BIOFOREST

Investigation of experimental methods to enhance biodiversity in plantation forests

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BIOFOREST PROJECT 3.1.3 FINAL REPORT, June 2006

COFORD NATIONAL COUNCIL FOR FOREST RESEARCH & DEVELOPMENT

epa Environmental Protection Agency

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EXECUTIVE SUMMARY

INTRODUCTION

This project is a component of a three-part project with the umbrella name BIOFOREST, funded by the National Development Plan through COFORD and the EPA. The three organisations conducting the work were University College, Cork, Trinity College, Dublin and Coillte Teoranta. The aim of this project was "*To identify those forestry management practices (with the possibility of using experimental plots) which are best suited to maintaining and enhancing biodiversity in plantation forests*". The focus of the project was open space within Irish forestry plantations. The project involved a literature review, a field survey of open spaces of different sizes and shapes in closed canopy Sitka spruce (*Picea sitchensis*) plantations and the establishment of an experiment on the manipulation of open space associated with roads within forests to enhance biodiversity.

LITERATURE REVIEW

A review of literature relating to the main factors affecting biodiversity in forests with relevance to Irish plantations was conducted. This review reports on the key elements that have been identified through international processes directing sustainable forestry practices, and scientific research underpinning the methods by which forest biodiversity is fostered or enhanced. Open space was identified as a practical focus for the research conducted during this project, and emphasis is placed on this aspect of forests and their diversity in the review. Strategies to enhance diversity through open space management are presented. The use of management spaces already present (e.g., roads and rides) is described, as well as the effect of orientation and aspect. The use of natural open areas is explored, such as retained unforested habitats. The creation of new spaces is also reviewed, as well as opportunities to alter plantation design at replanting or through change of management from clearfelling to continuous cover. Results of a survey to review policies on the management of open space in other countries are presented, and the regulations regarding open space in Ireland are discussed.

DATABASE

All data gathered during this project are stored in a custom-built fully interactive and user-friendly database system. The system also stores data from the other sub-projects of the entire BIOFOREST Project. The database is constructed in a geographic information system, enabling the user to access data through site locations. It includes relevant background data such as Ordnance Survey maps, forest cover distribution (Forest Service FIPS), officially designated conservation areas and CORINE land use data. Documentation is provided for all datasets using ISO compliant standards. BIOFOREST data are included in the different taxonomic categories: terrestrial vegetation, epiphytes, spiders, hoverflies and birds. Site history and management data are also included where dissemination of this information is permitted.

TERRESTRIAL VEGETATION

Terrestrial vascular plants, bryophytes and lichens were surveyed in a series of 151 4m^2 plots in 60 glades, rides and roads, and a complete species list was made encompassing each open space. Data were collected on several environment and management variables, and the light regime of open spaces was measured using hemispherical photography, calibrated for climatic and topographic conditions. We found that vegetation composition and diversity were primarily influenced by soil and climatic factors. The vegetation composition of roads was different to that of glades and rides, mainly because of the diversity of topographic features in roadsides and the influence of the gravel road surface. Solar radiation was an important factor influencing vegetation composition and diversity. Glades and wide rides supported species preferring open conditions, whereas shade-tolerant species, especially bryophytes, were characteristic of narrow rides. With increasing light levels, vascular plant species richness tended to increase while species richness of bryophytes and lichens tended to decrease. Vegetation evenness and Simpson's diversity were lower both in well-lit, grass-dominated situations and in heavily shaded, bryophyte-dominated conditions. Within glades, terrestrial bryophyte and lichen species richness and vegetation diversity increased from glade centres towards the forest edge. Roadside topographic features and contrasting surfacing material (i.e. limestone gravel vs. local sandstone or mica-schist stone) were responsible for much of the variation within and among road sites. The proportion of open space in the plantation as a whole had little influence on the vegetation of the open spaces surveyed. To increase the biodiversity of open habitat flora, rides and roads should be wide enough so that at least the centre is well-lit. Increasing the edge to area ratio of glades will increase the amount of ecotonal habitat between the glade and forest, and will probably increase vegetation diversity. Disturbance of roadside vegetation should be minimised during maintenance operations to maintain the diversity of roadside habitats.

EPIPHYTES

The main groups of epiphytes are in Ireland are mosses, lichens and liverworts and they have been found to constitute a major component of the total botanical diversity of semi-natural woodland in Ireland. However, there is no known information on epiphytes in Irish forestry plantations. This study investigated the diversity of epiphytes on trees adjacent to open spaces within commercially mature Sitka spruce plantations. Pairs of trees, one located at the north (south facing) edge of an open space and one in the forest interior, were studied at 12 sites. Epiphytes were studied in plots on the trunk and branches, at different heights and on the south (side facing the open space) and north sides of the trees. The main factors influencing epiphyte biodiversity were site elevation and tree density. Epiphyte species richness was negatively associated with elevation and positively associated with tree density. The main effect of open spaces on epiphytes observed in this study was an increase in the cover of bryophytes on the south side of the edge trees compared to the north side of the same trees and the south side of the interior trees. This was mainly related to the presence of live side branches over the entire height of the south side of the edge trees which appeared to shade the trunk and increase humidity levels. These live branches also formed a dense side canopy which may have closed the edge to light and air, and prevented the open spaces from affecting the epiphyte diversity of the adjacent trees. Further research is required before any recommendations regarding the management of open spaces for epiphyte biodiversity can be made.

SPIDERS

Sustainable forest management advocates the retention or creation of open space within plantation forests to enhance biodiversity. However, the biodiversity value of these open spaces will depend on the habitat type chosen, as well as open space size and shape. The present study investigated ground-dwelling spider assemblages in glades, rides and forest roads of various sizes in 12 mature Sitka spruce plantations across Ireland. Spiders were sampled along a transect from the open space into the forest using pitfall traps. Species richness and abundance declined along the open space - forest transect with the open space supporting a unique spider fauna, absent within the forest. Total species richness and richness of species associated with open habitats were significantly greater in the glades. There was a significant positive relationship between species variables and ride/road verge width and roads and rides <15m wide did not support an "open" spider fauna due to the influence of the canopy. No such "threshold" area was found for glades, probably because few small glades were sampled. Open space habitat type is an important determinant of spider assemblage structure, although open spaces with high shrub cover or unplanted broadleaves did not maintain a unique spider fauna vis-à-vis that supported within the plantation. At a large scale the total amount of open space within 200m of sampling plots was positively correlated with species richness and abundance. Forest management plans should encourage the retention of a range of habitat types in non-linear open space (glades), whereas the biodiversity value of linear open space (rides and roads) will be enhanced if wider than 15m.

HOVERFLIES

We used hoverflies as an indicator group to assess the role of open spaces in maintaining biodiversity within Sitka spruce plantation forests. We set out to determine the factors affecting hoverfly diversity in open spaces in plantation forests, and to make recommendations for planning and management of open spaces in plantation forests to enhance biodiversity. The majority (nearly 80%) of the species we recorded are associated with open space habitats rather than closed-canopy forest. The species richness of the fauna associated with large open spaces was slightly, but significantly, higher in unplanted glades compared to forest roads. The species richness of the open space fauna was positively correlated with forest road width but did not show any relationship with overall amounts of open space within the plantation. Species with larvae feeding on the foliage of trees and shrubs were associated with the presence of broadleaved woody vegetation. Species with larvae developing in surface water habitats were associated with wet habitat features. Planning and management for hoverfly biodiversity in Irish conifer plantations should focus on the open space component, and should encourage broadleaved trees and shrubs and wet habitat features. Wide forest roads and/or unplanted glades should be included to allow the maintenance of well-developed open space habitat in mature spruce forests.

BIRDS

Forest Birds

Open spaces are one of the most important contributory factors to bird diversity in plantation forests. We surveyed the birds occurring in and around a variety of open space types in twelve Sitka spruce plantations in Ireland. Shrub cover and cover of non-crop broadleaved trees appeared to have the greatest influence on bird diversity of any of the environmental variables we investigated. These variables were correlated with bird species richness at all scales we considered, from the level of stand or individual open space to that of the plantation. The relationship between bird diversity and these variables is largely due to the higher abundance and frequency of occurrence of a suite of relatively uncommon species associated with those gaps in the plantation forest canopy with high levels of broadleaved tree and shrub cover. Management for diverse bird communities in Irish plantation forests should focus on the creation of more and wider open spaces in which shrub and non-crop tree cover can develop, both in and around forests. In forests that are over-grazed by deer, control of deer populations may be necessary to achieve this.

Hen Harriers

The Hen Harrier (*Circus cyaneus)* was chosen for a special study because it is an open habitat species of high conservation concern whose distribution coincides with areas where commercial forest cover is typically high and increasing. Also, this species is the only rare Irish bird that occurs in areas with extensive forest cover whose breeding distribution is accurately known. Hen Harriers are a protected bird species under European law, and one of the birds of greatest conservation concern in Ireland. In recent decades, large tracts of Hen Harrier habitat in the Irish uplands have been afforested. Hen Harriers nest and forage in young plantations, but closed canopy forests are not used extensively by this species. The suitability of Irish plantation forests for Hen Harriers therefore depends on their age structure.

Using the results of a recent national survey, the NPWS outlined ten Indicative Areas for Hen Harriers (IAs). These cover 3.4% of the area of the Republic of Ireland, and at the time of the survey supported roughly 75% of the Irish Hen Harrier population. Hen Harriers were ten times less likely to occupy ranges in the IAs with less than 30% suitable habitat cover (within 1 km of their nest sites), than they were to occupy areas with more than 30% suitable habitat cover. Using the 30% habitat threshold, the proportion of the IAs that is unsuitable for Hen Harriers will increase from about 30% (at the time of the Hen Harrier survey in 2002) to about 50% by 2015. The persistence of Hen Harriers in some areas may depend critically on the value of young second rotation forests, relative to young first rotation forests and open habitats such as bog and heath.

Land-use change that reduces the cover of habitat in which Hen Harriers breed and hunt to less than 30%, within any area of the IAs of radius 1 km (approximately 3 km^2), should not be permitted. To facilitate decision-making over large areas like the IAs a custom-designed GIS is required, containing accurate, up-to-date information on land-use and habitats within these areas. There is a need for more detailed analysis of habitat in the areas where Hen Harriers occur, and for an improved understanding of Hen Harrier habitat requirements.

EXPERIMENTAL MANIPULATION

Strips of open spaces adjacent to forest roads can make a significant contribution to the biodiversity of forestry plantations. The extent of this contribution is partly dependent on the width of these unplanted strips. The possibility of using these strips as a focus for an experimental manipulation to be set up during this project was decided following a discussion session at the conference “Opportunities for enhancement of biodiversity in plantation forests”, 24 October 2002, Vienna Woods Hotel, Cork. This was attended by members of the BIOFOREST Steering Group and individuals from forest-related institutions both inside and outside of Ireland. Further exploration of the practicalities involved and possible sites focused the experiment on this subject.

The recommended between-trunk clearance across forest roads is currently 15m, with approximately 5m being the road surface and the other 10m being divided between the two sides of the road, leaving an average of 5m on each side. Branches tend to directly shade at least 2.5m of this, and an amount of the space is also used for positioning of drains and banks. Together with the shade from the maturing trees, there is little undisturbed open space on either side that is unshaded. The Research Group proposed to investigate the effect of doubling the clearance on the biodiversity of the area.

It is intended that this experiment will be maintained beyond the life of the BIOFOREST project and that the sites will be re-surveyed periodically. As such, the ownership of the sites was important, and therefore the project was restricted to using sites owned by Coillte. Study sites were chosen from several that were scheduled to undergo re-establishment (planting after harvesting) in 2004/2005. Plantations dominated by Sitka spruce were the focus of the experiment. In sections of forest road within these plantations two treatments were established: in the normal treatment trees were planted on either side of the road with a 15m clearance across the road between trunks; in the wide treatment, trees were planted with double clearance, i.e. 30m between trunks.

Baseline surveys were carried out during the summer of 2005 on vegetation, spiders, birds and hoverflies. Sorting and identification of specimens ensued, and the baseline data are included in the BIOFOREST database.

CONCLUSIONS AND RECOMMENDATIONS

The relevant features of open space for biodiversity in forest plantations are identified. For each feature we discuss the existing regulatory requirements, briefly summarise the relevant results from our research and then discuss the implications of these results. We make nine recommendations about forestry management practices that can influence the features and identify any modifications that may be required to the *Forest Biodiversity Guidelines*. All of our recommendations are made with the caveats presented in Section 10.2. Recommendations for further research are also identified.

The nine recommendations for design and management are as follows. Asterisks indicate where changes to the *Forest Biodiversity Guidelines* are needed:

1. Open spaces should be promoted in forest plantations as a method of biodiversity enhancement.
2. Recommended lower limits of 5-10% open space in forests needs to be reviewed with a view to increasing it.*
3. With consideration for forest fragmentation and other possible adverse effects, some areas of forest road should have a clearance substantially wider than the recommended 15m. This may be achieved by developing scalloped edges.*
4. Rides of standard width (6 m wide) should not be included in the 5-10% open space requirement. A minimum width (probably 15m) should be specified for rides or other linear open spaces to qualify for inclusion in the 5-10% open space requirement.*
5. A minimum glade size of 225m^2 should be specified for glades to qualify for inclusion in the 5-10% open space requirement, and larger glade sizes (at least 625-900m^2) should be encouraged.*
6. The protective zone around retained habitats should be at least 7m (on each side) for linear features such as hedgerows, treelines and small streams (not covered by the *Forestry and Water Quality Guidelines*), to ensure that they do not get shaded out as the plantation matures (the current recommended width is 3m). A mandatory minimum protective zone of 7m should be required for linear features in order for them to qualify for inclusion on the Area for Biodiversity Enhancement.*
7. The *Forest Biodiversity Guidelines* should be more specific on how to encourage shrub and non-crop tree patches/stands in plantations.*
8. The *Forest Biodiversity Guidelines* should emphasise the importance of small wet habitat features that are not mapped on Ordnance Survey six-inch maps, recommend that these be include in the Area for Biodiversity Enhancement, and provide specific guidance to help foresters to identify these features.*
9. Grazing pressure will need to be managed if broadleaved tree and shrub vegetation to develop in areas where open spaces within forests come under heavy grazing pressure. More information is needed on impacts of grazing.

The recommendations for further research are:

1. Research is required into the value for biodiversity of open spaces within forests in the habitat mosaic that occurs in lowland agricultural landscapes when significant proportions are afforested.
2. The effect of different forest types on the contribution of the internal open spaces to overall biodiversity should be further investigated, focusing on taxa that are likely to have distinct forest edge assemblages. Specific hypotheses about how variation in forest edge structure affects these taxa should be tested. The biodiversity of internal open spaces in semi-natural woodlands should also be determined to facilitate a comparison of this with that of plantation forests.
3. The importance of retained habitats within forests should be investigated with a view to drafting management guidelines for these to promote the overall biodiversity of the forest.
4. The potential significance of grazing (mainly by deer) as an influence on the biodiversity of open spaces in plantation forests needs to be researched.
5. This study recorded biodiversity in open spaces by investigating certain animal and plant groups. Other groups may have different relationships with open space and should be studied. Priority groups are suggested.

ACKNOWLEDGEMENTS

The authors would like to thank the following for their work at various stages of this project:

COFORD and the EPA, the bodies that funded this research, without which this work would not have been possible. In particular we thank Dr Eugene Hendrick, Joe O'Carroll and Lauren MacLennan of COFORD and Helen Walsh, Loraine Fegan, Dr Conor Clenaghan, Dr Karl Richards and Garret Kilroy of the EPA.

Coillte establishment managers Pat O'Brien, Dick Kelly, Jim O'Driscoll, Eamonn Drohan, Louise O'Reilly and John O'Mahoney for facilitating the setting up of the manipulations experiment.

Members of the BIOFOREST Steering Group, who made excellent observations and truly steered the project at the start when the details were being decided. They are: Prof. Tor-Björn Larsson, European Environment Agency; Prof. Jari Niemelä, Helsinki University; Dr Jonathan Humphrey, Forest Research, UK; Dr Allan Watt, Centre for Ecology and Hydrology, Banchory, UK, Noel Foley, Forest Service, Ireland, Dr John Cross, National Parks and Wildlife Service, Ireland.

The botanical teams thank Deirdre Ninaber, Siobhán McNamee and Bastian Egeter for assistance with fieldwork, Dr Laura French for help with sample identification and database management in the manipulations experiment and Dr Philip Perrin for help with planning the manipulations. We are grateful to T.H. Blackstock, Dr S. Bosanquet, Dr D.F. Chamberlain, Dr H. Fox, Dr B. Coppins, Dr M.O. Hill, Dr D.T. Holyoak, Dr P. Oksanen, R. Porley and G.P. Rothero for identification of difficult specimens.

The invertebrate teams would like to thank Blathnaid O'Loughlin, Julianna O'Callaghan and Eleanor O'Brien for their help with fieldwork and sample sorting. We also thank Dr Martin Speight, Robert Johnston and Dr Peter Merrett for verification of doubtful species.

The bird team sincerely thanks David Norriss of the National Parks and Wildlife Service, Stephen Newton of BirdWatch Ireland, and Kevin Collins and other members of the Irish Raptor Study Group, who collected and made available data on Hen Harrier nest sites from 1998-2003. Thanks to the British Trust for Ornithology for making available historical data on Hen Harrier distribution in Ireland from the two Breeding Atlases.

Thanks to Coillte Teoranta and to the Forest Service for granting access to the information in their geo-referenced forest inventories.

The GIS work, which spanned all disciplines, was supported greatly by the input of Dr Mark Wilson, and the individual discipline teams thank him for his input. Many thanks also to Helen Bradley (Geography, UCC) for help with some of the trickier GIS work.

1 INTRODUCTION

This project was one component of a three-part project with the umbrella name BIOFOREST. The overall BIOFOREST project was a large-scale project running from 2001 to 2006 with the aim of addressing some of the gaps that exist in the current information on biodiversity in Irish plantation forests. The project is funded from the National Development Plan funds through the Environmental Protection Agency (EPA) and the National Council for Forest Research and Development (COFORD) as part of the Environmental RTDI Programme 2000-2006. The three sub-projects were:

- **Project 3.1.1.** **Biodiversity assessment of afforestation sites**
- **Project 3.1.2** **Assessment of biodiversity at different stages of the forest cycle**
- **Project 3.1.3** **Investigation of experimental methods to enhance biodiversity in plantation forests (this project)**

The BIOFOREST research team is constituted from the following organisations:

- **Department of Zoology, Ecology and Plant Science and Environment Research Institute (ERI), University College, Cork (UCC)**
- **Department of Botany, Trinity College, Dublin (TCD)**
- **Coillte Teoranta, The Irish Forestry Board (Coillte)**

The Coastal and Marine Resources Institute, University College, Cork, provides expertise on database construction and management. This consortium brought together a team of researchers and partner organisations that have extensive experience in ecology, biodiversity assessment and forest biodiversity studies across a broad spectrum of botanical and zoological groups. The individuals involved in each team are listed in Appendix 1, as are the functional groupings for research, guidance and management.

Project 3.1.2 concluded first, and in the interests of avoiding repetition the final report from that project (Smith *et al*. 2005) will be referred to instead of duplicating information in the current report. Smith *et al*. (2005) give much of the background context to plantation forest biodiversity research in Ireland, and the reader is referred to that document.

1.1 PROJECT OBJECTIVES

The objective of Project 3.1.3, the current project, as stated in the COFORD/EPA scoping document, was:

To identify those forestry management practices (with the possibility of using experimental plots) which are best suited to maintaining and enhancing biodiversity in plantation forests.

The first task for the Research Group was to carry out a review of methodologies used to enhance biodiversity in plantation forests, to inform the further design of the field phase of the project. The different options open to the Group were discussed at a special session during the conference "Opportunities for enhancement of biodiversity in plantation forests", 24 October 2002, Vienna Woods Hotel, Cork. This was attended by members of the BIOFOREST Steering Group and individuals from forest-related institutions both inside and outside of Ireland, who had useful advice (see Appendix 2). A decision was made that this project should focus on the use of open space in forests for biodiversity enhancement.

Peterken (1996) identified the treatment of open spaces as being the single most important factor in the success or failure of nature conservation within plantations. The distribution, composition and management of open space within forests is a factor that is acknowledged to be important by the requirement under the *Forestry Biodiversity Guidelines* (Forest Service 2000b) for 15% open space and "retained habitats" in new plantations. It is a factor that is amenable to intervention for biodiversity enhancement, both at the forest planning stage, and through the subsequent treatment of the open space during the forest cycle. Management of open space, following its incorporation into a forestry plantation, can affect its biodiversity (Humphrey & Patterson 2000). Therefore, research on the biodiversity of open space in plantations would contribute significantly to biodiversity enhancement of plantation forests.

As there were only resources available to study one forest type in this project, and for reasons laid out by Smith *et al*. (2005), forests dominated by Sitka spruce (*Picea sitchensis*) were chosen as the subject. It was subsequently agreed that the project should proceed in two phases:

- An extensive survey, which would entail an examination of forests with different configurations of open space
- The establishment of an experiment on the manipulation of open space in the forest, focusing on roads.

1.2 SEQUENCE OF PRESENTATION WITHIN THIS DOCUMENT

This document is divided into three main sections, a literature review, the extensive survey of open spaces and the manipulation experiment. Within the second part, which constitutes most of the document, are presented the studies within the different taxonomic disciplines. Conclusions and recommendations form a final chapter, and draw on the information and expertise gained during the entire work of the project.

A focus study on a bird species of conservation concern was incorporated into the planning stages of Project 3.1.3, as it was decided that rare birds would be encountered infrequently, if at all, in BIOFOREST study sites. The Hen Harrier was chosen for the reasons outlined below. The special study was submitted in 2005 as a stand-alone report to COFORD and EPA (Wilson et al. 2005):

1. This species is of high conservation concern, being on the Irish Red Data list, and also on Annex 1 of the EU Birds Directive.
2. The Hen Harriers is essentially a species of open habitats, and while they forage and nest in young forest plantations, they do not use mature forests.
3. Nearly all areas where Hen Harriers now breed have experienced extensive afforestation in recent decades. Economic pressure to continue forestry in many of these areas remains high.
4. A national survey of Hen Harriers was completed in 2002, resulting in better data on breeding distribution than was available for any other rare bird species associated with Irish plantation forests.

2 REVIEW OF METHODOLOGIES TO ENHANCE BIODIVERSITY IN PLANTATION FORESTS THROUGH OPEN SPACE MANAGEMENT

2.1.1 Introduction.

The promotion of biodiversity in plantation forests demands consideration of all organisms that live in them: from the larger mammals which roam and use large areas to the invertebrates and plants that are more sedentary (Hunter 1990). In natural forests some organisms require closed canopy conditions while others are adapted to more open habitats created by natural processes such as extreme weather events, fires, windthrow and tree disease (Kirby 1992). In Europe there has been an increase in the overall area of forest since the 18^{th} and 19^{th} centuries, but this has been associated with continued loss of forest plant diversity (Grashof-Bokdam 1998). The reasons for this have been changes in the nature of the forests and the habitats. Specific impacts on forested landscapes include the conversion from mixed broadleaf to uniform coniferous forest and intensive drainage and fertilisation of agricultural land surrounding fragmented and small forest areas (Spiecker 2004; Wulf 2003).

A number of management methods to enhance biodiversity in plantation forests have been suggested, mainly through extending the features that are associated with areas of higher diversity. These have been identified during the international processes developing criteria and indicators for sustainable forest management (MCPFE 2003). The main actions that have been identified include:

- Diversifying the tree species planted.
- Increasing the amount of deadwood, particularly of large diameter.
- Retention of semi-natural or native habitats.
- Including over-mature (old) trees.
- Increasing the amount of open space.

At the start of the BIOFOREST project the Steering Group, composed of a range of diverse scientists from Ireland and abroad (see Appendix 1), took a decision that open space management would be the specific focus of this sub-project (3.1.3) to enhance biodiversity and, specifically, this review. The review was conducted using two approaches: a review of literature dealing with biodiversity enhancement with regard to open space management in forests and a survey of those agencies in a number of countries dealing with forest management policy regarding practices and regulations in use. In both cases an emphasis was put on the management of spaces generated by roads in forests, as this was the specific focus of the Manipulation Experiment set up during this project (see Section 9).

2.1.2 The importance of open space to forest biodiversity

The spatial and temporal characteristics of gaps in forests vary greatly. Their structure and distribution, as well as their temporal characteristics, impact on the biodiversity of a gap and affect the biodiversity of the surrounding forest. Factors include gap size, shape, distribution and the pattern made within the forest, in addition to temporal considerations such as gap age and corresponding ages of adjacent gaps. An overriding factor in these considerations is the nature of the forest itself, in particular whether it is composed of even-aged stands or is a more natural multi-aged entity. In regions where there is a relatively high cover of natural forest, such as Eastern Europe and the North-Western North America, an increase in the frequency and size of gaps has been found to correspond with a decrease in bird diversity

(Boncina 2000; Chambers *et al*. 1999). Moorman, (2001) and Greenberg (2001), however, have documented the opposite for other forest types.

In Irish plantation forests by far the most widespread forest type is that of one or more even-aged stands, none of which in general exceeds 55 years (Joyce 2002). While the management of open areas in forests has been promoted as a means by which to enhance biodiversity, studies documenting the biodiversity of these areas and their management effects are scarce (Bouget 2004). Documented effects can be ambiguous in their utility at a management level, in particular because what may benefit one species may exclude another. In the cases of Irish birds, for example, forest specialists are few (Smith *et al*. 2005) and therefore the inclusion of species attracted by open space may add very significantly to the diversity of these in the forest area. For plants, in contrast, there are many species that occur under the shade of forest cover (see Section 4.4.1), and open space can exclude these. The BIOFOREST study will provide data on a number of taxonomic groups that should facilitate a more complete analysis of the relationships between biodiversity and open space.

Ratcliffe and Peterken (1995) suggested methods for enhancing biodiversity in British spruce forests based on mimicking natural processes that occur in native forests. However, as British spruce forests (and Irish spruce forests) are composed of non-native species, there is no obvious native forest type upon which to model their management. Instead, Ratcliffe and Peterken (1995) suggested that aspects of four systems be taken into account, including the natural forests of north-west Europe and North America, where the species planted have their origins. Open space figures significantly in the maintenance of a good level of biodiversity in these systems (Peterken 1996). According to Zackrisson (1977), open space associated with mires, riparian strips, crags and treelines contributes in the region of 15% of the total forest area in Swedish boreal forests. However, the combination of short rotation lengths and large coupes (5-100ha) have no apparent parallel in comparable natural forests where a mosaic frequently exists from fine-grained to stand replacement (Quine 1999).

The structure and dynamics of plantation forests under a clearcut regime are therefore different from the natural forests that Ratcliffe and Peterken (1995) consider to be the most appropriate comparisons with British plantation forests. The associated flora and fauna of plantation forests are restricted to those species which can access and survive in these systems (Humphrey & Peace 2003). For example, ground layer plants which are predominantly clonal would have a greater chance to spread in natural systems than in short-rotation predominantly single-species forests (Wulf 2003). Smith *et al.* (2005) showed that the animal and plant assemblages that occur in plantation forests in Ireland change significantly over the growth cycle of the forest. One of the main factors that influences these changes is forest structure (Hansen *et al*. 1991; Smith *et al*. 2005); Sparks *et al.* (1996) recorded similar changes in even-aged coniferous plantations, such that particularly the ground layer communities may be adversely affected by shade created by maturing tree crops. Many species of former open areas disappear: rich invertebrate assemblages dependent on ground layer vegetation no longer have a suitable habitat in the dense shade. Foodplants for butterfly larvae and nectar sources for adults are generally associated with low to moderate shade conditions. Jukes and Peace (2003) recommended the inclusion of open spaces in forests particularly to enhance invertebrate biodiversity.

2.1.3 Strategies enhancing biodiversity

One of four main strategies for enhancing biodiversity in Britain's forests is: protecting incorporated semi-natural open habitats, and linking these to permanent open space networks along with open habitats on restock sites and beyond the forest (Hodge *et al*. 1996). Management intervention that promotes light penetration can improve the biodiversity complement significantly by providing open habitats that support species excluded from dark, closed-canopy forests (Mullen *et al*. 2003a). Forest rides provide a refuge for the range of communities typical of a site before planting (Sparks *et al*. 1996). By cutting back rows or blocks of trees along forest rides, gradients of light, leaf litter and disturbance are created which result in zonations of shrubs, field layer vegetation, open ground and grassland communities (Buckley *et al*. 1997). Some of the species these diverse ride-edges contain may have the potential to invade into the forest interior if conditions become more suitable later in the rotation. Grasses were found to typically occupy the edges of linear gaps, while annuals, short-lived perennials and ruderals occurred in the cut zone and common woodland perennials were found at the canopy edge or beneath it (Buckley *et al*. 1997). Thinning of trees along edges or towards the forest interior facilitates the spread of gap species into the forest itself (Anderson & Buckley 1991).

Aspect and orientation of linear gaps such as rides and roads affect the communities in them, such that proactive management for particular characteristics can favour particular assemblages. Ferris and Carter (2000b) and Ferris-Kaan (1995) show very effectively that the aspect and orientation of a gap should be rigorously reviewed at the design or planning stage for the gap (road, ride, clearing) because these dramatically impact on the resulting light regime of the gap. In the UK the south-facing edge of an east-west road or ride will get perhaps 10 hours of sunshine, whereas the north-facing edge may get 1.5 hours. The thinning of trees on the south-facing side will therefore have more effect on the penetration of sunlight into the forest than thinning on the north side (Ferris & Carter 2000b). Computer modelling can be used to optimise gap design for wildlife benefits (Ferris-Kaan 1995).

2.1.3.1 Creation of gaps

In natural forests, disturbance creates gaps of various sizes and shapes through time to form a dynamic occurrence of new gaps and a closing of old gaps. Random periodic disturbances can maintain high species richness (Connell 1978; Huston 1979), and a pattern of gaps at a landscape scale ensures a continual turnover of different successional stages. This provides niches for early successional habitat and light-demanding species groups (Greenberg & Lanham 2001; Kuusela 1990; Uutera *et al*. 1996). Continuity of the presence of open space animals and plants in the forest maintains the diversity of these spaces. The creation of gaps stimulates germination from the (persistent) seed bank of light demanding species that usually grow at forest edges and in clearings (Wulf 2003). Newly formed gaps isolated from a diverse source of colonisers will have lower biodiversity, especially of groups with restricted mobility and ability to disperse, than those to which propagules of open space species are available. Gap characteristics such as the presence of decaying large-diameter deadwood from tree falls promote biodiversity in certain groups, for example fungi and bryophytes (Ferris *et al*. 2000a; Ferris *et al*. 2000c).

Gap creation and infilling is part of the integral ecosystem dynamics of a natural forest, and biodiversity is linked to these processes (functional biodiversity, see Carnus *et al*., (2003). The availability and diversity of open space in plantation forests is often far less than in natural forests, and every effort should be made through forest design to maintain and recreate open space habitats (Ratcliffe & Peterken 1995). The highest species richness in Norway and Sitka spruce plantations in the UK was recorded during (a) the pre-thicket and (b) the over-mature stages: the stages at which light penetration is greatest (Humphrey *et al*. 2002). Although disturbance plays a major role in the penetration of light in natural forests, Quine *et al.* (1999) caution against applying a model of the pattern of disturbance in natural spruce forests to spruce plantations in Britain. The main reason for this is that climatic differences between Britain and more boreal climate zones are significant (temperature, cloud cover), so ecological processes prevailing in the two areas may be quite different, even though the dominant tree species is the same. Natural disturbance caused by fire is more frequent in boreal forests, whereas that caused by windthrow is more common in temperate forests

(Peterken 1996). The characteristics of gaps caused by these two different phenomena are quite distinct: for example windthrow gaps generally contain a significant amount of coarse woody debris (Bouget 2004). This debris attracts colonisation in particular by invertebrates and lower plants. Windfall gaps have been shown to have high diversity of saproxylic organisms (Alexander 1995), and the combination of sunlight with the presence of coarse woody debris and sunlight seems to favour specific species assemblages (Kaila *et al*. 1997; Sippola *et al*. 2002). Also, windthrow events in temperate forests tend to result in many small gaps, rather than few large ones (Faccio 2003; Greenberg & Lanham 2001; Lundquist & Beatty 2002). This has consequences for the range of open habitat and early-successional forest species that can occupy such gaps (Annand & Thompson 1997; Moorman & Guynn 2001).

Quine *et al.* (1999) propose that disturbance in natural forests can be used as a reference point for forest management rather than a model that should be mimicked. On a practical level, Kerr (1999) suggests that the size and frequency of clearcuts in British forestry plantations should be varied to maintain a diverse pattern over the landscape rather than focusing on the stand level. Oliver (1992) advocates a landscape-level plan that incorporates a dynamic balance of diverse forest structural stages. Although this approach focuses mainly on coupe size, gaps of the size caused by single tree falls (c. 100m^2) can be taken into account. Forests should therefore be harvested at a range of different scales (Delong & Tanner 1996; Hunter 1990).

2.1.3.2 Continuous forest cover

A movement has gained momentum over recent years to "transform" forests from a system of large clearcuts to systems promoting continuous cover, such as the "shelterwood system" (Pommerening & Murphy 2004; Spiecker 2003; Spiecker 2004). Forests can be managed to have structural characteristics that are more supportive of diverse species complements. In Poland the transformation to continuous cover forests in some areas is seen as imperative after dramatic forest decline (Malek 2004). The transformation of even-aged beech forests in experimental areas has resulted in greater species diversity of trees and other biota, and an increased diversity of gap sizes (Madas 2004). Although the transition to continuous cover forestry was seen as possible and economically viable, it needed two essential supports: (a) training for foresters and (b) establishment of demonstration plots to convince private forest owners that the system is a viable and favourable alternative (Ferris-Kaan 1995; Madas 2004).

A major advantage of continuous cover forestry has been identified as the facilitation of natural regeneration. The dense shade cast in spruce plantations was found to inhibit natural recruitment of seedling trees: more light would have to reach the forest floor for the trees to regenerate successfully (Hale 2001). To counteract this it would be possible to stock sites at levels that would yield lower volumes of wood than current standard stocking levels (Hale 2001; Page *et al*. 2001), but this would have economic implications for the forest owner. This is particularly so in Britain and Ireland, where high levels of cloud cover and consequent low levels of solar radiation increase the gap requirements of young, growing trees (Malcolm *et al*. 2001). For example, Sitka spruce and Douglas fir are relatively intolerant of shade, needing quite large gaps to regenerate in Britain and Ireland. Gap size requirements may be expressed as the ratio of a gap's diameter to the height of the surrounding trees, and regenerating Sitka spruce and Douglas fir require gap ratios of between one and two.

2.1.3.3 Patch shape

The diversity of habitat created by a combination of irregular shape and broad geographic distribution of forests promotes associated species diversity (Honnay *et al*. 1999; Wulf 2003).

Shape and size of habitat patches have been shown to affect species richness and quality (Forman 1995; Usher 1995). The Patton shape index, which measures patch shape irregularity, is correlated with the numbers of species in edges and clearings (Honnay *et al*. 1999). Patches with a high shape index seem able to support large numbers of species with no concurrent reduction in species quality. Studies carried out in rural mosaic landscapes show that a combination of the number of habitat types, their frequencies and the specificity of the species within the habitats reflects the biodiversity of the area (Duelli 1997; Wagner & Edwards 2001). Less frequently disturbed habitat types (e.g. hedgerows, grass verges) tend to contribute more to landscape species richness than frequently disturbed ones (e.g. arable fields, meadows) (Wagner & Edwards 2001).

As open spaces are obviously useful in the promotion of forest biodiversity in plantation forests, the use of the existing network of roads and rides in this regard has been the subject of biodiversity research (Carter & Anderson 1987; Ferris-Kaan 1995). The utility of these to the different elements of biodiversity varies with the nature of their edges and their width (Forman & Godron 1986; Spellerberg & Gaywood 1993). Edges have been associated with high biodiversity: structurally graded edges are preferable to abrupt changes from forest to open ground (Angelstam 1992; Ratcliffe & Peterken 1995). As these linear features can act as wind-funnels, "scalloping" of the forest edge along linear gaps in order to minimise windthrow of trees has been advocated as good forestry practice. This is beneficial from a biodiversity point of view, as it varies the width of the space and provides potentially more structural diversity (Ferris-Kaan 1995). Selective grazing by large herbivores can reduce the value of road edges for biodiversity, and in some forests, control of herbivore populations may increase the biodiversity supported by these open spaces (Fuller & Gill 2001), and open spaces can in turn be used for deer control (Ratcliffe 1985).

2.1.4 Policies on open space for biodiversity in forests.

Policies supporting forest management for biodiversity should be aimed at a range of geographical scales, for example, national, regional, and local (forest, stand) (Gill & Bell 1995; Quine & Humphrey 2003). Although international, national and regional policies can support management practices (e.g. MCPFE 2003), ultimately the policies that are specifically aimed at the local level will impact the practices most intensively (Gittings *et al*. 2004b). Results of management operations at the stand level are influenced by the nature and management of the surrounding land.

A survey conducted during the current review attempted to gather information on policies in different countries on open space in forests. A letter was sent (in French, Italian and Spanish versions where appropriate) to a number of officials in nineteen European countries requesting information (see Appendix 3). In addition to targeting some known government officials and forest researchers, letters requesting information were distributed at the IUFRO conference "Biodiversity and conservation biology in plantation forests", Bordeaux, 26-29 April, 2005. Some difficulty was experienced in obtaining responses that addressed the specific question on whether there was official promotion of open space management for biodiversity enhancement. Due to time and resources constraints on the BIOFOREST Project, it was seldom possible to follow up referrals to other contacts and this line of research was consequently abandoned. However, it seems that in temperate regions there is a lack of forest policies governing open space management for biodiversity enhancement. In particular, management of roads for biodiversity seemed not to be an item in many countries.

According to the responses received, the following information is offered. The authors thank the respondents for their time and input. As the discussion of policies can be a politically sensitive issue, names of most respondents are withheld. Countries are presented in alphabetical order, with Ireland at the end.

Finland.

Many thanks to Petri Heinonen, Metsähallitus Natural Heritage Services, who provided the following:

The open habitats with a defined management approach in Finnish forestry are:

1. Forest fire areas (natural fire dynamics in boreal forests)

It is a generally acknowledged problem that the extremely successful fire control in Finland has driven species dependent on charred wood material to be threatened. Since wild forest fires as such are great hazards to individuals and society it is acknowledged as well that the fire control policy cannot be changed. A solution to the biodiversity problem has been sought from two different approaches:

a) controlled man-made fire in the protected areas (national parks & other forested protected areas). Areas/landscapes have been defined where the objective is to produce regular forest fires in order to provide suitable habitat for specialized species

b) prescribed burning as a means of forest regeneration in commercial forests after regeneration cut. This method results in burned logging residuals as well as burned retention trees and supports the objective concerning protected areas. This is one of the targets defined in the Finnish Forest Certification System, Criterion 15. For more information on the fire/gap dynamics in boreal forests, see Korpilahti and Kuuluvainen (2002).

2. Woodland pastures (semicultural biotope declining due to changed animal husbandry)

The biggest single threat to the threatened species in Finland is estimated to be the closing (in growth) of the open spaces. It exposes the primary threat to 28 % of the threatened species. The open spaces are typically cultural areas: meadows, pastures, fallows, ditches in the fields, gardens, etc. The reason for the change is the change in agricultural methods. Woodland pastures & corrals are forest related cultural habitats. The management of these habitats in order to maintain dependent species is difficult and expensive. Typically it is done only with funding from external sources. Some farmers do it at their own cost due to respect for tradition. There is, however, no specified programme for these habitats in Finland. To some extent these areas are maintained in protected areas.

3. Sun-scorched southern eskers (special biotope in Southern Finland)

Post-glacial esker formations are common in Southern Finland. Their south and southwest facing slopes are subject to hot dry sunshine and the areas have been subject to frequent forest fires. This has resulted in a special habitat with specialised species. These are typically vascular plants and insects, butterflies dependent on the plants. Due the fire control and more ambitious forestry these areas are growing too dense. Here the question is to manage the forest in such a way that ground vegetation does not get suppressed. In many cases this means removal of Norway spruce and opening the canopy (thinning).

4. Herb-rich forests (spruce control in order to maintain vascular plants in the ground level)

Roughly half of the forest-dwelling threatened species occur in the herb-rich forests, although these forests represent today only 1.5 % of the forest area of Finland. Most of these habitats have been cleared for agricultural use in the past. The management needed here resembles that of eskers. Vascular plants need special conditions and spruce is harmful if found in larger quantities. The management question is spruce control (Metsähallitus 1995).

France.

The regulations in France vary according to *Departement*, each of which is relatively autonomous with regard to rural planning and policy. However, no clear answer was received on these specific questions from any Department official. This is not to say that there are no regulations: it seems that there are, but obtaining the information is not straightforward. Information gleaned from the survey did indicate that under the PEFC certification programme in Aquitaine, retention of broadleaf trees at pine forest edges and within the forest is a requirement. Designation of aquatic zones is also required for certification, but there are no other obligations to retain or create open spaces within the forest. Research is also ongoing on enhancement of the biodiversity of maritime pine forests in the Landes of Gacogne at the stand and landscape levels by establishing blocks of broadleaf species within the pine matrix (the ISLANDES project).

Greece

In 2000 the Ministry of Agriculture issued criteria and indicators for the sustainable management of forest in Greece. Some data on forest types, reserves, threatened species, stocking provenance, deadwood and other issues of interest are given here. It states that there is no special reference in the national forest legislation to the conservation of biological diversity or to threatened species and rare and vulnerable ecosystems. Nature is not protected effectively because the measures taken are usually fragmentary and without strategic and long-run perspectives. The respondent for Greece indicated that there is not any specific information available on enhancement of biodiversity through management of internal open spaces. Internal open spaces are not included as a special category in any known management plans that are applied in forests. However, Greece is a participant in an EU project http://www.forestbiota.org, and as such will be active on the subject of biodiversity in Greek forests.

Portugal.

Currently, policies promoting management practices to enhance biodiversity in forests cover only National Parks and protected areas. Several associations of forest producers are trying to establish systems for sustainability certification. However some research has been carried out in planted stands of *Pinus pinaster* which showed that biodiversity was promoted by a diversity of structural phases, the presence of older stands and habitat diversity at the landscape level.

Slovenia.

Guidelines on managing open spaces are partly covered by the *Rules on the protection of forests*, published in 2000. However, the guidelines are far from being concrete. They do not state where or how the management steps should be carried out. Although Slovenia fosters close-to-nature forest management, the details are often left to district foresters. Separate, more specific guidelines are being developed.

Spain.

Regulations in Spain are made at the level of regional departments rather than nationally.

Switzerland.

The respondent was unclear as to whether there was any specific official guideline or regulation governing management of open spaces for biodiversity in forests. However, there has been ample research on the subject of open spaces in Swiss forests, mainly after windthrow and forest fires. The importance of highly structured forest edges for regional biodiversity was highlighted.

UK.

Whether due to the availability of the publications or the presence of a member of the UK Forestry Commission (Forest Research) on the BIOFOREST Steering Group, or perhaps a combination of the two, information on UK forest standards and guidelines seemed most easily accessed. Much of this is mentioned in the review above; see references by Humphrey, Ferris, Quine, and others. In particular Ferris and Carter's Forestry Commission bulletin on the management of rides, roadsides and edge habitats (Ferris & Carter 2000a) seems unparalleled in other countries. It contains very precise detailed guidelines on how to design linear habitats in forests and the effects of different management scenarios. Other research in the UK has been reported in the review, above.

Ireland.

In Ireland it has been recognised that open space is important for biodiversity in forestry plantations. The *Forest Biodiversity Guidelines* (Forest Service 2000c) require that a combination of open space and other "Areas for biodiversity enhancement" such as "retained habitats" should comprise 15% of the area of each forest. The open space should be carefully designed in terms of its planning and management (Iremonger 1999). For example, an undulating forest edge is recommended with protrusions into adjoining open spaces, and the edge effect between the canopy and the open space should be maximised. Forest roads and extraction routes should be excluded from the retained habitats, and recommends a 3m "protective zone" around the retained habitat.

The *Code of best forest practice* (Forest Service 2000a) advocates that open spaces be (a) proactively managed with a view to capitalising on their biodiversity potential and (b) enhanced during (grant-aided) woodland improvement, generating local wildlife habitat development and protection. The Code states that generally forests greater than 10ha require a forest road network of 20 linear m per ha for operations and extraction, whereas those less than 10ha can generally use forest tracks. Recommendations exist stating the minimum distance between planting and aquatic zones and minimising the number of times a road must cross a water body. The *Forest Road Manual* (Ryan *et al*. 2004) has more detail regarding technical specifications for roads, but also states that pre-planning of these can improve opportunities to enhance forest biodiversity. A standard tree clearance of 15m is stipulated, composed of 5m for the road and an average of 5m tree-free area on either side. The stated purpose of this area is mainly to provide for the safe construction of roadside drains, to reduce shading and to allow sunlight and wind access, resulting in a regular drying of the road surface.

BIOFOREST Study of open space diversity and experimental manipulation.

The results of this study support the selection of open space for a focus of the project investigating methods to enhance biodiversity in Irish plantations. Additionally, the results support the establishment of an experiment to investigate the effects of different road widths on Irish plantation biodiversity (see Section 9).

3 THE BIOFOREST DATABASE FOR THIS PROJECT

3.1 INTRODUCTION

Ecologists are more frequently turning to innovative digital technologies when studying the natural world. Geographical Information Systems (GIS) technology is a digital mapping tool that not only aids in the creation of maps, but it also provides powerful facilities for drawing together data from various sources. GIS supports analysis, manipulation and visualisation of data, which enables the user to make decisions that must have an explicit spatial dimension.

The BIOFOREST 3.1.3 GIS uses this technology to bring together botanical and zoological field data, along with existing base data from national organisations such as Coillte, Environmental Protection Agency, Ordnance Survey Ireland and the National Parks and Wildlife Service. The BIOFOREST GIS also allows access to tabular data, imagery and metadata (data that is used to describe other data, examples of metadata include schema, table, index, view and column definitions) using ISO compliant standards for all of the datasets within the GIS. The BIOFOREST GIS is a fully functional, flexible and updateable GIS system. In tandem, the BIOFOREST Project is also utilising a licence-free and cost-free software system that can be run on any computer.

3.2 DATA TYPES

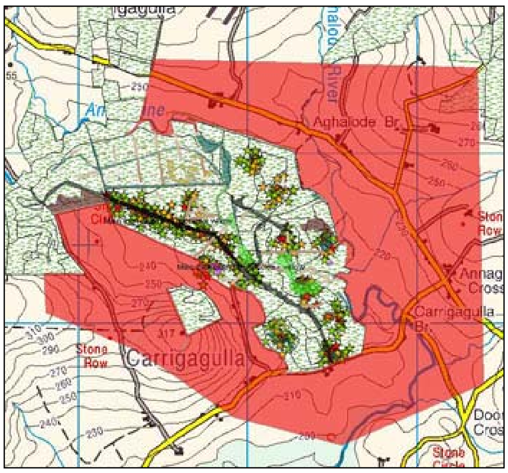
3.2.1 Base data

A number of base datasets are used within the BIOFOREST GIS. These include:

- Irish Coastline (EPA licence agreement);
- 1:50,000 Discovery Series (EPA licence agreement);
- 6 Inch Historical Maps (Coillte licence agreement);

Aerial Ortho Photographs (OSI Licence Agreement);

- Designated Areas (NPWS - www.heritagedata.ie);
- Forestry Data i.e. Properties, Compartments, Stands, Old Woodland Database (Coillte licence agreement);
- CORINE land use change (EPA licence agreement).



3.2.2 BIOFOREST Survey Data

BIOFOREST 3.1.3 survey data include:

- Vascular species data;
- Bryophyte species data;
- Epiphyte species data;
- Bird species and behaviour data;

- Spider species data;
- Hoverfly species data.

In addition, data on site management, environment and study design are integrated.

4 TERRESTRIAL VEGETATION

4.1 INTRODUCTION

Natural forests almost always contain some open, treeless areas within them. These may be temporary canopy gaps of varying sizes caused by disturbance agents, such as windthrow, fire or insect attack. More or less permanent open spaces can also be found in forests in places that are not favourable to tree growth because of waterlogged soils, rock outcrops or herbivory. Open spaces within forests provide suitable sites for plant species that cannot tolerate the shaded conditions of the forest interior (Peterken & Francis 1999). The additional habitats and species supported within open spaces serve to increase the biodiversity of the forest as a whole.

The value of open spaces for forest biodiversity is recognised by the Forest Service, which requires 5-10% of open space be created or maintained (as part of the Area for Biodiversity Enhancement (ABE)) within new forestry plantations in order to qualify for afforestation grant aid (Forest Service 2000c). Such open spaces can include ridelines, firebreaks, forest roads and turning bays, unplantable areas, areas left unplanted to facilitate ESB powerlines or other utilities and buffer zones for aquatic habitats and archaeological features. In essence, these open space types can be simplified into three: linear open spaces, non-linear open spaces (or glades) and roads. Although roads are also linear features, their management (e.g. surfacing with gravel) and the different roadside habitats (e.g. road cutting banks, roadside drains) provided for plants make them qualitatively different from other linear open spaces. A key aim of maintaining open spaces as part of the ABE within plantation forests is to "conserve and enhance the biodiversity value throughout the entire forest" (Forest Service 2000c). A secondary benefit is the provision of semi-natural open habitats that may be rare in intensively managed landscapes.

The objectives of this chapter are

1. To assess the biodiversity of terrestrial vascular plants, bryophytes and lichens in open spaces in plantation forests,
2. To investigate the major environmental and management factors influencing plant biodiversity at the plantation scale, between open spaces and within the open space, and
3. To recommend planning and management measures that can enhance the vegetation diversity of open spaces in plantation forests.

