical information on the impact of afforestation in different habitats is required to inform the selection of suitable afforestation sites for forest expansion.



Study site at Kilbraugh, Co. Kilkenny prior to afforestation in 2002 (OSI) and 10 years after planting (Bing Maps).

Comparative studies of forest biodiversity over time or between treatments are typically indirect in nature and they aim to infer a temporal trend from studies that measure biodiversity in different locations at a single point in time (Pickett, 1989). These space-for-time substitution studies (also referred to as chronosequence studies) have long been conducted in forest biodiversity research due to either convenience or necessity. While they are useful in many respects including the speed at which data can be made available and independence from climatic or stochastic impacts at the time of sampling, they cannot overcome the potential confounding effects of biotic and abiotic differences between study sites (Walker *et al.*, 2010). In order to be truly effective, the assessment of the impact of forest management practices on forest biodiversity must include long-term monitoring of the outcomes of alternative management practices. The identification of ecologically responsible, sustainable, management strategies requires appraisal of empirical data derived from such long-term monitoring studies, together with information on temporal trends in forest biodiversity (Gardner, 2010). The scope of long-term forest biodiversity monitoring studies is specific to the type of management practice and type of forest under investigation, particularly when considering forest rotation length in the experimental design. This is reflected in existing research projects such as the EMEND (Ecosystem Management Emulating Natural Disturbance) forest biodiversity project in Canada. The large-scale EMEND project aims to determine how forest management practices can best maintain biodiversity, and has been planned to run for one complete stand rotation of 80-100 years (Spence *et al.*, 2008). To date, no such studies have been undertaken in an Irish context, where assessments of the impacts of forest management practices on biodiversity have been short-term in nature and typically based on space-for-time substitution (Gittings *et al.*, 2006; Oxbrough *et al.*, 2010; Sweeney *et al.*, 2010b).

Objectives

This current study is the first time that the long-term monitoring approach has been used to assess the impacts of management on forest biodiversity in Ireland. Long-term biodiversity monitoring plots were established in Irish forests in 2002 during the BIOFOREST project (Biodiversity in Irish Plantation Forests) to study the impacts of afforestation (Iremonger *et al.*, 2006). The current study describes data from the 2010 resurvey of biodiversity in these forests. This is the first step towards achieving a long-term study of the effect of afforestation on biodiversity.

The specific objectives of this study were to:

- Complete a second survey of forests at BIOFOREST afforestation 'tracking study sites'.
- Investigate the effects of afforestation on biodiversity in a number of different habitats.

Methodology

Experimental design & site selection

The impacts of afforestation on biodiversity were investigated using two different methodologies:

1. **Paired chronosequence** (space-for-time substitution) at paired planted and unplanted sites.
2. **Within-site tracking** (long-term monitoring at permanent plots) at sites pre-planting and post-planting.

Within-site tracking was investigated using archived data collected during the earlier BIOFOREST project (2000-2006) (pre-planting) together with new data collected during the current project (post-planting). Paired chronosequence investigation was conducted using data collected at planted and unplanted sites during the BIOFOREST project. Study site locations covered a broad geographical and ecological range across the island of Ireland (Figure 1).

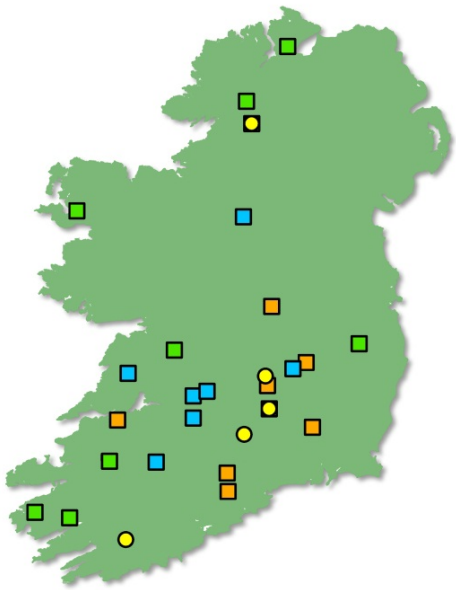


Figure 1. Study site locations: ■ = Chronosequence peatland site pairs (surveyed 2002 & 2004; ■ = Chronosequence improved grassland site pairs (surveyed 2002 & 2004); ■ = Chronosequence wet grassland site pairs (surveyed 2001 – 2004); ● = Within-site tracking sites (surveyed pre-planting in 2002 & post-planting in 2010).

The impacts of afforestation on the biodiversity of agriculturally improved grassland, wet grassland and peatland sites were investigated using the paired chronosequence method at planted and unplanted study sites. Data on the biodiversity of plants, invertebrates, birds and vegetation structure were collected from 24 pairs of chronosequence sites, eight pairs each in improved grassland, wet grassland and peatland during the BIOFOREST project (Iremonger *et al.*, 2006). Each pair consisted of an unplanted study site and a 5-year-old planted, first rotation Sitka spruce pre-thicket plantation. All site pairs were closely matched in terms of environmental conditions such as slope, altitude, drainage and soil type; they were within 5km of each other, with the majority of pairs (18) being immediately adjacent to each other.

The impacts of afforestation on the biodiversity of agriculturally improved and wet grassland sites were also investigated using the within-site-tracking method at permanent plots which were surveyed before and after afforestation. Plant, invertebrate, bird and vegetation structure surveys were conducted at five study sites prior to afforestation (pre-planting) in 2002 during the BIOFOREST project (Iremonger *et al.*, 2006), and during the current project in 2010 when plantations were between 4 and 7 years old (post-planting). These sites were planted with Sitka spruce, often mixed with other conifer species such as Japanese larch (*Larix kaempferi*) and broadleaf species such as alder (*Alnus glutinosa*).

Data collection

Surveys of plant, invertebrate and bird diversity and vegetation structure were carried out between May and September across a number of years between 2001 and 2011. All planted and unplanted paired chronosequence sites were surveyed in 2002 and 2004 and within-site tracking sites during 2002 (pre-planting) and 2010 (post-planting).

The percentage cover presence of all vascular plant species and of any bryophyte and lichen species covering an area of 50cm^2 or more was recorded in three 10m x 10m plots at each study site. Species cover of all identifiable plants including bryophytes and lichens was also recorded in two 2m x 2m subplots within each 10m x 10m plot. In these smaller plots the percentage cover was estimated to the nearest 5%, below which two categories (3.0% and 0.5%) were recorded. In addition to plant, cover, the height and percentage cover of each vegetation layer, and the cover of bare soil or rock, and leaf litter and standing water were also recorded.

For the within-site-tracking study, active ground-dwelling spiders were sampled using pitfall traps (plastic cups 7cm in diameter and 9cm tall), filled to a depth of 3cm with ethylene glycol. Each trap was placed in a pre-excavated hole ensuring that the rim was just below ground level. Five pitfall traps were established in a grid arrangement (hereafter called open plots) and placed adjacent to the three vegetation plots. Three additional transects with five pitfall traps in a linear arrangement were placed adjacent to hedgerows (hereafter called hedgerow plots), in order to explore the effect of afforestation on spider diversity within this important supplementary habitat type, which is typically found at field margins. Trap contents were collected three times over a period of approximately 63 days in all sites in summer 2010 and summer 2011. Further information on invertebrate biodiversity surveys can be found in Fuller *et al.* (2013a).

Hoverflies were also sampled at within-site-tracking sites using Malaise traps (Figure 2) and following the protocol in Smith et al. (2006). Two Malaise traps were established approximately 10m apart per site and

were placed in linear un-shaded areas. The contents of the traps were collected on three occasions at three week intervals between May and July in summer 2010. For further information see Fuller *et al.* (2013a).



Figure 2. Malaise trap used to sample hoverflies at study site in Kilbraugh, Co. Kilkenny.

At within-site-tracking and chronosequence study sites, birds were counted at 5 to 9 points (depending on site size) within each site using point counts, with a distance of 100m between point count locations. The number and species of all birds detected within a 50m radius, along with their estimated position and distance from the observer, were recorded using range-finding binoculars.

All bird counts were conducted twice during the breeding season in each year between 0800 hours and 1800 hours, with each count lasting 10 minutes. Birds in flight were excluded from the analyses. Bird surveys were not conducted in heavy or persistent rain, or in winds greater than Beaufort scale four. Bird densities for each point were then estimated using the computer programme *Density*. Further information on bird survey methods can be found in (Graham *et al.*, 2013).

Data analysis

For each 10m x 10m ground vegetation plot and the summed total of the two 2m x 2m subplots where they were also used, total plant species richness (SR), forest-associated plant SR, vascular plant SR and bryophyte SR were calculated. The effects of afforestation were analysed using generalised linear mixed models (GLMMs) using Poisson distribution with a log-link function, with backwards model selection and the dredging function and community data were explored using non-metric multi-dimensional scaling (NMS) ordination. Important factors governing composition and the distribution in ordination space were identified using permutational multivariate analysis of variance (PERMANOVA) analysis.

The difference in spider and hoverfly species richness between pre-planting and post-planting in open and hedgerow plots was analysed using generalised linear mixed modelling (GLMM). This analysis was carried out on total species richness of spiders and hoverflies and also for the identified sub-groups of open specialist spiders, forest specialist spiders, open specialist hoverflies, forest specialist hoverflies and aquatic habitat specialist hoverflies. Spider and hoverfly species composition was examined among plots using NMS ordination and permutational multivariate analysis of variance (PERMANOVA).

The within-site-tracking data collected during bird point counts were used to derive estimates of bird density and species richness at each survey point. Species richness was calculated as the cumulative number

of species recorded over both visits at each survey point in each sampling year. Differences between pre- and post- afforestation states at each individual sampling point were assessed using paired t-tests. Where assumptions of normality or homogeneity of variance were not met, differences between paired samples were tested using Wilcoxon's Signed Rank Tests. Generalised linear models (GLMs) were used to identify those habitat variables related to species richness and conservation value at each sampling point. NMS analysis was carried out to determine between site patterns in the bird community assemblages using Sørensen distance measures.

The bird diversity data from the paired chronosequence study were used to test the effect of afforestation on bird total density and species richness, birds of conservation concern (BOCC) density and species richness and the effect of preceding land use using general linear mixed models (GLMM). The hypothesis that the impacts of afforestation on bird community metrics are dependent not only on the gross difference between planted and unplanted sites, but also variations in the bird assemblages supported in the pre-afforestation habitats, was tested using the responses of the bird communities to afforestation. Differences between bird metrics of paired planted and unplanted forest sites were calculated for each paired site in each of the three habitat types. Differences in the response of total bird density and species richness, and BOCC density and species richness, and site conservation value between each of the three habitat types were assessed using one-way ANOVAs with Bonferroni corrected post-hoc tests.

Cross-taxon analyses were conducted at the site level. Compositional data for all taxonomic groups were compared to each other employing procrustes analyses. Partial correlation analyses (controlling for habitat type and unplanted/afforested sites) were used to test whether species richness of each taxonomic group was related to the species richness of any other taxonomic group. In addition, the difference in species richness after afforestation was calculated for each taxonomic group using each matching unplanted and afforested site pair. Partial correlations (controlling for habitat type) were then employed to investigate whether the species richness difference of each taxonomic group was related to the species richness difference of any other taxonomic group. For compositional and species richness data, correlations with r-values ≥ 0.7 were considered adequate for identifying meaningful surrogacy.

Results

Both within-site tracking and paired chronosequence space-for-time substitution showed that total plant species richness and vascular plant species richness increased in response to afforestation. By contrast non-vascular plant (bryophyte and lichen) species richness showed no significant response to tree planting. Total plant species richness and vascular plant species richness were significantly higher post-planting than pre-planting ($z = 4.91$, $P < 0.001$; $z = 4.24$, $P < 0.001$; Figure 3) at the study sites.

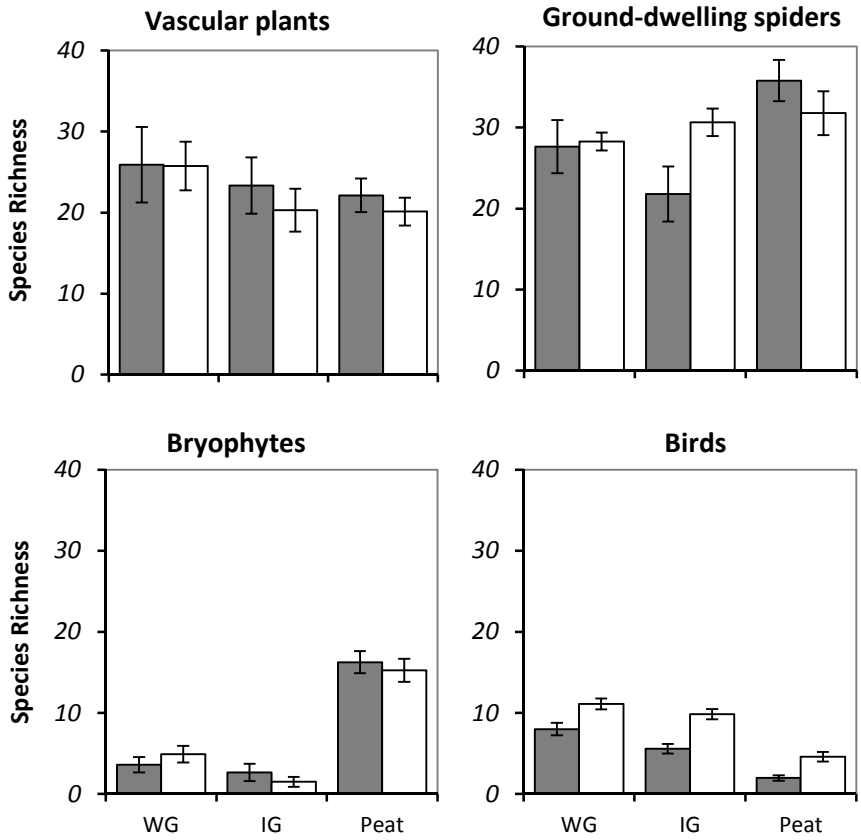


Figure 3. Mean species richness (± standard error) of the investigated taxonomic groups at within-site-tracking sites. Bars in dark grey indicate mean values at unplanted sites and bars in light grey indicate mean values at afforested sites. WG=wet grasslands, IG=improved grasslands, Peat=peatlands.

Vascular plant species richness was significantly higher post-planting than pre-plantings in 2m x 2m plots and 10m x 10m plots ($z = 4.24$, $P < 0.001$; $z = 7.12$, $P < 0.001$). Forest-associated species richness was also significantly higher post-planting ($z = 3.14$, $P < 0.01$; Figure 4). Analyses of both paired chronosequence and within-site-tracking data indicated that, in unplanted sites, wet grassland habitats were significantly more species rich than improved grasslands before planting ($z = 7.12$, $P < 0.001$ for chronosequence study).

Ordination of plant community data in the 2m x 2m plots showed that afforestation initiated a shift in plant community composition which was correlated with the increase in sapling and tree cover ($F_{1,80} = 4.53$, $P < 0.001$). The strength of this effect differed between wet and improved grasslands (tree & sapling cover x grassland type, $F_{1,80} = 4.67$, $P < 0.001$). Plant communities were also differently structured depending on the grazing intensity ($F_{2,80} = 2.57$; $P < 0.001$); plots with moderate and high grazing intensity were significantly different from plots that were not grazed. A significant interaction was found between grassland type and the tree species planted. In improved grasslands, plots afforested mainly with conifers were significantly different in their plant community composition compared with pre-planted sites ($t_{1,35} = 1.99$, $P < 0.01$); whereas, in wet grasslands, plant assemblages in plots afforested with broadleaf tree species significantly differed from pre-planting ($t_{1,18} = 1.47$, $P < 0.05$). Ordination of plant community data in the 10m x10m plots mirrored the 2m x 2m plot ordination. Plant assemblages differed between wet and improved grassland sites and chronosequence and tracking sites did not separate in ordination space (Figure 5).

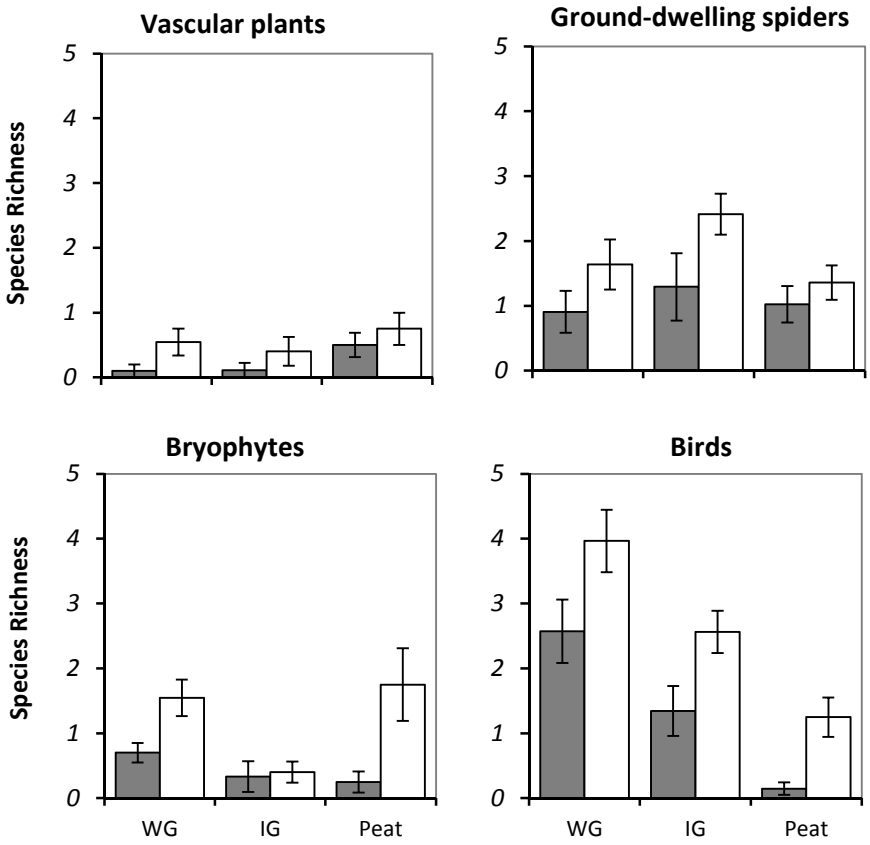


Figure 4. Mean forest-associated species richness (± standard error) of the investigated taxonomic groups at within-site-tracking sites. Bars in dark grey indicate mean values at unplanted sites and bars in light grey indicate mean values at afforested sites. WG=wet grasslands, IG=improved grasslands, Peat=peatlands

A total of 72 species of ground-dwelling spider (n = 909) and 52 species of hoverfly (n = 1,211) were recorded in pre-planting forest sites in the within-site-tracking study. At post-planting study sites 93 species of spider (n = 2,186) and 63 species of hoverfly (n = 617) were recorded. Both pre-and post-planting the spider species assemblage comprised mainly of habitat generalists and more open habitat specialists were present than forest specialists. In contrast the hoverfly species recorded both pre- and post-planting were a mix of open, forest and aquatic habitat specialists with very few generalist hoverfly species recorded. Data from the chronosequence study showed no difference in total spider species richness after planting however, forest-associated spider species significantly increased after planting with improved grasslands showing the largest increase in species richness Figure 4. Initial data exploration indicated very few differences between the spider and hoverfly communities of improved and wet grassland from the five within-site-tracking sites studied here. Therefore, both grassland types were grouped in order to examine broad trends of the effect of afforestation on these species groups. Analysis of the within-site tracking data revealed that total spider species richness was significantly higher in both open and hedgerow plots in post-planting than in pre-planting sites (open plots $t_{11} = 2.98$, $P < 0.01$; hedgerow plots $t_5 = 5.29$, $P < 0.01$). Forest-associated spider species richness also increased significantly in open plots in post-planting sites ($t_{11} = 3.81$, $P < 0.01$). No difference in open-associated spider species richness was detected in either open or hedgerow habitats. Afforestation had no significant effect on the measured hoverfly species metrics. Further information on the responses of spiders and hoverflies to afforestation can be found in Fuller *et al.* (2013a).

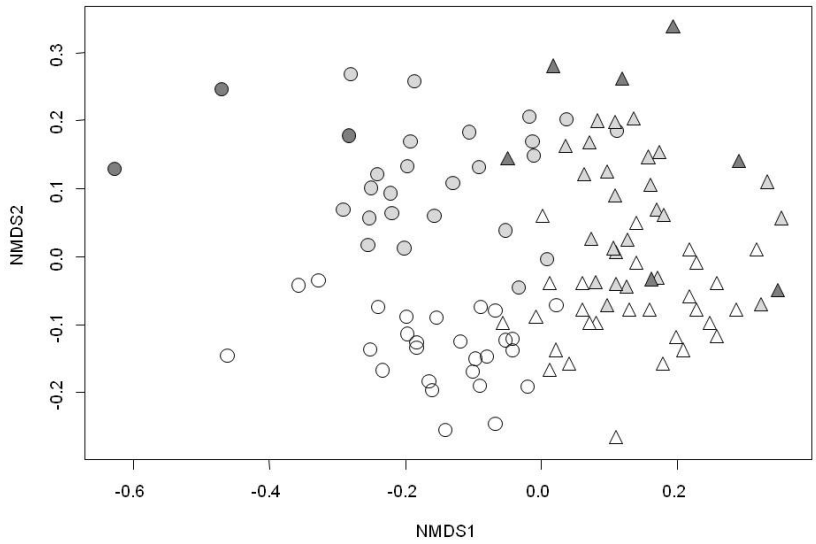


Figure 5. NMS ordination of the 10m x 10m ground vegetation plots: ○ = Unplanted Improved Grassland sites; ◌ = Improved Grassland sites planted mainly with conifers; ● = Improved Grassland sites planted mainly with broadleaves; △ = Unplanted Wet Grassland sites; ◬ = Wet Grassland sites planted mainly with conifers; ▲ = Wet Grassland sites planted mainly with broadleaves. Final stress for 2-dimensional solution = 0.19.

The responses of ground-dwelling spider species assemblages to afforestation differed between open and hedgerow habitats ($F_{1,46}$ = 1.83; P ≤ 0.05). While spider communities changed significantly in open habitat plots post-planting ($t_{1,4.1}$ = 1.59; P ≤ 0.05), no significant difference was seen in hedgerow plots ($t_{1,4\cdot1}$ = 1.21; P ≥ 0.05, Figure 6). Hedgerow spider communities were significantly different from open habitat communities in pre-planting sites ($t_{1,22}$ = 1.89; P < 0.001) but these differences disappeared post-planting on ($t_{1,24}$=1.10; N.S.) when communities became more homogenous (Figure 6). There was also a significant difference in hoverfly assemblages between pre- and post-planting sites ($F_{1,9}$ = 1.98, P < 0.01) with the assemblages becoming more homogenous post-planting (Figure 7).

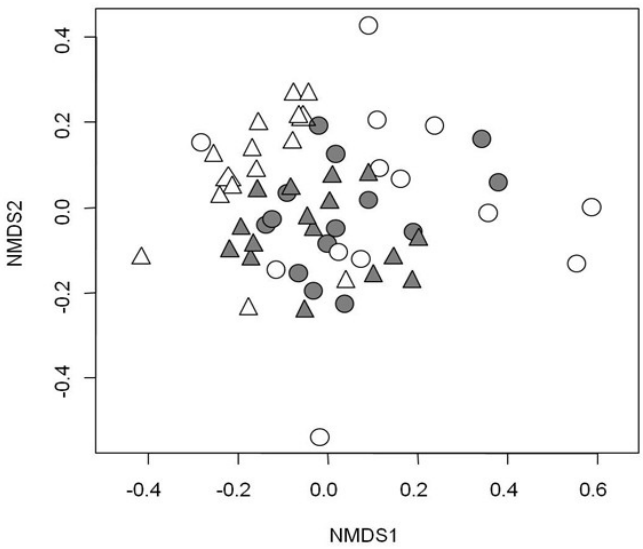


Figure 6. NMS ordination of ground-dwelling spider assemblages in the within-site-tracking study. △= Open plots pre-planting; ◬= Open plots post-planting; ○ = Hedgerow plots pre-planting; ◌ = Hedgerow plots post-planting. Final stress for 2-dimensional solution = 0.23.

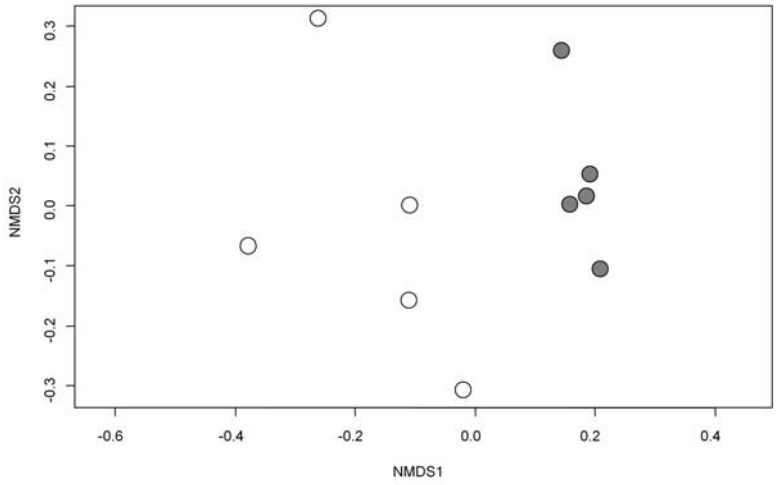


Figure 7. NMS ordination of hoverfly assemblages in the tracking study. ○ = pre-planting; ● = post-planting. Final stress for 2-dimensional solution = 0.09.

Thirty one bird species were recorded in the within-site-tracking study, of which five species (Grey Wagtail (*Motacilla cinerea*), Jackdaw (*Corcus monedula*), Skylark (*Alauda arvensis*) and Spotted Flycatcher (*Muscicapa striata*)) were recorded only in pre-planting sites. By contrast Blackcap (*Sylvia atricapilla*), Grasshopper Warbler (*Locustella naevia*), Goldfinch (*Carduelis carduelis*), Hooded Crow (*Corvus cornix*) and Lesser Redpoll (*Carduelis cabaret*) were all recorded exclusively in post-planting sites. A number of Amber listed species (Lynas *et al.*, 2007) were recorded including Spotted Flycatcher and Skylark at pre-planting grassland sites, Grasshopper Warbler at post-planting sites and Kestrel (*Falco tinnunculus*) and Swallow (*Hirundo rustica*) which occurred at both pre- and post-planting sites. There was a significant increase in bird species richness following afforestation (z = 3.44, P = 0.001; Figure 3) but no significant change in the density of birds (z = -1.7, N.S.).

There was considerably more variation between bird communities in pre-planting grasslands sites in comparison with the post-planting bird assemblages at the same sampling locations. This is indicated by the wider spacing of the pre-planting sites in the NMS ordination space compared with the post-planting sites, which occupy a relatively tight cluster in the ordination. This indicates that bird communities in these sites become less diverse following afforestation (Figure 8).

The paired chronosequence study revealed that the response of bird communities to afforestation varied markedly between pre-planting habitats. Bird densities and species richness in peatland sites, where the previous land use intensity was lowest, were negatively impacted by forest plantations while planting on previously intensively managed grassland sites increased richness and densities. Total bird density in unplanted sites was significantly different between habitat types ($F_{2,21}$ = 15.7, P < 0.001); it was highest in wet grassland and lowest in peatland sites and this trend was retained in planted sites, although the different in density between the wet and improved grassland sites was more of lower magnitude in planted sites. Total species richness also differed among unplanted sites of different habitat types ($F_{2,21}$ = 32.1, P < 0.001) which was also retained in planted sites ($F_{2,21}$ = 23.1, P < 0.001), and was highest in wet grassland and lowest in peatland sites. In terms of bird communities there was considerably more variation between unplanted grassland sites in comparison to the planted sites. This is indicated by the wider dispersal of the unplanted sites in the NMS ordination space compared to the planted sites, which occupy a relatively tight

cluster (Figure 9). This indicates that bird communities in these sites become more similar in response afforestation.

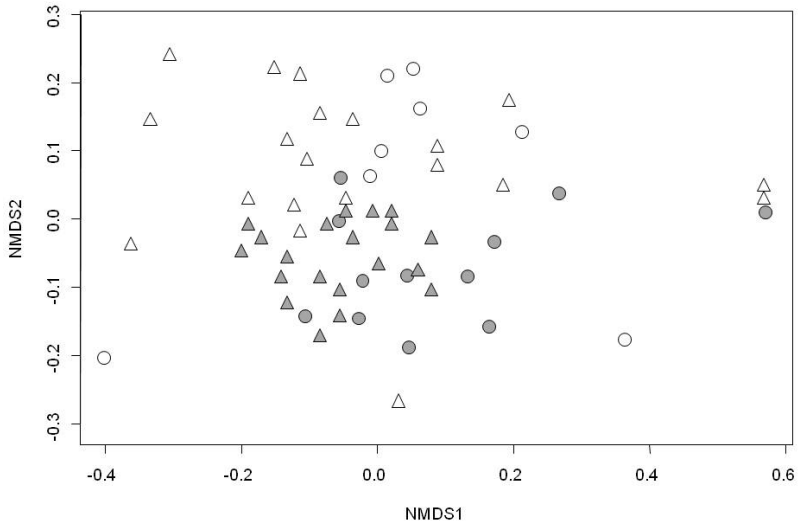


Figure 8. NMS ordination of bird assemblages in the within-site-tracking study. △= Wet Grassland sites pre-planting; ▲= Wet Grassland sites post-planting; ○ = Improved Grassland sites pre-planting; ● = Improved Grassland sites post-planting. Final stress for 2-dimensional solution = 0.18.

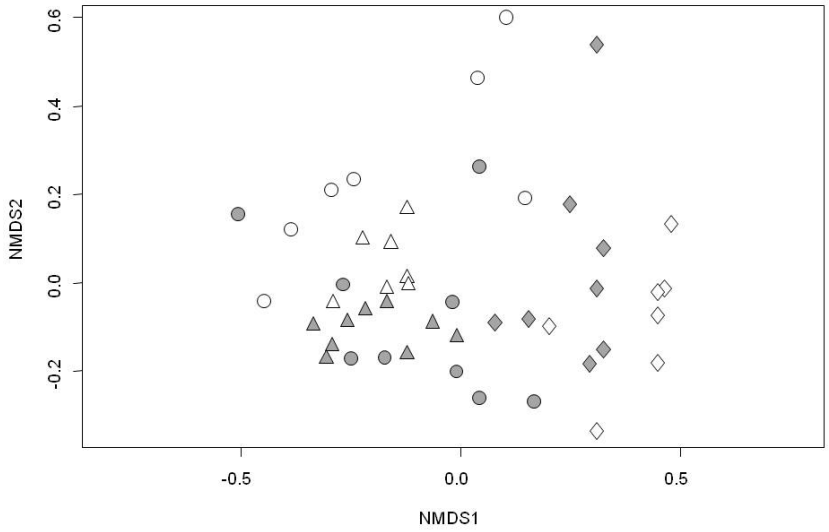


Figure 9. NMS ordination of bird assemblages in the chronosequence study. △= Wet Grassland sites pre-planting; ▲= Wet Grassland sites post-planting; ○ = Improved Grassland sites pre-planting; ● = Improved Grassland sites post-planting; ◇ = Peatland sites pre-planting; ◆ = Peatland sites post-planting. Final stress for 2-dimensional solution = 0.18.

Although each taxonomic group showed a significant shift in species composition and differences in diversity after afforestation, their responses differed from each other. With the exception of a significant partial correlation between vascular plant and bryophyte species richness ($t = 5.02$; $r = 0.58$; $P < 0.001$), none of the taxonomic groups' species richness was correlated with each other. Similarly, richness within functional groups like web-building and cursorial spiders was not significantly correlated with insectivorous

bird or plant species richness except for a significant negative correlation between bryophytes and cursorial spiders($t = -2.04$; $r = -0.29$; $P < 0.05$). The difference in vascular plant species richness was negatively correlated to total spider species richness difference ($t = -2.51$; $r = -0.48$; $P < 0.05$) but positively correlated to the bryophyte SR difference ($t = 4.77$; $r = 0.69$; $P < 0.001$).

Partial correlations of forest-associated species richness did not reveal any significant relationship between the investigated taxonomic groups while partial correlation of differences in forest-associated species richness after afforestation showed a positive relationship between forest vascular plants and forest bryophytes ($t = 2.11$; $r = 0.39$; $P < 0.05$). Procrustes analysis revealed significant correlations between ordinations for all pairs of taxonomic groups, indicating that on a broad spatial scale differences in species composition among sites of any taxonomic group were reflected by all other groups. Correlation coefficients were ≤0.7 in all cases.

Discussion

This study has shown that species richness of plants, ground-dwelling spiders and birds as well as community composition of these taxonomic groups and of hoverflies changed over the course of the initial years after afforestation of grassland habitats (Fuller *et al.*, 2013a; Graham *et al.*, 2013). In the early years following afforestation there was no significant effect of the tree species planted (conifer vs. broadleaf species) on either plant species richness but on the plant community or beta diversity. However, sites planted with broadleaf tree species tended to have different plant species assemblages compared with sites planted with conifer species or unplanted sites. Hence, afforestation affects affected plant diversity and community composition and this effect is was related to changes in tree and shrub cover, differences in the afforested habitat, the change in management regimes and the tree species planted.

There was a positive effect of afforestation on total spider species richness and forest specialist spider species richness. Analysis of the hoverfly data showed there was no effect of afforestation on total species richness or species richness of habitat specialists. The species composition of ground-dwelling spiders and of hoverflies changed following afforestation. Prior to afforestation supplementary hedgerow habitats supported more forest-associated species, but after afforestation supplementary hedgerow habitats no longer provided additional biodiversity benefits to ground-dwelling spiders within the habitat. Analysis of the overall bird community data revealed a positive impact of afforestation on both the overall density and species richness of birds (Graham *et al.*, 2013). However, when these parameters were examined by habitat type, analysis showed that afforestation increased the diversity and overall density of birds in the improved grassland and peatland sites, but only species richness and not total density in the wet grassland sites. Observations on bird communities were similar to the results on overall bird density and species richness and appear to be a result of the differences in the pre-afforestation vegetation composition in wet grassland sites compared to the peatland and improved grassland sites.

Although each taxonomic group showed a significant shift in species composition and differences in diversity after afforestation, their responses to afforestation differed from each other. With the exception of a significant correlation between vascular and non-vascular plant species richness, none of the taxonomic groups' species richness was correlated with the others. Similarly, richness within functional groups like web-building and cursorial spiders was not significantly correlated with insectivorous bird or plant species richness. With a few exceptions, these results support the findings of previous studies on Irish afforestation

(Oxbrough *et al.*, 2006a; Smith *et al.*, 2006; Buscardo *et al.*, 2008; Graham *et al.*, 2013). At this early stage of the forest cycle, afforested sites retained much of the original flora and fauna of the preceding grassland habitats, but new species were also recorded in the forest habitat. A number of factors that are associated with this change in biodiversity and community composition were identified.

Grazing intensity was significantly related to bryophyte species richness and plant community composition. This effect was most strongly expressed in improved grasslands. A reduction in grazing pressure associated with afforestation resulted in the plant community becoming more heterogeneous and is reflected in the loss of some species and the immigration of new species. This change in ground vegetation had, in turn, a positive effect on both ground-dwelling spider diversity and bird diversity. Structural heterogeneity and diversity have been shown to be positively correlated to invertebrate diversity (Greenstone, 1984; Humphrey *et al.*, 1999; Oxbrough *et al.*, 2005) and to bird diversity (MacArthur and MacArthur, 1961; Sweeney *et al.*, 2010b). Wet grassland habitats revealed higher plant species richness than improved grassland habitats. The initially higher plant diversity of unplanted wet grassland habitats supported a more diverse and specialist spider and bird fauna than the species-poor improved grassland habitats (Oxbrough *et al.*, 2007; Wilson *et al.*, 2012). The greater change of structural heterogeneity in improved grassland undoubtedly influenced changes in communities of the investigated taxa so that the positive effect of afforestation on e.g. bird diversity was stronger in improved grassland sites than in wet grassland sites.

As the young plantation becomes established, the increase in tree and sapling cover resulted in a higher structural diversity. Shading by young planted trees reduces the cover of forbs (Wallace and Good, 1995), in particular of open habitat species, and creates more favourable conditions for forest-associated species. Forest-associated species richness of plants and invertebrates increased after afforestation, and for total forest plant species richness this increase was greater in plots planted with broadleaf tree species compared to plots planted with conifers. In contrast to the negative impact of tree and sapling cover on vascular plant species richness, bryophyte diversity was not related to the initial increase in tree and sapling cover confirming that bryophytes diversity peaks at intermediate light levels (Bergamini *et al.*, 2001a).

Cross-taxon analysis revealed that although the taxonomic groups investigated mirrored each other's responses to afforestation more clearly with regard to species composition than to species richness, species composition were yet mainly affected by habitat type. Although changes in vascular plant species richness were positively associated with changes in bryophyte species richness, it is surprising that changes in vascular plant species richness were negatively related to changes in spider species richness. Spider diversity reflects the suitability of the habitat in terms of the litter cover and heterogeneity of vegetation (Oxbrough *et al.*, 2005; Oxbrough *et al.*, 2012). A less diverse plant community may still involve greater heterogeneity at this early stage in the forest cycle, e.g. in cases where the dominance of single species such as bramble (*Rubus fruticosus*) led to a species-poor community (Buscardo *et al.*, 2008) but increased both shrub cover and heterogeneity of vegetation. In contrast, changes in forest bryophyte diversity were positively related to changes in forest-associated plant diversity indicating an overall change from grassland to forest habitat.

Recommendation 1: Avoid planting of forests on peatland and biodiversity rich grassland, and focus new planting on intensively managed and/or structurally homogenous grassland where the opportunities for biodiversity enhancement are greatest.

Afforestation of grasslands provided an initial increase in the biodiversity value of the sites. This increase is unlikely to persist after canopy closure, which has been shown to result in a major change in species composition and a decrease in species richness for all of the taxa investigated (Oxbrough *et al.*, 2005; Wilson *et al.*, 2006; Sweeney *et al.*, 2011; Moore, 2012; O'Connell *et al.*, 2012). Managing for open spaces and areas that contain grassland characteristics and other characteristics typical for early afforestation within forest plantations may sustain the increased biodiversity value of these habitat types at a landscape level (Peterken and Francis, 1999; Oxbrough *et al.*, 2006a; Wilson *et al.*, 2006).

Recommendation 2: Management of plantation forests for biodiversity enhancement should aim to prevent extensive areas of canopy closure.

Conclusions

Afforestation of grassland areas will increase over the coming decades and this study demonstrated an initial increase in site biodiversity value following planting. Within site assessment of the effect of afforestation had not been researched in Ireland until now. In this context, the results are important in understanding the impact of afforestation. In general, the number of forest-associated species increased following planting, particularly in the presence of planted broadleaf trees. Factors such as grazing intensity, grassland type, tree and sapling cover and ground cover explained some of the observed changes. However, the consequences for biodiversity of afforestation will continue to develop over time, and current management practices, which allow complete closure of the forest canopy at later stages of the forest cycle, may mean that the advantages for biodiversity of afforestation, even where observed, will not be maintained in the longer-term (Smith *et al.*, 2006; Moore, 2012). Forest management to maintain the positive effects of afforestation should focus on providing low canopy cover through thinning. Alternatively, management could focus on the forest scale and integrate stands of different forest stages together with grassland habitats in forest plantations. Further, investigation of this is required through continued monitoring of these sites as the combined priorities of biodiversity and timber production are critical issues for future afforestation in Ireland, and can be achieved through management practices. It will be possible to increase the landscape scale biodiversity benefits of future forest expansion by ensuring that afforestation is planned and managed to maximise availability of the most beneficial forest stages over space and time.

Our analysis and the similarity in findings between the BIOFOREST study on chronosequence sites and the results on long-term monitoring sites presented here indicate that chronosequence studies during the early stages of the forest cycle seem to be a valid substitute for long-term monitoring sites, at least for most taxonomic groups (Oxbrough *et al.*, 2006b; Smith *et al.*, 2006). However, tracking long-term monitoring sites through time is the superior methodology for investigation of fine-scale, within-community trends and the interplay between functional, structural and compositional parameters (Pickett, 1989).

Chapter 2

Biodiversity of forest roads

Sandra Irwin, Anke Dietzsch, Linda Coote, Conor T. Graham, Lauren Fuller, Mark W. Wilson, Daniel L. Kelly, Fraser J.G. Mitchell, Thomas C. Kelly and John O'Halloran
Work Package 1B

The potential of forest roads to enhance habitat diversity within plantation forests is an important consideration in the management of production forests for biodiversity conservation. When properly managed these open spaces allow structurally diverse vegetation to grow at the road-verges, increasing the diversity of available habitat types within forests. Road-verges may therefore provide habitat that can support greater species abundance and richness, increasing overall forest biodiversity. Plant, invertebrate and bird diversity were investigated along forest road edges in young plantation forests in Ireland and examined the consequences of doubling the standard forest road-width currently used in Ireland. Biodiversity surveys were conducted in eight Sitka spruce plantations one year after planting and again six years into the forest cycle. Although this study did not reveal a significant impact of increased forest road width on biodiversity during the initial stages following planting, it has served to quantify biodiversity associated with forest roads, to identify the principal drivers that influence this diversity and to provide important baseline data against which future surveys of the effect of road-width can be monitored.

Background

Open spaces are an important component of forest habitat and an important factor for biodiversity in planted forests (Peterken and Francis, 1999; Pedley *et al.*, 2013b). Unplanted areas contribute to biodiversity by allowing open habitat species to persist in sites following afforestation, providing habitat for specialists of forest edges and glades, and accommodating other species whose ecological requirements lie between the extremes of the shaded forest interior and the surrounding landscape (Carter and Anderson, 1987; Peterken, 1999; Peterken and Francis, 1999). In order to promote biodiversity in Irish forests, the Forest Biodiversity Guidelines (2000b) state that between five and ten per cent of forest area should be retained as open space in plantations over 10ha in size. Open spaces in plantation forests include discrete open spaces (glades), rides, plantation edges, and road networks.

Although the primary function of roads and rides is to provide access for timber extraction and other forest management activities, their importance for biodiversity is increasingly being recognised (Peterken, 1996; Gittings *et al.*, 2006; Oxbrough *et al.*, 2006a; Smith *et al.*, 2007; Roycroft *et al.*, 2008; Pedley *et al.*, 2013a), particularly as such areas represent the only permanent open spaces within some forests. This is in contrast to the negative effects of roads on biodiversity in extensively forested landscapes where the construction of new roads increases fragmentation, alters the physical and chemical environment, increases disturbance and increases the spread of invasive species (Avon *et al.*, 2013; Johansson *et al.*, 2013). In Ireland at present, the recommended width of forest road corridors is 15m from tree trunk to tree trunk in order to qualify for

inclusion as an Area for Biodiversity Enhancement (Ryan *et al.*, 2004). However, the shade cast by the canopy of adjacent crop trees can limit the biodiversity supported by linear open spaces in forests (Greatorex-Davies *et al.*, 1994; Gittings *et al.*, 2006; Oxbrough *et al.*, 2006a). In this regard, a number of authors have advocated the widening of linear open spaces to promote biodiversity (Warren and Fuller, 1993; Gittings *et al.*, 2006; Oxbrough *et al.*, 2006a). From a management perspective, the implementation of this recommendation is more practical at the planting stage than at a later phase of the rotation, It is also vital that monitoring be continued as the forest moves through the forest cycle (Iremonger *et al.*, 2006).