4.2 METHODS

4.2.1 Study Sites

To reduce the effect of large-scale environmental variation, 12 study sites were chosen in two geographic clusters: a Cork cluster of six sites mainly in Co. Cork, but also including neighbouring parts of Co. Kerry and Co. Limerick, and a Wicklow cluster of six sites, mostly in Co. Wicklow, but including one in south Co. Dublin (Figure 1 and Appendix 4). The study sites were plantation forests comprised primarily of Sitka spruce (*Picea sitchensis*) ranging in age from 26 to 47 years old. The Wicklow sites were mostly situated on podzol or peaty podzol soils, whereas the Cork sites were largely on blanket peat.

Each study site contained a mixture of different types of open space surrounded by forest. All sites contained gravelled forest roads and unplanted forest rides. Five of the six sites in each cluster also contained unplanted glades of varying sizes. The sites were surveyed in the summer of 2003.

4.2.2 Vegetation Sampling

Five open spaces were surveyed within each site, including at least one open space of each type present in the site (road, ride and glade). In each open space, vegetation was sampled in two or more 4 m^2 plots, depending on the open space type. Plots were distributed between the centre of the open space and the forest edge. Preliminary surveys found that the ecotone between the forest interior and the adjacent open space was better defined by the ends of the branches than the tree trunks, and therefore the forest edge was defined as the edge of the forest canopy.

Orientation of open spaces and location within them is an important factor regulating the amount of light available to vegetation within the open space. The north sides (with maximum southern exposure) of wide rides oriented east-west receive more light than south sides (Carter & Anderson 1987; Ferris & Carter 2000b). Because ride and road orientation in plantation forests is usually constrained by other factors, such as local terrain and pre-existing access routes, we attempted to control this factor rather than allowing it to vary. Therefore, all vegetation plot sampling was carried out on the north, northeast or east sides of open spaces.

In rides, two plots were recorded: one in the centre of the ride and the second centred on the forest edge. In glades, three plots were recorded: one in the centre of the open space, the second centred on the forest edge and the third located midway between the others. The number of plots recorded in roads varied according to the characteristics of the individual road. Roadside vegetation and topographical features were separated into the following categories: *verge*- the zone immediately adjacent to the road surface, influenced by road gravel and vehicle disturbance, *ditch*- drains paralleling the road, *bank*- vertical or very steep faces exposed by road cutting, and *setback*- roadside open space that cannot be classified as one of the above features and is located between the forest edge and any of the above features. One plot was recorded in each of the features present. Whereas all other plots were 2 m × 2 m in dimension, it was often necessary to use 1 m × 4 m plots in verges and ditches, thus limiting the sampling to the feature while preserving the 4 m^2 plot size.

In each plot, percent cover of each terrestrial (including saxicolous and saproxylic) vascular plant, bryophyte and lichen species was recorded to the nearest 5%. Cover below 5% was recorded either as 3% (indicating cover of 1-5%) or 0.5% (indicating cover < 1%). For bryophytes and lichens, only species forming patches more than 5cm^2 were recorded.

In addition, a complete species list was compiled for glades and for the open space of rides and roads within 10 m on either side of the plots. All vascular plants were recorded, as were terrestrial bryophytes and lichens forming patches more than 50cm^2. The south and west sides of open spaces were included in the open space species list.

Nomenclature follows Stace (1997a) for vascular plants, Smith (2004a) for mosses, Paton (1999) for liverworts and Purvis *et al.* (1992) for lichens.

4.2.3 Environmental and Structural Data Sampling

In each 4 m^2 plot, the height and percent cover (nearest 5%, as above) of the following vegetation strata were recorded: small trees/large shrubs (2-5 m tall), saplings/small shrubs (< 2 m tall), brambles/briars, forbs, graminoids, bryophytes/lichens and the lowest live branches of the conifer crop (where present). The percent cover (nearest 5%, as above) of the following ground cover categories was recorded: bare soil, bare rock, standing water, leaf litter (non-conifer), conifer needle litter, fine woody debris (< 7 cm diameter), coarse woody debris (≥ 7 cm diameter) and live tree stems/roots. Soil drainage was estimated on a five point scale, based on soil physical characteristics. The distance from the centre of each plot to the bases of the trees at the forest edge was recorded. Plot location was marked with a GPS.

Five soil subsamples were collected to a depth of 5 cm from the corners and centre of each plot and bulked in the field. Soil pH was determined for the bulked, field moist samples at the earliest opportunity, using a glass electrode pH meter on a 2:1 suspension of distilled water and soil. Samples were later air-dried and loss-on-ignition determined.

Within each open space, soil type, slope and aspect were recorded and, for roads, the type of stone used for surfacing was identified. Height of trees at the forest edge was measured to the nearest 0.1 m, and the thinning status of the forest edge was recorded and placed on a semi-quantitative scale: 0 = unthinned, 1 = selection thinning only, 2 = line (and selection) thinning of 1 row in 6-7, and 3 = line thinning of greater intensity than 1 in 6. Intensity of grazing (mainly by deer and hares) was ranked on a scale of 0 - 3. Other silvicultural or recreational disturbance was noted, particularly whether the open space had been drained and ground-prepared. The width of rides and roads was measured from tree trunk to tree trunk. The area of glades was measured using aerial photographs. Ride and road orientation was measured in degrees (0-180°) and then transformed using the equation:

$$x' = 1 - \left(\frac{|x - 90|}{90} \right)$$

where x = orientation in degrees and x′ = transformed orientation. This places road orientation on a 0 – 1 scale, where an east-west orientation (90°) becomes 1, a north-south orientation becomes 0 and intermediate orientations (45° and 135°) become 0.5. This reflects east-west roads receiving maximum duration of direct sunlight (along the northern side) during the day and north-south roads receiving the minimum.

The light environment of the open spaces was measured using hemispherical photography. Hemispherical photos were taken in the centre plot of rides and glades, the centre of roads and in the edge plots of glades and rides over 20 m wide. Photos were taken using a tripod-mounted Canon AE-1 35mm camera with a Canon 7.5mm f/5.6 fisheye lens. The lens uses simple polar projection to represent a hemispherical distribution of points onto a circular image (Herbert 1987). Ilford FP4 Plus black-and-white film was used. The camera was erected at 1.3 m above ground, levelled and oriented towards magnetic north. At least four combinations of shutter speed (1/60 and 1/125) and aperture (5.6 and 8) were used at each point. Developed negatives were scanned using a Microtek ScanMaker X12 flatbed scanner. Negatives were converted to positive grey-scale images at 1200 ppi resolution and saved in JPEG format. Scanned images were then analysed using Gap Light Analyser 2.0 software (Frazer *et al*. 1999b). In addition to percentage canopy cover, transmitted direct and diffuse solar radiation were estimated with Gap Light Analyser 2.0 using site data collected in the field for latitude, longitude, elevation, slope and aspect. Direct radiation is that fraction of total solar radiation that emanates directly from the sun and not absorbed or reflected by the atmosphere. Diffuse radiation is the fraction that is scattered by the atmosphere towards the earth from all portions of the sky. Transmitted direct radiation is the portion of direct radiation that is transmitted through gaps in the forest canopy into the open space below. Transmitted diffuse radiation includes diffuse radiation that passes directly through canopy gaps and also direct radiation that has been scattered by the forest canopy. Above-canopy solar radiation parameters were estimated using the ratio of mean daily hours of sunshine to day length on a monthly basis in conjunction with the equation developed by McEntee (1980) and equations in the software manual (Frazer *et al*. 1999b). Sunshine data were based on thirty year averages obtained from the Met Éireann website (2005); for Wicklow sites, sunshine data from Dublin Airport were used and Cork Airport data were used for Cork sites. The clear-sky transmission coefficient was set at 0.8, following Garvey (1998).

We calculated the amount of habitat within 50 m, 100 m, 200 m and 500 m of the centre plot of the surveyed open spaces in each of nine categories: the area of broadleaf scrub, road, undeveloped plantation, windthrow, clearfell, young forestry, unplanted open space within the plantation and external open space and the length of rides. The habitat categories were mapped using aerial photographs, and the amounts of habitats within a specified distance (radius) of an open space centre were calculated using ArcView GIS.

4.2.4 Data Analysis

The following biodiversity metrics were calculated for each 4 m² plot: vascular plant species richness, bryophyte and lichen species richness, Simpson's diversity index (expressed as 1-D so that increases in index value represents increases in diversity) and the Berger-Parker index of evenness (high values of the index correspond to high dominance by one species, and hence low evenness and low diversity) (Magurran 2004). In addition, all plant species were classified as being typical of open habitats, characteristic woodland species or species frequently found in both wooded and non-wooded habitats. The classifications of species were determined using habitat and autecological information contained in Webb *et al.* (1996), Clapham *et al.* (1987), Stace (1997a), Fitter and Peat (1994), Jermy *et al.* (1982), Hubbard (1984), Paton (1999), Watson (1981), Smith (2004a), Purvis *et al.* (1992) and Dobson (2000). Vascular plants were classified as competitors, stress-tolerators, ruderals or combinations of these categories, according to Grime's CSR theory (Grime *et al*. 1988). The species richness of plants in these categories was calculated for each plot. Means of all of the above parameters were also calculated at the open space scale for use in analyses at the open space and plantation scales.

The effect of environmental and management variables on biodiversity metrics were investigated using linear regression or ANOVA as appropriate. Paired t-tests (within an open space) were used to investigate the effect of plot location relative to the forest edge in glades and rides on the biodiversity metrics.

Total area in eight habitat categories and total ride length within 100 m, 200 m and 500 m of open space centres were used as predictor variables in canonical correspondence analyses (CCAs) of species composition (presence/absence) at the open space scale. One CCA was performed for each distance increment. Habitat area and ride length parameters at the 50 m, 100 m, 200 m and 500 m scales were used in regression analyses of the biodiversity metrics.

Species composition in the open spaces was investigated using the species lists generated at the open space scale (presence/absence data) using NMS ordination (Legendre & Legendre 1998). Sørensen distance was the distance measure used. For each ordination, twenty preliminary ordinations were carried out, each one beginning with six dimensions and then stepping down in dimensionality to one. Monte Carlo tests were performed using 100 runs with randomised data. The optimal number of dimensions was determined, and the best of the preliminary ordinations with that number of dimensions was used as the starting configuration for the final ordination.

Vegetation structure and ground cover data in the plots were simplified by identifying structure/cover groups using flexible-beta hierarchical cluster analysis on Sørensen distance measures with the parameter β set to equal -0.25. This setting of β produces a solution intermediate between single-linkage and complete-linkage agglomerative clustering (Legendre & Legendre 1998).

Prior to parametric analyses, variables were inspected for conformity to the assumptions of parametric statistics. Variables were transformed and outliers were removed as needed. In particular, Simpson's diversity index was right-skewed and was therefore transformed to the reciprocal natural logarithm for most analyses. Berger-Parker evenness index was left-skewed and generally transformed using cube roots. Multivariate analyses were conducted using PC-Ord (McCune & Mefford 1997a) and univariate analyses were performed with SPSS 11.0 (SPSS 2001).

4.3 RESULTS

4.3.1 Diversity at Plantation Scale

A full list of plants recorded is given in Appendix 5. The mean site vascular plant species richness of 4 m^2 plots ranged from 5.4 (± 0.6 se) to 10.7 (± 3.5 se). However, there were no significant differences among sites in vascular plant species richness, Simpson's diversity index or Berger-Parker evenness index. The only significant differences among sites in mean plot bryophyte and lichen species richness were between the most species rich site and the two least species rich sites: MUCK, a Wicklow site without glades, had significantly higher bryophyte and lichen species richness (9.1 ± 1.1 se) than GLAN (4.0 ± 0.7 se) or LUGG (4.0 ± 0.4 se), according to Ryan's Q multiple comparisons tests following a significant ANOVA ($F_{11,48} = 2.73$, $p = 0.008$).

There were no significant relationships between biodiversity metrics calculated at the open space scale and the amount of non-forest habitat in nine categories (area of broadleaf scrub, road, undeveloped plantation, windthrow, clearfell, young forestry, unplanted open space within the plantation and external open space and length of rides) within 50 m, 100 m, 200 m or 500 m. We also calculated the total amount of unplanted open space (summing the broadleaf, road, internal and external unplanted areas) and total amount of non-forest habitat and tested the biodiversity metrics using these summary predictor variables. No significant relationships were found. The only exceptions were significant negative relationships between bryophyte and lichen species richness and total unplanted open space ($r^2 = 0.13$, $p = 0.005$) and total non-forest habitat ($r^2 = 0.14$, $p = 0.003$) within 50 m. These relationships reflect the lower species richness of bryophytes and lichens in glade plots which have more non-forest habitat within 50 m than do the more bryophyte-rich rides, and are therefore not truly plantation scale effects.

CCAs of the 43 glades and rides using the amount of nine non-forest habitat categories (area of broadleaf scrub, road, undeveloped plantation, windthrow, clearfell, young forestry, unplanted open space within the plantation and external open space and length of rides) nearby were not significant, according to Monte Carlo tests of 199 randomised runs. There were no significant relationships between any ordination axes and the amount of habitat within 100 m, 200 m or 500 m.

There were no significant relationships between the species composition of the roads and amount of non-forest habitat within 100 m or 200 m, according to the CCAs. For the CCA of road species composition and amount of non-forest habitat within 500 m, the second axis significantly explained 10% of the variation in the species data ($\lambda_2 = 0.231$, total variance = 2.30, $p = 0.01$). The species-environment correlation for the second axis was also significant ($r = 0.99$, $p = 0.04$). The first axis, however, did not significantly explain more variation than would be expected by chance alone ($\lambda_1 = 0.252$, $p = 0.21$). The strongest positive correlations between Axis 2 scores and the habitat areas were for area of windthrow ($r = 0.39$), external open space ($r = 0.31$), broadleaf scrub ($r = 0.22$) and young forestry ($r = 0.20$). Negatively correlated were area of clearfell ($r = -0.61$), forest road ($r = -0.48$) and undeveloped plantation ($r = -0.14$). (The above are "intra-set correlations" *sensu* ter Braak (1986); as they are not true correlation coefficients, they cannot be tested for significance.) Species with the highest positive correlations with axis 2 were primarily those associated with open habitats, especially humid grassland, and the strongest negative correlations were mainly with shade-tolerant bryophyte species. However, post-hoc correlation analyses with axis 2 scores found that they were positively correlated with transmitted diffuse radiation ($r = 0.58$, $df = 15$, $p = 0.015$), transmitted direct radiation ($r = 0.43$, $df = 15$, $p = 0.087$) and road width ($r = 0.35$, $df = 15$, $p = 0.163$). Therefore, the CCA axis 2 may be reflecting a gradient of increasing road width and light, rather than open space within 500 m.

4.3.2 Diversity Between Open Spaces

4.3.2.1 Species Composition

Preliminary analysis of the species list data for the open spaces indicated that the flora of the roads and the flora of rides and glades were quite different. Separate analyses were therefore conducted for the two groups of open spaces.

4.3.2.1.1 Glades and Rides

The NMS ordination of the 43 glades and rides (3-D solution: stress = 16.11, $p = 0.01$) showed that the major source of variation in species composition was due to geographic location between Cork and Wicklow. The two regions separated neatly in ordination space. Separate ordinations were then conducted for the two site clusters.

In the Cork glades and rides, the main contrast was between open spaces with higher mean pH and steeper slopes, where species such as *Juncus effusus, Holcus lanatus, Anthoxanthum odoratum* and *Rubus fruticosus* were present, and open spaces with lower pH on more organic rich soils, with species such as *Sphagnum quinquefarium, Rhytidiadelphus loreus* and *Vaccinium myrtillus* (Figure 2). The second NMS dimension mainly distinguished between narrow rides and larger glades and certain rides with higher levels of transmitted diffuse and direct radiation. Species typical of open heath or bog were characteristic of open spaces with high scores on dimension 2, such as *Erica tetralix*, *Eriophorum angustifolium, E. vaginatum, Juncus squarrosus* and *Polygala serpyllifolia*.

The ordination of the Wicklow glades and rides was similar to that of Cork. Open spaces on the left of the first NMS dimension generally had steeper slopes, lower above-canopy diffuse radiation (due largely to topographic factors), lower mean pH and less fertile soils, as demonstrated by the presence of such species as *Calypogeia muelleriana, Campylopus flexuosus, Blechnum spicant, Oxalis acetosella, Pleurozium schreberi* and *Rhytidiadelphus loreus* (Figure 3). In contrast, open spaces scoring highly on dimension 1 had species such as *Agrostis stolonifera, Rubus fruticosus, Holcus lanatus* and *Anthoxanthum odoratum*. The association of more intensive thinning with dimension 1 of the ordination appears to be somewhat spurious, as LUGG, a more fertile site, was thinned fairly heavily and sites on steeper slopes tended to be thinned less intensively or not at all. Dimension 2 of the ordination contrasted glades having higher transmitted direct and diffuse radiation with narrow rides. Typical species occurring in these glades were *Deschampsia flexuosa, Holcus mollis, Luzula sylvatica* and *Pteridium aquilinum*. The open glades also tended to have more acidic, organic soils and supported heavier grazing, mainly by deer. In contrast, shade-tolerant bryophytes, such as *Atrichum undulatum, Dicranella heteromalla, Pellia epiphylla* and *Thuidium tamariscinum*, occurred primarily in the narrow rides.

4.3.2.1.2 Roads

The most optimal ordination of the 17 roads sites was a two-dimensional solution, unlike the rides and glades ordinations (Figure 4). Dimension 1 contrasted narrow roads on steeper slopes on the left with wider roads on more gentle slopes towards the right. Dimension 2 reflected differences between the Wicklow and Cork clusters, with the former more heavily grazed and generally receiving greater above-canopy radiation due to climate. The type of gravel used in road surfacing was also an important factor in the roadside vegetation.

Wicklow and Cork roads gravelled with limestone were concentrated in the lower right of the ordination, Wicklow roads gravelled with local mica/schist material were plotted on the left and Cork roads surfaced with local sandstone were concentrated towards the top (Figure 4). Species characteristic of disturbed, base-rich habitats were more frequent on the verges of limestone surfaced roads, including *Bellis perennis, Euphrasia arctica, Plantago lanceolata, Potentilla reptans, Sonchus asper,* and *Trifolium dubium*.

4.3.2.2 Comparison of Open Space Type

Combining both geographical clusters, rides had lower vascular plant species richness and higher bryophyte species richness than glades and roads (Table 1). Roads had higher vascular plant species richness, numbers of species associated with open habitats and Simpson's diversity than the other two open space types (Table 1). For glades and rides, however, relationships with biodiversity metrics differed by geographical area. In Wicklow, mean vascular plant species richness at the open space scale was significantly higher in glades than in rides, whereas bryophyte and lichen species richness was higher in rides (Table 2). In Cork, bryophyte and lichen species richness was higher in rides than in glades, and the mean Berger-Parker evenness index was lower (Table 2). Accordingly, Simpson's diversity index was higher in Cork rides than in glades.

Table 1. Mean (± standard error) vascular plant species richness, bryophyte and lichen species richness, Simpson's diversity index, Berger-Parker evenness index and open habitat species richness in glades rides and roads. Values with the same letter are not significantly different according to Ryan's Q multiple comparisons tests following significant ANOVAs and a MANOVA of all biodiversity metrics.

	Glades	Rides	Roads
Vascular Plant SR	7.6 ± 0.6 A	5.6 ± 0.5 B	12.1 ± 1.1 C
Bryo & Lichen SR	5.3 ± 0.4 A	7.5 ± 0.4 B	6.1 ± 0.6 A
Simpson's Diversity	0.70 ± 0.03 A	0.75 ± 0.02 A	0.79 ± 0.02 B
Berger-Parker Evenness	0.43 ± 0.03 A	0.40 ± 0.02 A	0.35 ± 0.03 A
Open SR	2.7 ± 0.4 A	2.4 ± 0.4 A	5.6 ± 0.7 B

Table 2. Mean (± standard error) vascular plant species richness, bryophyte and lichen species richness, Simpson's diversity index and Berger-Parker evenness index in Wicklow and Cork glades and rides. Values shown in bold type indicate significant differences between glades and rides within the same geographic cluster, according to ANOVAs following a significant MANOVA.

	Wicklow			Cork		
	Glades	Rides	p	Glades	Rides	p
Vascular Plant SR	**9.0 ± 0.67**	**6.1 ± 0.82**	0.013	5.0 ± 0.23	5.1 ± 0.65	0.900
Bryo & Lichen SR	**5.7 ± 0.50**	**7.5 ± 0.67**	0.042	**4.5 ± 0.69**	**7.6 ± 0.51**	0.002
Simpson's Diversity	0.76 ± 0.021	0.72 ± 0.034	0.358	**0.60 ± 0.054**	**0.76 ± 0.027**	0.008
Berger-Parker Evenness	0.38 ± 0.027	0.42 ± 0.032	0.285	**0.54 ± 0.045**	**0.38 ± 0.031**	0.010

4.3.2.3 *Environment and Management*

4.3.2.3.1 Glades and rides

In Wicklow glades and rides, age of the plantation forest was positively associated ($r^2 = 0.34$) with mean plot vascular plant species richness (Table 3). Vegetation evenness, as measured by the Berger-Parker index, tended to increase ($r^2 = 0.17$), and Simpson's diversity index also generally increased ($r^2 = 0.13$) with greater forest age. Mean bryophyte and lichen species richness and Simpson's diversity were significantly higher on peaty podzol soils than on podzols (Table 3). Bryophyte and lichen species richness was significantly lower on shale bedrock than on granite or mica/schist (according to significant Tamhane's T2 multiple comparisons tests for unequal variances following an ANOVA: $F_{2,20} = 3.93$, $p = 0.036$); however, only one Wicklow site was on shale. This site was LUGG, a low elevation forest in south Co. Dublin, with vigorous vascular plant vegetation. Bryophyte and lichen species richness also tended to be lower in better drained sites and higher in open spaces that had been ground-prepared at forest establishment (Table 3).

Table 3. Significant relationships between diversity variables and environmental variables in Wicklow and Cork glades and rides. Positive relationships with the environmental variable are indicated '+', whereas negative relationships are shown as '– '. Diversity variables are: *VSR*- vascular plant species richness, *BSR*- bryophyte and lichen species richness, *1-D*- Simpson's diversity index, *d*- Berger-Parker evenness index. P-values are indicated as: (+) = p ≤ 0.1, + = p ≤ 0.05, ++ = p ≤ 0.01. Quantitative variables were tested using linear regression and categorical variables were tested using ANOVA.

	Wicklow				Cork			
	VSR	**BSR**	**1-D**	***d***	**VSR**	**BSR**	**1-D**	***d***
LOI					–			
Thinned					+			
ForestAge	++		(+)	(–)				
Drainage		(–)						
Shale [1]		–						
PeatyPodz [2]		++	+					
GroundPrep		(+)						
Cutover [3]							– –	++

[1] Wicklow plots (in LUGG) on shale compared with Wicklow plots on granite or mica/schist.
[2] Wicklow plots on peaty podzols compared with Wicklow plots on podzols
[3] Cork plots on cutover peat compared with Cork plots on intact peat and peaty podzols.

In Cork glades and rides, mean plot vascular plant species richness was negatively associated ($r^2 = 0.26$) with percentage of soil organic matter (as measured by loss-on-ignition) Table 3). Vascular plant species richness was also higher in the open spaces of forests that had been thinned than in unthinned forests (Table 3). Simpson's diversity was lower and Berger-Parker evenness index was higher in open spaces located on formerly cutover bog than on uncut peat (Table 3).

4.3.2.3.2 Roads

There were strong positive relationships between pH and vascular plant species richness and Simpson's diversity, and a negative relationship between pH and Berger-Parker evenness index (Table 4). On the other hand, increasing organic carbon as measured by loss-on-ignition and steeper slopes were negatively associated with vascular plant species richness and Simpson's diversity and positively associated with Berger-Parker evenness index (Table 4). Road verge plots adjacent to forest roads surfaced with limestone gravel had higher vascular plant species richness than roads surfaced with local sandstone or mica/schist (Table 4). The only significant relationship between bryophyte and lichen species richness in the road plots was a negative association with intensity of thinning (Table 4). This relationship emerged only when comparing Cork and Wicklow sites together; any association within each of the geographical clusters appeared unconvincing. Berger-Parker evenness index tended to decrease with increasing forest age in the Cork sites (Table 4).

Table 4. Significant relationships between diversity variables and environmental variables in roadside plots. Positive relationships with the environmental variable are indicated '+', and negative relationships are shown as '– '. Diversity variables are: *VSR*- vascular plant species richness, *BSR*- bryophyte and lichen species richness, *1-D*- Simpson's diversity index, *d*- Berger-Parker evenness index. P-values are indicated as: (+) = p ≤ 0.1, + = p ≤ 0.05, ++ = p ≤ 0.01, +++ = p ≤ 0.001, ++++ = p ≤ 0.0001. Quantitative variables were tested using linear regression and categorical variables were tested using ANOVA.

	VSR	**BSR**	**1-D**	***d***
pH	++++		++	–
LOI	– – –		–	(+)
Slope	– –		– –	++
Limestone surface [1]	++			
ThinIntens		–		
Forest age [3]				(–)

[1] Verge plots beside roads surfaced with limestone compared with sandstone and mica/schist road surfaces.
[2] Cork plots on sandstone compared with Cork plots on shales.
[3] Significant for Cork plots only.

4.3.3 Diversity Within Open Spaces

4.3.3.1 Plot Location

Vascular plant species richness was higher in roadside plots located on the road verge or ditch than in plots on banks or the road setback (Table 5). Open species richness was higher in verge plots than all other plot types. Ruderal species richness was higher in verge plots than in bank or setback plots. Berger-Parker evenness index was higher in setback plots than in verge plots. An ANOVA detected significant differences in Simpson's diversity index among the four roadside plot types (Table 5), but multiple comparisons tests were not able to distinguish subsets of plot types. No differences were found in bryophyte and lichen species richness.

Table 5. Mean vascular plant species richness, bryophyte and lichen species richness, open species richness, ruderal species richness, Simpson's diversity index and Berger-Parker evenness index in four roadside plot types. Values with the same letter superscript are not significantly different, according to Ryan's Q post-hoc tests following ANOVAs and a significant MANOVA.

	Verge	Ditch	Bank	Setback	p
Vascular Plant SR	17.9 ± 1.4 ^{A}	15.8 ± 1.9 ^{A}	9.3 ± 1.1 ^{B}	9.1 ± 1.0 ^{B}	**< 0.0001**
Bryo & Lichen SR	6.0 ± 0.7	4.5 ± 1.2	7.0 ± 0.9	6.0 ± 0.7	0.398
Open SR	11.3 ± 1.4 ^{A}	6.3 ± 1.5 ^{B}	2.5 ± 0.4 ^{B}	3.0 ± 0.5 ^{B}	**< 0.0001**
Ruderal SR	11.2 ± 1.2 ^{A}	7.2 ± 1.5 ^{AB}	2.6 ± 0.3 ^{B}	3.7 ± 0.6 ^{B}	**< 0.0001**
Simpson's Diversity	0.87 ± 0.03 ^{A}	0.86 ± 0.01 ^{A}	0.79 ± 0.05 ^{A}	0.73 ± 0.04 ^{A}	**0.048**
Berger-Parker Evenness	0.24 ± 0.04 ^{A}	0.29 ± 0.07 ^{AB}	0.35 ± 0.06 ^{AB}	0.42 ± 0.04 ^{B}	**0.024**

In glades, centre plots had significantly lower bryophyte and lichen species richness and Simpson's diversity than edge plots, according to paired t-tests (t_{19} = 2.46, p = 0.024 and t_{19} = 2.45, p = 0.024, respectively) (Figure 5). Edge plots had significantly lower Berger-Parker evenness index than centre or middle plots (t_{19} = 3.05, p = 0.007 and t_{19} = 2.13, p = 0.046, respectively) (Figure 5). There were no significant differences in open species richness among plot locations in glades. Vascular plant species richness and open species richness in the ride centre plot were significantly higher than in ride edge plots, according to a paired t-test (t_{22} = 2.55, p = 0.018 and t_{22} = 3.83, p = 0.0009, respectively).

4.3.3.2 Light and Open Space Size

Vascular plant species richness in 4 m^2 plots was significantly and positively associated with transmitted direct and diffuse radiation in Wicklow glades and rides, but not in Cork, where the relationships were positive, but not significant (Figure 6). In contrast, bryophyte and lichen species richness was negatively associated with transmitted radiation (Figure 7). The relationships were significant for transmitted diffuse and direct radiation in Cork, but not Wicklow, glades and rides. Also in Cork glades and rides, Simpson's diversity was negatively related to transmitted diffuse radiation (r^2 = 0.22, p = 0.01), and Berger-Parker evenness index was positively associated with transmitted diffuse radiation (r^2 = 0.21, p = 0.007). Open species richness was positively associated with direct and diffuse radiation in both geographical clusters, but the relationships were very weak and not significant.

Vascular plant species richness in the centre plot of rides was positively associated with ride width, but the relationship was weak and not significant (r^2 = 0.07, p = 0.226) (Figure 8a). Separate regressions for the two geographic clusters were also not significant. The three Wicklow plots above the regression line had all been disturbed by heavy machinery, unlike all other rides; the Cork plot was on extremely wet peat with water-filled drains. If these plots are removed, the relationship of vascular plant species richness with ride width becomes highly significant (r^2 = 0.41, p = 0.003). Ride centre plot bryophyte and lichen species richness was negatively associated with ride width, but again this relationship was weak and not significant (r^2 = 0.09, p = 0.092) (Figure 8b). Open species richness, Simpson's diversity index and Berger-Parker evenness index were also not well predicted by ride width. The ratio of ride width to tree height was no better predictor of biodiversity metrics than ride width alone.

When the relationships between biodiversity metrics in glade centre plots and glade area were investigated, the analysis was distorted by two plots located in a very large glade (in ATHN). With these outliers removed, vascular plant species richness was positively associated with glade area, but this relationship was not statistically significant (although nearly so in the Wicklow plots: $r^2 = 0.36$, $p = 0.051$) (Figure 9a). Bryophyte and lichen species richness was also positively related to glade area in both geographical clusters, and the relationship was stronger ($r^2 = 0.33$, $p = 0.013$) (Figure 9b). There was no clear relationship between open species richness, Simpson's diversity or Berger-Parker evenness index and glade area.

There were no meaningful relationships between biodiversity metrics and light variables, road width or road width : tree height ratio for road plots. Within some particular roadside features, for example ditch plots or bank plots, there appeared to be relationships between some biodiversity metrics and light or road width, but none of these were convincing due to small sample size.

Transmitted solar radiation at the centre of the open spaces was well-predicted by width of linear open spaces, but less well-predicted by glade area. For example, a highly significant logarithmic model was fit to transmitted diffuse radiation by width of rides and roads (Figure 10a), whereas the relationship between transmitted diffuse radiation and glade area (square root transformed) was much less strong, despite the omission of two glades in LUGG where well-developed broadleaf scrub reduced light transmission (Figure 10b). A quadratic regression gave a slightly better fit for rides and roads ($r^2 = 0.76$, $p \leq 0.0001$) than the logarithmic regression, but the quadratic regression fit to glade area was not significant ($r^2 = 0.23$, $p = 0.140$). Regressions of transmitted diffuse and direct radiation on the ratio of road/ride width to tree height were also fit. The regressions were similar to those conducted on road/ride width alone, but model fit was poorer in all cases.

Multiple linear regressions were calculated for transmitted diffuse and direct radiation in ride and road centres using road/ride width, tree height, branch length, and transformed road/ride orientation (Section 4.2.3). However, road/ride width was the only significant variable in the models. Multiple linear regression models were also calculated with the same independent variables using as dependent variables the residuals of the quadratic regressions of transmitted direct and diffuse light on road/ride width calculated above. No variables were significant for the regression using transmitted direct radiation residuals. However, tree height explained a significant amount of the residual variation from the transmitted diffuse light quadratic regression on road/ride width ($r^2 = 0.13$, $p = 0.020$).

4.3.3.3 Vegetation Structure

Cluster analysis produced five coherent groups of plots that differed substantially in vegetation structure and ground cover and one group of three plots that was discarded as an outlier. The three plots were in LUGG and BMUT and differed from all others in high cover of birch and bracken leaf litter, respectively. The groups differed primarily in cover of Sitka spruce, graminoids and bryophytes and each group generally included plots from both geographical areas and several sites (Table 6). The exception was Group E, the smallest group, which included roadside plots with high cover of bramble in four Cork sites. Group D had the highest cover of overhanging Sitka spruce branches, bryophytes and conifer needle litter; these plots were mainly at the edges of rides. Group B plots were mainly at the forest edge in glades and roadsides. In addition to high Sitka spruce cover, this group had the highest cover and height of broadleaf forbs (including ferns) and higher graminoid cover than the previous two groups. Group C had greater cover and height of graminoids and lower cover of bryophytes and Sitka spruce than other groups. Group A was a more "average" group than the others, with reasonably high covers of graminoids and bryophytes and a slightly higher cover of low shrubs than the other groups.

A MANOVA found significant differences in the biodiversity metrics among the five structural groups (Wilk's $\Lambda = 0.535$, $p \leq 0.0001$), and subsequent ANOVAs found significant differences among groups for all four metrics (vascular plant species richness $F_{4,143}$ = 4.68, p = 0.0014; bryophyte and lichen species richness $F_{4,143}$ = 8.84, p ≤ 0.0001; Simpson's diversity $F_{4,143}$ = 5.93, p = 0.0002; Berger-Parker evenness index $F_{4,143}$ = 9.12, p ≤ 0.0001). Vascular plant species richness was significantly lower in Group D, the group with highest Sitka spruce cover, than in Groups B and E (Table 7). Bryophyte and lichen species richness was lowest in the graminoid-dominated Group C and highest in Groups D and E. Simpson's diversity was higher and Berger-Parker evenness index was lower in Group A than in Group C, which was dominated by grasses, and Group D, which was dominated by bryophytes. There were no significant differences among groups in open species richness.

Table 6. **Vegetation structure and ground cover groups.** The five groups of 4 m^2 vegetation plots were identified by flexible-beta clustering with β = -0.25. Shown are the numbers of plots in glades, rides and roads and the numbers of plots at the centre and edge of glades and rides. Also shown are the means (± standard error) of several vegetation and ground cover and height classes. The highest value(s) for a cover/height class in a group are shown in bold.

	A (n = 51)	**B** (n = 39)	**C** (n = 25)	**D** (n = 25)	**E** (n = 8)
Glades (n)	22	17	14	3	0
Rides (n)	18	8	4	18	0
Roads (n)	11	14	7	4	8
Centre (n)	25	3	10	4	0
Edge (n)	4	21	1	17	0
Sitka spruce cover (%)	2.7 ± 1.2	**44.8 ± 4.1**	1.2 ± 1.0	**68.0 ± 2.9**	11.3 ± 7.6
Bramble Cover (%)	0.7 ± 0.5	1.5 ± 0.6	4.2 ± 1.9	0.6 ± 0.6	**40.0 ± 13.4**
Low Shrub Cover (%)	**15.1 ± 2.5**	6.0 ± 1.6	2.8 ± 1.0	8.4 ± 2.9	8.3 ± 3.6
Graminoid Cover (%)	57.0 ± 2.3	59.4 ± 3.6	**88.6 ± 1.8**	21.7 ± 3.2	23.1 ± 4.0
Graminoid Height (cm)	39 ± 3	36 ± 3	**66 ± 7**	27 ± 4	46 ± 11
Forb Cover (%)	7.8 ± 1.3	**29.5 ± 4.8**	6.2 ± 1.3	3.0 ± 1.2	11.8 ± 3.5
Forb Height (%)	21 ± 2	**43 ± 7**	30 ± 5	17 ± 2	16 ± 3
Bryophyte Cover (%)	43.5 ± 2.8	34.1 ± 3.3	12.5 ± 2.0	**51.1 ± 5.2**	28.1 ± 4.2
Needle Cover (%)	2.2 ± 0.6	7.1 ± 1.0	0.4 ± 0.2	**37.0 ± 5.0**	6.3 ± 5.0

Table 7. Mean values (± standard error) for vascular plant species richness, bryophyte and lichen species richness, open species richness, Simpson's diversity index and Berger-Parker evenness index in vegetation structure and ground cover groups. Groups are indicated by a letter and are described in Table 6 above. Means with the same letter superscript are not significantly different according to Ryan's Q multiple comparisons tests (for species richness) or Tamhane's T2 tests for unequal variances (for Simpson's diversity and Berger-Parker evenness index).

	A (n = 51)	B (n = 39)	C (n = 25)	D (n = 25)	E (n = 8)
Vascular Plant SR	8.4 ± 0.6 ^{AB}	9.4 ± 0.9 ^{A}	8.5 ± 1.0 ^{AB}	5.5 ± 0.7 ^{B}	13.3 ± 2.1 ^{A}
Bryophyte & Lichen SR	6.6 ± 0.3 ^{AB}	5.9 ± 0.4 ^{B}	4.0 ± 0.5 ^{C}	7.6 ± 0.5 ^{A}	8.0 ± 0.8 ^{A}
Open SR	3.4 ± 0.4	4.1 ± 0.8	3.5 ± 0.7	2.3 ± 0.4	5.8 ± 1.6
Simpson's Diversity	0.80 ± 0.01 ^{A}	0.79 ± 0.01 ^{AB}	0.61 ± 0.05 ^{B}	0.70 ± 0.04 ^{B}	0.82 ± 0.03 ^{AB}
Berger-Parker Evenness	0.33 ± 0.02 ^{A}	0.34 ± 0.02 ^{AB}	0.55 ± 0.05 ^{C}	0.45 ± 0.04 ^{BC}	0.32 ± 0.05 ^{ABC}

4.4 DISCUSSION

4.4.1 Light

The amount of light able to reach the understorey flora in forests or open spaces within forests depends on climatic, topographic and forest characteristics. Cork sites typically received lower levels of above-canopy radiation than Wicklow because of higher levels of cloud cover (Met Eireann 2005). Site slope and aspect and shading by surrounding topography also greatly influence above-canopy radiation. Forest managers, however, can do nothing about these factors. Forest structural characteristics, such as tree spacing and height, have a greater influence on solar radiation transmitted through or below the canopy. In the forests we surveyed for this study, width of rides and roads was the most important factor determining the amount of light reaching ride or road centre (Figure 10a). Tree height had a negative effect on the residual variation in diffuse radiation left unexplained by ride and road width. In glades, on the other hand, area had less effect on solar radiation reaching the glade centre (Figure 10b). Most of the glades we surveyed were large, compared with ride and road width, and therefore climatic and topographic factors have a stronger role than shading from the distant forest edge. It appears that at less than 2500 m^2 (see Section 4.4.4), the centre of a glade reaches levels of diffuse radiation that are near to maximum. Direct radiation in glades is more variable and more sensitive to differences in glade shape and characteristics of the surrounding forest; for example a group of tall trees on the southwest side of a glade will reduce direct solar radiation, whereas trees on the northeast side will not.

Solar radiation is one of the most important factors operating at the between and within open space scales, but its effects on plant communities are frequently obscured by other environmental factors. Within the forest, earlier BIOFOREST work has found that light levels, as influenced by forest structure, have a significant effect on understorey vegetation in Irish plantation forests (Smith *et al*. 2005). Similarly, we have found in this study that variation in light levels affects the composition and diversity of vegetation in open spaces. Differences in vegetation composition and diversity between glades and rides are largely the result of differences in light between two size classes of open space. Previous work has recommended that the width of rides and roads be 1-1.5 times the height of the adjacent trees to provide sufficient light for open habitat species (Carter & Anderson 1987), the socalled 1:1 rule of thumb (Ferris & Carter 2000b). However, this study has found that the width : height ratio is not a better predictor of transmitted light or plant diversity than ride/road width alone. This is most likely because our study sites were located in mountainous areas, largely on sloping ground. For a given ride or road width, the same tree height will provide more shade on a north-facing slope than a south-facing one. Ferris and Carter (2000b) state that the 1:1 rule "applies best in southern Britain and on level ground." For the purposes of devising simple management prescriptions, focusing on absolute open space width would provide a better target than width : height ratios.

In general, higher light levels in larger open spaces enable the presence of open habitat plant species (Figure 2 and Figure 3), increase vascular plant species richness and decrease bryophyte species richness. Previous research in Ireland (Mullen *et al*. 2003a) and Britain (Buckley *et al*. 1997; Sparks *et al*. 1996) has found similar results. Vegetation evenness was found to be lower in well-lit spaces, but also in poorly lit narrow rides (e.g. Group D in Table 7). At either extreme, fewer plant species are able to compete successfully for space, and the vegetation tends to become dominated at one extreme by one or a few vigorous graminoid species or at the other by shade-tolerant mosses. In this study, purple moor grass (*Molinia caerulea*) was the dominant species in most glades and large rides in Cork, whereas bent-grass species (mainly *Agrostis capillaris*) were dominant in Wicklow.

Not only does light vary among open spaces, but it also varies within open spaces, with implications for biodiversity as seen in Figure 5. Glade and ride edges effectively act as smaller open spaces, with higher evenness, higher bryophyte species richness and therefore higher diversity as measured by Simpson's index. In fact, the true differences between glade edges and glade centres are underestimated in this study, since we focused our sampling on the north and east edges of glades, which would be less shaded than the south and west.

The size of open spaces may have other effects on plant biodiversity in addition to those caused by light. One is the well-known species-area effect where increases in area lead to increases in species richness. In part, this is because larger areas have more scope for microsite heterogeneity. Island biogeography may be another reason, whereby larger islands (in the ecological sense) support greater numbers of species because of higher probabilities of immigration and lower probabilities of extinction (MacArthur & Wilson 1967). In this study, the species-area effect was observed in the number of species recorded in the complete open space lists increasing with larger sampling areas. An intriguing result is the increase in bryophyte and lichen species richness in glade centre plots with increasing glade area (Figure 9b). At first, this appears to contradict other findings that bryophyte and lichen species richness declines with increasing light levels. But since solar radiation does not increase very much at glade centre with increasing area, the relationship is probably due to factors other than light. It may be that larger glades have a larger pool of species available for "selection" for a given, defined sample area.

4.4.2 Environment and Management

4.4.2.1 Soil and Climate

Environmental factors besides light have strong influences on the vegetation composition and diversity of open spaces in plantation forests. Species compositions in glades and rides were primarily distinguished by geographical location. Cork sites were mostly on deep peats, whereas Wicklow sites were mainly on podzols or peaty podzols. There are also important climatic differences between the two site clusters. Within each of the geographical clusters, soil factors such as pH, drainage, organic carbon and probably mineral nutrients were most closely correlated with the main axis of compositional variation (Figure 2 and Figure 3), whereas transmitted radiation was mainly associated with the second axis of variation. Several soil factors also had significant relationships with biodiversity metrics in glades and rides (Table 3). These differences among open spaces interact with variation in solar radiation, thus obscuring the relationships between either set of factors and biodiversity metrics.

4.4.2.2 Roadside Features

In forest roadsides, soil conditions vary more than in glades and rides because of differences in roadside topography and road surfacing material. Roadsides are complex environments that can provide suitable microhabitats for a wide range of species, with the result that roads support much higher species richness of vascular plants and higher Simpson's diversity than glades or rides (Table 1). Because of this complexity, the effects of light on biodiversity metrics were generally obscured in this study. However, Figure 4 shows that the influence of light on the composition of roadside vegetation is important.

Immediately beside the road surface receiving most vehicular traffic is the verge, a frequently disturbed zone occupied mainly by ruderal plant species. Usually road gravel is knocked into the verge by passing traffic or washed into it by rain. If the gravel is not the same kind of stone as the soil parent material, such as limestone gravel in a forest on granite, it can have dramatic effects on the chemical properties of the verge soil and therefore on the vegetation of the verge (Section 4.3.2.1.2). Use of limestone gravel in a site with acid soils produces a strange roadside fringe of daisies, yellow-clover and other relatively calcicole species which contrast with the calcifuge species growing a meter or so further from the road surface. Even if the gravel is local stone, it may alter the physical characteristics of the verge. In deep peat sites, the road bed is made of sub-peat mineral soils; different species will be able to colonise the road surface and mineral soil verge than are found on the peaty road setback. The combination of soil effects and disturbance produce a road verge vegetation that is relatively rich in vascular plant species, particularly ruderal species and plants of open habitats and is heterogeneous in structure (Table 5).

A drainage ditch is frequently found beside the road verge. Ditches are normally at least seasonally wet, and can therefore support more species of plants that prefer wet, open conditions than most other parts of a plantation forest. We also found that many ditches are influenced by gravel spillage, leading to a wetland flora with more base-rich affinities than would normally be expected. Ditches also support high vascular plant species richness and tend to have more even vegetation structures (Table 5). Vertical roadside banks, where present, often provide habitat for many bryophyte species and ferns, particularly when damp or shaded. Level road setbacks between the above features and the forest edge typically had similar vegetation to rides of comparable width. In four Cork sites, however, we found that some roadsides had developed a distinctive vegetation structure characterised by high cover of bramble that supported relatively high species richness of vascular plants, bryophytes and lichens (Table 6 and Table 7).

4.4.2.3 Influence of Past and Present Management

There was very little active management of the open spaces used in this study. The only management we observed or were informed of by local forest managers was recent resurfacing of roads and ditch clearance in four sites and deer culling in one site. These practices made no detectable difference to vegetation biodiversity. Management of the forest, however, did have some influence on plant biodiversity in the open spaces. The association of forest age with vascular plant species richness, and to a lesser degree evenness and Simpson's diversity, in some open spaces (Table 3 and Table 4) suggests that the plant communities there are responding to environmental changes, as would be expected. As the forest matures, the open spaces retained within the forest become more sheltered and shaded, and possibly more humid. The positive relationship between thinning intensity and vascular plant species richness and the negative relationship with bryophyte and lichen species richness in some open spaces (Table 3 and Table 4) may also reflect forest maturation, as thinning intensity and forest age are correlated. On the other hand, thinning serves to open up the forest adjoining the open space, letting in a little more light. The strongest relationship between plant diversity and management was in fact with pre-forest management: the decrease in vegetation evenness and Simpson's diversity on cutover peat in Cork (Table 3). The decrease in evenness is due to greater dominance of purple moor-grass in cutover areas.

4.4.3 Plantation Scale Factors

Factors operating at the plantation scale generally had few detectable effects on plant biodiversity of the open spaces. One such factor that was investigated was dispersal. If ongoing immigration of species into the surveyed open spaces from other open spaces in the landscape was an important factor, the analyses of the open space categories within set distances of the survey locations should have yielded some evidence. The only apparent relationship, the area of certain kinds of open space within 500 m of forest roads, seems to be the result of correlations with road width and light (Section 4.3.1). The relationship between vascular plant species richness and forest age in Wicklow rides and glades (Table 3) discussed above may be partly attributable to species accumulation processes, but it is impossible to be certain. Dispersal was certainly an important factor in the past in at least some cases, for example permitting colonisation of the margins of limestone-gravelled roads by typical farmyard ruderal species and calcicoles. Nevertheless, it seems clear that, at the stages of forest maturity we surveyed, the importance of present-day dispersal mechanisms for vegetation composition and diversity is less than that of factors operating at the scale of the open space.

Other factors that would be expected to operate at the plantation scale are obscured by regional differences in this study. Density of grazing mammals, including deer, hares and trespassing sheep, would be expected to vary at scales comparable to their ranges: the plantation scale or greater. The impact of deer was much greater in the Wicklow sites than in Cork, and is probably one of the factors in addition to climate that differentiate the vegetation of the two site clusters (cf. Figure 4). In addition, grazing appeared to be more severe in glades than in rides in the Wicklow sites (Figure 3), probably because of greater abundances of food.

Factors operating at smaller scales can have effects that also manifest at the plantation scale. For example, the higher overall bryophyte and lichen species richness recorded for open space in MUCK compared with LUGG and GLAN is in part because no glades were present in MUCK and all but one of the rides and roads were narrow and well-shaded. (Strictly speaking, this is a sampling effect from failure to sample glades, but the sampling effect arises from the fact that there were no glades to sample.)

4.4.4 Open Space Management for Plant Biodiversity

The primary conservation goal of including areas of open space within plantation forests is to maximise biodiversity at the plantation scale by providing habitat for species not able to persist under the low light levels found in the forest understorey. The optimal sizes and types of open spaces incorporated into forests should therefore be those that increase biodiversity of those species, rather than shade-tolerant species. However, because of the continuous relationship between light levels and biodiversity metrics (see Section 4.4.1) and the variation in that relationship caused by other environmental factors, it is difficult to prescribe optimal open space sizes.

It appears from the relationship between ride and road width and diffuse solar radiation that widths from 15 to 25 m may be best (Figure 10a). Below 15 m width light levels decrease more steeply, and above 25 m there is much less increase in light for a given increase in width. It is important to note that the widths discussed in this chapter are from tree trunk to tree trunk. Actual canopy openness will be less, because of overhanging branches. Branch length increases with the height of the tree and the width of the open space; it averages 3.6 m (± 0.2 m se) over all the open spaces in this study and averages 4.9 m (± 0.2 m se) for trees greater than 20 m height. In a 10 m wide ride, the first value would allow only 3 m of open sky while the second would allow virtually none. Where multiple habitat/topographic features are present along roadsides, greater widths in places would be preferable so that some of these features are well-lit. However, because shaded banks can support a number of bryophyte and fern species, uniformly wide road setbacks may not be optimal.

The smallest glade we surveyed was 1220 m^2 in area, which appears adequate to maximise (or nearly so) the amount of diffuse radiation reaching the centre (Figure 10). Given that ride and road widths of about 25-30 m receive similar amounts of light to the smallest glades surveyed, a glade area of 625-900 m^2 (the above ride and road widths squared) may be sufficient. Regardless of light at the glade centre, parts of the glade nearer the forest edge will receive less light, depending on the shape and orientation of the glade and local topography. This could be beneficial to biodiversity, however, particularly if the central parts of the glade are likely to be occupied by competitive grasses. Creating glades with high edge to area ratios would similarly enhance biodiversity, by increasing the transition zone where neither the most competitive open space species nor most shade-tolerant species can dominate.

When choosing areas in which to create glades or to retain open habitats, it would be best to select pre-existing open habitats which already have relatively good biodiversity value. If no such areas stand out, then selecting areas with variation in microtopography, soil characteristics, etc. would help to maximise the number of species that can occupy the area.

Roads supported a greater diversity of plants as a result of variation in roadside features, and therefore the management of roadside vegetation should take this into account. Road maintenance should avoid disturbance of marginal vegetation as far as possible, for example cleaning drains only when necessary. Mechanical clearance of roadside scrub for safety purposes would be preferable to herbicide use.

It is worth noting that none of the plant species and community types encountered in this study were rare or of particular conservation value. Where a forest plantation is located in a landscape with a substantial number of high-quality open habitat plant communities, the open space within a plantation will add little to biodiversity at a landscape scale. In fact, it may be the plant species and communities of the forest understorey that serve to increase diversity at the landscape scale. In such situations, provision of open space within a plantation beyond that required for silvicultural purposes (e.g. roads) is unnecessary. Similarly, if characteristic forest species of conservation concern occur in a given plantation, biodiversity management should focus on these species, rather than those of open spaces. Of course the open spaces, such as roads, that are incorporated as needed should be managed in a manner that will maximise their biodiversity.

4.4.5 Further research required

Topics for further research that would provide useful additional insights include the contributions of (a) windthrow gaps and (b) spontaneous scrub in open areas to forest biodiversity, experimental manipulation of grazing intensity in open spaces and the vegetation diversity of (a) small (< 2000 m^2) glades within forests, (b) open spaces within broadleaf plantations and (c) clearfelled areas retained as open spaces in second-rotation forests.

4.5 CONCLUSIONS

Glades, rides and roads in Irish plantation forests can support reasonably diverse terrestrial plant communities. The primary causes of variation in vegetation composition and diversity were soil and climate factors. The vegetation of glades and wide rides was distinct from that of narrow, more shaded rides. The sides of forest roads supported species that did not occur in glades and rides as a result of roadside topographical features and in some cases the influence of limestone gravel. In general, vascular plant species richness increased and bryophyte and lichen species richness decreased with increasing solar radiation. Vegetation evenness and Simpson's diversity were lower both in well-lit, grass-dominated situations and in heavily shaded, bryophyte dominated conditions. Bryophyte species richness and vegetation diversity increased in the forest-open space ecotone at the edges of glades. Plantation scale factors, such as dispersal, appear to have little influence on the biodiversity of open spaces in mature forests, in contrast to the factors at scales of between- and within-open space.

To benefit terrestrial flora of open habitats, rides and roads should be a minimum of 15 m in width. Depending on local conditions, glade areas of 625-900 m^2 should be sufficient to have at least part of the glade well-lit. Creating glades with high ratios of edge to area will probably increase biodiversity by increasing the amount of ecotonal habitat. Selection of areas for open space retention should focus on areas of high biodiversity or environmental heterogeneity. Roadside vegetation should be managed in a sensitive fashion.

5 EPIPHYTES

5.1 INTRODUCTION

An epiphyte is an organism living on a plant, or in the dead outer tissues of a plant, without drawing water or food from its living tissues (Barkman 1958). Epiphytes can be found in every major group of the plant kingdom but in Ireland the main groups of epiphytes are mosses, lichens and liverworts.

The value of epiphytes as biological indicators is well known (e.g. Richardson 1987) and they have also been found to constitute a major component of the total botanical diversity of semi-natural woodland in Ireland (Kelly 2000). However, there is little published information on epiphytes in Irish woodlands. Richards (1938) described the bryophyte communities of Derrycunihy oakwood in Killarney on 7 habitats including tree bases, tree trunks and tree branches. Kelly (1975) studied epiphytes in the Killarney oak and yew woods between 0.5m and 14m height on tree trunks. Kirby (1982) listed the epiphytes occurring up to 2m in Shannawoneen wood, Co. Galway and Folan & Mitchell (1970) listed the lichens and lichen parasites of Derryclare Wood, Co. Galway. Fox *et al.* (2001) studied the epiphytes at 1.2m on 21 oak trees in Brackloon wood, Co. Mayo, as well as the lichens of the entire surface of a felled oak tree in the same wood.

There is no known published information on epiphytes in Irish forestry plantations. There is also little published work on the epiphytes of forestry plantations in Britain, or in the rest of Europe. Watson (1936) listed the bryophytes and lichens (including epiphytic species) of British woods, including coniferous woods. More recently, in their comparison of the diversity of spruce and pine plantations with semi-natural pine and oak woods in Britain, Humphrey *et al.* (2002) recorded the bryophytes and lichens occurring on mature trees up to 2m. Vanderpoorten *et al.* (2004) examined the diversity of obligate epiphytic bryophytes in the Semois river basin (Belgium, France), which contains large areas of Norway spruce (*Picea abies)* plantations.

Open spaces within forestry plantations can act as refuges for flora and fauna excluded from forest compartments and can be very species rich (Sparks *et al*. 1996). However, the presence of epiphytes within open spaces themselves is dependent on the presence of trees and shrubs. Neitlich and McCune (1997) found that gaps containing broadleaved trees and shrubs were important for epiphytic lichen diversity in managed conifer stands in Oregon. Where trees or shrubs are absent from an open space, it is the effect of the open space on the epiphytes of the planted trees surrounding the open space that is of interest.

The aims of this chapter are:

- To assess the biodiversity of epiphytes on trees adjacent to open spaces in plantation forests,
- To investigate the major environmental and structural factors influencing epiphyte biodiversity, and
- To recommend planning and management measures that can enhance the biodiversity of epiphytes adjacent to open spaces in plantation forests

5.2 METHODS

5.2.1 Study sites

To reduce the effect of large-scale environmental variation, 12 study sites were chosen in two geographic clusters: the Wicklow/Dublin Mountains (Wicklow cluster) and Cork and neighbouring parts of Kerry and Limerick (Cork cluster). (See Figure 1 and Table 8). All sites were commercially mature Sitka spruce (*Picea sitchensis*) plantations. The Wicklow cluster sites were typically well-drained upland sites with a mean elevation of 422m (±27se). The Cork cluster sites were typically poorly drained sites on peaty soils with a mean elevation of 288m (±23se).

Epiphytes were studied on trees adjacent to an open space at each site. Glades with south facing edges were preferentially selected. However, at three sites, where glades were not present or not suitable for sampling, the most suitable east-west running road or ride was selected (Table 8).

Table 8. The site code, location, age (*years after planting at time of study) type of open space and aspect of the edge studied at each site.

Site name	Site Code	County	Grid Ref.	Age*	OS Type	Edge aspect (°)
Wicklow Cluster:						
Lugg	LUGG	Dublin	T031243	31	Glade	180
Athdown	ATHN	Wicklow	T077 159	28	Glade	173
Ballysmuttan	BMUT	Wicklow	T048 145	38	Glade	160
Ballinastoe	STOE	Wicklow	T180 084	29	Glade	172
Mucklagh One	MUCK	Wicklow	T084 865	42	Road	155
Ballycurragh	CURA	Wicklow	T061 831	42	Glade	138
Cork Cluster:						
Knocknagoum	KNOC	Kerry	V958 217	32	Ride	168
Cleanglass	CLEA	Limerick	W244 219	31	Glade	211
Reanahoun	REAN	Cork	W256 200	39	Road	190
Meentinny	MEEN	Cork	W245 135	32	Glade	162
Glannaharee	GLAN	Cork	W444 887	38	Glade	205
Carrigagulla	CARR	Cork	W384 835	43	Glade	149

The sub-compartments in which epiphytes were studied were selected so that at least 80% of their net productive area was under Sitka spruce. In fact the sub-compartments at only two sites contained less than 100% Sitka spruce: KNOC, which contained 95% Sitka spruce and 5% alder (*Alnus glutinosa*) as edge planting; and CLEA, which contained 80% Sitka spruce and 20% lodgepole pine (*Pinus contorta*) as an intimate mixture.

5.2.2 Epiphyte sampling

Fieldwork was carried out between July and November 2003. All epiphyte surveying took place on the north side (i.e. south-facing side) of each open space. Epiphytes were studied on a pair of trees at each site, one tree at the edge of the open space and one tree in the forest interior. Each tree was identified by the site code (Table 8) followed by a number – 1 for the edge tree and 2 for the interior tree. The edge trees studied directly adjoined each open space, with the exception of KNOC1 where alder had been planted between the tree studied and the open ride. The forest interior trees studied were located at a distance from the edge of the open space greater than or equal to the height of the edge trees at that point. At two sites it was not possible to locate the interior tree a full tree height from the edge. The width of the sub-compartments at MUCK, between two roads; and CLEA, between the glade and an area of thinning, were not sufficient. The trees studied were not located adjacent to rows that had been entirely removed during thinning with the exception of BMUT2.

Epiphytes were studied in plots located on the trunk and branches at 4 different height zones in the tree– tree base (B), lower (L), middle (M) and upper (U). The sampling design is illustrated in Figure 11. The tree base zone began at the point where the trunk emerged from the soil or needle litter and reached to 0.5m above this point. In a number of cases this point was at a different height on the two sides of the trunk. All subsequent measurements were taken from the uphill side of the tree. The lower zone was centred on breast height (1.3m), and the middle and upper zones were centred on 1/3 and 2/3 of the height of the tree, respectively. Tree-climbing techniques were used to study the middle and upper plots. This involved the use of a rope and harness and climbing spurs. Trunk plots were located on the side of the trunk facing the open space and the opposite side (referred to as south and north sides). Plots were 50cm high and the width varied from a maximum of 25cm, to that required to sample a half cylinder of the trunk. A list of epiphyte species occurring in the trunk plots was made with percentage cover estimated to the nearest 5%. Below 5% two different cover-abundance units were distinguished: 3% (indicating cover of 1-5%), and 0.5% (indicating cover <1%). Total percentage cover of bryophytes, lichens, vascular epiphytes, others (algae, fungi etc.), needle litter, and total percentage bare bark were also recorded.

In the middle and upper zones the trunk plots were centred on a branch whorl. A branch from the north side and a branch from the south side of the whorl were removed for study on the ground. Three plots, 25cm long by 50cm wide, were studied on each branch. These plots ran perpendicular to the main axis and included the main axis and that side of the branch with the most cover of branchlets (Figure 11). The upper half cylinder of branches was studied. The first plot was placed at the base of the branch, the third plot was placed near the tip of the branch, but did not include the last 2 years growth on the main axis, and the second plot was centred half way between the previous two plots. Percentage of the plot taken up by branches (to the nearest 1%) and by needles was recorded and a list of epiphyte species was made with percentage cover estimated to the nearest 1%. Species with a cover less than 1% were assigned 0.5%. Total percentage cover of bryophytes, lichens, vascular epiphytes, others (algae, fungi etc.), and total percentage bare bark were also recorded.

5.2.3 Environmental and structural data sampling

At each site the slope and aspect of the site and the aspect of the edge at which trees were studied were recorded. Tree density and DBH were recorded from two 10m x 10m plots located within the forest. The first plot had its outer edge centred on the edge tree studied and ran perpendicular to the edge. The second was centred on the interior tree studied and was perpendicular to the edge. Stand basal area was later calculated from these data.

The area of glades was measured using aerial photographs and the width of rides and roads was measured from tree trunk to tree trunk. The light environment of the open spaces was measured using hemispherical photographs taken in the centre of each open space. See Section 4.2.3 for details.

The amount of habitat near the surveyed trees that was not under mature conifer forest was measured in nine categories: area of broadleaf scrub, road, undeveloped plantation, clearfell, windthrow, young forestry, unplanted open space within the plantation, external open space and length of rides. The habitat categories were mapped using aerial photographs, and habitat areas and ride length within 50 m, 100 m, 200 m and 500 m of each tree were calculated using ArcView GIS.

For each tree studied, the DBH (at 1.3m), tree height, heights to first live branch and base of live crown and the distance of the tree from the open space edge were recorded. The height above ground, girth and inclination at the centre of each trunk plot were recorded. For branches, the height above ground (at insertion), inclination, total branch length and the length of branch covered by foliage were recorded, as well as the distance from the trunk and diameter of the main axis at the centre of each plot.

5.2.4 Species Identification

Some of the epiphyte species encountered could be identified in the field, while others needed to be confirmed by microscopy. The plant species nomenclature used in this section follows Stace (1997b) for vascular plants, Smith (2004b) for Mosses, Paton (1999) for liverworts and Index Fungorum (2004) for lichens. Non-lichenised fungi were not recorded. Most epiphytes were identifiable to species level. However, a few were only identifiable to generic level due to lack of fruiting bodies or poor development. Those species identified only to generic level were treated in such a way that their inclusion did not cause an overestimation of species richness. If no other species belonging to the genus of the unidentified species was present in the unit under consideration, the specimen was included and identified to generic level only. If other species of the same genus were present, the data were amalgamated and the cover of the most abundant species for the genus was increased accordingly. Some sterile crustose lichens were not identifiable to genus level and were excluded from analyses.

5.2.5 Data analysis

5.2.5.1 Biodiversity measures

Total, bryophyte and lichen species richness; Simpson's diversity index and the Berger-Parker index of evenness were calculated for all sites and trees. The reciprocal of both Simpson's index (1/D) and the Berger-Parker index (1/d) were used so that increases in the value of the indexes represented increases in diversity and evenness, respectively (Magurran 2003).

The effect of environmental and structural variables on the biodiversity measures were investigated using linear regression. Total area in eight habitat categories (area of broadleaf scrub, clearfell, road, undeveloped plantation (areas of crop failure), windthrow, young forestry (pre-canopy closure), unplanted open space within the plantation and external open space) and total ride length within 50m, 100m, 200m and 500m of each tree were used in regression analyses of the biodiversity measures. In tests where multiple correlations are carried out there is an increased probability of making a Type 1 error. Bonferroni corrections can account for this error by means of adjusting the significance level by correcting for the number of tests. Bonferroni corrections were applied to the regression analyses of the biodiversity measures on the nine habitat variables.

Paired t-tests, or their non-parametric equivalents, were used to investigate differences between edge and interior trees for the five biodiversity measures.

5.2.5.2 Species composition

The epiphyte species composition on each tree was investigated using NMS ordination (Legendre & Legendre 1998). Presence-absence, rather than percentage cover data were used since the surface area available for epiphyte colonisation differed between trunk and branch plots. The distance measure used was Sørensen distance. For each ordination, forty preliminary ordinations were carried out, each one beginning with six dimensions and then stepping down in dimensionality to one. Monte Carlo tests were performed using 50 runs with randomised data. The optimal number of dimensions was determined, and the best of the preliminary ordinations with that number of dimensions was used as the starting configuration for the final ordination.

Groups of trees with similar epiphyte assemblages were identified using flexible-beta hierarchical cluster analysis on Sørensen distance measures with the parameter β set to equal -0.25. This setting of β produces a solution intermediate between single-linkage and complete-linkage agglomerative clustering (Legendre & Legendre 1998).

Indicator species analysis was used to identify species characteristic of the cluster analysis groups and of edge and interior trees. Good indicator species should be found mostly in a single cluster and should be present on most of the trees belonging to that cluster. The indicator value is 100% when a species is observed at all sites belonging to a single cluster and in no sites in the other clusters (Legendre & Legendre 1998). A random reallocation procedure of sites among the site groups was used to test the significance of the indicator values (Monte Carlo test).

ANOVAs, followed by post-hoc Tukey tests, or the non-parametric Kruskall-Wallis H Test followed by Mann Whitney U tests, were used to examine differences between the cluster analysis groups and between the different trunk plots on the edge and interior trees.

Prior to parametric analyses, variables were inspected for conformity to the assumptions of parametric statistics. Multivariate analyses were conducted using PC-Ord (McCune & Mefford 1997a) and univariate analyses were performed with SPSS 11.0 (SPSS 2001)

5.3 RESULTS

5.3.1 Site Diversity

A total of 68 species of epiphytes were found on the 24 trees – 28 bryophyte, 39 lichen and 1 vascular plant species. The vascular plant species was a juvenile pteridophyte, not identifiable to genus level. A list of all the epiphyte species recorded can be found in Appendix 6. The species richness, Simpson's diversity and Berger-Parker evenness of all sites is shown in Figure 12. The mean site species richness was 22.58 (+/- 1.80se). The most species rich site was KNOC with 36 species and the most species poor site ATHN with 16. The most diverse site according to Simpson's Diversity index was KNOC with a value for 1/D of 18.74. The least diverse site was CURA with a 1/D value of 8.06. CLEA was the most even site, as measured by the Berger-Parker index, with a value for 1/d of 7.91 and CURA was the least even site with a 1/d value of 3.54.

5.3.2 Tree Diversity

The species richness, Simpson's diversity and Berger-Parker evenness of all trees is shown in Figure 13. The mean tree species richness was 16.25 (+/- 0.92se). The most species rich tree was KNOC2 with 27 species and the most species poor tree was ATHN1 with 8 species. The most diverse tree according to Simpson's Diversity Index was BMUT1 with a 1/D value of 23.00. The least diverse tree was BMUT2, the interior tree from the same site with a 1/D value of 6.04. CARR2 was the most even tree according to the Berger-Parker index with a value for 1/d of 8.14 and BMUT2 was the least even tree with a 1/d value of 2.82.

5.3.3 Plot Diversity

A total of 478 plots were studied – 192 trunk plots and 286 branch plots. The species richness in plots ranged from 0 to 13 with a mean of 2.80 (+/- 0.10se). The most species rich plot was the middle branch plot at the middle plot height from the south side of KNOC1. A total of 60 plots contained no epiphyte species. Seventeen species occurred in only one plot and 23 species on only one tree. Only 16 species occurred in more than 5% of plots. These species are shown in Table 9.

5.3.4 Rare species

There is currently no Red Data Book for bryophytes and lichens in existence for Ireland, although all-Ireland bryophyte and lichen Red Data Books are in the process of being compiled. One of the species recorded, *Daltonia splachnoides*, is listed as vulnerable on the British Red Data List (Church *et al*. 2001) and is likely to appear on the Irish Red List (D.

Holyoak, pers. comm.). Another species recorded in this study, *Plagiothecium laetum*, is also likely to appear on the Irish Red List (D. Holyoak, pers. comm.) and its two records in this study constitute only the 3rd and 4th record of this species in Ireland.

Table 9. A list of the epiphyte species occurring in more than five percent of plots

Species	% of plots
Metzgeria temperata	32.22
Hypnum jutlandicum	29.29
Hypotrachyna revoluta	26.57
Fuscidea lightfootii	18.83
Ulota crispa agg.	15.90
Dimerella lutea	13.18
Lejeunea ulicina	10.67
Lepraria incana	9.00
Lophocolea bidentata	8.58
Colura calyptrifolia	7.95
Hypogymnia tubulosa	7.32
Frullania dilatata	6.90
Metzgeria furcata	6.90
Kindbergia praelonga	6.49
Gyalideopsis anastomosans	6.07
Ramalina farinacea	5.86

5.3.5 Relationship between diversity and habitat variables

Linear regression of the environmental and structural variables showed significant negative relationships for total species richness (r^2=0.459, p<0.001) and bryophyte species (r^2=0.540, p<0.001) with site elevation and for bryophyte species richness with slope (r^2=0.295 p=0.006) (Figure 14). However, the Wicklow cluster sites had a significantly higher slope (p=0.004) and elevation (p=0.001) than the Cork cluster sites, as well as having a significantly lower total (p<0.001) and bryophyte (p<0.001) species richness. For this reason the Cork and Wicklow sites were analysed separately and, although there were no relationships between species richness and site slope, the negative association between bryophyte species richness in the Wicklow sites and elevation was close to significance (r^2=0.300 p= 0.065) and there was a significant negative relationship with total species richness (r^2=0.426, p=0.021) and lichen species richness (r^2=0.426 p=0.033) in the Cork sites. There were also significant positive relationships for total species richness (r^2=0.459, p<0.001) and bryophyte species richness (r^2=0.308, p=0.005) with tree density and significant negative associations of total species richness (r^2=0.247, p=0.013) and bryophyte species richness (r^2=00.313, p=0.004) with average DBH in the 10m x 10m plots surrounding the studied trees Figure 15). There was also a significant negative relationship between average DBH and tree density (r^2=0.771, p<0.001). However there were no significant associations of the diversity measures with age of the plantation, site aspect, width of the open space, glade area or canopy openness at the centre of the open space.

The regression analysis of the total area in eight habitat categories (see 5.2.5.1) and total ride length within 50m, 100m, 200m and 500m of each tree, with Bonferroni corrections applied, showed no significant associations for the biodiversity measures.

5.3.6 Species composition

The NMS ordination of the species presence-absence data for the 24 trees shown in Figure 16 explains 55.5% of the variance in the original data. The variables with significant correlations with axis 1 and axis 2 are shown in Table 10. The main contrast was between sites of high elevation and slope and those of low elevation and slope. The length of road within 100m was also positively correlated with axis 1. The second NMS axis mainly distinguished between sites with a high tree density and more open stands which receive more above-canopy diffuse radiation. The proximity to areas of broadleaved scrub within 500m was correlated with both axes, however, scrub was only present at three of the study sites. The species with significant correlations with axis 1 and axis 2 are shown in Table 11. Species associated with damp shaded conditions and those associated with well-lit exposed situations were positively and negatively associated with both axes.

Table 10. Pearson correlations of the environmental variables with the NMS ordination axes of Figure 16.

Variable	Pearson correlation with Axis 1	Pearson correlation with Axis 2
Elevation	0.883***	-0.041
Slope	0.724***	0.011
Tree density	-0.102	0.640**
Average DBH	0.037	-0.590**
Diffuse radiation	-0.231	-0.554**
Broadleaved scrub within 500m	-0.443*	-0.523**
Length of road within 100m	0.443*	0.028

*= < 0.05; ** = < 0.01; *** = < 0.001

Table 11. Pearson correlations of the epiphyte species with the NMS ordination axes of Figure 16.

Species	Pearson correlation with Axis 1	Pearson correlation with Axis 2
Gyalideopsis anastomosans	0.669***	-0.136
Hypogymnia tubulosa	0.671***	0.456*
Micarea peliocarpa	0.471**	-0.245
Plagiothecium laetum	0.433*	0.176
Metzgeria furcata	-0.658***	-0.181
Kindbergia praelonga	-0.664***	-0.137
Frullania dilatata	-0.664***	0.230
Ulota crispa agg.	-0.600**	-0.024
Lejeunea ulicina	-0.521**	0.154
Hypnum resupinatum	-0.508**	0.378
Usnea esperantiana	-0.508**	0.378
Metzgeria fruticulosa	-0.491**	0.042
Physcia tenella	-0.457**	0.056
Hypotrachyna revoluta	0.421	0.623**
Evernia prunastri	-0.425*	0.512**
Colura calyptrifolia	-0.435*	0.450*
Ramalina farinacea	-0.424*	0.429*
Dimerella lutea	0.107	-0.670***
Pseudotaxiphyllum elegans	-0.181	-0.443*

*= < 0.05; ** = < 0.01; *** = < 0.001

Cluster analysis was used to identify discontinuities in the assemblage structure revealed by the ordination. The cluster analysis, combined with the NMS ordination, suggested that the best classification was the separation of the trees into four groups. With the four-cluster solution over 35% of the information was retained. The cluster analysis groups were compared in relation to a number of diversity, percentage cover, structural and environmental factors. Significant differences were revealed among the cluster analysis groups for total, bryophyte and lichen species richness; slope, elevation, tree density and average DBH. Post-hoc tests clarified these differences. This information is summarised in Table 12.

Table 12. Mean (± standard error) of species richness (SR), Simpson's diversity, Berger-Parker evenness, mean % trunk cover, structural and environmental variables for trees within the different cluster analysis groups. There is no significant difference between cluster analysis groups if designated with the same letter [a, b or c].

	Group 1 (n=4)	Group 2 (n=7)	Group 3 (n=2)	Group 4 (n=11)	ANOVA $df=_{3,23}$
Total SR	13.00 [a] (+/- 0.91)	18.14[a] (+/- 1.10)	26.00 [b] (+/- 1.00)	14.45 [a] (+/- 1.02)	F=11.00***
Bryophyte SR	6.75 [ab] (+/- 0.85)	9.86 [a] (+/- 0.51)	12.00 [a] (+/- 2.00)	5.91 [b] (+/- 0.90)	F=6.21**
Lichen SR	6.25 [a] (+/- 0.95)	8.00 [a] (+/- 0.72)	14.00 [b] (+/- 3.00)	8.55 [ab] (+/- 0.85)	F=4.25*
Diversity (1/D)	11.76 (+/- 3.84)	14.10 (+/- 1.17)	17.58 (+/- 0.92)	12.63 (+/- 0.90)	ns
Evenness (1/d)	4.60 (+/- 1.17)	6.39 (+/- 0.47)	7.55 (+/- 0.02)	5.32 (+/- 0.32)	ns
Total % cover	12.20 (+/- 2.85)	33.29 (+/- 8.07)	20.38 (+/- 2.62)	34.90 (+/- 7.62)	ns
Bryophyte % cover	8.281 (+/- 3.28)	32.02 (+/- 8.36)	19.91 (+/- 2.78)	31.44 (+/- 7.76)	ns
Lichen % cover	4.20 (+/- 1.53)	1.65 (+/- 0.47)	0.44 (+/- 0.31)	2.70 (+/- 0.80)	ns
Slope (°)	5.00 [a] (+/- 1.16)	4.57 [a] (+/- 1.34)	6.00 [ac] (+/- 0.00)	14.91 [c] (+/- 1.76)	H=13.78**
Elevation (m)	308.75 [ab] (+/-10.87)	297.14 [a] (+/-21.90)	180.00 [b] (+/-0.00)	440.91 [c] (+/-23.95)	H=15.30**
Tree density (no./100m²)	4.25 [a] (+/- 0.63)	18.57 [b] (+/- 1.77)	18.50 [ab] (+/- 1.50)	14.45 [b] (+/- 2.13)	H=11.18*
Average DBH (cm)	35.74 [a] (+/- 1.38)	20.43 [b] (+/-1.07)	17.45 [ab] (+/0.03)	25.50 [b] (+/- 1.99)	H=12.55**
Basal Area (m²/100m²)	0.46 (+/- 0.10)	0.66 (+/- 0.07)	0.51 (+/- 0.07)	0.69 (+/- 0.05)	ns

*= < 0.05; ** = < 0.01; *** = < 0.001

Indicator species analysis revealed 10 species with significant indicator values for these four groups. These 10 species are shown in Table 13 along with their indicator values.

Table 13. Indicator Species Analysis of epiphyte assemblages in the 4 cluster groups. The maximum indicator value is in bold type for each species.

Species	Group 1	Group 2	Group 3	Group 4	P value of max indicator
Hypnum resupinatum	0	0	**100**	0	0.004
Usnea esperantiana	0	0	**100**	0	0.004
Evernia prunastri	0	0	**92**	1	0.004
Lecanora pulicaris	0	0	**73**	10	0.004
Metzgeria fruticulosa	4	1	**72**	0	0.023
Hypogymnia tubulosa	0	0	17	**67**	0.001
Calypogeia muelleriana	0	**57**	0	0	0.037
Trapeliopsis flexuosa	0	**43**	0	0	0.024
Metzgeria temperata	9	**34**	9	28	0.046
Dimerella lutea	**43**	22	0	17	0.033

Cluster analysis group 1 contained both the edge and interior trees from LUGG and BMUT; both structurally mature Wicklow sites with relatively large widely spaced trees. These were also low elevation sites relative to the other Wicklow cluster sites. This group had significantly lower tree density and significantly higher average DBH than groups 2 and 4. It had the lowest species richness, diversity and evenness of all the groups studied and the lowest total and bryophyte cover. However, it had the highest lichen cover, with particularly high cover of *Dimerella lutea*, the indicator species for this group. *Dimerella lutea* is characteristic of humid shaded situations (Purvis *et al*. 1992).

Group 2 contained trees from all of the Cork sites with the exception of KNOC. They were all forest interior trees with the exception of CLEA1 and CARR1. This group was only exceeded by group 3 in its total and bryophyte species richness, diversity and evenness and had the highest bryophyte cover of all the groups, as well as the highest tree density. Both the bryophyte indicators, *Calypogeia muelleriana* and *Metzgeria temperata,* are characteristic of shaded or damp habitats (Smith 1990; Watson 1981).

Group 3 contained both trees from KNOC. This was the lowest elevation site studied and also the most species rich site. This group had significantly higher species richness than all the other groups. It also had the highest bryophyte and lichen species richness and was the most diverse and even of the groups. However, despite having high species richness, the epiphyte cover was low. It also had high tree density, with only group 2 having slightly greater density. It contained a large number of species not found at any of the other sites or in any other groups, hence the presence of five species with high indicator values for this group. Two of the lichen indicators for this group *Usnea esperantiana* and *Evernia prunastri* are characteristic of well lit windy situations (Dobson 2000; Purvis *et al*. 1992), while the bryophyte *Metzgeria fruticulosa* is characteristic of sheltered or damp situations (Smith 1990). *Evernia prunastri* is also more rarely found in shaded woodland and boggy sites (Dobson 2000).

Group 4 contained the trees from the four remaining Wicklow sites along with the forest edge trees from three sites in Cork. This group had significantly higher elevation than the other three groups and the highest slope. Bryophyte species richness was the lowest of all the groups but lichen species richness was relatively high. This group had the highest total epiphyte cover. The indicator for this group, *Hypogymnia tubulosa*, is characteristic of drier, more open habitats.

5.3.7 The effect of open spaces

5.3.7.1 Effect of open spaces on diversity and cover of epiphytes

A total of 14 species were recorded only on the open space edge trees and 17 species were recorded only on the forest interior trees (Appendix 6). This information is included in Appendix 6. However, 10 species and 13 species respectively were found on only a single edge or interior tree. Of the remaining 4 species occurring only on the edge trees, all were found on only two trees, and of the remaining 4 species occurring only on interior trees, 3 species were found on two trees and one species on 3 trees.

Comparisons of the edge and interior trees from each site in relation to the diversity measures and to epiphyte cover in the trunk plots revealed no significant differences in total, bryophyte and lichen species richness; Simpson's Diversity, Berger-Parker evenness, or epiphyte cover between them.

5.3.7.2 Effect of open spaces on species composition

In the NMS ordination of the 24 trees (Figure 16) the majority of the forest edge trees were positioned towards the top of the second axis, with LUGG1 and BMUT1 the exceptions. Indicator species analysis revealed only a single species with a significant indicator value for either edge or interior trees. *Dimerella lutea* had its maximum indicator value of 63 (p=0.025) for interior trees and an indicator value of 13 on edge trees. This species was also an indicator for cluster analysis group 1, which contained the two edge trees positioned towards the bottom of axis two of the NMS ordination (Figure 16).

5.3.7.2.1 Confounding factors

Comparisons of the stand structural measures taken from the 10m x10m plots surrounding each tree studied revealed differences between edge and interior plots. This information is summarised in Table 14. Average DBH and basal area were significantly greater in the edge plots than the interior plots and the difference in tree density approached significance (p=0.105).

Table 14. The tree density, average DBH and basal area from the 10x10m plots. Mean (se). Pairs having the same letter ([a or b]) indicates no significant difference between them (i.e. p>0.05).

	Edge (n=12)	Interior (n=12)
Tree density (no./m^2)	15.17[a] (+/-2.17se)	13.42[a] (+/-2.08se)
Average DBH (cm)	26.26[a] (+/-2.34se)	23.85[b] (+/-1.90se)
Basal area (m^2)	0.75[a] (+/-0.04se)	0.51[b] (+/-0.04se)

5.3.7.3 Effect of open spaces on distribution of epiphyte species richness and cover

The distribution of total, bryophyte and lichen species richness in the eight different trunk plots was examined separately for the edge and interior trees (Figure 17). The general trend was for a decrease in bryophyte species richness with height and a corresponding increase in lichen species richness. Comparisons of the species richness of the eight different plots revealed significant differences among trunk plots on the edge trees for total (H=15.41, p=0.031) and lichen (H=43.16, P<0.001) species richness and on the interior trees for lichen species richness (H=29.59, p<0.001). Subsequent comparison of plots highlighted a number of significant differences in species richness between individual plots. The edge trees showed more variation in species richness among trunk plots than the interior trees. In particular, there was more variation in species richness between north and south plots at the same height. The only significant difference in species richness between north and south plots at the same height was found for lichen species richness at the middle plot height on the edge tree. Subsequent comparisons of the average overall species richness of epiphytes on the north and south sides of the trunks of the edge and interior trees revealed no significant differences between them.

The distribution of epiphyte cover in the eight different trunk plots was also examined separately for edge and interior trees (Figure 18). Again the general trend was for a decrease in bryophyte cover with height and a corresponding increase in lichen cover. Analyses revealed significant differences among trunk plots for total ($H=23.08$, $p=0.002$), bryophyte ($H=31.22$, $p<0.001$) and lichen ($H=38.57$, $p<0.001$) cover on the edge trees and lichen cover ($H=41.13$, $p<0.001$) on the interior trees. Subsequent comparisons of plots highlighted a number of significant differences in epiphyte cover between them. Again there was more variation in cover among trunk plots for the edge trees and between north and south plots at the same height. Significant differences in cover between north and south plots at the same height were found for bryophyte cover at the lower plot height on the edge trees and for lichen cover at the upper plot height on the interior trees. Subsequent comparisons of the average overall cover of epiphytes on the north and south sides of the trunks of the edge and interior trees revealed a significantly higher bryophyte cover ($p=0.046$) on the south side of the edge trees compared with the south side of the interior trees and significantly higher total ($p=0.004$) and bryophyte cover ($p=0.013$) on the south side of the edge trees compared to the north side of the same trees.

5.4 DISCUSSION

5.4.1 Epiphyte diversity

Trees adjacent to open spaces in Sitka spruce plantations can support a reasonably diverse range of epiphyte species including a few relatively rare bryophyte species. The presence of a large number of infrequent species, as seen in this study, is a common phenomenon and is related to sampling area (Humphrey 2002). However, in this study, it was the effect of the open spaces that was the main focus. Further study of Irish Sitka spruce plantations would be required to better characterise their epiphytic flora.

Since there is no known published work on epiphytes in Irish forestry plantations, it is difficult to put these results into context. The most comparable study is that of Humphrey *et al*. (2002) in British forestry plantations. Although species lists are not given for the bryophytes and lichens recorded on Sitka spruce trees up to 2m, they reported only 2 species on substrates other than deadwood in mid-rotation or mature Sitka spruce plantations. However, they did not study plantations aged between 30 and 50 years; the main age range in this study.

5.4.2 Relationship between diversity and habitat variables

This research revealed a significant negative relationship between species richness and site elevation (Figure 14). When the effects of geographic location and elevation were separated, a significant negative association for total and lichen species richness in the Cork cluster sites was still present. Higher elevations are related to higher rainfall and humidity levels. Humphrey *et al.* (2002) found that upland Sitka spruce plantations (>1500mm annual precipitation) had higher species counts of lichens on deadwood than foothills plantations (800-1500mm annual precipitation) but found no difference for bryophytes. They attributed this to differences in moisture deficit. However, in their study, higher elevation sites had higher lichen species richness, the opposite of the findings in this study. Pearson (1969), however, has reported that epiphytic lichens prefer lower and more variable humidity and Frahm (2003) found that epiphytic lichens were characteristic of habitats with longer periods of desiccation than habitats with high bryophyte abundance.

This research has also revealed significant positive associations of total and bryophyte species richness with tree density, and significant negative associations of these factors with average DBH recorded from the 10m x 10m plots surrounding each tree. (Figure 15). Since there was a significant negative relationship between average DBH and tree density in the plots, due to the inverse relationship between tree growth and competition, the relationship with average DBH is likely to be the effect of tree density rather than an effect of tree size. Hazell *et al*. (1998) found that two bryophyte species growing on aspen (*Populus tremula*) were positively affected by higher density of Norway spruce (*Picea abies*) in the surrounding plantation. Conversely, Peck and McCune (1997) found that lichen biomass on retained trees was negatively correlated with the density of the surrounding forest. Higher tree densities can result in lower light and higher humidity levels. Hale (2001), found that, in Britain, only a very heavily thinned Sitka spruce stands showed substantial increase in the amount of light that penetrated to the forest floor. However, there were small differences between stands, with a general trend of increased light transmittance with decreased density. Many bryophytes are adapted to low light intensities and have low tolerance to desiccation (Trynoski 1982). Bryophytes also tend to be more competitive in more humid habitats than lichens (Frahm 2003). This may explain the positive correlation of bryophyte species richness with tree density in this study.

5.4.3 Species composition

In the NMS ordination, the same variables as have been previously discussed- elevation, slope, tree density, and average DBH- were strongly correlated with the first and second axes. These same factors showed significant differences among the four cluster analysis groups, which were also separated by significant differences in species richness. The association of higher species richness with higher tree density is again shown by these cluster groups and hence an association with lower light and higher humidity levels. Group 1 contained low-density Wicklow (and therefore drier) sites with low species richness and diversity. Group 2 contained more dense Cork (and therefore wetter) sites with high species richness and diversity and Group 4 contained intermediate sites where lower tree density increased light levels while at the same time, the higher elevation or geographic location kept humidity levels high. Group 3 contained two trees from KNOC, an exceptional site in terms of its species richness and diversity. The high tree density may be an important factor here, but other factors may also be affecting the species richness. This site had alder (*Alnus glutinosa*) planted between the Sitka spruce sub-compartment and the open ride. It is possible there may have been some colonisation of the Sitka spruce tree by the epiphytes occurring on these broadleaved trees. These fringing broadleaves may also have had an effect on light penetration into the stand, which will be discussed below.

Few indicator species were found for the cluster analysis groups, due to the presence of a small number of frequently occurring species and a large number of infrequent species. The presence of indicator species with differing habitat requirements in Group 3 highlights the fact that a range of microclimates can be present at different heights and different aspects on a single tree.

5.4.4 Effect of open spaces

If open spaces within forestry plantations were having an effect on epiphyte diversity, you would expect to see differences between the trees at the edge of the open spaces and those in the forest interior, both in terms of species composition and diversity. Those species that occurred only on edge or interior trees were too infrequent to allow any conclusions to be drawn. Only a single species, *Dimerella lutea,* was found to be a significant indicator of interior trees and this species was also found to be an indicator for cluster group 1 and was one of the more frequently occurring species in this study (Table 9). No significant differences in diversity were found between edge and interior trees. In the NMS ordination (Figure 16) the majority of edge trees were positioned towards the top of axis two, but there was no clear separation of edge and interior trees. Any differences may have been confounded by the fact that the difference in tree density between the edge and interior trees approached significance and this factor was strongly correlated with axis 1 (Table 10).

A number of studies have compared the edge and interior of forests in relation to epiphytes (e.g. Esseen 1998; Kivisto 2000; Pearson 1969; Sillett 1995). Some of these studies have found reduced abundance (Esseen 1998) or diversity (Kivisto 2000) of epiphytes at the edge compared with the forest interior. However, these studies have looked at edges created by clearcutting rather than edges present from the planting stage and the effects were mainly due to the effects of wind damage (Esseen 1998) and sudden exposure to high levels of irradiance (Kivisto 2000). Pearson (1969) studied a more natural edge between a bog and its surrounding woodland and found that the cover of epiphytic lichens was significantly higher on trees at the edge than in the surrounding woodland. This was related to the lower humidity and higher light levels at the edge. An overall difference in epiphyte cover between edge and interior trees was not seen in this study.

The main difference between edge and interior trees in this study was the significantly higher cover of bryophytes on the south side of the edge trees compared to both the south side of the interior trees and the north side of the edge trees. You would expect the south side of the edge trees to receive the most light and have the lowest humidity levels. Since bryophytes have a low tolerance to desiccation, you would therefore expect their cover to be lower on this aspect, and not higher as was found in this study. One possible reason for this high bryophyte cover may be the presence of live branches over the entire height of the south side of the edge trees. These live branches may prevent light penetration to the inner branches and trunk of the south side of the tree. On the north sides of the edge trees and both sides of the interior trees only the upper plots were within the live crown. In this situation, paradoxically, the south side of the edge tree may receive the least amount of light and have the highest humidity. This would explain the significantly higher bryophyte cover on the south side of the edge trees. These live branches also form a side canopy which will close the edge to light and air (Matlack 1999), resulting in no significant gradients in light levels or light related factors from the open space into the forest (Matlack 1993). This side canopy may be preventing the open space from affecting the epiphyte biodiversity of the adjacent trees. It is interesting to note that KNOC, the most species rich site in this study, had the greatest height to first live branch of any trees and was the only site where the first live branch on the edge tree was higher than the first live branch on the interior tree. The fringing broadleaves at this site seemed to prevent closure of the Sitka spruce side canopy and may have increased light penetration into the stand.

5.4.5 Further Research

Topics for further research that would provide useful additional insights include studies of the epiphyte diversity adjacent to open spaces with fringing broadleaves or scrub, the epiphyte diversity at the north facing edges of open spaces, the epiphyte diversity of broadleaved trees and shrubs within open spaces, the epiphyte diversity at different thinning intensities, and the epiphyte diversity of broadleaved plantations. Basic research on the epiphytes of Irish semi-natural woodlands is also urgently required to put any studies of Irish forestry plantations into context.

5.5 CONCLUSIONS

Trees adjacent to open spaces in Sitka spruce plantations can support a reasonably diverse range of epiphyte species. The main factors influencing epiphyte biodiversity in this study were elevation and tree density. In contrast to other studies, the effect of increased elevation on epiphyte species richness was found to be negative. The positive association of tree density with bryophyte species richness highlights the adaptation of bryophytes to low light levels and their low tolerance to desiccation.

The effect of open spaces on epiphyte diversity was to increase the cover of bryophytes on the south side of the edge trees compared to the north side of the same trees and the south side of the interior trees. This was mainly related to the presence of live branches over the entire height of the south side of the edge trees which appeared to shade the trunk and increase humidity levels. These live branches also formed a dense side canopy which may have closed the edge to light and air, and prevented the open spaces from affecting the epiphyte diversity of the adjacent trees.

The results of this study suggest that stand management in relation to tree density may be more important for epiphyte diversity than open spaces within the forestry plantation. However, further study is required before any concrete recommendations can be made.

6 SPIDERS

6.1 INTRODUCTION

One of the goals of sustainable forest management is the enhancement of biodiversity within plantation forests (Coillte 2005). This can include measures to promote plantations as woodland habitats to the benefit of forest specialist species. Such strategies include the promotion of deadwood (Ferris & Humphrey 1999), longer rotation lengths (Jukes *et al.* 2001), and the enhancement of field-layer vegetation (Oxbrough *et al.* 2005). However, measures to promote biodiversity must also examine the effect of afforestation on landscapes, which can lead to the loss of habitats supporting rare or specialised species. More specifically, sustainable forest management must address how species that are typical of pre-planting habitats, and cannot survive in a forest environment, can be retained within forests. The Irish *Forest Biodiversity Guidelines* (Forest Service 2000b) state that 15% of the forest area should be incorporated into Areas for Biodiversity Enhancement (ABE), recommending that these areas should comprise approximately 5-10% retained habitats and 5-10% open space in plantations greater than 10 hectares. In order for the maximum biodiversity value to be derived from these ABEs, forest managers need to know which areas to target for open space. More specifically, what habitats should be retained to maximise biodiversity value? And, what size and shape should the open space be in order to facilitate the retention of open space species?

For plants and invertebrates the level of shade in open space within forests is a key factor affecting the species present (Greatorex-Davies *et al*. 1992; Sparks *et al*. 1996; Warren 1989). Shade levels are primarily determined by open space width, height of surrounding trees and orientation (Warren & Fuller 1993). Current guidelines vary in the minimum width recommended to promote species associated with open habitats. Often quoted as a 'rule of thumb' is the 1:1 ratio of tree height to ride width (Carter 1989; Warren & Fuller 1993). However Irish guidelines recommend that forest rides should be 6m wide, and forest road corridors should be 15m wide, in order to qualify for inclusion as an Area for Biodiversity Enhancement (Forest Service 2003). Furthermore, Warren and Fuller (1993) recommend that some forest glades should be at least 0.25 hectares in size to encourage biodiversity.

Previous research has examined the influence of orientation and width on diversity of invertebrates within rides and forest roads, with particular interest in the affects on butterflies (Greatorex-Davies *et al*. 1992; Greatorex-Davies *et al*. 1993; Sparks *et al*. 1996; Warren 1989) Other invertebrate groups examined include Coleoptera and Hemiptera (Greatorex-Davies & Sparks 1994); and mixed groups of arthropods (Carter 1989; Mullen *et al*. 2003b). Fewer studies have focused on the biodiversity value of different types of open space (such as forest roads, rides and glades) and different widths and areas of open space. There is also a need to investigate the influence of open space within forests on different invertebrate taxa, enabling forest management plans to try to reach a consensus on the best way to manage open habitats for a range of invertebrate groups. Spiders are useful as indicators of habitat change as they are primarily affected by changes in habitat structure (Uetz 1991). Spiders also occupy an important position in terrestrial food webs as both predators and prey and hence have the potential to be used as surrogate indicators of invertebrate diversity (Marc *et al*. 1999).

We aim to assess the influence of open space in plantation forests on spider assemblages by addressing the following questions:

1. How does open space enhance spider diversity within plantation forests?
2. How does the type of open space and its size influence spider diversity within plantation forests?

Forest management plans for enhancing biodiversity, specifically for invertebrates will be considered when interpreting the findings from this study.

6.2 METHODOLOGY

6.2.1 Study areas

Twelve commercially mature Sitka spruce (*Picea sitchensis*) stands of at least 80 ha were sampled within Ireland. The stands were located in two geographical clusters of six sites that were matched for environmental variables such as altitude, soil and geology (Figure 1). The Wicklow cluster (in Co.s Wicklow and Dublin), were typically well-drained upland sites with peaty-podzol soils and the unplanted open space was predominately humid acid-grassland/dry heath. The Cork cluster (in Co.s Cork, Limerick and Kerry), were typically poorly drained sites on peaty soils with modified blanket bog as the predominant habitat type in the unplanted open space.

6.2.2 Open space configurations

We categorised the open space into three types: forest road edges; rides; and glades (areas of non-linear open space). We used digitised aerial photographs to identify the open space within each site and to select suitable areas for sampling. Five open spaces were sampled per site, with at least one from each open space type where possible (Table 15). However, three sites did not contain any glades and two sites had only one large glade (> 6 ha), so in this case, two sampling plots were established within the open space, with plots always separated by a minimum of 100m (Table 15). We sampled a total of 60 plots of open space comprising 20 glades, 22 rides and 18 roads. The plots were all located on the south facing side (or southwest/west where south facing was not possible) of the open space in a homogenous area of vegetation which was typical of the open space being sampled.

Table 15. Configuration of open space sampled within each site

Site	Number of plots		
	Glade	Ride	Road
Wicklow cluster			
Athdown	3*	1	1
Ballinastoe	1	3	1
Ballysmuttan	3	1	1
Ballycura	3	1	1
Lugg	3	1	1
Mucklagh	0	3	2
Cork cluster			
Carrigagula	2	1	2
Cleanglass	2*	1	2
Glanharee	2	1	2
Knocnagoum	0	3	2
Meetinny	2	2	1
Reanahoun	0	3	2
Total	21	21	18

* Two plots established in the same glade.

6.2.3 Spider sampling

We used pitfall traps to sample the ground-dwelling spider fauna. Pitfalls consisted of a plastic cup, 7cm in diameter by 9cm depth. Each trap had several drainage slits pierced approximately 2cm from the top of the cup and was filled with antifreeze (ethylene glycol) to a depth of 1cm to act as a killing and preserving agent. The traps were placed in holes dug with a bulb corer so that the rim was flush with the ground surface.

Sampling plots consisted of pitfall traps arranged in a transect from the open space into the forest. Each sample point on the transect consisted of three pitfall traps, each set two metres apart, which were arranged perpendicular to the forest edge. Two of these traps were used in the analysis with the third to be used only if traps were lost due to flooding or animal damage. Five sampling points were established on the transect in the following locations: Open (centre of the open space); Open-boundary (2m into the open space from tree trunks); Boundary (tree trunk); Forest-boundary (2m into the forest from the tree trunk); Forest (5m into the forest interior). The traps were set in May 2004 and were active for nine consecutive weeks, being emptied approximately every three weeks.

6.2.4 Environmental variables

The percentage cover of vegetation was recorded in a $1m^2$ quadrat surrounding two of the pitfall traps in each sample point on the transect in the following structural layers: ground vegetation (0-10cm); lower field layer (>10cm - 50cm) and upper field layer (>50cm – 200cm). Cover of other features such as deadwood and litter were also recorded using this scale and litter depth was measured within each quadrat. All cover values were estimated using the Braun-Blanquet scale (Mueller-Dombois & Ellenberg 1974), which involves giving numerical rankings to a range of percentages (+ = <1% cover; 1 = 1 - 5%; 2 = 6 - 25%; 3 = 26 - 50%; 4 = 51 - 75%; 5 = 76 - 100%).

Soil samples were taken from the Open, Boundary and Forest sampling points on the transect using a bulb corer which extracted the top layer of substrate to a depth of 15cm. Organic content of the soil was calculated using the method outlined in Grimshaw (1989). Hemispherical photographs were used to measure canopy openness in the centre of each open space with the percentage of open space calculated from the scanned images using Gap Light Analyser 2.0 software (Frazer *et al*. 1999a).

Within each ride and road open space plot the distance between tree trunks was measured. Digitised aerial photographs were used to estimate glade area and also estimate the area of open space within 100m, 200m and 300m of each plot in the following categories: unplanted, rides (>10m wide), clearfell, young forestry (pre-canopy closure), broadleaved, undeveloped (areas of crop failure), windthrow, outside (open space outside the plantation), forest road.

6.2.5 Species identification

Spiders were sorted from the pitfall trap debris and stored in 70% alcohol. The species were identified using a x50 magnification microscope and nomenclature follows Roberts (1993). Difficult species verified by Robert Johnston and Dr Peter Merrett and voucher specimens have been retained. Only adult specimens were identified due to the difficulty in assigning juveniles to species. The species were categorised according to the literature into the following habitat associations: open-associated species; forest-associated species; or generalists (Cawley 2001; Harvey *et al*. 2002; McFerran 1997; Nolan 2002; Roberts 1993; van Helsdingen 1996a; van Helsdingen 1996b).