Forest road at a Sitka spruce forest study site in Cardtown, Co. Laois.

Objectives

Using data on plants, invertebrates and birds, the impacts of experimental widening of forest roads in second rotation Sitka spruce plantations on biodiversity during the early years of the forest cycle was examined. The specific objective of this work was to:

- Investigate the effects of forest road-width on biodiversity during the early forest cycle.

Methodology

Experimental design & site selection

The effect of doubling the width of forest roads was investigated at a suite of 8 study sites located in second rotation Sitka spruce plantation forests (Figure 10). Planting at these sites was undertaken during the winter of 2003/2004 at all except one site, which was planted in the following winter. At all sites the road structures were in place before planting.

Each site included two study road sections, a ‘standard’ treatment and a ‘wide’ treatment. Each of these treatments was at least 200m long, typically adjacent to each other and, as far as possible, differed only in the width of unplanted land at the side of the road. The ‘standard’ treatment represents current forestry practice under Irish forestry guideline (Ryan et al. 2004). In this treatment the total width of the forest road gap was 15m, consisting of a 5-6m open space gap at each side of the 3-5m wide road surface. The ‘wide’ treatment had a total width of the forest road gap of 30m, comprised of a 12-14m open space gap at each side of the 3-5m wide road surface. Therefore, the difference between treatments was in the width of the

open unplanted roadside space (hereafter referred to as 'road gap') at each side of the pre-existing road surfaces (Figure 11).

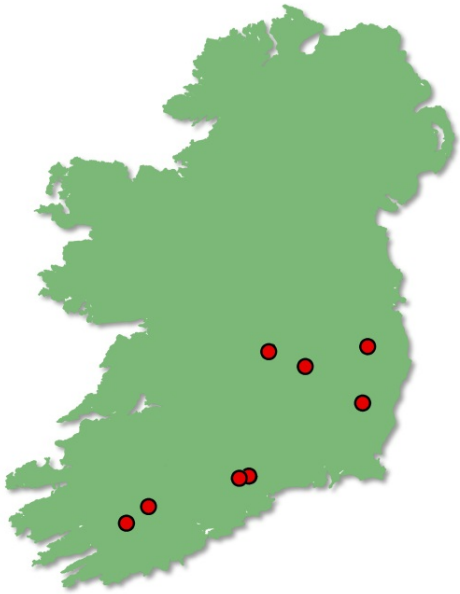


Figure 10. Forest road site locations (n = 8, surveyed in 2005 & 2010).

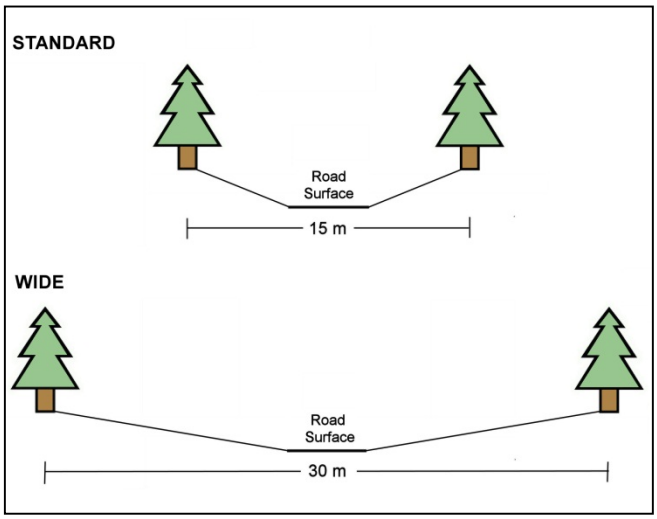


Figure 11. Schematic diagram of the two treatments in each of the road-width experimental sites. In the 'standard' road sections, there was a 15m gap from tree base to tree base on each side of the road, while in the 'wide' road sections, this distance was increased to 30m. The width of the road (3-5m) was the same in each treatment.

Data collection

Surveys of plant, invertebrate and bird diversity and vegetation structure were carried out between May and September during 2005 (one year post-planting, data collected during the BIOFOREST project) and 2010 (6 years post-planting). Six 20m long sub-sections (three in each of the standard and wide treatments) along the north, northeast or east sides of each road were selected. A list of plant species occurring between the

forest edges on both sides of the road and on the road surface was recorded and terrestrial and deadwood-inhabiting bryophyte and lichen species were recorded where they formed patches more than 50cm^2. Transects perpendicular to the road were centred on each of the six sub-sections. Three 4m^2 plots were surveyed along each transect: one adjacent to the gravel road surface (hereafter referred to as 'verge'), one at 3m from the forest edge ('open') and one at 2m into the forest for plants ('forest'). In each plot percentage cover data of all plants, bryophytes and lichens were collected as described for the 2m x 2m plots in Chatper 1A. In addition the number, species and height of tree saplings were also recorded.

Invertebrates were sampled at three plot types which were placed adjacent to the vegetation survey plots described above: 'Verge' (placed midway between the road edge and the tree-line), 'Open' (also placed midway between the road edge and the tree-line in the standard road-width and placed 3m before the tree-line in the wide treatment) and 'Forest' (placed 5m into the forest for invertebrates). Active ground-dwelling spiders were sampled using pitfall traps (plastic cups 7cm in diameter and 9cm tall), filled to a depth of 3cm with ethylene glycol. Each trap was placed in a pre-excavated hole ensuring that the rim was just below ground level. Three pitfall traps were placed 2m apart in a line at each plot. Two of the pitfall traps were used in analyses and the third was used as a contingency in case of loss or damage to the other traps. Trap contents were collected three times over a period of approximately 63 days. Further information on invertebrate biodiversity surveys in the forest road study is provided in (Fuller *et al.*, 2013b).

Birds were sampled by transect count along each of the standard and wide road sections, with each transect at least 500m in length. The number and species of all birds detected within 100m of the observer, along with their estimated position and distance from the observer, were recorded using range-finding binoculars. All bird counts were conducted twice during the breeding season in each year between 0800 hours and 1800 hours, with each count lasting 10 minutes. Birds in flight were excluded from the analyses. Bird surveys were not conducted in heavy or persistent rain, or in winds greater than Beaufort scale four. Further information on bird surveys in the forest road study is provided in Graham *et al.* (2012).

At each forest plot, environmental and management information were collected including location (latitude, longitude), elevation, soil drainage (poor, moderate or good), thinning history and grazing intensity (none, low, moderate or heavy grazing). In addition, soil pH and other soil variables were determined. A number of structural vegetation metrics were recorded at each site including ground vegetation cover, conifer and deciduous tree cover, shrub cover, deadwood, leaf litter, bare soil, fine woody debris and coarse woody debris, canopy cover and tree height, diameter at breast height of all trees and mean distance between tree stems

Data analysis

The effect of road-width treatment on ground vegetation was analysed using NMS ordination analysis and beta diversity using Sørensen pairwise dissimilarity. PERMANOVA was used to investigate factors influencing plant community composition. Invertebrate species richness and abundance were compared between plot position and treatment within each sampling year using paired t-tests for normally distributed data and paired Wilcoxon signed rank tests for non-normally distributed data. The effects of plot position and road-width treatment on spider assemblages within each sampling year were compared with a PERMANOVA. Variation partitioning was used to examine how much of the variation in species assemblages in the road-verges was explained by the subsets of the measured variables: habitat structure, treatment and plot

position. Redundancy analysis was then used to examine the effect of significant subsets on species composition.

Differences in the density of ground-nesting shrub-associated birds and of open habitat associated birds as well as the total density of birds, and the species richness within 100m of the road were calculated. Differences were tested separately between treatments within year, and within treatment between years using Mann-Whitney U-tests. All statistical analyses were conducted on the number of bird detections as this provided comparable density estimates in these habitats, which had low levels of scrub and tree cover. The P-values from the Mann-Whitney U-tests were corrected to counteract the increased risk of Type I error resulting from multiple comparisons

Results

Forest roads in this study were found to support a large number of plant species, the majority of which were vascular plants (57.3 ± 1.3%) and a large proportion were species that would be unlikely to occur in the forest interior. There was no significant change in the number of plant species supported in the road sub-sections from 2005 to 2010, but there was a significant change in community composition. The numbers of shrub and graminoid species increased significantly, while no change was seen in the species richness of forbs or bryophytes. There was no significant change in species richness of any of the groups in the verge plots while total species richness decreased between 2005 and 2010 in the forest plots.

The NMS ordination of the sub-sections (not shown; final stress = 10.46, final instability ≤0.00001, overall r^2 = 89.0) indicated that the floristic composition was similar within each site, both within and between years. Edaphic differences between sites (soil type, drainage and pH), had the greatest influence on plant community composition. Grazing was also important in determining plant community composition along forest roads, with its presence preventing dominance by shrubs. Intermediate levels of shrub cover (3-40%) were associated with the highest species richness. The presence of bare ground was also an important feature, being positively associated with species richness up to approximately 30% cover, above which species richness dropped off sharply. Heterogeneity at the site and plot level was also important in explaining variation in species composition.

No effect of doubling the road width was identified by 6 years post planting; however, abundant natural regeneration of conifers observed at a number of sites will likely negate the effect of the widening unless controlled. In the NMS ordination of the ‘verge’, ‘open’ and ‘forest’ plots (not shown) there was some separation of the plots from 2005 and 2010 with the ‘verge’ plots from all sites forming a relatively distinct group, while the ‘open’ and ‘forest’ plots were intermixed. The PERMANOVA for the plots indicated that seven variables (shrub cover, pH, year, bare ground cover, plot position, drainage and heterogeneity) and two interactions (shrub cover * plot position and bare ground cover * plot position) had significant influences on plant community composition. Road-width was again not found to have a significant influence on composition.

In total 141 species of spider were recorded in this study, 29 of which were forest- and shade- associated species, 35 were open habitat specialists and 77 were habitat generalists. As with the ground vegetation, doubling the width of forest roads had no effect on ground-dwelling spider assemblages by six years post planting ($F_{1,42}$ = 1.39, N.S.), and there was no difference in assemblages between forest-interior and road-

verge transects. Species assemblages did not differ between the plot positions ('Verge', 'Open' and 'Forest') of the road-verge and forest in either the baseline survey ($F_{2,42}$ = 0.41, N.S.) or the repeat survey ($F_{2,42}$ = 0.46, N.S.). There was also no effect of plot position on any of the species metrics measured in the baseline or repeat surveys. However, in 'Forest' plots total species richness was significantly greater in the standard road-width treatment than in the wide treatment (33.88 ± 2.02 and 28.75 ± 3.07 respectively; $t_{1,7}$ = 3.30, P < 0.05). A significant difference in web-building spider species richness was also seen between the standard treatment and the wide treatment (26.88 ± 1.42 and 22.63 ± 2.27 respectively; $t_{1,7}$= 2.92, P < 0.05). In 'Verge' plots the observed species richness of rare species was significantly greater in the wide treatment than in the standard treatment (2.13 ± 0.13 and 1.5 ± 0.19 respectively; $U_{1,7}$ = 0, P < 0.05).

There was a significant difference in ground-dwelling spider assemblages between the sampling years, with a change from an open-associated fauna to a mix of forest- and open-associated species. The significant increase in shrub cover between the years likely accounts for the change in assemblage. Variation partitioning of the measured variables revealed that habitat structure explained 19% of the variation in species composition in the road-verges ($F_{6,57}$ = 3.42, P < 0.01) while road-width and plot position had no influence and produced values of 0%, indicating these explanatory variables performed worse than random variables.

A total of 32 bird species was recorded across all sites, with the most frequently encountered species along forest roads being Willow Warbler, Wren (*Troglodytes troglodytes*), Robin (*Erithacus rubecula*), Meadow Pipit and Chaffinch (*Fringilla coelebs*), which together accounted for almost 70% of all detections. Although the number of species detected within 100m of the road was slightly higher in the wide road-width treatment both in 2005 and 2010 no significant difference was observed (U = 19, N.S. and U = 23, N.S. respectively). No significant effect on bird densities between standard and wide road-width treatments was observed in 2005 or 2010 (U = 12, N.S. and U = 31, P < 0.05).

Although there was no significant effect on bird diversity of doubling forest road width by six years post planting, as seen for spiders, there were considerable changes in the bird communities over time. Open habitat specialists quickly, but briefly, colonized the plantations following tree harvesting. However, only six years after replanting, these species were replaced by ground nesting migrant species, presumably in response to the development of the shrub layer. There were no significant differences between treatments in the numbers of ground nesting birds or open habitat associated birds in 2005. However, the number of ground-nesting shrub-associated birds within 100m of the road increased significantly in both treatments between 2005 and 2010 (standard: U = 0, P < 0.05; wide: (U = 2, P < 0.05). There was a corresponding decrease in the number of open habitat associated birds detected in the wide treatment between 2005 and 2010 (U = 8, P < 0.05).

Discussion

Plantation forest road corridors in this study were found to support a wide range of plant, invertebrate and bird species. A large proportion of those recorded were non-woodland, generalist and open habitat species that would be unlikely to occur in the post-canopy closure forest interior. This is in keeping with the findings of previous studies which have found that forest roads can increase the number of species supported at woodland scale (Sparks *et al.*, 1996; Peterken and Francis, 1999; Roycroft *et al.*, 2008). The conifer plantations in these study sites were composed of relatively young (newly planted and six year old) trees in

the pre-canopy closure stage. The pre-canopy closure stands of Sitka spruce adjacent to both treatments in 2010 comprised habitat that can support species more typical of open, unforested habitats which are lost during the later parts of the forest cycle (Wilson *et al.*, 2006).

The retention of small patches of non-forest habitat within forest plantations may provide a 'life-boat' function for species of conservation concern (Johansson *et al.*, 2013) and two species of endangered spider and eight vulnerable spider species were recorded in the road-verges in young plantation forests in this study. In 2005, in recently clear-felled sites, an average of 18 months after felling and replanting, the forest roads in this study also supported reasonable numbers of Stonechat and Meadow Pipit, both typical open habitat species. Skylark and Wheatear (*Oenanthe oenanthe*), also associated with open habitats, and both of conservation concern in Ireland (Lynas et al., 2007), were also recorded in 2005, although in very low numbers.

This study is among the first to examine the biodiversity of recently clear-felled sites, and it is noteworthy that between 2005 and 2010 successional changes were observed in plant, invertebrate (Fuller *et al.*, 2013b) and bird (Graham *et al.*, 2012) communities. These changes were related in all cases to vegetation structure. Between 2005 and 2010 there was a significant increase in shrub cover in all study plots. This suggests that the disturbance created by felling and reforestation activities has led to successional processes that are leading towards the colonisation and establishment of woodland (Ferris and Carter, 2000; Sweeney *et al.*, 2010b). However, in the verge plots, there was no significant change in the cover of bare ground or in the species richness of any of the plant groups under investigation, and the biodiversity of the verge plots in both years was similar. This suggests that the disturbances caused by road use and management and by grazing animals are resulting in deflection from the expected succession (Ferris and Carter, 2000).

There was no advantage or disadvantage for biodiversity of young plantation forests of increasing the width of forest roads during the first five years of the forest cycle. The experimental widening of the road corridor in second rotation Sitka spruce plantations had little impact on the biodiversity in the first six years after planting. The positive effect of forest roads on biodiversity is mediated primarily through effects of light penetration which is greater at forest roads than it is in the forest interior (Watkins *et al.*, 2003). The trees in this study were approximately 2m tall and so cast little shade and the ground flora was well-developed along the road-verges. The biodiversity present at the study sites was related to the structure of the sites, which will change significantly as the forests mature. Light penetration will be reduced as they mature and the canopy closes (Warren and Fuller, 1993; Avon *et al.*, 2010) and therefore the effects of increased road width will not be fully realised until the crop canopy is tall and dense enough to cast considerable shade on adjacent vegetation. It is anticipated that these benefits will result in the biodiversity supported by the two road width treatments becoming increasingly divergent as the adjacent conifers mature (Peterken, 1996; Sparks *et al.*, 1996). The vegetation and moisture requirements of the open-habitat specialist spider species recorded in this study make it unlikely that they would be found in the interior of plantation forests, particularly after canopy closure where the ground vegetation diversity is typically reduced due to the decreasing availability of light, nutrients and moisture (Fuller *et al.*, 2013b). It is therefore likely that the impact of increased road width will only start to be realised towards the end of the pre-thicket stage, in another five years. It is expected that the major benefit of experimental widening of forest road corridors will be realised from the time of canopy closure onwards. That the widening of forest roads is likely to augment the biodiversity of these forest plantations by providing suitable habitat for scrub- and forest-dependent species.

Natural regeneration of Sitka spruce trees was observed along the road-verges during the repeat survey at several of the forests in this study. Therefore management of forest roads is required to prevent regeneration of these and other non-native trees in areas along road-verges. If a wider road-width is found to be beneficial to biodiversity at later stages of the forest cycle it will be important to actively manage the road-verges and remove any regeneration of the planted tree species and of other non-native tree species that may cause heavy shading.

Recommendation 3: Manage natural regeneration along forest road verges to maintain the integrity of open space through the forest cycle.

Previous studies have shown that forest biodiversity evolves over the course of the forest cycle in both first and second rotation plantation forests (Oxbrough *et al.*, 2005; Sweeney *et al.*, 2010d), Therefore it will be necessary to continue the monitoring of this long term experiment until commercial maturity in order to determine how biodiversity is affected by changes in canopy cover and habitat succession in plantation forest road-verges. This information will allow forest managers to effectively tailor management strategies to maximise opportunities for biodiversity conservation.

Conclusions

The study of the biodiversity of forest roads clearly demonstrated the importance of forest road-verges for open specialists and habitat generalist species. This research has found that, in the absence of shading by the tree crop, forest road corridors mainly support plant species and communities of open habitats. The vulnerable and endangered species found in the open habitat of these young plantation forests indicate that open areas within plantation forests support rare species. While they are unlikely to act as substitutes for semi-natural open habitats (Eycott *et al.*, 2006), they may still play an important role in increasing diversity at the plantation scale (Sparks *et al.*, 1996; Peterken, 1999; Ferris and Carter, 2000), and possibly also the landscape scale (Peterken, 1996; Peterken and Francis, 1999). During later, closed canopy, stages in plantation forests they may also provide important refuges for woodland species that would not otherwise persist in the forest interior. However, the shading influence of the tree crop later in the cycle in narrow road corridors, and of naturally regenerated conifers in wider road corridors may negate these benefits. Longer-term monitoring of plant diversity within the standard and wide road sections in sites with and without conifer regeneration will give further insights into this. The effect of forest roads on forest biodiversity is an important conservation and management issue. Forest road-verges provide important open habitat for in plantation forests. Their importance extends to species of conservation importance, where they make a valuable contribution to the conservation of biodiversity, providing further support for their inclusion in forest management plans. These findings support the retention of road-verges in plantation forests, and demonstrate the importance of this open habitat for rare spider species. Forest management should include consideration of the importance of these areas for forest biodiversity.

Chapter 3

Long-term biodiversity monitoring

Sandra Irwin, Anke Dietzsch, Linda Coote, Conor T. Graham, Lauren Fuller, Mark W. Wilson, Daniel L. Kelly, Fraser J. G. Mitchell, Thomas C. Kelly and John O'Halloran
Work Package 1C

Long-term monitoring of forest biodiversity is essential to monitor changes in biodiversity over time and to assess the effectiveness of biodiversity conservation measures. This study set out to establish forest sites in Ireland which can be used for long-term biodiversity monitoring. Four Sitka spruce and four lodgepole pine sites were selected across the island of Ireland. These are the two most commonly planted tree species in Irish forests at present and surveys of plant, invertebrate and bird biodiversity were undertaken to provide baseline data against which future monitoring can take place. Long-term monitoring is essential to provide insights into environmental change and biodiversity conservation at time-frames appropriate to forest ecological processes which short-term approaches cannot provide. Studies of this kind support the integration of biodiversity maintenance with timber production, which is a goal of national and global afforestation policies.

Background

Because trees are relatively long-lived, processes in forests have long-term cycles long-term studies play a central role in ecology and in resolving environmental issues (Franklin, 1989). In addition to its role in monitoring the effectiveness of management practices for biodiversity conservation, long-term monitoring of baseline forest biodiversity is also essential to guide management standards and practices and for environmental policies (Hartmann *et al.*, 2010; Magurran *et al.*, 2010). Therefore the benefits of establishing experimental forest plots can only be fully reaped when the plots are monitored over long periods. In order to fully understand forest biodiversity, long-term studies that include repeat visits to sites and systematic data collections, over periods preferably longer than a forest rotation, are required. Inadequacies in the quality and quantity of data on the biodiversity of local forests have been identified in many areas, compromising the ability of management approaches to enhance the reforestation process (Gardner, 2010; Lorenz, 2013). Thus there is a requirement for basic research to inform monitoring programmes and aid decision makers (Magurran *et al.*, 2010).

Lindenmayer *et al.* (2012) define the five key values of long-term ecological studies as follows:

1. Quantifying ecological responses to drivers of ecosystem change.
2. Understanding complex ecosystem processes that occur over prolonged periods.
3. Providing core ecological data that may be used to develop theoretical ecological models and to validate simulation models.
4. Acting as platforms for collaborative studies, thus promoting multidisciplinary research.
5. Providing data and understanding at scales relevant to management, and hence critically supporting evidence-based policy, decision making and the management of ecosystems.

Against this background the EU LIFE project "Further Development and Implementation of an EU-level Forest Monitoring System (FutMon)" was established by 22 EU-Member States, and now operating in 41 countries, with a view to standardising forest ecosystem monitoring (Harrington *et al.*, 2010). The aim of this project is to gather long-term information suitable for modelling relationships between forest health, forest growth, carbon fluxes, climate change, air pollution, and biodiversity (Lorenz, 2013). This work commenced in 1991 and related work began in Ireland in 1988 (Cummins *et al.*, 2011). In recognition of the value of long-term baseline biodiversity monitoring the survey of biodiversity at a number of plots at FutMon sites suitable for use in long-term monitoring was included in this study. These sites were plantations of Sitka spruce and Lodgepole pine, the two most common tree species planted in Ireland at present, making up 52.3% and 10.1% of the planted forest area respectively (Forest Service, 2007).

Objectives

The specific objective of this study was to:

- Establish long-term biodiversity monitoring plots in planted forests in Ireland.

Methodology

Study sites

Eight mature plantation forest sites were selected including four Sitka spruce-dominated and four Lodgepole pine-dominated (Figure 12). All sites are also part of the pan-European FutMon Level 1 Long-term Monitoring Plot Scheme which includes sites throughout Europe (Lorenz, 2013).

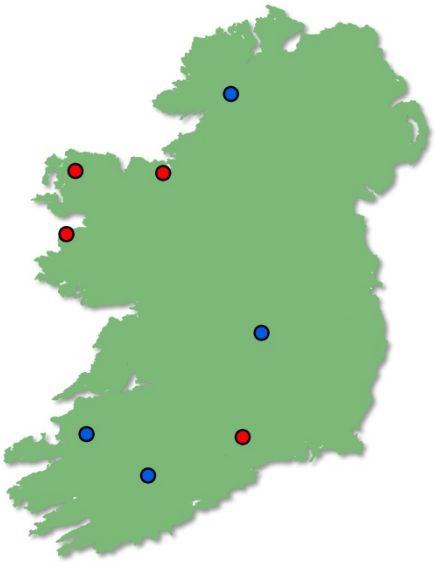


Figure 12. Study site locations: ● Sitka spruce plantations (n = 4); ● Lodgepole pine sites (n = 4).

Data collection

Surveys of plant, invertebrate and bird diversity and vegetation structure were carried out between May and September 2010 and 2011. Ground vegetation biodiversity surveys were undertaken in three 10m x 10m plots at each site that were chosen to represent the dominant habitat at the site. At all plots, height

and percentage cover of each vegetation layer were recorded together with canopy openness, amount and quality of deadwood, non-floristic cover variables, soil variables and floristic variables. All vascular plant species present in each plot were recorded and bryophytes and lichens were recorded where they formed patches more than 10cm x 10cm. Species not forming patches of this size but which were frequently occurring were also recorded.

In order to sample ground-dwelling spider diversity three plots were established at each site, each of which was ≥50m from the forest edge with a distance of at least 50m between each of the three plots. Each plot had five pitfall traps placed in a line at 2m intervals. Active ground-dwelling spiders were sampled using pitfall traps (plastic cups 7cm in diameter and 9cm tall), filled to a depth of 3cm with ethylene glycol. Each trap was placed in a pre-excavated hole ensuring that the rim was just below ground level. Trap contents were collected three times over a period of approximately 63 days in all sites in summer 2010 and summer 2011. Hoverflies were sampled using four Malaise traps per site, which were placed ≥100m apart in linear un-shaded areas. The contents of the traps were collected three times at three week intervals between May and July in summer 2010 and summer 2011.

Birds were counted at five locations within each site using point counts. One count was carried out at the central position of the FutMon plot and the remaining four were located at each of the cardinal directions (N, E, S, W). All bird counts were conducted twice during the breeding season in each year between 0800 hours and 1800 hours, with each count lasting 10 minutes. Birds in flight were excluded from the analyses. Bird surveys were not conducted in heavy or persistent rain, or in winds greater than Beaufort scale four. Bird densities for each point were then estimated using the computer programme *Density.*

At each forest plot, environmental and management information were collected including location (latitude, longitude), elevation, soil drainage (poor, moderate or good), thinning history and grazing intensity (none, low, moderate or heavy grazing). In addition, soil pH and other soil variables were determined. A number of structural vegetation metrics were recorded at each site including ground vegetation cover, conifer and deciduous tree cover, shrub cover, deadwood, leaf litter, bare soil, fine woody debris and coarse woody debris, canopy cover and tree height, diameter at breast height of all trees and mean distance between tree stems.

Results

Biodiversity surveys at the four Sitka spruce-dominated long-term monitoring study sites found a total of 199 species of all taxa combined and at the four lodgepole pine-dominated study sites found 186 species. Lists of species are available in the BIOPLAN project GIS database and a summary of the breakdown of species across the individual taxa in each forest type surveyed are shown in Figure 13. The flora under lodgepole pine was somewhat richer than under Sitka spruce, especially for bryophytes and, by contrast, the invertebrate and bird fauna was considerably poorer under lodgepole pine.

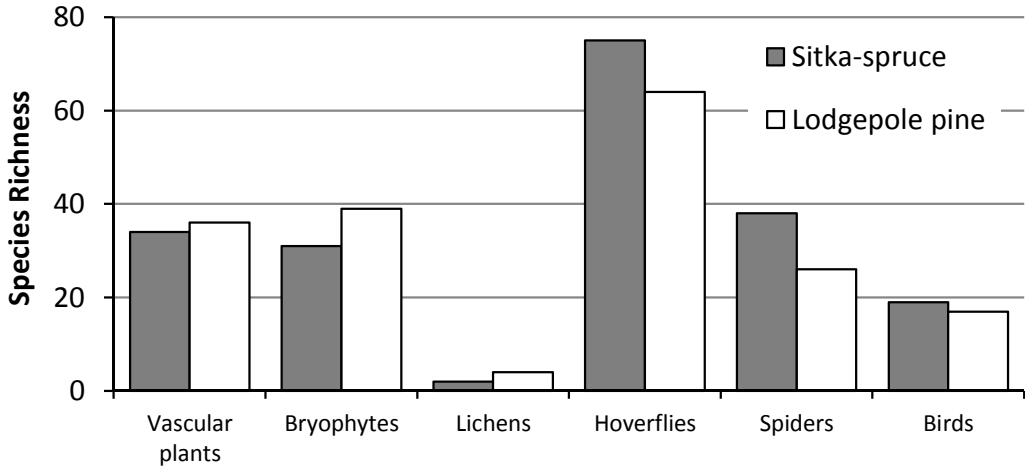


Figure 13. Total species richness of the taxa surveyed in the two long-term monitoring forest site types: ■ Sitka spruce dominated plantations, and □ Lodgepole pine dominated plantations.

Discussion

Biodiversity surveys at the four Sitka spruce-dominated long-term monitoring study sites found a total of 199 species and at the four lodgepole pine-dominated study sites found 186 species. Long-term biodiversity monitoring is essential to determine what has been achieved from particular management options and to monitor effectiveness against biodiversity targets (Magurran *et al.*, 2010; Collen *et al.*, 2013). This in turn is required to inform practice for ecologically responsible forest management, and biodiversity monitoring makes a meaningful contribution towards improving management policy and practice (Gardner, 2010; Oxbrough *et al.*, 2014). Direct comparisons of outcomes of alternative management are useful, but the effectiveness of conservation efforts can only be measured against targets or benchmarks (Huggard *et al.*, 2009). Forest biodiversity monitoring for this purpose is necessarily a long-term activity and the data collected here on the diversity of plants, invertebrates and birds at the four Sitka spruce and four lodgepole pine sites selected as long-term monitoring sites represents the first, in what will hopefully be, a long-term study of forest biodiversity over time in Ireland. These data were also used to investigate spider diversity in three types of conifer plantation in chapter 4 and for testing biodiversity indicators in chapter 6.

Recommendation 4: Continue monitoring of BIOPLAN study sites (afforestation, forest roads and long-term sites) at five year intervals to determine biodiversity impacts across the forest cycle and to provide benchmark data against which the performance of biodiversity conservation measures can be evaluated.

Chapter 4

Planted tree species and spider diversity

Lauren Fuller, Thomas C. Kelly, John O'Halloran and Sandra Irwin
Work Package 1D

The selection and management of canopy tree species in production forests is intrinsically linked to their biodiversity value as it influences light penetration, micro-climate, and vertical structural layers. Considering the requirement for forest plantations to deliver both timber yield and environmental services, such as biodiversity conservation, and the prevalence of introduced tree species used in plantation forests in Ireland and throughout much of Europe, the potential for native tree species in this regard is of interest. Spiders are abundant forest-floor arthropods and are useful for assessing the influence of canopy tree species on forest biodiversity. The diversity of active ground-dwelling spiders was examined in Scots pine plantation forests compared with two commonly used non-native conifer specie: Sitka spruce and lodgepole pine. Results indicated that Scots pine forests supported the highest species richness and number of associated species of ground-dwelling spiders, while Sitka spruce supported an intermediate level of species richness and fewer associated species, and lodgepole pine supported the lowest species richness and no significantly associated species. These differences were mediated by higher canopy openness, vegetation cover and structure in Scots pine plantation forests, and potentially the historical presence of this tree species in the Irish landscape. These results indicate that planting Scots pine offers the potential to enhance biodiversity in plantation forests.

Background

Forest biodiversity is influenced by a number of factors, including the selection and management of the canopy tree species, which can alter understory structure and species composition through changes in microclimate, soil chemistry, litter and vegetation (Palik and Engstrom, 1999; Horgan *et al.*, 2003). Many plantation forests throughout Europe are dominated by non-native tree species and Ireland has one of the highest percentages of introduced species in Europe with almost 70% of the forests consisting of exotic conifer species (Forest Europe UNECE and FAO, 2011). As already noted, the most commonly planted tree species in Ireland are Sitka spruce and lodgepole pine, comprising 52.5 % and 9.7 % of the total forest area of Ireland respectively (Forest Europe UNECE and FAO, 2011; Forest Service, 2013). These non-native species, introduced from North America in the 19th century, are well-suited to the soils and climate of Ireland, and are favoured by forest managers for their fast growth (Carey and Hendrick, 1986; Farrelly *et al.*, 2009). Another, less commonly planted conifer species used in production forestry in Ireland is Scots pine, which comprises just 1.2 % of the total forest area. This species was formerly a major component of Irish native woodlands, but dwindled and virtually disappeared from Ireland in the course of the first millennium A.D. (Roche *et al.*, 2009). Scots pine is considered to be 'semi-native' by the Irish forestry sector and is regarded as the only naturally occurring conifer with forestry potential (Forest Service, 2000a). Considering the global expansion of plantation forestry (FAO, 2012) and the high percentage of introduced species used,

research that directly compares plantations of non-native species with those of native origin is required to address knowledge gaps in sustainable forest management. In particular, investigating the capacity for non-native species to support biodiversity, to inform policy for biodiversity conservation (Peterken, 2001; Carnus *et al.*, 2006).



Scots pine plantation forest (Photo by Jim Campion)

Objectives

- The aim of this study was to use data collected as part of Work Packages 1 and 2 to investigate differences in spider diversity between non-native and semi-native conifer plantations.

Methodology

Study sites

Four replicate stands each of Scots pine, Sitka spruce and lodgepole pine plantation forests were selected for study across Ireland (Figure 14). These 12 stands were selected to represent even-aged, commercially mature monocultures for each forest type. The Sitka spruce stands and one lodgepole stand were sampled in the summer of 2010 and the remaining three lodgepole pine and all Scots pines stands were sampled in the summer of 2011.

Data collection

Active ground-dwelling spiders were sampled using pitfall traps which were dug into the ground and positioned so the rim of the cup was slightly below the ground surface. Three sampling plots were located in each stand, a minimum of 50m apart and a minimum of 50m from the forest edge and selected to represent the stand as a whole, in homogenous areas. In each plot five traps were set 2m apart in a linear arrangement. The traps were plastic cups, 7cm in diameter and 9cm in depth, filled with 3cm of anti-freeze. The contents of each trap were collected every three weeks from May to July.

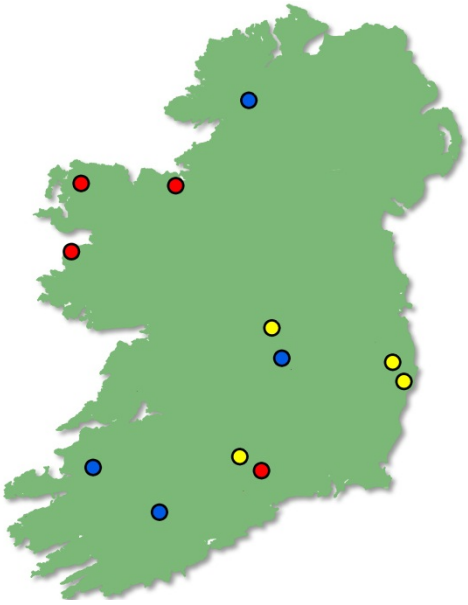


Figure 14. Study site locations: ● Scots pine plantations (n = 4); ● Lodgepole pine plantations (n = 4); ● Sitka spruce plantations (n = 4).

Habitat surveys were carried out at each plot using a 1m x 1m quadrat placed over each of the five pitfall traps. The percentage covers of the following environmental variables were recorded: leaf and needle litter, fine woody debris (<10cm diameter), coarse woody debris (>10cm diameter), ground vegetation (0 – 10cm), lower field-layer vegetation (10 – 50cm) and upper field-layer vegetation (50 – 100cm). The mean percentage cover for each variable was calculated from the five quadrats at each plot. Litter depth was measured and one soil sample was taken from each corner and the centre of a 10m x 10m plot placed adjacent to each pitfall trap plot at a distance of 10m to reduce disturbance to the pitfall traps. The soils were pooled for each plot and soil pH and organic carbon content were measured. Organic carbon content was measured as per cent loss on ignition at 550˚C for 5 hours. The percentage of canopy cover was also calculated using Gap Light Analyser 2.0 from a hemispherical photograph taken at the centre of each 10m x 10m plot at a height of 1.3m.

Data analysis

Data from the four Sitka spruce and four lodgepole pine long-term monitoring study sites were used together with data from Scots pine study sites surveyed as part of chapter 6 to investigate the spider diversity in these plantation types. The effect of planted conifer species on the relative abundance and species richness of ground-dwelling spiders was examined using generalised linear mixed modelling (GLMM) and, where appropriate, this was followed by post-hoc tests adjusted for multiple comparisons. The environmental variables were also compared between forest types using GLMM. The effect of forest type on species composition was tested using a PERMANOVA on Hellinger transformed species abundance data. Following a significant effect of forest type, PERMANOVA was used to test for significant differences between pairs of forest types. P-values were adjusted using the Bonferroni correction for multiple comparisons.

Results

Generally, Scots pine plantation forests supported the highest species richness and lodgepole pine plantation forests supported the lowest species richness of ground dwelling spiders. Overall, Sitka spruce had fewer significant differences with the other two plantation forest types and supported an intermediate level of species richness. In particular, Scots pine and Sitka spruce plantation forests supported significantly higher total species richness compared with lodgepole pine plantations ($F_{2,30} = 5.89$, $P < 0.05$, Figure 15). The species richness of forest habitat specialist spider species was significantly higher in Scots pine plantations compared with lodgepole pine plantations ($F_{2,30} = 4.24$, $P < 0.05$, Figure 15). PERMANOVA revealed that forest type had a significant effect on spider species composition ($F_{2,32} = 3.95$, $P < 0.001$) and that the species composition of each forest type was significantly different: Scots pine vs. lodgepole pine ($F_{1,19} = 4.14$, $P < 0.001$); Scots pine vs. Sitka spruce ($F_{1,20} = 4.46$, $P < 0.001$); lodgepole pine vs. Sitka spruce ($F_{1,21} = 3.30$, $P < 0.001$).

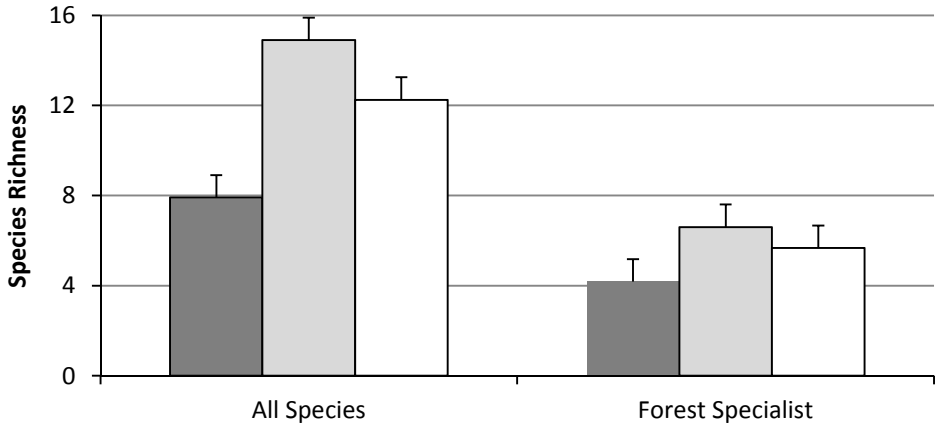


Figure 15. Mean (± standard error) of ground-dwelling spider relative abundance and species richness in three types of conifer plantation: ■ Lodgepole pine, □ Scots pine and □ Sitka spruce.

Discussion

All of the plantation forest types studied differed from each other in terms of species composition; however the Scots pine plantations supported the most distinct spider species assemblage. Overall they also had greater species richness than lodgepole pine plantations, but not Sitka spruce plantations. The canopy was more open and there was greater vegetation cover in Scots pine plantation forests than in the other types. These findings support previous studies and indicate that, due to greater light penetration through the canopy and thus having greater vegetation cover and structure, Scots pine plantation forests support a greater number of spider species and of forest habitat specialist species. Scots pine plantations had a greater cover of upper layer vegetation compared with the other conifer types, which was probably as a result of increased light availability from the more open canopy, which benefits plant diversity in forests (Thomas *et al.*, 1999; Ferris *et al.*, 2000). Shade tolerant and forest habitat specialist spider species are adapted to the shade created by canopy cover but open habitat specialists and many cursorial species are negatively affected by this, thus overall spider diversity may decrease with forest canopy closure (Oxbrough *et al.*, 2006a). Therefore, the more open canopy and presence of upper field layer vegetation in Scots pine forests is likely to benefit spiders and in particular specialist species, which prefer open habitats or forest edges and would normally be excluded from the interior of non-native conifer plantation forests.

Recommendation 5: Increase the diversity of tree species in plantations, particularly to include planting of native or lightly shading species, such as Scots pine, to enhance biodiversity.

These results suggest that appropriate management of plantation forests to mimic the habitat of Scots pine forests by increasing vegetation cover and structural diversity could benefit ground-dwelling spider diversity and composition in low diversity forest types, such as lodgepole pine. This can be achieved by increasing canopy openness and promoting the growth of ground vegetation through the creation of canopy gaps, frequent thinning, and retaining over-mature trees, or by allowing them to succeed naturally, by increasing the rotation length (Peterken *et al.*, 1992; Quine and Humphrey, 2010). The longer rotation required before Scots pine reaches commercial maturity plus the need for more specialised soil and climate conditions make Scots pine a less favourable forestry species in Ireland. While trade-offs between timber yield and biodiversity are inevitable, the findings of the present study in the context of current interest in using a mix of tree species in plantation forests suggest that Scots pine might also be beneficial as a secondary species in a mix (Lust *et al.*, 1998) due to the advantages it confers on forest biodiversity. However, this requires further research to determine the proportions and planting patterns which would provide the optimum benefits to biodiversity, with due regard for productivity and timber yield.

Recommendation 6: Undertake research to identify forest plantation types that offer the optimal combination of economic and ecological benefits.

Conclusions

The selection of tree species for planting has implications not only for timber yield, but also for forest biodiversity. This study of the effects of planted tree species on ground-dwelling spiders demonstrated that Scots pine forests provide a greater number of microhabitat niches supporting a variety of species which fulfil different ecological functions, whereas the two most commonly used North American conifer tree species performed the worst for ground-dwelling spider diversity. These results suggest that using native or semi-native tree species or species of European provenance in plantation forests will increase ground-dwelling spider diversity. Scots pine requires a longer rotation length before reaching commercial maturity and more specialised soil and climate conditions making it a less favourable forestry species in Ireland (Forest Service, 2012). However, the habitat changes that occur as a result of a longer rotation length are probably also the reason for Scots pine's higher diversity value (Lust *et al.*, 1998). Increasing canopy openness and promoting the growth of ground vegetation by either managing forests to mimic these changes or allowing them to naturally succeed will improve their diversity value for spiders. These results have implications for forest policy and management in Ireland and suggest that longer rotation lengths will benefit ground-dwelling spider diversity in the conifer types used for forestry (Peterken *et al.*, 1992; Quine and Humphrey, 2010). The current interest in using a mix of cover tree species in plantation forests, due to perceived benefits to biodiversity, implies that Scots pine might be beneficial as a secondary species in a mix (Lust *et al.*, 1998). However, further research is required to determine the proportions and planting patterns which would provide the optimum benefits to biodiversity.

Chapter 5

Fungus gnats in clear-fell and thinning debris

Rob Deady, Sandra Irwin, Thomas C. Kelly and John O'Halloran
Work Package 1E

The deadwood habitat of forests and its saproxylic insect inhabitants are becoming increasingly well understood and incorporated in forest management plans with environmental objectives. Most studies of this habitat, however, tend to focus on coarse woody debris (CWD, ≥10cm diameter) where its conservation value is thought to be higher than for fine woody debris (FWD, ≤10cm diameter). 'Brash' or 'Slash' is a common fine woody debris component of plantation forests and is composed of limbs, branches and twigs. Up until now, systematic saproxylic fly studies have been scarce with most work having been carried out on saproxylic beetles. This study provides the first description of the fungus gnat fauna (Diptera, Sciaroidea) utilizing Sitka spruce brash in commercial plantations in Ireland. Sixteen emergence traps were used to collect emerging adult fungus gnats from thinning brash lines and clear-fell brash in commercial forest plantations. In total, 1794 specimens comprising 80 species of fungus gnat were recorded. The findings of this study suggest that fungus gnats use brash in exposed areas where clear-fell operations have ceased and that these brash communities differ to those of post-thinning brash environments. It is likely that gnat larvae use this unique anthropogenic habitat to consume fungi coursing through the debris but also that which resides in the soil horizon while using the soil as a pupation medium also. It is therefore recommended that brash should be left in situ after both clear-felling and thinning operations to allow colonisation by fungi and to enhance biodiversity in planted forests.

Background

Thinning and pruning are management operations that are commonly used in Irish plantation forests to remove inferior trees, to minimise competition between trees and to promote growth in the trees selected for retention until final harvest. The vast majority of commercial plantations in Ireland are harvested by clear-felling, where all trees are removed at the end of the forest cycle. The deadwood logging residues, or brash, produced by thinning and clear-felling operations are typically left *in situ* as they are unsuitable for conventional timber processing. In Irish plantation forests brash serves to protect forest soil from erosion and compaction caused by heavy machinery during harvesting (Booth *et al.*, 2007) and it is estimated that between 50 and 100 oven dry tonnes of residue per hectare remain following harvesting (Mitchell and Hankin, 1993). Due to the escalating costs associated with fossil fuels, and the negative impact that they have on climate change, brash is attracting attention as a potential bio-fuel feedstock (Gunnarsson *et al.*, 2004). Although this is not practice in Ireland at present (Whelan, 2010), the extraction of residual brash for use as bio-fuel has increased in recent decades, particularly in Scandinavian countries (Hoyne and Thomas, 2001). The resultant lack of deadwood in Irish forests presents a potential conflict with biodiversity conservation, as deadwood is a key component for many aspects of forest biodiversity (Ehnstrom, 2001; Jonsson, 2005; Sweeney *et al.*, 2010d).

In terms of invertebrate biodiversity, particularly at risk from these practices are the saproxylic species i.e. those species reliant on deadwood or moribund trees during some phase of their lifecycle (Speight, 1989). Saproxylic invertebrates are an important functional group in forest ecosystems and their conservation relies on scientific information on how well remaining habitat patches in forests can support these species, and what kinds of woody debris should be retained in forests to ensure the persistence of these species (Siitonen, 2001; Grove, 2002). If the needs of these species are not considered in forest management plans their persistence, and the persistence of associated ecosystem services, is threatened (Grove, 2002). Little is known about saproxylic Diptera (true-flies) in Irish forests, particularly fungus gnats. Therefore a detailed study was undertaken on the fungus gnat fauna in a sub-set of study sites.



Clear-fell plantation forestry site.

Objectives

The main aims of the study were to describe the fungus gnat fauna associated with fine woody debris brash in Irish plantations and to examine the differences in fauna between clear-fell debris and thinning debris. The rationale for the second aim was to investigate whether or not an alteration to the forest environment such as a clear-fell operation could lead to fungus gnat absence/diminishment with risk of desiccation, change in deadwood profile and a loss in fungal hosts likely being the main drivers.

Methodology

Study sites

The study of saproxylic invertebrates in brash lines was conducted at four forest study sites (Figure 16) from the 28th June 2010 to the 9th September 2010 (73 days). Two of these were pre-thicket Sitka spruce plantation sites that were part of the study of forest road biodiversity, where brash lines consisted of thinning debris from previous thinning operations. The other two study sites were mature Sitka spruce plantation sites that were part of the long-term biodiversity monitoring study where brash lines consisted of thinning debris from previous thinning operations as well as brash that had been raked into lines following clear-felling of the first rotation.

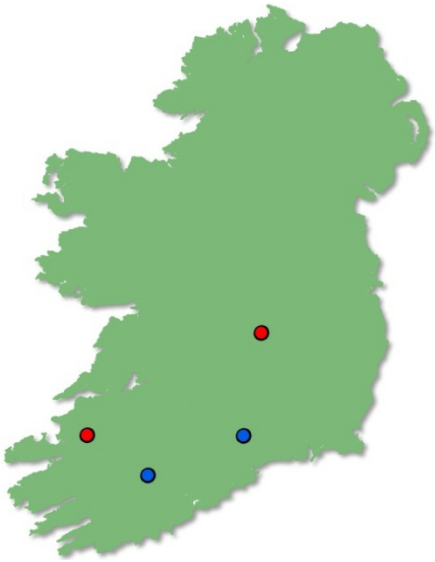


Figure 16. Study site locations: ● Thinning study sites (n = 2); ● Clear-fell study sites (n = 2).

Data collection

Emergence trapping was used to sample flying invertebrates (predominantly adult Diptera) emerging from the substrate. A detailed description of trap design is provided in Deady (2013). Two standard emergence traps were established on each of two brash-lines from six typical and representative brash-lines in each site spaced approximately 8m apart from each other on thinning brash-lines and clear-fell brash-piles (Figure 17). These traps collected emerging adult fungus gnats (Mycetophilidae, Keroplatidae, Bolitophilidae and Diadocidiidae (Diptera: Sciaroidea)) from thinning brash-lines and clear-fell brash piles. Environmental and habitat data were collected to investigate the factors determining abundance, species richness, dominance and diversity of fungus gnats. Brash height and width, canopy cover, vegetation cover, bark cover, leaf litter, bare soil and Leaf Area Index and radiation transmission were measured at each study site.



Figure 17. Emergence traps on thinning brash

Data analysis

Differences in fungus gnat abundance and species richness between thinning and clear-fell debris and the relevance of physical characteristics of brash were investigated using negative-binomial generalised linear modelling (NBGLM). NMS was used to examine the relationships between the variables measured and to assess dissimilarities at the species level between traps, treatments and sites.

Results

Data on the fungus gnat communities at two of the forest roads study sites and two of the long-term monitoring study sites were used in this analysis. Eighty species of fungus gnat, *Sensu lato*, were recorded in this study (n = 1794), 75% of these species were saproxylic species which rely on deadwood. These included two species of fungus gnat that had not previously been recorded in Ireland (*Trichonta vulcani* and *Exechiopsis fimbriata*), and two other species whose status in Ireland was previously uncertain (*Mycetophila abiecta* and *Phthinia humilis*). Significantly fewer fungus gnats were found in thinning debris than in clear-fell debris (690 ± 0.649, 948 ± 7.48, respectively; $Z_{1,46} = -4.15$, $P < 0.001$). The abundance of fungus gnats also differed significantly between debris types, both at species ($H = 8.8$, d.f. = 3, $P < 0.05$) and genus ($H = 8.6$, d.f. = 3, $P < 0.05$) levels. Brash height was a significant positive driver of fungus gnat abundance at both species ($Z_{1,46} = 3.3$, $P < 0.001$) and genus ($Z_{1,46} = 3.198$, $P < 0.01$) levels. The species richness of fungus gnats was not significantly influenced by brash height, brash width, lower field layer cover, upper field layer cover, mean minimum temperature or debris treatment at either species or genus level. The NMS on fungus gnat assemblage data generated 3 axes which accounted for 72% of cumulative variance in assemblages across all traps and over the course of the study (Figure 18).

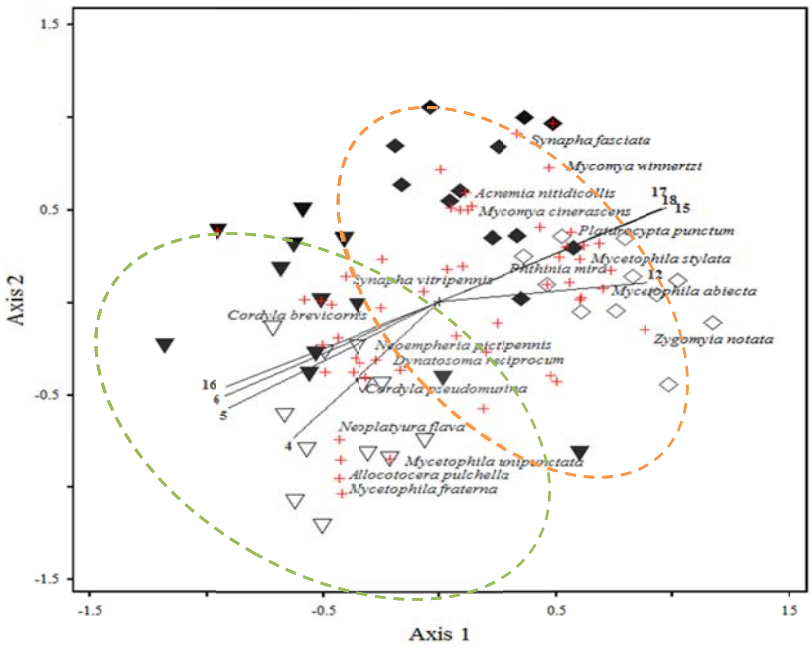


Figure 18. NMS ordination of traps with fungus gnat species abundance data (+) with singletons removed. This was overlaid with joint vectors of the environmental variables and labels of site specific indicator species. Axis 1, $r^2 = 0.347$; Axis 2, $r^2 = 0.19$. Final stress for a 3D solution = 17.452, final instability = 0.03558. Diamonds = clear-fell debris; triangles = thinning debris. Environmental variables (cut off r^2 value = 0.6): (4) Ground layer; (5) Litter; (6) Bark cover; (12) Mean rainfall; (15) % Canopy Openness (16) LAI (Leaf Area Index) (17) Direct solar radiation transmitted (18) Diffuse solar radiation transmitted.

Using an r^2 cut-off of 0.6 to ensure high correlations, visual examination of the NMS ordination revealed a distinct separation of plots along Axis 1 correlating differing exposure regimes in terms of sunlight that plots received. There was a marked difference between clear-fell debris fungus gnat species assemblages and thinning debris fungus gnat species assemblages along Axis 1 ($H = 29.68$, d.f. = 1, $P < 0.001$). A significant difference was also found between clear-fell debris assemblages and thinning debris assemblages along Axis 2 ($H = 17.93$, d.f. = 1, $P < 0.001$).

Discussion

This study provides the first description of the fungus gnat communities utilising Sitka spruce clear-fell and thinning deadwood debris in commercial forest plantations in Ireland. The results show that fine woody Sitka spruce debris in the form of brash is an important habitat for supporting Mycetophilidae and other closely related Diptera. This is particularly the case for saproxylic species and, despite the fact that Sitka spruce is a non-native tree species (Green *et al.*, 2007), at least 75% of the fungus gnat species found in this study were saproxylic. Many of the other species collected may utilize brash as a source of cover during their pupation phase and/or feed on subterranean mycelia. It is therefore recommended that brash should be left *in situ* after clear-felling and thinning operations, to allow colonisation by fungi and, consequently, saproxylic invertebrates. The gnat communities inhabiting the two types of debris were found to be distinct from one another, with suspected specialist species utilizing the more exposed brash habitats at clear-fell sites, probably in response to changes in deadwood fungal communities. The habitat and environmental variables measured did not explain differences in diversity or species richness of gnats, though there was a positive relationship between brash height and overall gnat abundance. The patchy distribution and infrequent occurrence of many species suggests fungus gnats occupy unique micro-niches within the fractal make-up of brash.

The brash habitat is a complex one that plays host to organisms from many different trophic levels, and with the shortfall of coarse woody debris in Irish forests (Sweeney *et al.*, 2010d), brash and fine woody debris are important in countering the overall deadwood deficit (Nordén *et al.*, 2004). The importance of conservation of coarse woody debris in plantation forests for invertebrate biodiversity is well recognised, but the findings of the current study show just how important it is to also promote fine woody debris in managed forests.

Recommendation 7: Preserve fine woody debris in plantation sites after both thinning and clear-fell operations to enhance and maintain biodiversity of organisms that are dependent on it.

Conclusions

This study demonstrated that fine woody Sitka spruce debris in the form of brash is an important habitat for fungus gnats and other closely related invertebrates and concluded that it should not be extracted. The fungus gnat communities inhabiting the thinning and clear-fell debris were found to be distinct from one another, with suspected specialist species utilizing the more exposed brash habitats at clear-fell sites. It is therefore recommended that brash should be left in situ after clear-felling and thinning operations, to allow colonisation by fungi and, consequently, saproxylic invertebrates. There was a positive relationship between brash height and overall gnat abundance suggesting that brash stacking should continue at the currently recommended 2m height. With the low levels of deadwood in Irish forests (Sweeney *et al.*, 2010d), brash and fine woody debris are important in countering the overall deadwood deficit. The importance of conservation of coarse woody debris in plantation forests for invertebrate biodiversity is well recognised, but the findings of the current study show just how important it is to also protect finer debris too.

Chapter 6

Biodiversity indicators

Linda Coote, Nadia Barsoum, Lauren Fuller, Mark Wilson, Anke Dietzsch, Sandra Irwin, Daniel L. Kelly, Fraser J. G. Mitchell, Thomas C. Kelly and John O'Halloran
Work Package 2A

Plantation forests comprise a considerable proportion of the total forest area in Ireland and the UK and the identification of management practices to enhance biodiversity is essential if the goals of sustainable forest management are to be achieved. Since complete biodiversity assessments are rarely possible, efforts have been increasingly focussed on the use of indicators. Stand-scale structural, functional and compositional biodiversity indicators for vascular plant, bryophyte, ground-dwelling spider and bird diversity have previously been derived from Sitka spruce and ash plantation forests in Ireland, but remained to be tested on independent data. In this study, these provisional biodiversity indicators were tested on data from 15 Scots pine monocultures, 14 oak monocultures and 14 intimately mixed Scots pine/oak stands from three regions (Ireland, the New Forest in southern England and Thetford Forest in eastern England), as well as in four Sitka spruce and four lodgepole pine plantations from Ireland, to determine their applicability across a broader range of forest types and geographical areas. Confirmed biodiversity indicators included conifer canopy cover, high shrub cover for bird species richness, coarse woody debris for forest-associated bryophytes, litter cover for vascular plants and forest-associated spiders, proximity to old woodland for forest-associated vascular plants and stand age for forest-associated vascular plants and forest-associated spiders. These biodiversity indicators can be assessed without the need for specialist knowledge, are ecologically meaningful and applicable to a range of forests managed under a clear-felling system. The indicators can be used to assess the potential value of stands for the taxonomic groups to which they apply, as well as giving insights into management practices that may enhance diversity in these groups. However, it is important to note that these indicators may not be applicable across a broader geographical area, where climate variables, e.g. precipitation levels, and land-use histories can differ. Therefore consideration of individual stand characteristics must be included in forest management plans for biodiversity conservation.

Background

Despite the popularity of the indicator concept in forest biodiversity monitoring, few indicators have been adequately tested or validated (Noss, 1999; Niemi and McDonald, 2004). Three types of biodiversity indicators were identified by Noss (1990): structural indicators (the physical organisation of a habitat, e.g. horizontal structural layers); functional indicators (ecological or evolutionary processes, e.g. disturbance, nutrient cycling and gene flow); and compositional indicators (species or species groups which can predict the response of other species). Biodiversity indicators from the categories proposed by Noss (1990) that are correlated with species richness are particularly useful since species richness is the most basic and universally accepted method of quantifying biodiversity (Magurran, 2004). Following an extensive study as part of the BIOFOREST project (Iremonger *et al.*, 2006), a set of indicators of biodiversity were developed for

plantation forests in Ireland, which were also expected to be applicable over a wider area with similar climates (Oxbrough *et al.*, 2005; Smith *et al.*, 2008). These compositional, structural and functional indicators of biodiversity covered five taxonomic groups—bryophytes, vascular plants, spiders, hoverflies and birds—and used data from 44 Sitka spruce and ash (*Fraxinus excelsior*) plantation forests. However, they remained provisional until tested on independent data and their applicability to plantations of other tree species and in other countries was unknown. The aim of the present study was to test these indicators in plantations of monoculture and mixed stands of tree species in both Ireland and Britain in order to assess their broader applicability. The stand types selected for study were monoculture Scots pine plantations, monoculture oak (*Quercus petraea/robur*) plantations and Scots pine/oak intimate mixes. Oak planted with Scots pine is recommended as a tree species mix in many parts of Europe (Río and Sterba, 2009), and increasingly in Ireland where it has been specifically promoted in recent years in forestry grant schemes (Guest and Huss, 2012). Traditionally Scots pine has been considered as a temporary nurse crop for oak, but this mix is gaining interest because of the ecological and socio-economic value of both tree species (Río and Sterba, 2009).

Objectives

- To determine a set of tested indicators of forest biodiversity for the range of forest types in Ireland.
- To support collaborative forest biodiversity research and exchange of information between Britain and Ireland.

Methodology

Site selection

Forty three stands of three different forest types were selected for this study across southern, eastern and central Ireland and at two locations in Britain (Figure 19). These forest types were monoculture Scots pine plantations, monoculture oak (*Quercus petraea/robur*) plantations and Scots pine/oak intimate mixes (hereafter referred to as 'mixed'). The minimum proportion of oak in mixed stands selected was 10%. A total of 13 stands were studied in Ireland (5 Scots pine, 4 oak plantations and 4 mixed) and 30 in Britain (10 of each type) (Figure 19). The minimum size of stand selected in Ireland was 5 ha, in order to be able to space sampling points for bird surveys adequately. In Britain, the size of all stands was at least 1.5 ha and bird surveys were not conducted. Stands with a high proportion of *Rhododendron ponticum* cover or where there was significant presence of tree species other than those mentioned above were not selected. The Sitka spruce and lodgepole pine stands from Work Package 1 were also included in some parts of the analysis.

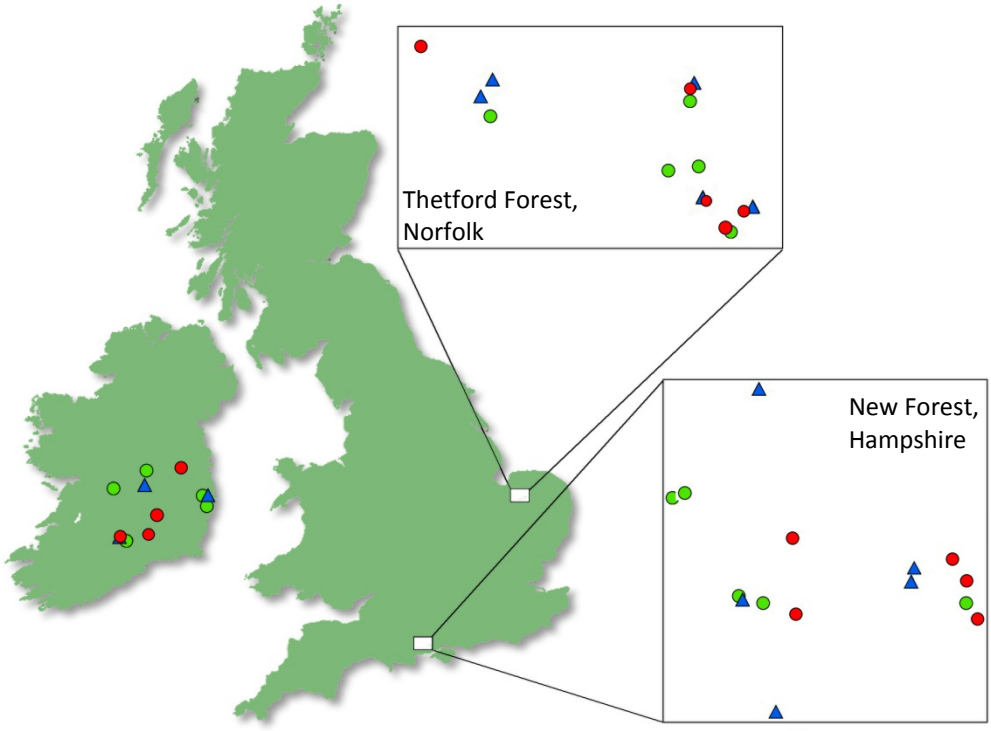


Figure 19. The location of Irish and British Scots pine ●, oak ● and Scots pine/oak mix ▲ sites.

Biodiversity Surveys

Data on the species richness and abundance of vascular plants, bryophytes, spiders and birds were collected at all study sites in Ireland, and of vascular plants, bryophytes and spiders in Britain, along with several structural and functional variables. The majority of data collection was carried out within 10m x 10m plots, each containing a single nested 2m x 2m plot; in Ireland there were three 10m x 10m plots per stand, while there were eight in Britain. The cover of bryophytes and of vascular plants was estimated within these plots and structural and soil data collected. Spiders were sampled using pitfall traps, which were plastic cups filled with 3cm of ethylene glycol. Irish traps that were vulnerable to disturbance were covered with a 10cm x 10cm corrugated plastic cover, which was suspended over the trap by metal pins, and all British traps were covered by a 19cm x 19cm steel cover. In Ireland, the traps were arranged in three transects of five traps adjacent to the 10m x 10m ground vegetation plots (15 traps per stand), while in Britain, there was one trap within each nested 2m x 2m plot (8 traps per stand). In Ireland, additional information on vegetation structure was collected in 1m x 1m plots around each pitfall trap. Birds were surveyed twice at study sites in Ireland, in May/early June and in June/early July, at between three and six point counts (Bibby *et al.*, 1992) per site. Point counts were located at least 100m apart and were conducted for 10 minutes, during which time the number and species of birds detected within 50m of the observer were recorded and their positions estimated. Structural data were also collected within 30m of each point. Further information on survey methodology can be found in Coote *et al.* (2013) and Fuller (2013).

Data analysis

The species richness of each taxonomic group was calculated at the plot/transect and stand level, as well as the species richness of various subgroups of species: (i) forest-associated species - species characteristic of forest/woodland in Ireland; (ii) open-associated species - species characteristic of open habitats; and (iii) generalist species –species either not entirely dependent on forest or open habitats, or having associations with both habitats. The relationships between the indicators identified by Smith *et al.* (2008) and Oxbrough *et al.* (2005) (Table 1) and species richness in Scots pine, oak, mixed, Sitka spruce and lodgepole pine stands were investigated using ANOVAs or t-tests for categorical variables and Pearson's correlation analysis for continuous variables. Additionally, generalised linear mixed models (GLMM) were used to test whether the relationships between the indicators and species richness were consistent across three stand types (Scots pine, oak and mixed stands) and three study areas (Ireland, New Forest and Thetford Forest).

Table 1. The structural, functional and compositional indicators developed by Smith et al. (2008) and Oxbrough et al. (2005), for which data were collected in the present study.

Structural	Functional	Compositional	
		Plants[a]	Birds[b]
Canopy cover	Age	*Rubus fruticosus*	Wren (*Troglodytes troglodytes*)
Shrub cover	Elevation	*Dryopteris dilatata*	Dunnock (*Prunella modularis*)
Field layer cover	Available P[c]	*Agrostis capillaris*	Blackbird (*Turdus merula*)
Ground layer cover		*Thuidium tamariscinum*	Pheasant (*Phasianus colchicus*)
Conifer litter cover		*Hypnum jutlandicum*	Robin (*Erithacus rubecula*)
Coarse woody debris volume		*Dicranum scoparium*	Treecreeper (*Certhia familiaris*)
Distance to forest edge		*Kindbergia praelonga*	Stonechat (*Saxicola torquata*)
Distance to old woodland		*Plagiothecium undulatum*	Greenfinch (*Carduelis chloris*)
Area of old woodland within 1km			Great Tit (*Parus major*)
			Blue Tit (*Parus caeruleus*)

[a] Presence of the full set of four species in each case
[b] Abundance of these species
[c] Indicators not tested on British data

Results

A number of the biodiversity indicators identified by Smith *et al.* (2008) were confirmed for Irish plantations in this study (Table 2). Canopy cover was confirmed as a structural indicator, particularly in conifer plantations, with bryophyte species richness higher under relatively high canopy cover plantations on poorly-drained soils. Bryophyte species richness declined under very high canopy cover, however. Canopy cover also had an important influence on shrub, field layer and litter cover, which in turn influenced the species richness of various groups. Bird species richness was found to be higher in more open plantations with high shrub cover, confirming this indicator. The relationship of spider species richness with field layer cover was found to be positive rather than negative and no relationships were found with ground layer or conifer litter cover. The species richness of forest-associated bryophytes was positively associated with the volume of coarse woody debris. The number of forest-associated vascular plants increased significantly with proximity to old woodland though the area of old woodland within 1km was not confirmed as an indicator.

Table 2. Summary of the stand-scale structural and functional indicators confirmed (P ≤ 0.05 for each) for Ireland and the taxonomic groups to which they apply.

Indicator	Taxonomic group
Structural	
Canopy cover	Bryophytes[a]
Shrub cover	Birds
CWD	Forest-associated bryophytes[a]
Proximity to old woodland	Forest-associated vascular plants
Functional	
Stand age	Forest-associated vascular plants & forest-associated spiders

[a] In conifer plantations only

Stand age was the only functional indicator confirmed for both forest-associated vascular plants and forest-associated spiders, with the species richness of both groups increasing with age (Figure 20). The relationship between ground-nesting birds and elevation was found to be negative rather than positive and no relationship was found between available phosphorus in the soil and vascular plant species richness. None of the vascular plant, bryophyte or bird species indicators were confirmed. The relationships between the findings of Smith *et al.* (2008) and those the present study are discussed in detail in Coote *et al.* (2013).

Testing of the indicators in only Scots pine, oak and mixed stands across a broader geographical area resulted in some differences in the relationships with species richness compared to those described above. Only two structural indicators – canopy cover and total litter cover - were confirmed to have the same relationships with bryophyte, vascular plant and ground-dwelling spider species richness as found by Smith et al. (2008) and, whilst these relationships held across the three stand types, few of these relationships were found to be consistent across Ireland, the New Forest and Thetford Forest (Table 3). Canopy cover had a positive relationship with total bryophyte ($z = 2.84$, $P < 0.01$) and forest-associated bryophyte species richness ($z = 2.33$, N.S.) in Thetford Forest, but a negative relationship with total bryophyte species richness in the New Forest ($z = -2.55$, $P < 0.01$), and no significant relationship with bryophyte species richness in Ireland. Conversely, in Ireland, canopy cover had a positive relationship with forest-associated vascular plant species richness ($z = 2.42$, $P < 0.05$) and negative relationship with open-associated spider species richness ($z = -3.39$, $P < 0.01$) but no effect on these species groups in the New Forest or Thetford Forest. Total litter cover was the only indicator to consistently have a negative relationship with total vascular plant species richness across all three study areas (Ireland: $z = -2.52$, $P < 0.01$; the New Forest and Thetford Forest: $z = -3.95$, $P < 0.001$). Litter cover again had a contrasting relationship with forest-associated vascular plant species richness in the British study areas, exhibiting a negative relationship in the New Forest ($z = -2.67$, $P < 0.01$) but a positive relationship in Thetford Forest ($z = 2.49$, $P < 0.01$). Total bryophyte species richness was also negatively affected by litter cover in Ireland ($z = -2.99$, $P < 0.01$), but not in the New Forest or Thetford Forest, whereas forest-associated spider species richness was positively affected by litter cover in the New Forest and Thetford Forest ($z = 2.22$, N.S.), but not in Ireland.

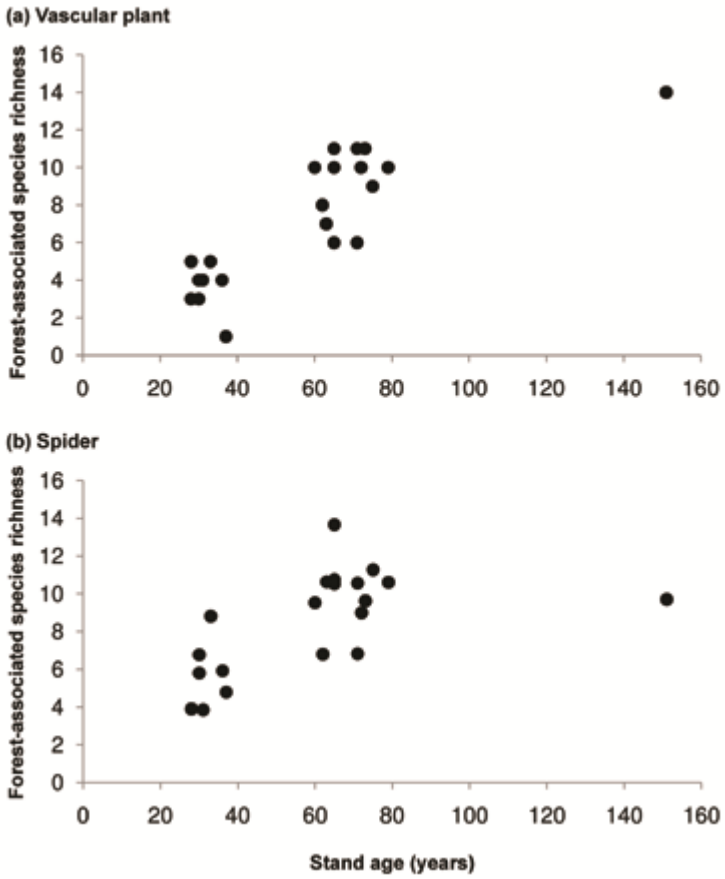


Figure 20. The relationship between stand age and the species richness of (a) forest-associated vascular plants and (b) forest-associated spiders in Irish plantations.

Table 3. Summary of the positive (+) and negative (-) structural indicators confirmed for Scots pine, oak and mixed stands and the taxonomic groups to which they apply. Relationships hold across all stand types.

Indicator	Ireland	New Forest	Thetford Forest
Canopy cover	Forest-associated vascular plants (+) Open-associated spiders (-)	Total bryophytes (-)	Total bryophytes (+) Forest-associated bryophytes (+)
Total litter cover	Total vascular plants (-) Total bryophytes (-)	Total vascular plants (-) Forest-associated vascular plants (-) Forest-associated spiders (+)	Total vascular plants (-) Forest-associated vascular plants (+) Forest-associated spiders (+)

Discussion

Several structural indicators and a single functional indicator have been confirmed as stand-level indicators of diversity in one or more taxonomic groups in Irish plantations. Conifer canopy cover was confirmed as an important biodiversity indicator, most likely due to the lower light intensities in the shade of conifers (Gates *et al.*, 1965). Bryophytes can survive at relatively low light levels compared to many vascular plants, and have a preference for high humidity conditions away from the competition of vascular plants (Trynoski and Glime, 1982; Bergamini *et al.*, 2001b). Although bryophyte richness was positively associated with canopy cover, the relationship was non-linear, with richness falling to low values at high canopy covers, as

previously identified in Sitka spruce plantations (Smith *et al.*, 2008; Moore, 2012). Light levels in these plantations fall below the threshold even of shade-adapted species (Hill, 1979; French *et al.*, 2008). Increased light levels at lower conifer canopy cover allow the vegetation to develop greater below-canopy species diversity and structural diversity (Hill, 1979; Eycott *et al.*, 2006; Moore, 2012). These in turn are important for supporting diverse bird (MacArthur and MacArthur, 1961; Sweeney *et al.*, 2010b), and invertebrate assemblages (Humphrey *et al.*, 1999; Oxbrough *et al.*, 2005). The requirements for bryophytes and the other groups studied may seem to be somewhat conflicting; it is however important to note that many of the bryophyte-rich plantations in the present study were on poorly drained peat soils in exposed, high rainfall areas, which are generally thinned to a limited extent or left un-thinned for tree stability reasons (Phillips, 2004). In terms of Scots pine, oak and mixed plantation forests in Britain, canopy cover was also an important positive indicator for total bryophyte and forest-associated bryophyte species richness in Thetford Forest. The contrastingly negative result of canopy cover on bryophytes in the New Forest may be due to a long history of free-ranging cattle and ponies meaning that the ground layer in these forests is adapted to a more open forest habitat (Mountford and Peterken, 2003). Overall, canopy cover did not predict consistent results across Ireland, the New Forest and Thetford Forest, potentially due to differences in land-use history and the rainfall gradient across these regions, affecting species composition and therefore species responses (Whitehouse, 2006; Grant and Edwards, 2008; Dolman *et al.*, 2010).

Total litter cover consistently indicated lower vascular plant species richness in Ireland and Britain and bryophyte species richness in Ireland, most likely due to its acting as a suppressant on vegetation diversity (Xiong and Nilsson, 1999). However, despite the negative effects of litter cover on ground vegetation species richness, it may benefit forest-associated plants, some of which are adapted to penetrate leaf litter, in contrast to other, more competitive plants which cannot (Sydes and Grime, 1981). Furthermore, it has been suggested that broadleaf litter is important in coniferous plantation forests to help mitigate the acidifying effect of needle litter, which may reduce the number of forest-associated plant species (Petit *et al.*, 2004). Litter cover also indicated forest-associated spider species richness in Britain and it provides vital habitat for many organisms, including litter-dwelling arthropods, which play an important role in nutrient cycling and provide prey for ground-dwelling spiders and other predatory species groups of the forest floor (Wise, 1993; Lawrence and Wise, 2000).

CWD volume was confirmed as a positive indicator of forest-associated bryophyte diversity in conifer plantations in Ireland. Deadwood is an important substrate for bryophytes in conifer plantations (Humphrey *et al.*, 2002) and the greater the volume present, the greater the chance of a range of decay classes being present, thus the greater the range of species supported (Andersson, 1991). Deadwood is also used by a number of other groups of organisms, including vascular plants, birds, invertebrates and fungi (Harmon *et al.*, 1986) and a large proportion of the species living in forests are dependent on it (Larsson, 2001).

Proximity to old woodland was confirmed as a positive indicator of forest-associated vascular plant diversity in Ireland. These woodlands act as seed sources for these species, which often have limited powers of dispersal (Brunet and von Oheimb, 1998; Dolman and Fuller, 2003; Rackham, 2003). Dispersal limitation may be an issue for other groups such as certain epiphytic lichens (Sillett *et al.*, 2000; Hauck, 2011), and invertebrates (Lindo and Winchester, 2008), but is less likely to be an issue for spiders, which can disperse over greater distances by ballooning (Duffey, 1998), or for birds (Harrison *et al.*, 1992).

Recommendation 8: Broadleaved tree species, particularly native species, should be favoured for planting on sites adjacent to semi-natural woodland. Conifer plantations adjacent to these woodlands should be considered or conversion to plantations of native tree species.

Stand age was confirmed as a positive indicator of forest-associated vascular plant and spider species richness in Ireland. The dispersal limitation of vascular plants has been discussed above, with the chance of additional species colonising therefore increasing with time (Brunet and von Oheimb, 1998); the same will likely also be the case for other dispersal-limited groups. For spiders, which are less dispersal-limited, the development of suitable habitat, such as shrub, field, ground or litter layers (Harvey *et al.*, 2002; Oxbrough *et al.*, 2005; Oxbrough *et al.*, 2012), is probably more important.

All of the identified indicators can be assessed without need for specialist taxonomic or technical knowledge and can be used to assess the potential value of stands for the taxonomic groups to which they apply. In order for stands of potentially high overall biodiversity to be identified, further research is required to identify additional biodiversity indicators, particularly those for other taxonomic groups. However, the current indicators can give insights into management practices that could enhance the diversity of the relevant taxonomic groups.

Recommendation 9: Use forest biodiversity indicators with caution, particularly when applying them to forest types or geographical areas other than those in which they have been developed or tested.

Recommendation 10: Use appropriate forest biodiversity indicators across different species and taxonomic groups.

Conclusions

Our results indicate that the structural and functional indicators tested here are not broadly applicable across forest stand types and geographical regions. Out of the tested indicators canopy openness, litter cover and field layer vegetation cover were the best predictors of species richness across both forested regions in England but these indicators had few similar relationships with species richness across England and Ireland. Biodiversity indicators for plantation forests should be used with caution as their ability to predict species richness can depend on other factors such as tree species, stand age, land-use history, climate and geographical region. Therefore, the use of stand scale indicators cannot be relied upon to ensure biodiversity conservation and enhancement in plantation forests and this method should be incorporated into forest management plans which also take into consideration other important landscape and regional scale factors.

Chapter 7

Ground-dwelling invertebrate diversity of mixed forest plantations

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Work Package 2B

The use of a mixed tree species composition has gained considerable attention in recent times as a means of increasing habitat heterogeneity and supporting greater biodiversity in plantation forests. However, although international forest policy is increasingly advocating stands of mixed tree species, not all scientific studies find supporting evidence. This study investigated the effects of a mixed tree species composition on ground-dwelling invertebrates compared with those in monoculture stands. Active ground-dwelling spiders and carabid beetles were sampled in 42 plantation oak monocultures, Scots pine monocultures, and intimate Scots pine and oak mixtures, across three different geographical regions in the UK and Ireland. Results revealed no significant consistent difference of spider and carabid beetle species richness in mixed stands compared with monocultures across the three regions investigated. There were also very few differences in species composition between the stand types in each region and few species of spider or carabid beetle species exhibited high associations with any of the forest stand types. In terms of the ground-dwelling invertebrates and tree species studied here, these results do not support the perception that mixed stands benefit biodiversity in plantation forests, nor do they support current forest policy which specifically promotes planting mixed forest stands for this purpose. However, there may be benefits for other taxonomic groups, such as canopy-dwelling invertebrates, and mixed stands merit further investigation to determine the optimum mixing percentages and planting patterns required in order to support forest biodiversity.

Background

Although in Ireland, and many other places, forests planted for timber production typically consist of just one, often exotic, tree species of just one age, mixed tree species composition is frequently proposed as a way to increase habitat heterogeneity and enhance biodiversity in forest plantations (Ishii *et al.*, 2004; Gamfeldt *et al.*, 2013). The use of mixed tree species plantations to enhance biodiversity in commercial forests has received considerable attention in forest policy and management plans in recent times (Spence *et al.*, 1997; Forest Service, 2000b; Spiecker, 2003). In practice, however, the impact of mixed tree species plantations on biodiversity is variable and depends on local conditions, planted tree species and the taxa under investigation (Uliczka and Angelstam, 2000; Work *et al.*, 2004; Jactel and Brockerhoff, 2007; Cavard *et al.*, 2011). In mixed forests, each tree species has slightly different physiological requirements, favouring the maximum use of site diversity. These forests have higher vertical and structural diversity and provide a greater variety of habitats for wildlife than their monoculture counterparts. Careful selection of tree species is required to ensure combinations of trees that use environmental resources efficiently and provide maximum benefits for biodiversity while, at the same time, being economically favourable. Because of the many variables involved, the usefulness of mixed species forest plantations for biodiversity conservation

requires investigation at local scales, to directly inform management strategies. The aim of the study of mixed plantations was to compare ground-dwelling spider and carabid beetle species assemblages and species richness in mixed and monoculture stands of Scots pine and oak, in three geographical regions across Ireland and Britain. Ground-dwelling spiders and carabid beetles were chosen for the study because they are useful indicators of forest management impacts, as they are sensitive to environmental change, have a broad geographic range and can be sampled and identified effectively (Uetz, 1979; Uetz, 1991; McGeogh, 1998; Pearce and Venier, 2006; Cameron and Leather, 2012).

Objective

- To compare the ground-dwelling spiders and beetles of mixed and monoculture plantation forests.
- To support collaborative forest biodiversity research and exchange of information between Britain and Ireland.

Methodology

Site selection

The 43 study sites are described in detail in chapter 6 (Figure 19). These forest types were monoculture Scots pine plantations, monoculture oak (*Quercus petraea/robur*) plantations and Scots pine/oak intimate mixes (hereafter referred to as 'mixed').

Biodiversity Surveys

Data on active ground-dwelling spiders and carabid beetles were collected at all study sites, along with environmental variables known to affect these species e.g. cover of vegetation layers, litter cover and canopy cover. The majority of data collection was carried out within 10m x 10m plots, each containing a single nested 2m x 2m plot. In Ireland there were three 10m x 10m plots per stand, while there were eight in Britain. Spiders and carabid beetles were sampled using pitfall traps, which were plastic cups filled with 3cm of ethylene glycol and dug into the ground so the rim was just below the ground surface. Irish traps that were vulnerable to disturbance were covered with a 10cm x 10cm square corrugated plastic cover, which was suspended over the trap by metal pins, and all British traps were covered by a 19cm x 19cm square steel cover. In Ireland, three pitfall trapping plots were established, placed at least 50m apart. These plots consisted of five traps placed 2m apart in a linear arrangement (15 traps per stand). In Britain, eight plots were established, placed at least 15m apart. One trap was placed in each of the eight plots (8 traps per stand). Further information on survey methodology can be found in Fuller (2013).

Data analysis

Data from each region (New Forest, Thetford Forest, and Ireland) were analysed separately. For all analyses, data were pooled across collection periods and forest stands. To test whether mixed plantation stands support a greater species richness of ground-dwelling spiders and carabid beetles than monocultures, the effect of stand type on the species richness of all species, habitat generalist species, forest specialist species, and open specialist species was analysed. This was conducted using Kruskal-Wallis rank sum tests followed by post-hoc tests using Wilcoxon pairwise rank sum tests with Bonferroni corrections.

Species composition was examined using permutational multivariate analysis of variance (PERMANOVA) to determine whether there were any differences in the species composition between each stand type in each region. Where stand type was found to have an effect, post-hoc pairwise comparisons were conducted with Bonferroni corrected P values for multiple comparisons. Indicator species analysis was used to identify species that have a high affinity for the different stand types. Additionally, stand level averages for each environmental variable were calculated from the plots and differences between stand types were tested for using the same Kruskal-Wallis rank sum test procedure as described above.

Results

Ground-dwelling spiders

In total 81 (n = 2,279), 86 (n = 3,418) and 81 (n = 2,463) ground-dwelling spider species were sampled in the New Forest, Thetford Forest and Ireland, respectively. Of these species, forest specialists constituted 44% of all spiders captured in the New Forest, while in Thetford Forest, forest specialists comprised only 27% of all identified spiders. However, in Ireland, forest specialist spiders dominated the stands, comprising 70% of all spiders captured. Habitat generalist ground-dwelling spiders constituted most of the remaining fraction of spiders in each region, with very few open habitat specialist species occurring in any of the regions (4%, 1% and 1% of all spiders in the New Forest, Thetford Forest and Irish stands, respectively). No significant difference in spider species richness was found between forest stand types in Ireland or the New Forest. However, in Thetford Forest species richness in Scots pine monocultures was 46% greater than in oak monocultures and 26% greater than in mixed stands ($\chi^2_{(2)}$ = 9.05, P < 0.05; Figure 21). Differences in specialist spider species among stand types showed no consistent trend between regions.

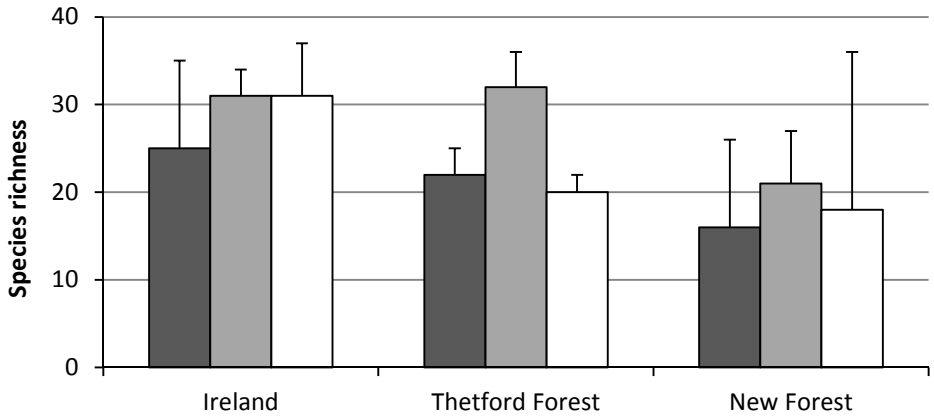


Figure 21. Median ± interquartile range of ground-dwelling spider species richness in oak monocultures (■), Scots pine monocultures (□) and Scots pine and oak mixes (□) in each region.