6.2.6 Data Analysis

6.2.6.1 Trends along the **Open** *to* **Forest** *transect*

We used one-way ANOVA and Tukey post-hoc tests to assess trends in species richness and abundance (plot as the replicate) along the Open to Forest transect. We used global non-metric multi-dimensional scaling analysis (NMS) to examine differences in assemblage structure across the Open to Forest transect. This ordination method has been successfully used in several studies of invertebrates within forests (Huhta 2002; Oxbrough *et al*. 2005; Siira-Pietikainen *et al*. 2003; Siira-Pietikainen *et al*. 2001). The NMS used mean relative abundance of each species per site, for each location on the Open to Forest transect. We used relative abundance rather than absolute abundance data as variation in vegetation structure (as may be present across the Open to Forest transect) can affect the efficiency of pitfall traps (Melbourne 1999). The NMS ordination diagram was presented as a joint biplot which uses correlation analyses to relate habitat variables (measured at each transect location) with the NMS ordination axes. We used the Bray-Curtis distance measure and the following parameter set-up for the NMS: 6 axes; 20 runs with real data; stability criterion = 0.001; 10 iterations to evaluate stability; 250 maximum iterations; step down in dimensionality used; initial step length = 0.20; Random starting coordinates; 50 runs of the Monte Carlo test. We also carried out Indicator Species Analysis on the spider assemblages at the different sampling points on the transect to identify species associated with the different habitat types.

6.2.6.2 The influence of open space type and size

Preliminary analysis indicated that only traps in the centre of the open space supported an open spider fauna so only data from these traps were used in following analyses. We used one-way ANOVA with Tukey post-hoc tests to examine differences in species richness and abundance among the open space types (with plot as the replicate). Trends in spider assemblage structure among the open space types were assessed using NMS (parameter set-up as above). We used Pearson's correlation analyses to examine the relationship between species richness and abundance and the following open space dimensions: glade area; ride/road width (trunk to trunk); ride/road verge width (trunk to road edge). Ride and road were combined as previous analyses indicated that their species richness, abundance and assemblage structure were similar. We used flexible-beta cluster analysis (with $\beta = -0.25$) to explore the relationship between spider assemblage structure and open space size in more detail. One-way ANOVA was used to examine differences in species variables, open space size and habitat variables among the assemblage groups identified by cluster analysis. Some open space plots had a high shrub or broadleaf cover (of unplanted origin) whereas others were more typical of a grassland habitat with high lower-field layer cover. Preliminary analysis indicated these two types of open space were different in spider assemblage structure and so the open space plots within each cluster group were split into these broad habitat groupings (shrub/deciduous cover or lower-field layer cover) and the assemblages were analysed using NMS (parameters as above) and Indicator Species Analysis.

6.2.6.3 Large scale influence of open space

We used Pearson's correlation analyses to examine the relationship between species variables and the total amount of open space within 200m of each plot using the open space categories described above. We used 200m (rather than 100m or 300m) as preliminary analyses indicated that the spider assemblages responded to the open space amounts at this scale. The open space categories were also combined into total unforested open space (road, ride, outside and unplanted) and total open space (all categories). We also used one-way

ANOVA to examine the effect of open space amount in the following groups: <5%; 5-10% and >10% on species richness and abundance.

Where the assumptions of normality and homogeneity of variance were not met, data were square root transformed, however if the data still did not conform to the assumptions of ANOVA we used the Kruskal-Wallis (H) test, with a Tukey-type post-hoc comparison (Zar 1996). ANOVA and correlation analyses were carried out in SPSS (SPSS 2002). Multivariate analyses (NMS and cluster analysis) as well as Indicator Species Analysis were carried out using PC-ORD (McCune & Mefford 1997b).

6.3 RESULTS

Two of the plots (a glade in Ballysmuttan and a ride in Mucklagh) had a substantial number of traps (33%) disturbed and so were excluded from the analyses. This gave a total of 58 plots used in the analyses: 20 rides, 20 glades and 18 roads. There were a total of 11,872 individual spiders captured in 13 families and 122 species. Of these 2690 were juveniles and so were excluded from the analyses. Twenty-four species were classified as being associated with open habitats and 14 with forested habitats. A full list of species and their habitat association is given in 0. The species were predominately from the Linyphiidae family (87 species), however there were also 9 species of Lycosidae and 6 species of Therididae. *Monocephalus fuscipes*, *Lepthyphantes zimmermanni* and *Diplocephalus latifrons* were the most abundant species accounting for 32% of the total adult catch and occurring in 97% of the traps.

6.3.1 Trends along the Open to Forest transect

Across the Open to Forest transect mean species richness and abundance decreased (Table 16), the centre of the open space being significantly richer than the other points on the transect ($F_{4,59}$ = 33.0, p = <0.001 and $F_{4,59}$ = 9.80, p = <0.001 respectively). Species richness was also significantly greater at the Open-boundary point on the transect than the Forest-boundary and Forest sampling points. Richness and abundance of open-associated species was also significantly greater in the open space compared to the other points along the transect ($F_{4,59}$ = 51.1, p = <0.001 and $H_{4,59}$ = 36.3, p = <0.001 respectively), whereas richness and abundance of forest-associated species was significantly lower in the open space compared to the other points along the transect ($F_{4,59}$ = 8.33, p = <0.001 and $F_{4,59}$ = 39.1 , p = <0.001 respectively). Furthermore, at either end of the transect there were 52 species in the centre of the open space which did not occur 5m into the forest, whereas only 6 species occurred within the forest but not in the centre of the open space. There were 50 species shared among these two groups of traps.

Table 16. Mean species richness and abundance (±SD) per plot across the Open to Forest transect: Open (centre of the open space); Open-boundary (2m into the open space); Boundary (tree base); Forest-boundary (2m into the forest); Forest (5m into the forest).

	Open	Open-boundary	Boundary	Forest-boundary	Forest
Species richness	15.2 (±1.9)	12.1 (±1.5)	10.4 (±1.1)	9.5 (±1.3)	8.8 (±1.7)
Open-associated species richness	3.0 (±0.8)	0.8 (±0.5)	0.3 (±0.3)	0.3 (±0.3)	0.2 (±0.2)
Forest-associated species richness	2.6 (±1.0)	4.1 (±0.8)	4.1 (±0.8)	4.2 (±0.8)	4.2 (±0.9)
Abundance	46.7 (±12.8)	30.5 (±9.0)	28.8 (±8.7)	25.9 (±9.0)	25.3 (±8.3)
Open-associated species' relative abundance	0.31 (±0.12)	0.05 (±0.04)	0.02 (±0.02)	0.01 (±0.01)	0.01 (±0.01)
Forest-associated species' relative abundance	0.20 (±0.10)	0.50 (±0.12)	0.61 (±0.09)	0.64 (±0.09)	0.69 (±0.09)

The NMS ordination of spider assemblages across the Open to Forest transect explained 85% of the variation in the species data, with Axes 1 and 2 accounting for 50% and 35% respectively (Figure 19). Across Axis 1 the assemblages of spiders in the open space are separated from those positioned closer to the boundary along the Open to Forest transect, whereas Axis 2 represents a separation of the Cork and Wicklow sites (with the exception of Mucklagh in the Wicklow region which is more similar to the Cork cluster of sites). The spider assemblages found at the Open-boundary sampling point on the transect represent a transition of assemblages in the centre of the open space to those within the forest. The spider assemblages at the Boundary and those within the Forest form a tight cluster of points whereas those in the open space and at the Open-boundary exhibit much greater variation across both axes. Cover of lower-field layer vegetation is associated with the spider assemblages in the Open, whereas needle litter and twig cover are associated with spider assemblages within the Forest. Cover of ground vegetation was associated with the assemblages at the Open-boundary (2m into the open space).

There were 17 species with high indicator values in the centre of the open space, however no indicators were identified at any of the other locations on the transect (Table 17). Many of these were species associated with open habitats such as *Pardosa pullata*, *Oedothorax gibbosus, Pocadicnemis pumila*, *Alopecosa pulverulent*a, *Silometopus elegans* and *Pardosa uliginosus*. Several species also had relatively high indicator values for the Open-boundary, compared to those within the Forest, including *P. pumila, Bathyphantes gracilis, Oxyptila trux* and *Maso sundevalli*.

Table 17. Indicator Species Analysis of spider assemblages across the Open to Forest transect. The maximum indicator value and associated significance (Monte Carlo test) are in bold type for each species. Parentheses indicate species associated with open habitats (O)

Species	Percentage indicator value				
	Open	Open-boundary	Boundary	Forest-boundary	Forest
Pardosa pullata (O)	**95*****	1	0	0	0
Dicymbium tibiale	**74*****	6	1	1	1
Oedothorax gibbosus (O)	**71*****	3	3	0	0
Pocadicnemis pumila (O)	**67*****	15	3	2	0
Pardosa nigriceps	**58*****	3	1	0	0
Alopecosa pulverulenta (O)	**55*****	1	0	0	0
Bathyphantes gracilis	**52*****	19	2	1	6
Walckenaeria vigilax	**51*****	18	2	2	0
Dismodicus bifrons	**49*****	8	2	2	0
Oxyptila trux	**42****	15	0	2	0
Bathyphantes nigrinus	**41****	4	0	1	0
Pachygnatha degeeri	**41****	0	0	0	0
Silometopus elegans (O)	**40****	2	0	0	1
Pirata piraticus	**37****	1	0	0	1
Pirata uliginosus (O)	**36****	2	2	1	0
Maso sundevalli	**34****	14	0	0	0

* = < 0.05; ** = < 0.01; *** = < 0.001

6.3.2 The influence of open space type and size

Mean species richness among the open space types is shown in Table 18. Total species richness and richness of open-associated species was significantly greater within the glades than in the rides or roads ($F_{2,57} = 4.50$, $p = <0.05$ and $F_{2,57} = 5.27$, $p = <0.01$ respectively). However forest-associated species richness did not differ significantly among the open space types. Mean spider abundance exhibited a similar trend among the open space types (Table 18), where total abundance and abundance of open-associated species was greater in the glades compared to either the roads or rides ($F_{2,57} = 6.57$, $p = <0.01$ and $F_{2,57} = 4.37$, $p = <0.05$

respectively). However the abundance of species associated with forests was significantly lower in the glades compared to the roads ($F_{2,57} = 5.98$, $p = <0.01$).

Table 18. Mean species richness and abundance per plot among the open space types. Parentheses indicate standard error.

	Glade (n = 20)	Ride (n = 20)	Road (n = 18)
Total species richness	17.6 (± 1.10)	13.9 (± 0.86)	14.2 (± 0.97)
Open-associated species richness	9.9 (± 0.88)	7.0 (± 0.76)	6.4 (± 0.76)
Forest-associated species richness	1.7 (± 0.27)	2.0 (± 0.32)	2.4 (± 0.24)
Total abundance	65.4 (±7.99)	36.7 (±5.77)	37.6 (± 4.78)
Open-associated species' relative abundance	0.73 (± 0.05)	0.57 (± 0.05)	0.50 (± 0.06)
Forest-associated species' relative abundance	0.06 (± 0.03)	0.15 (± 0.04)	0.20 (± 0.05)

The NMS ordination of spider assemblages among the open space types accounted for 65% of the variation in the species data with Axis 1 explaining 40% and Axis 2 25% of the variation respectively (Figure 20). The plots in the glades generally separate along Axis 1 from the rides and roads. The roads and rides have similar assemblage structure to each other. Axis 2 represents a separation of the Cork and Wicklow open space plots.

The relationship between spider species variables and open space size is shown in Table 19. Three ride plots were identified as outliers and were removed from the correlation analyses involving open-associated species richness and abundance (Figure 21, Table 19). These wide rides were atypical of the rides sampled and contained features which may have affected the number of open-associated species present: two rides in Knocknagoum were bordered by several rows of planted birch trees and were originally to be planned forest roads, whereas in a ride in Ballysmuttan a large rowan tree was present where the traps where located.

There was no relationship between total species richness and ride/road verge width (Table 19). However open-associated species richness was significantly positively correlated with ride/road verge width whereas forest-associated species richness was negatively correlated, though not significantly so. Total abundance and abundance of open-associated species were both significantly positively correlated with ride/road verge width whereas the abundance of forest-associated species was significantly negatively correlated. The relationships between ride/road width and species variables were weaker (and mostly non-significant) than for ride/road verge width (with the exception of total species richness), however they did exhibit the same directional trend.

Glade area was positively related to total species richness and open-associated species richness, however these relationships were not significant. There was no relationship between glade area and total abundance although the habitat specialists did exhibit relationships: abundance of open-associated species was positively related to glade area whereas abundance of forest-associated species was negatively related, however these relationship were not significant.

Using cluster analysis the spider assemblages were separated into four groups (Table 20). Cluster Group 1 contains most of the glades, and the plots are predominately from the Wicklow region, whereas cluster Group 2 consists mostly of road and rides plots that are all from the Cork region. Cluster Groups 3 and 4 consist mainly of road and ride plots; however in Group 3 these are predominately from the Cork region whereas in Group 4 the majority of plots are from the Wicklow region. The assemblages of cluster Groups 1 and 2 were initially split from Groups 3 and 4 in the analysis.

Table 19. Correlations between spider assemblage variables and the following open space dimensions: ride/road-verge width (m); ride/road width (m); Glade area (m^2).

Variable	Pearson r		
	Ride/Road-verge (m) n = 36	Ride/Road (m) n = 38	Glade (m^2) n= 20
Total species richness	0.09	0.18	0.31
Open-associated species richness	0.58[a]***	0.26	0.38
Forest-associated species richness	-0.15	-0.08	0.00
Total abundance	0.47**	0.38*	-0.04
Open-associated species' relative abundance	0.61[b]***	0.30#	0.15
Forest-associated species' relative abundance	-0.52**	-0.30#	-0.18

= 0.05 - 0.10; * = <0.05; **<0.01; ***<0.001

[a] 3 outliers removed

Table 20. The distribution of open space plots within the cluster groups

No. of plots	Cluster 1	Cluster 2	Cluster 3	Cluster 4
Total	17	10	20	11
Cork	3	10	14	3
Wicklow	14	0	6	8
Glade	11	4	4	1
Ride	4	1	10	5
Roads	2	5	6	5

The mean width of rides and roads as well as glade area is greater in cluster Groups 1 and 2 compared to cluster Groups 3 and 4 however these differences are not significant (Table 21). Cluster Groups 1 and 2 were characterised by significantly greater canopy openness than Groups 3 and 4, however other layers of vegetation do not differ significantly across the cluster Groups. Total species richness and open-associated species richness was significantly greater in cluster Group 1 than in the other Groups (Table 21), whereas richness of forest-associated species was significantly greater in cluster Group 4 than in Group 1. Total abundance and that of the open-associated species followed a similar trend, however cluster Group 2 was also significantly greater than Groups 3 and 4. Abundance of forest-associated species was significantly greater in Groups 3 and 4.

The open space plots within each cluster group were classified by the predominant type of vegetation cover (Table 22). Cluster Groups 1 and 2 do not contain any plots that have a shrub/deciduous cover whereas at least half of the total number of plots in Groups 3 and 4 do. The road/ride widths of cluster Groups 1 and 2 range from 15–34m (Table 22), all of which have lower-field layer cover. In Groups 3 and 4, however, the plots with lower-field layer cover have a much smaller range of widths (7-14m); furthermore, this range does not overlap with those in Groups 1 and 2. This would suggest that the roads and rides with lower-field layer cover that are less than 15m wide support a different assemblage of species than those in Groups 1 and 2 (which are wider than 15m). Furthermore these plots (<15m wide) with lower-field layer cover are more similar to those plots under shrub/deciduous cover.

All of the glades present in cluster Groups 3 and 4 were under shrub/deciduous cover, with the exception of the very small glade (80 m^2) in cluster Group 3.

The spider assemblages within the open space plots which were characterised by shrub/deciduous cover did not form a distinct group from those with a lower-field layer cover (Figure 22) or from those assemblages sampled on the forest interior transect position. Moreover, the assemblages within the shrub/deciduous open space plots represented a transition between the forest interior and the lower-field layer cover habitats. Several species with significantly high indicator values in the shrub/deciduous cover open space were identified (Table 23), including *Lepthyphantes alacris*, a forest-associated species and *O. gibbosus*, a species associated with open habitats, as well as several generalist species. The three species with high significant indicator values within the forest traps were all associated with forest habitats.

Table 21. Mean environmental and species variables among cluster groups (± standard error).

	1	2	3	4	ANOVA df = $_{3,57}$	Post-Hoc
Mean area (m^2) of glade	12991 (±4994)	8406 (±1967)	3818 (±1798)	3083[a]	n.s	
Mean width of ride (m)	16.4 (±1.0)	28.8	13.3 (±1.8)	9.0 (±1.4)	n.s	
Mean width of road (m)	23.6 (±3.0)	25.2 (±3.3)	20.5 (±1.9)	16.6 (±3.3)	n.s	
Mean width of road verge (m)	9.3 (±2.2)	11.8 (±1.4)	9.1 (±1.4)	9.9 (±1.5)	n.s	
Canopy openness (%)[a]	61 (±0.04)	55 (±0.04)	37 (±0.04)	18 (±0.04)	F = 17.9***	1 & 2 > 3 & 4 3 > 4
Ground vegetation	0.50 (±0.07)	0.20 (±0.08)	0.34 (±0.07)	0.51 (±0.1)	n.s	
Lower field layer vegetation	0.51 (±0.08)	0.74 (±0.05)	0.59 (±0.07)	0.38 (±0.1)	n.s	
Upper field layer vegetation	0.01 (±0.01)	0.05 (±0.03)	0.15 (±0.05)	0.13 (±0.06)	n.s	
Total species richness	19.1 (±0.86)	14.4 (±1.27)	13.6 (±0.91)	13.3 (±1.27)	F = 7.59***	1 > 2 & 3 & 4
Open-associated species richness	12.0 (±0.60)	7.8 (±0.73)	5.9 (0.64)	4.9 (±0.73)	F = 23.10***	1 > 2 & 3 & 4
Forest-associated species richness	1.5 (±0.19)	1.6 (±0.27)	2.1 (±0.27)	3.1 (±0.49)	H = 9.33*	1 < 4
Total abundance	80.0 (±7.8)	46.0 (±6.1)	29.5 (±3.3)	28.2 (±4.1)	F = 22.81***	1 > 2 & 3 & 4 2 > 3 & 4
Open-associated species relative abundance	0.84 (±0.03)	0.72 (0.05)	0.45 (±0.05)	0.40 ±(±0.07)	F = 22.43***	1 > 2 & 3 & 4 2 > 3 & 4
Forest-associated species relative abundance	0.04 (±0.01)	0.05 (0.01)	0.17 ± (0.03)	0.29 (±0.06)	H = 22.87***	1 < 3 & 4 2 < 4

* = < 0.05; ** = < 0.01; *** = < 0.001

[a] 2 data points missing hence ANOVA df = $_{3,55}$

Table 22. The number of plots within each cluster group according to shrub/deciduous cover and the range of open space size among ride/roads and glades

Cluster group		Number of plots: Shrub/deciduous cover	Number of plots: Lower-field layer cover	Range of open space size: Shrub/deciduous cover	Range of open space size: Lower-field layer cover
1	Ride/Road	0	6	-	15 – 27m
2	Ride/Road	0	6	-	16 – 34m
3	Ride/Road	10	6	10 – 27m	7 – 14m
4	Ride/Road	5	5	7 – 26m	9 – 14m
1	Glade	0	11	-	1105 – 45211m^2
2	Glade	0	4	-	4166 – 11753 m^2
3	Glade	3	1	1396 - 6898 m^2	80 m^2
4	Glade	1	0	3083 m^2	-

Table 23. Indicator Species Analysis of spider assemblages in the open space with shrub/deciduous habitat type and adjacent forest traps. The maximum indicator value (percent) and associated significance (Monte Carlo test) are in bold type for each species. Species habitat associations are given in the parentheses: open-associated (O); forest-associated (F).

	Percentage indicator value: Shrub/deciduous cover	Percentage indicator value: Forest traps
Agyneta ramosa	**49***	3
Bathyphantes gracilis	**43***	4
Bathyphantes nigrinus	**57****	0
Dicymbium tibiale	**43***	0
Diplocephalus latifrons (F)	12	**62***
Lepthyphantes alacris (F)	**63****	9
Lepthyphantes zimmermanni (F)	32	**65***
Monocephalus fuscipes (F)	10	**67****
Oedothorax gibbosus (O)	**43****	0

* = < 0.05; ** = < 0.01; *** = < 0.001

6.3.3 Large scale influence of open space

Correlations between the amount of open space (within 200m of the plots) and species variables revealed several significant relationships (Table 24). The total number of species, and individuals, as well as the number of open-associated species, were significantly positively correlated with the area of unplanted open space, whereas these variables were significantly negatively correlated with ride area. Forest-associated species abundance however, showed the opposite trend. There were no significant relationships between the species variables and the any of the following open space types: road, outside, undeveloped, windthrow, clearfell, broadleaf, total unforested and total open space.

Mean species richness increased with increasing amounts of unplanted open space within 200m of each plot: <5% open space (13.9±0.8 SE); 5-10% open space (15.5±0.9 SE); >10% open space (17.5±1.3 SE). Furthermore plots which had >10% unplanted open space were significantly greater in mean species richness than those with <5% ($F = 3.09$ 2,57, $p = 0.05$). A similar trend was exhibited between mean richness of open-associated species and unplanted open space: <5% (6.5±0.7 SE); 5-10% (8.6±0.9 SE); >10% (9.3±1.0 SE) where plots with >10% unplanted open space have significantly greater richness than those with <5% ($F = 3.39$ 2,57, $p = 0.04$). There was no significant difference between forest-associated species or species abundance and unplanted open space amounts; or between the other open space categories and the species variables.

Table 24. Correlations (Pearson r) between the area of open space within 200m of the sample plots and species variables (n = 58)

Species variable	Open space type	
	Unplanted (m^2)	Ride (m^2)
Total species richness	0.36**	-0.28*
Open-associated species richness	0.35**	-0.31*
Forest-associated species richness	-0.04	-0.01
Total abundance	0.34**	-0.35**
Open-associated species relative abundance	0.20	-0.31*
Forest-associated species relative abundance	-0.25*	0.30*

*= < 0.05; ** = < 0.01; *** = < 0.001

6.4 DISCUSSION

6.4.1 Trends along the Open to Forest transect

This study suggests that open space within forest plantations can support a wide array of spider species that are not present within the forest. The open space supports a greater number of generalist species as well as providing a suitable refuge for species associated with open habitats. This is consistent with studies of plants (Mullen *et al*. 2003b; Peterken & Francis 1999; Sparks *et al*. 1996) and other groups of invertebrates (Carter 1989; and see Section 7). Furthermore, the present study found that the areas under canopy in plantation forests supported fewer species than the open space. Previous studies have also found that, in terms of invertebrates, mature plantation forests are relatively species poor compared to more open habitats (Butterfield *et al*. 1995; Day & Carthy 1988; Oxbrough *et al*. 2005).

The spider assemblages at the Open-boundary sampling point on the transect represent a transition between the open and forested habitats. This is consistent with other studies which have found an 'edge effect' at the Open to Forest ecotone with the boundary zone being able to support species from both habitat types (Downie *et al*. 1996; Terrel-Nield 1986). In the present study, the traps at the Open-boundary were under variable amounts of canopy cover depending on the length of branches above a particular trap (personal observation). This created varied vegetation cover at a small scale, where some of the lower field layer vegetation is shaded out to the benefit of ground vegetation, predominantly more shade tolerant mosses. Spider diversity is positively influenced by vegetation structure (Uetz 1991). The vegetation facilitates greater prey abundance and diversity, web attachment points, protection from predators, stable microclimates, and hiding places for active hunters. In the present study the open-boundary 'transition zone' supported more species than those in the forest, though not more than those in the open, suggesting that some open-associated species can take advantage of the conditions in the Open-boundary area. Downie *et al.* (1996) also found species with a particular preference for the Boundary zone, however there did not appear to be any species which were particularly specialised to the Open-boundary within this study.

Spider species richness and abundance declined dramatically once the traps were under the influence of the canopy. The spider assemblages at the Boundary (tree base) were indistinguishable in assemblage structure from those two metres and five metres into the forest but different from those at the Open-boundary (only 2m away). Vegetation structure declined across the Open to Forest transect with lower-field layer cover associated with the open space and ground vegetation cover associated with the Open-boundary. It is well known that vascular plant cover is lower under the canopy (Ferris *et al*. 2000b; Oxbrough *et al*. 2005). This agrees with the findings of Bedford and Usher (1994) and Downie *et al*. (1996)

which suggest that even at a distance of a few metres the movement of open species into the forest is limited.

6.4.2 Influence of open space type and size

The present study found that glades support more species and individuals, as well as a distinct fauna from the rides and road edges. The non-linear shape of glades means that they have a larger area away from the influence of the forest canopy, probably allowing them to support a greater number of species associated with open habitats. The relationship between area and species richness is well-studied, with larger areas having a greater potential for habitat heterogeneity, less chance of random extinctions and greater likelihood of random immigration affecting the spider population (MacArthur & MacArthur 1961; Pianka 1966). This suggests that the glades have a greater potential than roads or rides to retain open species associated with the pre-planting habitat. The spider assemblages were also distinct among the geographical clusters in the open space, (though not in the traps in the forest interior). The Cork and Wicklow site clusters were characterised by different habitats: in Cork the sites were predominately modified blanket bog in poorly drained areas, whereas the Wicklow sites were predominately humid-acid grassland/dry heath in better drained areas. This suggests that habitat type is an important factor in determining the spider species present in an open space, indicating that the species present are not just generalist species, but may be retained from the pre-planting habitat.

The relationship between open space size and spider diversity was confounded by the influence of the plantation canopy and the habitat type of the open space. Several of the open space plots were characterised by a heavy shrub layer or deciduous woodland cover, with more forest-associated species and fewer open species. These plots were similar in assemblage structure to the rides and roads less than 15m wide. This suggests that open spaces less than 15m wide are not able to support a fauna of spiders associated with typical open habitats. Rides and roads <15m wide are more shaded, which probably leads to vegetation structure and microclimatic conditions similar to those of a mature open forest.

It has been recommended that ride width should be between 1-1.5 times tree height to provide adequate light conditions for open-associated species (Carter 1989; Greatorex-Davies 1989; Warren & Fuller 1993). In the present study, mean height of mature spruce was 15.3m (±4.3SD), giving a ride width of 15-23m to support open species. Therefore our results would appear to support the recommended ratio of tree height to width if it is taken as the minimum needed to support an open spider fauna. This also suggests that the inclusion of rides with a width of 6m as Areas for Biodiversity Enhancement in Irish plantations may be too low for spiders (Forest Service 2003), whereas the 15m width recommendation for roads should be taken as a lower limit. Furthermore, as species richness showed no indication of reaching a plateau with increasing road verge width, this would suggest that widening roads above the 15m standard width will further enhance biodiversity.

A number of the glades were characterised by areas of deciduous woodland and shrubs; these were more similar in spider assemblage stucture to the narrower roads/rides than the glades which have a high lower-field layer cover. The indicator species identified in the shrub/deciduous open space plots were mostly generalist species, although an open-associated species (*O. gibbosus*) and a forest-associated species (*L. alacris*) were identified as indicators, both of these species however are relatively common. In Ireland *L. alacris* has frequently been found within more open Sitka spruce plantations (mature stands which have been thinned or thicket stage) (Smith *et al*. 2004). This might suggest that these deciduous/shrub areas represent intermediate habitat between the plantation forest and the lower field layer-type open space, though the lack of specialist species (for instance, forest specialists which are not supported within the plantation), indicates that their potential for adding to plantation biodiversity may be negligible.

The glades sampled in this study did not exhibit a similar 'threshold' size as was found for the roads and rides. However, one glade was similar in assemblage structure to the glades that were characterised by deciduous woodland and shrubs, although it was characterised by lower-field layer vegetation. This glade was only 80m² in area, whereas the next smallest glade in area was 1000m². This might suggest that this very small glade was under the influence of the forest plantation canopy and so was not large enough to support an open spider fauna. To identify a threshold area (above which open species can be supported), areas between 80m² and 1000m² need to be studied; the 15m threshold for ride/road width might suggest 225 m² as a minimum area for glades.

6.4.3 Large-scale influence of open space

The overall amount of unplanted open space within a plantation was positively related to both species richness and abundance. Similarly, Peterken and Francis (1999) found that the number of open space species supported by woodlands was far greater in large woods, which they attributed to the presence of more open space across the whole wooded area. In Section 7 no relationship was found between hoverfly richness and amount of open space at a large scale; this may be a reflection of the different response of this different invertebrate group to open space habitats.

Whilst there was a relationship with unplanted open space at a large scale, there was no relationship with non-linear unplanted open space at a smaller scale (within each open space). This may suggest that more open space at a larger scale encourages the movement of individuals among open space. Spiders utilise both aerial (Duffey 1956) and ground dispersal (Thomas *et al*. 1990) as a means to colonise habitats, so the amount of open space surrounding the sampling plots will directly affect the ability of open-associated species to disperse within the forested landscape. There was a negative relationship between ride area and spider assemblages. However it is likely that ride area indirectly represents the amount of forested area within 200m of the sampling points i.e. the greater the amount of planted forest, the greater potential for more rides.

The Irish *Forest Biodiversity Guidelines* (Forest Service 2000b) recommend that 5-10% of forest plantations larger than 10ha should be kept as retained open space. However as spider species richness increased with the amount of unplanted open space in the three categories of <5%, 5-10% and >10%, this suggests that the number of species which can be supported in areas with 5-10% has not reached a maximum and hence the 5-10% area may not be adequate to support a full suite of species associated with open habitats.

6.5 CONCLUSIONS AND FURTHER RESEARCH REQUIRED

Open space within plantation forests supports spider species associated with open habitats and enhances overall plantation biodiversity. However, this study focused on ground dwelling spider assemblages and to sample the entire spider community present within plantations, future research would need to sample all vegetation layers present within the open space and forest interior (e.g. from lower vegetation to shrubs, and to tree canopies). More comprehensive surveys of invertebrate fauna would also benefit from the inclusion of other invertebrates groups which may have different ecological requirements from spiders (which are generalist predators and relatively mobile), such as phytophagous or saproxylic invertebrates.

This study suggests that an absolute minimum width of 15m is needed for forest roads and rides to support an open spider fauna. For non-linear open space, a stratified sampling approach that varies glade area may reveal a similar 'threshold' size, above which open species are supported. The present study also highlights the need to examine the biodiversity value of a range of habitat types that could potentially be selected as retained habitat, specifically with regard to whether the unique and rare species associated with pre-planting habitat persist.

7 HOVERFLIES

7.1 INTRODUCTION

Sustainable forest management is now a key concept underpinning forest policy, and includes the aim of maintaining forest biodiversity. An important component of forest biodiversity is the biota associated with open space habitats within forests (Ferris & Carter 2000a; Ferris-Kaan 1991; Warren & Fuller 1993) and Peterken (1996) considered that "the treatment of the open spaces is the single most important factor in the success or failure of nature conservation within plantations". Open spaces can contribute to maintenance of forest biodiversity in two ways. Firstly, in the highly fragmented forests of northwestern Europe, many of the characteristic forest species are, strictly speaking, species of forest edges and glades, rather than forest interior species (Kirby 1992). Secondly, in intensively farmed landscapes, open spaces within forests may provide suitable habitat for species characteristic of semi-natural open space habitats, which no longer occur within the surrounding landscape.

Research on open space habitats within forests has focused on the management of forest roads and rides, particularly in relation to the effects of shade and the requirement for vegetation control (Ferris-Kaan 1991; Greatorex-Davies *et al*. 1992; Greatorex-Davies & Sparks 1994; Greatorex-Davies *et al*. 1993; Sparks & Greatorex-Davies 1992). This research has resulted in the development of practical guidelines for forest managers (Ferris & Carter 2000a; Warren & Fuller 1993). However, this research has been focused on intensively managed lowland woods in southern England, and on a limited range of taxonomic groups (mainly vascular plants, butterflies and birds, but see Greatorex-Davies & Sparks 1994). Some of the management prescriptions, such as regular vegetation management, may not be practicable in other types of woodland. More generally, there is a need for information about the effects of open space design and management on a wider range of taxonomic groups, and about the contribution of open spaces (including unplanted glades, as well as forest roads and rides) to overall forest biodiversity.

The effect of afforestation on biodiversity, and the potential to plan and manage forests to enhance biodiversity, is a particularly relevant issue in the Republic of Ireland where there is a target afforestation rate of 20,000 ha/year (Department of Agriculture Food and Forestry 1996). All grant-aided afforestation must comply with the *Forest Biodiversity Guidelines* (Forest Service 2000b), which includes requirements for plantations to contain 5-10% open space (although this requirement can be modified in small plantations). This requirement provides an opportunity to mitigate potential adverse impacts of afforestation on biodiversity, and, in some situations, to enhance biodiversity. However, more detailed knowledge of the biodiversity of open spaces within commercially managed plantation forests is required to make the most of this opportunity. The rationale behind our study was to contribute towards the development of this knowledge, using hoverflies (Diptera, Syrphidae) as an indicator group.

Hoverflies have been recommended as a suitable group for use in site evaluation due to the relative ease of identification, the availability of reliable species lists, good knowledge of species habitat associations and larval microhabitats, occurrence in nearly all terrestrial and freshwater habitats, the range of generation times providing information about short and longer term changes in site conditions, and the availability of standardised sampling techniques (Speight 1986; Speight *et al*. 2000). Hoverflies have been used as indicators of agricultural pollution, habitat disturbance and habitat quality (Sommagio 1999). Some examples include their use as indicators of ancient woodland in Britain (Stubbs 1982) and assessment of biodiversity and ecosystem function in alluvial habitats in France, Ireland and the Netherlands (Castella & Speight 1996; Castella *et al*. 1994; Reemer *et al*. 2005).

In recent years, information about European hoverflies has become widely accessible through the development of the Syrph The Net database (Speight *et al*. 2004). This includes coded information on species macrohabitats, microsites, traits and range and status; it is updated annually. The database can be used to analyse recorded species assemblages in relation to their habitat associations. The database also includes a detailed review of the Irish hoverfly fauna (Speight 2000a). The availability of this database has made hoverflies a powerful tool for biodiversity assessment (see Speight 2000b; Speight & Castella 2001).

Therefore, using hoverflies as an indicator group, the objectives of our study were:
1. To assess the role of open spaces in maintaining biodiversity within plantation forests.
2. To determine the factors affecting hoverfly biodiversity in open spaces in plantation forests.
3. To make recommendations for planning and management of open spaces in plantation forests to enhance biodiversity.

Nomenclature follows Stace (1997a) for vascular plants and Speight *et al.* (2004) for hoverflies.

7.2 METHODS

7.2.1 Study Design

To reduce the effect of large-scale environmental variation among sites, we used a clustered approach to study site location. We surveyed 12 sites, located in two geographic clusters: Cork (Co. Cork, including adjacent areas of Cos. Kerry and Limerick) and Wicklow (Co. Wicklow, including adjacent areas of Co. Dublin). We selected sites from a GIS forest inventory database, and chose sites that had a range of configurations of open spaces. Within each cluster, we standardised, as far as possible, environmental factors such as soil type. In particular, we took care to make sure that the habitat/vegetation types of the open spaces in all the sites within a cluster were broadly comparable. All sites were mature plantations of Sitka spruce (*Picea sitchensis*) of at least 80 ha in size. The sites in the Wicklow cluster were on podsols with rock outcrops and with dry humid-acid grassland/dry heath vegetation in the unplanted open spaces. The sites in the Cork cluster were on deep blanket peats and peaty podsols with modified blanket bog vegetation in the unplanted open spaces.

7.2.2 Hoverfly sampling

We used Malaise traps to sample hoverflies. In each site we installed two Malaise traps on forest roads, and two Malaise traps in unplanted open spaces away from forest roads (glades). The position of each trap in relation to nearby trees and shrubs was selected to ensure that it was exposed to continuous sunlight between, at least, the southeast and southwest (i.e. for the period of the day when hoverfly activity is greatest). The traps were located at least 100 m apart, and at least 100 m from the edge of the plantation, where possible. The traps were located within 10 m of the edge of the open space, so that they sampled both the open space and the forest fauna.

The forest road traps were located, where possible, on east-west sections of forest roads. In some sites, the shading and distance criteria effectively determined the location of the forest road traps. Where there was a choice of locations for the forest road traps in a site, the traps were located to represent the range of variation in the forest road habitat: e.g., forest road sections with and without broadleaved trees/shrubs.

The glade traps were located in the two largest non-forest road open spaces in each site. Where a site only contained one non-forest road open space of sufficient size to meet the shading criterion, both traps were located in this open space, and the 100 m separation criterion was relaxed. These open spaces were mainly habitats that had been retained without physical modification from before the plantation had been established. However, in one case ground preparation had been carried out, and in two other cases the open spaces were located along wide rides that had been occasionally used by motor vehicles.

The Malaise traps were operated continuously from early May to early September 2003 to cover the main period of flight activity. The contents of the traps were collected approximately every three weeks. All hoverflies caught in the Malaise traps were identified to species.

At two sites, damage to some of the Malaise traps prevented us from obtaining a full sample. Therefore, these sites are excluded from analyses carried out at the site scale, but the traps that were successful in these sites are included in the analyses carried out at the trap scale.

7.2.3 Habitat recording

We used the Syrph The Net macrohabitat classification (Speight *et al*. 2004). This classification is based upon the CORINE classification (Commission of the European Communities 1991), but with modifications to reflect habitat characteristics of importance to hoverflies that are not covered by CORINE. We recorded the spatial extent of each major macrohabitat type, and the frequency of supplementary habitats, in a 100 m radius around each malaise trap.

We recorded habitat structure using the categories defined in Table 25. These are based on the Syrph The Net microhabitat classification (Speight *et al*. 2004), because this work codifies the relationships of hoverfly species with these microhabitat categories. We estimated the cover of these habitat structure categories using the Dominant-Abundant-Frequent-Occasional-Rare (DAFOR) scale.

For the traps located on forest roads, we recorded the average width of a 200 m length of the forest road centred on the trap location. To do this, we divided the forest road into sections of approximately constant width and then calculated the overall average width as the average of the individual sections weighted by their length. We measured the widths as the distance between the trunks on each side of the road (inter-trunk width) and the distance between the edge of the canopy on each side of the road (inter-canopy width). The forest road widths include the paved surface and unplanted verges between the paved surface and the forest edge. The paved surface was 2.5-4 m wide (with 84% of the measurements between 2.5-3 m), so the variation in forest road width is almost entirely due to the width of the unplanted verges.

We used digitized aerial photography to measure the overall amount of open space habitat in the vicinity of each trap, and at the site scale. We used three distance bands of 100 m, 200 m and 300 m from each trap. The 300 m distance band was selected as the upper limit because above this distance, the open space component becomes dominated by habitats outside the plantation boundary. For the site scale, we created combined buffers of the same distance bands from all the traps. We classified open spaces as broadleaved (areas of broadleaved scrub/woodland), clearfell, forest roads, outside (open space habitat outside the plantation boundary), undeveloped (where tree crop failure has resulted in gaps in the canopy), unplanted, young forestry (pre-canopy closure), and windthrow.

Table 25. Habitat structure categories.

Category	Definition[1]
Mature trees[2]	Canopy trees that have reached the age of fructification without yet developing features of "overmature/senescent" trees.
Understorey trees[3]	Trees of more than 2 m in height that at maturity do not reach the forest canopy, e.g. *Crataegus monogyna*, *Sorbus aucuparia*, or are immature specimens of canopy-forming species.
Tall shrubs[3]	Woody plants between the heights of 0.5 and 2 m, e.g. *Ulex europaeus*, *Salix* sp., *Rubus fruticosus* and young trees (saplings).
Low shrubs (bramble)	*Rubus fruticosus* up to the height of 0.5 m.
Low shrubs (dwarf shrubs)	Ericoids (e.g. *Vaccinium myrtillus*, *Calluna vulgaris*, *Erica tetralix*) and gorse (*Ulex* sp.) up to the height of 0.5 m.
Low shrubs (conifers)[4]	Conifers up to the height of 0.5 m.
Tussocks	Tussocks formed by grasses, sedges and rushes (Graminae, Cyperaceae, Juncaceae).
Tall herbs	Tall, strong forbs over 0.5 m in height, e.g. *Digitalis purpurea*, *Cirsium palustre*, *Senecio jacobaea*, *Urtica dioica*.
Short herbs	Ground-living, non-woody flowering plants up to 0.5 m in height, and including non-tussocky grasses exceeding this height.
Submerged sediment/debris	Permanently submerged sediment or debris in running or standing waters.
Water-saturated ground	Permanently or temporarily (at least for some weeks) water-logged soil surface layer.

[1] modified from Speight *et al*. (2004).
[2] only broadleaved trees were recorded in this category.
[3] cover of broadleaved and coniferous trees/shrubs recorded separately in these categories.
[4] this category is not included in Speight *et al.* (2004).

7.2.4 Species groupings

In order to compare different facets of hoverfly biodiversity, we have used a number of species groupings based on the recorded macrohabitat and microhabitat associations in the Syrph The Net database (Speight *et al*. 2004). A special feature of the macrohabitat classification is the concept of supplementary habitats. A supplementary habitat is a small habitat feature that can occur in association with a macrohabitat (e.g., a wet flush in a forest). Supplementary habitats are used to refine the coding of the association of hoverfly species with macrohabitats: in many cases, a hoverfly species is only considered likely to occur in a particular macrohabitat if the supplementary habitat is present.

The primary classification divided the recorded species into four groups, based on their predicted association with open space macrohabitats: forest species, small open space species, large open space species, and open scrub species. Forest species are those that are predicted to occur in mature spruce plantations (Syrph The Net macrohabitat code 1811) without requiring the presence of supplementary habitats and, therefore, should not require open space habitats. Small open space species are those that are predicted to occur in mature spruce plantations with the supplementary habitats tall herb clearings/tracksides (211f), grassy clearings/tracksides (234f), and small open areas with flushes (731f). Various additional supplementary habitats are also coded, but none of these add any additional species to the predicted mature spruce plantation fauna, apart from *Eristalis lineata*, coded for brook edges (7442f). We defined large open space species as those that are not coded to occur in mature spruce plantations, even with the presence of supplementary habitats, but are coded to occur in the open space macrohabitats that were present in most of the sites that we studied: unimproved humid grassland (23113) and moorland (24); note that in the Syrph The Net macrohabitat classification, drying blanket bog dominated by *Molinia caerulea* is classified within the oligotrophic sub-category (231132) of unimproved humid grassland. Two additional macrohabitats, unimproved dry acidophilous grassland (231121) and lowland heath (251), were frequent in the Wicklow sites, but these macrohabitats do not add any additional species to those already predicted by the unimproved humid grassland and moorland macrohabitats. We defined open scrub species as those that are not coded to occur in closed-canopy mature spruce plantations, but are coded to occur in Atlantic thickets (122) with the supplementary habitats tall herb clearings/tracksides (211f) and grassy clearings/tracksides (234f). Note, that some species in this category overlap with the small and large open space species.

We also used three additional classifications, based on macrohabitat associations, to distinguish anthropophobic and anthropophilic (Boycott 1934; Speight & Castella 2001) species and species associated with surface water features. Anthropophobic species are species which generally do not survive in habitats subject to intensive use by humans: i.e., they are dependent upon semi-natural habitats and will not persist in intensively farmed landscapes, Anthropophilic species are species which generally do survive in habitats subject to intensive use by humans and can utilise habitats that typically occur in intensively farmed landscapes. We defined anthropophilic species as including all species predicted to occur in any of the following habitats: heavily-grazed improved grassland (23212), intensive grassland (233), and cultural macrohabitats (5) apart from orchards (54) and urban parks (55). We defined anthropophobic species as species that are not predicted to occur in any of these habitats. We included species associated with conifer plantations in our anthropophobic category, if they are not associated with any other anthropophilic habitat, because an objective of our analyses was to determine whether plantation forests can support species that otherwise cannot persist in intensively farmed landscapes. We defined surface water associated species as those that are coded for standing (71) and running (72) freshwater macrohabitats; the latter includes flushes and springs.

We used classifications, based upon microhabitat associations, to define species groups that might be associated with trees and shrubs and with wet habitat features. These classifications follow the codings of species in the microhabitats spreadsheet of the Syrph The Net database (Speight *et al*. 2004).

7.2.5 Data analysis

We used paired t-tests to examine differences in species richness between open space types, with mean species richness per site as the response variable. We used blocked multi-response permutation procedures (MRBP) to test for differences in species assemblages between open space types, with Euclidean distance measures and median alignment within blocks. OS TYPE was the grouping variable and SITE was the blocking variable with the log transformed summed abundance in pairs of forest road and glade traps as the response variable. Caution is required in interpreting abundance data from Malaise trap catches. However, we consider that, in this context, it is appropriate to use abundance data because the comparisons are being made within sites.

We used Pearson's correlations to investigate the relationships of forest road width and open space area with species richness of the forest, small open space, large open space and open scrub hoverfly species groups. We carried out separate analyses of the relationships with open space areas at the trap and site scales. For the analyses at the site scale we divided the amount of each open space type by the total area of the buffer, as the latter varied between sites due to overlapping buffers for individual traps.

In order to investigate relationships between habitat structure and hoverfly species richness we first carried out ordination analyses of the habitat parameters followed by correlation of species richness of selected hoverfly groups with the ordination axes. We used non-metric multidimensional scaling analysis (NMS) because the datasets were not suitable for Principal Components Analysis (PCA), a method commonly used for this purpose, due to the presence of a large number of zero values in the datasets. However, exploratory analyses with PCA produced very similar results. For the NMS analyses we used Sørensen (also known as Bray & Curtis) distance measures and the parameter set-up shown in Table 26. In this way, we examined the relationships of numbers of tree and shrub associated species with vegetation structure, and between numbers of wet habitat associated species with aquatic microhabitats and supplementary habitats. For the latter analysis we had to exclude two traps that had no wet habitat features, as samples with no non-zero values cannot be included in NMS analyses. We did not examine relationships of habitat structure with herb layer, ground layer and root zone species because these species groups are ecologically heterogeneous and unlikely to respond in a simple way to the habitat structure parameters that we recorded.

We used PC-Ord (McCune & Mefford 1997b) for multivariate analyses, and SPSS (SPSS 2004b) for all other analyses. We tested data for normality and homogeneity of variance before using parametric statistics.

Table 26. Standard parameter set-up used for NMS.

Parameter	Value used
Number of axes	6
Number of runs with real data	20
Stability criterion	0.001
Iterations to evaluate stability	10
Maximum number of iterations	500
Step down in dimensionality	Yes
Initial step length	0.20
Starting coordinates	Random
Number of runs of Monte Carlo test	50

7.3 RESULTS

7.3.1 Hoverfly assemblages

We recorded a total of 75 species (see Appendix 8), of which 65 are associated with closed canopy spruce forest, small open spaces, large open spaces or scrub habitats, and five are associated with miscellaneous macrohabitats that occurred in, or adjacent to, particular sites. Therefore, only five species were recorded whose occurrence could not be related to macrohabitats in, or adjacent to, the trapping locations. The majority (nearly 80%) of the recorded species are associated with open space habitats rather than closed-canopy forest (Figure 23). Overall, more of the recorded species are associated with large open spaces compared to small open spaces, but the mean species richness per site was similar in these two categories. The most common habitat association of the recorded species is with humid grassland habitats, but there were more anthropophobic species associated with moorland and surface water habitats (Figure 24). In fact, most (73%) of the anthropophobic species associated with humid grassland and moorland are also associated with surface water habitats. While the total and mean per site species richness of scrub-associated species was relatively high, very few of these species are anthropophobic.

We recorded three species that are listed as threatened by Speight *et al.* (2004): *Platycheirus amplus*, *Xanthandrus comtus*, and *Xylota florum*. However, the latter species was probably not associated with habitats present inside the plantation.

7.3.2 Influence of open space type

The numbers of species associated with large open spaces were slightly, but significantly, higher in glades compared to forest roads, although there was no difference in the numbers of species associated with small open spaces (Table 27). Assemblage structure was significantly different between forest roads and glades (MRBP analysis: probability of a smaller or equal delta = 0.0008), although the effect size was small (A = 0.10), and the indicator species analysis only identified six indicator species (Table 28).

Table 27. Comparison of hoverfly species richness between forest roads and glades.

Species group	Mean species richness/site (SD)		Paired t-test	
	Forest road	Glade	t	p
Forest	7.8 (1.3)	8.1 (1.0)	-0.79	0.45
Small open spaces	10.5 (1.6)	10.9 (1.6)	-0.76	0.46
Large open spaces	5.7 (2.1)	7.5 (1.4)	-2.84	0.02
Open scrub	4.3 (1.6)	4.3 (1.6)	0.22	0.83
Anthropophobic	6.4 (1.1)	6.7 (1.3)	-0.89	0.40

Table 28. Indicator species for forest roads and glades. The max IndVal is indicated in bold.

	Species group	Forest Road	Glade	P value of max IndVal
Eristalis interrupta	Large open spaces	8	**59**	0.033
Melanostoma mellinum	Small open spaces	**68**	26	0.001
Meliscaeva cinctella	Forest	**56**	44	0.037
Sphegina clunipes	Forest	41	**59**	0.041
Syrphus ribesii	Forest	**62**	31	0.056
Volucella bombylans	Small open spaces	**42**	3	0.099

7.3.3 Influence of open space amount

The numbers of species associated with small and large open spaces were positively correlated with the average width of the forest road within 100 m of the Malaise traps (Figure 25 and Table 29). There were no significant relationships between the richness of these species group with forest road width at the trap location, or between the richness of other species groups and forest road width.

Table 29. Pearson correlations between hoverfly species richness and forest road width (n = 21).