No effect of stand type was seen in the species composition of ground-dwelling spiders in the New Forest ($F_{2,12}$ = 1.42, P = 0.06) or in Ireland ($F_{2,9}$ = 1.21, P = 0.24). In Thetford Forest, however, there was a significant effect of stand type on species composition ($F_{2,12}$ = 2.02, P = 0.004); this was significantly different in Scots pine monocultures compared with the other two stand types, which were not significantly different from one another.

Carabid beetles

In total 21 (n = 4,059), 37 (n = 16,015) and 28 (n = 3,314) carabid beetle species were sampled in the New Forest, Thetford Forest and Ireland, respectively. Forest specialist species comprised a high percentage of all carabid beetles caught in the New Forest stands (55%) and in the Irish stands (49%). By contrast, forest specialist carabid beetle species were notably scarce in Thetford Forest stands, occurring in only 0.46% of all carabid beetles caught. The only region in which stand type significantly affected carabid species richness was Thetford Forest. Here, total carabid species richness was 35% higher in oak monocultures than in mixtures and 40% higher than in Scots pine monocultures ($\chi^2_{(2)} = 7.53$, $P < 0.05$) (Figure 22). There was no effect of stand type on the species composition of carabid beetles in the New Forest ($F_{2,12} = 1.37$, N.S.), Thetford Forest ($F_{2,12} = 1.78$, N.S.) or in Ireland ($F_{2,7} = 0.87$, N.S.). Additionally, there were no significant differences in the measured environmental variables between the stand types in each region.

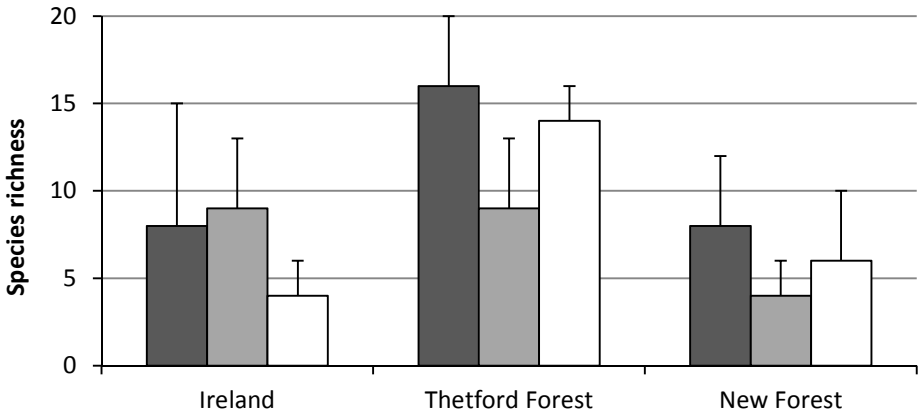


Figure 22. Median ± interquartile range of carabid beetle species richness in oak monocultures (■), Scots pine monocultures (■) and Scots pine/oak mixes (□) in each region.

Discussion

In all three regions, the forest stand types (Scots pine monocultures, oak monocultures and mixed stands) exerted a limited influence on the species composition and richness of both ground-dwelling spiders and carabid beetles. Where significant stand type effects were observed, monoculture stands supported higher richness than mixed stands, but the magnitude of these effects differed between regions. Therefore, our findings do not support the hypothesis that mixed tree species stands support higher species richness of ground-dwelling spiders and beetles than monocultures of either conifer or broadleaved tree species. The lack of any significant difference in the measured environmental variables between the three forest stand types studied, and the similarity of ground-dwelling invertebrate communities across these forest types, indicate a high degree of overlap in the ecological resource provisioning of the three stand types.

A high degree of similarity in habitat characteristics between mixed and monoculture stands has previously been reported in Norway spruce/Scots pine mixtures and Norway spruce-oak mixtures compared with Norway spruce monocultures (Oxbrough *et al.*, 2012). The authors of this study concluded that the lack of differences in habitat was related to the poor mixing ratio of oak with Norway spruce (15-40%). In the current study the mixing ratio of the broadleaf component was comparatively high, particularly in the English study sites (at least 40% oak in mixed stands), so if distinct environmental conditions were created by a Scots pine and oak mixture at this level, these should have been evident.

While this study and others (Barbier *et al.*, 2008; Oxbrough *et al.*, 2012) do not lend support to current practise of using mixed stands to enhance biodiversity (Spiecker, 2003; Fuller *et al.*, 2008; Taboada *et al.*, 2010), benefits may be seen for canopy dwelling organisms and other mixtures may be beneficial. Tree species mixes of interest in this regard would be intimate mixtures with more main canopy tree species and/or different tree species to those studied here. Both oak and Scots pine are native to Britain, and as such innately likely to support high numbers of insect and mite species (Kennedy and Southwood, 1984). Adding a native broadleaf component to a non-native conifer plantation, where there are likely to be fewer associated insects, might substantially increase the abundance and diversity of, for example, herbivorous invertebrates which fall from the canopy and, therefore, their associated predators (Butterfield and Malvido, 1992; Magura *et al.*, 2002). Adding a broadleaf tree species to a densely shading conifer plantation may also increase stand light levels and, consequently, levels of understory vegetation cover, potentially increasing stand structural diversity (Ishii *et al.*, 2004). However, Scots pine is a relatively lightly shading conifer, with some native Scots pine forests having notably high levels of shrub cover. In this study, canopy openness did not differ significantly between stand types. Finally, it should be noted that, the diversity of canopy-dwelling groups such as birds and canopy invertebrates is enhanced by the inclusion of oak in production forest stands (Sweeney *et al.*, 2010c; O'Connell *et al.*, 2012). The lack of an observed effect of oak on the diversity of ground-dwelling invertebrates in this and other studies should not be interpreted as evidence that mixes do not benefit biodiversity, merely that they do not enhance the diversity of the ground-dwelling species under investigation. It is likely that the benefits of mixed planting will vary between different taxa and, within taxa, between different guilds and across the range of ecosystem services (Cavard *et al.*, 2011; Gamfeldt *et al.*, 2013).

When evaluating these findings, it is important to note that the availability of potential study sites was limited by historical planting trends. The sites selected for this study were all commercially mature stands, but at the time that these were established, intimately mixed planting of conifers and broadleaved species were rare. As a result, suitable mixed stands were difficult to find, particularly in Ireland. Ideally, all mixed stands selected for this study would have been planted as mixes from the start, but in some of the stands the oak component appeared to have been established after the pine was planted. In some or all of these sites the majority of canopy oaks may have derived from natural regeneration as they were, for the most part, poorly developed, comprising an understorey below the main Scots pine canopy. Planting of broadleaves, particularly in mixes with conifers, has become more frequent in recent decades. As a result, it is likely that the types of forest required for a study such as this one will become more available in the future. Ideally, experimental stands would be established for the direct study of the effect of mixes on biodiversity, to best inform future management of the forest estate. Ideally research would be undertaken to examine the significance of the planting of different tree species mixes across a range of ecosystem services, including timber yield and biodiversity conservation.

Recommendation 11: Undertake research to identify mixed tree species plantation types that offer the optimal combination of economic and ecological benefits and establish experimental mixed plantations for long-term evaluation.

Conclusions

The inclusion of more than one tree species in a forest stand, particularly native broadleaf species, can increase habitat heterogeneity and enhance forest biodiversity. However, our study found no significant consistent effect of mixed or monoculture tree species on ground-dwelling spider and carabid beetle diversity. At the levels of mixing considered within this study (10 - 50% broadleaf component), and considering the two tree species under study (Scots pine and oak), mixed stands showed no influence on spider or carabid beetle diversity compared to monocultures of these species. This supports previous research suggesting that additional broadleaf canopy species confer no clear biodiversity benefits for ground-dwelling species (Barbier *et al.*, 2008; Oxbrough *et al.*, 2012), although they may have an influence at greater broadleaf to conifer mixing ratios. Further research is needed to establish whether a greater broadleaf component in mixtures can improve their biodiversity value over stands of simpler species composition and to identify species mixtures and configurations that are most beneficial.

However, it is important to consider the effects of mixed stands on taxa that occupy other strata of the forest, such as birds and canopy-dwelling invertebrates, as these species groups can benefit from mixed stands (Sweeney *et al.*, 2010c; O'Connell *et al.*, 2012). Therefore, the lack of an effect on the ground-dwelling invertebrates studied here should not be interpreted as evidence that mixed stands do not benefit biodiversity. These results have important implications for current forest policy in Ireland and the UK, which recommends the use of a mixed species composition at much lower ratios than considered here. Further investigation is needed to determine optimum forest management of mixed stands for biodiversity conservation of both ground-dwelling species and species which occupy higher strata of the forest environment. Potential areas of review for current policy and practice include investigating the benefits of admixing broadleaf tree species to a densely shading conifer tree species, to increase light levels and, consequently, levels of understory vegetation structural diversity and associated invertebrate and bird diversity (Ishii *et al.*, 2004). Increasing the percentage of broadleaf tree species in a mix from the current 5% broadleaf component in the UK (Forestry Commission, 2011) and 20% secondary species component in Ireland (Forest Service, 2000b) to above 50% of the mix should also be considered. Further investigation of the tree species used in a mix and the planting patterns used, e.g. grouped or banded mixtures rather than intimate mixtures, should also be undertaken.

Chapter 8

Bird diversity in semi-natural and plantation oak forests

Conor T. Graham, Mark W. Wilson, Sandra Irwin, Thomas C. Kelly and John O'Halloran
Work Package 2C

The planting of native broadleaved tree species has gained considerable attention in recent times as a means of increasing habitat heterogeneity and supporting greater biodiversity in plantation forests. This study investigated whether oak plantations can support comparable bird assemblages to semi-natural oak woodlands, and to assess whether high levels of ungulate grazing impact on the quality of semi-natural oak woodland habitats for birds. Bird and vegetation surveys were conducted in commercially mature oak plantations (n = 4), semi-natural oak woodlands (n = 10) and very heavily grazed semi-natural oak woodland (n = 4). Bird densities and species richness were compared between forest types. This study found that bird diversity in plantation oak and semi-natural forests was similar, with no difference in species richness, total bird density or the density of either warbler or hole-nesting bird species. However, heavily grazed semi-natural oak woods had lower species richness than either of the other two study site types, and a lower density of warblers than oak plantations. These observed differences in bird communities are related to grazing mediated differences in habitat complexity between the forest types. These findings demonstrate that plantation forests of native tree species may support comparable bird communities to semi-natural woodlands in areas with a generalist bird fauna from which forest specialist species are absent. Bird diversity in woodlands subject to high levels of grazing is likely to be limited, unless ungulate populations and their access to these woodlands are managed with a view to promoting the development of a more complex understorey.

Background

Prior to historical deforestation, the island of Ireland was largely covered in woodlands of oak, elm and other broadleaved species, but today semi-natural forests are rare in Ireland, covering just 1.1% of the country (Forest Service, 2013). Extensive afforestation schemes which have increased forest cover in Ireland use mostly non-native conifer tree species with broadleaf plantations currently accounting for just 2.7% of Ireland's land cover (Forest Service, 2013). However, broadleaf planting has increased substantially in recent decades and made up 36% of all planting between 2009 and 2012 (Forest Service, 2012). Ireland is conspicuously lacking in forest specialist birds even when compared to the bird assemblages of forests in Britain, where there are far fewer forest specialists than on mainland Europe (Fuller *et al.*, 2007). This is due to a combination of Ireland's geographical location as an island at the western extremes of Europe and the paucity of forest cover in Ireland over the last few centuries (Fuller *et al.*, 2007; O'Halloran and Kelly, 2012). The on-going expansion of plantation forest cover in Ireland and many other European countries (FAO 2007), in combination with the decline in woodland bird species throughout the continent (Fuller et al. 2005, Gregory et al. 2007), mean that assessments of the bird communities of plantation forests are timely (Sweeney et al. 2010a). This is particularly true in Ireland were plantations are potentially of greater

conservation importance than in regions where forest cover is dominated by native woodlands (Berndt et al. 2008, Bremer & Farley 2010).

While many studies have shown that plantations of exotic species contribute to the conservation of biodiversity (Humphrey *et al.*, 2003; Wilson *et al.*, 2006; Brockerhoff *et al.*, 2008), native species are generally considered preferable for biodiversity conservation because of their higher value as habitat for native species (Brockerhoff *et al.*, 2008; Bremer and Farley, 2010). Several studies have noted that oak trees (*Quercus robur* & *Q. petraea*) have the highest number of associated canopy invertebrates of any tree in Britain (Southwood *et al.*, 1982; Kennedy and Southwood, 1984), with beneficial impacts for forest species such as birds that rely on invertebrates (Whittingham *et al.*, 2001). The value of oak forest as a habitat may however be fundamentally affected by herbivores. Deer, in particular, are increasing in abundance in Ireland (Carden *et al.*, 2010) and can exert a considerable influence on the vegetation structure of woodlands (Fuller and Gill, 2001; Pellerin *et al.*, 2010). The impacts of browsing can have cascading impacts on the biodiversity of birds (Martin & Joron 2003, Allombert et al. 2005a, Gill & Fuller 2007, Newson et al. 2012) through destruction of nest sites, increased vulnerability to nest predation and changes in food supply (Allombert *et al.*, 2005; Newson *et al.*, 2012).

Objectives

The main objectives of this study were to:

- Evaluate whether oak plantation forests can support similar bird communities to those of semi-natural oak woodlands.
- Assess the extent to which variation in the bird communities of semi-natural woodlands was related to intensive browsing by ungulates.

Methodology

Study sites

Bird assemblage and habitat data were collected from fourteen semi-natural oak forests and four commercially mature oak plantations (ranging in age from 72 to 151 years) in Ireland (Figure 23). Of these fourteen semi-natural oak woodlands, four sites were located in one area around Killarney in County Kerry and are subject to particularly intensive grazing by wild populations of Sika (*Cervus nippon*) and Red (*Cervus elaphus*) deer.

Data collection

Vegetation variables were estimated visually at each of the six point count locations at each site. Variables recorded were canopy height (m), canopy cover including understorey cover (%), percentage shrub cover (woody vegetation between 0.5-2m in height) and percentage ground vegetation cover (0.5m in height or less). Measurements from individual point locations were averaged to generate site means.

Birds communities were sampled at six point locations in each study site using point count methodology (Bibby *et al.*, 1992). Surveys of the plantations and the browsed semi-natural oak forests were carried out in the summer of 2011, and of the semi-natural oak forests in the summers of 2007 and 2008. Data were collected at each site over two visits, one in May/early June and the second in June/early July, between the

hours of 07:00 and 18:00. One morning and one afternoon survey were conducted at each site. This minimised effects of time of day on bird detectability, by excluding periods in the early morning and evening when birds are known to be particularly active. During each five minute count, the species, position and behaviour of all birds detected within 100m of the observer were recorded. Bird surveys were not conducted in winds greater than Beaufort scale 4 or in heavy or persistent rain. Clusters of birds of the same species were recorded as having a maximum number of two individuals to reduce the influence of family parties with fledglings on density estimates. Flying birds were excluded from the analyses as their presence could not be assumed to indicate an association with that habitat.

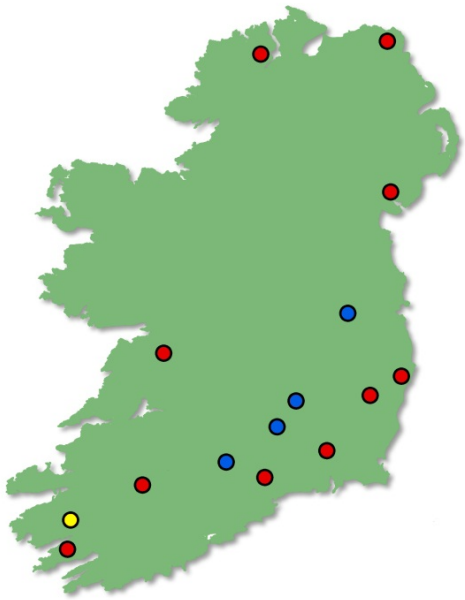


Figure 23. Study site locations: ● Natural oak (n = 10); ○ Grazed natural oak (n = 4); ● Plantation Oak (n = 4).

Data analysis

Distance software was used to derive individual species densities from field observations collected during point counts, with the distributions of distances at which birds were recorded being used to derive detection functions. Each species was assigned to one of four detection groups based on aspects of the ecology and behaviour likely to influence species' detectability. Species richness, total bird density, density of warblers (Blackcap, Chiffchaff, Grasshopper Warbler, Garden Warbler, Whitethroat and Willow Warbler) and density of hole-nesting species (Blue Tit, Coal Tit and Great Tit) were calculated for each study site. Species richness was calculated as the cumulative number of species recorded over both visits in each site.

General linear models were used to test for differences between each of the habitat variables between sites. We used an information theoretic model averaging framework based on Akaike's Information Criterion corrected for small sample sizes (AICc) to model relationships between site environmental variables between forest types. As forest type was collinear with each of the vegetation variables, generalised linear models assuming Gaussian and Poisson distributions for densities and species richness, respectively, with forest type as the single response variable, were run to calculate the residuals from these models. The remaining variation in the data was then analysed using generalised linear models using a Gaussian distribution to identify how much residual variation in species richness, total bird density and

density of warbler and hole-nesting species was related to the recorded vegetation variables. Variables included in each analysis were canopy cover and height, shrub cover and ground cover. Top models were assessed based on AICc. To determine the most important explanatory variables in the GLMs, we applied model averaging (Burnham & Anderson 2002) using the dredge function in the R library. Further details of data collection and analysis are provided in (Graham *et al.*, 2014).

Results

Bird species richness was lower in the heavily grazed semi-natural oak forests in Killarney than in the other forest types (Figure 24). There was no significant difference in total bird density or the density of hole-nesting bird species between the three forest types. The density of warblers was lower in the heavily grazed semi-natural oak forests than in the oak plantations, but there was no difference in warbler density between the semi-natural oak forests and the other forest types.

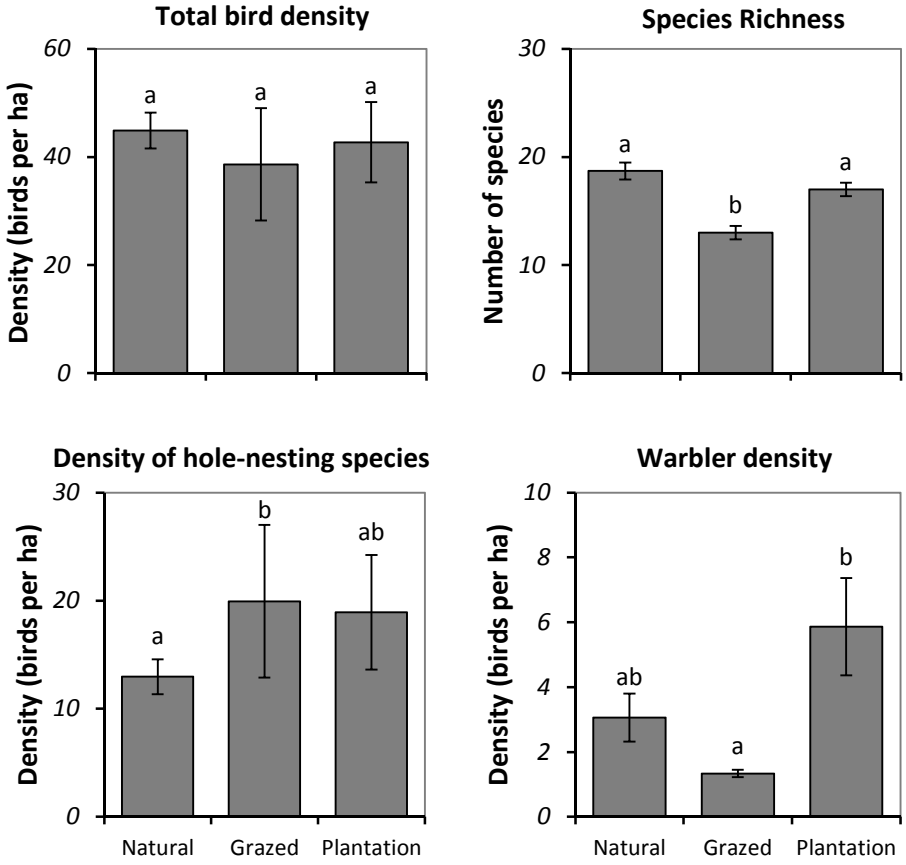


Figure 24. Mean (± standard error) total density of birds, bird species richness, density of hole nesting species and density of warblers, in each of the oak forest types. Lowercase letters above error bars refer to homogenous subsets indicated by general linear models.

The percentage cover of both canopy and shrub were significantly lower in the heavily grazed semi-natural oak forests than in the other two forest types (Figure 25). Canopy height was significantly taller in the oak plantations than in the other two oak forest types (Figure 25). While ground cover was greater in the heavily grazed semi-natural oak forests relative to other semi-natural oak forests, there was no significant difference in ground cover between the oak plantations and the other two forest types (Figure 25). There

was a difference in the relative importance of the habitat variables in determining bird species richness, total bird density, warbler density and hole-nesting species density (Table 4). Canopy cover was the best predictor of residual variation in species richness and total bird density, with the remaining vegetation variables being of lower relative importance (Table 4). Although canopy cover was the most important variable in models of residual density of hole-nesting species, none of the explanatory variables of this model had high importance values. Shrub cover had the highest relative importance value for modelled residual variation of warbler density. However, as with the hole-nesting species density model, all of the vegetation variables of this model had relatively low variable importance values.

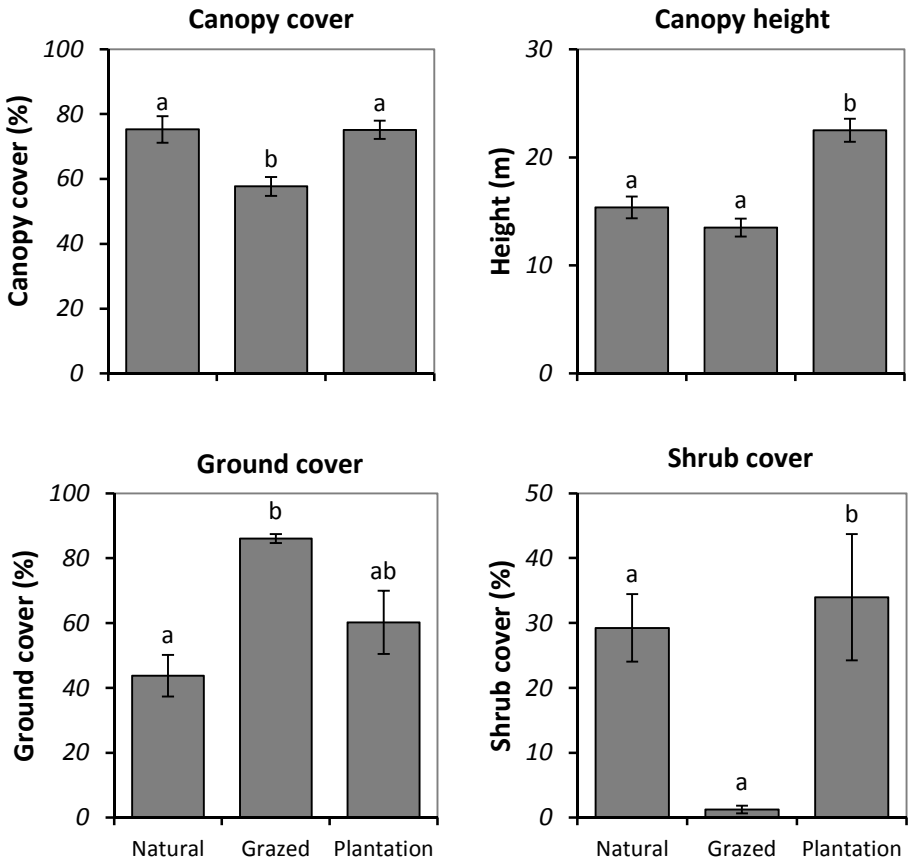


Figure 25. Mean (± standard error) canopy cover, canopy height, ground cover and shrub cover in each of the oak forest types. Lowercase letters above error bars refer to homogenous subsets indicated by general linear models.

Table 4. Explanatory variable parameter estimates, standard errors (± standard error) and significance values of each of the top models, and the relative variable importance (RVI) from the model averaging, from the general linear models demonstrating the relationship of measured environmental variables on the bird species richness and density of all birds, hole nesting birds and warbler species in the three types of oak forest. Note warbler density was square root transformed prior to analyses to meet assumptions of the test.

Explanatory	Total Density			Species Richness			Warbler Density			Hole-nester Density		
Variable	Estimate	P	RVI	Estimate	P	RVI	Estimate	P	RVI	Estimate	P	RVI
Intercept	14.09(19.5)	0.48	-	-1.13(0.62)	0.085	-			-			-
Canopy Cover	0.41(0.27)	0.152	0.46	0.016(0.01)	0.083	0.72			0.17			0.36
Canopy Height			0.21			0.37			0.15			0.12
Shrub Cover			0.11			0.17			0.35			0.14
Ground Cover			0.23			0.28			0.32			0.25

Although the oak plantations sites were separated from the semi-natural oak forest sites in ordination space (Figure 26), this separation was on axis 1, which explained just 8% of the variation in bird communities between all sites. Axis 1 scores were significantly negatively correlated with canopy height and shrub cover as were the shrub associated species, Blackcap, Wren, Chiffchaff and Dunnock. The canopy associated bird species Goldcrest, Chaffinch, Siskin and Treecreeper were weakly positively correlated with axis 1 of the ordination as was canopy cover. There was no separation between the forest types on axis 2, which accounted for 82% of variation between sites, indicating that the bird assemblages of the three types of oak forests were broadly similar to one another. However, hole-nesting birds and Goldcrest were negatively correlated with axis 2 of the ordination, which was associated with canopy cover, the most important variable from the model averaging of hole-nesting bird density. Further details of these results are presented in (Graham *et al.*, 2014).

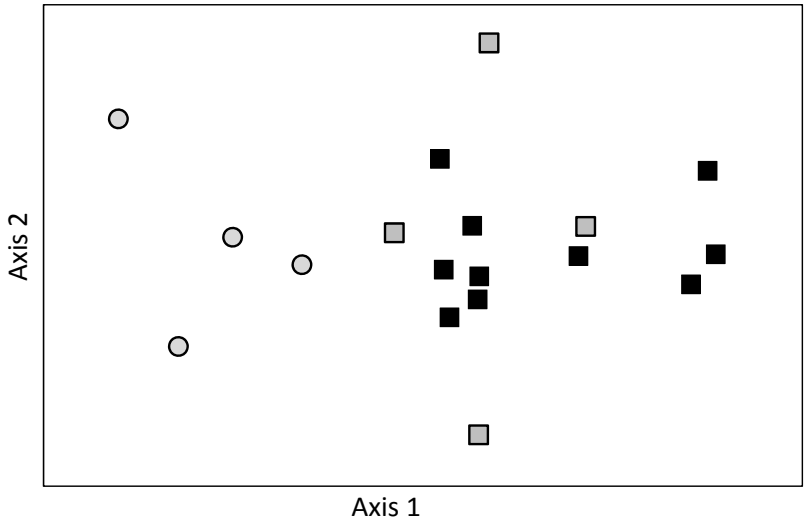


Figure 26. NMS ordination plot of bird community data from the three types of oak forest: ■ Natural oak (n = 10); □ Grazed natural oak (n = 4); ○ Plantation Oak (n = 4). Final stress from two-dimensional solution = 12.9, final instability = 0.00042. Axis 1 and 2 explain 6 and 82% of the variation in the dataset, respectively.

Discussion

The bird diversity of oak plantations in this study compared favourably to that of semi-natural oak forests, with no difference in species richness, total bird density, warbler density or density of hole-nesting birds between these forest types. The heavily grazed semi-natural oak woodlands of Killarney National Park, however, had lower bird species richness than the other two forest types and a lower density of warbler species than the oak plantations. These observed differences in bird communities were related to variation in habitat structural complexity, which is a well-established driver of forest bird communities (Wilson *et al.*, 2006; Quine *et al.*, 2007; Sweeney *et al.*, 2010b; Sweeney *et al.*, 2011; O'Connell *et al.*, 2012).

The bird communities of the semi-natural oak forests and oak plantations also compare very favourably to previous studies in Irish semi-natural forests. The mean species richness in both semi-natural oak forests (18.7) and oak plantations (17.0) in this study were comparable to the mean species richness of the semi-natural forests in previous studies, which ranged from 18.6 to 21.0 (Batten, 1976; Wilson, 1977; Nairn and Farrelly, 1991). Comparisons of bird densities between our study and these previous studies are

impracticable, as because we used *Distance* analyses to correct for the influence of distance from the observer on detectability, while these studies did not.

The oak plantations included in this study supported very similar bird communities to the semi-natural oak forests, demonstrating that plantation forests of native tree species can support comparable bird communities to semi-natural woodlands in areas such as Ireland with a generalist bird fauna lacking lacks forest specialists. The role of plantations in the conservation of forest biodiversity has been debated extensively (Brockerhoff *et al.*, 2008; Bremer and Farley, 2010; Paillet *et al.*, 2010; Coote *et al.*, 2012). However, the lower levels of biodiversity commonly reported in plantation forests may, in part, be due to the fact that most of these plantations are comprised of exotic or non-native species (Lindenmayer *et al.*, 2003; Sweeney *et al.*, 2011).

These results clearly show that the oak plantations and semi-natural oak forests supported bird communities with similar species richness, total density and density of typical woodland groups such as hole-nesting species and warblers. The diversity of bird species recorded in the oak plantations here (17.0 bird species per site) is higher than has been previously been found by studies of bird communities in Irish conifer plantation forests. For example, Sweeney *et al.* (2010c) reported average species richness values of between 11.0 and 14.5 species in commercially mature Norway spruce (*Picea abies*) mix plantations, respectively. Studies in plantations of Ireland's most widely planted tree species, Sitka spruce (*Picea sitchensis*) also found lower bird diversity than the oak plantations of this study, with averages of between 12.3 (Sweeney *et al.*, 2010b) and 12.6 (Wilson *et al.*, 2006) species per site. The bird communities of closed canopy conifer plantations in Ireland tend to be dominated by relatively few generalist bird species, particularly Coal Tit, Goldcrest and Chaffinch, and typically support few (if any) bird species of conservation concern (Wilson *et al.*, 2006) (Wilson et al. 2006, Sweeney et al. 2010c). This demonstrates that, at least in regions without forest specialist species, plantations of native tree species can support comparable bird biodiversity, in terms of communities and species composition, to those of semi-natural woodlands.

Recommendation 12: Plantations of native tree species should be planted where biodiversity enhancement is a priority.

Somewhat surprisingly, densities of hole-nesting birds in the oak plantations sites were comparable to those in the semi-natural oak forests. Populations of cavity-nesting species in plantations can be limited by the lack of old, cavity-rich trees, due to the fact that plantations in Ireland are typically harvested before partial and complete tree death can give rise to an abundance of cavities. Even though oak trees rarely develop cavities or hollows, the vast majority of forests in Ireland have, in any case, been subjected to harvesting of wood for timber and fuel, and low densities of cavities may also result from historic exploitation of older trees (Sweeney *et al.*, 2010d). Another contributing factor may be the historical absence of woodpeckers, which have only recently re-colonised Ireland after an absence of several centuries (McDevitt *et al.*, 2011).

Despite lower species richness and warbler densities in the heavily grazed oak semi-natural forests than at other study sites, the bird communities of these woodlands, as indicated by NMD ordination, were very similar to those of the other semi-natural oak forests. The results of the ordination indicate, similar to the models on total density, species richness and density of warbler species and of hole-nesting species, that

what little variation there was between bird assemblages was related to aspects of the forest vegetation structure, and in particular, shrub cover and canopy cover.

There were significant differences in bird diversity between the heavily grazed and other semi-natural oak woodlands in this study, with woodlands subjected to intensive browsing by deer having significantly lower species richness than the other two forest types. Warbler density was positively related to shrub cover, and it is therefore likely that the lower density of warblers in the grazed sites was related to the high levels of deer browsing at these sites, a phenomenon which has previously been reported by studies of woodland sites in Britain (Perrins and Overall, 2001; Gill and Fuller, 2007) particularly for species of warbler (Perrins and Overall, 2001). The amounts of both canopy cover and shrub cover in browsed semi-natural oak forests were lower than the other two forest types. In fact, canopy cover was positively related to species richness, total bird density and density of hole-nesting species. It is therefore likely that the development of shrub and understorey vegetation in heavily grazed semi-natural oak forests was suppressed by deer rather than by shading. The low levels of canopy cover in the heavily grazed oak woodlands were due principally to a lack of understorey vegetation in these forests, presumably as a result of high levels of ungulate browsing. The high levels of canopy cover in the oak plantations were not due to dense understorey vegetation but, rather, to the high density at which crop trees were planted. As populations of deer are increasing throughout Ireland and much of temperate Europe (Carden *et al.*, 2010), the suitability of heavily grazed semi-natural oak forests for several bird species can be improved through the development of a more complex understorey. This would require steps to facilitate a reduction in the browsing of these woodlands by ungulate species, through increased management of their density within such landscapes.

Recommendation 13: The suitability of heavily browsed semi-natural oak woodlands for several bird species can be improved through management aimed at promoting the development of a complex understorey.

Conclusions

This study revealed that, in this context, plantation forests of native tree species can support bird communities comparable to those of semi-natural woodlands. This finding is highly relevant at this time as plantation forest cover is expanding across the world and in particular in Europe, at the same time that populations of forest birds are declining. While the planting of native tree species in plantations is increasing in countries such as Ireland, exotic conifers continue to dominate new plantings. Populations of deer are increasing throughout temperate Europe, with deleterious effects on woodland birds as a result of browsing mediated differences in vegetation complexity. This study found that the suitability of heavily grazed semi-natural oak forests for several bird species can be improved through management aimed at promoting the development of a more complex understorey and moderate grazing levels. This would require steps to facilitate a reduction in the browsing of these woodlands by ungulate species, through increased management of their density within such landscapes.

Chapter 9

Indicators of woodland biodiversity potential

Linda Coote, Daniel L. Kelly and Fraser J. G. Mitchell
Work Package 3

This study set out to examine the use of indicators to assess the potential biodiversity of native woodlands in Ireland. In particular, the use of indices that combine a number of woodland biodiversity indicators in a single measure (composite indices) were examined. Following an extensive review of indicators and composite indices, the French 'Potential Biodiversity Index' was identified as the best model on which to base an index for Irish woodlands. The National Survey of Native Woodlands (NSNW) was identified as the best dataset on which to test this index of woodland biodiversity potential This index was modified as required and applied to the National Survey of Native Woodlands dataset using eight biodiversity indicators (five that can be influenced by forest management and three that describe the woodland context) to obtain composite scores of potential biodiversity. The potential of the index to summarise information relating to semi-natural woodland biodiversity, and scope for its inclusion as a component of reporting on national-scale biodiversity potential was demonstrated.

Background

Although the term 'biodiversity' was coined just twenty-five years ago (Wilson, 1988), it is now recognised as a fundamental concept in modern conservation. Recognition of the need to protect and enhance woodland biodiversity is reflected in numerous global agreements including the 'Forest Principles' and Agenda 21, which deal with sustainable forest management and sustainable development respectively, and the Convention on Biological Diversity. The need for monitoring and reporting progress under such international agreements and the impossibility of recording all species has created an interest in the development of indicators to assist signatories to monitor forest biodiversity and to assess performance of policy measures put in place for the conservation and sustainable management of woodland biodiversity (Puumalainen *et al.*, 2003; Boutin *et al.*, 2009; Lamb *et al.*, 2009). Integral also to the compliance monitoring process is a 'benchmark' or a measure of an acceptable standard of what actually represents a healthy or desirable condition (Ferris and Humphrey, 1999). However, not all indicators developed are applicable at both the national and woodland scales (Newton and Kapos, 2002), with the latter most useful to forest managers, allowing indicator information to be used to guide action on the ground (Ferris and Humphrey, 1999).

The indicator concept is based on the principle that easily measured features that affect or derive from variation in biodiversity, can be used as a proxy for fine-scale measurement of biodiversity (Landres *et al.*, 1988; Ferris and Humphrey, 1999). Indicators of biodiversity are always only an expression of the *potential* rather than the *actual* biodiversity that is present, as the creation of new habitats and niches does not guarantee that they will be filled by the expected organisms (Noss, 1990). Indicators of woodland biodiversity can be separated into those indicators that can be influenced by woodland management and

those that cannot (Table 5), both of which have an important influence on woodland biodiversity potential (Larrieu and Gonin, 2008). While the former are the most useful in terms of informing woodland management, the latter are important in terms of identifying woodlands of potentially high biodiversity value independent of the management regime.

Table 5. Indicators of woodland biodiversity potential.

Influenced by woodland management	Independent of woodland management
Tree species composition	Area
Structural diversity	Woodland continuity
Deadwood	Associated habitats
Veteran trees	
Open areas	

As detailed in chapter 6 and reflected in the abundance of scientific literature, no single indicator of biodiversity can be devised (Duelli and Obrist, 2003; Larrieu and Gonin, 2008). Something more useful to decision makers than long lists of individual indicators is concise information that combines lists of indicators into bottom-line messages (Failing and Gregory, 2003), and the optimal approach is to select a 'basket' of indicators (Duelli and Obrist, 2003). An index of biodiversity is a mathematical construct that summarises the effect of two or more indicators in a single number or value. Such aggregations of several indicators into indices of biodiversity enable policy makers and stakeholders to have a clear indication about forest management performance with respect to biodiversity objectives (Montreal Process, 1995; Failing and Gregory, 2003). The suite of indicators included in such an index must be sufficiently comprehensive to capture the full range of biodiversity that occurs in forests and woodlands and yet concise enough to function as a practical tool for land managers (McElhinny et al., 2005). Finally, in order to achieve reliability and broad acceptance indices must be tested for their correlation with a substantial and quantifiable portion of actual biodiversity to test whether the biodiversity *potential* identified by the index is reflected in the *actual* biodiversity present (Duelli and Obrist, 2003).

A number of authors have proposed such composite indices that are applicable at the forest management unit scale (Van Den Meersschaut and Vandekerkhove, 1998; McElhinny *et al.*, 2006; Larrieu and Gonin, 2008). While many such indices involve complex, time-intensive recording and some degree of specialist knowledge, the French Potential Biodiversity Index (Larrieu and Gonin, 2008) is quick, easy to use and requires no specific level of expertise beyond the knowledge of native tree genera. The index includes a wide range of biodiversity indicators, many of which are recognised in international agreements as being important to forest biodiversity. It most notably also includes indicators that are independent of forest management that are frequently overlooked despite their potential to greatly enhance woodland biodiversity. The Potential Biodiversity Index also has the advantage of having been widely tested and refined by its authors in woodlands throughout France (Larrieu and Gonin, 2008). Although the authors suggest that this index is applicable in Atlantic, continental and European boreal regions (Larrieu and Gonin, 2008) it has, to date, not been formally applied in woodlands outside France.

Objectives

This study set out to identify indicators of woodland biodiversity potential for Ireland and, to apply these indicators to native woodlands and in order to evaluate their performance test the validity of selected indicators against actual biodiversity data. The specific objective of this study was to:

- Determine forest condition criteria for Native Woodlands in Ireland.

Methodology

Data source

Following an intensive review of available Irish datasets the Irish National Survey of Native Woodlands (NSNW) dataset (Perrin *et al.*, 2008) was identified as the most appropriate for use in this study. The NSNW collected data for the majority of the indicators identified as being important in the compilation of an index of biodiversity potential, and was the only dataset containing data at the stand rather than plot scale. This dataset contains data from 1,217 woodland sites located across all 26 counties of the Republic of Ireland, collected between 2003 and 2007. The sites ranged from relatively young plantations and naturally regenerated woodlands on previously unforested sites to ancient semi-natural woodlands, with sites of varying ages, site histories and degrees of management intervention between. The data are based on a general site survey during which the presence or abundance (on the DAFOR scale: Dominant, Abundant, Frequent, Occasional, Rare) of plant species and other features within the entire site were noted. Further information on methodology can be found in Perrin *et al.* (2008).

Index of woodland biodiversity potential

A review was undertaken of available indices of woodland biodiversity potential include the Belgian Stand-scale Forest Biodiversity Index (Van Den Meersschaut and Vandekerkhove, 1998), the French Potential Biodiversity Index (Larrieu and Gonin, 2008) and the Austrian Forest Biodiversity Index (Geburek *et al.*, 2010). The index of biodiversity potential developed in this study was based on the *Potential Biodiversity Index* (Indice de Biodiversité Potentielle, Larrieu and Gonin, 2008), which uses seven indicators that are related to woodland management (native tree species, vertical structure, large diameter standing deadwood, large diameter fallen deadwood, large living trees, living trees bearing microhabitats and open areas) and three context-specific indicators (woodland continuity, aquatic habitats, rocky habitats) that describe characteristics of the site. A score of zero, two or five is given to each indicator based on threshold values that reflect the expected proportional increase in biodiversity associated with each indicator. The combined indicator scores for the indicators associated with woodland structure and management and the contextual indicators can be given separately as well as being added together to give a total IBP score. The index can also be expressed as a percentage of the maximum possible score and be broken down into scoring ranges that can be assigned to potential biodiversity ratings from 'Low' to 'High'.

As the NSNW data were collected for a different purpose, they were not always in the form needed for the application of the *Potential Biodiversity Index*. Therefore some modifications were required in order to apply the *Potential Biodiversity Index* to the NSNW data (Table 6). Since two of the indicators (open areas and large living trees) could not be assessed using the NSNW data, eight indicators were included in the present study and the maximum possible score for the structure and management indicators was 25 and the maximum total score was 40. The scoring ranges were therefore adjusted from those used in the *Potential Biodiversity Index* to maintain the same percentage scores for the classes (Table 7).

Table 6. The scoring system used in the index of woodland biodiversity potential.

Indicator	Score
STRUCTURE AND MANAGEMENT	
NATIVE TREE SPECIES • Number of native genera (following Meikle (1984) for willows and Webb *et al.* (1996) for other species) • Alive, no height restriction • Score capped at 2 if cover of native species less than 10%	0 = 0-2 genera 2 = 3/4 genera 5 = ≥5 genera
VERTICAL STRUCTURE • 4 layers: a) Herbaceous and semi-woody (most abundant of the categories 'herbs' and 'low woody species') b) Low woody (most abundant tree species DBH[a] <7cm, ht 25.1cm-2 m) c) Intermediate woody (most abundant tree species DBH[a] <7cm, ht >2m) d) High woody (most abundant tree species DBH[a]≥7 cm) • Woody species only counted in a single layer • Layers not counted unless scored D, A or F on DAFOR[b] scale	0 = 1/2 layers 2 = 3 layers 5 = 4 layers
LARGE-DIAMETER STANDING DEADWOOD • Dead trees, snags/snapped (most abundant of the 2 categories), any height • No DBH[a] cut-off • AFOR[b] rating given with absent/R equivalent to <1/ha, O to ≥1 & <3/ha and A/F to ≥3/ha	0 = <1/ha 2 = ≥1 & <3/ha 5 = ≥3/ha
LARGE-DIAMETER FALLEN DEADWOOD • D ≥ 5cm (any length) • AFOR[b] rating given with absent/R equivalent to <1/ha, O to ≥1 & <3/ha and A/F to ≥3/ha • No capping possible	0 = <1/ha 2 = ≥1 & <3/ha 5 = ≥3/ha
LIVING TREES BEARING MICROHABITATS • Type of microhabitat: major branches lost or crown damage • Abundance of trees with microhabitat assessed on AFOR[b] scale with absent/R equivalent to <1/ha, O/F to ≥1 & <6/ha and A to ≥6/ha • No maximum applied	0 = <1/ha 2 = ≥1 & < 6/ha 5 = ≥6/ha
CONTEXTUAL WOODLAND CONTINUITY • All or part of woodland present approximately 160 years ago = old woodland, which can be evaluated from the 1st edition 6 inch to 1 mile (1830-1843) Ordnance Survey maps (maps.osi.ie) • Partially cleared or probable old woodland not assessable	0 = Not part of old woodland 2 = Partially cleared or probable old woodland 5 = Clearly part of old woodland
AQUATIC HABITATS • Types (natural or artificial): spring, watercourse, lake/pond, bog, fen/flush, marsh, swamp • Within or bordering	0 = absent 2 = 1 type (homogeneous) 5 = ≥2 types (diverse)
ROCKY HABITATS • Types: rock or boulders; walls or ruined buildings • Within or bordering • Rock or boulders within stand counted if O or above on DAFOR[b] scale, adjacent to stand counted if present; walls or ruined buildings within and adjacent counted if present [a] DBH = Diameter at Breast Height (1.3m). [b] (D)AFOR = (Dominant), Abundant, Frequent, Occasional, Rare.	0 = absent 2 = 1 type 5 = ≥2 types

Table 7. The scoring classes used in the index of woodland biodiversity potential.