Species group	Inter-canopy (100 m)	Inter-trunk (100 m)	Inter-trunk (trap location)
Forest	-0.090, p = 0.70	-0.041, p = 0.86	0.095, p = 0.68
Small open spaces	0.387, p = 0.083	**0.462, p =0.035**	0.365, = 0.134
Large open spaces	**0.489, p =0.024**	**0.479, p =0.028**	0.402, p = 0.071
Open scrub	0.192, p = 0.40	0.237, p = 0.30	0.163, p = 0.48

There were no significant relationships between any of the measures of open space areas and the numbers of hoverfly species.

7.3.4 Habitat parameters

The three axes of the NMS ordination of the vegetation structure parameters explain 53%, 17% and 22%, respectively, of the variance in the species data. Axis 1 represents a gradient from broadleaved trees and shrubs to coniferous shrubs (Table 30). The numbers of tree/tall shrub foliage species were negatively correlated with axis 1, even when conifer-associated species were considered separately, or when this group was split into anthropophobic and anthropophilic species (Table 31).

Table 30. Pearson's correlations of vegetation structure parameters with the axes derived from NMS ordination of these parameters (n = 43).

	Axis 1		Axis 2		Axis 3	
	r	r-sq	r	r-sq	r	r-sq
Mature trees	-.303	.092	.160	.026	-.040	.002
Understorey trees (broadleaved)	-.887**	.786	-.300	.090	.099	.010
Tall shrubs (broadleaved)	-.897**	.805	-.226	.051	.160	.026
Low shrubs (bramble)	-.753**	.568	-.457**	.208	.107	.012
Low shrubs (dwarf shrubs)	.090	.008	.342	.117	-.784**	.614
Tall herbs	-.101	.010	-.536**	.288	.692**	.479
Tussocks	-.266	.071	-.210	.044	-.514**	.264
Short herbs	.391	.153	-.032	.001	.720**	.518
Understorey trees (conifers)	.391	.153	-.631**	.398	-.043	.002
Tall shrubs (conifers)	.525**	.275	-.432**	.187	.007	.000
Low shrubs (conifers)	.500**	.250	-.007	.000	.115	.013

** p < 0.01

Table 31. Pearson's correlations of hoverfly species richness with the axes derived from NMS ordination of the vegetation structure parameters (n = 43).

Species group	Axis 1	Axis 2	Axis 3
Trees, understorey trees, and tall shrubs			
Conifer species	-0.41**	-0.11^{ns}	-0.13^{ns}
Non-conifer species	-0.49**	-0.29^{ns}	-0.14^{ns}
Anthropophobic species	-0.44**	-0.12^{ns}	-0.22^{ns}
Anthropophilic species	-0.47**	-0.24^{ns}	-0.09^{ns}
Low shrubs, excluding species associated with trees/understorey/tall shrubs	0.43*	0.20^{ns}	-0.29^{ns}

** p < 0.01, *p < 0.05, ^{ns} not significant.

The two axes of the NMS ordination of the wet habitat parameters explain 65% and 23%, respectively, of the variance in the species data. Axis 1 represents a gradient of increasing influence of most wet habitat features, except drainage ditches (Table 32). The numbers of species associated with submerged sediment, water-saturated ground and surface water habitats were positively correlated with axis 1 (r = 0.36, r = 0.43 and r = 0.35, respectively, p < 0.05). There were no significant correlations of species richness with axis 2.

Table 32. Pearson's correlations of wet habitat parameters with the axes derived from NMS ordination of these parameters (n = 41).

	Axis 1		Axis 2	
	r	r-sq	r	r-sq
Submerged sediment/debris	0.651**	0.423	0.036	0.001
Water-saturated ground	0.764**	0.583	-0.472**	0.223
Brook edge	0.648**	0.420	-0.108	0.012
Drainage ditch	0.141	0.020	0.786**	0.618
Open flush	0.586**	0.343	-0.509**	0.259
Seasonal brook	0.746**	0.556	-0.350	0.123

** p < 0.01

7.4 DISCUSSION

Our results illustrate the importance of open spaces for the maintenance of hoverfly biodiversity in forestry plantations. Overall, nearly 80% of the hoverfly fauna was associated with some form of open space habitat, and would not be predicted to occur in closed-canopy forests, and there was a greater number of open space species compared to closed canopy species in each individual plantation that we studied. Studies of carabid beetles in plantation forests have also found greater species richness in open spaces compared to the closed canopy habitat (Butterfield *et al*. 1995; Day & Carthy 1988), although a study of vascular plants found generally greater biodiversity of shade species compared to open space species in individual woods (Peterken & Francis 1999). The greater number of open space hoverfly species was not simply due to the open spaces being colonised by widespread generalist species: around one-third of these open space species are anthropophobic, and also include some scarce or rare species. Open spaces may even be important for species associated with closed-canopy spruce plantations, as indicated by their association with broadleaved trees and shrubs (which, in the plantations that we studied, tend to be restricted to the edges of open spaces).

We found few differences between hoverfly biodiversity in forest roads and unplanted glades. There was a higher number of species associated with large open spaces in the latter, and significant differences in assemblage structure between forest roads and glades. However, the magnitudes of the differences were small, and the species identified by the indicator species analysis as characteristic of the different assemblages were not very informative. Our results may be partly confounded by the movements of adult flies, and, in particular, the use of forest roads as flight paths. This may cause the forest road assemblages to be inflated by species that are merely passing through the habitat rather than breeding there. While, in general, such effects should be less likely to bias our comparisons of assemblage structure, as this included relative abundance as well as species occurrence, abundances of migratory species along forest roads might not be closely related to their use of the forest road habitats for breeding. However, we repeated the MRBP analysis, excluding species coded as strongly migratory by Speight *et al.* (2004), and obtained very similar results (MRBP analysis: probability of a smaller or equal delta = 0.0002, effect size A = 0.07). Tree-lined field boundaries have been shown to act as barriers to dispersal to three species of hoverfly (Wratten *et al*. 2003), so forest roads might also be expected to increase the permeability of the plantation to "open country" species (i.e., species occurring in habitats outside the plantation). However, we found no difference in the numbers of open space species in glades connected to forest roads compared with the numbers in glades isolated from other open space habitat by mature spruce.

It is possible that differences between sites could obscure differences between open space types. However, we also carried out exploratory ordination analyses and these also only indicated weak structure in the assemblage variation linked to open space type. The lack of differences between the hoverfly assemblages of forest roads and glades may reflect the association of many species with small-scale habitat features that are equally likely to occur on forest roads and glades. This particularly is likely to be true of the nearly 50% of the fauna that is associated with surface water features. However, our results should not be interpreted as showing that the hoverfly biodiversity of forest roads and unplanted glades is always similar. For the purposes of our study, we deliberately selected forests with open space habitats that were comparable between sites and, therefore, represented widespread habitats. Open spaces with localised semi-natural habitats may well contain a more specialised hoverfly fauna that will not be maintained in the disturbed habitat conditions that occur along forest roads.

The biodiversity of various invertebrate groups in open spaces within forest plantations has been shown to increase with decreasing levels of shade (Greatorex-Davies & Sparks 1994; Greatorex-Davies *et al*. 1993) and shade levels will tend to decrease as open space size increases (see Section 4.4.1). We found that hoverfly species richness was strongly related to forest road width but showed no relationship with overall amounts of open space. It could be argued that relationships between forest road width and hoverfly species richness may just reflect better trapping conditions in wider forest roads. However, we were careful to locate the Malaise traps in positions where, even on narrow forest roads, they would be exposed to sunlight through much of the day. Moreover, the relationships that we found were with the species groups that, *a priori*, were considered likely to respond to wider forest roads. Also, the relationships with forest road width occurred at the 200 m scale, not at the trap location (as would be expected if they were due to trapping effects). It is interesting to contrast the significant relationships of species richness with forest road width, with the lack of any relationships with open space area. This difference may be because open space area, at the resolution at which we measured, is a very crude measure, and says little about habitat quality. Increasing forest road width, on the other hand, is likely to be associated with increased habitat diversity, as very narrow forest roads simply do not have the space to develop representative open space habitats. It is difficult to say from our data whether there is a threshold width, although it would seem likely that one should occur as there is no obvious reason why species richness should continue to increase in very wide forest roads. In fact, species richness might be expected to decrease in very wide forest roads due to exposure and wind tunnel effects. The Irish *Forest Road Manual: Guidelines for the design, construction and management of forest roads* (Ryan *et al*. 2004) recommends a width of 15 m (inter-trunk) for new forest roads. Our data would certainly suggest that forest roads narrower than this width will have reduced hoverfly biodiversity, and that even 15 m may be too narrow for optimum hoverfly biodiversity. In the forest roads that we surveyed an inter-trunk width of 15 m would approximate to around 2 m widths for each verge. In many cases, this may be too narrow for well-developed open space habitat to develop, given factors such as shading, steep banks, drainage ditches, etc.

A relationship between the presence of broadleaved trees and shrubs in conifer plantations and invertebrate biodiversity has been suggested (Ferris & Carter 2000a; Warren & Fuller 1993), but has not been previously demonstrated to our knowledge. The relationship that we found of tree and shrub-associated hoverfly species with broadleaved trees and shrubs is not surprising with regard to the species not associated with closed-canopy spruce, but it is notable that this relationship remained even when the analysis was restricted to the species that are associated with the latter. This may reflect a requirement of adult hoverflies of many woodland species for sources of pollen and nectar in spring (see Branquart & Hemptinne 2000), such as can be provided by broadleaved trees and shrubs. The main broadleaved tree species that we recorded were *Betula pubescens* Ehrh. and *Salix cinerea* L., with some *Alnus glutinosa* (L.) Gaertn., and *Sorbus aucuparia* L. *S. cinerea* and *S. aucuparia* are good sources of nectar for hoverflies (M.C.D. Speight, pers. comm.). The relationships that we have found are not with open space habitat *per se*. However, in practice the only opportunity in spruce plantations for significant cover of broadleaved trees and shrubs to develop is in open spaces, because they will usually be out-competed by the more vigorous growth of the conifers within the closed-canopy areas. Natural regeneration of spruce along some forest roads resulted in the presence of understorey, tall shrub and low shrub conifers. The negative association of tree and shrub-associated hoverfly species with the presence of naturally regenerating conifers reflects the negative relationship in our study sites between broadleaved tree and shrubs and conifer regeneration. This latter relationship could be due to competitive effects, or may just reflect differences in soil conditions.

We are not aware of any previous studies that have examined the contribution of small wet habitat features to invertebrate biodiversity in forest plantations. Our results illustrate the value of these features for the maintenance of hoverfly biodiversity. Nearly 50% of the fauna that we recorded has associations with surface water features (mainly streams and flushes) and there were generally consistent relationships between richness of species groups associated with wet habitat features and the relevant habitat structure parameters. However, one of the two traps that had to be excluded from the NMS analysis (because it had no non-zero values for the wet habitat parameters) did not conform to this pattern. This trap was located on a hillside above a river valley, and, while there were no wet habitat features within 100 m of the trap, a high number of wet habitat species were recorded, presumably originating from the river valley. The combination of open space and wet habitat is likely to be important in maintaining the biodiversity of hoverfly species associated with wet habitat features, as over 90% of the species predicted to occur in surface water habitats are not predicted to occur in closed-canopy spruce forests (from data in Speight *et al*. 2004). Indeed, Peterken (1999) recommended that open spaces in plantation forests should be concentrated around wet ground as this is where open spaces are concentrated in natural conifer forests. The Irish *Forestry and Water Quality Guidelines* (Forest Service 2000d) requires that afforestation must include unplanted buffer zones (10-25 m wide depending upon slope and soil type) around aquatic habitats. Therefore, in addition to mitigating water quality impacts from afforestation (which is the purpose of this measure), this requirement is also likely to promote the maintenance of hoverfly biodiversity associated with wet habitat features. However, small wet habitat features that are not marked on the standard six-inch Ordnance Survey mapping are not covered by this requirement. In the areas that we surveyed, only 20% of the 33 wet habitat features (excluding drainage ditches) that we recorded were shown on these Ordnance Survey maps.

7.4.1 Further research required

Our results and conclusions are based upon Malaise trapping of adult hoverflies. As discussed above, movement of adults away from their breeding habitats can complicate the interpretation of this kind of data. Therefore, research to confirm the larval microhabitats of the hoverfly fauna would improve our understanding of the responses of this group to open space design and management. This kind of research could be carried out by emergence trapping, for species that develop in the ground and in low vegetation, and by direct searching for species that develop on tree and shrub foliage.

Another potentially important aspect of hoverfly ecology that our results have highlighted is the role of broad-leaved trees and shrubs as pollen and nectar sources for adult hoverflies. Further research, by direct observation of adult hoverfly behaviour, would be useful to confirm this, and to investigate the value of different species of trees and shrubs.

In Section10.12, we discuss issues where further research would be likely to yield results of direct relevance to the development of guidelines for open space management in plantation forests. Hoverflies would be a suitable group for most of the research topics suggested. However, hoverfly surveys by Malaise trapping may not be very sensitive in detecting responses to manipulations of grazing pressure because of the difficulties of detecting responses at small spatial scales.

7.5 CONCLUSIONS

Open space habitats are of major importance for the maintenance of hoverfly biodiversity in conifer plantations. If the same is true of other indicator groups, then biodiversity planning and management in conifer plantations should focus on the open space habitats rather than the closed-canopy areas. For hoverflies, it is the quality of the open space habitat that is important, rather than the overall amount. Open space habitats containing broadleaved trees and shrubs and wet habitat features (including small-scale features such as wet flushes and temporary streams) should be promoted. Forest roads appear to be able to support broadly similar hoverfly biodiversity to unplanted glades, where the latter do not contain particularly localised habitat features. However, the standard forest road width currently recommended for Irish plantations is probably too narrow, in many cases, for maintaining well-developed open space habitat in mature spruce forests. Therefore, wider sections of forest road and/or unplanted glades should be included in such forests.

8 BIRDS

8.1 BIRDS IN THE FOREST

8.1.1 Introduction

Open spaces contribute to biodiversity of forests by providing habitat both for specialists of forest edges and glades, and for biota of open habitats which are rare or absent in the landscape outside the forest. In a study of disturbance regimes in a range of broadleaved woodlands in England, Blackburn (1996) found that while "semi-natural woodlands have high potential genetic variability, high probability of persistence of gap species, and a high potential to support 'edge' species, the opposite is the case for plantations". However, there are opportunities for biodiversity enhancement in modern forest plantations, and Peterken (1996) considered management of open space to be the most important factor in determining the contribution of plantations to conservation. Fuller and Peterken (1995) state that the most important function of management in plantations from a biodiversity point of view is to enable wildlife-rich open spaces to be maintained and renewed, and that woodland managers should "seize every opportunity" to diversify new woodland.

The presence of forest gap specialists and some birds of open habitat and scrub has been found to increase the diversity of bird assemblages in and around forest gaps created by natural processes such as tree senescence (Fuller 2000; Keller *et al*. 2003) and storm events (Faccio 2003). Bird species richness can also be increased by the artificial gaps that are created by various tree-harvesting techniques (Greenberg & Lanham 2001; Moorman & Guynn 2001). However, other studies have found that gaps have a negligible (Robinson & Robinson 1999; Tomialojc & Wesolowski 2004) or even negative (Wardell-Johnson & Williams 2000) effect on bird species richness. Studies in landscapes where a relatively high proportion of natural forest cover has been retained, such as in eastern Europe (Boncina 2000) and western USA (Chambers *et al*. 1999), have found that frequency and size of gaps increase with intensity of forest management, restricting many forest specialist bird species to areas where the influence of management on forest structure is low or altogether absent. In NE Amazonia, Thiollay *et al.* (1997) attributes decreases in bird species richness that follow selective felling to increases in the density of understorey vegetation, increased exposure to humans and predators, and avoidance of gaps by some forest specialist species. However, in the same area, nearly half of the rare species found in forests were associated with edges or gaps, and were more abundant in habitats outside the forest zone.

Open space is probably of particular importance in Irish forests, as the suite of forest specialist bird species found in continental Europe and the UK, is almost entirely absent from Ireland. In a study of the breeding bird assemblages in Bialowieza forest in Poland, Fuller (2000) compared the abundance of birds at gaps with that of birds in closed canopy forest. The species he found to be more abundant in non-gaps (Wood Warbler *Phylloscopus syllybatrix* and Red-breasted Flycatcher *Monocedula parvus*) are either absent from or extremely rare in Irish forests. In contrast, those species he found to be more common in gaps (Dunnock *Prunella modularis*, Blackcap *Sylvia atricapilla*, and Chiffchaff *Phylloscopus collybita*) are widespread in Ireland. Fuller (1996) lists 17 British bird species that are confined to woodland habitats, and nine others that, while not confined to woodland, are more abundant within it. Of these 26 species, only one forest specialist breeds in Ireland (Crossbill *Loxia curvirostra*), as opposed to seven (Blackcap, Goldcrest *Regulus regulus*, Long-tailed Tit *Aegithalos caudatus*, Blue Tit *Parus caeruleus*, Great Tit *Parus major*, Coal Tit *Parus ater* and Treecreeper *Certhia familiaris*) of the nine forest generalists. The near-absence of bird species adapted to high forest habitat means that the creation of open space in forest does not run the risk of displacing forest specialist species. On the contrary, it is more likely that, in closed-canopy Irish plantations, bird diversity will largely be a function of the species supported by habitats found at the forest edge, and in open spaces within the forest.

A few studies have investigated the importance of open space for birds of Irish forests. Nairn and Farrelly (1991) found that, in a semi-natural woodland in County Wicklow, vegetation at the edge of a wide road gap constitutes the preferred habitat of summer migrants. Duffy *et al.* (1997) surveyed a mixed woodland in County Meath and found that the most important factor contributing to bird diversity was the presence of very wide ridelines there. Pithon *et al.* (2004) reported that a landscape level effect of open habitats in forest/agricultural matrices, where proportion of non-forest farm habitats may determine the suitability for some declining farmland bird species. However, no published study has expressly addressed the effect of open space on birds in a representative sample of Irish commercial forests. The primary aim of this study was to determine the effect of different types and amounts of open space on Irish forest bird assemblages. In order to pursue this objective, we conducted a survey of the breeding birds in twelve conifer plantations containing a variety of types and sizes of open space.

8.1.2 Methods

8.1.2.1 Study site selection

We selected 12 sites in two geographic clusters referred to as Cork (in counties Cork, Kerry and Limerick) and Wicklow (counties Wicklow and Dublin, see Figure 1). We selected sites that had a wide range of configurations of open spaces from a GIS forest inventory database. Within each cluster, we standardised, as far as possible, soil type and habitat/vegetation types of the open spaces. All sites were mature plantations of Sitka spruce (*Picea sitchensis*), at least 80 ha in size. The sites in the Wicklow cluster were on podsols with rock outcrops and with dry-humid acid grassland/dry heath vegetation (as defined by Fossitt (2000)) in the unplanted open spaces. The sites in the Cork cluster were on deep blanket peats and peaty podsols with modified blanket bog vegetation in the unplanted open spaces.

8.1.2.2 Road transects

Two visits were made to each site to sample birds, the first in May and the second in June. Birds were not censused during persistent or heavy rain, or in windy (Beaufort scale 4 or more) conditions. Approximately 1km of road was censused in each study site, between 0800 hours and 1800 hours. The species of all birds detected while walking along the road were recorded, along with their estimated position and distance from the observer. Each length of road was broken down into between 3 and 10 sections that were more or less homogenous according to the following environmental variables: shrub cover (woody vegetation 0.5 – 2 m high) within the road gap, broadleaved tree cover (broadleaved vegetation > 2 m high) within the road gap, brash cover within the road gap, crop tree height, and road gap width. Percentage cover of these variables was estimated for each road section, apart from inter-canopy road gap width, which was measured at a point judged to be representative of the road section. Road section length was measured from aerial photographs. Birds flying over the forest canopy and birds estimated to be more than 10m from the edge of the road gap were excluded from the analysis.

8.1.2.3 Point counts

Bird communities were also sampled using point counts (Bibby *et al*. 2000). Twelve points were counted in each site. Points were situated at a minimum of 100 m apart, to cover a wide a range of open space types and configurations. Points were located in the field using a Garmin GPS 12, accurate to within approximately 5 m (though this distance can increase if satellite cover is compromised by extensive canopy cover), and aerial photographs. Point counts were conducted for 10 minutes, during which time the identity of all birds detected was recorded. Each detected bird was placed into one of three estimated distance categories: 50 m from the observer, 100 m from the observer, and >100 m from the observer. Point counts were conducted between 0700 hours and 1100 hours and between 1300 hours and 1700 hours (GMT). Each point was visited once in the morning and once in the afternoon. Birds flying over the forest canopy were excluded from the survey. The following variables were estimated for an area 50 m around the point: area of shrub cover, area of non-crop tree cover, area of brash cover, total area of open space, crop tree canopy cover and crop tree height.

8.1.2.4 GIS Measurement of Open Space

The area within 300m of each point count location was mapped from digitised aerial photographs using ArcView version 3.2a GIS, and areas not covered by closed canopy conifer forest assigned to seven categories: Unplanted (all non-linear areas within the forest and rides wider than 10 m that had not been planted with trees), Undeveloped (patches within areas of closed canopy forest in which tree growth had been insufficient for the canopy to close), Young (areas of forest planted too recently to be closed canopy), Clearfell, Outside (open areas outside the forest), Road (roads, including all associated open areas such as verges and turning bays) and Woodland (areas of broadleaved woodland both within and outside the forest). The area in each of these categories was calculated for within 50m, 100m, 200m and 300m of each of the point counts.

8.1.2.5 Statistical Analysis

During the road survey, detectability of birds was assumed not to vary greatly between the observer and 10m beyond the forest edge, so numbers of birds detected in each road section were treated as relative abundances. Species richness and abundances of different species and species groups (see below) in the different road sections were standardised for length of road section by taking residuals from linear regressions of these variables on road length.

The numbers of birds detected during point counts was affected both by distance from the observer and by configuration of open space so, in order to minimise the effect of open space on detectability, analyses were restricted to evaluating presence/absence data for each species. Species richness of birds detected during point counts was calculated for three distance categories: within 50m of the observer, within 100m of the observer and all species detected. Measures of species richness within 50m and 100m were used to investigate relationships with open space at the same scales. Species richness for all species detected was used to investigate relationships with open space within 200m and 300m of the point count locations.

The following bird species have been identified by other studies (Bibby *et al*. 1989a; Fuller 1995; Fuller 1996) as being associated with broadleaved woodland: Blackcap, Blue Tit, Bullfinch *Pyrrhula pyrrhula*, Chiffchaff, Great Tit, Long-tailed Tit, Treecreeper and Willow Warbler *Phylloscopus trochilus*. However, these species occurred too infrequently along roads for their abundances to be evaluated separately, so for analysis of the road survey data these species were therefore combined into a single group of birds associated with broadleaved woodland. Chiffchaffs and Willow Warblers were detected sufficiently often during point counts for their presence/absence to be analysed separately, but the other species grouped for the road survey analysis were also grouped for analysis of point count data.

Pearson's R was used to test for associations except where data did not conform to parametric assumptions, in which instances the non-parametric Kendall's τb correlation coefficient was used. For comparison of means of independent samples, t-tests were used except where data did not conform to parametric assumptions, in which case non-parametric Mann-Whitney U tests were used. Where t-tests were used, equal variances were not assumed unless Levene's Test for equality of variance indicated that the two samples did not have significantly different variances. The relationships between binary presence/absence data and GIS-derived open space variables was analysed using binary logistic regression. The statistical significance of the relationships was evaluated using the Wald statistic, which is the ratio of the regression co-efficient B to its standard error. A nested ANOVA was used to test for an effect of site cluster on bird species richness of road sections. SPSS (SPSS 2004a) was used for all analyses.

8.1.3 Results

8.1.3.1 Roads

Mean bird species richness along roads was slightly higher in Cork sites than in Wicklow sites (Figure 26). Sections of Cork road had higher levels of shrub cover (Mann-Whitney U=259, n=27, 37, p=0.001) and broadleaf cover (Mann-Whitney U = 139, n= 36, 27, p < 0.0005). Bird species richness was positively correlated with shrub cover (Figure 27) and with broadleaved tree cover (Figure 28). There was no significant relationship between species richness along roads and road gap width, crop height or brash cover. Shrub cover was positively correlated with relative abundances of Chaffinches *Fringilla coelebs*, Goldcrests, Wrens *Troglodytes troglodytes*, and species associated with broadleaved woodland (Table 33). Broadleaf cover was positively correlated with abundances of Goldcrests and species associated with broadleaved woodland (Table 33). Road sections of 15 m or wider had significantly higher cover of shrubs (Mann-Whitney U = 115.5, n = 9, 54, p = 0.012) and broadleaved trees (Mann-Whitney U = 115.5, n = 9, 54, p = 0.028) than narrower road sections.

Table 33. Correlations between relative abundance of birds found along forest roads and cover of shrub layer vegetation and broadleaved trees. Correlation statistic quoted is the non-parametric Kendall's τb. Statistically significant correlations are shown in bold

Bird Species	Shrub cover (d.f. = 64)		Broadleaf cover (d.f. = 63)	
	τb	p	τb	p
Chaffinch	0.07	0.39	0.01	0.883
Coal Tit	**0.23**	**0.007**	0.04	0.665
Goldcrest	**0.24**	**0.005**	**0.29**	**0.002**
Robin *Erithacus rubecula*	0.07	0.41	**0.18**	**0.050**
Wren	**0.36**	**0.001**	0.10	0.290
Broadleaved birds	**0.33**	**<0.001**	**0.40**	**< 0.001**

8.1.3.2 Point Counts

The mean number of bird species detected during point counts in Cork sites was not significantly different than in Wicklow sites (t = 0.26, d.f. = 135.82, p = 0.79). However, the areas around Cork points had significantly higher cover of shrubs (U = 1814, n = 72, 72, p = 0.001) and broadleaved trees (U = 1897, n = 72, 72, p < 0.0005) than the areas around Wicklow points. Of the environmental variables measured in the field, bird species richness within 50m was positively correlated with shrub cover (τb = 0.13, n = 144, p = 0.039) and broadleaved tree cover (τb = 0.29, n = 144, p < 0.001). Species richness was not significantly correlated with brash cover, crop tree canopy cover or total area of open space. Of the cover variables estimated from aerial photographs, Woodland cover was positively correlated with bird species richness at every scale we investigated (Table 34). Bird species richness was also positively correlated with Road area at a 50m scale, and with Clearfell area and total area of open space at the 300m scale (Table 34). No other remotely measured open space variables were significantly correlated with bird species richness at any scale.

Table 34. Correlations between bird species richness and different elements of open space, at different scales. Scale is given for both open space measurements (e.g. "OS 50m") and measures of species richness (e.g. "R 50m"), the latter referring to the cut-off distance from the observer within which detected birds contributed towards species richness. Correlation statistic quoted is the non-parametric Kendall's τb. Statistically significant correlations are shown in bold.

Scale:	OS 50m, R 50m		OS 100m, R 100m		OS 200m, R all		OS 300m, R all	
OPEN SPACE	τb	p	τb	p	τb	p	τb	p
Clearfell	-0.67	0.37	0.03	0.67	0.11	0.11	**0.15**	**0.028**
Road gap	**0.21**	**0.002**	0.11	0.09	0.08	0.18	0.07	0.25
Unplanted	0.05	0.50	-0.06	0.34	0.02	0.77	-0.08	0.18
Undeveloped	-0.01	0.87	0.01	0.85	0.08	0.25	0.08	0.19
Young forest	0.17	0.82	0.05	0.47	-0.01	0.94	0.00	0.95
Outside	-0.08	0.25	0.11	0.12	-0.00	0.97	0.07	0.29
Woodland	**0.15**	**0.043**	**0.21**	**0.005**	**0.21**	**0.003**	**0.27**	**<0.001**
Total	0.06	0.32	0.10	0.11	0.06	0.29	**0.12**	**0.040**

More bird species were detected in the three sites with an element of Woodland cover than in the nine other sites (Figure 29). Within the three sites that had a Woodland element, more bird species were detected from points that had greater than 0.5ha Woodland within 200m than from other points (Figure 30).

In all sites, Woodland cover within 300m was positively related to occurrence of Chiffchaffs, Jackdaws *Corvus monedula*, Rooks *Corvus frugilegus*, Willow Warblers and other species associated with broadleaved woodland (Blackcap, Blue Tit, Bullfinch, Great Tit and Long-tailed Tit) (Table 36). Area Outside the forest and Total open space within 300m were positively related to occurrence of Meadow Pipits and Skylarks (Table 36). Cover of Woodland within 300m was positively correlated with area of open habitat Outside the forest within 300m (τb = 0.18, n = 144, p = 0.009).

Table 35. Wald statistic for logistic regression of presence/absence of six species groups (MPS = Meadow Pipit and Skylark, RO = Rook, JD = Jackdaw, CC = Chiffchaff, WW = Willow Warbler, and BLS = other species associated with broadleaved woodland) detected during point counts, on area of eight open space habitat categories within 300 m of point counts. Sign before statistic indicates direction of relationship. All independent variables were tested separately from one another. For all regressions n = 144, degrees of freedom = 1.

	Clear	Outside	Road	Undev.	Unpl.	Wood.	Young	Total
MPS	+0.07	+23.61***	-4.59*	+0.71	+0.06	+1.26	+1.73	+33.72***
RO	+0.24	+3.57	-0.13	+4.10*	-0.75	+16.29***	-0.00	+6.07*
JD	+7.02**	+1.03	-0.31	+0.54	-0.91	+24.16***	-0.74	+3.29
CC	-0.12	-0.51	-1.15	+4.59*	+0.16	+13.59***	+0.14	-0.00
WW	+2.63	+0.60	-1.91	+1.13	-0.41	+11.78***	+0.38	+3.18
BLS	-0.12	+1.90	-0.03	+1.80	-0.67	+11.49***	+3.09	+6.22*

* p<0.05
** p<0.01
*** p<0.001

8.1.4 Discussion

The relationship between species richness along forest roads and cover of both shrub layer vegetation and broadleaved trees indicates that these types of vegetation may be important determinants of the bird assemblages along forest roads. The main plant species contributing towards shrub cover were bramble *Rubus fruticosus* and, to a lesser extent, heather *Calluna vulgaris*, bilberry *Vaccinium myrtillus*, and gorse *Ulex* spp. and willow *Salix* spp. Broadleaved tree cover was predominantly composed of willow but also included birch *Betula* spp., holly *Ilex aquifolium*, hawthorn *Crataegus monogyna*, rowan *Sorbus aucuparia*, and alder *Alnus glutinosa*. Many forest bird species in Britain will use a wide range of tree species, exhibiting little evidence of preference for one or a few particular species (Fuller 1996). It is likely that Irish forest birds are similar in this regard, and that the effects of broadleaves on bird assemblages reported in this study are, broadly speaking, a general property of most native, broadleaved tree species. However, some broadleaved tree species will have more specific effects on the value of an area of forest for birds, through the provision of fruits and/or microhabitats that suit the foraging strategies of particular species.

Many other studies in Ireland and Britain have also found a positive effect of shrub cover on forest bird diversity (e.g., Bibby *et al*. 1989b; Currie & Balmford 1982; Duffy *et al*. 1997). Species associated with broadleaved woodland were among the birds identified as being more numerous in shrubby areas. Several studies have found the abundance of one or more of these species to be positively correlated with shrub density (Duffy *et al*. 1997; Fuller & Henderson 1992; Pearson 1979; Smith *et al*. 1992). Numbers of Wrens were also positively correlated with shrub cover. Although Wrens are classed as birds of high forest in North America (Imbeau *et al*. 2001), in Ireland and Britain this species typically nests and forages in habitats with high structural complexity near the ground, and several Irish and British studies have found Wren abundance to be highest in shrubby habitats (Bibby *et al*. 1989b; Duffy *et al*. 1997; O'Halloran *et al*. 1998).

Shrub cover is also correlated with abundances of Coal Tits and Goldcrests, and cover of non-crop broadleaved trees is correlated with abundance of Goldcrests. In Britain and Ireland, both of these species are known to be associated with conifers (Avery & Leslie 1990; Fuller 1995; Hutchinson 1989), so it may be that they are correlated with other environmental sources of variation relevant to birds. Indeed, a study of Welsh forests in which high levels of shrub cover were found to support greater densities of Goldcrests than in other forests attributed this to the greater depth of canopy foliage in shrubby forests (Currie & Balmford 1982). Both shrub cover and canopy depth must depend to some extent on the penetration of light to lower layers of forest vegetation, and could therefore be enhanced at the edge of some wide roads. Alternatively, these vegetation types could exert an indirect influence on the value of an area for birds, if, for example, they increased the numbers of invertebrates in the area. Such an effect of broadleaved trees and shrubs has been suggested by authors of published guidance on management of forest open space for conservation in the UK (Ferris & Carter 2000a; Warren & Fuller 1993), and is also suggested by the findings of the hoverfly component of this study (Section 7). We found that the presence of broadleaved trees and shrubs was positively correlated not only with the numbers of hoverfly species whose larvae feed on broadleaves, but also with numbers of conifer-feeding species. This suggests that broadleaved trees and shrubs may have a positive effect on the invertebrate assemblages of adjacent conifers, where Goldcrests and Coal Tits may be foraging.

Scrub is generally a transient vegetation type (Fuller & Peterken 1995) which, in the context of a natural woodland, would typically occur as part of the succession from open space to closed canopy forest. In many Irish plantation forests, this transitional stage is initiated at harvest, when large areas of forest are converted to open, shrubby habitats following clearfelling and replanting. Areas of pre-thicket forest can support scrub specialists, but in Irish conifer plantations these species disappear after about 10 years, when the crop-tree canopy closes (Wilson *et al*. In Press). The uniform age structure of these forests results in large contiguous areas of closed canopy conifer plantations with no shrub-rich pre-canopy closure stands. In such areas, other sources of shrubby habitats can greatly enhance the bird assemblages, which might otherwise consist entirely of forest specialist and generalist species (Wilson *et al*. In Press).

The lack of a significant relationship between species richness and road gap width is not surprising given the fact that neither shrub layer nor broadleaved tree cover are significantly correlated with this variable. This is because many wide forest road gaps have very low shrub cover. Variables other than width that could influence the shrub layer of a road gap include age of the road gap, soil type, competition with grasses and other plant groups, management history and grazing pressure. However, road gaps less than 15m wide had lower shrub and non-crop broadleaf cover than wider road gaps. The potential for road gaps of this size to support a diverse assemblage of birds may, therefore, be limited.

The greater number of bird species observed along Cork road sections than in Wicklow road sections is consistent with differences in the levels of shrub and broadleaved cover between the two site clusters. Grazing levels in the Wicklow forests were generally much higher than in the Cork sites (Section 4.4.3), due to the much larger densities of deer in the former area. Sustained heavy grazing pressure from deer has negative repercussions for many components of forest biodiversity (Fuller & Gill 2001), including birds, which are impacted primarily through the reduction of shrub and ground layer vegetation (Fuller 2001; Gillings & Fuller 1998). It is possible that high levels of grazing in the Wicklow sites may have restricted the development of shrub and non-crop broadleaved tree layers along the forest roads.

The importance of shrubs and, in particular, broadleaves for birds is further emphasised by the point count data. Shrub layer was positively correlated with bird species richness at the 50m scale, but we were not able to assess its influence at larger scales, as it was not possible to infer shrub cover from aerial photographs. Broadleaved trees and Woodland, on the other hand, were positively correlated with bird species richness at every scale we investigated, from 50m up to 300m, and even at the level of the forest. Moreover, when examined at two scales within the same data set, the relationship between species richness and Woodland cover persists independently at both scales.

The increased species richness associated with broadleaves can be explained by the more regular occurrence of a number of different species in areas where broadleaves are present. Many of these, such as Chiffchaff, Willow Warbler and the group comprising Blackcap, Blue Tit, Bullfinch, Great Tit and Long-tailed Tit, are species that, in Britain and Ireland, are known to occur primarily in broadleaved woodland (Fuller 1995), and whose abundance was found by us to be positively correlated with shrub cover. Each of these species are benefited by broadleaves directly, either through the provision of foraging habitat, nesting locations, or both. Jackdaw and Rook also occurred more frequent in areas of forest with broadleaves. Both of these species typically forage in open, non-forest habitats, and their apparent preference for Woodland cover may be strengthened by the greater amounts of Woodland in areas with high cover of Outside open space. However, in Ireland, both of these species nest in broadleaved trees more frequently than they do in conifers, Jackdaws in trunk cavities or clefts of trees, and Rooks in colonies in the crowns of large trees.

The occurrence of Meadow Pipits and Skylarks is strongly and positively correlated with cover of Total open space. These are the only species that we recorded on more than one occasion that are associated exclusively with open habitats. However, the only component of Total open space with which the presence of these species is strongly correlated is cover of Outside open space. Woodland cover is weakly correlated with occurrence of these species, but this variable is higher around points near the forest edge, as evidenced by the positive correlation between Woodland cover and cover of Outside open space. None of the within-forest open space components explain any variation in Meadow Pipit and Skylark occurrence. This suggests that open space in Irish plantation forests is not suitable for bird species which are typical of non-forest open habitats. This could be because most forest open spaces are not large enough to accommodate such species. Avery and Leslie (1990) came to a similar conclusion regarding open space in British forest plantations.

In census plots in upland conifer plantations in North Wales, Bibby *et al.* (Bibby *et al*. 1989a) found that bird diversity was positively correlated with proportion of broadleaved trees. Furthermore, although overall abundance of birds wasn't correlated with area of broadleaves, the abundance of twelve species associated with broadleaved woodland was. They state that their data show that a given area of broadleaves will have the most impact on the bird diversity of a conifer forest if distributed throughout the area in many small patches, rather than few large ones. However, at least towards the upper range of sizes of the broadleaved patches they surveyed (0.1-2ha), large patches appear to support more broadleaved specialists than the equivalent area of smaller patches. Williamson (1972) recommends setting aside approximately 10% of a new plantation to comprise broadleaved trees and scrub, and warns against making these too small, advising that broadleaves should, where possible, be distributed in discrete or paired areas totalling minima of approximately 0.75ha. Whether such patches are located in internal forest blocks, or in 'bulges' along the edge of the forest, they have the potential to greatly enhance the diversity of birds found in closed canopy plantations.

8.1.4.1 Further research required

The effects of open space on bird diversity in Irish plantations are worthy of further study. These include the effect of broadleaved trees and shrubs in a wider variety of plantation types than investigated here (i.e. of different tree species, and in lowland as well as upland landscapes), and in different configurations throughout the forest, as well as the effects of different broadleaved species. The effect of deer populations on forest bird assemblages, mediated by the effect of grazing on open space vegetation, should also be investigated more thoroughly, in order to determine the grazing regimes that maximise bird diversity in different forest types.

8.1.5 Conclusions

The most important aspects of open space for bird diversity in Irish upland conifer plantations are cover of shrubs and of non-crop broadleaved trees. This is largely due to a suite of relatively uncommon species that rely on these elements of open space vegetation for foraging and/or nesting habitat. Irish forest managers seeking to increase the number of bird species found in their plantations can encourage broadleaf trees and shrubs through a variety of measures. The creation of wider roads and rides would result in unshaded strips through the forest in which shrubs and non-crop broadleaves could grow. This applies especially to rides, as the vast majority of rides in Irish plantations are far too narrow (less than 10m in width) to support a high cover of shrubs or broadleaves. However, disturbance and grazing pressures in such areas must be sufficiently low to allow shrubs and broadleaves to establish and grow. In a similar vein, leaving unplanted margins at the edge of plantations will allow the development of a 'soft' vegetational transition between forest and open habitat, which can be as valuable for birds as interior patches of scrub. In many areas, broadleaved tree species will readily establish in an area if it is released from shade and grazing pressure. However, where local sources of seed are not sufficient to colonise such areas, planting of strips and patches of broadleaves may be necessary. Finally, where deer numbers are high, over-grazing of forest open space is likely to have a negative impact on bird diversity. Control of deer populations in these areas will be a necessary precursor to the development of broadleaves and shrubs within forest open spaces.

8.2 THE DISTRIBUTION OF HEN HARRIERS IN IRELAND IN RELATION TO LAND-USE COVER IN GENERAL AND FOREST COVER IN PARTICULAR

Note: Section 8.2 was submitted to COFORD and EPA in 2005 as a stand-alone report.

8.2.1 Introduction

Hen Harriers *Circus cyaneus* were once widespread in Ireland, but have declined in range and population over the past 200 years, through a combination of habitat loss/degradation and persecution (O'Flynn 1983; Whilde 1993). This decline was reversed between 1950 and 1970, when many upland areas were planted with coniferous forest (O'Flynn 1983). Although the traditional breeding habitat of Hen Harriers in Ireland and Britain is open moorland (Gibbons *et al*. 1993), the ground vegetation of young plantation forests can be more suitable for Hen Harrier nesting and foraging than that of surrounding open habitats, where heather and long grass cover can be limited by heavy grazing or burning (Madders 2003). Hen Harriers in Ireland used newly established conifer plantations for both hunting and nesting, and reached an estimated peak of between 200 and 300 pairs (Watson 1977). However, since 1970, the Hen Harrier population in Ireland has undergone a rapid decline, (Newton *et al*. 1999; Whilde 1993), and more recent estimates have placed the breeding population at about 120 pairs (Gibbons *et al*. 1993; Norriss *et al*. 2002).

This decline has been attributed to agricultural improvement of marginal rough pasture, bogland and scrub, and to the maturation of the Irish forest plantation estate (O'Flynn 1983; Whilde 1993). Hen Harriers cease to use plantations after canopy closure and, until recently, available evidence has suggested that Hen Harriers make little use of young second rotation forests either for nesting or for hunting (Madders 2000; Petty & Anderson 1986). A survey of Hen Harriers conducted from 1998-1999 found that, in some parts of Ireland, nests were often located in restocked conifer forest (Norriss *et al*. 2002). However, in areas such as Wicklow, where there is now little afforestation, Hen Harriers have disappeared, despite wide availability of young, open second rotation forests (Gibbons *et al*. 1993; Norriss *et al*. 2002). Reforested sites may be less suitable for foraging than young new plantations due to the presence of brash and a lower cover of ground vegetation (Madders 2000; Norriss *et al*. 2002). Moreover, forest areas generally have a closed canopy for about two thirds of the forest cycle. This means that even if pre-thicket first and second rotation forests are as valuable to Hen Harriers as the pre-planting open habitats they replace, afforestation will still result in a net loss of habitat to Hen Harriers (Bibby & Etheridge 1993).

In May 2002, nine Indicative Areas (IAs) ranging from 61 to 744 km2 were identified by National Parks and Wildlife as being sufficiently important for breeding Hen Harriers to warrant Special Protection Area (SPA) status, under European Council Directive 79/409/EEC on the conservation of wild birds. All of these areas have relatively high levels of forest cover, and stakeholders in these areas are anxious to allow further afforestation. While it is likely that Hen Harriers will require substantial areas of open habitats if they are to persist in afforested landscapes the size of such areas has not yet been objectively determined. There is, therefore, a pressing need for information on the habitat requirements of Hen Harriers. If the activities of the farming community and other stakeholders are to be curtailed in Hen Harrier SPAs, this should be done in the knowledge of the impact that these activities would have been likely to have on Hen Harriers. Furthermore, even if no further afforestation is sanctioned in these areas, their value to Hen Harriers is likely to change. The high level of forest cover in the SPAs means that their suitability for Hen Harriers is likely to be affected by the relative proportions of open and closed canopy forest within them. It is possible that in some places, Hen Harriers could benefit from further tree-planting if, at some stage in the future, this would provide them with areas in which to hunt or nest at a time when these activities were not well catered for by non-forest habitats.

This paper aims to address some of the gaps in our knowledge about the habitat requirements of Hen Harriers in Ireland. Specifically, we examine whether changes in Hen Harrier distribution over the past three decades can be related to changes in land use, forest cover, intensity of human activity, in order to determine whether this species uses open areas and different stages of the forest cycle more or less often than one might expect from the availability of these habitat types within their ranges. We also estimate threshold areas of foraging and nesting habitat, below which landscapes appear to become unsuitable for Hen Harriers. We use these thresholds to predict the effects that the maturation of the forest estate will have on the Hen Harrier population in the IAs, and generate some preliminary guidelines for those responsible for regulating afforestation in areas of Ireland where breeding Hen Harriers occur.

8.2.2 Methods

8.2.2.1 Data preparation

8.2.2.1.1 Hen Harrier data

Hen Harrier distribution data were taken from three sources. The main dataset that this study uses is from the recent nation-wide survey co-ordinated by Dúchas, Birdwatch Ireland, and the Irish Raptor Study Group (Norriss *et al*. 2002). A concerted effort was made during this survey to census all Hen Harriers breeding in Ireland, covering all areas where they have been known to occur, and a selection of other areas that contain potential breeding habitat. Survey effort was concentrated on obtaining evidence of breeding, so any pairs for which there was no definite or probable evidence of breeding were excluded from our analyses. The position of most breeding pairs found during the survey was estimated to an accuracy of 100 m; the remainder of records were accurate to within 1 km. Most of these data were gathered between 1998 and 2000, but they were supplemented by data collected in 2001-2003. This survey shall henceforth be referred to as the 1998-2003 survey.

The other datasets we analysed were those collected during the surveys for *The Atlas of Breeding Birds in Britain and Ireland* (Sharrock 1976), henceforth referred to as the Old Atlas survey, and *The New Atlas of Breeding Birds in Britain and Ireland: 1988-91* (Gibbons *et al*. 1993), henceforth referred to as the New Atlas survey. Data from the Old Atlas survey referred to 10 km squares, while data from the New Atlas survey referred to 2 km squares (henceforth referred to as 'tetrads'). The highest resolution at which data from both surveys could be analysed was the 10 km square - 100 km^2). For each 10 km square surveyed in both Old and New Atlas surveys, Hen Harriers were recorded as definitely/probably breeding, possibly breeding, or not observed. We combined the first two categories into a single category which we entitled Present; and defined squares in which Hen Harriers were not observed as Absent.

8.2.2.1.2 Environmental data

Four sources of geo-referenced environmental data were used. The two sources of data for forest cover were the Forest Inventory and Planning System (FIPS), a geo-referenced database compiled by the Forest Service that covers all forest stands present in Ireland in 1997, and allocates them to forest type and age categories and the Coillte inventory, which contains more detail about each stand (e.g. planting and projected felling years), but deals only with forests managed by Coillte. We had access to Coillte inventory data for all Coillte-owned and managed forests in the Hen Harrier IAs, but not for Coillte forests in the rest of the country. For nation-wide analyses we therefore derived our forest data entirely from FIPS; whereas for analyses restricted to the Hen Harrier IAs, we used information in the Coillte inventory to classify all forests owned by Coillte, and FIPS data to classify all forests outside of the Coillte estate.

We needed to distinguish forests that could be used by Hen Harriers for either hunting or nesting from forests that were not suitable for these purposes. Broadly speaking, plantations have the potential to be used by Hen Harriers until canopy closure, after which time they will be of little value to them. We identified twelve years as being an age beyond which most commercial forest in Ireland will have closed canopy.
The FIPS database categorises forests into the following types:

1. mature conifer forest,
2. mature broadleaved forest,
3. young conifer forest,
4. young broadleaved forest,
5. PGA (land for which a planting grant had been approved at the time of the database's compilation in 1996. Most of this land would have constituted pre-canopy closure forest during the time of the 1998-2003 Hen Harrier survey) and clearfelled forest.

We worked out the age ranges that these categories correspond to for Coillte forests in the Hen Harrier IAs from the information in Coillte's inventory (Figure 31). Approximately half of the forest area classed as young was more than twelve years old at the time of the 1998-2003 survey; while almost 20% of the forest area classed as mature in FIPS had been felled and restocked by this time (Table 36). On their own, it is therefore highly unlikely that these two categories can be used to reliably discriminate between areas that were used by Hen Harriers and areas that were not.

In the Hen Harrier IAs, however, 63% of the 93594 ha of forest in the FIPS database belongs to Coillte, and so can be aged precisely according to planting year. Moreover, the private estate is characterised by having either very old or very young stands and hence its value to Hen Harrier is more readily assessed than the Coillte estate (Table 37). Coillte afforestation peaked between 1950 and 1980, whereas commercial planting in the private sector was negligible until the mid 1980s, after which time it rapidly overtook afforestation by the state (Government of Ireland 2001). Privately owned forests classed as mature in FIPS are largely old estates, rather than commercial plantations. An indication of this is the proportion of broadleaved and mixed woodland in privately owned mature forests (68%) compared to that in mature Coillte forests (9%). Very few of these old woodlands would have been felled and replanted 12 years or less before the 1998-2003 survey, so on the whole they would not have provided Hen Harriers with suitable forest habitat at this time. Privately owned forests classed as young in FIPS, will nearly all have been planted after 1985. In the Hen Harrier IAs, forests in the Coillte inventory were classified according to their planting and felling years (pre-thicket if 12 years or less since planting for at least some of the time between 1998-2000, and post-thicket if more than 12 years old during this period), with the exception of stands that were noted in the Coillte inventory as being undeveloped, blown and burnt (these were classified as pre-thicket). Mature forests not in the Coillte inventory were classed as post-thicket, while young forests were classed as pre-thicket.

Table 36. The area of three age-classes of Coillte-owned forest in Hen Harrier IAs, in the two FIPS age categories).

Planting year	Mature FIPS (km²)	Young FIPS (km²)
pre-1985	91252	49709
1985-1995	2278	49052
post-1995	18127	2318

Table 37. Areas of forest and planned forest in the FIPS inventory that are a. in the Coillte estate and b. privately owned.

a.

Forest type	Area	
	ha	%
Mature conifer	29330	30
Young conifer	30167	31
Mature broadleaf and mixed	2907	3
Young broadleaf and mixed	464	0
Cleared	17868	18
PGA	12741	13
Scrub and other forest	52	0

b.

Forest type	Area	
	ha	%
Mature conifer	2894	8
Young conifer	2839	8
Mature broadleaf and mixed	5979	17
Young broadleaf and mixed	481	1
Cleared	3756	11
PGA	17279	50
Scrub and other forest	150	0

Broad categories of open habitats were distinguished using County Landcover Thematic Maps, also owned by the Forest Service, whose resolution was to 25 m. Four land cover types were taken from these data: Bog/heath, Cutaway bog, Dry grassland and Wet grassland.

The final source of geographical data was 1:50000 digitised Ordnance Survey (OS) maps, with a resolution of 4.5 m, which we used to derive seven categories, comprising areas of four elevation classes (0-100m, 101-200m, 201-400m and >400m), three road types (Major roads, National roads and Private roads) and built land.

We calculated the areas of these environmental variables within the relevant data sampling units (i.e. 10 km squares, or fixed radius circles around Hen Harrier nests and randomly located points) e.g.Figure 32. The areas of map representing writing and symbols within a sampling unit were assigned to each of the seven OS-derived elevation, road and built environment classes described above, in proportion to the relative area of each class within the sampling unit. GIS operations were carried out using ESRI ArcView 3.2 GIS. All statistical analyses were performed using SPSS v.11.0.

8.2.2.2 Analysis

8.2.2.2.1 Changes in distribution between Hen Harrier surveys

To determine whether the changes in Hen Harrier distribution between earlier and later surveys were significantly related to environmental variables, we looked only within groups of sampling units (10 km squares or tetrads) where Hen Harriers were present in the earlier survey, or within groups of sampling units where Hen Harriers were absent in the earlier survey. Within these groups, we carried out logistic regression for each environmental variable, using Hen Harrier presence/absence as the dependent variable, and the proportional cover of the environmental variable within the survey unit as the independent variable. For these analyses we used all 17 of the forest cover, land cover and OS-derived variables described above.

8.2.2.2.2 Current distribution of Hen Harriers within IAs

To determine whether the current breeding distribution of Hen Harriers within the nine IAs in the Republic of Ireland was non-random, we plotted all sites where breeding Hen Harriers have been found in the IAs in the last seven years. When Hen Harrier pairs in different years bred within 1 km of each other, only the earliest documented breeding site was retained in the dataset, and sites recorded in later years were discarded. There were two records of two Hen Harrier pairs breeding within 1 km of each other in the same year. In neither instance was the separate identity of each pair a matter of certainty, and so in both cases the better-documented breeding site was retained in the dataset, while the other was discarded. This left 134 Hen Harrier breeding sites. We calculated the total area of each of the 17 environmental variables within a 500 m radius of each of these sites. We then generated 20 sets of 134 points, which we distributed randomly throughout the IAs, constraining their position only so that all the points in a set were at least 1 km apart, and each IA contained the same number of points from each set as it did breeding Hen Harrier sites. For each of these 20 sets, we then calculated the total area of each of the 17 environmental variables within 500 m of each point, as we had done for the set of Hen Harrier breeding sites. If the total area of a given variable within 500m of the nests was either greater than or less than the range of total areas within 500 m of each of the 20 sets of random points, we concluded that Hen Harrier breeding sites were located non-randomly with respect to that variable.

8.2.2.2.3 Separating the influences of young and mature forestry

Forests differ widely in their value to Hen Harriers depending on their age. However, forests are not spread evenly throughout the landscape, but have a clustered distribution, so that most areas with a high cover of pre-thicket forest will tend also to have a high cover of older stages of forest. We generated a single set of randomly located points within the IAs, all of which were more than 1 km from any other point or Hen Harrier breeding site. These points were distributed among the IAs in proportion to the number of Hen Harrier nests in each IA. The highest number of points that could be distributed among the IAs in this way was 670 (five times the number of Hen Harrier breeding sites within the IAs). For each of these random points, and for each of the 134 nest sites, the total areas of pre-thicket forest and post-thicket forest within 500 m were calculated. In order to investigate the influence of pre-thicket forest cover on Hen Harrier distribution, the 804 points were separated into four groups, each of 201 points, on the basis of post-thicket forest cover (each group covered a discrete range of post-thicket forest cover within 500 m). Within each group, we examined the association between Hen Harrier presence/absence and pre-thicket forest cover using logistic regression. To investigate the influence of post-thicket forest cover on Hen Harrier distribution, we divided the 804 points into four groups according to the level of pre-thicket forest cover around the points, and examined the association between Hen Harrier presence/absence and pre-thicket forest cover using logistic regression. We repeated this procedure using areas of pre- and post-thicket forest cover within 1000 m of the points.

8.2.2.2.4 Hen Harrier habitat requirements

We defined suitable habitat as comprising bog/heath, cutover bog, wet grassland and pre-thicket forests, these being the land-use classifications that equate most closely to Hen Harrier nesting and hunting habitat as described in the literature (Gibbons *et al*. 1993; Norriss *et al*. 2002; Watson 1977). For the 670 random points and 134 nest sites in the IAs, we calculated the proportional area of suitable habitat within a radius of 1000m. We allocated each point to one of ten groups, each of which represented a 10 percentile class of suitable habitat area, and calculated the proportion of points in each group that were occupied by Hen Harriers. We then repeated this procedure using adjusted values of pre-thicket forest cover to projected levels in 2015. We compared the proportions of points occupied by Hen Harriers in low availability of suitable habitat with the proportion occupied in high availability of suitable habitat, at three different scales (500 m, 1 km and 2 km radius), and using different cut-off points to define low and high availability of suitable habitat. In this way, we determined the threshold of suitable habitat availability that best distinguishes between points occupied by Hen Harriers and points without Hen Harriers. We also estimated the change in Hen Harrier population that could be expected in 10 years due to maturation of the forest estate, assuming that the loss or gain of a given proportion of suitable habitat will result in an equivalent change in the Hen Harrier population.

8.2.3 Results

8.2.3.1 Changes in distribution between Hen Harrier surveys

Squares where Hen Harriers were present in both the Old Atlas and New Atlas surveys had less land under 200 m, less dry grassland, and more of all four categories of forest (mature, young, PGA and Clearfell) than squares which lost Hen Harriers between these two surveys (Table 38). Squares where Hen Harriers were present in the New Atlas and the 1998-2003 surveys had less built land, more land between 200-400 m, and more land in each of the four forest categories than squares which lost Hen Harriers between these two surveys (Table 39).

8.2.3.2 Current distribution of Hen Harriers within IAs

Comparing the area of different landcover types within 500 m of Hen Harrier nests found between 1998 and 2003 in the Hen Harrier IAs with the equivalent values for randomly distributed points in these areas (Table 40) shows that the Hen Harrier nests were located non-randomly within the IAs with respect to a number of environmental variables. Areas of mature conifer, young conifer, clearfell and PGAs within 500 m of Hen Harrier nest sites were higher than expected from random chance, while areas of mature broadleaf were lower. Areas of bog/heath and cutover bog were higher and areas of both wet and dry grassland lower than expected. The length (as estimated by area) of all sizes of road within 500 m of Hen Harrier nest sites was less than expected. Nests were also located non-randomly with respect to elevation, there being a greater area of land between 200 and 400 m and a lower area of land less than 200 m than would be expected if nests were randomly distributed.

In order to separate the effect of dry grassland cover from the effect of altitude (there is a strong negative relationship between the two variables), we compared the cover of dry grassland around the Hen Harrier nests with that around the randomly distributed points within three different elevation categories. Dry grassland cover around Hen Harrier nests was significantly lower than expected by random chance at all altitudes (Table 41).

Table 38. The percentage cover of different landcover types in the 192 10 km squares where Hen Harriers were seen during the Old Atlas Survey, grouped according to whether Hen Harriers were (n=60) or were not (n=132) seen during the New Atlas Survey, and the results of logistic regressions relating the probability of Hen Harrier occurrence to each environmental variable. For regressions that are statistically significant the logit coefficient (B) is given, along with its standard error, and Nagelkerke r2 (this value expresses the proportion of the variance in Hen Harrier occurrence accounted for).

	HH present		HH absent							Nagelkerke
Landcover	Mean	n	Mean	n	Wald	d.f.	Sig	B	S.E.	r^2
<200m	52.4	50	61.6	116	4.54	1	0.033	-1.40	0.65	0.038
200-400m	23.0	50	11.8	116	11.57	1	0.001	3.11	0.92	0.100
>400m	1.4	50	2.7	116	1.23	1	0.268			
Tracks	1.4	50	1.5	116	0.24	1	0.628			
National roads	0.9	50	0.8	116	0.54	1	0.462			
Major roads	0.4	50	0.4	116	1.04	1	0.308			
Mature forest	7.1	50	4.6	128	8.42	1	0.004	9.62	3.32	0.068
Young forest	12.9	50	5.4	128	26.53	1	0.000	15.07	2.30	0.269
Clearfell	3.0	50	1.7	128	7.33	1	0.007	16.16	5.97	0.060
PGA	3.4	50	1.3	128	20.58	1	0.000	49.09	10.82	0.226
Bog/heath	10.9	50	8.2	106	2.16	1	0.141			
Cutover bog	2.2	50	4.1	106	2.61	1	0.106			
Wet grassland	9.2	50	7.5	106	1.24	1	0.265			
Dry grassland	58.2	50	66.8	106	4.03	1	0.045	-1.39	0.69	0.036
Built land	0.7	50	1.2	106	0.93	1	0.334			

Table 39. The percentage cover of different landcover types in the 123 10km squares where Hen Harriers were seen during the New Atlas Survey, grouped according to whether breeding Hen Harriers were (n=33) or were not (n=90) found in the period 1998-2003, and the results of logistic regressions logistic regressions relating the probability of Hen Harrier occurrence to each environmental variable. For variables that are statistically significant predictors of Hen Harrier presence, the logit coefficient (B) is given, along with its standard error, and Nagelkerke r2 (this value expresses approximately the proportion of variation in Hen Harrier occurrence accounted for by the variable in question).

	HH present		HH absent							Nagelkerke
Landcover	Mean	n	Mean	n	Wald	d.f.	Sig	B	S.E.	r2
<200m	54.1	33	57.4	74	0.47	1	0.494			
200-400m	24.1	33	12.0	74	7.60	1	0.006	3.19	1.16	0.113
>400m	1.2	33	1.5	74	0.10	1	0.748			
Tracks	1.4	33	1.4	74	0.12	1	0.728			
National roads	0.9	33	0.8	74	1.02	1	0.314			
Major roads	0.4	33	0.4	74	0.01	1	0.907			
Mature forest	8.1	33	4.2	86	10.36	1	0.001	16.25	5.05	0.175
Young forest	15.4	33	8.2	86	11.59	1	0.001	10.19	2.99	0.197
Clearfell	4.3	33	1.2	86	15.65	1	0.000	50.69	12.81	0.336
PGA	3.8	33	2.4	86	4.42	1	0.035	17.40	8.27	0.073
Bog/heath	13.4	33	12.5	66	0.09	1	0.767			
Cutover bog	2.1	33	2.3	66	0.03	1	0.852			
Wet grassland	11.2	33	10.4	66	0.15	1	0.699			
Dry grassland	52.0	33	58.7	66	1.51	1	0.219			

Built land	0.2	32	1.1	62	2.89	1	0.089	-98.58	58.03	0.11

Table 40. The total area (ha) of different landcover types within 500m of 134 Hen Harrier nests found between 1998 and 2003 in the Hen Harrier IAs in the Republic of Ireland, and the corresponding maximum and minimum values for 20 sets of points distributed randomly throughout these IAs. The final column in this table indicates, for each landcover type, whether the value for the Hen Harrier nest sites was outside the range of values found for the 20 sets of points, and thus significantly different from random.

Landcover type	500m from nests	Random iteration range	Difference ($p<0.05$)
←200m	2886	3686 - 5044	Less
200-400m	5412	3687 - 4680	Greater
400-500m	268	32 - 369	-
Tracks	79	82 - 117	Less
National roads	92	114 - 161	Less
Major roads	10	14 - 44	Less
Mature conifers	1900	815 - 1347	Greater
Mature broadleaves	9	21 - 82	Less
Young conifers	1938	747 - 1371	Greater
Young broadleaves	2	0 - 22	-
Clearfell	1452	476 - 876	Greater
PGA	908	321 - 753	Greater
Bog/heath	2731	1569 - 2408	Greater
Cutover bog	719	266 - 555	Greater
Wet grass	1122	1181 - 1546	Less
Dry grass	1435	3414 - 4790	Less
Built land	4	1 - 60	-

Table 41. The percentage cover of dry grassland in three different elevation categories within 500m of 134 Hen Harrier nests found between 1998 and 2003 in the Hen Harrier IAs in the Republic of Ireland, and the corresponding maximum and minimum values for 20 sets of points distributed randomly throughout these IAs. The final column in this table indicates, whether the percentage cover of dry grassland for the Hen Harrier nest sites was outside the range of values found for the random points.

Elevation	% Dry grass 500m from nests	Range of % dry grass in random iterations	Difference ($p<0.05$)
0-100m	20	39-66	Less
100-200m	14	37-53	Less
200-400m	12	23-35	Less

8.2.3.3 Separating the influences of young and mature forest

Cover of pre-thicket forest within both 500m and 1000m is strongly and positively related to Hen Harrier occurrence, when controlling for variation in post-thicket forest cover (Table 42). The relationship between post-thicket forest cover and Hen Harrier occurrence is not as robust, persisting at intermediate levels of pre-thicket forest cover, but weakening or disappearing altogether when pre-thicket cover is either low or high (Table 43). No points with greater than 82% of post-thicket forest cover within 500m, or 68% post-thicket cover within 1000m, were occupied by Hen Harriers.

Table 42. The results of logistic regression of Hen Harrier occurrence on area of pre-thicket cover within a) 500m and b) 1000m, for four groups defined by the level of post-thicket forest cover within 500m of these points. For variables that are statistically significant predictors of Hen Harrier presence, the logit coefficient (B) is given, along with its standard error.

a)

Post-thicket	Mean pre-thicket cover (n) HH present	HH absent	Wald	d.f.	Sig	B (10^6)	S.E. (10^6)
0%	28% (17)	9% (184)	14.19	1	0.000	5.49	1.46
0-3%	38% (21)	14% (180)	16.81	1	0.000	4.27	1.04
3-21%	50% (40)	23% (161)	28.77	1	0.000	5.36	1.00
21-97%	30% (56)	24% (145)	4.11	1	0.043	2.08	1.03

b)

Post-thicket	Mean pre-thicket cover (n) HH present	HH absent	Wald	d.f.	Sig	B (10^6)	S.E. (10^6)
0-1%	25% (14)	8% (187)	17.63	1	0.000	2.56	0.61
1-8%	31% (17)	15% (184)	12.80	1	0.000	1.43	0.40
8-21%	36% (44)	22% (157)	20.82	1	0.000	1.55	0.34
21%-90%	31% (59)	26% (142)	4.18	1	0.041	6.42	0.31

Table 43. The results of logistic regression of Hen Harrier occurrence on area of post-thicket cover within a)500m and b) 1000m, for four groups defined by the level of pre-thicket forest cover within 500m of these points. For variables that are statistically significant predictors of Hen Harrier presence, the logit coefficient (B) is given, along with its standard error.

a)

Pre-thicket	Mean post-thicket cover (n) HH present	HH absent	Wald	d.f.	Sig	B (106)	S.E. (106)
0%	10% (8)	3% (193)	1.97	1	0.160	3.46	2.47
0-12%	26% (23)	15% (178)	3.38	1	0.066	1.70	0.93
12-32%	32% (33)	15% (168)	15.11	1	0.000	4.26	0.12
32-99%	17% (70)	16% (131)	0.03	1	0.855	2.07	0.11

b)

Pre-thicket	Mean post-thicket cover (n) HH present	HH absent	Wald	d.f.	Sig	B (10^{-6})	S.E. (10^{-6})
0-6%	29% (1)	8% (200)	0.26	1	0.620	-3.32	6.70
6-16%	35% (32)	15% (169)	3.74	1	0.053	6.22	0.32
16-29%	25% (38)	16% (163)	13.73	1	0.000	1.35	0.37
29-78%	13% (63)	10% (138)	2.28	1	0.131	0.50	0.33

8.2.3.4 Hen Harrier habitat requirements

The difference in Hen Harrier occupancy between points with low and high availability of suitable habitat is maximised when the cut-off point (separating points with low and high proportions of suitable habitat around them) is 30%, and when suitable habitat availability is calculated for the area within a 1000m radius of each point (Figure 33).

Points were distributed quite evenly between the 10 percentile classes of suitable habitat within 1km, except for points in the top two classes (80-90% and 90-100%), which were much less numerous than in classes lower than 80% (Figure 34). No points with less than 20% suitable habitat cover, and only a small proportion (6%) of points with 20-30% habitat cover, were occupied by Hen Harriers (Figure 34). Among points with greater than 30% suitable habitat cover, the percentage of points occupied by Hen Harriers ranged between 15% and 31%; and was positively correlated with suitable habitat cover (Pearson's r=0.82, n=7, p=0.02).

Maturation and harvesting of the forest estate will by 2015 have resulted in substantial changes in the frequency distribution of points within the 10 percentile classes of suitable habitat from the 1999 distribution (Figure 35). Points with more than 60% suitable habitat within 1km will decline to almost a quarter of their numbers in 1999, from 265 to 71. This decrease will be mirrored by an increase in the number of points with less than 40% suitable habitat within 1km, from 337 to 557. If the same proportion of points in each 10 percentile class are occupied by Hen Harriers as during the period 1997-1999 (Figure 34), there will be fewer points with enough suitable habitat to be occupied by Hen Harriers. This shift may result in an overall decrease in the carrying capacity of the IAs (Figure 36), equivalent to a decrease in the number of points occupied by Hen Harriers by approximately 30%.

8.2.4 Discussion

8.2.4.1 Hen Harriers and agriculture

The changes in Hen Harrier distribution over the past 40 years indicate that the species has moved away from lowland areas, where disturbance has increased and agricultural intensification has reduced the availability of Hen Harrier habitat, into areas at higher elevation, where new plantations have provided an abundance of suitable habitat for them. A similar pattern is apparent in the Hen Harrier's current distribution within the IAs, where they appear to strongly avoid dry grassland and areas at low elevations, both of which are closely associated with improved agricultural land. These variables are not independent of one another; dry grassland cover dominating at low altitudes, but giving way in upland areas to bog and forest habitats. A possible reason for the relatively low cover of dry grassland around Hen Harrier nest sites would therefore have been that Hen Harriers were selecting for vegetation types associated with higher elevations, rather than avoiding land that had been improved for agriculture. However, in three different elevation categories, percentage dry grassland cover around Hen Harrier nests was significantly lower than expected by random chance (Table 41). This indicates that the Hen Harriers avoid dry grassland at a range of altitudes. Agricultural intensification therefore has the potential to reduce the carrying capacity of land for Hen Harriers at high as well as low elevations.

The 1998-2003 survey targeted areas known to hold extant populations of Hen Harriers, as well as a random selection of areas containing suitable habitat but not known to hold breeding Hen Harriers. Within these areas, surveyors concentrated their time and effort on the habitat that looked best for Hen Harriers (Dúchas 1998-2003, unpublished data). This could have led to a bias in the results of the survey, whereby Hen Harriers occupying habitats perceived to be less favourable for them would be detected less efficiently than Hen Harriers in more traditional habitats. If such a bias were strong enough, it could result in the pattern observed in this study, and the false conclusion that Hen Harriers avoid areas of intense agriculture. However, Norriss et al. (2002) claim the vast majority of Hen Harriers breeding in the Republic of Ireland were detected by the 1998-2003 survey, in which case if such a bias existed, it would apply only to a small number of Hen Harriers. Such a small bias would not be sufficient to generate the relationships between Hen Harrier distribution and habitat described here. Furthermore, all of the improved agricultural land within the IAs is situated within 10km of areas where Hen Harriers were found during the 1998-2003 survey. Hen Harriers breeding on improved agricultural land in the IAs were therefore more likely to be found than those breeding in similar habitat elsewhere in Ireland. It is therefore likely that agricultural intensification has a real and pronounced negative effect on the value of land to Hen Harriers.

To maintain the populations of Hen Harriers within the IAs at present levels, further agricultural intensification within these areas should be minimised. In a recent review of the status of Hen Harriers in Ireland, Maclochlainn (2003) wrote "the belt of… marginal land above the cultivation line but below the heather tops is diminishing inexorably" due to the replacement of rough grassland and other semi-natural habitat with re-seeded grassland for improved grazing. However, a recent statement made by Dúchas maintained that existing farming practices are almost certain to be fully compatible with the conservation requirements of Hen Harriers, and that there will consequently be no need to impose restrictions on existing farming activity (Canny 2003). If farming activity is taken to include the ongoing intensification of rough and marginal agricultural habitats, then this assumption may need to be re-examined. The new single premium system, which will result in a decoupling of stocking from agricultural scheme payments, will be introduced to Ireland in early 2005, and may result in an increase in the amount of agricultural land that is suitable for Hen Harrier. It will almost certainly lead to lead to an overall decrease in grazing pressure, which might result in the 'roughening' of grassland areas, to the benefit of the Hen Harrier. However, in other areas, small farms may be amalgamated into larger holdings in order to improve their efficiency, accompanied by agricultural intensification.

8.2.4.2 Hen Harriers and forestry

The strength of the relationship between Hen Harrier occurrence and pre-thicket forest cover at all levels of post-thicket forest cover indicates that young forests are selected for by the species. The relationship between post-thicket cover and Hen Harrier occurrence, when variation in pre-thicket cover is accounted for, is contrastingly weak, especially when pre-thicket cover is either very high or nearly absent. This is consistent with the conclusion that the positive association between mature forest cover and Hen Harrier occurrence (found in the analyses of Hen Harrier distribution changes; and in the comparison of the habitat around Hen Harrier nest sites with the habitat around sets of random points – see Table 38, Table 39 and Table 40) is due in large part to the proximity in the landscape of old and pre-thicket forest. However, post-thicket forest cover is a predictor of Hen Harrier occurrence at low to intermediate levels of pre-thicket forest cover. This may be because, prior to the mid 1990s, new forests were nearly all established in the uplands, typically on unenclosed areas of bog and rough pasture (Fahy & Foley 2002). In contrast, recent planting has typically taken place on relatively improved agricultural land, in landscapes that are unsuitable for Hen Harrier. Very few areas with Hen Harrier will have no young forests at all; and (at least in the IAs) most areas where young forest cover is abundant will be suitable for Hen Harriers. However, among areas that have low levels of pre-thicket forest cover, upland areas (i.e. those areas containing the majority of good Hen Harrier habitat) are likely to have higher levels of post-thicket forest cover than lowland areas, where agricultural activity is more intensive. Thus, where pre-thicket forest cover is low, post-thicket cover could be positively related to Hen Harrier occupancy.

A limitation of this study is that although we were able to distinguish broadly between habitats that have some value to Hen Harrier and others that do not, it was not possible for us to distinguish low quality habitats (where Hen Harriers forage and nest with relatively little success) from high quality habitats (where Hen Harriers enjoy high levels of hunting and breeding success). This is partly because the resolution of the habitat data we used was quite coarse, but by far the biggest obstacle to determining habitat quality is our lack of knowledge about the value of different habitat types to Hen Harriers. This lack of knowledge is particularly critical in relation to the quality of second-rotation forests. While current indications are that young second-rotation forests are being used by Hen Harriers for both nesting and foraging, we have insufficient data to judge the value of this habitat in relation to either young first rotation forestry or open habitats such bog and wet grassland. The availability of second rotation forestry will increase greatly over the next few decades, during which time the persistence of Hen Harriers in many heavily forested areas may hinge on the value of young second rotation forestry to this species.

Our estimate of suitable habitat cover in 2015 does not take account of any of the afforestation that will have occurred between 1999 and 2015. Despite the recent move of afforestation in Ireland away from the most marginal lands for agriculture (Fahy & Foley 2002), the majority of land currently put forward by farmers for afforestation is still relatively unproductive from an agricultural perspective, and could potentially be used by Hen Harriers for foraging. Unless financial incentives are put in place to encourage the establishment of new forests on high quality pasture land, it is likely that the majority of afforestation will continue to occur on marginal agricultural land. If this is the case, then new plantations will not result in the creation of substantial areas of entirely new Hen Harrier habitat, as many of these marginal areas will have been used by Hen Harriers before planting. Therefore, while the area of habitat suitable for Hen Harriers in the IAs will be influenced by forest maturation, and by the felling and replanting of mature forest stands, it is unlikely to be greatly increased by the afforestation of previously open habitats.

The distribution of Hen Harriers within the IAs in relation to percentile classes of suitable habitat indicates that areas with less than 30% cover of bog, rough pasture or young forest are avoided by Hen Harriers. Due to the maturation of the forest estate, this threshold will be exceeded by a far larger proportion of the IAs in 2015 than at present, with the likely consequence that the carrying capacity of these areas for Hen Harriers will decrease. In predicting that this decrease will be in the region of 30% we assume that the populations of Hen Harriers in the IAs are currently at carrying capacity. We also assume that different areas of suitable habitat are comparable in quality (i.e. their ability to support Hen Harriers); especially between habitat types that will contribute different proportions of the total area of suitable habitat in 2015 than they did in 1999 (e.g. young first rotation forest and young second rotation forest). If these assumptions are incorrect, then the loss or gain of a given proportion of suitable habitat will not necessarily result in a change of equal magnitude in the Hen Harrier population.

If the value to Hen Harriers of new forests planted between 1999 and 2015 greatly exceeds that of the habitats they replace, the carrying capacity of the IAs may, at least during the period under consideration, be less affected by the maturation of the forest estate than we predict. However, such a 'buffering' effect of afforestation would be temporary at best, as Hen Harriers can only use a piece of land for a third of the time after it has been planted with trees, and, as we have discussed, the value of second-rotation forestry in relation to other suitable habitats is not known. If second-rotation forestry is inferior to first-rotation forest and open habitats, then the maturation of the forest estate may result in an even greater impact on Hen Harriers than we predict. Likewise, if land covered by first rotation pre-thicket forests during the 1998-2003 survey was more valuable to Hen Harriers than some or all of the open components of suitable habitat (i.e. bog and wet grassland), then the canopy closure of these forests could decrease the carrying capacity of the IAs by a larger extent than we predict.

8.2.5 Recommendations

1. Afforestation and agricultural improvement should be regulated in the IAs, to minimise further decreases in the carrying capacity of these areas for Hen Harriers. Wherever possible, afforestation should target improved agricultural land in the IAs, rather than bog, rough pasture, and semi-natural habitats.
2. The findings of this study suggest that 3 km^2 may be an appropriate scale at which to evaluate habitat composition within the IAs, as there is a clear association between Hen Harrier occupancy and habitat composition within a radius of 1 km. If a proposed change in land use would decrease the proportion of any 3 km^2 area of land in the IAs to below 30% (below which threshold Hen Harrier occupancy is substantially lower than at higher levels of suitable habitat cover), it should be regarded as being potentially damaging to Hen Harriers.
3. Where Hen Harriers occupy heavily afforested areas a mosaic of different age classes should be developed, so that forests within any 3 km^2 area are composed of close to one third pre-canopy closure forest at any one time. In areas of continuous forest, blocks of greater than 100ha that are composed stands within 14 years of each other should be avoided. Such large, contiguous areas of similarly-aged forest could reduce the value of the surrounding landscape to Hen Harriers by reducing the overall availability of suitable habitat within 1 km to below 30%. This threshold assumes that Hen Harriers will continue to breed in areas of extensive forest cover if sufficient young second rotation forest is available.

4. The development of a custom-designed GIS would help to facilitate such a decision-making process. This would allow the effects of a proposed change in land use on the proportion of suitable habitat cover in the surrounding area to be easily evaluated in the context of existing land uses. The GIS would also enable landscape change to be predicted, allowing proposals to be evaluated in the light of future impacts on suitable habitat available to Hen Harriers. In conjunction with data from future Hen Harrier surveys, it would be used to test and refine the predictions of the model.
5. More detailed habitat data should be collected from the IAs. In particular, a detailed inventory of all forests (both private and Coillte-owned) in the IAs, to include planting species, planting year and projected felling year, should by compiled and kept up to date. Such habitat data would be essential in implementing the recommendations presented here; and would greatly facilitate further research on the habitat requirements of Hen Harriers. They would also enable validation and/or refinement of the associations between Hen Harriers and land use described here, and possible refinement of the recommendations.
6. Our understanding of Hen Harrier habitat requirements also needs to be improved, through combined satellite- or radio-tracking study of foraging adults, and monitoring of the fledging success of Hen Harrier nests in different habitat configurations.
7. Although preliminary indications are that Hen Harriers will use second-rotation forests for both hunting and foraging, we need to acquire a better understanding of the value of young second-rotation forest for breeding Hen Harriers before we can be certain that forest habitats will continue to provide suitable habitat for Hen Harriers in the long term.

A combined limit of 70% should therefore apply to improved agricultural land and plantation forestry, when considering proposals to convert an area of bog or rough pasture to either of these land cover types in Hen Harrier IAs.

9 EXPERIMENTAL MANIPULATION OF OPEN SPACE

9.1 INTRODUCTION

The main objective of this part of Project 3.1.3 was to investigate the effect of open space configuration on the biodiversity of Irish forestry plantations using experimental manipulations. The results will inform forestry management theory and practice on some options available to enhance biodiversity through the arrangement of open space in and around a forest. Open spaces in typical forest plantations can be broadly divided into four main types: forest roads, rides, discrete open spaces (glades) and plantation edges. In selecting which of these components to investigate, the factors considered included the potential effect of the open space components on biodiversity, the potential for normal forestry practices to incorporate manipulation of these components and the practicalities of manipulating these components within the resources available to the BIOFOREST project.

9.1.1 Rationale for the specific focus of the manipulation experiment.

Preliminary analysis of data from work carried out on the BIOFOREST Project in existing forest plantations during 2001-2003 suggested that strips of open spaces adjacent to forest roads and plantation edges can make a significant contribution to the biodiversity of forestry plantations. The extent of this contribution is partly dependent on the width of these unplanted strips. Forest roads are likely to be more significant than plantation edges, because they can provide open space habitat in the middle of large forest blocks. Plantation edges are adjacent to unplanted areas, but do have the potential for the development of scrub habitat that is generally lacking in unplanted farmland. The BIOFOREST Research Group considered that the width of unplanted road margins would be more amenable to change than the configuration of non-linear open space within forests, without major disruptions to standard forestry procedures. It was anticipated that it might be easier to plan, establish, manage and eventually survey linear open spaces along roads and edges than specific arrangements of discrete open space. Rides are generally established along compartment boundaries or the boundaries of other management units. Therefore, felling and planting schedules will often be different on either side of a ride, thus making establishment of manipulations problematic. The review of open space literature (Section 2) also informed the Research Group and indicated that roads would be a good focus for the experiment. In order to further examine the possibilities of using roads as a focus, the options on the nature of the manipulations were also considered.

9.1.2 Development of research design

Manipulation of the clearance between the trees on either side of forest roads would affect the space available for open habitats to develop. The recommended between-trunk clearance across the road is currently 15m, with approximately 5m being the road surface and the other 10m being divided between the two sides of the road, leaving an average of 5m on each side (Ryan *et al.* 2004). Branches tend to directly shade at least 2.5m of this, and an amount of the space is also used for positioning of drains and banks. Together with the shade from the maturing trees, there is little undisturbed open space on either side that is unshaded. The Research Group proposed to investigate the effect of doubling the clearance on the biodiversity of the area.

An important factor to take into consideration in the experimental design was that data collection would be restricted to a single field season. Data collected on the different taxa would serve as a baseline for comparison with later surveys of the sites, after the lapse of certain time periods. Deciding that the sites would be owned by the State forestry agency, Coillte, ensured the best chance of (a) "earmarking" certain forests for the experiment and (b) maintaining the treatments into the future. As Coillte is not currently carrying out much afforestation on its lands, the experiment had to be established at the planting stage of second rotation forest.

This design was guided in part by a discussion session at the conference "Opportunities for enhancement of biodiversity in plantation forests", 24 October 2002, Vienna Woods Hotel, Cork. This was attended by members of the BIOFOREST Steering Group and individuals from forest-related institutions both inside and outside of Ireland, who had useful advice about the practicalities of various different options. The design proposed (above) was formally agreed upon by the Steering Group in December 2003.

9.2 EXPERIMENTAL ROAD WIDTH MANIPULATIONS

9.2.1 Study Design

Eight manipulation sites were established in the winter of 2003/2004. Two are located in Wicklow (Bawnogue and Ballingate), two in Laois (Fossy Hill and Cardtown), two in Waterford (Lismore and Tooranaraheen) and two in Cork (Cloontycarthy and Carrigagulla). Note that planting in Carrigagulla took place in the winter of 2004/2005. Each site contains two sections of forest road that, as far as is possible, differ only in the width of unplanted land at the side of the road. Each site contains a "normal" treatment, and a "wide" treatment. We have specified the forest road widths for these treatments, based upon measurements taken during our extensive survey of forest roads in 2003, from discussions with foresters, and with reference to the draft *Forest Road Manual* (see Figure 38).

- The "standard" treatment represents normal forestry practice. In this treatment, the total width of the forest road gap is 15 m (see Section 9.1.2).
- The "wide" treatment represents a modification of normal forestry practice for biodiversity enhancement. In this treatment, the total width of the forest road gap was 30 m, equivalent to an unshaded strip on each side of the road of 10 m, using the assumptions in Section 9.1.2).

The length of each treatment is usually at least 200m, and longer than this where possible, in order to accommodate bird surveys.

9.2.2 Baseline survey

Baseline surveys of the manipulation sites were carried out in the summer of 2005. At this stage, the planted trees were too small to influence the biodiversity of the road verges. The road verges have had one full growing season to recover from disturbance occurring during the planting operations.

The objective of the baseline survey was be to detect any differences in biodiversity between the two treatments in each site. Any differences will reflect underlying site differences between the treatments and will inform interpretation of future monitoring of the treatments.

9.2.2.1 Vegetation

9.2.2.1.1 Sampling Design

Because of the scale of roadside features and for comparison with the extensive survey, a plot size of 4 m^2 was used. A partial transect approach was used in which plots were arranged from the road edge and into the forest at each sampling point. There were three sampling points in each treatment, for a total of six per site and 48 for the entire baseline survey. These sampling points were also used for pitfall trapping for spiders. Three sampling points were located along each road section at approximately 50 m, 100 m and 150 m; however, the distances were adjusted where local conditions did not facilitate plant or spider sampling (e.g. heavy brash) or where the road length was greater or less than 200 m.

These points were be placed on the north, northeast and east sides (i.e. south, southwest and west facing sides) of the roads, as these will intercept more afternoon sunlight when the trees mature.

One open plot and one forest interior plot were surveyed, each paired with (adjacent to) the appropriate spider pitfall trap plot (see Section 6.2). The forest interior plot was 2 m from the forest edge defined by the tree stems, towards the interior of the forest. The open plot was 3 m from the forest edge, towards the road.

In addition, an unpaired vegetation plot was recorded immediately beside the gravel road surface ("verge plot"). Where possible, this was also 2 × 2 m, but where this size plot did not fit, the plot size was changed to 1 × 4 m (at the same time retaining the 4 m^2 plot area). No plots were recorded on nearly vertical, largely unvegetated banks where they are present, but species occurring on banks were noted in the species list (see below).

The location of sampling points was recorded with a GPS and the centre of each plot was permanently marked with a short length of PVC pipe. In addition, the bottom left and upper right corners were marked with a large nail and washer driven into the ground so that plots may be refound using a metal detector if the PVC marker is disturbed. Bottom left and upper right was defined as seen from the road, facing into the forest.

9.2.2.1.2 Recording

Site Data

At each sampling point, the following environmental and management information was recorded:

- Site variables:
    - Site and treatment type
    - Site slope (°) (slope in direction of topographic site aspect)
    - Aspect (°)
    - Soil type (determined by observation in field)
    - Road orientation (°)
    - Stone type used for road surface
- Frequency and intensity of natural or human activities, including land management:
    - Grazing (e.g. amount of dung; evidence of grazing, trampling or damage to ground vegetation and shrub layer)- described and ranked from 0 to 3.
    - Road management (e.g. drain maintenance, resurfacing, etc.)
    - Recreational use (e.g. pathways present; spent cartridges; signs of rubbish etc.)- described and ranked in intensity from 0 to 3.
    - Vegetation and Other site management, particularly burning, turf-cutting
- Silvicultural variables:
    - Height of edge trees (to nearest 0.1 m)
    - Method of ground preparation for planting of adjoining areas
    - Open space width (m) from tree to tree at the sampling point (to calculate sampling area for species list below)

Five soil subsamples were collected from the corners and centre of each plot and bulked in the field. These were collected to a depth of 5 cm using a trowel. Litter layers were not sampled where possible, however, there were cases where the top 10 cm or more of "soil" was a highly disturbed mixture of litter, well-humified organic matter and mineral soil. Soil pH was determined for the bulked, field-moist samples at the earliest opportunity. Samples were retained, air-dried and stockpiled for possible future chemical analysis.

A photo was taken from the road verge opposite each sampling point.

Plot data:

The following were recorded for each 4 m^2 plot:

- Distance to forest edge (m) (as defined by the nearest tree stem)
- A list of species with percentage cover estimated to the nearest 5%. Below 5% two different cover-abundance units was distinguished: 3% (indicating cover of 1-5%), and 0.5% (indicating cover <1%). Numbers of individuals may be counted if the species in question is rare or otherwise significant.
- Number and species of tree saplings ≥ 0.25 m tall
- Height and percent cover of vegetation strata:
    - o height to lowest live branches of planted conifers
    - o small tree/large shrub stratum (woody veg 2-5 m tall), if any
    - o sapling/small shrub stratum (woody veg < 2 m tall, including subshrubs and planted conifers)
    - o brambles/briars (*Rubus* and *Rosa*)
    - o forbs (vascular, broadleaved herbs)
    - o graminoids (grasses, sedges, rushes)
    - o bryophytes
    - o lichens
- Percent cover of other ground cover types:
    - o bare soil
    - o bare rock
    - o standing water
    - o leaf litter (non-conifer)
    - o (conifer) needle litter
    - o fine woody debris (< 7 cm diameter)
    - o coarse woody debris (≥ 7 cm diameter), including stumps
    - o live tree stems and roots
- Soil drainage on a five-point scale:
    - o very poor (e.g. very wet peats or standing water present)
    - o poor (e.g. wet gleys)
    - o damp
    - o mesic (ideal soils for cultivation)
    - o dry (very well-drained, e.g. dry road banks)
- Presence/absence of microtopography features in the below categories:
    - o ditch
    - o hollow
    - o flat
    - o hummock
    - o soil mound (from ground prep for affor)
    - o stump
    - o bank
- Microtopographical heterogeneity on a 3-point scale (1 = fairly uniform, 2 = heterogeneous, 3 = very heterogeneous)
- Plot dimensions (2x2 m or 1x4 m)
- Plot type (verge, ditch, open, forest)
- Matching spider pitfall trap number (O1, O2, F)

All vascular plant species were recorded. For bryophyte and lichen recording, only species forming patches on soil, rock, litter or woody debris more than 5cm^2, i.e. 2.2cm × 2.2cm, were recorded. Algae and fungi were not recorded.

In addition, a supplementary species list was compiled, where the presence of species not present in plots was noted. Recording took place 10 m along the road on either side of the sampling point. Species occurring between the forest edges on both sides of the road and on the road surface were recorded. All vascular plants were recorded. For terrestrial and dead-wood inhabiting bryophytes and lichens, only species forming patches more than 50cm² (7.1 × 7.1 cm) were recorded. Algae and fungi were not recorded. Species associations with unusual microhabitats, such as the road surface or wet drains, were also noted.

Each plot was identified by a unique alphanumeric code indicating site (4 letters), treatment (S or W), sampling point (1-3) and plot (a, b, c). Plot lettering began at the road verge and extended into the forest, such that "a" indicates the verge plot, "b" the open plot and "c" the forest interior plot. For example, FOSS S3b indicates the second (open) plot from the roadside at the third sampling point in the standard treatment at the Fossy Hill site.

9.2.2.2 Spiders

This survey was carried out in 8 sites of reforested Sitka spruce (*Picea sitchensis*) stands. Within each site two experimental road widths were established: 15m (standard treatment) and 30m (wide treatment). Each road treatment was approximately 200m in length.

9.2.2.2.1 Sampling Protocol

Within each site six sampling plots were established, three within each road width treatment (standard treatment plots coded S1-S3; wide treatment plots coded W1-W3. The plots were located approximately at 50m, 100m and 150m along the manipulation area in areas with typical vegetation and which are relatively homogenous for that particular area of road edge. The plots were all located on the south facing side (or southwest/west where south facing was not possible).

The spiders were sampled using pitfall traps. These consisted of a plastic cup, 7cm in diameter by 9cm depth. Each trap had several drainage slits pierced approximately 2cm from the top of the cup and was filled with antifreeze (ethylene glycol) to a depth of 1cm to act a killing and preserving agent. The traps were placed in holes dug with a bulb corer so that the rim was flush with the ground surface.

Each plot comprised of nine pitfall traps arranged in three sampling points. Sample points consisted of three pitfall traps set two metres apart which were arranged parallel to the forest edge (tree bases). Two of these traps were used in the analysis with the third to be used only if traps were lost due to flooding or animal damage. Within the standard treatment (15m) two sample points were established midway between the road edge and the forest edge, approximately 2-4m apart and were coded Open 1 (O1) and Open 2 (O2). The third sampling point was located within the forest (5m from the forest edge) and was coded Forest (F). Within the wide treatment (30m) sampling point Open 1 was established midway between the road edge and the forest edge, sampling point Open 2 was established three metres into the open space from the forest edge. The forest sampling point location follows the method used for the standard treatment. This arrangement of traps was used so that the data is comparable with the 3.1.3 extensive survey.

9.2.2.2.2 Environmental variables

The percentage cover of vegetation was recorded in a 1m2 quadrat surrounding two of the pitfall traps in each sample point on the transect in the following structural layers: ground vegetation (0-10cm); lower field layer (>10cm - 50cm) and upper field layer (>50cm – 200cm). Cover of other features such as deadwood and litter were also recorded using this scale and litter depth was measured within each quadrat. All cover values were estimated using the Braun-Blanquet scale (Mueller-Dombois & Ellenberg 1974), which involves giving numerical rankings to a range of percentages (+ = <1% cover; 1 = 1 - 5%; 2 = 6 - 25%; 3 = 26 - 50%; 4 = 51 - 75%; 5 = 76 - 100%). This follows the protocol used in the 3.1.3 extensive survey.

9.2.2.2.3 Fieldwork schedule

Fieldwork was undertaken between May-July 2005 and traps were changed 3 times during this time period, approximately every 3 weeks (Table 44). The environmental variables were measured during change 1. For logistical reasons fieldwork began one week earlier than the 2003, 3.1.3 extensive survey. However as spiders are most active and abundant from the beginning of May-July this should not affect comparisons with the 3.1.3 extensive survey.

Table 44. Schedule of pitfall trap changes

	Start date	Change 1	Change 2	Change 3 - End
Ballingate	10/05/05	30/05/05	17/06/05	12/07/05
Bawnogue	10/05/05	30/05/05	17/06/05	12/07/05
Carrigagula	12/05/05	01/06/05	20/06/05	14/07/05
Cardtown	09/05/05	30/05/05	16/06/05	11/07/05
Clootycarthy	12/05/05	01/06/05	20/06/05	14/07/05
Fossyhill	09/05/05	30/05/05	16/06/05	11/07/05
Lismore	11/05/05	02/06/05	21/06/05	13/07/05
Tooranaraheen	11/05/05	02/06/05	21/06/05	13/07/05

9.2.2.2.4 Species identification

Spiders were sorted from the pitfall trap debris and stored in 70% alcohol. The species were identified using a x50 magnification microscope and nomenclature follows (Roberts 1993). Only adult specimens were identified due to the difficulty in assigning juveniles to species.

9.2.2.2.5 Data

To date the spiders from the first round of pitfall samples have been sorted and identified to species level. The remaining samples will be sorted and identified by March 10th 2006 and will be included in the final version of the BIOFOREST GIS Database.

9.2.2.3 Hoverflies

9.2.2.3.1 Rationale for the sampling design

The survey design and the nature of the survey sites posed certain problems for effective sampling of the hoverfly fauna:

- All the sites were very exposed, generally being located in upland areas with little or no shelter along the forest roads. This creates problems for passive sampling methods that are dependent upon hoverfly flight activity. Our previous experience (from sites surveyed for BIOFOREST project 3.1.1 in 2002) suggested that Malaise traps placed in these types of exposed conditions would not catch a sufficient number of hoverflies to constitute and adequate sample of the fauna.
- In each site, the two treatments are located adjacent, or nearly adjacent to each other. As adult hoverflies fly, it is likely that there would be a significant mixing of the faunas originating from the two sections of forest road. This problem would be exacerbated by the lack of any physical barriers, or habitat differentiation, between the treatments, and the relatively short length (200 m) of each section. Therefore, any sampling method using adult hoverflies would be unlikely to detect differences between the treatments if they exist.

The above considerations suggested that a sampling design based on placement of Malaise traps in pre-determined positions in each treatment would be unlikely to detect differences between the treatments and may not even provide an adequate overall baseline or the fauna of each site. Alternative sampling methods for adult hoverflies would also have similar problems.

Therefore, instead of attempting to compare the hoverfly faunas between treatments in each site, we placed Malaise traps in whatever shelter was present in each site. Our objective in doing this was to catch a sufficient number of hoverflies to provide an adequate representation of the overall hoverfly fauna in each site. This will provide a baseline against which the future development of the hoverfly fauna, as the forest matures, can be compared. Our results will not be suitable for determining whether there were within-site differences between treatments in 2005.

9.2.2.3.2 Methods

Malaise trapping

We used four Malaise traps per site so that the baseline data will be comparable with any future sampling involving pairs of Malaise traps in each forest road section. These traps were placed in the most sheltered locations that we could find within each site. Depending upon the availability of suitable sheltered conditions, these traps were variously arranged: in some cases pairs of traps were placed next to each other, while in other cases all four traps were fairly evenly distributed across the site. Most traps were place adjacent or close to the forest roads, but a few were more distant (Table 45).

Table 45. Distances from forest road of the malaise traps in metres.

	M1	M2	M3	M4
Bawn	1	16	2	1
Card	18	30	1	1
Cloo	8	1	9	20
Foss	2	5	20	25
Gate	0	5	0	10
Gull	13	28	65	65
More	50	100	67	75
Toor	3	1	10	10

The traps were operated continuously for a period of 63 days, between 9-12 May and 11-14 July 2005. The contents of the traps were collected three times during this period, at approximately three-week intervals, with the final collection at the end of the trapping period. Two samples were lost due to damage to the Malaise traps: BawnM3, trapping period 2 (30 May-17 June 2005); and ToorM2, trapping period 2 (2-21 June 2005).

Habitat recording

We recorded macrohabitats and microhabitats within a 100 m radius of each site, using the methods described in Section 7.

9.2.2.3.3 Results

Sorting and identification of the hoverfly samples will be completed in early 2006, and the results will be included in the final version of the BIOFOREST GIS database.

9.2.2.4 Birds

9.2.2.4.1 Methods

Road transects

Birds were sampled during two visits to the sites, one in May and one in June. Visits were timed to take place between 0800hrs and 1800hrs, and were not made during persistent or heavy rain, or in winds stronger than force 4 on the Beaufort scale. During each visit, the position, identity and behaviour of each bird detected by the observer was recorded while walking along each of the experimentally manipulated road sections.

9.2.2.4.2 Results

The results of this survey will be included in the final version of the BIOFOREST GIS database. When the data from this baseline survey are compared with bird surveys of the experimental road sections at later stages of the forest cycle, care should be taken to ensure that birds are sampled in an equivalent manner. Because birds are much more detectable in open habitat than in forest, any comparison should restricted to birds within a short distance (e.g. 10m) of the forest edge, be made after controlling for detectability of birds in open and forested situations, or densities of birds in the forest adjacent to the road gap should be sampled using point counts.

10 CONCLUSIONS AND RECOMMENDATIONS

10.1 INTRODUCTION

A large component of Irish biodiversity is associated with forest habitats, and much of this biodiversity is dependent upon areas of closed-canopy tree cover: for example vegetation of shaded forest floors and epiphytes and invertebrates associated with cool, humid microhabitats in trees. However, another important component of biodiversity in forest plantations is the flora and fauna associated with open space habitats within forests. Coniferous plantation forests in Ireland and the UK are generally darker than the natural broadleaf forests and have been found to lack elements of biodiversity associated with open spaces and less dense canopies in natural forest. Many of the characteristic forest species remaining in Ireland are, strictly speaking, species of forest edges and glades, rather than forest interior species. In intensively farmed landscapes, open spaces within forests may provide suitable habitat for species characteristic of semi-natural open space habitats, which no longer occur within the surrounding landscape.

In this report we have described the results of our studies on the terrestrial vegetation, epiphytes, spiders, hoverflies and birds associated with open spaces in Irish plantation forests. This section provides a synthesis of the key findings that we have described. Our focus is on results that have implications for the management of open spaces to enhance biodiversity. We have identified nine relevant features of open space in forest plantations, which we consider in turn below. For each feature, we discuss the existing regulatory requirements, briefly summarise the relevant results from our research and then discuss the implications of these results. We then make recommendations about forestry management practices that can influence the feature and identify any modifications that may be required to the *Forest Biodiversity Guidelines* (Forest Service 2000b; referred to without citation hereafter). All of our recommendations are made with the caveats presented in Section 10.2.

10.2 CAVEATS

10.2.1 The application of simple management prescriptions

For practical management purposes, and for ease of ensuring regulatory compliance, it is desirable to have simple criteria, such as requirements for fixed percentages of open space. However, in the application of ecological management principles, there will always be exceptions to simple rules. Where our recommendations include specific criteria (such as minimum width of forest roads), these should be interpreted as general principles, and provision should be made for exceptions. In particular, priority should usually be given to existing features of biodiversity importance such as retained habitats of conservation value. There are also likely to be sites where there is limited potential for developing any open space habitat of significant biodiversity interest. The potential contribution of open space habitats in forest plantations to maintaining open space biodiversity at the landscape level will depend largely upon the overall amount of open space habitat in the landscape. In landscapes where there are significant areas of high quality open habitats, open spaces within plantations (that are not important retained habitats) may not contribute much to the overall biodiversity of the landscape. Conversely, in landscapes where semi-natural open space habitat is rare, due to agricultural intensification or widespread afforestation, open spaces in plantations may have a significant role in maintaining landscape-level biodiversity. The above considerations show that there are complex issues involved in assessing the potential biodiversity value of open spaces in forest plantations. Therefore, we would warn against an uncritical "one size fits all" approach to biodiversity regulation of forestry, and recommend consideration of more focused approaches.