Structure and Management Indicators			Contextual Indicators			Total		
Score		Rating[a]	Score		Rating[a]	Score		Rating[a]
Total	%		Total	%		Total	%	
0-5	0-20	Low	0-5	0-33	Low	0-8	0-20	Low
6-10	21-40	Rel. low	6-10	34-67	Mod.	9-16	21-40	Rel. low
11-15	41-60	Mod.	7-15	68-100	High	17-24	41-60	Mod.
16-20	61-80	Rel. high				25-32	61-80	Rel. high
22-25	81-100	High				33-40	81-100	High

[a] Rel. = Relatively, Mod. = Moderate

Data analysis

The index of woodland biodiversity potential was applied to the Irish National Survey of Native Woodlands (NSNW) dataset. Eight biodiversity indicators were applied to obtain indicator-specific and combined composite scores of potential biodiversity. Analyses involved examination of all 1217 NSNW sites combined, regardless of woodland type, as well as the separate examination of the different Irish semi-natural woodland types defined by Fossitt (2000). The latter was carried out to determine whether any unique compositional or structural features of the different woodland types influenced their scoring by the index. Yew woodlands (WN3, Fossitt, 2000), that is woodlands dominated by yew (*Taxus baccata*), are very rare in Ireland (Fossitt, 2000; Perrin *et al.*, 2006), and only two sites were surveyed in the NSNW. Therefore, this woodland type was excluded from the separate analysis, but the two sites were included in the combined analysis. The six remaining semi-natural woodland types are detailed in Table 8. Only woodlands with ≥90% of their area assigned to a particular woodland type during the NSNW survey were included in the separate analysis by semi-natural woodland types, with woodlands classified as highly modified/non-native excluded from this part of the analysis. This resulted in a subset of 760 sites (62% of all sites surveyed) used in the separate analysis by semi-natural woodland type.

Table 8. Classification of semi-natural woodlands included in this study (After Fossitt, 2000).

Woodland Classification
Oak-birch-holly (WN1)
Oak-ash-hazel (WN2)
Wet pedunculate oak-ash (WN4)
Riparian (WN5)
Wet willow-alder-ash (WN6)
Bog (WN7)

Principal Components Analysis (PCA) was carried out on the matrix of the scores of each of the eight indicators at 1213 of the sites (4 sites had missing values for certain indicators) in order to derive information on the potential biodiversity of each woodland and to examine how each of the six semi-natural woodland types compared overall in their scores for each of the eight indicators. Analyses were carried out in PC-ORD 6.07.

In order to assess the performance of the index, the total IBP scores were correlated with woodland area. These scores were also compared for all woodlands greater than 5 ha between those classified as both ancient and possibly ancient woodland (continuously wooded since 1660) by Perrin et al. (2010) and more recent woodlands, as well as between those woodlands designated as candidate Special Areas of Conservation (cSAC) under the EU habitats Directive and/or Statutory Nature Reserves under the Wildlife Act, 1976 and the Wildlife (Amendment) Act, 2000 and undesignated sites.

The scores for individual indicators and the ratings for the structure and management, contextual and total IBP for all woodlands in the dataset and for the six semi-natural woodland types were also examined in order to assess the suitability of the indicator thresholds for the range of woodland types. For example, if the majority of sites receive the highest score for an indicator the threshold may be too low, whereas if the majority receive the lowest score it may be too high. The differences in the scores received by the six semi-natural woodland types for the individual indicators and in their ratings for the structure and management, contextual and total IBP were tested with Kruskal-Wallis tests, followed by Mann-Whitney U tests. Significant differences in scores could indicate that the different woodland types required different scoring thresholds.

Results

The Principal Component Analysis revealed that standing deadwood, tree microhabitats, woodland continuity and rocky habitats had greater weight than the other indicators in determining the variation among sites. The orientation of the vectors in the two-dimensional space suggests that native tree species and rocky habitats have a strong positive correlation. All other vectors are relatively distinctly oriented, suggesting little correlation between them. The Principal Component Analysis plots (Figure 27), which indicates the location of the sites of each of six semi-natural woodland types in the combined ordination, show that the different woodland types are quite evenly spread in the reduced space defined by the first two principal components. However, the first principal component, which was positively correlated with woodland continuity and rocky habitats and negatively correlated with aquatic habitats, somewhat separated the oak-birch holly (WN1) and oak-ash hazel (WN2) sites from all other semi-natural woodland types.

The total IBP scores for the woodlands were significantly but relatively weakly positively correlated with woodland area ($n = 1212$, Spearman's rho = 0.359, $P < 0.001$) and were also significantly higher ($Z = -4.650$, $P < 0.001$) for ancient or possibly ancient woodlands ($n = 107$, total IBP = 25.5±0.5) than for more recent woodlands ($n = 653$, total IBP = 22.8±0.2). Designated woodlands had significantly higher ($Z = -8.068$, $P < 0.001$) total IBP scores ($n = 302$, total IBP = 24.4±0.3) than undesignated ones ($n=910$, total IBP = 21.3±0.2).

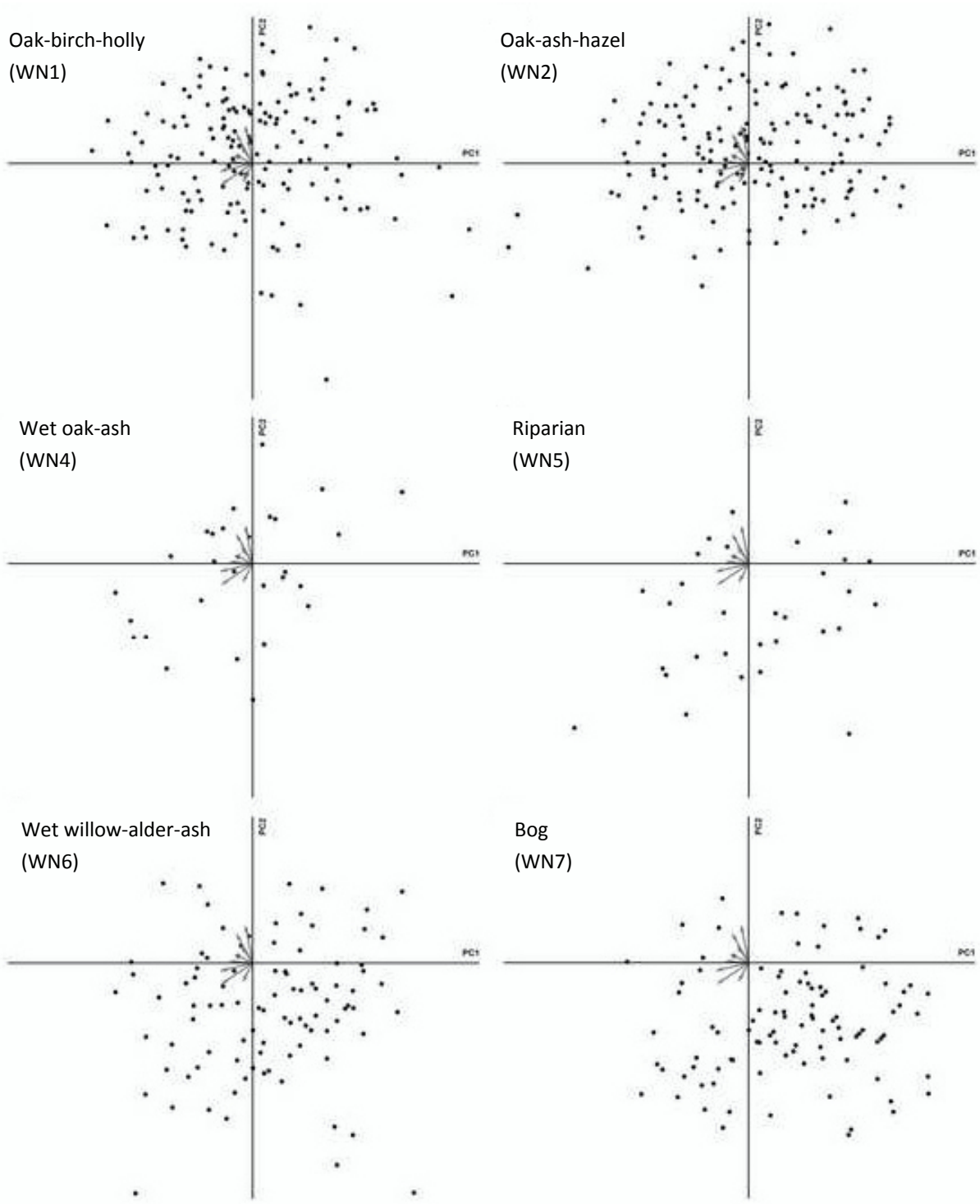


Figure 27. The first two principal components (PC1 = 20% and PC2 = 16% of variance extracted) of the PCA ordination of the eight indicator scores for the 1213 woodlands.

For all sites combined, the composite scores for the woodland structure and management indicators rated few of the sites of semi-natural woodland type as having either low or high potential biodiversity, with the majority rated relatively high (Table 9). For the contextual indicators, a low proportion were rated high and for the total IBP score the majority of semi-natural woodlands of all types were rated as having moderate potential biodiversity, with few rated as either low or high. When the six semi-natural woodland types were examined separately, bog woodlands had a significantly lower rating than all other woodland types for their

composite scores for the contextual indicators and for the total IBP. Riparian and wet willow-alder-ash woodlands also had a low proportion of sites scoring high for contextual and total IBP.

Table 9. The percentage of all NSNW sites and of each of the six individual semi-natural woodland types receiving the various ratings for woodland structure and management, contextual and total index. Differences in ratings between woodland types (χ^2) and p-value presented, with post-hoc Mann-Whitney U tests indicated by lowercase letters. Woodland types are significantly different (P < 0.05) if they do not share the same letter.

Rating	All	Oak-birch-holly (WN1)	Oak-ash-hazel (WN2)	Wet oak-ash (WN4)	Riparian (WN5)	Wet willow-alder-ash (WN6)	Bog (WN7)
n	1217[a]	203 [a]	247	30	32	110	138 [a]
Structure and Management							
χ^2= 19.0	P < 0.01	a	b	ab	abc	abc	c
Low	1.6	1.9	1.6	3.3	0	1.8	1.5
Relatively low	12.5	10.4	10.5	0	18.8	11.8	21.3
Moderate	37.8	40.1	31.2	36.7	37.5	40.9	41.2
Relatively high	41.7	43.6	48.2	50.0	37.5	38.1	30.1
High	6.4	4.0	8.5	10.0	6.25	7.3	5.9
Contextual							
χ^2=134.6	P < 0.0001	a	b	abc	cd	d	e
Low	35.2	20.2	32.4	40.0	50	55.5	76.1
Moderate	45.2	50.7	51.0	36.7	40.6	37.3	22.5
High	19.6	29.1	16.6	23.3	9.4	7.3	1.4
Total							
χ^2=77.0	P < 0.0001	a	a	a	ab	b	c
Low	1.1	1.0	0	0	0	0.9	4.4
Relatively low	18.2	11.9	17.8	13.3	21.9	25.5	40.4
Moderate	45.8	43.1	44.9	46.7	50.0	49.1	44.9
Relatively high	32.3	41.1	34.4	36.7	28.1	23.6	10.3
High	2.6	3.0	2.8	3.3	0	0.9	0

[a] Missing values - ALL: n=1213, WN1: n=202, WN7: n=136 for Structure and Management and Total

Discussion

Using the National Survey of Native Woodlands dataset and a modified version of the French Potential Biodiversity Index (Larrieu and Gonin, 2008) this study demonstrated the feasibility of assessing the potential biodiversity of a woodland using a set of relatively easily measured forest biodiversity indicators combined in an index. The PCA results showed that the majority of the individual indicators provided unique information about the potential biodiversity of the NSNW sites. However, native tree species and rocky habitats were strongly positively correlated, despite describing quite different characteristics of the woodlands. This may be explained by the fact that the best examples of semi-natural woodland in Ireland, often only survived because of their position on rocky ground that could not be cultivated (McEvoy, 1944). Also, only two types of rocky habitat could be distinguished (rock/boulders and walls/ruined buildings) using

the NSNW data compared with the nine types proposed in the original index (Larrieu and Gonin, 2008). Although most of the remaining seven types of rocky habitat are likely to have been encompassed in the two types that were assessed, assessment of all nine types individually may not have resulted in the observed correlation.

The eight indicators assessed did not have equal weight in determining the variation among sites in their total index scores. There is an inherent weighting towards deadwood in the original index (Larrieu and Gonin, 2008), as it is assessed separately in both the standing and lying deadwood indicators, and also to some extent in the microhabitats associated with trees indicator. This weighting is based on the large proportion of woodland species known to be dependent on deadwood. These aforementioned deadwood indicators were not found to be correlated in the current study, suggesting that they all provide unique information on the sites. However, the low or absent diameter cut-offs for assessing deadwood using the NSNW data meant that large diameter deadwood was not assessed, as prescribed by the original index and this may have influenced this result. Previous research has found Irish woodlands to be lacking in large diameter deadwood (Sweeney *et al.*, 2010d), suggesting the sites may have scored poorly for the deadwood indicators had the proposed diameter cut-offs been applied. However, both large and small diameter deadwood are known to be valuable for biodiversity and so its assessment quantifies an important biodiversity resource (Humphrey *et al.*, 2005; Caruso *et al.*, 2008).

The examination of the IBP scores showed a significant positive correlation with woodland area, although this was weak. There is usually a direct relationship between the size of a stand and the number of species it contains (Connor and McCoy, 2001). Since the majority of the NSNW woodlands were small or very small (Perrin *et al.*, 2008), and a number of the IBP indicators are assessed on a per hectare basis the weak relationship is perhaps unsurprising. Ancient woodlands had significantly higher IBP scores than more recent woodlands and the former have been found to support significantly higher species diversity of plants (Perrin and Daly, 2010) and other groups (Assmann, 1999; Alexander *et al.*, 2003). Designated woodlands had significantly higher IBP scores than undesignated ones. Perrin and Daly (2010) found that the majority, although not all, of the NSNW woodlands they considered to be of high conservation value were protected by some form of designation. These results suggest that the relationship between the index scores and biodiversity is relatively robust. However, addition of area as an indicator in the contextual part of the IBP may greatly improve this relationship.

Recommendation 14: Investigate the addition of further indicators, including woodland area, to the potential biodiversity index in order to improve the correlation between potential biodiversity and the index scores.

Overall the proportion of semi-natural woodlands in this study rated as having high potential biodiversity was low. This is surprising given that the dataset should contain a large proportion of the better examples of Irish woodlands in terms of naturalness. However, even the best examples of Irish woodland have been highly fragmented and modified, hence their description as semi-natural rather than native woodlands (Fossitt, 2000). This may result in a lower potential biodiversity than might be expected in ancient woodlands that have not been influenced by human activities.

This study has also demonstrated that the index of woodland biodiversity potential has the potential to be applied to semi-natural woodlands, particularly in regions such as Ireland, where such woodlands are much modified and focussed management has the potential to greatly improve their biodiversity status. The use of an index of woodland biodiversity, such as that derived in the current study would allow woodland management to be focussed on areas requiring improvement, as well as assisting with long-term monitoring. Ideally, the decision should ideally be made to apply this index prior to surveying, and purpose collected data covering the full set of recommended indicators included. Further examination of the application of the index to a range of woodland types, including plantation forests, in Ireland, using the original methodology would be informative.

Recommendation 15: Further test the potential biodiversity index using purpose collected data covering the full set of recommended indicators in both native woodlands and plantation forests in Ireland and the UK, using taxa that have not been used to generate the index across a range of spatial scales and forest ages.

Conclusions

The implementation of an index of woodland biodiversity such as that derived in the current study could be used to allow woodland management to focus on areas requiring improvement, as well as assisting with long-term monitoring. Ideally, the decision should be made to apply this index prior to surveying, as retrospective application may result in certain indicators being inadequately assessed. Because the measurement of total species diversity is not possible, other than at very small scales (Lawton et al., 1998), the relationship between the potential biodiversity as assessed by the index and the actual biodiversity of a woodland cannot be examined and there must be a reliance on the results of research to guide the selection of indicators. Further examination of the results of application of the index to a range of woodland types in other regions, and in Ireland, using the original methodology would be informative.

Chapter 10

Forest connectivity

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Work Package 4

Habitat fragmentation is a major threat to biodiversity in Ireland's forests. Forest fragmentation is linked to biodiversity loss and Ireland's forests estate is currently very fragmented within a predominantly agricultural landscape. Despite recent increases of over 5,500 ha of forest area per annum, the predominantly private nature of afforestation in Ireland offers little in terms of increasing connectivity between existing forest patches. Conserving and improving landscape connectivity is gaining acceptance as a key management objective to mitigate the negative effects of forest fragmentation. This study set out to analyse how forest landscape connectivity impacts on the diversity of ground dwelling spiders and birds of Irish forests. The Probability of Functional Connectivity (PFC), which accounts for both within and between forest patch connectivity, had no significant impact on spider diversity but was negatively related to total bird diversity (including generalist species and those of open habitats) and positively related to broadleaved-associated bird species diversity. Thus forest specialist bird species are negatively affected by habitat fragmentation and decreasing patch size in Irish forests. These findings demonstrate the importance landscape connectivity for the conservation of forest-associated biodiversity.

Background

Within existing forest habitats, woodland structure and habitat quality are important factors for species conservation, while at the landscape scale both structure and connectivity are important determinants of biodiversity conservation (Dolman *et al.*, 2007). Because of its history of extensive deforestation followed by commercial afforestation, most recently with small, privately owned plantings, the forest estate in Ireland is highly fragmented. Recent planting trends in Ireland, particularly the increase in private plantings, means that forest fragmentation will be increased in the future, even where total forest area increases (ITGA, 2012; Forest Service, 2013). With the proposed expansion of the national forest estate to as much as 1 million ha (circa 14.5% of land area) by 2030 and the requirement to manage these new forests in an ecologically sustainable manner, information on the importance of connectivity between these new forests is essential to inform policy and practice.

Forest fragmentation is one of the main threats to forest biodiversity, with the responses of different species to fragmentation varying in magnitude and direction (Fahrig, 2003; Henle *et al.*, 2004; Echeverría *et al.*, 2007). There is a large, diverse, body of literature on effects of habitat fragmentation on biodiversity, with different authors measuring fragmentation in different ways and, as a consequence, drawing different conclusions regarding both the magnitude and direction of its effects (Fahrig, 2003). What is clear is that that species in isolated patches are more likely to become extinct than those in larger habitat patches (MacArthur and Wilson, 1967), and that with increasing fragmentation of forests, the degree to which these fragments are functionally linked becomes increasingly important (Calabrese and Fagan, 2004). This

connectivity has many implications for ecological services, particularly biodiversity conservation in fragmented landscapes (Summerville, 2004; With, 2004).

Forest connectivity is important for species level biodiversity, species dispersal, pollination and genetic interchange (Bailey, 2007; Shanthala Devi *et al.*, 2013) and reduces the negative effect of patch size (Martensen *et al.*, 2008). Forest connectivity in a landscape is influenced by the physical location of habitat patches and by the characteristic of the surrounding habitat matrix (Baskent and Jordan, 1995). Landscape connectivity is the degree to which the landscape facilitates or impedes movement among resource patches (Taylor *et al.*, 1993). It refers to both 'structural connectivity' and 'functional connectivity' and is a vital component of landscape structure with implications for the survival of plant and animal populations. Structural connectivity refers to the physical relationship between habitat patches (Kadoya, 2009) and functional connectivity refers to the degree to which the landscape facilitates or impedes the movement of organisms and processes (Taylor *et al.*, 1993). The response of different plant and animal species to connectivity is species specific and is related to the behaviours, habitat preferences and dispersal ability of each species (Calabrese and Fagan, 2004).

Managing forest landscapes to sustain functional connectivity is one of the key strategies to counteract the negative effects of climate and human-induced landscape changes on forest biodiversity (Bailey, 2007; Rubio *et al.*, 2012; Ernst, 2014), and improving connectivity between woodland patches offers reduced isolation and helps to maintain biodiversity (Ricketts, 2001; Bailey, 2007). The maintenance of hedgerows is the only measure for improving forest connectivity included in the *Forest Biodiversity Guidelines* (Forest Service, 2000b). Despite being a frequently recommended management strategy, there are no accepted standards for the measurement of connectivity or its application to forest management strategies (Kindlmann and Burel, 2008; Watts and Handley, 2010; Saura *et al.*, 2011a). This study set out to use data collected during the BIOFOREST and FORESTBIO projects to provide information on forest connectivity in Ireland and its role in biodiversity conservation, with input from Forest Research, UK.

Objectives

The specific objective of this study was to:

- Test the forest connectivity factor of a forest monitoring programme using GIS databases amassed on previous projects.

Methods

The effect of landscape scale forest connectivity on the biodiversity of Irish forests was investigated using data collected by surveys conducted during the BIOFOREST, FORESTBIO and BIOPLAN projects. The diversity of ground-dwelling spiders and birds from a range of forests of various ages and types were collated, representing the full geographical and ecological range of the Republic of Ireland (Table 10).

Table 10. The number of each forest site type included in the analyses of the impact of forest connectivity on the diversity of plants, ground-dwelling spiders and birds in the Irish landscape.

Forest Type	Ground-dwelling spiders	Birds
Semi-natural forests		
Ash	10	10
Oak	17	13
Yew	2	2
Broadleaf plantations		
Ash (pre-thicket)	0	4
Ash (thicket)	4	4
Ash (mature)	4	4
Oak (mature)	4	4
Conifer & conifer dominated plantations		
Norway Spruce/Oak mix	2	2
Norway Spruce/Scots pine mix	4	4
Scots pine/Oak mix	4	4
Sitka spruce/Lodgepole pine dominated	8	8
Norway Spruce	6	6
Scots Pine	4	5
Sitka spruce dominated (pre-thicket)	30	34
Sitka spruce dominated (thicket)	13	13
Sitka spruce dominated (mid-rotation)	9	9
Sitka spruce dominated (mature)	12	17
TOTAL	133	143

Data analysis

For each site, the total ground-dwelling spider and forest-associated spider species richness, and total bird and broadleaved associated bird species richness were calculated. As the diversity of invertebrates and birds within forests is related the vegetation structure and structural variables within forests (Oxbrough *et al.*, 2006a; Wilson *et al.*, 2006; Sweeney *et al.*, 2010b; Sweeney *et al.*, 2011), these site level environmental variables also collected during the BIOFOREST and FORESTBIO surveys were also collated (Table 11).

In order to calculate landscape level metrics that potentially influence the diversity of forest habitats, a 1 x 1m resolution map of the Republic of Ireland of those habitats that influence forest habitat connectivity in the landscape, was assembled using ArcGIS in the form of a raster catalogue. Each of the approximately 80 billion 1 x 1m cells comprising this map was assigned a habitat type of either: semi-natural forest, hedgerow, young or commercially mature broadleaf, conifer or mixed (conifer and broadleaf) plantation forest, water or other habitat. Young forests were classified as those 20 – 30 years since first planting and commercially mature plantations were classified as those plantations established for 30 or more years. Plantations younger than 20 years were not included in analyses as they were considered not to have been established for a sufficient period in order to play a role in functional habitat connectivity. Using this habitat map of the Republic of Ireland, a series of landscape scale connectivity metrics were calculated in ArcGIS

(Table 11). Each of these landscape metrics were calculated for each individual site at a patch level, with site patch defined as the patch of adjoined forest within which each study site was located.

Table 11. Details of all metrics included in the general linear models identifying the relationship between biodiversity and site environmental and landscape scale metrics. Forest type was included in all models as a categorical variable.

Metric	Spider diversity	Bird diversity
Site environmental metrics	✓	
% ground vegetation cover (0 - 10cm high)	✓	✓
% lower field vegetation cover (10 - 50cm high)	✓	
% shrub cover (50 - 200cm high)	✓	✓
% fine woody debris cover (≤ 10cm diameter)	✓	
% coarse woody debris cover (≥ 10cm diameter)	✓	
% needle and leaf litter cover	✓	
% cover of bare ground	✓	
% cover of tree canopy		✓
Canopy height (m)		✓
Site location metrics	✓	✓
Elevation (m)	✓	✓
Latitude	✓	✓
Longitude	✓	✓
Patch level metrics		
Area of semi-natural forest within patch (m^2)	✓	✓
Area of young broadleaf plantation within patch (m^2)	✓	✓
Area of mature broadleaf plantation within patch (m^2)	✓	✓
Area of young mixed plantation within patch (m^2)	✓	✓
Area of mature mixed plantation within patch (m^2)	✓	✓
Area of young conifer planation within patch (m^2)	✓	✓
Area of mature conifer plantation within patch (m^2)	✓	✓
Perimeter of patch (m)	✓	✓
Area of broadleaf patch (m^2)	✓	✓
Perimeter of broadleaf patch (m)	✓	✓
Patch landscape scale metrics		
Distance to nearest semi-natural woodland (km)	✓	✓
Distance to nearest broadleaf plantation (km)	✓	✓
Distance to nearest mixed plantation (km)	✓	✓
Distance to nearest conifer (km)	✓	✓
Probability of functional connectivity (PFC)	✓	✓

Using the ArcGIS raster of habitat in Ireland, the probability of functional connectivity (PFC), which incorporates the least-cost distance between forest habitat patches and accounts for both within and between forest patch connectivity, was calculated for each study site (Watts and Handley, 2010). Least-cost distance values (Table 12) reflect the ability or cost of a species to move through the landscape matrix of various habitat types and colonise a forest habitat. Habitats with a low least-cost value have a high permeability to forest species, with high least-cost values indicating a hostile habitat for forest species. Two versions of the PFC metric was calculated for each study site, the first (generalist) with least-cost values assigned for a forest generalist to move through the landscape matrix and the second (specialist) with least-cost values assigned for a forest specialist to move through the landscape matrix (Table 12).

General linear models, assuming a Gaussian distribution and using forward and backward model selection, were used to identify those site level environmental variables and landscape scale connectivity metrics related to: total ground dwelling spider and forest-associated spider species richness, and total bird and broadleaved associated bird species richness. Variables included in each of these models are detailed in Table 12. Due to slight differences in sampling effort between sites, prior to analyses, species richness data was rarefied and standardised the R package, Vegan (Oksanen *et al.*, 2010).

Table 12. Least-cost distance values for each of the habitats in the ArcGIS raster of habitat in Ireland for both forest generalist and forest specialist species, which were incorporated in the PFC landscape metric.

	Generalist	Specialist
Native	1	1
Broadleaf (Mature)	1	2
Broadleaf (Young)	1	12
Conifer (Mature)	1	10
Conifer (Young)	1	12
Mixed (Mature)	1	5
Mixed (Young)	1	12
Hedge	10	10
Water	50	50
Everything Else	20	20

Results

Bird diversity

The total diversity of bird species was dependent on the vegetation structure within sites, the location of sites and landscape scale metrics but not on forest type. The amount of both shrub and canopy cover within a site were strongly, positively related to total species richness of birds, with increasing canopy height being negatively associated with total species richness of birds. Both elevation and latitude were negatively but longitude positively related to the diversity of total bird species richness (Table 13). The area and perimeter of broadleaf patch were negatively and positively related to total bird species richness, respectively. At a landscape scale, distance to the nearest patch of semi-natural woodland and the probability of functional connectivity metric (PFC) were both negatively related to total bird species richness.

Table 13. Results of the general linear model on the impact of forest type, site environmental and location metrics as well as patch scale and landscape scale connectivity metrics on the total and broadleaf associated bird species richness.

	Total bird species richness		Broadleaf associated bird species richness	
Metric	**Estimate (±se)**	**P**	**Estimate (±se)**	**P**
Intercept*	55.3 (20.7)	0.009	2.76 (0.61)	<0.0001
Forest type				
Ash – Semi-natural			2.59 (0.66)	
Ash – Pre-thicket plantation			-1.3 (0.80)	
Ash – Thicket plantation			0.36 (0.80)	
Norway spruce – Mature plantation			0.21 (0.73)	
Norway spruce/oak – Mature plantation			2.51 (0.98)	
Norway spruce/Scots Pine – Mature plantation			0.39 (0.80)	
Oak – Mature plantation			2.06 (0.79)	
Oak – Semi natural			2.00 (0.65)	
Scots pine – Mature plantation			0.59 (0.82)	
Scots pine/oak – Mature plantation			1.46 (0.82)	
Sitka spruce – Mature plantation			-0.18 (0.67)	
Sitka spruce – Mid rotation plantation			0.57 (0.72)	
Sitka spruce – Pre-thicket plantation			-0.69 (0.62)	
Sitka spruce – Thicket plantation			1.34 (0.68)	
Sitka spruce/Lodgepole pine – Mature plantation			-0.45 (0.73)	
Yew – Semi natural			-4.85 (2.03)	
Site environmental metrics				
% shrub cover (50 - 200cm high)	0.06 (0.01)	<0.0001	0.03 (0.01)	<0.0001
% tree canopy cover	0.07 (0.01)	<0.0001		
% coarse woody debris cover (≥ 10cm diameter)	-0.01 (0.05)	0.011		
Site location metrics				
Elevation (m)	-0.01 (0.00)	<0.0001	-3.52E-03 (1.35E-03)	0.01
Longitude	-0.74 (0.37)	0.049		
Latitude	0.78 (0.28)	0.006		
Patch level metrics				
Area of young conifer plantation within patch (m^2)			-5.86E-08 (3.74E-08)	0.12
Area of patch that is broadleaf (m^2)	-2.96E-06 (1.61E-06)	0.067	-1.75E-06 (7.24E-07)	0.018
Perimeter of patch that is broadleaf (m^2)	3.69E-04 (1.23E-04)	0.003		
Patch landscape scale metrics				
Distance to nearest semi-natural woodland (km)	-3.91E-04 (2.09E-04)	0.064	-2.08E-04 (8.41E-05)	0.015
Probability of functional connectivity (PFC)	-1.83E-06 (6.53E-07)	0.006	1.71E-06 (7.38E-07)	0.022

*Mature ash plantation was included in the intercept

Spider diversity

Forest type, vegetation structure and site location were all important determinants in total spider species richness (Table 14). Nearly all of the forest types had significantly higher total species richness than mature ash plantations, including any site type that contained oak, Norway spruce, pre-thicket and thicket Sitka spruce and mature Scots pine plantations (Table 14). Canopy cover was negatively related to total spider

species richness, while the amount of coarse woody debris and site elevation were positively associated with total spider species richness (Table 14).

Table 14. Results of the general linear model on the impact of forest type, site environmental and location metrics, patch scale and landscape scale connectivity metrics on the total and forest-associated spider species richness. (se = standard error)

	Total spider species richness		Forest-associated spider species richness	
Metric	**Estimate (±se)**	**P**	**Estimate (±se)**	**P**
Intercept*	20.05 (3.12) [±]	<0.0001*	9.86 (1.41)[±]	<0.0001*
Forest type				
Ash – Semi natural	7.40 (2.74)	0.008*	1.47 (0.90)	0.10
Ash – Thicket plantation	3.86 (3.63)	0.29	-3.72 (1.08)	0.0008*
Norway spruce – Mature plantation	10.52 (3.02)	0.0007*	3.83 (0.98)	0.0002*
Norway spruce/Oak – Mature plantation	10.31 (3.96)	0.011*	3.10 (1.31)	0.019*
Norway spruce/Scots Pine – Mature plantation	11.59 (3.30)	0.0006*	3.53 (1.07)	0.0014*
Oak – Mature plantation	6.54 (3.23)	0.046*	2.42 (1.07)	0.025*
Oak – Semi natural	9.69 (2.66)	0.0004*	3.42 (0.87)	0.0002*
Scots pine – Mature plantation	10.37 (3.39)	0.0028*	1.64 (1.09)	0.13
Scots pine/Oak – Mature plantation	6.63 (3.31)	0.048*	1.91 (1.16)	0.10
Sitka spruce – Mature plantation	4.68 (2.75)	0.092	1.30 (0.92)	0.16
Sitka spruce – Mid rotation plantation	4.25 (2.92)	0.15	1.38 (0.98)	0.16
Sitka spruce – Pre-thicket plantation	14.86 (2.93)	<0.0001*	-2.52 (0.83)	0.003*
Sitka spruce – Thicket plantation	14.30 (2.93)	<0.0001*	-0.22 (0.93)	0.81
Sitka spruce/Lodgepole pine – Mature plantation	3.27 (3.02)	0.28	0.15 (1.00)	0.88
Yew – Semi natural	0.44 (3.99)	0.91	-2.36 (1.35)	0.082
Site environmental metrics				
% shrub cover (50 - 200cm high)	-0.04 (0.02)	0.077*		
% tree canopy cover	-0.15 (0.03)	<0.0001*		
% coarse woody debris cover (≥ 10cm diameter)	0.22 (0.07)	0.004*		
Site location metrics				
Elevation (m)	0.02 (0.01)	0.002*	4.33E-0.3 (1.91E-03)	0.025*
Longitude			0.56 (0.16)	0.0006*
Patch level metrics				
Area of semi-natural forest within patch (m^2)			6.33E-07 (2.17E-07)	0.004*
Area of young mixed plantation within patch (m^2)			-1.16E-04 (6.18E-05)	0.064
Patch landscape scale metrics				
Distance to nearest broadleaf plantation (km)	-4.36E-04 (2.42E-04)	0.075		

*Mature ash plantation was included in the intercept

Forest type, location and patch level metrics were important in the model of forest-associated spider species richness (Table 14). Forest-associated spider species richness was significantly lower in the pre-thicket (ash and Sitka spruce) sites, while any sites that contained either oak or Norway spruce had a significantly positive impact on the diversity of forest-associated spider species richness (Table 14). Forest-associated spider species richness was positively related to elevation, longitude and the area of semi-natural forest within site patch (Table 14).

Discussion

The biodiversity of spiders and birds in terms of species richness in this study was a function of forest type, vegetation structure and location, as well as both patch level and landscape scale metrics, all of which impact on the species richness of both spiders and birds in Irish forests. The differing impact of young plantation forest sites on total spider diversity and forest-associated spider diversity was related to the land use change between unplanted to planted habitats. The positive association between total spider species richness was due to the persistence of open habitat associated species in these habitats after planting, while the negative relationship between forest-associated spider species richness in these young plantations was due to insufficient time for colonisation of these specialists into this newly created habitat.

Lower levels of biodiversity are typically reported in plantation forests compared with natural woodlands (Barlow *et al.*, 2007; Brockerhoff *et al.*, 2008; Sweeney *et al.*, 2010a), and in this study that both broadleaf associated bird species richness and total spider species richness were significantly greater in the semi-natural broadleaf forest types included. Forest-associated spider species richness was significantly greater in oak semi-natural forests but not in ash semi-natural woodlands. In general, ash plantations had lower levels of spider diversity in comparison with other forest types. In contrast, any of the plantation forest types that had an oak component had higher diversity of broadleaf associated bird species and both total and forest-associated spider species richness. Several studies have recorded high species diversity associated with oak (Southwood *et al.*, 1982; Whittingham *et al.*, 2001). Positive impacts of Norway spruce within a plantation on spider diversity have also been recorded in Britain where five species groups of invertebrates all had higher species richness in exotic Norway spruce plantations compared to native oak plantations. The results of the current study suggest that in future afforestation, more oak should be planted (Quine and Humphrey, 2010). It should be noted that this analyses was conducted on species richness data and takes no account of the species composition of assemblages. For example, while ash plantations had relatively low levels of spider diversity, these plantations do support several spider species of conservation concern (Irwin *et al.*, 2013). The biodiversity of plantations of exotic conifers such as Sitka spruce would benefit from being mixed, where site conditions allow, with oak. As Norway spruce plantations had greater spider diversity in comparison to Sitka spruce, it may be advisable to plant more Norway spruce, where site conditions allow.

Recommendation 16: Where possible plantation forests should include an oak component where biodiversity conservation and enhancement is a priority.

Unlike broadleaf associated bird species richness and spider diversity, forest type was not an important variable in total bird species richness. As expected total bird species richness, as well as broadleaf associated bird species richness were strongly and positively related to the habitat complexity within forests (Wilson *et al.*, 2006; Gill and Fuller, 2007; Quine *et al.*, 2007; Sweeney *et al.*, 2010a; Sweeney *et al.*, 2010d). Irish forests are characterised by a lack of large diameter coarse woody debris (Sweeney *et al.*, 2010d) which provides habitat for significant biodiversity in forests (Ehnstrom, 2001; Humphrey *et al.*, 2002; Humphrey *et al.*, 2005; Lassauce *et al.*, 2011; Deady, 2013). The negative relationship between canopy cover and total spider species richness is most likely due to the reduction in habitat complexity due to shading which results in a reduction of all understorey structural levels and hence exclusion of many species which rely on a complex shrub and understorey vegetation structure (Oxbrough *et al.*, 2006a). It is apparent from these

results, and from other studies, that increasing the levels of coarse woody debris and promoting the amount of non-crop vegetation within plantations would benefit biodiversity at a site scale.

Recommendation 17: Coarse woody debris should be retained and non-crop tree regeneration encouraged in forest plantations.

The negative impact of elevation on the total species richness of birds and positive impacts on spider diversity are most probably related to separate factors. Much of the planted upland areas in Ireland are peatland and such habitat, although it supports habitat specialists, tend to have low bird diversity (Wilson *et al.*, 2012) and passerine birds are more abundant in plantations at lower altitudes and on better soils (Newton, 1986) and at higher latitudes (Avery and Leslie, 1990). The diversity of spiders in contrast to total bird species richness, was positively related to elevation. Many of the sites at low elevation included the ash pre-thicket and mature sites which were planted on better soil quality which tends to be better draining and hence have a drier forest floor. The sites at higher elevations tended to be on peatland dominated sites, which are wetter. Oxbrough *et al.* (2007) reported that wet flushes were indicators of spider diversity and therefore this positive association of spider species richness with elevation may be more related to provision of microhabitats that favour wet habitat associated species which boost the diversity of spiders in these wetter sites at higher elevations. There is a gradient of increasing bird species diversity from west to east across Europe, that appears to be not just a consequence of insularity, which supports our findings of positive relationship between biodiversity and longitude (Fuller *et al.*, 2007).

Both vegetation structure and forest type were related to the biodiversity within forest sites, as this has been shown in many other studies for a range of taxa including both birds and spiders (Oxbrough *et al.*, 2005; Oxbrough *et al.*, 2006a; Sweeney *et al.*, 2010b; Wilson *et al.*, 2010). However, this study, the first of its kind in Ireland, investigated the influence of landscape scale forest connectivity on forest biodiversity, and metrics including the vegetation structure and forest type were included as previous research has identified their importance in determining the biodiversity of Irish forests. The area of semi-natural woodland within a forest patch was the only patch or landscape scale metric which had a significant impact on spider diversity. This may be due to the fact that the source of such forest-associated spider species is our remaining patches of semi-natural forest. Smaller patches of habitat host relatively small populations of species which increases the probability of localised extinction (Ewers and Didham, 2006). And so, to maintain species diversity, it is important to protect Irelands semi-natural woodland which presently covers just over 1% of land area and is already highly fragmented (Forest Service, 2013).

Somewhat surprisingly, there was a negative impact of the area of broadleaf forest immediately adjacent to sites on the species richness of broadleaf associated birds. However, there was a positive impact on total bird species richness of the perimeter of such patches. Ireland is conspicuously lacking in forest specialist birds even when compared with the bird assemblages of forests in Britain, where there are far fewer forest specialists than on mainland Europe (Fuller *et al.*, 2007). This is probably due to a combination of Ireland's geographical location as an island at the western extremes of Europe, and the paucity of forest cover in Ireland over the last few centuries (O'Halloran and Kelly, 2012). Fragmentation and decreasing area of continuous forest habitat have been shown to negatively affect species richness of forest specialist birds (McCollin, 1993; Cerezo *et al.*, 2010). However, overall species richness is negatively correlated with distance from the forest edge due to a mixing of distinct fragment and matrix faunas at habitat edges, giving

rise to a zone of overlap with greater overall species richness (Magura, 2002). Therefore, this observed negative relationship of broadleaf forest immediately adjacent to sites on the species richness of broadleaf associated birds and a positive relationship of bird total species richness with the perimeter of these patches suggests a positive impact of edge effects on bird diversity. Certainly, there appears to be no negative impact of patch size on Irish broadleaf associated bird assemblages.

The probability of functional connectivity (PFC) landscape scale quantifies how well each site is connected to forest patches in its vicinity. The negative relationship between total bird species richness and PFC may be due to the fact that high PFC values in this model are generated with continuous cover of forestry, regardless of type and may again represent a lack of forest edge that appears to have a positive impact on bird diversity. The positive relationship of broadleaf associated bird species richness with PFC may be due to the different cost distance values applied in the PFC metric used in this model. The PFC used in the model of broadleaf associated birds species richness, attributed greater functional landscape connectivity when sites are located close to or adjacent to broadleaf forests, as these habitat had the lowest cost distance values. Therefore, the diversity of broadleaf associated birds appears to be positively related to the proximity of semi-natural woodland and broadleaved plantations within the vicinity and so functionally connected to our sites without necessarily being directly connected. This would therefore provide functional connectivity while also maintaining the positive edge effect that appears to have a positive impact on bird diversity. This is supported by the fact that there was also a significant negative impact of increasing distance between our study sites and the nearest patch of semi-natural forest on the diversity of broadleaf associated bird species richness.

Recommendation 18: Ensure connectivity to existing forest habitat, particularly to semi-natural woodland and areas of high conservation value for new and existing forests.

The contrasting relationships of landscape scale connectivity between species richness of broadleaf associated birds and forest-associated spiders most likely reflect differences in both the definition and the dispersal ability of these taxa groups. The only connectivity metric related to the diversity of forest-associated spiders was the positive association with the patch scale metric of the amount of semi-natural woodland within a site patch, whereas broadleaf associated bird diversity was associated with the patch scale edge effect and landscape scale functional connectivity. This difference between taxa is due to the fact that these spiders were true forest specialists whereas Ireland is lacking forest specialist birds (Fuller *et al.*, 2007). Forest specialists have been shown to be more negatively affected by habitat fragmentation and decreasing patch size compared to more generalist species, which have shown to be respond positively at forest edges (Dorp and Opdam, 1987; Niemelae, 2001).