10.2.2 Limitations of our research

This study has documented the terrestrial and epiphytic vegetation, spider, hoverfly and bird biodiversity of open spaces in mature Sitka spruce plantations and has developed recommendations for enhancing the biodiversity of these plantations. Inevitably, however, with a study of this nature we have not been able to address all the relevant issues. Therefore, in interpreting the results of this study it is necessary to bear in mind the caveats we discuss below.

Our study was limited to plantations dominated by Sitka spruce in upland areas on poor soils in two regions of Ireland. Other crop species may have different effects on biodiversity in small open spaces due to differences in effects on the light regime. Soil type has a major effect on open space biodiversity, determining the types of open space habitat (see Sections 4.3.2.1.1 and 4.3.2.3.1) and also influencing responses to grazing pressure, or lack of grazing pressure (and we discuss this issue in Sections 10.3, 10.9 and 10.11).

In our surveys, we focused on sections of forest roads and rides that were predominantly orientated east-west. The orientation of open spaces can have strong effects on the light levels within these open spaces (Yallop & Hohenkerk 1991). Light levels affect vegetation and invertebrate biodiversity in open spaces (Greatorex-Davies & Sparks 1994; Sparks *et al.* 1996). Therefore, the precise quantitative form of the relationships that we have found between forest road and ride width and open space biodiversity may not apply to sections of forest roads and rides that are orientated generally north-south.

Like any biodiversity study we have had to be selective about the taxonomic groups that we studied. While our study has covered a broad range of taxonomic and functional groups, there are important components of biodiversity that we have not sampled. In particular, among the invertebrate groups, we have studied one group of mainly ground-dwelling predators (spiders), and another group of trophically and functionally diverse species (hoverflies). Inevitably there are many diverse invertebrate groups that we did not survey. The important question is the extent to which the invertebrate groups that we have covered represent the range of functional diversity that exists in open space habitats in plantation forests. In this context, significant gaps include arboreal spiders and host-specific phytophagous invertebrates.

Arboreal spiders are a major component of the predatory invertebrate fauna in tree and shrub canopies. While the hoverfly fauna that we studied includes a significant component of predators, these are mainly specialists on aphids. Arboreal spiders favour other prey such as springtails (Collembolla) and mites (Acari) and could, therefore, show other effects of open space type and management.

While *Cheilosia* hoverflies are phytophagous, they did not form a significant component of the fauna in the sites that we sampled. Phytophagous insects associated with typical plants of the open space types that we surveyed (such as many Lepidoptera) could well show other effects of open space type and management.

10.3 THE CONTRIBUTION OF OPEN SPACES TO FOREST BIODIVERSITY

Our work supports the results of previous research that has emphasised the contribution of open spaces to the biodiversity of plantation forests. Open spaces contain vegetation communities that cannot develop in closed canopy conditions and usually support higher numbers of vascular plant species than are found under closed canopies. There were 52 species of spiders that we only found in samples from open spaces in the forests, in contrast to just six species that were only present in samples from closed canopy areas, and average spider species richness per plot was significantly higher in the open spaces. Indicator species analysis showed that a distinctive spider assemblage occurred in the open spaces. Our hoverfly survey sampled both the forest and open space components of the hoverfly fauna (see Section 7.2.2). Nearly 80% of the hoverfly fauna that we recorded was associated with open space habitats, and around one-third of these are mainly associated with semi-natural habitats.

The bird fauna does not closely follow the patterns described above, as typical open space specialists, such as Skylark and Meadow Pipit, that are widespread in habitats just outside the plantation, were absent from most of the open spaces within the forest plantations that we surveyed. However, open spaces provide the main opportunity for the development of broadleaved tree and shrub cover within conifer plantations, and such vegetation is associated with higher bird biodiversity (see Section 10.9).

Overall, our results support the requirement in the *Forest Biodiversity Guidelines* for including an open space component in forest plantations. However, different types of open spaces may favour different components of biodiversity: for example, bramble thickets promote bird biodiversity but may be not be of much value for ground dwelling spider biodiversity. Moreover, in some sites, such as cutover bogs or fertile lowland plantations, open spaces may be occupied by competitive grass species or rapidly invaded by dense bramble thickets. While these open spaces may still have some biodiversity interest (spiders in cutover bogs invaded by competitive grass species and common woodland edge bird species in dense bramble thickets), these habitat types are likely to be common in the surrounding landscape and the overall biodiversity gain may be very limited. In these situations, management such as bramble control, or planting of non-crop broadleaved trees and shrubs may greatly increase the contribution of open space to plantation biodiversity.

Recommendation: Open spaces should be promoted in forest plantations as a method of biodiversity enhancement.

Modifications to the *Forest Biodiversity Guidelines*: None required.

10.4 EFFECTS OF OPEN SPACE AMOUNT

A figure of 15% has been suggested as the amount of open space that should be included in conifer plantations in Britain because this corresponds to the amount of open space that occurs in natural coniferous forests (Peterken 1999). In Ireland, the *Forest Biodiversity Guidelines* require plantations to contain 5-10% open space, except in plantations of less than 10 ha in size. However, we are not aware of any work that has explicitly examined the relationship between the proportion of open space in a forest plantation and the contribution of the open space habitat to the biodiversity of the plantation.

In our research, we found that the species richness of open space associated spiders was positively related to the total area of unplanted open space within 200 m, but there was no relationship with the area of the unplanted open space within which the plot was located. Therefore, spider biodiversity appears to be enhanced by increased amounts of open space at the plantation scale. We also found that spider species richness increased with the amount of unplanted open space in the three categories of <5%, 5-10% and >10%, suggesting that the open space amounts of 5-10% may not be adequate to support a full suite of species associated with open habitats.

By contrast with our results from the spider surveys, we did not find any relationships between overall amounts of open space and biodiversity in the other groups that we studied. This may be because these other groups are more ecologically heterogeneous. The spider fauna sampled by pitfall traps is dominated by money spiders (Linyphiidae), a large family of spiders containing many species with similar broad ecological requirements, mainly dependent upon high amounts of vegetation cover in the field layer. By contrast, the composition and diversity of vegetation communities was more strongly influenced by factors operating at larger scales, such as climate and biogeography, and smaller scales, such as soil conditions and light regime. Hoverfly and bird assemblages consist of species occupying a range of trophic levels and utilising microhabitats from the ground layer to the tree canopy. Therefore, other factors are more likely to confound any relationships between open space amounts and biodiversity in these groups. The relationship between ground-dwelling spider biodiversity and open space amount suggests that when the effects of habitat variation are removed, increasing the overall amounts of open space in a forest plantation will increase the biodiversity of open-associated species in the plantation.

Our results suggest that the open space requirement in the *Forest Biodiversity Guidelines* of 5-10% may not be sufficient for the maintenance of open space associated spider biodiversity in forest plantations. In fact, an even higher amount of open space than the 10% threshold suggested by our data may be required. The relationships that we found between open space amounts and spider biodiversity were with unplanted open spaces (including wide rides). The current open space requirement in the *Forest Biodiversity Guidelines* also includes forest roads. We found weaker correlations between spider biodiversity and combined amounts of unplanted open space and forest roads.

In considering the adequacy of the 5-10% open space requirement in the *Forest Biodiversity Guidelines*, it also necessary to consider the potential contribution of the additional 5-10% retained habitats, which may often also be open space habitat. Therefore, the specific open space requirement could be flexible, depending upon the amount of open space habitat included in the retained habitat component.

Finally, while we have discussed above the relationships between open space amounts and simple biodiversity measures, in formulating policy a more fundamental issue needs to be considered: what is the overall contribution of the open space habitat within forest plantations to maintaining biodiversity at the landscape level, and how is this contribution affected by variation in open space amounts within the plantation. As discussed in Section 10.2.1, this contribution is likely to vary depending upon both the nature of the open space habitat within the plantation and the distribution of open space habitats in the wider landscape.

Recommendations: Increasing the amount of open space habitat in plantations will generally have a positive impact on the biodiversity of these forests. Benefits to biodiversity do not necessarily level off at 10% open space, and, in some plantations, larger amounts of open space should be considered. However, the contribution of open space habitat within plantations needs to be considered at a landscape scale and simple universal prescriptions about open space amounts may not be desirable.

Modifications to the *Forest Biodiversity Guidelines*: The prescription for 5-10% open space amounts in the *Forest Biodiversity Guidelines* should be reviewed in relation to both the adequacy of the existing specification and the desirability of a universal prescription.

10.5 FOREST ROAD WIDTH

The *Irish Forest Road Manual: Guidelines for the design, construction and management of forest roads* (Ryan *et al*. 2004) recommends a minimum clearance width of 15 m between trunks (inter-trunk) for new forest roads, and forest roads can be included in the 5-10% open space required in new plantations greater than 10 ha in size (Forest Service 2003). A clearance of 15 m reduces requirement for maintenance of the road surface by promoting drying and heating effects of absence of canopy cover. Forest roads are necessary only in forest areas greater than 10 ha: generally forest tracks serve for operations in smaller forests (Forest Service 2000a).

The biodiversity of various invertebrate groups in open spaces within forest plantations has been shown to increase with decreasing levels of shade (Greatorex-Davies & Sparks 1994; Greatorex-Davies *et al*. 1993) and shade levels will tend to decrease as open space size increases. We found relationships between forest road width and various components of open space biodiversity.

Vegetation composition of wide roads differed from that of more shaded, narrow roads. The flora of well-lit road verges made an important contribution to vegetation biodiversity of roads and plantations as a whole. Diffuse solar radiation reaching road centres increases with increasing road width up to 25-30 m width.

The species richness and abundance of open-associated spiders was positively correlated with the width of forest road verges. The species richness showed no indication of reaching a plateau with increasing road verge width and continued to increase in road verges much wider than the verge width that corresponds to a forest road width of 15 m wide. Open-associated hoverfly species richness was positively correlated with forest road width and our data suggest that forest road widths of 15 m may be too narrow for optimum hoverfly biodiversity. Although road gap width was not found to be related to bird species richness, road gaps less than 15 m wide had significantly lower cover of shrubs and broadleaved trees; both variables appear to have a positive influence on bird diversity. On average, in our datasets an inter-trunk forest road width of 15 m is equivalent to an inter-canopy width of 9.2 m and a road verge width of 7 m (see Figure 37).

For some groups, therefore, it is likely that forest road widths of greater than 15 m would enhance biodiversity. A wider minimum width (20 m) is recommended for reversal roads (roads that are constructed by taking mineral soil from beneath the peat layer and placed as an embankment on top of the peat ; Ryan *et al*. 2004). However, in general, forest roads much wider than 15 m are not preferred for forestry operations, as wide verges are difficult for machinery to cross during harvesting operations (Noel Foley, pers. comm.). Additionally there is concern about the fragmentation and changes in hydrological conditions that forest roads cause. A compromise between encouraging wide forest roads for biodiversity reasons and operational requirements might be achieved by developing forest roads with wide scallops, i.e. alternating sections of road of standard and wide widths. In fact this type of road design would probably be better for biodiversity than a road of uniformly wide width, as the scallops will provide shelter from wind tunnel effects and also will increase the length of forest edge habitat. These scallops should be placed mainly on the side of the forest road with a more southerly aspect, as this will be the side that receives more sunlight. Varying the light conditions of roadside features such as banks and ditches will also likely increase plant diversity. Scalloped edges to forest roads and rides are recommended in British guidelines on managing open spaces for biodiversity (Ferris & Carter 2000a; Warren & Fuller 1993).

Recommendations: Fifteen metres is the minimum width of gap required by many open space biota. Some areas of road should have a clearance substantially wider than 15 m and this may be achieved by developing scalloped edges to forest roads. However, the placing of wider forest roads must be planned with due consideration for side-effects which may be deleterious to the forest biota.

Modifications for *Forest Biodiversity Guidelines*: It may be necessary to modify the *Guidelines* to specifically promote the use of scalloped forest edges along forest roads, creating wide open spaces at intervals along the road. Only sections of forest roads wider than 15m may then count towards the specified open space requirement.

10.6 RIDE WIDTH

Rides can be included in the 5-10% open space required in new plantations greater than 10 ha in size and ridelines would normally be 6 m wide (Forest Service 2003)[1], although Iremonger (1999) recommended that they should be much wider if possible (to 1.5 times the size of adjacent trees).

Our vegetation and spider surveys included a number of narrow (6-10 m wide, inter-trunk) rides. The vegetation composition of wide and narrow rides was distinct, and the species richness of vascular plants was positively associated with ride width. The abundance and species richness of open-associated spiders was positively correlated with ride width and species richness was higher in open spaces large enough to be unshaded (15 m or wider). We did not specifically include narrow rides in our hoverfly and bird surveys. However, rides that are 6 m wide in mature plantations are too narrow to support well-developed open space habitats and will not, therefore, support a significant open space-associated hoverfly fauna, or bird species associated with shrubs and broadleaved trees. As with roads, diffuse solar radiation at ride centre increases steeply with ride width, levelling off at 25-30 m. Given that the average length of branches at forest edges is 3.6 m (Section 4.4.4), very few rides 6 m wide will not be completely overhung by branches.

Recommendations: A gap of 6 m (such as that made by a standard rideline) is too narrow to be treated as open space from a biodiversity perspective and rides (or other linear features currently treated as open spaces under the *Forest Biodiversity Guidelines*, such as drains) may need to be at least 15 m wide to constitute well-developed open space habitat.

Modifications for *Forest Biodiversity Guidelines*: Rides of standard width (6 m wide) should not be included in the 5-10% open space requirement. A minimum width (probably 15 m) should be specified for rides or other linear open spaces to qualify for inclusion in the 5-10% open space requirement.

10.7 GLADE SIZE AND SHAPE

Unplanted open spaces (glades) can be included in 5-10% open space required in new plantations greater than 10 ha in size, and there is no minimum size specified (Forest Service 2000b; Forest Service 2003). The *Forest Biodiversity Guidelines* recommends design of open spaces to create an undulating forest edge and, thereby, maximising the forest edge habitat.

We did not find any relationships between glade size and biodiversity. However, apart from one glade with an area of 80 m^2 sampled for spiders, all the glades were at least 1000 m^2. Therefore to identify a threshold area (over which open species can be supported), glades smaller than 1000 m^2 would need to be studied. However, the 15 m threshold for ride width might suggest that 225 m^2 should be the minimum area for partially-lit glades, while glade areas of 625-900 m^2 should be sufficient to have the centre of the glade well-lit, depending on local conditions, such as slope and aspect.

In addition to glade size, glade shape can affect biodiversity, and our results support the recommendation in the *Forest Biodiversity Guidelines* about maximising forest edge habitat. Creating glades with high edge to area ratios will enhance vegetation biodiversity, by increasing the transition zone where neither the most competitive open space species nor shade-tolerant species can dominate. High edge to area ratios will also maximise the forest edge habitat and, where broadleaved trees and shrubs develop at the forest edge, will enhance hoverfly and bird biodiversity (see Section 10.9).

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[1] Forest Service (2003) does not specify whether this means 6 m between trunks or 6 between the edges of the canopy. However, as this reference also refers in a similar way to the 15 m width for forest roads (which is defined as 15 m between trunks by Ryan *et al*. 2004), it is reasonable to assume that the 6 m figure refers to the width between trunks.

Recommendations: A minimum glade size of 225 m^2 is probably required to allow development of some characteristic open space biodiversity, while glade sizes of at least 625-900 m^2 may be necessary to develop light conditions at the glade centre unaffected by the adjacent forest canopy. The potential value of glades for biodiversity is likely to be enhanced by creating high edge to area ratios.

Modifications to the *Forest Biodiversity Guidelines*: A minimum glade size of 225 m^2 should be specified for glades to qualify for inclusion in the 5-10% open space requirement, and larger glade sizes (at least 625-900 m^2) should be encouraged.

10.8 PROTECTIVE ZONE AROUND RETAINED HABITATS

The *Forest Biodiversity Guidelines* recommend that the sustainability of retained habitats can be enhanced by enforcing a 3 m protective zone. We did not specifically examine the biodiversity implications of varying the width of the protective zone around retained habitats. However, our results on the relationship of forest road and ride widths with biodiversity are applicable to the issue of the width required for linear retained features, such as hedgerows and treelines. A 3 m protective zone around such features would probably usually equate to a gap of less than 10 m (i.e., a 3 m protective zone on either side and the width of the feature). Our results discussed above (see Sections 10.5 and 10.6) show that this is likely to be too narrow to support well-developed open space habitats and that a protective zone that would create a gap of at least 15 m is probably required.

The buffer zone around streams would generally be determined by the *Forestry and Water Quality Guidelines* (Forest Service 2000d; referred to without citation hereafter) and, under these guidelines, would be a minimum of 10 m on either side. However, small streams not marked on the Ordnance Survey six inch maps and wet flushes are not covered by this requirement (see Section 10.10), and, in such cases, it will be necessary to specify an adequate buffer zone under the *Biodiversity Guidelines* if these features are to contribute towards the Area for Biodiversity Enhancement.

Recommendations: The protective zone around retained habitats should be at least 7 m (on each side) for linear features such as hedgerows, treelines and small streams (not covered by the *Forestry and Water Quality Guidelines*), to ensure that they do not get shaded out as the plantation matures (the current recommended width is 3 m).

Modifications to the *Forest Biodiversity Guidelines*: A mandatory minimum protective zone of 7 m should be required for linear features in order for them to qualify for inclusion on the Area for Biodiversity Enhancement.

10.9 BROADLEAVED SHRUBS AND TREES

The *Forest Biodiversity Guidelines* recommends that shrubs and native broadleaves should be encouraged through planting and/or appropriate management of open and retained habitats.

We found that the presence of broadleaved trees and shrubs was associated with increased biodiversity of vegetation, hoverflies and birds. Roadsides that have developed substantial bramble cover have relatively high diversity and species richness of both vascular plants and bryophytes. There was a positive relationship between species richness of hoverflies with larvae developing in the foliage of broadleaved trees and shrubs and the frequency of broadleaved woody vegetation (including bramble, but excluding dwarf shrubs as defined in Table 25). Bird species richness and the abundance of several bird species was positively related to the cover of broadleaved trees and shrubs, and, at the plantation scale, forests with a broadleaved woodland component had higher bird species richness than those without.

The relationships that we have found are not with open space habitat per se. However, in practice the only opportunity in spruce plantations for significant cover of broadleaved trees and shrubs is in open spaces, because they will usually be out-competed by the more vigorous growth of the conifers within the closed-canopy areas.

Almost all the broadleaved trees and shrubs in our sites were native species so we are not able to examine whether there are differences in biodiversity gain as between the presence of native or non-native species of broadleaved trees and shrubs. Alien broadleaved evergreen shrubs such as *Rhododendron ponticum* greatly reduce plant diversity in the field (herbaceous) layer and ground (bryophyte) layer (Kelly 1981).

Our results support the existing recommendations in the *Forest Biodiversity Guidelines* about encouraging broadleaved trees and shrubs. In most sites, natural regeneration processes will probably be sufficient to generate adequate cover of this vegetation. However, in exposed upland sites and/or on deep peats some planting may be necessary. It is also important to maintain the habitat diversity of the open space resource. In fertile lowland sites open spaces may very quickly become completely colonised by dense thickets of bramble and other shrubs. In these sites, the management problem is not how to develop cover of broadleaved trees and shrubs but is how to maintain areas of non-woody open space habitat. This is particularly important when the Area for Biodiversity Enhancement includes valuable areas of retained habitat such as species-rich semi-natural grassland.

Recommendation: Broadleaved shrubs and trees make important contributions to forest biodiversity and open spaces provide the main opportunity for the development of this vegetation in conifer plantations. More specific guidance for foresters could help them to encourage shrub and non-crop tree patches/stands in plantations.

Modifications to the *Forest Biodiversity Guidelines*: The *Forest Biodiversity Guidelines* should include more specific guidelines on how to encourage shrub and non-crop tree patches/stands in plantations. For example: not clearing roadside scrub any more than strictly necessary for safety purposes; using mechanical clearance methods rather than herbicides (see Section 4.4.4); and providing open spaces nearby existing broadleaved seed sources (Smith 2003).

10.10 Small wet habitat features

The *Forestry and Water Quality Guidelines* require that aquatic zones identified on Ordnance Survey six-inch maps are protected and the *Forest Biodiversity Guidelines* require the mapping of biodiversity considerations.

In many forests small wet habitat features such as temporary streams and wet flushes occur, which are not included on standard Ordnance Survey six-inch maps. We found that the species richness of hoverflies with larvae that develop in wet habitats was positively associated with the frequency of these features, and these types of features are also likely to be important for other biota such as bryophytes, and bird species such as Snipe (*Gallinago gallinago*) and Woodcock (*Scolopax rusticola*). Based on the known habitat associations of the hoverfly species, the combination of open space and wet habitat is likely to be important in maintaining their biodiversity, as most are not associated with closed-canopy spruce forests. However, the important consideration here is the absence of conifer planting around the features; development of scrub or native woodland would be acceptable. In comparing plant biodiversity associated with different types of native woodland, Kelly and Iremonger (1997) found the highest values in communities typical of flushed habitats.

In the areas that we surveyed, only 20% of the 33 wet habitat features (excluding drainage ditches) that we recorded were shown on the six-inch maps. The remainder would not, therefore, be protected by the requirements of the *Forestry and Water Quality Guidelines*. The requirements of the *Forest Biodiversity Guidelines* provide an alternative mechanism for achieving this protection, as if these features are identified as part of the retained habitat component of the Area for Biodiversity Enhancement, the protective zone requirements (see Section 10.8) will then apply to them. However, it is likely that, without specific guidance, many such features would not be recognised because of their very small scale. This highlights the need for more specific guidance in the *Forest Biodiversity Guidelines* on identifying and mapping habitats, as discussed in more detail in our report *Biodiversity Assessment in Preparation for Afforestation* (Gittings *et al*. 2004a).

Recommendations: Small wet habitat features, such as temporary streams and wet flushes, that are not mapped on Ordnance Survey six-inch maps can be important for biodiversity. Such features should be identified and assessed for inclusion in the Area for Biodiversity Enhancement. More specific guidance is required to help foresters to identify and assess these features.

Modifications to the *Forest Biodiversity Guidelines*: The *Forest Biodiversity Guidelines* should emphasise the importance of small wet habitat features that are not mapped on Ordnance Survey six inch maps, recommend that these be include in the Area for Biodiversity Enhancement, and provide specific guidance to help foresters to identify these features.

10.11 GRAZING

The *Forest Biodiversity Guidelines* recommends protection against deer in areas where natural regeneration is being encouraged. It is a condition of grant aid schemes and of felling licenses that forests must be adequately fenced to prevent intrusion of domestic grazing stock (Forest Service 2002). Guidance on deer control is provided by (Forest Service 2002)

In semi-natural woodlands, the two management extremes of zero grazing and heavy grazing have both been shown to be associated with reduced levels of diversity in vegetation structure and plant species composition (Kelly 2000; Kirby *et al*. 1994; Mitchell & Kirby 1990). A low level of grazing may create maximal diversity among small mammals, birds and invertebrates (Mitchell & Kirby 1990).

Our study was not designed to investigate the effect of grazing on forest biodiversity. However, levels of grazing differed markedly between our two site clusters, being much higher in the Cork than in the Wicklow sites. We found differences in the vegetation communities between the Cork and the Wicklow sites, some of which may have been caused by differences in grazing pressure between the clusters. Shrub cover and broadleaved tree cover were significantly higher in Cork than in the Wicklow sites, both along roads and within 50 m of bird point count locations. It seems likely that this difference was at least partly due to differences in grazing pressure between the two sites, especially given that the more fertile sites in Wicklow would, other factors being equal, be expected to be more suitable for development of broadleaved species than the peaty soils of our Cork sites. If high levels of grazing can retard or prevent the development of broadleaved tree and shrub cover, they would be expected to have a negative impact on hoverfly and bird diversity. Indeed, species richness of birds and of tree and shrub associated hoverflies along forest roads was higher in the Cork than in the Wicklow sites.

Recommendations: More research needs to be done to determine the optimal grazing regimes for biodiversity in forest open spaces. However, in areas where open spaces within forests come under heavy grazing pressure, it is likely that grazing pressure will need to be managed if broadleaved tree and shrub vegetation to develop.

Modifications to the *Forest Biodiversity Guidelines*: More research is required before specific recommendations can be made.

10.12 FURTHER RESEARCH

10.12.1 Context

Our research has identified some important aspects of open space configuration and management that affect the contribution of open spaces to biodiversity in Sitka spruce plantation forests in Ireland. However, we have also identified areas where further research would be useful. In the individual chapters, we discuss topics for further research that will help to improve our scientific understanding of the processes affecting the biodiversity of the groups involved. However, this chapter focuses on the management of open spaces to enhance biodiversity (see Section 10.1). Therefore, in this section, we highlight areas where further research would be likely to yield results of direct relevance to the development of guidelines for open space management in plantation forests.

10.12.2 Landscape type

Our study was restricted to plantations in upland landscapes, on poor soils, and usually with extensive areas of semi-natural open space habitat in the vicinity. However, a large proportion of future afforestation is likely to take place in more-or-less intensively farmed lowland landscapes. In these landscapes, open spaces in plantations will usually have more fertile soils and there will be different, pools of potential open space species to colonise the habitat. Also, there may be management problems in maintaining useful open space habitat in these situations (see Section 10.3). However, these types of open spaces may have the greatest potential to contribute towards biodiversity maintenance at the landscape scale (Peterken & Francis 1999), depending on the intensity of management in the surrounding landscape. Therefore, research into the biodiversity of open spaces in plantations in agricultural lowland landscapes would be useful in establishing the value of these open spaces and providing management guidelines to realise their potential. Such research should take into account the open habitats present in the landscape outside the forest boundary and differing agricultural management regimes (e.g. REPS and non-REPS farms).

10.12.3 Forest type

Our study was restricted to plantations dominated by Sitka spruce. In theory, there may be different relationships between open space and biodiversity in plantations dominated by other conifer species or by broadleaved species. However, the effect of canopy tree species on open space biodiversity is likely to be limited to the biota of the forest edge and is likely to depend more on major structural differences (e.g., deciduous vs. evergreen trees, difference in self-pruning of lower branches) than the precise species involved. Therefore, further research in this area should focus on taxa that are likely to have distinct forest edge assemblages, and should test specific hypotheses about how variation in forest edge structure affects these taxa.

It would also be useful to investigate the biodiversity of open spaces in semi-natural woodlands to provide reference data to put studies of open spaces in plantation forests into context.

10.12.4 Open space habitat

The focus of our study was on identifying relationships between biodiversity and open space amounts and configuration. Therefore, to achieve adequate replication, and to avoid confounding factors, we had to restrict our sampling to a single broad open space habitat type in each region, which inevitably meant that we focused on widespread and mundane open space habitats. More interesting open space habitats occur in plantation forests. For example, in preliminary site selection visits for this project we saw habitats such as acid fen

(Fossitt 2000) and alder carr with tussock sedge (Forest Service 2005). Research into the biodiversity of important open space habitats such as these would help develop guidelines for the management of important retained habitats.

10.12.5 Grazing

We have already discussed the potential significance of grazing (mainly by deer) as an influence on the biodiversity of open spaces in plantation forests and have highlighted the need for further research on this topic (see Section 10.11).

10.12.6 Other taxa

We have already discussed the limitations of our research in terms of the restricted range of taxonomic groups that we surveyed (see Section 10.2.2). Research on the biodiversity of other taxonomic and functional groups that are likely to have different ecological responses to open space configuration and management would be useful. These could include: epiphytes on broadleaved trees and shrubs, spider fauna in shrubs and trees, moths and ground beetles. Moths and ground beetles have already been extracted from our Malaise trap and pitfall trap samples and could, therefore, be investigated relatively easily.

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APPENDICES

Appendix 1 BIOFOREST Staff and administrative groupings

Individuals involved in the BIOFOREST Project met periodically to plan and review. The following were the main groups that met.

1. Research Group:

Department of Zoology, Ecology and Plant Science and the Coastal and Marine Resources Centre, University College, Cork (UCC): Professor Paul Giller, Professor John O'Halloran, Dr Tom Kelly, Dr Tom Gittings, Dr Mark Wilson, Dr Josephine Pithon, Ms Anne Oxbrough

Botany Department, Trinity College, Dublin (TCD): Dr Daniel Kelly, Dr Fraser Mitchell, Dr Paul Dowding, Dr George Smith, Dr Laura French, Ms Linda Coote, Dr Susan Iremonger, Dr Anne-Marie McKee and Ms Saoirse O'Donoghue

Coillte Teoranta: Dr Aileen O'Sullivan, Mr Pat Neville, Dr Alistair Pfeifer.

Others joined this Research Group at different stages of the project, in particular:

Coastal and Marine Resources Centre, University College, Cork: Ms Valerie Cummins, Ms Vicki O'Donnell

Temporary research students and associates:
Ms Erika Buscardo, Ms Jacqueline Bolli, Ms Julianna O'Callaghan.

2. Management Group:

COFORD: Joe O'Carroll
EPA: Helen Walsh, Dr Conor Clenaghan, Dr Garret Kilroy, Dr Karl Richards
UCC: Prof. Paul Giller, Prof. John O'Halloran, Dr Tom Gittings
TCD: Dr Daniel Kelly, Dr George Smith
Coillte: Dr Aileen O'Sullivan
Project manager: Dr Susan Iremonger

3. Steering Group:
This Group was composed of the other two Groups, plus:

National Parks and Wildlife Service: Dr John Cross
Forest Service: Noel Foley
Centre for Ecology and Hydrology (UK): Dr Allan Watt
Forestry Commission (UK): Dr Jonathan Humphrey
University of Helsinki (Finland): Dr Jari Niemelä
European Environment Agency (Denmark): Dr Tor-Björn Larsson

Appendix 2 Discussion group members

During the conference conference "Opportunities for enhancement of biodiversity in plantation forests", 24 October 2002, Vienna Woods Hotel, Cork a special session discussed the various options for the focus of BIOFOREST Project 3.1.3. This was attended by members of the BIOFOREST Steering Group and individuals from forest-related institutions both inside and outside of Ireland

Tom Bolger (UCD)

Linda Coote (TCD)

Noel Foley (Coillte Teoranta)

Tom Gittings (UCC)

Jonathan Humphrey (Forestry Commission, UK)

Susan Iremonger (TCD)

Daniel Kelly (TCD)

Garret Kilroy (EPA)

Pat Neville (Coillte Teoranta)

Joe O'Carroll (COFORD)

Saoirse O'Donoghue (TCD)

John O'Halloran (UCC)

Aileen O'Sullivan (Coillte Teoranta)

Flemming Rune (Danish Forest and Landscape Research Institute)

George Smith (TCD)

Mark Wilson (UCC)

Ian Wright (FIE)

Appendix 3 Letter for open space survey

BIOFOREST

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Date

Dear,
The BIOFOREST Project is currently investigating the effects of open space management on biodiversity in Irish forestry plantations. This is a part of a larger five-year study on different aspects of plantation forest biodiversity, involving researchers in Trinity College Dublin, University College Cork and Coillte, the Irish Forestry Board. In this section of the study we hope to gather information from other countries regarding the enhancement of forest biodiversity through management of internal open spaces, including glades, rides and roads. In particular we are interested in any policy or research documents that specify methods of encouraging biodiversity through manipulation of open spaces.
For example, there may be a policy of leaving small areas within the forest unplanted to create discrete open spaces, or perhaps a recommendation to either clear forest rides periodically or to leave them so that scrub develops. As regards forest roads there may be regulations stating that any road in a forest must be a minimum width, or trees must not be planted within a given distance from a forest road. Although there is little on biodiversity specifically contained in the Irish government publications, guidelines regarding considerations for biodiversity given to Irish foresters on forest roads are addressed in recent documents including the Code of Best Forest Practice (section 14) and the Guidelines for Forestry and Biodiversity (both available for download at: http://www.agriculture.gov.ie/index.jsp?file=forestry/publications/publications.xml).
There is also a new Forest Roads Manual produced by COFORD, the Council for Forest Research and Development (available at: http://www.coford.ie/reports/ForestRoadManual.pdf).
Through contacting government forestry departments (or equivalent agencies) in a number of countries we hope to be able to discuss the Irish policies and practices in the context of those in other countries. If you could inform us of any policies regarding the biodiversity management of open spaces within forests in {your country}, or indicate to us where we could find documentation regarding this, we would be most grateful. We would be very happy to share with you the results of information received in this survey if that would be of interest to you.
Information on our BIOFOREST Project is available at http://bioforest.ucc.ie, and my particulars are available at http://www.esatclear.ie/~siremonger.
Many thanks.

Yours sincerely,

Susan Iremonger, Ph.D.

Appendix 4 Site names for the survey of open spaces.

Wicklow/Dublin		Cork/Kerry/Limerick	
CURA	Ballycurragh, Co. Wicklow	**CARR**	Carrigagulla, Co. Cork
MUCK	Mucklagh One, Co. Wicklow	**GLAN**	Glannaharee West, Co. Cork
ATHN	Athdown, Co. Wicklow	**MEEN**	Meentinny, Co. Cork
STOE	Ballinastoe, Co. Wicklow	**REAN**	Reanahoun, Co. Cork
BMUT	Ballysmuttan, Co. Wicklow	**CLEA**	Cleanglass, Co. Limerick
LUGG	Lugg, Co. Dublin	**KNOC**	Knocknagoum, Co. Kerry

Appendix 5 Terrestrial vascular plant, bryophyte and lichen species recorded.

Terrestrial plant species recorded in the extensive survey. *Taxon* indicates: V- vascular plant, B- bryophyte, L- lichen. *Open / Woodland* indicates: 1- species typical of open habitats, 2- species sometimes found in woodlands or species characteristic of woodland edges, 3- typical woodland species. *C* indicates if the species is a competitor, *S* indicates if the species is a stress-tolerator and *R* indicates if the species is a ruderal, under Grime's CSR theory (Grime *et al*. 1988). Nomenclature follows Stace (1997a) for vascular plants, Smith (2004a) for mosses, Paton (1999) for liverworts and Purvis *et al.* (1992) for lichens.

Species	Taxon	Open / Woodland	C	S	R
Acer pseudoplatanus	V	3	1	0	0
Achillea millefolium	V	1	1	0	1
Agrostis canina sl.	V	2	1	1	1
Agrostis canina ssp. canina	V	2	1	1	1
Agrostis capillaris	V	2	1	1	1
Agrostis stolonifera	V	2	1	0	1
Agrostis vinealis	V	2	1	1	1
Aira praecox	V	1	0	1	1
Alnus glutinosa	V	3	1	1	0
Anthoxanthum odoratum	V	2	0	1	1
Arrhenatherum elatius	V	1	1	0	0
Athyrium filix-femina	V	3	1	0	0
Bellis perennis	V	1	0	0	1
Betula pubescens	V	2	1	0	0
Blechnum spicant	V	3	0	1	0
Brachypodium sylvaticum	V	3	1	1	0
Callitriche stagnalis	V	1	0	0	1
Calluna vulgaris	V	2	1	1	0
Cardamine flexuosa	V	2	0	1	1
Cardamine pratensis	V	2	1	1	1
Cardamine species	V	.	.	1	1
Carex binervis	V	1	0	1	0
Carex curta	V	1	0	1	0
Carex disticha	V	1	1	1	0
Carex echinata	V	2	0	1	0
Carex flacca	V	1	0	1	0
Carex laevigata	V	3	0	1	0
Carex nigra	V	1	1	1	0
Carex ovalis	V	1	0	1	0
Carex panicea	V	1	0	1	0
Carex pilulifera	V	1	0	1	0
Carex rostrata	V	1	1	1	0
Carex viridula	V	1	0	1	0
Centaurea nigra	V	1	1	1	1
Cerastium fontanum	V	1	0	0	1
Chamerion angustifolium	V	2	1	0	0
Cirsium arvense	V	1	1	0	0
Cirsium palustre	V	2	1	1	1
Cirsium vulgare	V	1	1	0	1
Crataegus monogyna	V	2	1	1	0
Cynosurus cristatus	V	1	1	1	1

Species	Taxon	Open / Woodland	C	S	R
Dactylis glomerata	V	2	1	1	1
Danthonia decumbens	V	1	0	1	0
Deschampsia flexuosa	V	2	1	1	0
Digitalis purpurea	V	2	1	1	1
Drosera rotundifolia	V	1	0	1	0
Dryopteris affinis	V	3	1	1	0
Dryopteris carthusiana	V	3	1	1	0
Dryopteris dilatata	V	3	1	1	0
Dryopteris filix-mas	V	3	1	1	0
Dryopteris juvenile	V	3	1	1	0
Epilobium brunnescens	V	1	0	1	1
Epilobium montanum	V	2	1	1	1
Epilobium obscurum	V	1	1	1	1
Epilobium parviflorum	V	1	1	1	1
Equisetum fluviatile	V	1	1	0	1
Erica cinerea	V	1	0	1	0
Erica tetralix	V	1	0	1	0
Eriophorum angustifolium	V	1	0	1	0
Eriophorum vaginatum	V	1	1	1	0
Euphrasia arctica	V	1	0	1	1
Fagus sylvatica	V	3	1	1	0
Festuca ovina	V	1	0	1	0
Festuca rubra	V	1	1	1	1
Fuchsia magellanica	V	1	1	0	0
Galium saxatile	V	1	0	1	0
Geranium robertianum	V	2	1	1	1
Glyceria fluitans	V	1	1	0	1
Hedera helix	V	3	1	1	0
Holcus lanatus	V	1	1	1	1
Holcus mollis	V	2	1	0	0
Hyacinthoides non-scripta	V	3	1	1	1
Hypericum pulchrum	V	1	0	1	0
Hypochaeris radicata	V	1	1	1	1
Ilex aquifolium	V	3	1	1	0
Juncus acutiflorus	V	2	1	1	0
Juncus articulatus	V	1	1	1	1
Juncus bufonius	V	1	0	0	1
Juncus bulbosus	V	1	0	1	1
Juncus conglomeratus	V	1	1	1	0
Juncus effusus	V	1	1	0	0
Juncus squarrosus	V	1	0	1	0
Linum catharticum	V	1	0	1	1
Lolium perenne	V	1	1	0	1
Lotus corniculatus	V	1	0	1	0
Lotus uliginosus	V	1	1	1	0
Luzula multiflora	V	2	0	1	0
Luzula sylvatica	V	3	1	1	0
Molinia caerulea	V	2	1	1	0
Myosotis secunda	V	1	1	1	1
Myrica gale	V	1	1	1	0
Nardus stricta	V	1	0	1	0

Species	Taxon	Open / Woodland	C	S	R
Oreopteris limbosperma	V	2	1	1	0
Osmunda regalis	V	2	1	1	0
Oxalis acetosella	V	3	0	1	0
Pedicularis sylvatica	V	1	0	1	1
Phyllitis scolopendrium	V	3	0	1	0
Picea sitchensis	V	3	1	1	0
Pinus contorta	V	3	1	1	0
Pinus sylvestris	V	3	1	1	0
Plantago lanceolata	V	1	1	1	1
Plantago major	V	1	0	0	1
Poa annua	V	1	0	0	1
Poa humilis	V	1	1	1	1
Poa pratensis	V	1	1	1	1
Poa trivialis	V	2	1	0	1
Polygala serpyllifolia	V	1	0	1	0
Polypodium vulgare	V	3	0	1	0
Potentilla erecta	V	2	0	1	0
Potentilla reptans	V	1	1	0	1
Prunella vulgaris	V	1	1	1	1
Pteridium aquilinum	V	2	1	0	0
Ranunculus acris	V	1	1	1	1
Ranunculus flammula	V	1	1	1	1
Ranunculus omiophyllus	V	1	0	1	1
Ranunculus repens	V	2	1	0	1
Rubus fruticosus agg.	V	2	1	1	0
Rubus idaeus	V	2	1	1	0
Rumex acetosa	V	1	1	1	1
Rumex acetosella	V	1	0	1	1
Sagina procumbens	V	1	0	0	1
Salix × multinervis	V	2	1	1	0
Salix aurita	V	1	1	1	0
Salix caprea	V	2	1	0	0
Salix cinerea	V	2	1	0	0
Senecio jacobaea	V	1	0	0	1
Senecio vulgaris	V	1	0	0	1
Sonchus asper	V	1	0	0	1
Sorbus aucuparia	V	2	1	1	0
Stellaria graminea	V	1	1	1	1
Stellaria holostea	V	2	1	1	1
Stellaria uliginosa	V	1	1	0	1
Succisa pratensis	V	1	0	1	0
Taraxacum officinalis agg.	V	1	0	0	1
Teucrium scorodonia	V	2	1	1	1
Trichophorum cespitosum	V	1	0	1	1
Trifolium dubium	V	1	0	0	1
Trifolium pratense	V	1	1	1	1
Trifolium repens	V	1	1	1	1
Tussilago farfara	V	1	1	0	1
Ulex europaeus	V	1	1	1	0
Ulex gallii	V	1	1	1	0
Ulex juvenile	V	1	1	1	0

Species	Taxon	Open / Woodland	C	S	R
Urtica dioica	V	2	1	0	0
Vaccinium myrtillus	V	2	1	1	0
Veronica chamaedrys	V	2	0	1	0
Veronica officinalis	V	1	0	1	0
Veronica serpyllifolia	V	1	0	0	1
Viola palustris	V	2	0	1	0
Atrichum undulatum	B	3	.	.	.
Aulacomnium palustre	B	1	.	.	.
Brachythecium rutabulum	B	2	.	.	.
Breutelia chrysocoma	B	1	.	.	.
Calliergonella cuspidata	B	2	.	.	.
Calypogeia muelleriana	B	2	.	.	.
Campylopus atrovirens	B	1	.	.	.
Campylopus flexuosus	B	2	.	.	.
Campylopus introflexus	B	1	.	.	.
Campylopus paradoxus	B	2	.	.	.
Campylopus pyriformis	B	2	.	.	.
Cephalozia bicuspidata	B	2	.	.	.
Cratoneuron filicinum	B	2	.	.	.
Dicranella heteromalla	B	2	.	.	.
Dicranum scoparium	B	2	.	.	.
Didymodon spadiceus	B	1	.	.	.
Diplophyllum albicans	B	2	.	.	.
Eurhynchium striatum	B	3	.	.	.
Fissidens taxifolius	B	2	.	.	.
Hylocomium splendens	B	2	.	.	.
Hypnum andoi	B	3	.	.	.
Hypnum cupressiforme	B	2	.	.	.
Hypnum jutlandicum	B	2	.	.	.
Hypnum resupinatum	B	2	.	.	.
Isothecium myosuroides	B	3	.	.	.
Kindbergia praelonga	B	3	.	.	.
Leiocolea badensis	B	2	.	.	.
Lophocolea bidentata	B	2	.	.	.
Lophozia ventricosa	B	2	.	.	.
Marsupella emarginata	B	2	.	.	.
Metzgeria furcata	B	3	.	.	.
Mnium hornum	B	3	.	.	.
Nardia scalaris	B	2	.	.	.
Odontoschisma sphagni	B	1	.	.	.
Oligotrichum hercynicum	B	1	.	.	.
Pellia endiviifolia	B	2	.	.	.
Pellia epiphylla	B	2	.	.	.
Peltigera lactucifolia	B	2	.	.	.
Philonotis fontana	B	1	.	.	.
Plagiochila asplenioides	B	3	.	.	.
Plagiochila porelloides	B	2	.	.	.
Plagiothecium denticulatum	B	3	.	.	.
Plagiothecium succulentum	B	3	.	.	.
Plagiothecium undulatum	B	2	.	.	.
Pleurozium schreberi	B	2	.	.	.

Species	Taxon	Open / Woodland	C	S	R
Pogonatum aloides	B	2	.	.	.
Pogonatum nanum	B	1	.	.	.
Pogonatum urnigerum	B	1	.	.	.
Polytrichastrum formosum	B	3	.	.	.
Polytrichum commune	B	2	.	.	.
Polytrichum juniperinum	B	1	.	.	.
Pseudoscleropodium purum	B	2	.	.	.
Pseudotaxiphyllum elegans	B	3	.	.	.
Racomitrium aquaticum	B	2	.	.	.
Racomitrium lanuginosum	B	1	.	.	.
Rhytidiadelphus triquetrus	B	2	.	.	.
Rhytidiadelphus loreus	B	2	.	.	.
Rhytidiadelphus squarrosus	B	2	.	.	.
Riccardia chamedryfolia	B	2	.	.	.
Riccardia multifida	B	2	.	.	.
Scapania undulata	B	2	.	.	.
Sphagnum capillifolium	B	2	.	.	.
Sphagnum cuspidatum	B	1	.	.	.
Sphagnum denticulatum	B	2	.	.	.
Sphagnum fallax	B	2	.	.	.
Sphagnum fimbriatum	B	2	.	.	.
Sphagnum inundatum	B	2	.	.	.
Sphagnum palustre	B	2	.	.	.
Sphagnum papillosum	B	1	.	.	.
Sphagnum quinquefarium	B	3	.	.	.
Sphagnum subnitens	B	1	.	.	.
Sphagnum tenellum	B	1	.	.	.
Thuidium tamariscinum	B	3	.	.	.
Baeomyces rufus	L	2	.	.	.
Cladonia fimbriata	L	1	.	.	.
Cladonia gracilis	L	2	.	.	.
Cladonia polydactyla	L	2	.	.	.
Cladonia portentosa	L	1	.	.	.
Cladonia squamosa	L	2	.	.	.
Lepraria incana	L	2	.	.	.

Appendix 6 Epiphyte species recorded

Epiphyte species recorded in the survey plots. *Taxon* indicates: B - bryophyte, L - lichen, and V - vascular plant. *Edge* and *Interior* indicate the number of open space edge and interior trees the species occurred on respectively. Nomenclature follows Smith (2004b) for mosses, Paton (1999) for liverworts and Index Fungorum (2004) for lichens.

Species	Taxon	Edge	Interior
Atrichum undulatum	B	1	0
Calypogeia muelleriana	B	1	3
Campylopus introflexus	B	2	0
Campylopus sp.	B	0	1
Colura calyptrifolia	B	6	7
Cryphaea heteromalla	B	1	0
Daltonia splachnoides	B	1	1
Frullania dilatata	B	5	4
Frullania tamarisci	B	1	1
Hypnum andoi	B	3	1
Hypnum jutlandicum	B	12	12
Hypnum resupinatum	B	1	1
Isothecium alopecuroides	B	2	0
Kindbergia praelonga	B	5	8
Lejeunea cavifolia	B	1	4
Lejeunea ulicina	B	6	6
Lophocolea bidentata	B	7	5
Metzgeria fruticulosa	B	2	2
Metzgeria furcata	B	6	6
Metzgeria temperata	B	11	9
Mnium hornum	B	1	0
Plagiothecium laetum	B	1	1
Plagiothecium undulatum	B	5	5
Pseudotaxiphyllum elegans	B	1	0
Radula complanata	B	1	1
Rhytidiadelphus loreus	B	0	1
Thuidium tamariscinum	B	0	2
Ulota crispa agg.	B	10	9
Ulota phyllantha	B	2	0
Anisomeridium biforme	L	0	1
Anisomeridium polypori	L	0	1
Byssoloma subdiscordans	L	1	0
Candelariella reflexa	L	0	1
Cladonia chlorophaea	L	0	1
Cladonia sp.	L	1	1
Dimerella lutea	L	5	11
Dimerella pineti	L	5	7
Dimerella sp.	L	0	0
Evernia prunastri	L	2	1
Fellhanera bouteillei	L	1	3
Fuscidea lightfootii	L	12	10
Graphis elegans	L	2	1
Graphis scripta	L	1	0
Graphis sp.	L	1	0
Gyalideopsis anastomosans	L	4	7

Hypogymnia sp.	L	0	1
Hypogymnia tubulosa	L	7	5
Hypotrachyna revoluta	L	10	11
Hypotrachyna sp.	L	1	1
Lecania cyrtella	L	0	1
Lecanora chlarotera	L	0	1
Lecanora pulicaris	L	2	4
Lecidella elaeochroma	L	1	0
Lepraria incana	L	7	8
Melanelia fuliginosa	L	0	1
Micarea peliocarpa	L	1	5
Micarea lignaria	L	1	1
Micarea prasina	L	3	5
Micarea sp.	L	0	0
Parmelia sulcata	L	3	1
Parmotrema chinense	L	1	2
Phaeographis smithii	L	2	1
Physcia adscendens	L	0	1
Physcia aipolia	L	1	0
Physcia sp.	L	2	0
Physcia tenella	L	5	3
Pseudevernia furfuracea	L	1	0
Ramalina farinacea	L	6	3
Ramalina fastigiata	L	0	1
Ramalina sp.	L	0	1
Rinodina biloculata	L	0	1
Trapeliopsis flexuosa	L	0	3
Usnea esperantiana	L	1	1
Usnea filipendula	L	0	1
Usnea flammea	L	2	0
Usnea sp.	L	0	2
Xanthoria polycarpa	L	1	0
Juvenile pteridophyte	V	0	2

Appendix 7 Spider species recorded

The total number of individuals within each spider species and their habitat associations; and number of individuals in the open (centre of the open space) and forest (5m into the forest) sampling points on the transect. Nomenclature follows Roberts, 1993.