Recommendation 19: Improve availability of data on non-forest habitats and landscape features such as roads, buildings and hedges, so that these can be accounted for in future assessments of connectivity.

Chapter 11

Impacts of large herbivores

Work Package 5

Miles Newman, Daniel L. Kelly and Fraser J G Mitchell

The aim of this study was to inform the development of effective strategies for the management of large herbivores in wooded areas in Ireland through the assessment of the impacts of large herbivores on semi-natural and plantation forests. Surveys of plant diversity and structure were conducted at exclosure/control sites in Killarney National Park, Wicklow Mountains National Park and Glenveagh National Park in 2010 and 2011, while plant, invertebrate and bird diversity surveys were conducted at paired plots in Killarney National Park. The main findings indicate that: low grazing levels should be maintained through culling and wild herbivore management; large-scale, long-term fencing of oak woodlands should be replaced by large herbivore management programmes, in order to ensure the conservation of diverse woodland ecosystems; more taxonomic groups should be included in biological assessment for management and conservation when stressors to ecosystem functioning, such as grazing, are present; and , assessments of grazing pressure should be conducted in areas of plantation forests, in order to maintain levels of grazing sufficient to promote plant community diversity.

Background

Management of large herbivore grazing is one of the main factors in temperate forests and spans the range from no control (high grazing pressure), to controlled grazing (by fence and/or deer culling or via woodland grazing plans), to zero grazing in fenced plots (Worrell and Long, 2010). It is common practice in many environmentally and legally protected woodland areas to cull or exclude (through fencing) wild and domestic herbivores in order to maintain desired ecosystems or promote conservation. Fencing as a conservation measure is not a new concept and is one which has developed the dual purposes of enclosing and excluding certain species (Figure 28) (Hardin, 1968; Hayward and Kerley, 2009). Fencing for biodiversity conservation may ultimately be an acknowledgement of our failure to successfully embrace the maintenance of biodiversity in tandem with other land uses (Hayward and Kerley, 2009). Fencing and culling programmes are often costly both ecologically and economically (Pérez and Pacheco, 2006; Hayward and Kerley, 2009). The complete exclusion of wild herbivores may not be desired, however, as they are a natural element of Irish woodlands (Mitchell, 2005) and optimal culling quotas may be difficult to estimate due to inaccuracies in survey methods (Swanson *et al.*, 2008; Koh *et al.*, 2010). Large herbivore grazing has been highlighted as a conservation management issue requiring attention in Irish temperate woodlands (NPWS, 2005a, b, 2008; Purser *et al.*, 2009). There is now a major need to investigate the impacts of grazing levels in temperate woodlands (Hester *et al.*, 2000; Pellerin *et al.*, 2010; Perrin *et al.*, 2011), in order that targeted grazing management prescriptions can be developed.



Figure 28. Forest exclosure showing dense holly regeneration and community change within the exclosure, 39 years since fencing in Killarney National Park. Photo by Miles Newman

Objectives

The aim of this study was to investigate the impact of different grazing levels on plant communities, woodland structure, tree regeneration, and relationships among different taxonomic groups. This included the quantification of the long-term changes in plant communities within deer exclosures, assessment of the effects of exclosures on multiple taxa, and the impact of grazing on the vegetation community composition in plantation forest.

The specific objective of this work was:

- To devise protocols for appropriate management of grazing animals in Irish woodland areas, to include both semi-natural woodlands and plantation forests.

Methodology

The maintenance of long term deer exclosures over several decades within semi natural woodlands provides an ideal resource for addressing the central research objective. This study was thus conducted within semi-natural oak woodlands in three National Parks in Ireland: Killarney National Park (KNP), Wicklow Mountains National Park (WMNP), and Glenveagh National Park (GNP) (Figure 29).

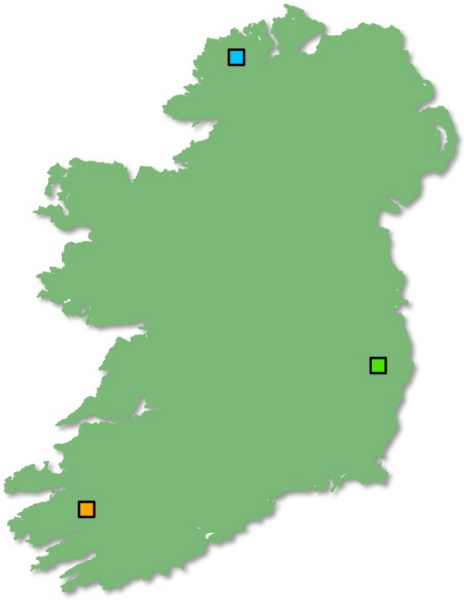


Figure 29. Study sites at Killarney National Park (■), Wicklow Mountains National Park (■) and Glenveagh National Park (■).

In addition to national designations, these woodlands are protected as Special Areas of Conservation (SAC) under the European Union's Habitats Directive (92/43/EEC) as they contain representative areas of old sessile oak with holly (*Ilex aquifolium*) and hard fern (*Blechnum spicant*) (Annex 1 EU code 91A0). Additional data for plantation forests were collected as part of Work Package 1 from a total of 173 forest plots distributed throughout Ireland as described in chapters 1-3.

Impacts of herbivore grazing

The impact of grazing on woodland ground flora was investigated using a paired exclosure and control experiment using existing sets of exclosures (established between 1969 and 1988) in KNP, WMNP and GNP. Plant species cover abundance (%) was recorded in ten $1m^2$ relevés within each plot (ranging from 400-11,000 m^2) with the cover (%) of vegetation within structural height classes (0-5cm, >5-25cm, >25-50cm, >50-100cm, and >100-200cm) also being recorded. Numbers of individual seedlings (<25cm in height) and saplings (≥ 25cm in height, but with a DBH of less than 3.2cm at 1.3m) of each species, and their cover, were recorded within the $1m^2$ relevés. Plant life-forms were categorised, by summing species cover values, into: graminoids – grasses, rushes, and sedges; forbs – herbs excluding graminoids and ferns; ferns; shrubs – woody plant that are not trees (Stace, 2010) or climbers; bramble – *Rubus fruticosus* agg.; climbers – *Lonicera periclymenum* and *Hedera helix*; and bryophytes – mosses and liverworts.

A total of 18 paired plots were surveyed (nine exclosed from deer grazing and nine open to grazing), with five pairs in KNP, two in WMNP, and two in GNP. Grazing impacts were classified into three categories (absent, low, and medium-high) using the parameters set out in the online Woodland Grazing Toolbox (Forestry Commission Scotland, 2013). This toolbox qualitatively categorises the recent grazing pressure (within 12 months) using the criteria of ground disturbance, bark stripping, swards composition (grass, rush, and low herb vegetation), preferentially browsed species, tree seedlings and saplings, epicormic shoots, and basal shoots on trees. Non-metric multidimentional scaling (NMS) ordination and PERMANOVA were used for community level analysis, while paired t-tests and General Linear Models (GLM) were used to test for differences among life-forms, structural classes, and tree regeneration, at different grazing levels.

Long-term exclosure monitoring

The effects of wild herbivore removal in the woodland ecosystem were studied using an existing network of seven large-herbivore exclosures (two in KNP, three in WMNP, and two in GNP) surveyed for up to 41 years. Ground flora species cover-abundance (%) was recorded at varying intervals across this time-frame. Baseline and subsequent species cover abundance records were transformed into per cent scale. In order to compare changes in diversity through time, the data points were divided into three *a-priori* age groups. These groups were delimited by gaps in the 'time since fencing' scale. The three natural groups in the data represent the time since fencing groups of: time 1 (0-12 years, n = 16), time 2 (16-28 years, n=8), and time 3 (32-41 years, n=9). The multi-site dissimilarity measures (based on Baselga, 2010) of Sørensen beta ($\beta_{SØR}$ - species turnover and nestedness – subsets of the same species occurring among sites) was used to highlight dissimilarities in species turnover and nestedness among time groups. Beta diversity was used as a proxy for homogeneity as it measures the degree of varying types of similarity on an index between 0 (complete similarity) and 1 (complete dissimilarity). Ordination and PERMANOVA were used to assess community change, ANOVA and post-hoc comparison tests were used for beta diversity comparisons, and a similarity percentage routine (SIMPER) was used to assess species abundance differences among the time groups.

Multi-taxonomic impacts of herbivore exclusion

The effect of herbivore exclusion on birds, ground-dwelling spiders and vegetation was assessed in grazed and ungrazed sites in Killarney National Park. Three exclosures (five years old) and three grazed sites were surveyed for vegetation and ground-dwelling spider biodiversity, and two of these exclosures and two grazed sites for bird richness, between May and August 2011. Vegetation variables were recorded at plot ($10m^2$) or sub-plot ($1m^2$) level. Ground-dwelling spiders were collected using transects of six pitfall traps, over a total of 84 trapping days. Birds were surveyed using point counts, with a total of eight point counts (4 in exclosures and 4 in grazed area), surveyed once in the early breeding season and once later. Multivariate Procrustes analysis was used to assess similarity between ground spider and vegetation community composition. Bird data were insufficient for this analysis. Surrogacy between species richness, and relative abundance of taxonomic groups was investigated through correlation analysis.

Impacts of grazing in forest plantations

A separate study of the impacts of grazing animals on forest structure and ground vegetation composition in plantation forests was carried out in at a number of study sites in Work Package 1 including the chronosequence and tracking plots (n = 53) in chapter 1, the road-width plots (n = 96) in chapter 2 and the FutMon plots (n = 24) in chapter 3. Grazing intensity was recorded through categorisation (none, low, moderate, heavy). Grazing intensity was then used as a factor in ANOVAs and as an environmental variable in ordination analysis in chapters 1 and 2; the data collected in chapter 3 were not analysed.

Results

Impacts of herbivore grazing

Plant community composition differed among grazing levels; it was significantly different between absent and medium-high levels (Figure 30). Plots where grazing was absent showed increased bramble (*Rubus fruticosus* agg.) cover, climber cover, vegetation cover between 50-200cm high, and tree sapling cover and number, with a decreased graminoid species richness, bryophyte cover and richness, total plant species richness and tree seedling numbers.

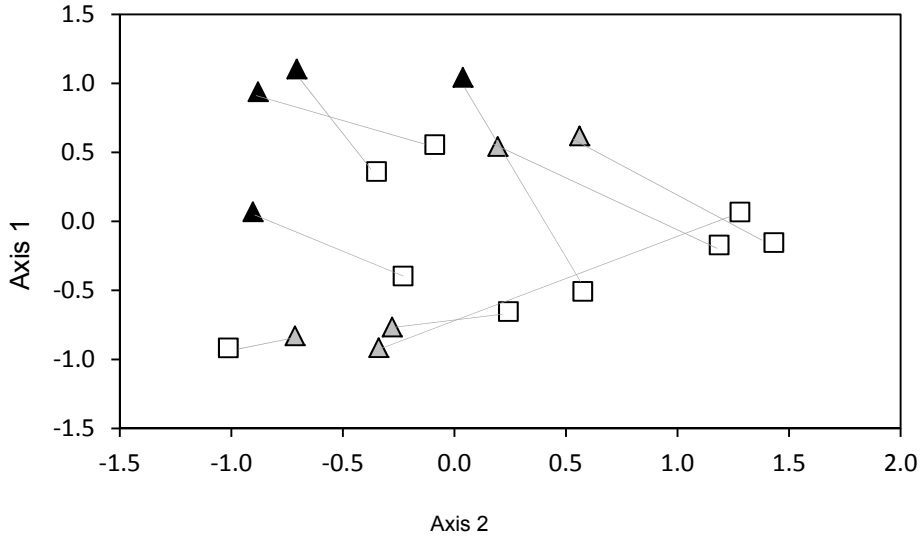


Figure 30. NMS ordination of plots showing grazing levels. □ = ungrazed plots, △ = low grazing plots, ▲ = medium-high grazing plots. Vectors (solid grey lines) show pairing of plots. Final stress was 12.78, Monte Carlo Randomisation test results for axis 1 and 2 were 0.036 and 0.004, and instability was ≤0.00001. The r^2 of Axis 1 and 2 was 39.1% and 45.5%, respectively.

At sites where low grazing levels were observed, plots exhibited greater cover in vegetation between 100-200cm, greater tree seedling cover and number of individuals, and greater sapling cover, species richness, and numbers of individuals, compared with either absent or medium-high grazing levels. Sites with medium-high grazing level showed greater graminoid species richness, forb cover, bryophyte cover and species richness, lower climber cover, vegetation cover between 50-200cm, tree sapling cover, species richness and number of individuals, compared with other grazing levels.

Long-term exclosure monitoring

There was a significant effect of time since fencing ($P < 0.001$), site ($P < 0.001$), and the interaction between TSF and site ($P < 0.001$) on community composition. When species presence/absence was tested instead of abundance the effect of TSF and site were significant ($P = 0.01$ and $P = 0.001$, respectively), but the interaction was not significant. The beta diversity results show a similar pattern to the ordination, where an increase in community composition similarity is seen through time (Figure 31). Sørensen's beta diversity (β_{SOR}) shows that all three time groups are significantly different from each other, with an increase in similarity among plots with time. The abundance of *Luzula sylvatica* and *Pteridium aquilinum* were identified as species contributing to the dissimilarity between each time group comparison, while the abundance of *Luzula sylvatica* and *Vaccinium myrtillus* contributed greatest to group dissimilarities in each comparison.

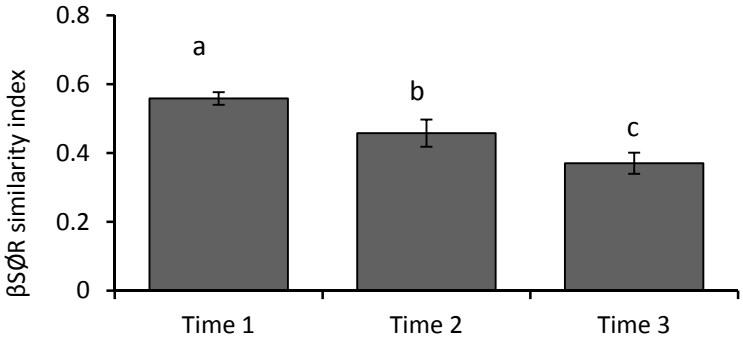


Figure 31. Comparison of Sørensen's beta diversity (βSOR) among time groups, where 0=complete similarity and 1=complete dissimilarity. Significant difference (Tukey post-hoc, P < 0.05) between lowercase letters. Time 1=0-12 (n = 16) years, time 2=16-28 years (n = 8), and time 3=32-41 years (n = 9).

Multi-taxonomic impacts of herbivore exclusion

A significant correlation was observed in the symmetric rotation of vegetation community cover and ground spider community relative abundance, within exclosures ($r = 65.7\%$, $P = 0.0356$). In grazed sites however, the correlation in the symmetric Procrustes rotation was not significant. Within exclosures, significant ($P < 0.05$) positive correlations were found between woodland specialist spider species richness and vascular plant cover ($r = 0.69$), forb cover ($r = 0.74$), and total vegetation cover (vascular plants and bryophytes) ($r = 0.73$). The relative abundance of woodland specialist spiders was significantly ($P < 0.05$) positively correlated with moss species richness ($r = 0.74$) and bryophyte cover ($r = 0.79$), but negatively correlated with fern species richness ($r = -0.75$). Within exclosures, bird species richness was significantly ($P < 0.05$) negatively correlated with the cover of tree seedlings greater than 1 year old ($r = -0.97$).

Impacts of grazing in forest plantations

Plantation sites with high or very high grazing intensities were not recorded, however, analysis of the data from the WP1 sites found that bramble cover was significantly lower at medium grazing intensity than where grazing was absent ($P = 0.022$). Conversely, grass & sedge cover was significantly lower where grazing was absent, as compared to low ($P < 0.001$) and medium ($P < 0.001$) grazing intensities. There were no significant differences for the other layers. Analysis of the vascular plant species composition at the different grazing levels revealed that the composition at sites where grazing was absent was significantly different to those with low ($P < 0.009$) or medium ($P = 0.002$) grazing intensities. The vegetation at these ungrazed sites was often dominated by bramble. The monitoring of afforestation sites revealed that grazing intensity was significantly related to both bryophyte species richness and plant community composition. This effect was most strongly expressed in improved grassland habitats (Chapter 1). Grazing was also found to be important in determining plant community composition along forest roads, with its presence preventing dominance by shrubs (Chapter 2).

Discussion

Impacts of herbivore grazing

This research has illustrated the effects of different levels of grazing impact on the vegetation communities of semi-natural oak woodlands. The overlap of absent and low grazing level plots in the ordination, combined with the results of the PERMANOVA, suggests species composition is influenced by the gradient of grazing levels. Other studies have also shown that deer grazing can change woodland plant communities (Putman *et al.*, 1989), with increased grazing shifting dominance from woodland specialists to generalists (Pellerin *et al.*, 2010) (Pellerin *et al.*, 2010), increasing ruderal plants (McEvoy *et al.*, 2006), and generally simplifying the woodland ecosystem (Stockton *et al.*, 2005) though this was not investigated in the current study. Concurrent with previous studies, *Rubus fruticosus* agg. was negatively affected by large herbivore grazing (Kirby and Thomas, 2000; Kirby, 2001; McEvoy *et al.*, 2006; Pellerin *et al.*, 2010). However, zero grazing may also lead to a loss of plant diversity; in particular, competitive generalist species may be favoured and woodland specialists may be lost (Perrin *et al.*, 2011). Previous studies have focused on the vegetation differences between grazing and grazing exclusions, however, the findings of this study allow us to identify the likely vegetation responses of not only presence or absence but also at different intensities when grazers are present.

Recommendation 20: Maintain low grazing levels through management of wild herbivores.

#Long-term exclosure monitoring

Using beta diversity as a proxy for homogeneity, a clear increase in homogenisation in vegetation composition is seen with increased time since fencing. This suggests that woodland vegetation is becoming more homogenous over time, when large herbivores are excluded, and confirms the findings of previous studies on some of the Killarney research sites (Kelly, 2000; Perrin *et al.*, 2011). The proportion of species shared among plots within a time group (β_{SIM}), decreased significantly between 16 and 41 years since fencing, compared to recently fenced plots (0-12 years). This has implications for biodiversity conservation as the results indicate a potential reduction in the number of plant species found in areas where large herbivore grazing has been removed for long periods (e.g. > 12 years).

Of the four key species identified as the greatest contributors to species abundance difference between the time groups, two species (*Luzula sylvatica* and *Pteridium aquilinum)* contribute to the differences between all the time group comparisons. *L. sylvatica* can become dominant following cessation of grazing (Mitchell, 1990), but may not respond to large herbivore removal in the same way as other graminoids. Reductions in graminoids were noted after 16 years in woodland exclosures in Wisconsin (Rooney, 2009) and after 22 years in England (Putman *et al.*, 1989). The abundance of the fern, *Pteridium aquilinum,* showed an initial decline in the first two time periods followed by a slight increase in the third time period. Ferns may be negatively impacted by large herbivore exclusion as a declining trend in overall fern abundance was noted after 32 years of deer exclusion by Perrin *et al*. (2011), while the loss of the locally frequent (Page, 1997) Tunbridge filmy-fern (*Hymenophyllum tunbrigense*) was recorded after 26 years of herbivore exclusion in Killarney (Kelly, 2000). This may be a result of increased competition from recovering browse sensitive plant species. Complete removal of large herbivores is undesirable other than for short-term objectives to be completed (Hester *et al.*, 2000), as large herbivores perform integral ecosystem functions, such as disturbance and driving succession through selective herbivory, within woodlands (e.g. Vera, 2000). Short-term fencing, directed at increasing tree regeneration or biodiversity maintenance for example, could still be used with the knowledge that community homogenisation may take place after 12 years. The findings of this research, combined with those of others, suggest a general homogenisation of temperate oak woodlands will occur with long-term large herbivore removal.

Recommendation 21: Replace large-scale, long-term fencing of broadleaved woodlands with large herbivore management programmes in order to ensure the conservation of diverse woodland ecosystems.

Multi-taxonomic impacts of herbivore exclusion

The community composition results imply that the effects of exclosures on vegetation and ground-dwelling spiders are comparable. This may be because of the reduced complexity of the ecosystem when large herbivores are removed. Large herbivore grazing may cause a greater diversity in vegetation composition, which has been shown to have knock-on effects for invertebrate communities (Takada *et al.*, 2008). Although a range of significant positive and negative correlations were found between ground-dwelling spider and vegetation variables, no significant relationship was noted between the same two taxa variables in both exclosures and grazed areas. This study has thus shown that the relationships between taxa may be altered, and sometimes reversed, when comparing large herbivore exclosures with grazed areas. Consequently care must be taken when using surrogates in habitats subjected to different levels of grazing.
Results from this research have revealed the complexity of the relationships between birds, vegetation, and spiders. However, it should be noted that explaining these complex interactions is not always possible, as direct and meaningful cause and effect relationships are not always present (Landres *et al.*, 1988).

Recommendation 22: Increase the number of taxonomic groups used in biological assessment of management and conservations objectives where stressors to ecosystem functioning, such as grazing, are present.

Impacts of grazing in forest plantations

As with the findings of the semi-natural woodlands section, bramble (*R. fruticosus* agg.) cover was significantly higher in areas of lower grazing pressure, most likely due to the palatability of the species (Putman, 1996). Again, as with findings in semi-natural woodlands, graminoid cover was higher in grazed areas. This reflects the 'grazing lawn' which is often created when large herbivore selectively graze and browse out preferential species, thus allowing graminoids to dominate (Putman, 1996). Whilst complete absence of grazing may be undesirable for forest biodiversity, there is little information on the threshold densities for plantations above which crop damage becomes significant and below which the impact is acceptable (Putman, 1996). Further research is required in this area. The plantation forest type was identified as being a more important factor for both ground flora structure and vegetation community composition, than grazing pressure. The dominant influence of forest type in determining vegetation structure and composition has previously been identified in plantation forests (Coote *et al.*, 2012). In conifer plantations, particularly spruce (*Picea* spp.) plantations, the shading influence of the tree crop results in an impoverished ground flora once the canopy closes, concentrating grazing in open areas, such as road corridors; the impacts of grazing in road corridors are discussed in chapter 2 on the biodiversity of forest roads.

Recommendation 23: Conduct assessment of grazing pressure in plantation forests to maintain levels of grazing sufficient to promote plant community diversity.

It is concluded that long-term fencing of woodlands may not be desirable as exclusion of large herbivores has a negative impact on community composition. General homogenisation of temperate oak woodlands will occur with total large herbivore removal. The relationships among ground flora, ground-dwelling spiders, and birds are stronger in exclosures than in grazed areas, likely due to the varying degrees of disturbance encountered in grazed areas. The vegetation community responds to grazing pressure in similar ways in both semi-natural Oak woods and commercial plantation forests.

Conclusions

The study of herbivore grazing in Irish forests concluded that long-term fencing of woodlands may not be desirable as exclusion of large herbivores has a negative impact on community composition. General homogenisation of temperate oak woodlands will occur with total large herbivore removal. The relationships among ground flora, ground-dwelling spiders, and birds are stronger in exclosures then in grazed areas, likely due to the varying degrees of disturbance encountered in grazed areas. The vegetation community responds to grazing pressure in similar ways in both semi-natural Oak woods and commercial plantation forests. The use of non-permanent moveable deer fences would allow targeted management of priority area for large herbivore removal. These fences could then be moved when desired management aims (such as tree regeneration or increased cover of vegetation at ground-level) are achieved

Chapter 12

Climate change impacts

Nova Sharkey, David Bourke and Mike Jones
Work Package 6

Global climate is rapidly changing and, while many studies have investigated the potential impacts of climate change on the distribution of woodland species in the wider European context, few have focused on the impacts of climate change on the distribution of woodland species in Ireland. This study evaluated the potential impact of climate change on the distribution of species characteristic of Irish woodlands and assessed the implications for conservation. A number of species distribution modelling techniques were applied to atlas data for 104 vascular plant, bryophyte, bird and butterfly specie with baseline (1961-1990) and projected future (2031-2060) scenarios. Of the 104 species, 45 modelled well in at least 6 of the 8 models. Under a scenario of unlimited dispersal, almost 50% of species were projected to lose climate space, with the largest range contraction (-47.7%) projected for the Ringlet butterfly. All other species were projected to have increased climate space. When a scenario of limited long-range dispersal was considered, all species were projected to lose at least some of their current range, from a decrease of 52.4% for Thin-spiked wood-sedge to a decrease of 10.9% for St. Patrick's cabbage. These results have significant implications for the future of woodlands in Ireland. Conservation management plans for these habitats require greater focus on predicted climate change impacts in order to ensure the long-term survival of forest ecosystems.

Background

In its infancy the sole objective of forest management was the production of a high timber yield. Since the 1970s an increasing awareness of the role of forests in the provision of ecosystem services and their importance for biodiversity conservation has led to more ecosystem-based forest management that takes account of the multifunctionality of forests (Fürst *et al.*, 2007). Forest management practice and policy is continually evolving to bridge conflicting management objectives and the United Nations Forum on Forests (UNFF) defines sustainable forest management as "*a dynamic and evolving concept aiming to maintain and enhance the economic, social and environmental value of all types of forests, for the benefit of present and future generations*". Over the coming century the Irish climate is likely to become drier and hotter (up to 4˚C) with increased risk of extreme weather events such as storms and floods which potentially threatens both the economic and ecological benefits that forests provide to people (Lindner *et al.*, 2010; Hanewinkel *et al.*, 2013; Liu *et al.*, 2013). The design of adaptive management strategies that incorporate climate change scenarios is therefore essential to the sustainable management of forests to protect biodiversity and ecosystem services into the future (Beever and Belant, 2012; Johnston and Hesseln, 2012; Day and Pérez, 2013). Thus there exists a continuous challenge to provide scientific knowledge on climate change impacts in support of forest management and policy (Lindner *et al.*, 2010; Liu *et al.*, 2013; Mori *et al.*, 2013).

Changes in climate are projected to have significant impacts on biodiversity and may lead to widespread changes in species distribution and community composition in many parts of the world (Root *et al.*, 2005; Parmesan, 2007). While forest ecosystems are particularly sensitive to climate change due to the long life span of trees forests have proved resilient to past changes in climate (De Frenne *et al.*, 2013). However today's fragmented and degraded forest ecosystems are more vulnerable, particularly in landscapes where plantation forests dominate and climate change is a major threat to biodiversity over the coming century (Noss, 2001; Heller and Zavaleta, 2009). When faced with a change in their environment beyond their normal tolerance, such as climate change, species and populations can respond in one of two ways, either by adaptive evolution so that the new climate is tolerated, or movement to another area with a more suitable climate (Beever and Belant, 2012). Should either adaptive evolution or geographic dispersal prove too slow to keep up with a shifting climate, and should they lack sufficient phenotypic plasticity to tolerate changing environmental conditions, these species will face local extinction (Pearson, 2006; Thomas *et al.*, 2006).

The projected climate change will impact on forest biodiversity at species level and will impact on the interactions between species, and hence on forest ecosystem dynamics (Lindner *et al.*, 2010). Climate change impacts will vary spatially and be related to local forest management practices including the species of tree selected for planting (Yu *et al.*, 2013). While the effects of climate change on woodland species in Ireland have not previously been studied directly, adverse effects of climate change on species distributions and abundances have been reported in British woodlands (Thomas *et al.*, 2004; Kirby *et al.*, 2005) where the majority of native broadleaved trees are predicted to become unsuitable for commercial timber production (Broadmeadow *et al.*, 2005). The effects of climate change on forests and forest biodiversity must be evaluated and potential vulnerability identified locally in order to determine how best to respond to this threat and minimise the negative impacts of climate change (Spittlehouse, 2005; Ogden and Innes, 2007).

The most commonly used methods to assess the impacts of projected climate change on the distribution of biodiversity are species distribution models (SDMs), also known as Ecological Niche Models (ENM), which statistically relate current species distributions to climatic variables, enabling projections of distributions under future climate change scenarios (Figure 32) (Pearson and Dawson, 2003; Thuiller *et al.*, 2005; Beever and Belant, 2012; Crimmins *et al.*, 2013). Species distribution models have been used extensively to determine the potential impact of climate on both species and communities, and can be important in informing conservation strategies (Midgely *et al.*, 2003; Carroll, 2010). Because we have little knowledge of the physiological factors that govern the distribution of many species, SDMs can provide valuable insight into the potential impacts of climate change on these species using broader climatic and environmental variables as proxies for a variety of eco-physiological processes.

Objective

Risk associated with climate change can be reduced through appropriate forest management which relies on current, local knowledge on the impacts of climate change. The specific objective of this study was to:

- Synthesise predictions of the future species composition of Irish forests under different climate change scenarios.

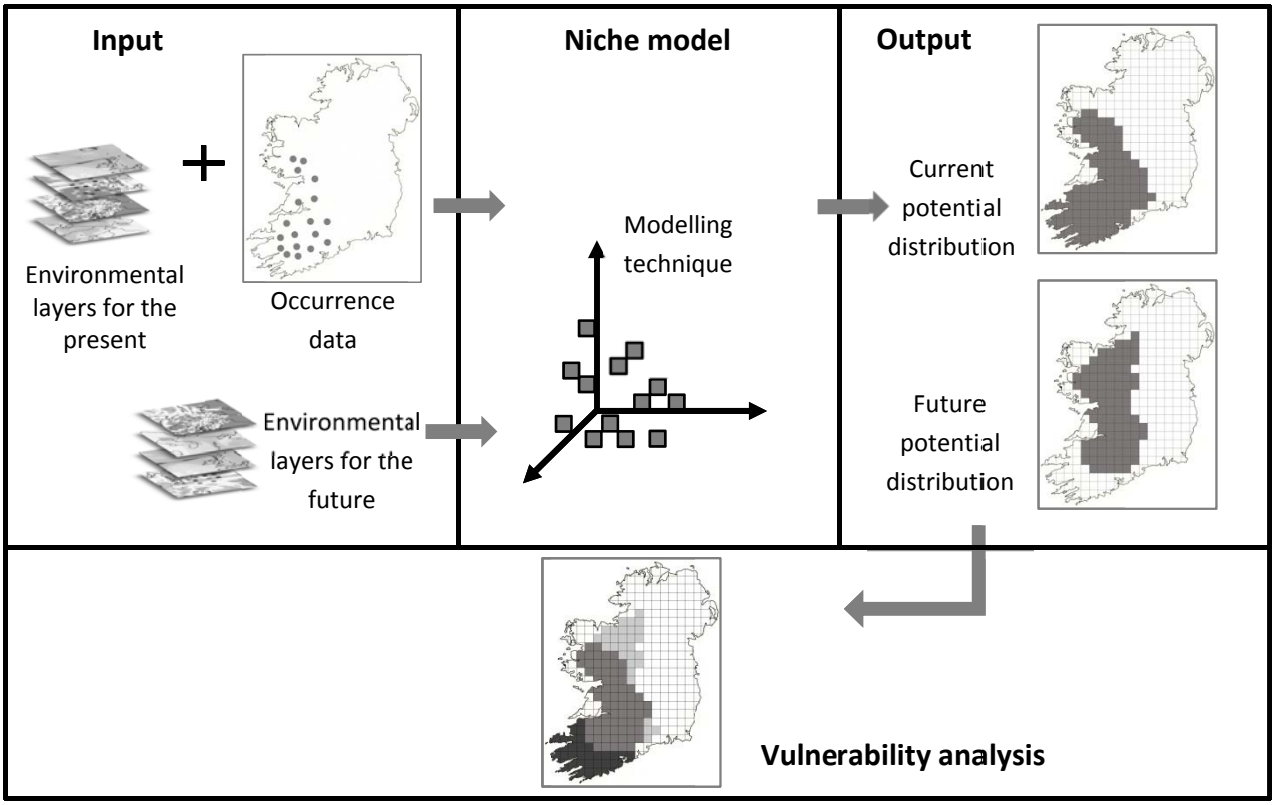


Figure 32. Diagrammatic representation of the Species Distribution Modelling process (After Martínez-Meyer, 2012).

Methodology

This study was undertaken across the whole island of Ireland. Plant and animal species from four broad taxonomic groups (vascular plants, bryophytes, butterflies and birds) which were either confined to, or characteristic of, a variety of semi-natural woodlands were included in the study. Environmental variables used were minimum annual monthly temperature, continentality index (difference between mean maximum and minimum temperatures), net annual precipitation (all based on either baseline climate data (1961-1990) or projected future climate data (HadCM3 GCM; 2031-2060), (both at 10 km^2 resolution) and elevation (at 10 km^2 resolution). The climate change data used in the current study include the mean values of the IPCC A2 and B2 scenario outputs (IPCC, 2000), obtained by statistically downscaled outputs from the HadCM3 Global Climate Model (Sweeney and Fealy, 2003). The B2 family of scenarios places more emphasis on the projected finding of local solutions to economic, social and environmental sustainability, whereas the A2 family of scenarios projects a greater increase in global carbon dioxide (CO_2) emissions associated with different development trajectory assumptions (IPCC, 2000). Detailed information on variable selection and species distribution and climate change data sources can be found in Sharkey *et al.* (2013).

The distributions (current and future projections) of 104 target species were predicted using eight modelling techniques available in the BIOMOD library (Thuiller *et al.*, 2009), run and calibrated within the R environment using Generalized Linear Models (GLM), Random Forests (RF), Generalized Boosting Models (GBM), Artificial Neural Networks (ANN), Classification Tree Analysis (CTA), Multiple Adaptive Regression Splines (MARS), Flexible Discriminant Analysis (FDA), and Generalized Additive Models (GAM) (Thuiller *et al.*, 2009). A split-sample cross-validation procedure was used to evaluate the models (i.e. different data portions are used to construct and to evaluate the models). Models were calibrated for each species on 80%

of the data and then evaluated on the remaining 20% using the area under the curve (AUC) of the receiver operating characteristic (ROC), and the True Skill Statistic (TSS) (Allouche *et al.*, 2006). Species were considered to have modelled well if they had an AUC > 0.90 , a TSS > 0.60, or if AUC > 0.7 and TSS > 0.4.

Differences between range changes in different grouping categories were compared and analysed for significance with Kruskal-Wallis tests and Mann-Whitney U tests. Corrections for multiple comparisons, where necessary, were made using the Dunn-Šidák correction. The simulated current and future distributions of the species were compared for the models using both climate and topography variables and the percentage decrease or increase in range for each species calculated. Central to possible range changes of species under changing climate is the ability of those species to colonise new potentially suitable areas. This mostly depends on the dispersal ability of a species.

Detailed dispersal distances are not available for most plant and animal species and so two extreme scenarios were investigated:

1. Unlimited dispersal: where the entire projected future range of the species is taken to be the actual future distribution.
2. No long-range dispersal: where the future distribution results solely from the overlap between current and projected future range of the species.

Results

The models generally performed well, with mean AUC values of greater than 0.7 for all models except for CTA. Of the 104 species modelled, 39 did not model 'well' according to the AUC/TSS scores. Using these parameters, twenty-one species modelled well in all 8 of the models used in BIOMOD and 37 modelled well in all except CTA. When corrected for multiple comparisons, median AUC scores were significantly higher ($P < 0.05$) for vascular plants than bryophytes in five of the models (GBM, GLM, MARS, FDA and RF), while median TSS scores were significantly higher ($P < 0.05$) in four models (GLM, MARS, FDA and RF).

Projected range changes

Model performances in BIOMOD were in broad agreement and so ensemble forecasting results were chosen to produce data on range changes. Range changes (for both the unlimited dispersal and no long-range dispersal scenarios) were calculated for those species that performed well in at least six of the eight BIOMOD models. A mean increase in range of 65.9% by 2055 was projected under the unlimited dispersal scenario, while a mean reduction in range (-26.9%) was projected under the no long-range dispersal scenario (Table 15).

Bryophytes showed a significantly greater increase in range than vascular plants under the unlimited dispersal scenario (145.9% ± 149.0 vs 17.4% ± 109.3; $p<0.05$, Table 15). Almost 50% of species which modelled well were projected to lose climate space, with the largest range contraction projected for the butterfly *Aphantopus hyperantus* (Ringlet) (-47.7%). The remainder of species were projected to have increased climate space. The rare bryophyte *Telaranea nematodes* (occurring in only 17 grid squares) was projected to have an increase in suitable climate space of over 400%, while *Sphagnum quinquefarium*, a moss of acidophilous woodland, which is currently more widespread (Atherton *et al.*, 2010) was also projected to have increased climate space of over 450%.

Table 15. Mean percentage range changes for plant species (± standard deviation), under unlimited dispersal and no long-range dispersal, for plant type, biome and eastern limit categories.

	Number of species	Range change (Unlimited dispersal)	Range change (No long-range dispersal)
Bryophyte	11	145.9 ± 149.0	-23.5 ± 6.5
Vascular plant	27	17.4 ± 109.3	-28.4 ± 9.9
Biome			
Wide-boreal	1	-25.2	-29.0
Boreal-montane	2	218.8 ± 330.9	-19.1 ± 8.3
Boreo-temperate	14	29.1 ± 57.2	-27.0 ± 6.6
Temperate	17	34.5 ± 134.8	-29.5 ± 11.0
Southern-temperate	4	167.3 ± 175.0	-19.5 ± 5.8
Eastern limit			
Hyperoceanic	5	140.0 ± 164.3	-19.1 ± 5.7
Oceanic	4	52.7 ± 57.2	-26.3 ± 8.6
Suboceanic	10	89.8 ± 139.0	-29.1 ± 12.7
European	10	-11.9 ± 25.2	-30.1 ± 6.9
Eurosiberian	6	72.3 ± 225.5	-27.4 ± 7.8
Eurasian	1	-4.8	-14.8
Circumpolar	2	-22.0 ± 4.6	-26.6 ± 3.4
All species		**54.6**	**-27.0**

Of the butterflies, only *Aphantopus hyperantus* was projected to lose climate space under the unlimited dispersal scenario. *Gonepteryx rhamni* (Brimstone) was projected to have a potential range increase (unlimited dispersion) of almost 90.0%, but a limited dispersal range change of -27.0%. No significant differences were found between range changes for the biome or eastern limit categories of plant species, although some trends were evident. Species in the boreal-montane and southern temperate biomes show the greatest increase in range under unlimited dispersal, while species in the wide-boreal, boreo-temperate and temperate show the greatest losses in suitable climate space (Table 15).

While there was a general trend for an increase in suitable climate space under the unlimited dispersal scenario, this was not the case for those species in the European, Eurasian and Circumpolar eastern limit categories, all of which showed a mean decrease (Table 15). The hyperoceanic category was projected to gain the largest amount of suitable climate space, at 140.0% ± 164.3. Under the no long-range dispersal scenario, all categories were projected to have decreased ranges, with all but the hyperoceanic and Eurasian eastern limit categories projected to have large (>25%) mean decreases.

When the no long-range dispersal scenario was considered, no species were projected to maintain their entire current range and all species were projected to lose suitable climate space. *Saxifraga spathularis* was the species with the smallest projected decrease in range, at 10%, while *Carex strigosa*, *Aphantopus hyperantus*, *Lysimachia nemorum* and *Cardamine flexuosa* were all projected to lose more than 40% of their current climate space. Under this scenario, bryophytes were projected to show a mean range loss of 23.5% ± 6.5 compared with a loss of 28.4% ± 9.9 for vascular plants (Table 15).

Patterns of distribution change

Changes in distribution fell into one of two main groups. In the first, suitable climate space was lost in the centre of the country, and ranges were either maintained or expanded around the periphery. The other pattern of range shift was for a loss of climate space in the west and south, with ranges maintained or gained in the north and east. This was linked to eastern limit categories, with hyperoceanic, oceanic and suboceanic eastern limit categories showing a decrease in the centre of the country and an increase near the coasts (e.g. *Dryopteris aemula* (hyperoceanic), *Saccogyna viticulosa* (oceanic), and *Lysimachia nemorum* (sub-oceanic); Figure 33), reflecting the projected change in continentality index (i.e. an increase in the difference between mean maximum and minimum temperatures; Figure 34). Species in the Eurosiberian, Eurasian and circumpolar categories showed a trend for a range shift to the north east (e.g. *Betula pubescens* (Eurosiberian), *Glechoma hederacea* (Eurasian), and *Phalaris arundinacea* (circumpolar); Figure 35), which is similar to the projected change in minimum February temperature (Figure 36). Distribution maps for all other species that modelled well are presented in Sharkey *et al.* (2013).

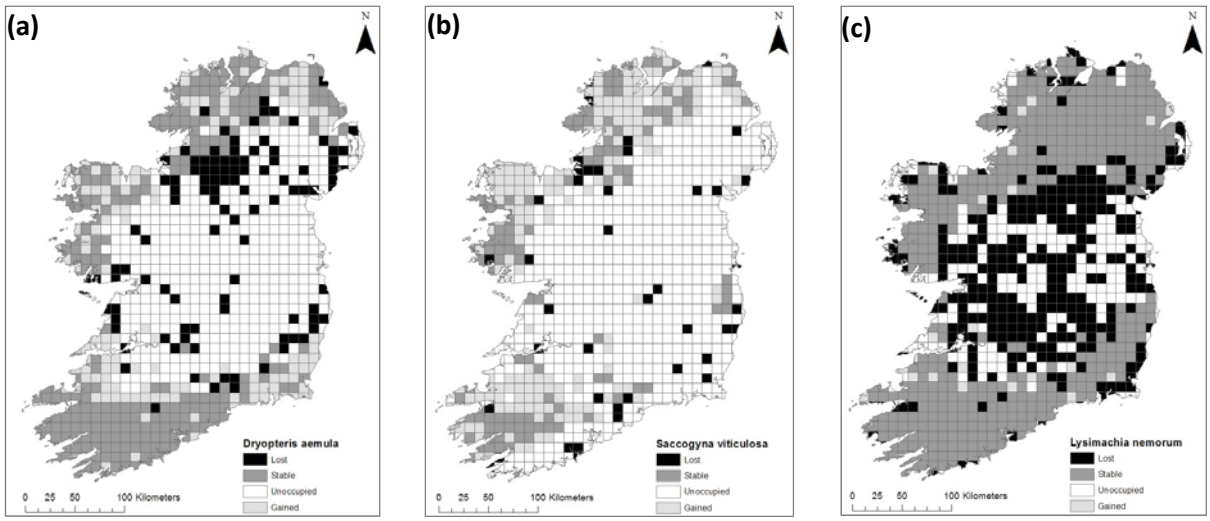


Figure 33. Species distribution maps showing the projected change in distribution under unlimited dispersal for (a) Dryopteris aemula, a hyperoceanic species, (b) Saccogyna viticulosa, an oceanic species, and (c) Lysimachia nemorum, a sub-oceanic species.

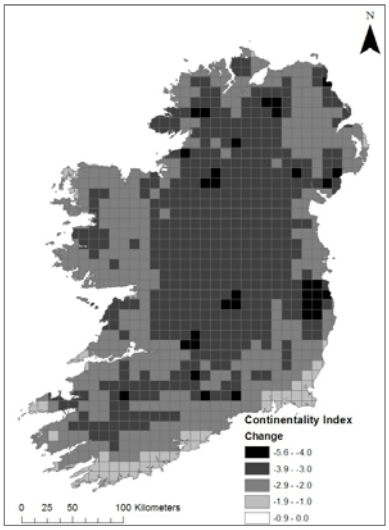


Figure 34. Projected change in continentality index. The darker colours indicate the greatest decrease (i.e. a larger difference between mean maximum and minimum temperatures).

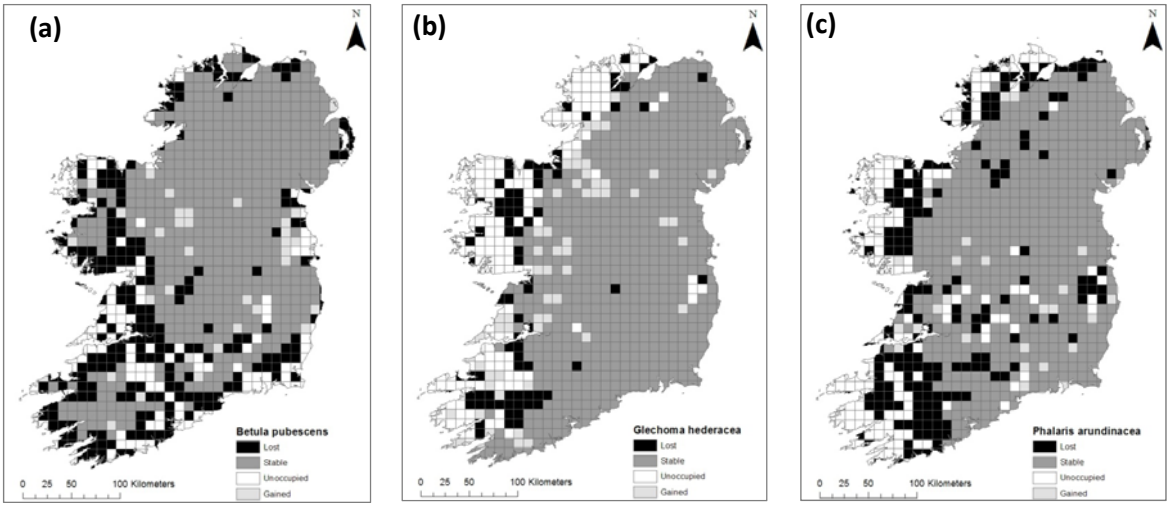


Figure 35. Species distribution maps showing the projected change in distribution for (a) Betula pubescens, a Eurosiberian species, (b) Glechoma hederacea, a Eurasian species, and (c) Phalaris arundinacea, a circumpolar species.

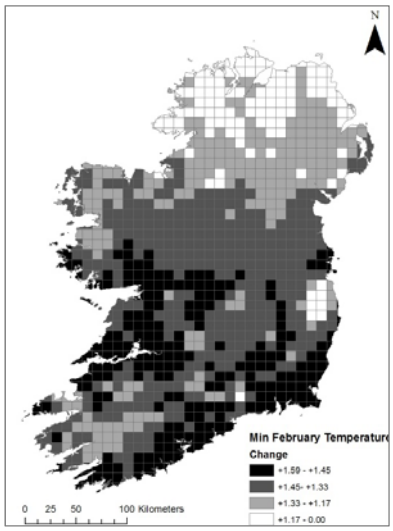


Figure 36. Projected change in minimum February temperature where the darker colours indicate the largest increase in minimum February temperature.

Discussion

The species included in this study modelled well using three climatic variables together with elevation, and the projected range changes for individual species can contribute to the understanding of the impacts of climate change on woodland species in Ireland. There is an overall picture of a loss of suitable climate space, although this varies greatly from species to species, with some species projected to gain large increases on their current range.

Under the unlimited dispersal scenario, a mean increase in range across all species is projected. This is similar to findings in a recent report on the implications of climate change for woodland biodiversity in the UK (Berry *et al.*, 2012). The majority of species modelled here are projected to experience a decrease in potentially suitable climate space under both climate change scenarios tested (24 of 46 species under the unlimited dispersal scenario, with all species losing at least some of their current range under the no long-

range dispersal scenario). The community composition in our woodlands may therefore change drastically, as individual species respond to climate change in different ways. The composition of butterfly communities in the UK has been shown to become dominated by generalist species as the climate warms (Menendez *et al.*, 2006). This pattern may be repeated across taxonomic groups, as dispersal limitations are likely to matter less to generalist species than to specialists. Generalists may also be more likely to find suitable habitat than specialists.

In the present study, all of the woodland butterfly species which modelled well were projected to lose large areas of suitable climate space under the no long-range dispersal scenario; *T. betulae* was projected to lose 14.8% with all others projected to lose 25% or more of their current range. When the unlimited dispersal scenario was used, however, only *Aphantopus hyperantus* was projected to lose climate space. Of the other four species, two were projected to increase their current range by small amounts, while the range of *Gonopteryx rhamni* was projected to increase by 89.6%, and that of *T. betulae* was projected to increase by 866.7%. The differences between range change for the unlimited and no long-range dispersal scenarios can be large, and while the limited dispersal scenario may not be applicable to such mobile organisms as butterflies, when the potential distribution of suitable food plants (*T. betulae* larvae, for example, feed primarily on *Prunus spinosa*) and any potential phenological mismatches (Buse and Good, 1996; Visser and Holleman, 2001) are taken into account, future range contractions may be even larger than projected.

Berry *et al.* (2012) examined the impacts of climate change on a number of woodland species in the UK, and some of their results are at odds with the results in this study, for example while both *Betula pubescens* and *Quercus robur* are projected to lose climate space here (at -30% and -6% of their current range under unlimited dispersal respectively), in the Berry et al. study *Q. robur* is projected to have an 'insignificant gain', and *B. pubescens* an 'insignificant loss'. These differences may reflect differences in modelling techniques, but it is likely that the larger area and more varied climate of the UK will provide more potentially suitable climate space under climate change than can Ireland. Ireland's relatively depauperate flora means that woodland communities found here are subtly different to those found in the UK, and these communities may also respond differently to climate change, as community composition shifts in favour of species more suited to the changed climate. These differences underline the necessity of research into the impacts of climate change on Irish woodland species.

The projected expansion in the distributions of hyperoceanic and oceanic plant species reflects the patterns of projected decrease in continentality index (movement towards coastal areas where smallest changes in continentality index are projected). The projected shifts in distribution of Eurosiberian, Eurasian and Circumpolar species reflect the patterns in projected change in minimum February temperature, with a general shift in distribution to the north east. The projected range shifts to the north east is a pattern which was also observed at a European scale by Bakkenes *et al.* (2002). Bryophytes show a greater increase in range under unlimited dispersal, and a smaller decrease in range under limited dispersal than vascular plants, indicating that bryophytes and vascular plants are likely to respond to climate change in different ways. Even within these broad taxonomic groups, differing projected responses to climate change were seen. These types of varied responses to climate change have been predicted/observed by many studies and suggest that range responses of individual species to climate change will lead to changes in the species composition of many communities (Le Roux and McGeoch, 2008; Lenoir *et al.*, 2008).

The results presented here have a number of implications for woodlands in Ireland. Under the no long-range dispersal scenario, all of the species which modelled well were projected to have a decreased area of suitable climate space by 2055. The 'unlimited dispersal' scenario showed that, for some species, climate change will result in either an increased area of suitable climate or a shift in potential range, however, large range shifts over a short time period are very improbable. These results show that both plant and animal species will experience changes in their ranges, related to changes in areas of suitable climate, in the future. Many of the dominant or characteristic species of Irish woodlands have limited dispersal abilities, and long-range dispersal is likely to be further restricted by a lack of suitable habitat for colonisation. Modelling of the suitable climate space of these woodland species has shown that there are species-specific responses to climate, for example *Betula pubescens* is projected to lose 30% of its current range under the unlimited dispersal scenario, while *Quercus robur* is projected to lose just 6%. The species composition of existing woodlands and woodland communities may therefore change, as some species will tolerate climate change more readily than others. Ireland has a small area of woodland, much of the landscape being occupied by agricultural land. This means that the area available for colonisation by woodland plants is smaller than in many other European countries. The maintenance and promotion of connectivity in the wider landscape and between woodlands and/or Natura 2000 sites is therefore vital to ensure that species can reach new areas of suitable climate space. The creation of green infrastructure will help maintain a heterogeneous landscape, facilitating dispersal of species to these new areas of suitable climate and habitat.

Recommendation 24: Maintain and promote connectivity in the landscape and between woodlands and Natura 2000 sites to ensure that species can disperse to new areas of suitable climate space.

A number of the species included in this study are of conservation interest. *Frangula alnus* is listed as 'rare' in the Irish Red Data list for vascular plants (Curtis and McGough, 1988), and the increase in its potential range (under the unlimited dispersal scenario) may aid in its conservation. *Sematophyllum demissum* is a bryophyte listed as 'near threatened' by Lockhart *et al.* (2012). It was projected to have a large increase in potential range (under the unlimited dispersal scenario), while *Sphagnum quinquefarium* was projected to more than quadruple its current range, although availability of suitable habitat will restrict any actual range expansion. *Leptidea sinapis*, the wood white butterfly, which is listed as 'near threatened' by Regan and Fleischer (2010), is projected to experience a contraction of its current range under climate change.

Recommendation 25: Conservation measures for plant and animal species should take into account the projected future distributions of species and the habitats on which they depend.

There are a number of ways in which the impacts of climate change on the distribution of woodland species in Ireland might be mitigated. The most important is probably the maintenance and conservation of existing woodlands, especially protecting them from other stressors (e.g. excessive grazing pressures) to increase the likelihood of survival of the species which are present. The ecological impacts associated with climate change will not occur in isolation; rather climate-driven changes will combine with, and exacerbate, existing stresses on Ireland's natural systems. An understanding of those interactions will become increasingly critical in defining and implementing effective conservation measures.

Recommendation 26: Maintenance and conservation of existing woodlands, including their protection from stressors, should be undertaken to mitigate the negative effects of climate change.

Under future climate change existing woodlands of one type might also become suitable habitats for species which currently live in other types of woodland. Some species in the future will not be capable of migrating to new areas of suitable climate and habitat or adapting to new conditions. The translocation of vulnerable species from areas which are projected to become inhospitable to areas within the new projected range could be attempted, a process known as 'assisted migration' (McLachlan *et al.*, 2007) or 'assisted colonisation' (Hoegh-Guldberg *et al.*, 2008). While this is an unpredictable and potentially expensive method of conservation, it has been used with some success to re-colonise areas with species formerly indigenous to that area and could potentially be an important part of climate change mitigation in Ireland.

Recommendation 27: The viability of translocation to areas with suitable climate and habitat as a conservation measure should be investigated, especially for species that are currently vulnerable or red listed to avoid extinction.

Managed, commercial woodlands could be made more profitable/viable through the use of plants of a more southerly provenance, e.g. plantations could be stocked with plants sourced from regions with climates currently similar to those projected (Broadmeadow *et al.*, 2005). This type of management could help to maintain some species in the woodland, but the overall species composition is likely to change, due to changes in suitable climate space for other species.

Recommendation 28: Plantation forests should be stocked with trees sourced from regions with climates currently similar to those projected, in order to maintain productivity.

Conclusions

This study has shown that, for many of the characteristic species of semi-natural woodland in Ireland, climate change is a real threat. For others, however, it may represent a real opportunity. The area of suitable climate space is projected to shrink for all species, given a no long-range dispersal scenario. While the reality may be that actual dispersal lies between the unlimited and no long-range dispersal scenarios, other factors such as land use and biotic interactions will also impact on future distribution. Further information on responses to biotic and abiotic factors, dispersal ability and phenological responses will be required to determine appropriate conservation measures for woodlands and woodland species in Ireland. The relatively small area and fragmented nature of Irish semi-natural woodlands makes their conservation a priority. Threats from climate change are high, as outlined above, and further investigations into the impact of climate change are required to gain an increased understanding of the potential range changes for these species and communities of high conservation value.

Chapter 13

BIOPLAN GIS database

Rory Scarrott, Kathrin Kopke and David Roig Cervera

Supporting Work Packages 1, 2 and 5

Forest biodiversity science, where the collection of primary data is expensive and time consuming, has a lot to gain from the sharing of data. To ensure maximum exploitation of biodiversity data after collection data management plans, with strict standards relating to data accessibility, metadata and quality control are required for all forest biodiversity research projects. Data collected during the BIOPLAN project were gathered, harmonised and integrated into a GIS database, which allows access, visualisation and further analyses of the datasets, and can be easily distributed and updated with future research. Each of the datasets collated within the Work Packages 1, 2 and 5 are described fully by INSPIRE-compliant metadata, which is also held in the GIS database. Creation of the database involved the implementation of a GIS strategy within BIOPLAN. Researchers and research support staff were provided with education and training, as well as technical support. The output is a BIOPLAN GIS database, consisting of a geodatabase containing data and associated metadata, with a visualisation aid to encourage and facilitate data exploration, and re-use. The geodatabase forms the core of a store of baseline data, which can be updated and enhanced by future studies. Furthermore, the application of rigorous metadata standards allows for enhanced opportunities for partner institutions and the state, to demonstrate the skills and potential of Irish collaborative forest and biodiversity research.

Background

Forest biodiversity science is the study of the variety of life in our forests which increasingly seeks to address questions that cover large temporal and spatial scales, for example the effects of climate change, afforestation or forest management practices (Enke *et al.*, 2012). Biodiversity data can be numerical, categorical or pictorial and includes inventories of species name, locations and analysis (Costello *et al.*, 2013). By its nature this data is expensive and time consuming to collect and so the availability of archived biodiversity data can make a significant contribution to the future of this discipline. Data sharing is therefore an important issue in modern biodiversity research (Enke *et al.*, 2012; Costello *et al.*, 2013) and can create opportunities for more integrative forest biodiversity research and analysing data in additional ways (Bendix *et al.*, 2012; Costello and Wieczorek, 2013). Scientific questions can be answered over larger spatial scales due to the opportunities for integration of data collected by researchers and projects over wide geographic scales.

A geographic information system (GIS) integrates hardware, software, human resources and data for capturing, managing, analysing, and displaying all forms of geographically referenced information. It allows us to link features commonly seen on maps (such as roads, town boundaries, water bodies) with related information not usually presented on maps, such as type of road surface, population, type of agriculture, type of vegetation, or water quality information.

The benefits of implementing a GIS database within a research project fall into three categories:

1. **Supporting better decision making** by storing data in a framework, from which is it relatively easy to extract meaningful and informative visuals, and summaries of conclusions which have stemmed from BIOPLAN research.
2. **Allowing for better record keeping** of datasets collated over the course of BIOPLAN
3. **Enabling better understanding** of what is happening in geographical terms in forests studied within the framework of BIOPLAN.

Previous to the PLANFORBIO project, the potential benefits of implementing a GIS strategy to support the work of biodiversity researchers were highlighted by the BIOFOREST project (Iremonger *et al.*, 2006). Data collated were spatially referenced, and stored in a single file location, together with metadata (information about each dataset), and accompanied by a visualisation tool, allowing users with basic GIS knowledge to explore, extract and re-use the data collected over the course of BIOFOREST.

PLANFORBIO further exploited the potential of implementing a GIS strategy within all four of the funded sub-projects - FORESTBIO, HENHARRIER, RHODO, and BIOPLAN. At its most basic implementation level, the focus of this strategy is (i) to create a geographically referenced data store within each sub-project, (ii) to provide easily accessible information about each dataset (metadata) within the store, compliant to standards outlined by the INSPIRE Directive (2007/2/EC) and its subsequent metadata implementation regulation (EC Regulation No. 1205/2008), and (iii) to provide a useful visualisation tool to encourage future users to further explore the datasets, and facilitate presenting aspects of the data held within. This fundamental implementation goal has been achieved in all four PLANFORBIO sub-projects, with more highly analytical and research-enabling strategies implemented in a number of studies (within HENHARRIER for example).

The technology implemented has also advanced considerably since the completion of the BIOFOREST GIS. Datasets collated within PLANFORBIO are now held in a single geodatabase architecture file (as opposed to a folder containing a large number of ESRI shapefiles), together with their metadata which is also linked to the data file. This allows for a considerably higher data to disk-space ratio, improving the efficiency of storing data collated within the PLANFORBIO project. The visualisation tool (a simple ArcMap document), simply accesses this geodatabase. It is also notable that datasets held within a geodatabase file are also fully updateable, allowing for additional datasets to be added into the future, or amendments and further quality control to be implemented on existing data.

This chapter presents the details of the geodatabase constructed in support of the BIOPLAN sub-project. This is the largest pool of spatial data to have been gathered over the course of the PLANFORBIO project. It is worth noting that the same fundamental principals have been applied in all GIS implemented within PLANFORBIO, these being:

1. Where practical, all data with a spatial location has been collated into spatially referenced datasets held within a Geodatabase.
2. All datasets collected using PLANFORBIO resources (excluding third party datasets) are fully described with INSPIRE-compliant metadata.
3. All datasets held within the geodatabase are integrated within the visualisation map document, and are accessible via this document.

This "BIOPLAN GIS" (referring to the collated data, metadata and visualisation document file) contains and presents zoological, botanical, and environmental field data, collected during the BIOPLAN Project. The Geodatabase contains 70 Feature Class datasets and 4 Raster datasets, and is accompanied by ArcMap (licenced software) and ArcReader (licence-free software) templates for visualisation. The data are contained alongside four supporting baseline datasets giving visually interpretive information on relief derived from NASA's SRTM mission, imagery derived from ENVISAT MERIS and Landsat TM data, and land administration boundaries derived from data provided by the Irish Central Statistics Office.

Accessing the information via licenced software (ArcMap) allows access to tabular data, imagery and metadata (information about the data). Examples of information contained in the metadata include descriptions of schema, tables, indexes, column definitions, data collection methods, contact information, use constraints etc.). The metadata for all datasets contained in the BIOPLAN Geodatabase has been compiled to the INSPIRE-compliant ISO-19115 standard for metadata (ISO, 2003; Craglia, 2013). The GIS is fully-functional, flexible and updateable. The information contained in the geodatabase can also be accessed using a licence-free (and cost-free) software system (ArcReader) that can be run on any computer. However it is worth noting that ArcReader has comparatively limited functionality to that possible using licenced ArcInfo software, allowing less efficient access to the datasets. Furthermore, when only ArcReader is available, the metadata for each dataset must be viewed using *.html documents also stored on the product disc.

The BIOPLAN GIS allows access, visualisation and further analysis of the spatially referenced data collated over the course of the BIOPLAN project. It allows for further integration of this data into the overall PLANFORBIO database and beyond, with the additional capacity to be updated with future research records if required.

Implementation of the BIOPLAN GIS strategy required a number of goals to be set, and achieved practically. In general these can be summarised as:

- Educating and enabling researchers through demonstrations and technical advice to successfully carry out basic GIS tasks.
- Ensuring researchers geo-located every location from which data were sampled throughout their individual research projects.
- Providing guidance and education to BIOPLAN researchers on the information requirements of the INSPIRE-compliant ISO19115 metadata standard, and how to practically implement it.
- Providing technical support to achieve all of the above.
- Harmonising and quality-controlling incoming finalised datasets and metadata from all partner institutions, and integrating these into a tailored BIOPLAN geodatabase and associated visualisation documents (see Figure 37).

This guidance, support, and GIS compilation was principally implemented by University College Cork's Coastal and Marine Research Centre (Now Beaufort Research), with considerable support from the PLANFORBIO project management team and research staff. Information on the practical aspects on how each goal was achieved can be found in Table 16.

Table 16. Key goals and practical elements of implementation which were used to implement the BIOPLAN GIS strategy, and produce a GIS database for the project.

Goal	**Educating and enabling researchers**
Practical elements	Researchers from all partner institutions were up-skilled with fundamental GIS skills and knowledge through: • UCC's advanced GIS capable staff from the CMRC presenting informative instructions at Kick-off and progress meetings • UCC's CMRC and Department of Geography providing links and access codes to ESRI's suite of online courses. • UCC's CMRC staff providing experiential knowledge, technical support and advice via email and phone to individual researchers.
Goal	**Ensuring researchers geo-located all data**
Practical elements	All data sampled over the course of BIOPLAN research was geo-located through: • Researchers integrating the recording of each sample site's location into their survey strategies and site assessment plans. • CMRC staff liaising with the BIOPLAN project management team and Principal Investigators in the various institutions, to ensure that the more junior researchers adhered to recording the spatial location of their samples as standard practice.
Goal	**Providing guidance and education on the ISO-19115 metadata standard**
Practical elements	A number of avenues were used to ensure researchers throughout the BIOPLAN project, and the wider PLANFORBIO programme, were being educated in, and implementing, INSPIRE-compliant metadata standards within their research. This was achieved through: • UCC's advanced GIS staff from the CMRC presenting informative instructions at Kick-off and progress meetings • Instructions in compiling metadata being provided for the researchers who chose to use ArcCatalog software. • An excel sheet with all the necessary fields to be filled in for each dataset being distributed amongst all researchers. CMRC staff then integrated the information into each dataset's ArcCatalog metadata. • UCC's CMRC staff providing experiential knowledge, technical support, and advice via email and phone to individual researchers. • PLANFORBIO's project management team ensuring researchers availed of the supports available, and remained aware of the project's obligations pertaining to metadata.
Goal	**Providing technical support**
Practical implementation	• UCC's CMRC staff providing experiential knowledge, technical support, and advice via email and phone to individual researchers.

Goal	**Harmonising, quality-controlling, and integrating datasets into the geodatabase and visualisation documents**
Practical implementation	The following process was carried out in the CMRC to derive a BIOPLAN GIS from datasets received from all research partners (Figure 37). 1. The geodatabase framework was created in ArcCatalog, alongside the ArcMap Visualisation document in ArcMap. All ArcGIS work was done using ArcInfo 10.0 to 10.2 software. 2. Datasets received by the CMRC arrived in two formats (i) a pre-prepared shapefile, or (ii) and excel sheet containing X and W coordinates in either the WGS1984 Lat/long datum, or in Irish National Grid. In situations where (ii) was the case, the X and Y coordinates were used to create point shapefiles. 3. Shapefiles were re-projected, where necessary, into Irish National Grid based on the TM65 datum. 4. The metadata were added to the shapefile dataset in the geodatabase. Upon completion, *.html documents suitable for display using Webpage software such as Internet Explorer and Firefox, were derived from the Metadata held in the geodatabase. 5. Shapefiles (data + metadata) were imported into the geodatabase, and introductory aspects of their data visualised within the map document. 6. Metadata and data in the Geodatabase were quality checked. Where necessary, metadata were edited within the Geodatabase to account for amendments requested by the researchers. 7. Upon completion of all datasets integration into the geodatabase and the Visualisation document, versions of the Map Document compatible with ArcGIS versions 9.2, 9.3 were also created. 8. The ArcMap Document was also published as an ESRI Published Map, compatible with ArcReader software. 9. A helpful Manual for novel users of the BIOPLAN GIS was compiled and finalised as a *.pdf document. 10. The geodatabase, ArcGIS map documents, ArcReader map document, metadata in *.html, and BIOPLAN GIS Manual were compiled into a BIOPLAN GIS product CD (a copy of which accompanies this report).

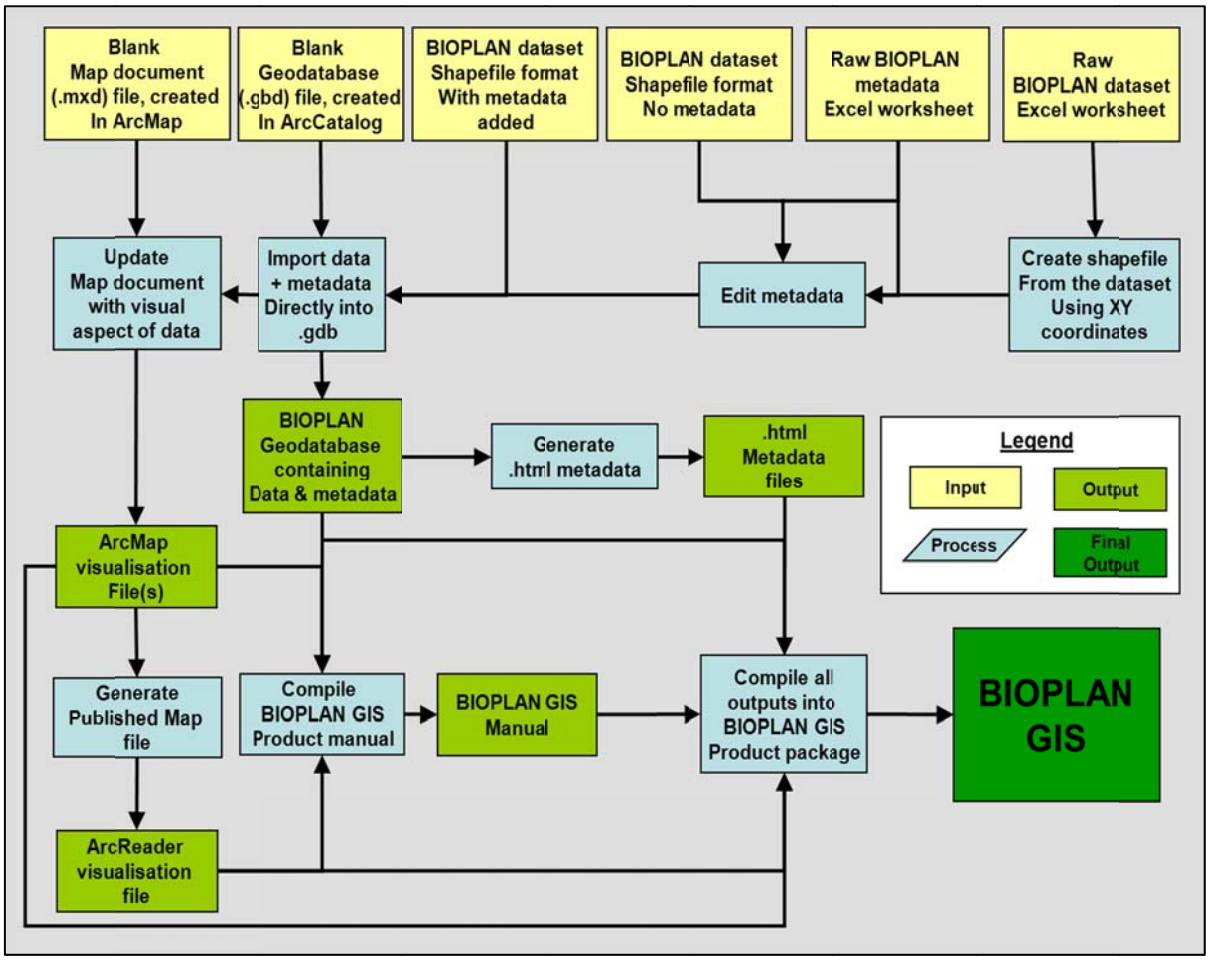


Figure 37. Harmonising and processing datasets and metadata, and compiling the BIOPLAN GIS.

Output - the BIOPLAN GIS

The BIOPLAN GIS captures any data with a spatial element collated by researchers participating in the BIOPLAN project. It provides an updateable system that allows access, visualisation and further analysis of the spatial data component within the BIOPLAN project. The BIOPLAN database is compatible with the overall PLANFORBIO research program suite of databases. This provides a 'one-stop-shop' for the spatial data component of data collected within the framework of PLANFORBIO.

BIOPLAN GIS description

The BIOPLAN GIS consists of a product CD containing a number of files, or groups of files:

- A data and metadata storage file in the form of an ArcCatalog geodatabase (.gdb).
- Data visualisation file in the form of an ArcMap map document (.mxd) - visualising basic elements of the data in the *.gdb file. ArcMap documents compatible with earlier versions of ArcGIS software.
- A helpful instruction manual for novices to exploring GIS databases
- Metadata for all the datasets in web-browser readable *.html format

The data and metadata storage component

The datasets and their associated metadata descriptions are held in the geodatabase (*.gdb) file. This contains 70 Feature Class datasets and 4 Raster datasets, and is accessible using licenced ArcCatalog

software. Informative summaries of the datasets of interest can be examined through looking at the metadata, and exported if needed into a more flexible shapefile (*.shp) or GeoTiff (*.tif) format for further processing and analysis. All datasets use the Irish National Grid geo referencing system, based on the TM65 Datum. This is the standard referencing system applied to all datasets incorporated into any of the four PLANFORBIO Geodatabases.

Figure 38 shows the list of datasets as they would appear in ArcCatalog. The datasets have been collated within the three subject areas of interest for BIOPLAN, namely bird survey data, invertebrate survey data, and Ground-vegetation survey data. Biotic, abiotic, habitat assessment and summary metrics are available for all three areas of interest. These are supported in the geodatabase by contextual baseline data, to aid interpretation, navigation, and exploration of potential linkages between survey results. A summary of the datasets collated within areas of interest and for the baseline data can be found in Table 11.

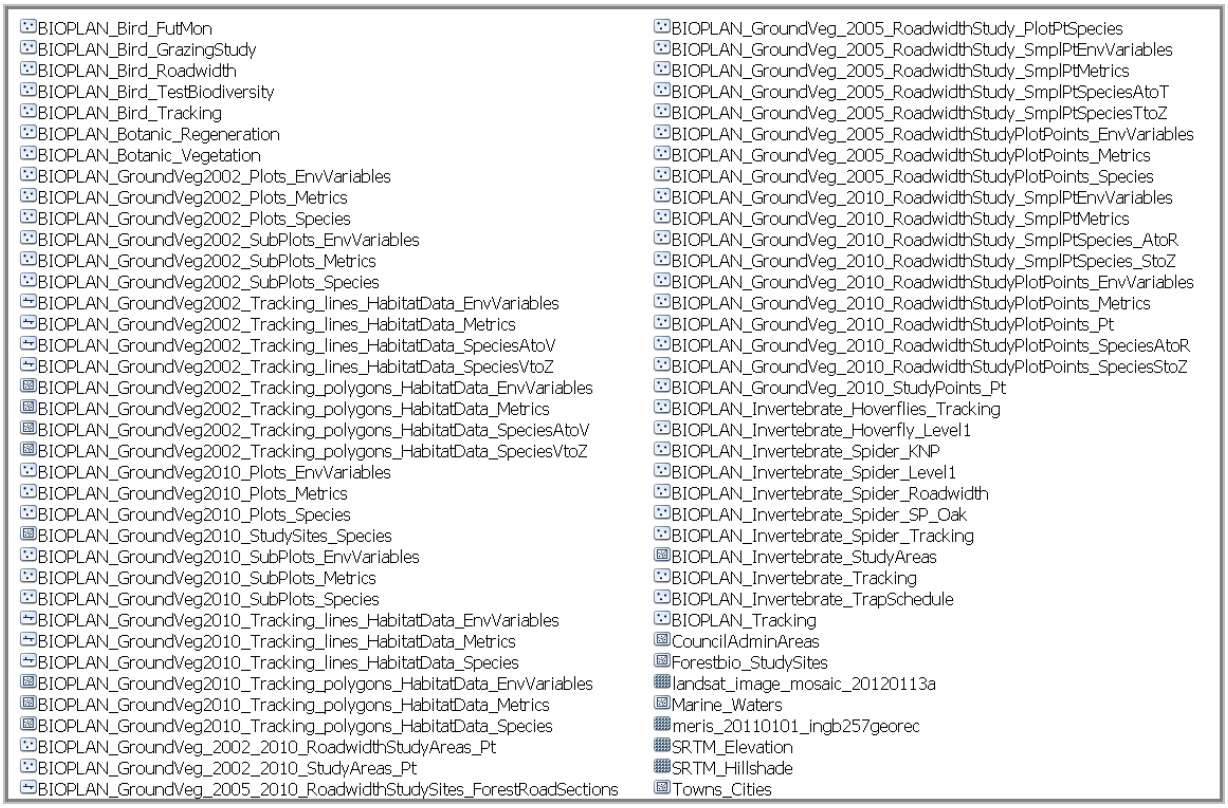


Figure 38. Overview of BIOPLAN Geodatabase in ArcCatalog containing Feature Datasets with Feature Classes and Tables (figure prepared in ArcCatalog Version 10.1).

Table 11. Datasets held within the BIOPLAN geodatabase

Dataset	Dataset File Name	Originators
Baseline		
Irish Coastline	Marine_waters	CMRC (UCC)
Irish Admin Boundaries	CouncilAdminAreas	CMRC (UCC)
30m Landsat TM mosaic	landsat_image_mosaic_20120113a	CMRC (UCC);NASA
250m ENVISAT MERIS image	meris_20110101_ingb257georec	CMRC(UCC);ESA
50m Elevation SRTM DEM	SRTM_Elevation	NASA
DEM-derived hillshade	SRTM_Hillshade	CMRC(UCC); ASA
ROI Town & city locations	Towns_Cities	CSO
FORESTBIO sites 2007/2008	Forestbio_StudySites	BEES (UCC)
Ground Vegetation		
tree regeneration monitoring	BIOPLAN_Botanic_Regeneration	DOB (TCD)
ground veg. monitoring	BIOPLAN_Botanic_Vegetation	DOB (TCD)
2002 & 2010 Ground vegetation survey sites	BIOPLAN_GroundVeg_2002_2010_StudyAreas_Pt	DOB (TCD)
2002 & 2010 road width study sites	BIOPLAN_GroundVeg_2002_2010_RoadwidthStudyAreas_Pt	DOB (TCD)
2005 & 2010 forest road study sites	BIOPLAN_GroundVeg_2005_2010_RoadwidthStudySites_ForestRoadSections	DOB (TCD)
2010 Locations of the ground vegetation study sites	BIOPLAN_GroundVeg_2010_StudyPoints_Pt	DOB (TCD)
2005 road width study plot sites	BIOPLAN_GroundVeg_2005_roadwidthStudy_PlotPtSpecies	DOB (TCD)
2005 road width study plots - species counts	BIOPLAN_GroundVeg_2005_RoadwidthStudyPlotPoints_Species	DOB (TCD)
2005 road width study plots community metrics	BIOPLAN_GroundVeg_2005_RoadwidthStudyPlotPoints_Metrics	DOB (TCD)
2005 road width study plots environmental conditions	BIOPLAN_GroundVeg_2005_RoadwidthStudyPlotPoints_EnvVariables	DOB (TCD)
2005 road width sample sites - species counts A-T	BIOPLAN_GroundVeg_2005_RoadwidthStudy_SmplPtSpeciesAtoT	DOB (TCD)
2005 road width sample sites - species counts T-z	BIOPLAN_GroundVeg_2005_RoadwidthStudy_SmplPtSpeciesTtoZ	DOB (TCD)
2005 road width sample sites community metrics	BIOPLAN_GroundVeg_2005_RoadwidthStudy_SmplPtMetrics	DOB (TCD)
2005 road width sites environmental conditions	BIOPLAN_GroundVeg_2005_RoadwidthStudy_SmplPtEnvVariables	DOB (TCD)
2005 road width study plot sites	BIOPLAN_GroundVeg_2010_RoadwidthStudyPlotPoints_pt	DOB (TCD)
2010 road width study plots - species counts - Species A-R	BIOPLAN_GroundVeg_2010_RoadwidthStudyPlotPoints_SpeciesAtoR	DOB (TCD)
2010 road width study plots - species counts - Species S-Z	BIOPLAN_GroundVeg_2010_RoadwidthStudyPlotPoints_SpeciesStoZ	DOB (TCD)
2010 road width study plots environmental conditions	BIOPLAN_GroundVeg_2010_RoadwidthStudyPlotPoints_Metrics	DOB (TCD)
2010 road width study plots - species counts - Species A-R	BIOPLAN_GroundVeg_2010_RoadwidthStudy_SmplPtSpecies_AtoR	DOB (TCD)
2010 road width study plots community metrics	BIOPLAN_GroundVeg_2010_RoadwidthStudyPlotPoints_EnvVariables	DOB (TCD)

Dataset	Dataset File Name	Originators
Ground Vegetation cont.		
2010 road width sample sites - species counts - Species S-Z	BIOPLAN_GroundVeg_2010_RoadwidthStudy_SmplPtSpecies_StoZ	DOB (TCD)
2010 road width sample sites community metrics	BIOPLAN_GroundVeg_2010_RoadwidthStudy_SmplPtMetrics	DOB (TCD)
2010 road width sample sites environmental conditions	BIOPLAN_GroundVeg_2010_RoadwidthStudy_SmplPtEnvVariables	DOB (TCD)
2002 Subplot species counts	BIOPLAN_GroundVeg2002_Plots_Species	DOB (TCD)
2002 Plot community metrics	BIOPLAN_ GroundVeg2002_Plots_Metrics	DOB (TCD)
2002 Plot Environmental conditions	BIOPLAN_GroundVeg2002_Plots_EnvVariables	DOB (TCD)
2002 Subplot species counts	BIOPLAN_GroundVeg2002_SubPlots_Species	DOB (TCD)
2002 Subplot community metrics	BIOPLAN_GroundVeg2002_SubPlots_Metrics	DOB (TCD)
2002 Subplot Environmental conditions	BIOPLAN_GroundVeg2002_SubPlots_EnvVariables	DOB (TCD)
2002 Track line species counts A-V	BIOPLAN_GroundVeg2002_Tracking_lines_HabitatData_SpeciesAtoV	DOB (TCD)
2002 Track line species counts V-Z	BIOPLAN_GroundVeg2002_Tracking_lines_HabitatData_SpeciesVtoZ	DOB (TCD)
2002 Track line community metrics	BIOPLAN_GroundVeg2002_Tracking_lines_HabitatData_Metrics	DOB (TCD)
2002 Track line Environmental conditions	BIOPLAN_GroundVeg2002_Tracking_lines_HabitatData_EnvVariables	DOB (TCD)
2002 Track area species counts A-V	BIOPLAN_GroundVeg2002_Tracking_polygons_HabitatData_SpeciesAtoV	DOB (TCD)
2002 Track area species counts V-Z	BIOPLAN_GroundVeg2002_Tracking_polygons_HabitatData_SpeciesVtoZ	DOB (TCD)
2002 Track area community metrics	BIOPLAN_GroundVeg2002_Tracking_polygons_HabitatData_Metrics	DOB (TCD)
2002 Track area Environmental conditions	BIOPLAN_GroundVeg2002_Tracking_polygons_HabitatData_EnvVariables	DOB (TCD)
2010 species summaries per sample site	BIOPLAN_GroundVeg2010_StudySites_Species	DOB (TCD)
2010 Plot species counts	BIOPLAN_GroundVeg2010_Plots_Species	DOB (TCD)
2010 Plot community metrics	BIOPLAN_GroundVeg2010_Plots_Metrics	DOB (TCD)
2010 Plot environmental conditions	BIOPLAN_GroundVeg2010_Plots_EnvVariables	DOB (TCD)
2010 Subplot species counts	BIOPLAN_GroundVeg2010_SubPlots_Species	DOB (TCD)
2010 Subplot community metrics	BIOPLAN_GroundVeg2010_SubPlots_Metrics	DOB (TCD)
2010 Subplot environmental conditions	BIOPLAN_GroundVeg2010_SubPlots_EnvVariables	DOB (TCD)
2010 Track line species counts	BIOPLAN_GroundVeg2010_Tracking_lines_HabitatData_Species	DOB (TCD)
2010 Track line community metrics	BIOPLAN_GroundVeg2010_Tracking_lines_HabitatData_Metrics	DOB (TCD)

Dataset	Dataset File Name	Originators
2010 Track line environmental conditions	BIOPLAN_GroundVeg2010_Tracking_lines_HabitatData_EnvVariables	DOB (TCD)
2010 Track area species counts	BIOPLAN_GroundVeg2010_Tracking_polygons_HabitatData_Species	DOB (TCD)
2010 Track area environmental conditions	BIOPLAN_GroundVeg2010_Tracking_polygons_HabitatData_EnvVariables	DOB (TCD)
Birds		
2010 Track area community metrics	BIOPLAN_GroundVeg2010_Tracking_polygons_HabitatData_Metrics	DOB (TCD)
Biodiversity surveys	BIOPLAN_Bird_TestBiodiversity	BEES (UCC)
Grazing impact surveys	BIOPLAN_Bird_GrazingStudy	BEES (UCC)
FutMon surveys	BIOPLAN_Bird_FutMon	BEES (UCC)
Bird Tracking study sites & survey dates	BIOPLAN_Tracking	BEES (UCC)
Tracking surveys	BIOPLAN_Bird_Tracking	BEES (UCC)
Road-width impact surveys	BIOPLAN_Bird_Roadwith	BEES (UCC)
Invertebrates		
2005-2010 road width manipulation surveys spiders	BIOPLAN_Invertebrate_Spider_Roadwidth	BEES (UCC)
2005-2010 road width manipulation surveys spiders	BIOPLAN_Invertebrate_Tracking	BEES (UCC)
2005-2010 road width manipulation surveys spiders	BIOPLAN_Invertebrate_Hoverflies_Tracking	BEES (UCC)
2005-2010 road width manipulation surveys spiders	BIOPLAN_Invertebrate_Spider_Tracking	BEES (UCC)
Active ground dwelling spider surveys (Killarney Nat. Park)	BIOPLAN_Invertebrate_Spider_KillarneyNatPark	BEES (UCC)
Active ground dwelling spider surveys (Scots Pine & Oak)	BIOPLAN_Invertebrate_Spider_SP_Oak	BEES (UCC)
Species counts survey plot sites	BIOPLAN_Invertebrate_StudyAreas	BEES (UCC)
Species counts plot trapping schedule	BIOPLAN_Invertebrate_TrapSchedule	BEES (UCC)
Species counts spider	BIOPLAN_Invertebrate_Spider_Level1	BEES (UCC)
Species counts Hoverflies	BIOPLAN_Invertebrate_HoverflyLevel1	BEES (UCC)

The data visualisation component

The ArcMap (*.mxd) document allows for visualisation of the Geodatabase data (Figure 39). Using ArcMap, the datasets can be queried as in the habitats example shown in Figure 40, providing detailed information on data acquired over the course of the BIOPLAN project. ArcMap facilitates data query and analyses via several tools e.g. the identifier selecting a specific feature, or the attribute table for a specific layer as demonstrated in Figure 40. The Attribute table is shown for the "Bird_point" data in Figure 41, also illustrating that a further amount of information apart from the information that is visualised in ArcMap is stored within the database and is also available for investigation via ArcMap for each individual data layer.

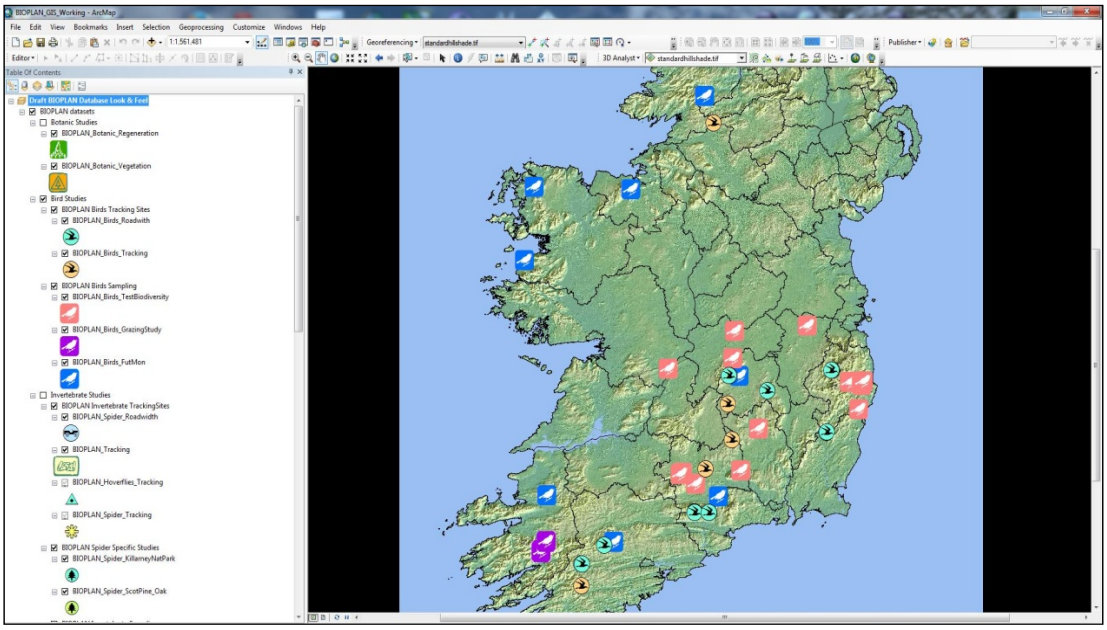


Figure 39. Visualising the data using the BIOPLAN GIS visualisation document (figure prepared in ArcMap Version 10.1.).

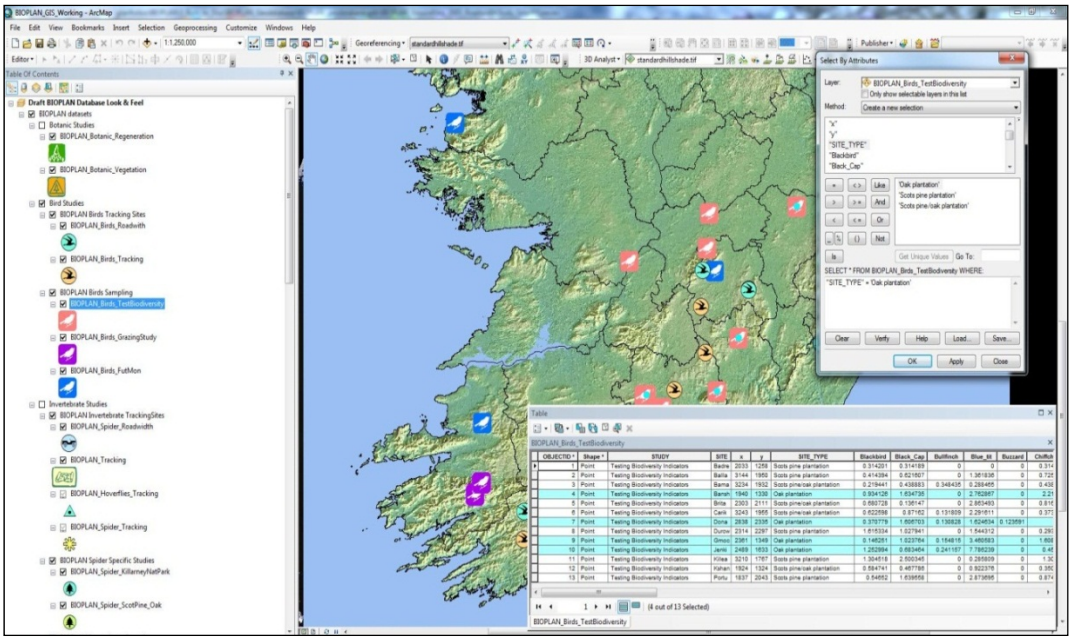


Figure 40. Exploring a dataset through looking at its attribute table, and querying it using SQL in ArcMap (figure prepared in ArcMap Version 10.1.).

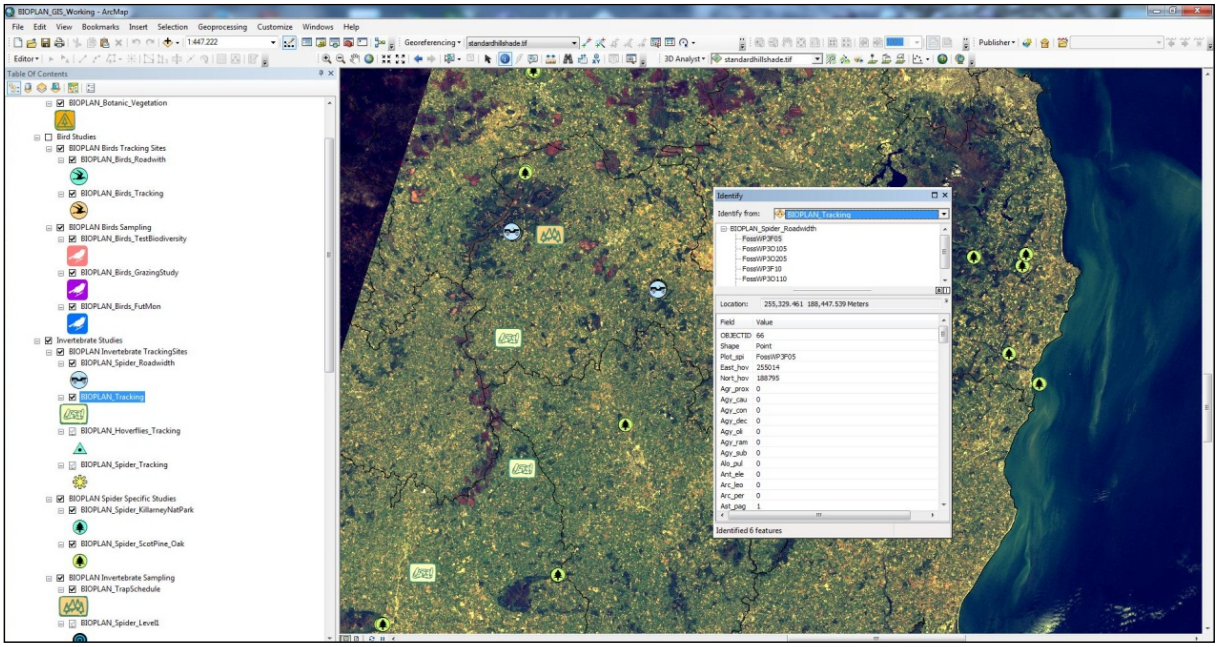


Figure 41. Using the simplest form of data point query in ArcMap. Individual points of information can be examined, whilst held visually in context with baseline information on their location (in this case a Landsat TM satellite-derived image). Figure prepared in ArcMap Version 10.1.

Note that interrogation of the Geodatabase is also possible directly via ArcCatalog. Single data points, polygons and locations can be interrogated for data using the "identifier" tool as shown in Figure 42. This also demonstrates how contextual baseline information in the form of satellite imagery is held within the geodatabase.

The ArcMap visualisation document has been specifically designed to encourage the user to explore datasets held in the Geodatabase. It allows access, visualisation and further analysis of the spatial data component of the BIOPLAN project. It also allows for further integration of these data into the overall PLANFORBIO suite of datasets, whilst having the additional capacity to be updated with future research records if required. More extensive information on the datasets held in the Geodatabase, and visualised in the GIS, and basic exploration instructions, can be found in the BIOPLAN GIS User Manual provided on the BIOPLAN-GIS CD accompanying this report. Also note that the visualisation document is available on the accompanying CD as an ArcMap Document for licensed users of ArcGIS software, and also as a less functional, though useful, ArcReader Published Map document for users of un-licensed ArcReader software.

Help guide for novices

More extensive information on the datasets held in the Geodatabase, and visualised in the GIS, and basic exploration instructions, can be found in the BIOPLAN GIS User Manual provided on the BIOPLAN-GIS CD accompanying this report. This basic guide also contains useful information for novice GIS users who are initially examining the complex array of data held in the BIOPLAN GIS, and information on the technical software requirements of starting the GIS on the one's computer. It is highly recommended that this document is consulted before the GIS is accessed.

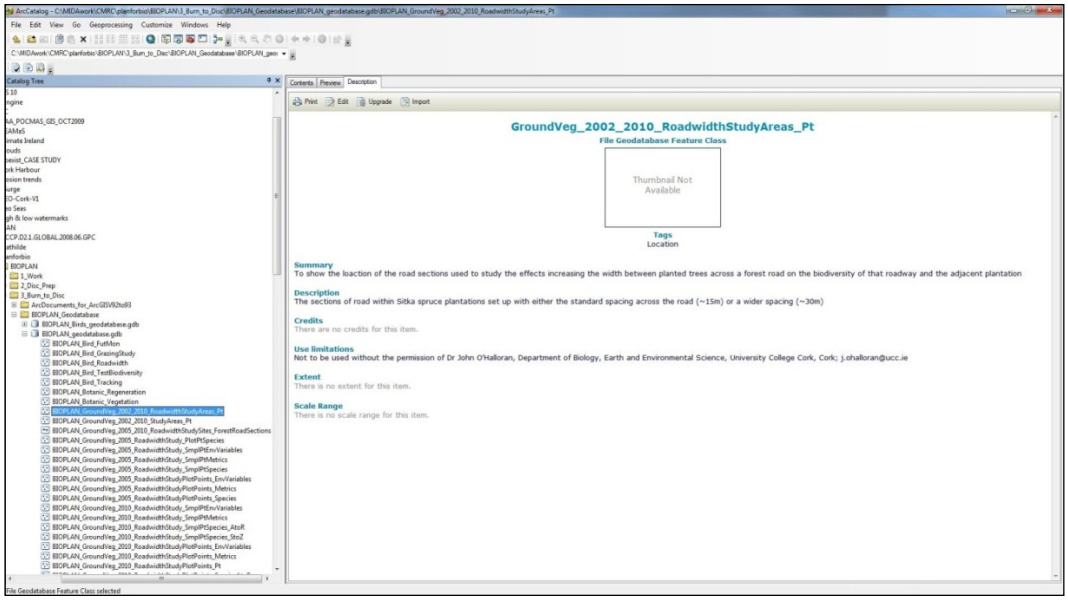


Figure 42. Viewing Metadata in ArcCatalog. The metadata shown here is for one of the ground vegetation datasets (figure prepared in ArcCatalog Version 10.1).

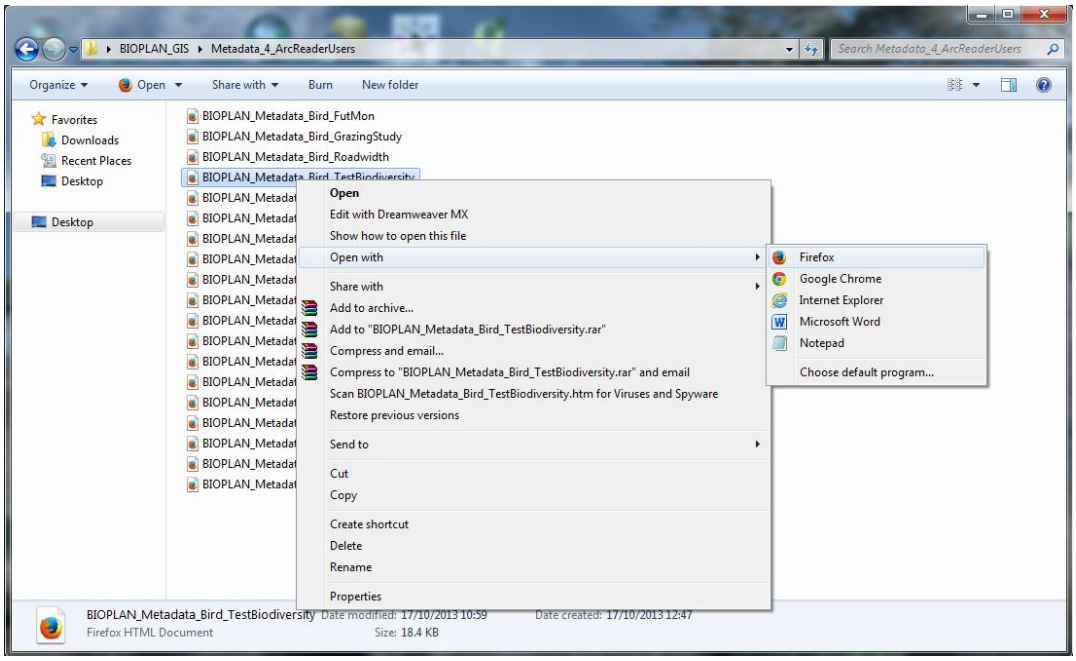


*Figure 43. Viewing Metadata in using the web-browser enabled *.html versions. The metadata shown here is for one of the bird datasets (figure prepared in ArcCatalog Version 10.1).*

Metadata explaining the data

Appraising the information about the data before using it is absolutely critical in the view of the BIOPLAN project. The data available are extensive, and often highly complex, and to fully exploit its potential, it is strongly advised that the metadata be consulted.

The BIOPLAN GIS strategy involved implementing two ways through which metadata can be examined by:

i. Using ArcCatalog to view the metadata records stored within the geodatabase (see Figure 42 as an example).

ii. Opening the *.html files in a web-browser (such as Firefox, or internet explorer which have both been tested whilst compiling the GIS). An example can be seen in Figure 43.

Further information can be found in the BIOPLAN GIS User's Manual provided on the BIOPLAN-GIS CD accompanying this report.

Discussion

The BIOPLAN GIS provides a single repository for all data with a spatial (i.e. where) element collected over the course of BIOPLAN. At its core is a series of 74 datasets recording in detail the data collated over during BIOPLAN. The GIS allows for data access, further analysis, visualisation, and exploration of this data. Furthermore, it is updateable, and flexible, allowing it to form a baseline of biological, and environmental data for use into the future. As such is it a recommendation that future work stemming from the BIOPLAN (PLANFORBIO) project or complimentary to the BIOPLAN (PLANFORBIO) project implement a similar GIS strategy within its framework, collating data that are compatible to that already held in the PLANFORBIO Geodatabases.

Recommendation 29: Future research conducted in Ireland of a similar nature to that carried out on the BIOPLAN project should implement a similar GIS strategy involving data collection and storage within the project framework. Data collected should be compatible with archived BIOPLAN data.

The GIS allows users to view, understand, question, interpret, visualise and analyse the extensive and often complex data in many ways that can reveal relationships, patterns, and trends. It also allows for aspects of the datasets already held within it to be visualised in the form of maps, globes, reports, and charts. The collection of datasets, held together and spatially harmonised, offers opportunities for further cross-disciplinary research to be conducted on the existing set of data, exploring the linkages between species, their environments and forest-use practices.

The PLANFORBIO project and researchers have invested considerable effort into collating both (i) the Data and (ii) information about the data (Metadata), which has been completed to the ISO19115 standard, and is INSPIRE compliant. With this in mind, initiatives such as the Irish Spatial Data Exchange (ISDE) should be examined to ascertain the potential for the existence of the BIOPLAN (and PLANFORBIO) suite of datasets to be promoted to an international, in addition to national, audience. Compliance with INSPIRE opens the possibility for a pan-European audience for Irish collaborative forestry research, allowing future partners to identify datasets and skills which can form the foundations of future collaborative work, and contact the originators and custodians of the data.

Recommendation 30: A clear strategy and process through which the data holdings and skills resulting from collaborative Irish research efforts in the area of forest ecology and management can be actively promoted to a pan-European audience through use of the archived INSPIRE-compliant metadata should be implemented.

Implications for policy and practice

During the BIOPLAN project a large amount of research was conducted on forest biodiversity in both Irish and British forests in the areas of afforestation site selection, planted tree species, forest management for biodiversity, biodiversity indicators, herbivore management and climate change. The breadth of data collected by this project has practical applications and implications for the management of Irish plantation forests and semi-natural woodlands. The findings provide evidence-based recommendations to underpin forest management policies to protect biodiversity and future-proof our forest estate in the face of climate change. A series of recommendations have emerged from this work which have been peer-reviewed in international publications and are translated here for the purposes of policy formation and practice. Biodiversity conservation in Ireland's forest estate is central to sustainable development in this sector, ecosystem functioning, the provision of ecosystem services and human well-being (Balvanera *et al.*, 2006; Department of Agriculture Food & the Marine, 2010; Bastian, 2013).

A major driver of biodiversity loss is human-induced land use change, and so the impacts of forest expansion on biodiversity are an important consideration in afforestation site selection (Carnus *et al.*, 2006; Brockerhoff *et al.*, 2008). The research on afforestation of grassland habitats described in chapter 1 confirmed the successional processes of early forest development and identified favourable habitats for afforestation of open lands in Ireland. The observed differences in biodiversity between improved and wet grassland sites, and the greater change of biodiversity in improved grassland habitats following afforestation (Fuller *et al.*, 2013a; Graham *et al.*, 2013) have implications for afforestation site selection. Priority should be given to planting on improved grasslands at low altitudes in Ireland, and planting on marginal lands of high conservation importance should be avoided. In practice, this means that trade-offs with farming activities, the common agricultural policy and carbon sequestration need to be addressed through incentive schemes. Improved grasslands and other highly modified land such as arable land are usually the most productive and thus least likely to be afforested (Kearney, 2001; Wilson *et al.*, 2012; Upton *et al.*, 2014). Yet their biodiversity value and their value as a carbon sink is expected to increase after afforestation (Laganière *et al.*, 2010). Prioritising these land types for afforestation will support the government's strategy to increase forest cover, augment the capacity of the national carbon sink and preserve more biodiverse habitats (COFORD Council, 2009; Wilson *et al.*, 2012).

The study of the biodiversity of standard- and double-width forest roads in young forest plantations described in chapter 2 reported biodiversity rich plant, invertebrate and bird species associated with forest roads in plantation forests (Graham *et al.*, 2012; Fuller *et al.*, 2013b). This proves the usefulness of forest roads and corridors for increasing stand scale biodiversity in landscapes with highly fragmented forests and supports the inclusion of forest road networks in forest policy for biodiversity. No advantage in terms of biodiversity was related to doubling of the standard road width by 6 years post-planting, but biodiversity benefits are expected to accrue later in the forest cycle and this issue should be revisited when this data becomes available.

Long-term biodiversity is the cornerstone of forest policy development and during the BIOPLAN project appropriate sites for long-term monitoring were selected and baseline surveys of biodiversity undertaken to pave the way for long-term monitoring of biodiversity in Ireland's forests. Long-term biodiversity monitoring

such as this is essential to provide insights into environmental change and biodiversity conservation at time-frames appropriate to forest ecological processes to support evidence based policy. Analysis of the baseline data, described in chapter 3, provided scientific evidence for the relationship between planted tree species and biodiversity and revealed that different tree species provide habitat for different flora and fauna. This re-emphasises the need for diversity in planting not just at the scale of the forest stand, but also at landscape scale. Further scientific evidence for implementation of forest policy on maintaining diversity of planted tree species was provided by the study of insect biodiversity in different conifer plantations in chapter 4. Higher biodiversity was recorded in plantations of Scots pine than in Sitka spruce or Lodgepole pine forests (Fuller, 2013). The native provenance of this species lends support to the theory that the use of native tree species in forest planting can help to enhance biodiversity. Furthermore, the detailed study of bird communities in oak semi-natural and plantation forests in chapter 8 revealed that plantation oak forests can support bird communities comparable to those of semi-natural oak woodlands (Graham *et al.*, 2014). These findings have significant implications for forest policy in Ireland as plantation forest cover is expanding and this research demonstrates that the planting of a diversity of tree species, including broadleaves and those of native provenance, should be promoted to enhance biodiversity, particularly where exotic conifer plantations do not support biodiversity comparable with native species.

Chapter 5 describes a study of deadwood-reliant fungus gnats which demonstrated that deadwood brash derived from forest thinning and clear-felling operations harbours a wealth of invertebrate biodiversity. The presence of large-diameter deadwood in Ireland's forests is particularly low (Sweeney *et al.*, 2010d), and these findings demonstrate that in order to provide habitat for these species forest management should aim to preserve fine deadwood brash from both thinning and clear-felling (Deady, 2013).

The research on forest biodiversity indicators described in chapter 6 failed to identify forest biodiversity indicators that are robust across different plant and animal species, or across geographic regions. Monitoring of the effectiveness of biodiversity conservation measures requires routine monitoring of forest biodiversity, and these findings have implications for the use of indicators for this purpose. Canopy cover, coarse woody debris, proximity to old woodland and stand age are some of the structural indicators that proved useful for various plant and animal groups (Coote *et al.*, 2013; Fuller, 2013). All of these indicators can be assessed without the need for specialist knowledge, are ecologically meaningful, and applicable to a range of managed forests. They can be used to assess the potential value of stands for the taxonomic groups to which they apply, as well as giving insights into management practices to enhance diversity in these groups. Biodiversity indicators for plantation forests should however, be used with caution as their ability to predict species richness can depend on other factors such as tree species, stand age, land-use history, climate and geographical region. Furthermore, biodiversity surveys for multiple taxonomic groups should be continued as provide valuable information required to inform policy and practice.

The planting of mixed tree species is commonly promoted as a means of increasing biodiversity in forest plantations. Chapter 7 looked at invertebrate diversity in mixed and found that mixed forest plantations did not benefit the species richness of ground-dwelling spiders and carabid beetles compared with monoculture stands across three regions in Ireland and Britain (Barsoum *et al.*, 2014). These findings refer only to ground-dwelling species and are likely due to the similarity of habitat characteristics between the mixed and monoculture stands e.g. high canopy openness and vegetation cover in both mixed and pure stands of Scots pine and oak. Therefore this should not be interpreted as evidence that mixed stands do not benefit biodiversity, but that further, purpose-designed, research is required to determine the importance of

different mixed tree species woodlands for biodiversity across a range of ground- and canopy-dwelling plants and animals. In the interim policy should promote the diversity of planted tree species, both at forest stand and landscape scales.

Because the measurement of total species diversity is not possible, other than at very small scales, an index for assessing the potential biodiversity of woodlands at the stand scale was developed and is described in chapter 9. This index combines a number of biodiversity indicators and worked well in describing the biodiversity potential of native woodlands. Further research and broadening of the scope of this work, particularly in relation to plantations, is recommended to facilitate its adoption as a method for assessing potential biodiversity in Irish woodlands. Where such an index is adopted for assessment of biodiversity potential, it should be supplemented by recording of biodiversity across a range of taxonomic groups.

Modern forest management is more focussed on landscape scale processes than traditional stand scale management (Saura *et al.*, 2011b), and management of landscape connectivity affords us the opportunity to enhance biodiversity as prescribed by international agreements and legislation for biodiversity conservation. Understanding the effects of forest connectivity on biodiversity is essential for the formulation of appropriate forest management plans (Zuidema *et al.*, 1996) and the research described in chapter 10 demonstrated a significant effect of landscape scale connectivity on the biodiversity of forest bird species, but not on generalist bird species or spiders. While further work is needed in this area in an Irish context afforestation policy should aim to promote proximity to areas of native woodland and increase connectivity between new and existing forest habitat patches.

The research described in chapter 11 highlighted the role that wild herbivores play in driving the species composition and woodland structure in both semi-natural oak woods and plantation forests. The grazing intensity in woodland has implications for floristic composition, plant community structure, and tree regeneration (Newman *et al.*, 2014). This is particularly important for regeneration and conservation objectives in woodland management. Future policy should focus on managing wild grazing animals as part of the overall biodiversity objective. This research and that described in chapter 8, show that deer and grazers are important component of woodland habitats. Low levels of wild herbivore grazing pressure are desirable in semi-natural Oak woodlands where the maintenance of plant and bird biodiversity is a priority (Newman, 2013; Graham *et al.*, 2014; Newman *et al.*, 2014). Litter-dwelling invertebrate species were found to be unaffected by herbivore grazing, and derive no benefit from their exclusion (Fuller *et al.*, 2014). Management, rather than exclusion, of herbivores should therefore be the priority and, where fencing is essential it should be limited to a short-term deployment. Qualitative grazing impact surveys, such as the 'Woodland Grazing Toolbox' used here, should be employed to provide quick and cheap assessments of the current grazing levels and also to provide information on yearly and seasonal impact levels.

The conservation requirements of individual plant and animal species change in line with climate and widespread changes are already occurring in natural systems. The scale and extent of changes will continue to accelerate over coming decades and beyond as greenhouse gas emissions are likely to continue unabated. The influence of climate change on the distribution of woodland species in Ireland will have large implications for conservation policy and practice. Given the relatively small land area on which woodlands are currently found, together with the limited dispersal ability of many woodland species, shifts in suitable climate space are likely to drastically change the species composition of many Irish woodlands, and may lead to significant decreases in the populations of some species, including those currently of conservation

concern (Sharkey *et al.*, 2013). Conservation decisions will have to be made based on longer timescales than has traditionally been the case and synergy between carbon sequestration and biodiversity policy goals is essential for climate change mitigation strategies (Rittenhouse and Rissman, 2012).

Conclusions and recommendations

The BIOPLAN project looked into the relationship between forest management and biodiversity. This research clearly identifies ways in which biodiversity conservation can benefit from changes in forest management practices. A number of draft recommendations were developed based on the findings of the research. In order to ensure that these were practice focussed they were subjected to review at a national workshop held in Portlaoise in December 2013. At this workshop the research findings arising from the BIOPLAN project were presented to an audience of forest policy makers, practitioners and researchers where they were then reviewed by focus groups. Following the consultation carried out during this process the following key recommendations are made, subject to the limitations of this project:

1. Avoid planting of forests on peatland and biodiversity rich grassland, and focus new planting on intensively managed and/or structurally homogenous grassland where the opportunities for biodiversity enhancement are greatest.
2. Management of plantation forests for biodiversity enhancement should aim to prevent extensive areas of canopy closure.
3. Manage natural regeneration along forest road verges to maintain the integrity of open space through the forest cycle.
4. Continue monitoring of BIOPLAN study sites (afforestation, forest roads and long-term sites) at five year intervals to determine biodiversity impacts across the forest cycle and to provide benchmark data against which the performance of biodiversity conservation measures can be evaluated.
5. Increase the diversity of tree species in plantations, particularly to include planting of native or lightly shading species, such as Scots pine, to enhance biodiversity.
6. Undertake research to identify forest plantation types that offer the optimal combination of economic and ecological benefits.
7. Preserve fine woody debris in plantation sites after both thinning and clear-fell operations to enhance and maintain biodiversity of organisms that are dependent on it.
8. Broadleaved tree species, particularly native species, should be favoured for planting on sites adjacent to semi-natural woodland. Conifer plantations adjacent to these woodlands should be considered or conversion to plantations of native tree species.
9. Use forest biodiversity indicators with caution, particularly when applying them to forest types or geographical areas other than those in which they have been developed or tested.
10. Use appropriate forest biodiversity indicators across different species and taxonomic groups.
11. Undertake research to identify mixed tree species plantation types that offer the optimal combination of economic and ecological benefits and establish experimental mixed plantations for long-term evaluation.
12. Plantations of native tree species should be planted where biodiversity enhancement is a priority.
13. The suitability of heavily browsed semi-natural oak woodlands for several bird species can be improved through management aimed at promoting the development of a complex understorey.
14. Investigate the addition of further indicators, including woodland area, to the potential biodiversity index in order to improve the correlation between potential biodiversity and the index scores.
15. Further test the potential biodiversity index using purpose collected data covering the full set of recommended indicators in both native woodlands and plantation forests in Ireland and the UK,

using taxa that have not been used to generate the index across a range of spatial scales and forest ages.

16. Where possible plantation forests should include an oak component where biodiversity conservation and enhancement is a priority.
17. Coarse woody debris should be retained and non-crop tree regeneration encouraged in forest plantations.
18. Ensure connectivity to existing forest habitat, particularly to semi-natural woodland and areas of high conservation value for new and existing forests.
19. Improve availability of data on non-forest habitats and landscape features such as roads, buildings and hedges, so that these can be accounted for in future assessments of connectivity.
20. Maintain low grazing levels through management of wild herbivores.
21. Replace large-scale, long-term fencing of broadleaved woodlands with large herbivore management programmes in order to ensure the conservation of diverse woodland ecosystems.
22. Increase the number of taxonomic groups used in biological assessment of management and conservations objectives where stressors to ecosystem functioning, such as grazing, are present.
23. Conduct assessment of grazing pressure in plantation forests to maintain levels of grazing sufficient to promote plant community diversity.
24. Maintain and promote connectivity in the landscape and between woodlands and Natura 2000 sites to ensure that species can disperse to new areas of suitable climate space.
25. Conservation measures for plant and animal species should take into account the projected future distributions of species and the habitats on which they depend.
26. Maintenance and conservation of existing woodlands, including their protection from stressors, should be undertaken to mitigate the negative effects of climate change.
27. The viability of translocation to areas with suitable climate and habitat as a conservation measure should be investigated, especially for species that are currently vulnerable or red listed to avoid extinction.
28. Plantation forests should be stocked with trees sourced from regions with climates currently similar to those projected, in order to maintain productivity.
29. Future research conducted in Ireland of a similar nature to that carried out on the BIOPLAN project should implement a similar GIS strategy involving data collection and storage within the project framework. Data collected should be compatible with archived BIOPLAN data.
30. A clear strategy and process through which the data holdings and skills resulting from collaborative Irish research efforts in the area of forest ecology and management can be actively promoted to a pan-European audience through use of the archived INSPIRE-compliant metadata should be implemented.

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Full list of outputs from the project

Peer-reviewed papers

Graham, C.T., Wilson, M.W., Gittings, T., Kelly, T.C., Irwin, S., Sweeney, O.F. & O'Halloran, J. 2014. Factors affecting the bird diversity of planted and semi-natural oak forests in Ireland. Bird Study, X:X-X.

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* These papers were produced by the BIOPLAN team using data collected during earlier DAFM funded projects.

Papers in preparation

Coote, L., Dietzsch, A.C., Smith, G.F., Iremonger, S., Mitchell, F.J.G., Walsh, A.T. & Kelly, D.L. *In preparation*. The plant community composition of plantation forest road corridors.

Coote, L. & Barsoum, N. *In preparation*. Assessing the broad applicability of a composite index of woodland biodiversity.

Dietzsch, A.C., Coote, L., Wilson, M.W., Graham, C.T., Fuller, L., Gittings, T., Walsh, A.T., Iremonger, S., Oxbrough, A., Smith, G.F., Irwin, S., Kelly, D.L., Mitchell, J.G., Kelly, T.C. & O'Halloran, J. *In preparation*. A comparison of the initial responses of plants, ground-dwelling spiders and birds to the afforestation of open Irish habitats.

Dietzsch, A.C., Coote, L., Smith, G.F., Iremonger, S., Walsh, A.T., Mitchell, F.J.G. & Kelly, D.L. *In preparation*. Tracking the changes in plant biodiversity associated with the afforestation of grassland habitats: is space-for-time substitution a valid method?

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Fuller, L., Barsoum, N., Coote, L., Irwin, S., Kelly, T.C. & O'Halloran, J. *In preparation*. Can biodiversity indicators for plantation forests be applied across a broad range of forest stand types and regions?

Graham, C.T., Wilson, M.W., Gittings, T., Kelly, T.C., Irwin, S., Quinn, J. & O'Halloran. *In preparation.* Biodiversity impacts of land-use change depend on the habitat provided by both the new and former land-use. Submitted to Journal of Applied Ecology.

Graham, C.T., Wilson, M.W., Irwin, S. & O'Halloran, J. *In preparation*. The relative importance of landscape and stand-scale factors in explaining forest biodiversity: a multi-taxonomic approach.

Newman, M., Mitchell, F.J.G. & Kelly, D.L. *In preparation.* Impacts of wild herbivores on the plant community structure and regeneration in ancient oak woodlands.

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Popular articles

Anon, 2011. Forestry tips and advice on offer in Birr. Irish Independent, 5 May 2011.

Coote, L. 2011. Progress on PLANFORBIO. Trinity Centre for Biodiversity Research. Issue 2, September 2011.

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Newsletters

A twice-yearly newsletter was produced to share the findings of this research with interested stakeholders which also served to provide updates on research progress.

Presentations at workshops and conferences (including posters)

Graham, C.T., Wilson, M.W., Gittings, T., Kelly, T.C., Irwin, S., Sweeney, O.F., Quinn, J. & O'Halloran, J. 2013. Differential impacts of afforestation on bird diversity is dependent on previous land use. 6th Irish Ornithological Research Conference, November 23rd, University College Cork, Ireland.

O'Halloran, J. 2013. Biodiversity of planted forests and provision of habitat. IUFRO Task Force on 'Forest Biodiversity and Ecosystem Services' workshop on Biodiversity and Ecosystem Services in Planted Forests Workshop. Estoril, Portugal.

Coote, L., Dietzsch, A.D., Wilson, M.W., Graham, C., Fuller, L., Gittings, T., Walsh, A.T., Irwin, S., Kelly, D.L., Mitchell, F.J.G., Kelly, T.C. & O'Halloran, J. 2012. Indicators of biodiversity in plantation forests. Second international conference on biodiversity in forest ecosystems and landscapes, August 2012, Cork, Ireland. Poster presentation.

*Coote, L., Dietzsch, A.C., Wilson, M.W., Oxbrough, A., Sweene. y, O.F.McD., Moore, K., Martin, R., Irwin, S., Kelly, D., Mitchell, F.J.G., Kelly, T.C. & O'Halloran, J. 2012. Can spruce-dominated plantations support biodiversity equivalent to semi-natural woodlands? International conference on managing forests for ecosystem services: Can spruce forests show the way? October 2012, Edinburgh, Scotland. Poster presentation.

Coote, L. 2012. Plant diversity in Irish plantation forests. Second international conference on biodiversity in forest ecosystems and landscapes, August 2012, Cork, Ireland.

Deady, R. 2012. Biodiversity in Irish plantation forests: Saproxylic invertebrates: brash as a safe haven. Second international conference on biodiversity in forest ecosystems and landscapes, August 2012, Cork, Ireland.

*Dietzsch, A.C., Coote, L., Wilson, M.W., Graham, C., Fuller, L., Gittings, T., Walsh, A.T., Iremonger, S., Oxbrough, A., Smith, G.F., Irwin, S., Kelly, D.L., Mitchell, F.J.G., Kelly, T.C. & O'Halloran, J. 2012. A comparison of the initial responses of plants, invertebrates and birds to the afforestation of grassland habitats. Second international conference on biodiversity in forest ecosystems and landscapes, August 2012, Cork, Ireland.

*Dietzsch, A.C., Coote, L., Wilson, M.W., Oxbrough, A., Sweeney, O.F.McD., Moore, K., Irwin, S., Kelly, D.L., Mitchell, F.J.G., Kelly, T.C. & O'Halloran, J. 2012. Knowing one, knowing all? Comparing the diversity responses of plants, spiders and birds in first and second rotation Sitka spruce (*Picea sitchensis*) plantations across the commercial forest cycle. International conference on managing forests for ecosystem services: Can spruce forests show the way? October 2012, Edinburgh, Scotland.

Fuller, L. 2012. Biodiversity in Ireland's native woodlands: BIOPLAN grazing experiment. Second international conference on biodiversity in forest ecosystems and landscapes, August 2012, Cork, Ireland. Poster presentation.

Fuller, L., Oxbrough, A., Irwin, S., T. C. Kelly & O'Halloran, J. 2012. Impact of experimental road-verge management on ground-dwelling spider diversity in young Sitka spruce plantation forests. International conference on managing forests for ecosystem services: Can spruce forests show the way? October 2012, Edinburgh, Scotland. Poster presentation.

Kelly, D. & Mitchell, F. 2012. Biodiversity in Ireland's native woodlands: yew and oak woodland. Second international conference on biodiversity in forest ecosystems and landscapes, August 2012, Cork, Ireland.

Newman, M., Mitchell, F.J.G., & Kelly, D.L. 2012. Impacts of wild herbivores on diversity, structural composition, and regeneration in ancient oak woodlands. Irish Plant Scientists' Association Meeting (IPSAM), 3rd April 2012, National Botanical Gardens, Dublin, Ireland.

Newman, M., Mitchell, F.J.G., & Kelly, D.L. 2012. Impacts of wild herbivores on diversity, structural composition, and regeneration in ancient oak woodlands. Botany-Zoology Postgraduate Symposium 2012, 16th-17th April, Trinity College Dublin, Ireland.

Newman, M., Mitchell, F.J.G., and Kelly, D.L. 2012. Exclusion of large herbivores: Long-term monitoring of plant community composition in Irish semi-natural oak woodlands. Second international conference on biodiversity in forest ecosystems and landscapes, August 2012, Cork, Ireland.

O'Halloran, J. 2012. Ireland does have forests. The importance of highly managed forest fragments in agricultural landscapes. Second international conference on biodiversity in forest ecosystems and landscapes, August 2012, Cork, Ireland. Keynote presentation.

Wilson, M., 2012. Woodland Forests and Scrub. Bird Habitats in Ireland – a one day symposium. RDS Dublin, May 18th 2012.

*Wilson, M. 2012. Biodiversity in Irish plantation forests: Bird conservation. Second international conference on biodiversity in forest ecosystems and landscapes, August 2012, Cork, Ireland.

Deady, R., Irwin, S., Kelly, T.C., Chandler, P.J. & O'Halloran, J. 2011. What Diptera diversity does thinning debris and clearfell debris support in Irish plantations? ENVIRON 2011, University College Cork.

Deady, R., Irwin, S., Kelly, T.C., Chandler, P.J. & O'Halloran, J. 2011. What Diptera diversity does thinning debris and clearfell debris support in Irish plantations? On Dynamics and Ecological services of deadwood in forest ecosystems, Quebec, May 2011. Poster presentation.

Fuller, L., Irwin, S., Deady. R., Kelly, T.C. & O'Halloran, J. 2011. Can roads be used to enhance forest biodiversity? BES Annual Symposium, 2011, Forests and Global Change, University of Cambridge, March 2011. Poster presentation.

Fuller, L., Irwin, S., Deady. R., Kelly, T.C. & O'Halloran, J. 2011. Can roads be used to enhance forest biodiversity? Environ 2011, University College Cork, April 2011. Poster presentation.

*Larsson, T-B., Barbati, A., Chirici, G., Gardfjell, H., Lombardi, F., Michalak, R, O'Halloran, J. & Sweeney, O. 2011. Deadwood in European Forests. International Symposium on Dynamics and Ecological services of deadwood in forest ecosystems, Quebec, May 2011.

O'Halloran, J., Wilson, M.W., Sweeney, O. F. McD., Kelly, T. & Irwin, S. 2011. Bird communities of native and plantation forests in Ireland: Can plantation forests benefit bird communities? 8th Conference of the European Ornithologists' Union, Riga, Latvia, August 2011. Poster presentation.

O'Halloran, J. Kelly, D., Kelly, T., Mitchell, F.J.G., Giller, P., Iremonger, S., Irwin, S. & Wilson, M. 2011. The environmental challenges facing the Irish forestry industry: Forests and biodiversity. Augustine Henry Forestry Lecture, RDS, March 2011.

O'Halloran, J., Wilson, M.W., Sweeney, O. F. McD., Kelly, T. & Irwin, S. 2011. Bird communities of native and plantation forests in Ireland: Can plantation forests benefit bird communities? Environmental Research Institute Outreach Day 2011. Poster presentation.

O'Halloran, J. Forests: From Biodiversity to Ecotoxicology. Environmental Research Institute Outreach Day 2011.

*Coote, L., French, L.J., Moore, K., Mitchell, F.J.G. & Kelly, D.L. 2010. Can plantation forests support species and vegetation communities typical of semi-natural woodlands? British Ecological Society Annual Meeting, 9th September 2010, Leeds, UK.

Irwin, S., Kelly, D., Kelly, T., McCarthy, N., Mitchell, F., Coote, L., Oxbrough, A., Wilson, M., Martin, R., French, V., Fox, H., Sweeney, O., Moore, K. & O'Halloran, J. 2010. PLANFORBIO: Planning and management tools for biodiversity in a range of Irish Forests. EPA National Research Conference, 23 June 2010, Croke Park Conference Centre, Dublin.

*Kelly, T.C. 2010. Monitoring biodiversity in Irish Forests. Meeting of European National Forest Research Institutes. Dublin, July 2010.

* These presentations were produced by the BIOPLAN team using data collected during earlier DAFM funded projects.

Input to policy and practice development

The findings of the BIOPLAN project were disseminated to interested stakeholders, including policy makers, practitioners and scientific researchers at an-end-of-project workshop in December 2013.

Theses

Fuller, L. 2013. Invertebrate diversity in Irish and British forests. PhD Thesis, University College Cork. 268pp.

Newman, M. 2013. Woodland vegetation change through space and time: Impacts of large herbivores. PhD Thesis, Trinity College Dublin.

Deady, R. 2013. The importance of brash from felling and thinning activity in Irish plantation forests for fungus gnats. MSc Thesis, University College Cork. 136pp.

Involvement in Framework RTDI

We were part of a consortium that submitted a proposal as a Collaborative Project, on Theme 2 (Food, Agriculture and Fisheries, and Biotechology, FP7-KBBE-2012-6-singlestage) answering the topic KBBE.2012.1.2-02. The title of this proposal was *European Coalition for Improving Ecological Services through Enhanced Biodiversity* (EUROVERSITY). Other collaborators on this proposal were Wageningen University (Netherlands), Consejo Superior de Investigaciones Cientificas (Spain), Federal Institute of Agricultural Economics (Austria), University of Novi Sad (Serbia), ISARA Lyon (France), Umwelt Büro (Austria), Institute of Nature Conservation of the Polish Academy of Sciences, Institute of Ecology and Botany of the Hungarian Academy of Sciences, and it was coordinated by the University of Hohenheim (Germany).

Databases and data management

All data collected during this project has been compiled in an easily accessible GIS database as described in chapter 13. All data is compliant to standards outlined by the INSPIRE Directive (2007/2/EC), thus maximising opportunities for future exploitation of this data.

Internet presence

A project website has been published at: www.ucc.ie/en/planforbio/Projects/BIOPLAN. This site is regularly updated and includes the findings of the project. It also provides scientific information and links to other resources on forest management and biodiversity conservation. All project outputs are available to download from the outputs section of the project web page.

Other outreach activities

Members of the PLANFORBIO team were invited to host a display at the Irish Forestry, Woodland & Bio Energy Show at Birr Castle Estate, Co. Offaly on 6th and 7th May. The Irish Timber Growers' Association (ITGA), with support from the Forest Service, hosted the Education and Conference Centre at this show. This provided talks, seminars and information on subjects relating to forestry, woodlands and bioenergy to a varied audience. At the PLANFORBIO stand a number of project posters were displayed and there was a display of plants and invertebrates displaying the range of biodiversity found in our forests. Photographs of common woodland birds were projected onto a screen accompanied by recordings of their calls and songs. The following poster and oral presentations were also made by team members:

- Irwin, S. 2011. Bird diversity in Irish forests.
- Irwin, S. 2011. Biodiversity in Irish forests.
- O'Halloran, J. 2011. Bird diversity in Irish forests.
- Wilson, M. 2011. Bird diversity in Irish forests.
- Hanley, E., Kelly D.L. and Coote, L. 2011. Impacts of grazing on tree regeneration and biodiversity in young forest plantations.
- O'Sullivan, R. and Kelly, D.L. 2011. Biodiversity, light and forest roads.

In order to raise the profile of the research being conducted on this project, the IUFRO (International Union of Forest Research Organizations) Second International Conference on Biodiversity in Forest Ecosystems and Landscapes was hosted at UCC in August 2012. This was the first time that an international conference on forest biodiversity was held in Ireland, and helped to showcase the BIOPLAN research project, allowing Irish scientists, forest managers and policy makers to interact with international experts and attend scientific presentations on cutting edge research in this field. This conference was held from 28th to 31st August 2012. Three days of oral and poster presentations were held at UCC, with a mid-conference excursion, including presentations by Irish researchers, to Killarney National Park. The event was attended by 145 delegates from 33 countries and included 64 oral presentations and 50 posters. A book of abstracts was produced and distributed to conference delegates and is also publicly available on the conference website at www.ucc.ie/en/iufro2012. Twenty papers, including one by the conference organisers were published in a special issue of the journal Forest Ecology & Management in early 2014.