Species	Number of individuals			Habitat
	Open	Forest	Total	association
Agroeca proxima (O.P.-Cambridge, 1871)	5	0	7	Generalist
Agyneta conigera (O.P.-Cambridge, 1863)	0	1	10	Generalist
Agyneta decora (O.P.-Cambridge, 1871)	2	1	3	Generalist
Agyneta olivacea (Emerton, 1882)	30	2	66	Generalist
Agyneta ramosa (Jackson, 1912)	71	17	194	Generalist
Agyneta subtilis (O.P.-Cambridge, 1863)	142	41	448	Generalist
Alopecosa pulverulenta (Clerck, 1757)	49	0	50	Open
Antistea elegans (Blackwall, 1841)	6	0	6	Open
Aphileta misera (O.P.-Cambridge, 1882)	1	0	1	Open
Asthenargus paganus (Simon, 1884)	6	42	145	Forest
Bathyphantes gracilis (Blackwall, 1841)	78	13	146	Generalist
Bathyphantes nigrinus (Westring, 1851)	21	0	31	Generalist
Bathyphantes parvulus (Westring, 1851)	8	3	12	Generalist
Centromerita concinna (Thorell, 1875)	3	0	4	Generalist
Centromerus arcanus (O.P.-Cambridge, 1873)	0	0	3	Generalist
Centromerus dilutus (O.P.-Cambridge, 1875)	19	37	204	Generalist
Centromerus sylvaticus (Blackwall, 1841)	0	0	1	Open
Centromerus prudens (O.P.-Cambridge, 1873)	5	3	19	Generalist
Ceratinella brevipes (Westring, 1851)	11	4	32	Generalist
Ceratinella brevis (Wider, 1834)	0	0	1	Generalist
Clubiona reclusa (O.P.-Cambridge, 1863)	8	0	9	Generalist
Clubiona trivialis (C.L.Koch, 1843)	1	0	1	Generalist
Cnephalocotes obscurus (Simon, 1884)	2	0	3	Generalist
Cryphoeca sylvicola (C.L.Koch, 1834)	0	0	1	Generalist
Dicymbium nigrum (Blackwall, 1834)	11	0	12	Generalist
Dicymbium tibiale (Blackwall, 1836)	67	2	88	Generalist
Diplocephalus latifrons (O.P.-Cambridge, 1863)	52	332	943	Forest
Diplocephalus permixtus (O.P.-Cambridge, 1871)	3	1	6	Generalist
Diplostylor concolor (Wider, 1834)	0	1	3	Generalist
Dismodicus bifrons (Blackwall, 1841)	30	1	43	Generalist
Drassodes cupreus (Blackwall, 1834)	2	0	3	Open
Enoplognatha ovata (Clerck, 1757)	2	0	2	Open
Erigone atra (Blackwall, 1833)	10	0	10	Open
Erigone dentipalpis (Wider, 1843)	20	0	21	Open
Erigonella hiemalis (Blackwall, 1841)	11	0	20	Forest
Ero cambridgei (Kulczynski, 1911)	2	0	2	Generalist
Ero furcata (Villers, 1789)	0	0	1	Generalist
Gonatium rubens (Blackwall, 1833)	1	0	2	Open
Gongylidiellum vivum (O.P.-Cambridge, 1875)	38	18	134	Generalist
Gongylidiellum latebricola (O.P.-Cambridge, 1871)	0	0	2	Generalist
Gongylidum rufipes (Linnaeus, 1758)	1	0	1	Generalist
Hahnia nava (Blackwall, 1841)	1	0	2	Open
Haplodrassus signifier (C.L.Koch, 1839)	5	2	9	Generalist
Hilaira excisa (O.P.-Cambridge, 1871)	2	0	2	Generalist
Hypomma cornutum (Blackwall, 1833)	0	0	1	Forest
Lepthyphantes alacris (Blackwall, 1853)	110	122	584	Forest
Lepthyphantes cristatus (Menge, 1866)	10	0	10	Generalist
Lepthyphantes ericaeus (Blackwall, 1853)	37	13	156	Generalist

Lepthyphantes flavipes (Blackwall, 1854)	5	49	395	Forest
Lepthyphantes mengei (Kulczynski, 1887)	15	7	44	Generalist
Lepthyphantes obscurus (Blackwall, 1841)	5	6	48	Forest
Lepthyphantes pallidus (O.P.-Cambridge, 1871)	1	1	7	Generalist
Lepthyphantes tenebricola (Wider, 1834)	32	42	171	Forest
Lepthyphantes tenuis (Blackwall, 1852)	12	3	31	Generalist
Lepthyphantes zimmermanni (Bertkau, 1890)	97	227	940	Forest
Leptorhoptrum robustum (Westring, 1851)	0	0	10	Generalist
Lophomma punctatum (Blackwall, 1841)	3	0	4	Generalist
Macrargus rufus (Wider, 1834)	0	0	2	Forest
Maro minutus (O.P.-Cambridge, 1906)	10	22	158	Generalist
Maso sundervalli (Westring, 1851)	24	1	32	Generalist
Meioneta saxatilis (Blackwall, 1844)	61	1	71	Generalist
Meta mengei (Blackwall, 1869)	7	1	13	Generalist
Meta merianae (Scopli, 1763)	0	0	1	Cryptic
Meta segmentata (Clerck, 1757	0	0	3	Generalist
Metopobactrus prominulus (O.P.-Cambridge, 1872)	8	0	9	Generalist
Micaria pulicaria (Sundevall, 1832)	0	0	1	Open
Micrargus herbigradus (Blackwall, 1854)	16	7	47	Generalist
Microlinyphia pusilla (Sundevall, 1830)	2	0	2	Generalist
Microneta viaria (Blackwall, 1841)	0	0	3	Forest
Monocephalus casteneipes (Simon, 1884)	0	0	5	Generalist
Monocephalus fuscipes (Blackwall, 1836)	66	232	1183	Forest
Neon reticulates (Blackwall, 1853)	1	0	1	Generalist
Nereine clathrata (Sundevall, 1830)	3	0	5	Generalist
Neriene Montana (Clerck, 1757)	0	1	1	Forest
Neriene peltata (Wider, 1834)	3	3	18	Generalist
Nesticus cellulanus (Clerck, 1757)	0	0	1	Cryptic
Oedothorax fuscus (Blackwall, 1834)	1	0	1	Open
Oedothorax gibbosus (Blackwall, 1841)	96	1	112	Open
Oedothorax retusus (Blackwall, 1851)	8	0	8	Open
Oxyptila trux (Blackwall, 1846)	36	0	48	Generalist
Pachygnatha clercki (Sundevall, 1823)	5	0	7	Generalist
Pachygnatha degeeri (Sundevall, 1830)	47	0	48	Generalist
Pardosa amentata (Clerck, 1757)	10	0	13	Open
Pardosa nigriceps (Thorell, 1856)	52	0	58	Generalist
Pardosa palustris (Linnaeus, 1758)	1	0	1	Open
Pardosa pullata (Clerck, 1757)	521	0	531	Open
Pelecopsis nemoralis (Blackwall, 1841)	2	2	16	Forest
Pelecopsis parallela (Wider, 1834)	2	1	4	Generalist
Pepnocranium ludicrum (O.P.-Cambridge, 1861)	4	0	4	Generalist
Pholcomma gibbum (Westring, 1851)	3	0	5	Generalist
Pirata piraticus (Clerck, 1757)	17	1	21	Generalist
Pirata uliginosus (Thorell, 1856)	45	1	53	Open
Pocadicnemis juncea (Locket & Millidge, 1853)	9	0	11	Open
Pocadicnemis pumila (Blackwall, 1841)	300	3	390	Open
Poeciloneta globosa (Blackwall, 1841)	0	1	6	Open
Porrhomma campbelli (O.P.-Cambridge, 1894)	1	0	2	Cryptic
Porrhomma convexum (Westring,1861)	1	0	1	Cryptic
Porrhomma pallidum (Jackson, 1913)	5	42	154	Generalist
Porrhomma pygmaeum (Blackwall, 1834)	0	1	5	Generalist
Robertus arundineti (O.P.-Cambridge, 1871)	1	0	1	Generalist
Robertus lividus (Blackwall, 1836)	68	74	341	Generalist
Saaristoa abnormis (Blackwall, 1841)	32	65	243	Generalist
Saaristoa firma (O.P.-Cambridge, 1905)	3	6	18	Generalist
Silometopus elegans (O.P.-Cambridge, 1872)	41	2	46	Open

Tapinocyba pallens (O.P.-Cambridge, 1872)	3	2	27	Forest
Tapinocyba praecox (O.P.-Cambridge, 1873)	0	0	1	Generalist
Taranucnus setosus (Simon, 1884)	6	0	9	Generalist
Theonoe minutissima (O.P.-Cambridge, 1879)	7	13	105	Generalist
Theridion pallens (Blackwall, 1834)	0	0	2	Generalist
Tiso vegans (Blackwall, 1834)	34	0	48	Generalist
Trochosa spinipalpis (O.P.-Cambridge, 1895)	0	0	1	Generalist
Trochosa terricola (Thorell, 1836)	27	6	64	Generalist
Walckenaeria acuminata (Blackwall,1833)	21	19	100	Generalist
Walckenaeria antica (Wider, 1834)	1	0	1	Generalist
Walckenaeria atrobtibialis (O. P.-Cambridge, 1878)	5	0	13	Generalist
Walckenaeria cuspidate (Blackwall, 1833)	12	2	35	Generalist
Walckenaeria dysderoides (Wider, 1843)	4	14	80	Generalist
Walckenaeria nodosa (O.P.-Cambridge, 1873)	1	0	1	Generalist
Walckenaeria nudipalpis (Westring, 1851)	10	5	31	Generalist
Walckenaeria vigilax (Blackwall, 1851)	66	1	103	Generalist
Xysticus cristatus (Clerck, 1757)	4	0	4	Open
Zora spinimana (Sundevall, 1833)	3	0	4	Generalist
Total	2770	1521	9438	

Appendix 8 Hoverfly species recorded

	Number of sites[1]	Open space association				Anthropophobic
		Forest	Small open spaces	Large open spaces	Open scrub	
Arctophila superbiens (Muller), 1776	2			√		√
Baccha elongata (Fabricius), 1775	6		√			
Cheilosia albitarsis (Meigen), 1822	3			√	√	
Cheilosia bergenstammi Becker, 1894	2			√	√	
Cheilosia pagana (Meigen), 1822	1			√	√	
Cheilosia variabilis (Panzer), 1798	1					√
Chrysotoxum bicinctum (L.), 1758	10		√			
Chrysotoxum fasciatum (Muller), 1764	8		√			√
Dasysyrphus albostriatus (Fallen), 1817	6	√				
Dasysyrphus tricinctus (Fallen), 1817	3					√
Didea fasciata Macquart, 1834	1	√				√
Episyrphus balteatus (DeGeer), 1776	10	√				
Eristalis abusiva Collin, 1931	1			√		√
Eristalis arbustorum (L.), 1758	1			√		
Eristalis interrupta (Poda), 1761	8		√			
Eristalis intricaria (L.), 1758	2			√		√
Eristalis lineata (Harris), 1776	3			√		
Eristalis pertinax (Scopoli), 1763	9		√			
Eristalis tenax (L.), 1758	3			√		
Eupeodes bucculatus (Rondani), 1857	2					√
Eupeodes corollae (Fabricius), 1794	4			√	√	
Eupeodes latifasciatus (Macquart), 1829	5			√		
Eupeodes luniger (Meigen), 1822	6			√	√	
Helophilus hybridus Loew, 1846	5			√		√
Helophilus pendulus (L.), 1758	10		√			
Lapposyrphus lapponicus (Zetterstedt), 1838	1	√				√
Lejogaster metallina (Fabricius), 1781	1			√		
Leucozona glaucia (L.), 1758	1					
Leucozona lucorum (L.), 1758	7		√			
Melangyna arctica (Zetterstedt), 1838	1					√

	Number of sites[1]	Open space association				Anthropophobic
		Forest	Small open spaces	Large open spaces	Open scrub	
Melangyna lasiophthalma (Zetterstedt), 1843	2	√				
Melanogaster hirtella (Loew), 1843	7			√		
Melanostoma mellinum (L.), 1758	10		√			
Melanostoma scalare (Fabricius), 1794	10		√			
Meligramma cincta (Fallen), 1817	0[2]					√
Meliscaeva auricollis (Meigen), 1822	10	√				
Meliscaeva cinctella (Zetterstedt), 1843	10	√				
Myathropa florea (L.), 1758	1					
Neoascia podagrica (Fabricius), 1775	5			√		
Neoascia tenur (Harris), 1780	2			√		
Paragus haemorrhous Meigen, 1822	3			√	√	√
Parasyrphus punctulatus (Verrall), 1873	1	√				√
Pipiza austriaca Meigen, 1822	0[2]				√	
Pipizella viduata (L.), 1758	3			√	√	
Platycheirus albimanus (Fabricius), 1781	10		√			
Platycheirus amplus Curran, 1927	1			√		√
Platycheirus angustatus (Zetterstedt), 1843	8			√		√
Platycheirus clypeatus (Meigen), 1822	10			√		
Platycheirus granditarsus (Forster), 1771	10			√		
Platycheirus manicatus (Meigen), 1822	7			√		
Platycheirus nielseni Vockeroth, 1990	10		√			√
Platycheirus occultus Goeldlin, Maibach & Speight, 1990	9			√		√
Platycheirus ramsarensis Goeldlin, Maibach & Speight, 1990	2			√		√
Platycheirus rosarum (Fabricius), 1787	2			√		

	Number of sites[1]	Open space association				Anthropophobic
		Forest	Small open spaces	Large open spaces	Open scrub	
Platycheirus scambus (Staeger), 1843	1			√		√
Platycheirus scutatus (Meigen), 1822	10		√			
Rhingia campestris Meigen, 1822	9			√		
Scaeva pyrastri (L.), 1758	2			√	√	
Scaeva selenitica (Meigen), 1822	10		√			√
Sericomyia lappona (L.), 1758	10		√			√
Sericomyia silentis (Harris), 1776	10		√			
Sphaerophoria fatarum Goeldlin, 1989	5			√		√
Sphaerophoria philantha (Meigen), 1822	7			√		√
Sphegina clunipes (Fallen), 1816	10	√				√
Syritta pipiens (L.), 1758	0[2]			√		
Syrphus ribesii (L.), 1758	10	√				
Syrphus torvus Osten-Sacken, 1875	10	√				
Syrphus vitripennis Meigen, 1822	6	√				
Volucella bombylans (L.), 1758	6		√			
Volucella pellucens (L.), 1758	4				√	
Xanthandrus comtus (Harris), 1780	1					√
Xylota florum (Fabricius), 1805	1					√
Xylota jakutorum Bagatshanova, 1980	10	√				√
Xylota segnis (L.), 1758	10	√				
Xylota sylvarum (L.), 1758	2					√

[1] Number of sites recorded, excluding the two sites with incomplete samples.

[2] Recorded in one of the two sites with incomplete samples.

FIGURES

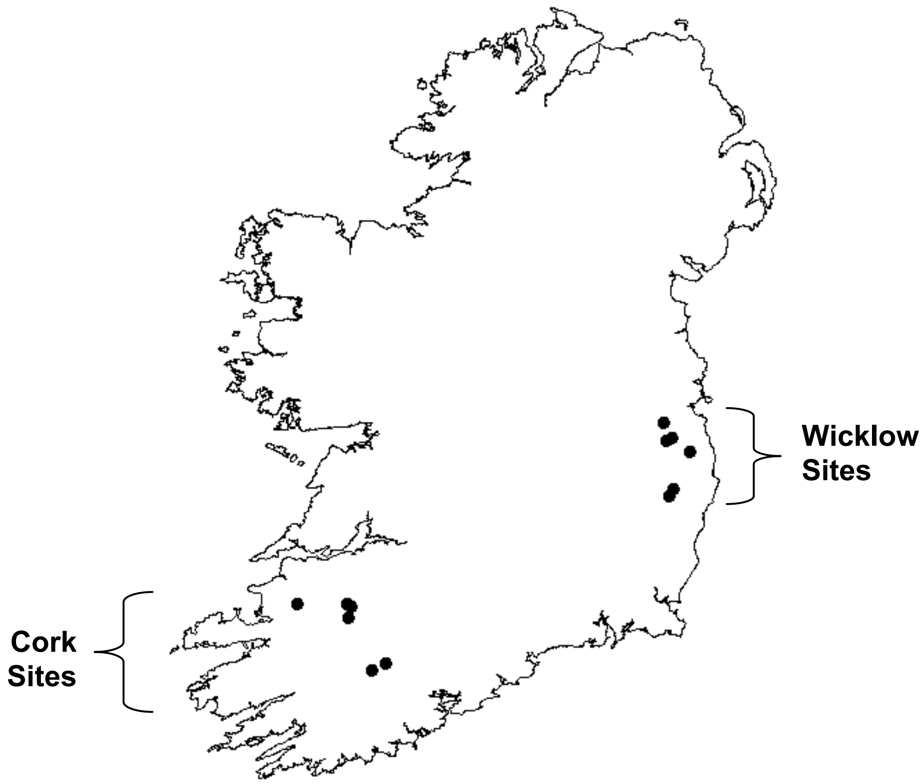

Figure 1. Location of the 12 study sites.

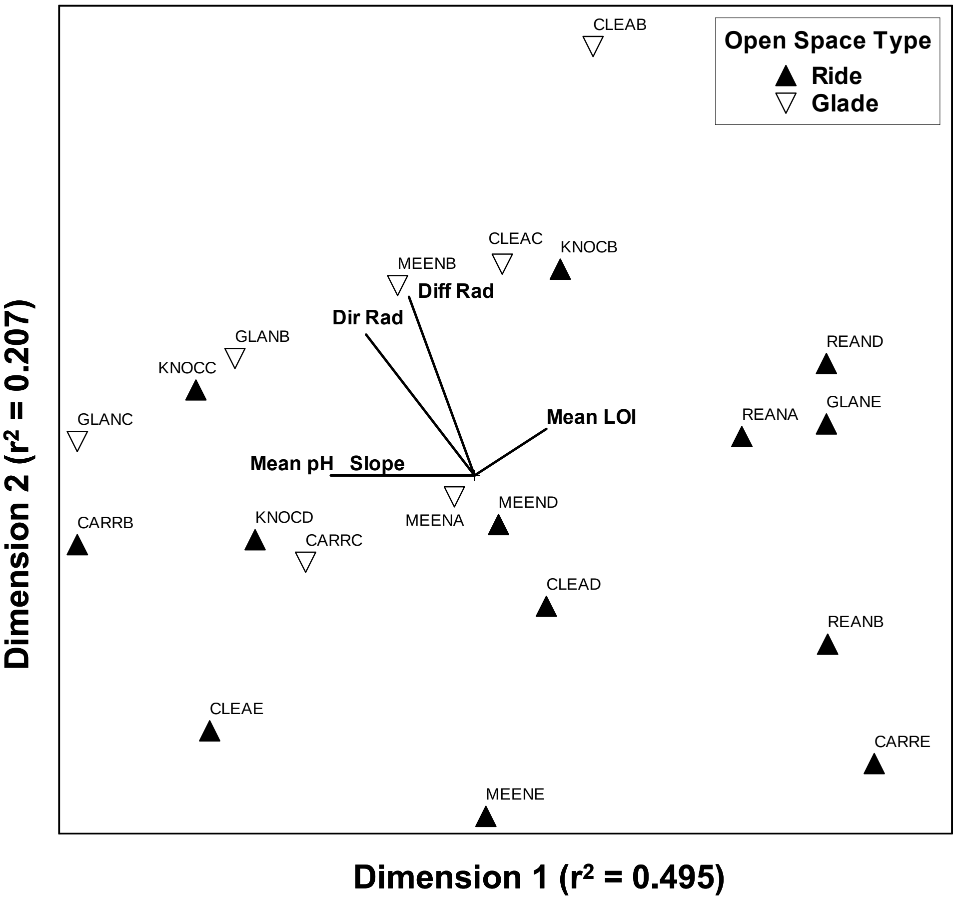

Figure 2. Joint plot of NMS ordination of species presence data in Cork glades and rides (3-D solution: stress = 13.12, p = 0.01). The r^2 values indicate the amount of variance in the original data explained by the ordination axes. The five environmental variables shown by vectors in the joint plot are: *Dir Rad*- transmitted direct radiance, calculated from the open space centre hemiphoto; *Diff Rad*- transmitted diffuse radiance, calculated from the open space centre hemiphoto; *Mean LOI*- mean loss-on-ignition from plot samples; *Slope*- slope in degrees; *Mean pH*- mean pH from plot samples.

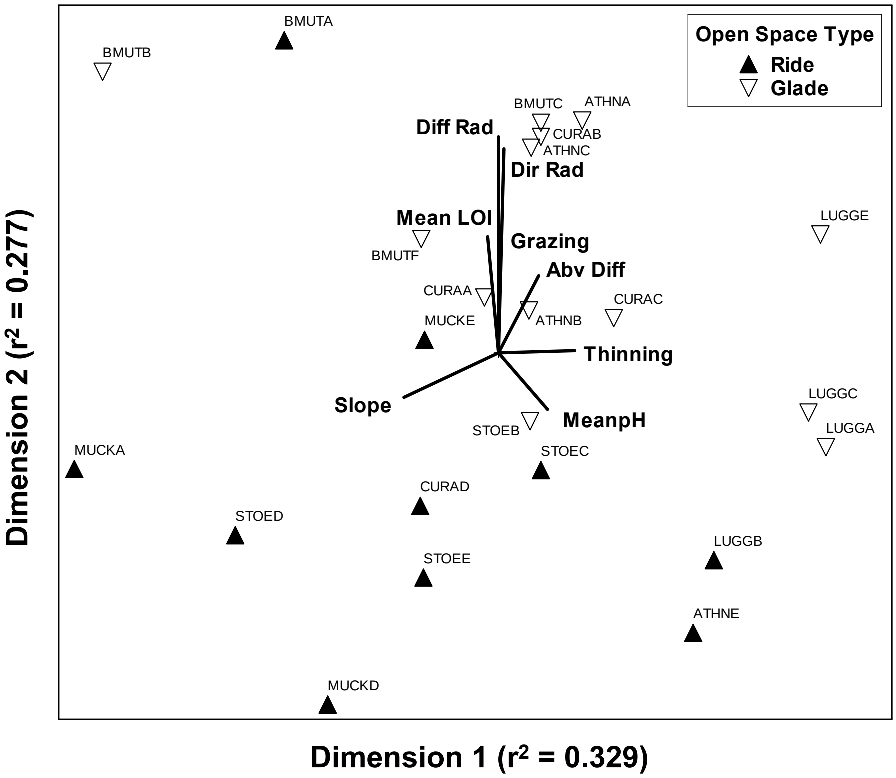

Figure 3. Joint plot of NMS ordination of species presence data in Wicklow glades and rides (3-D solution: stress = 14.04, p = 0.01). The r^2 values indicate the amount of variance in the original data explained by the ordination axes. The eight environmental variables shown by vectors in the joint plot are: *Dir Rad*- transmitted direct radiance, calculated from the open space centre hemiphoto; *Diff Rad*- transmitted diffuse radiance, calculated from the open space centre hemiphoto; *Mean LOI*- mean loss-on-ignition from plot samples; *Slope*- slope in degrees; *Mean pH*- mean pH from plot samples; *Grazing*- grazing intensity estimated on a 0-3 scale; *Abv Diff*- above-canopy diffuse radiance, calculated from the open space centre hemiphoto; *Thinning*- thinning intensity estimated on a 0-3 scale.

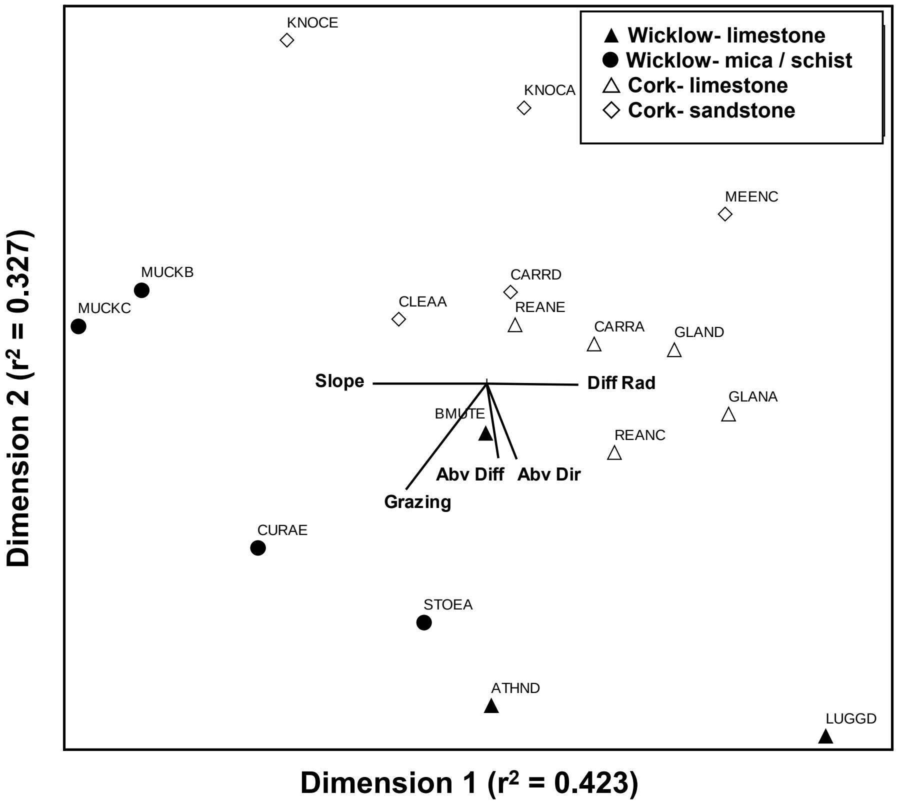

Figure 4. Joint plot of NMS ordination of species presence data in the roads (2-D solution: stress = 14.96, p = 0.01). The r^2 values indicate the amount of variance in the original data explained by the ordination axes. The road surface material is indicated by different symbols; Wicklow sites are shown with filled symbols and Cork sites with open symbols. The five environmental variables shown by vectors in the joint plot are: *Diff Rad*- transmitted diffuse radiance, calculated from the open space centre hemiphoto; *Slope*- slope in degrees; *Grazing*- grazing intensity estimated on a 0-3 scale; *Abv Diff*- above-canopy diffuse radiance, calculated from the open space centre hemiphoto; *Abv Dir*- above-canopy direct radiance.

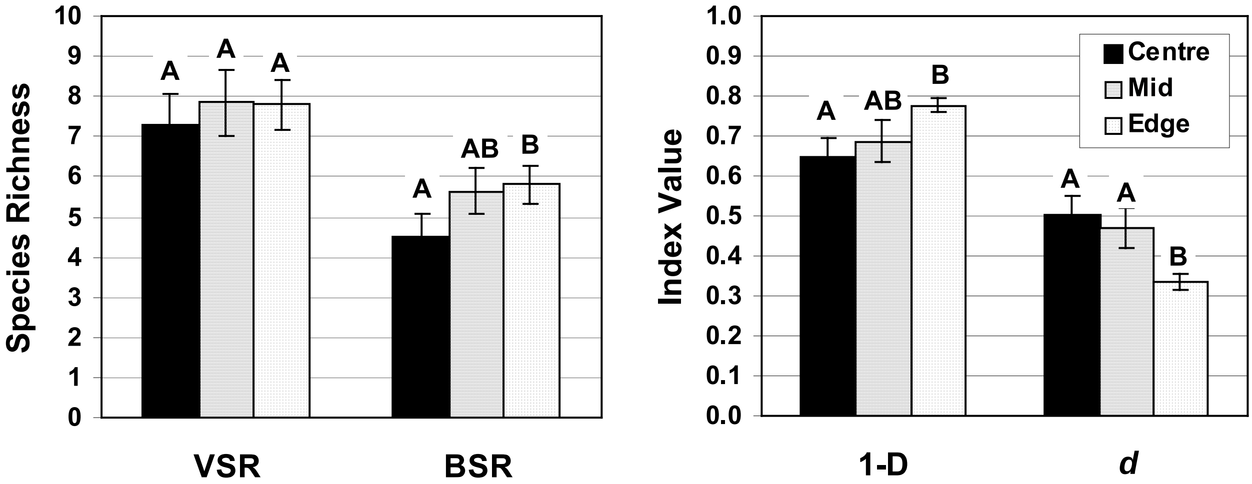

Figure 5. Mean vascular plant species richness (VSR), bryophyte and lichen species richness (BSR), Simpson's diversity (1-D) and Berger-Parker evenness index (*d*) in 4 m^2 plots in the centre, middle and edge of glades. Plot location means with the same letter are not significantly different according to paired t-tests.

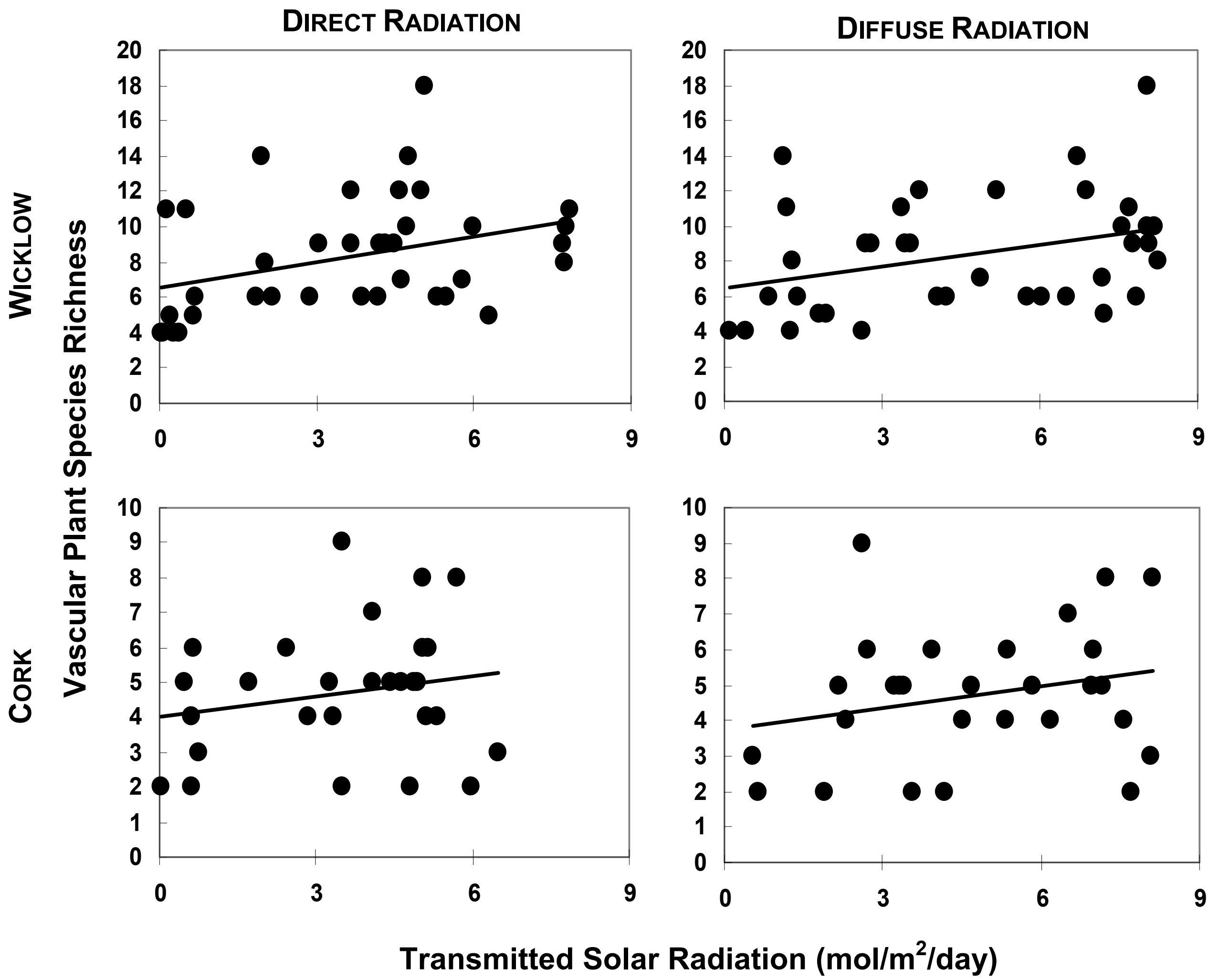

Figure 6. Linear regressions of vascular species richness on transmitted solar radiation in 4m^2 plots in glades and rides. a) Direct radiation vs species richness in Wicklow glades and rides ($r^2 = 0.13$, $p = 0.029$). b) Diffuse radiation vs species richness in Wicklow glades and rides ($r^2 = 0.12$, $p = 0.034$). c) Direct radiation vs species richness in Cork glades and rides ($r^2 = 0.04$, $p = 0.309$). d) Diffuse radiation vs species richness in Cork glades and rides ($r^2 = 0.06$, $p = 0.199$).

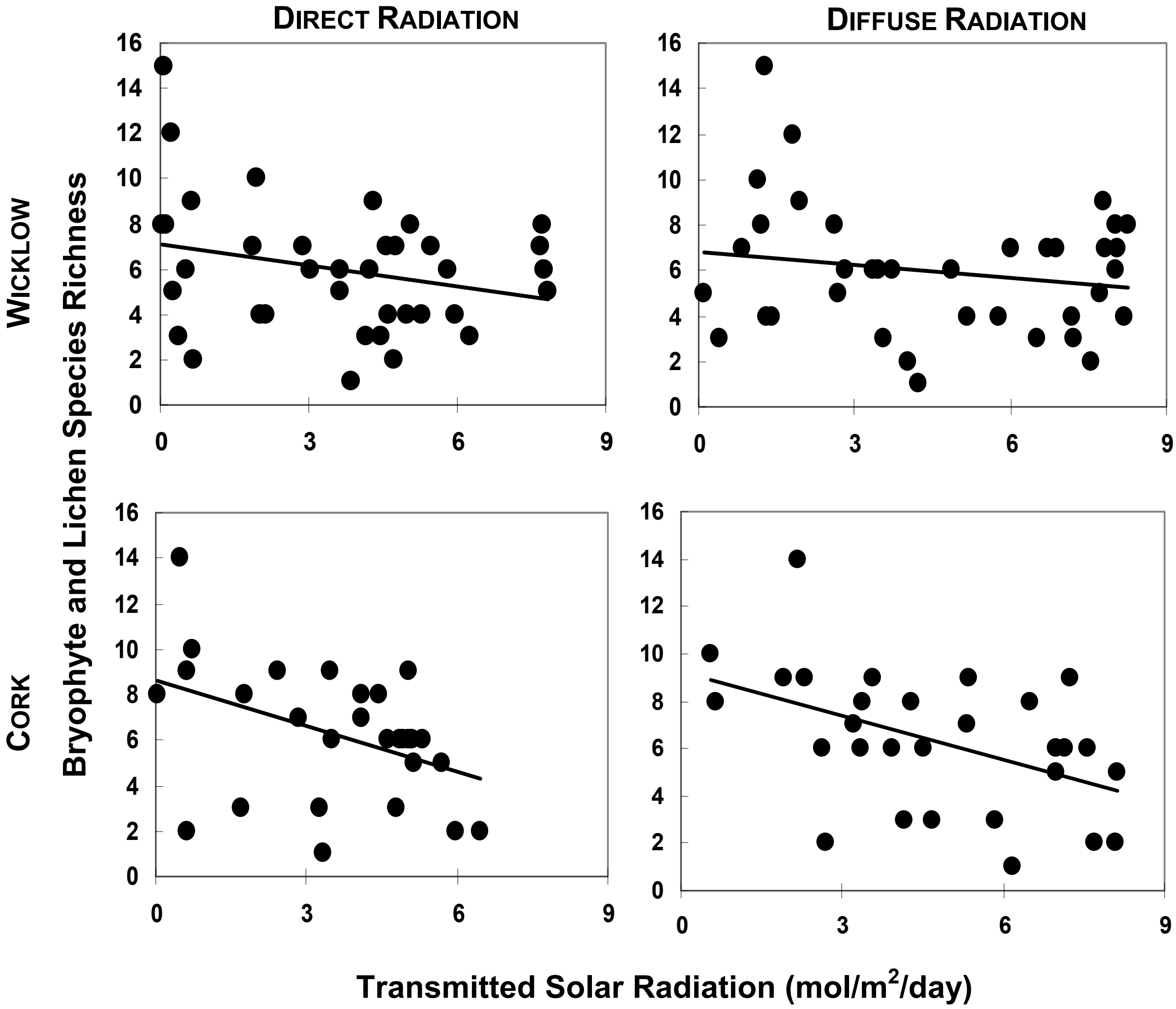

Figure 7. Linear regressions of bryophyte and lichen species richness on transmitted solar radiation in 4m² plots in glades and rides. a) Direct radiation vs species richness in Wicklow glades and rides ($r^2 = 0.07$, $p = 0.106$). b) Diffuse radiation vs species richness in Wicklow glades and rides ($r^2 = 0.03$, $p = 0.273$). c) Direct radiation vs species richness in Cork glades and rides ($r^2 = 0.19$, $p = 0.017$). d) Diffuse radiation vs species richness in Cork glades and rides ($r^2 = 0.23$, $p = 0.009$).

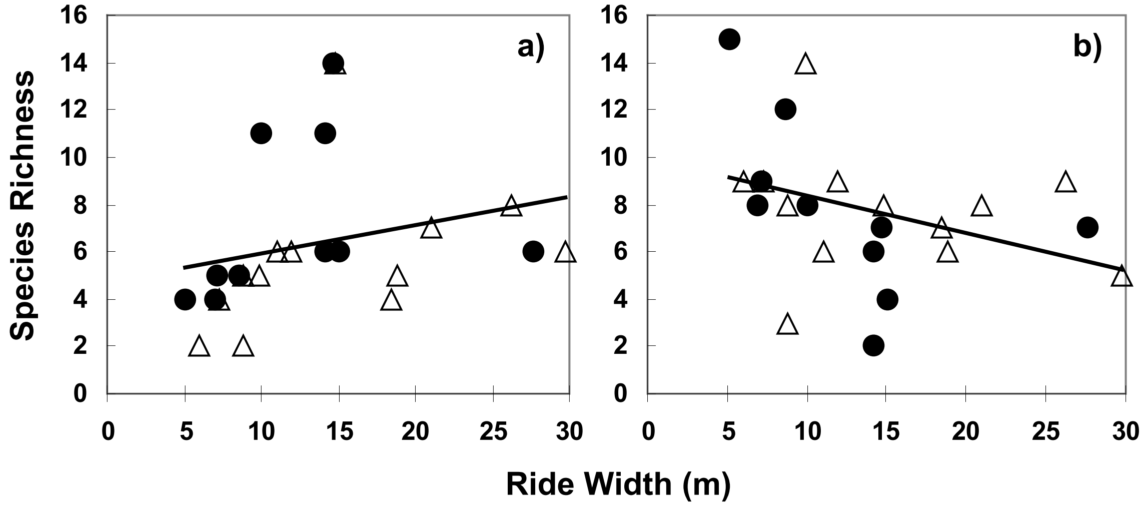

Figure 8. Scatter plots of a) vascular plant species richness and b) bryophyte and lichen species richness in ride centre plots against ride width. ● = Wicklow and △ = Cork. No relationships are statistically significant.

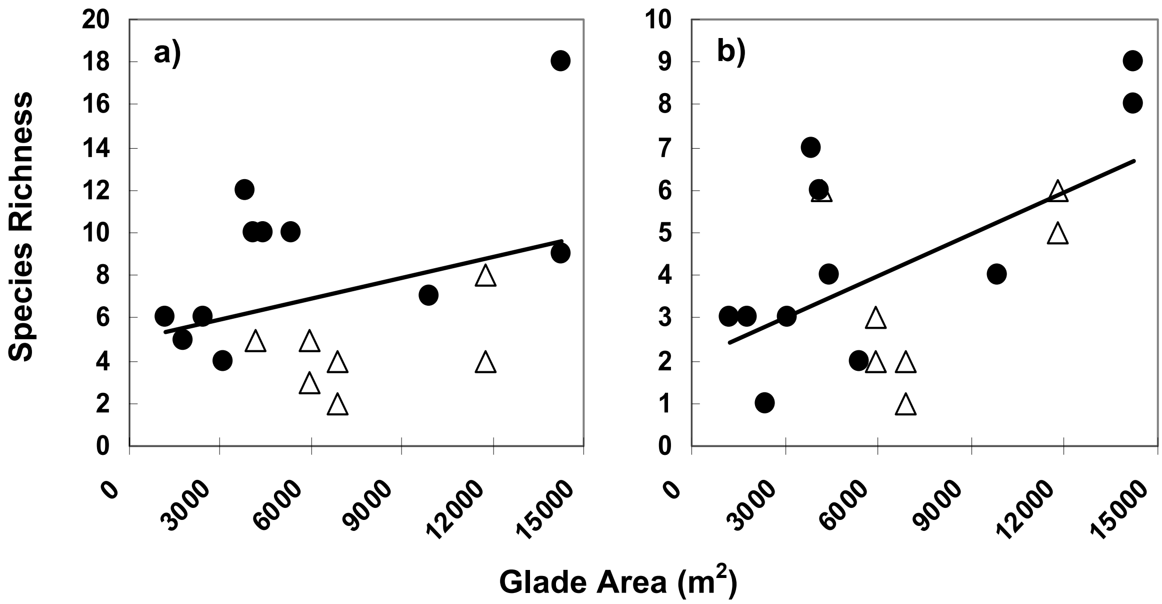

Figure 9. Scatter plots of a) vascular plant species richness and b) bryophyte and lichen species richness in glade centre plots against glade area. ● = Wicklow and △ = Cork. Bryophyte and lichen species richness is significantly predicted by glade area ($r^2 = 0.33$, $p = 0.013$).

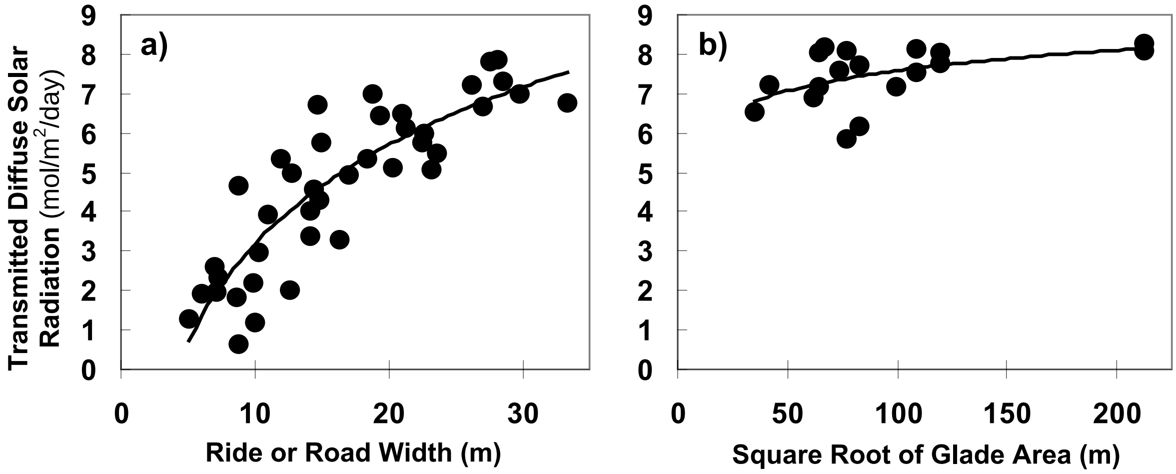

Figure 10. Regressions of transmitted diffuse radiation on: a) ride and road width (m) and b) glade area square root transformed (m). Logarithmic regressions were fitted to both sets of data: a) y = 3.63(ln(x)) - 5.17, $r^2 = 0.75$, $p \leq 0.0001$; b) y = 0.75(ln(x)) + 4.15, $r^2 = 0.23$, $p = 0.043$.

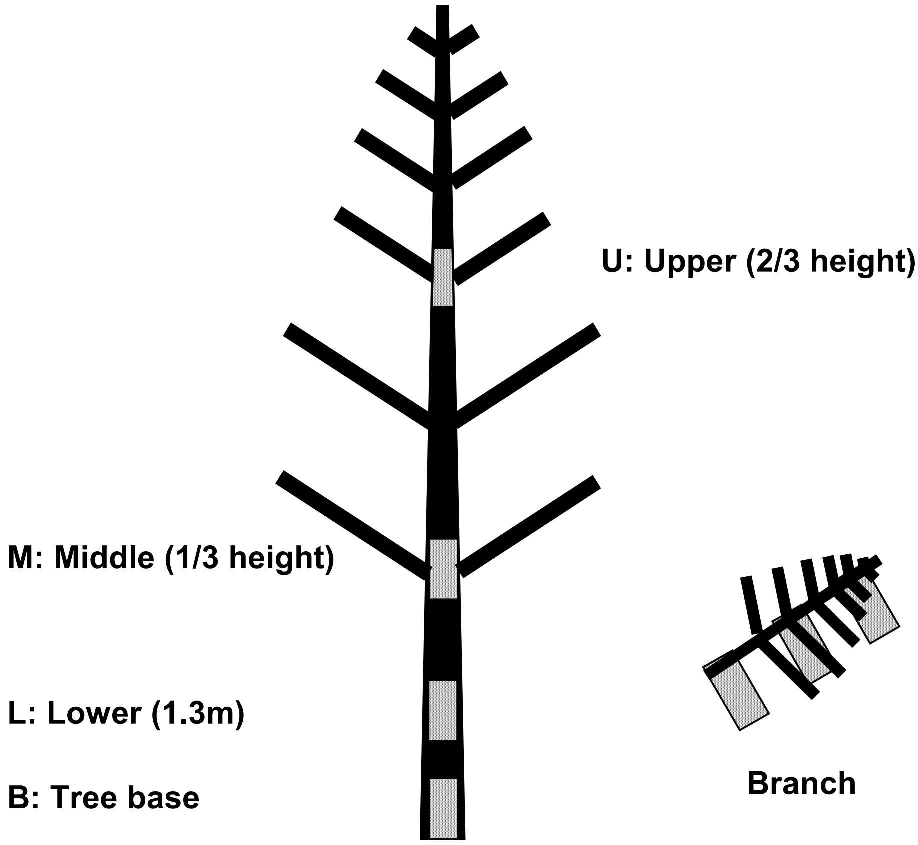

Figure 11. The plot design used for epiphyte sampling.

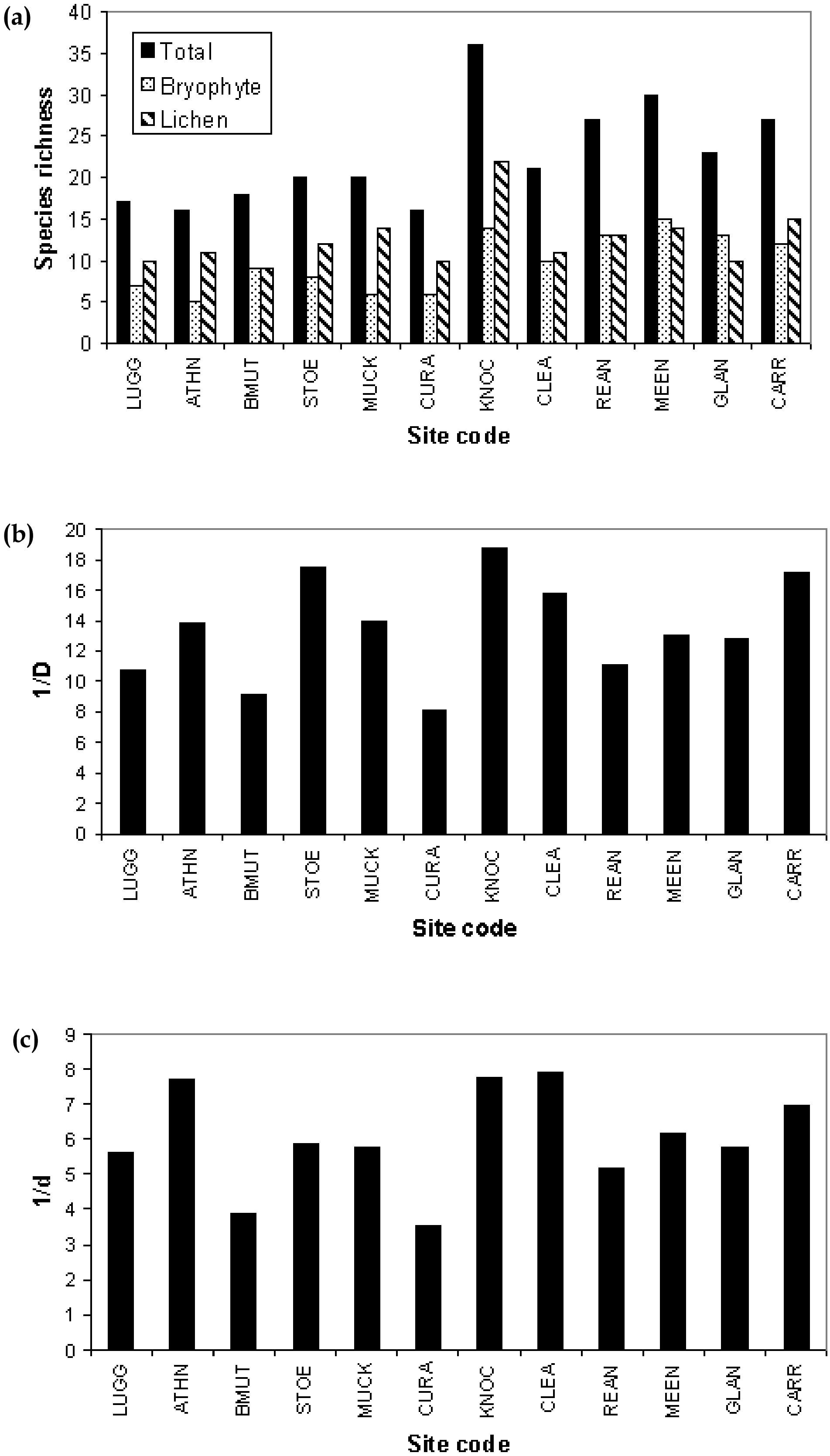

Figure 12. (a) Total, bryophyte and lichen species richness (b) Simpson's Diversity and (c) Berger-Parker Evenness at the 12 sites.

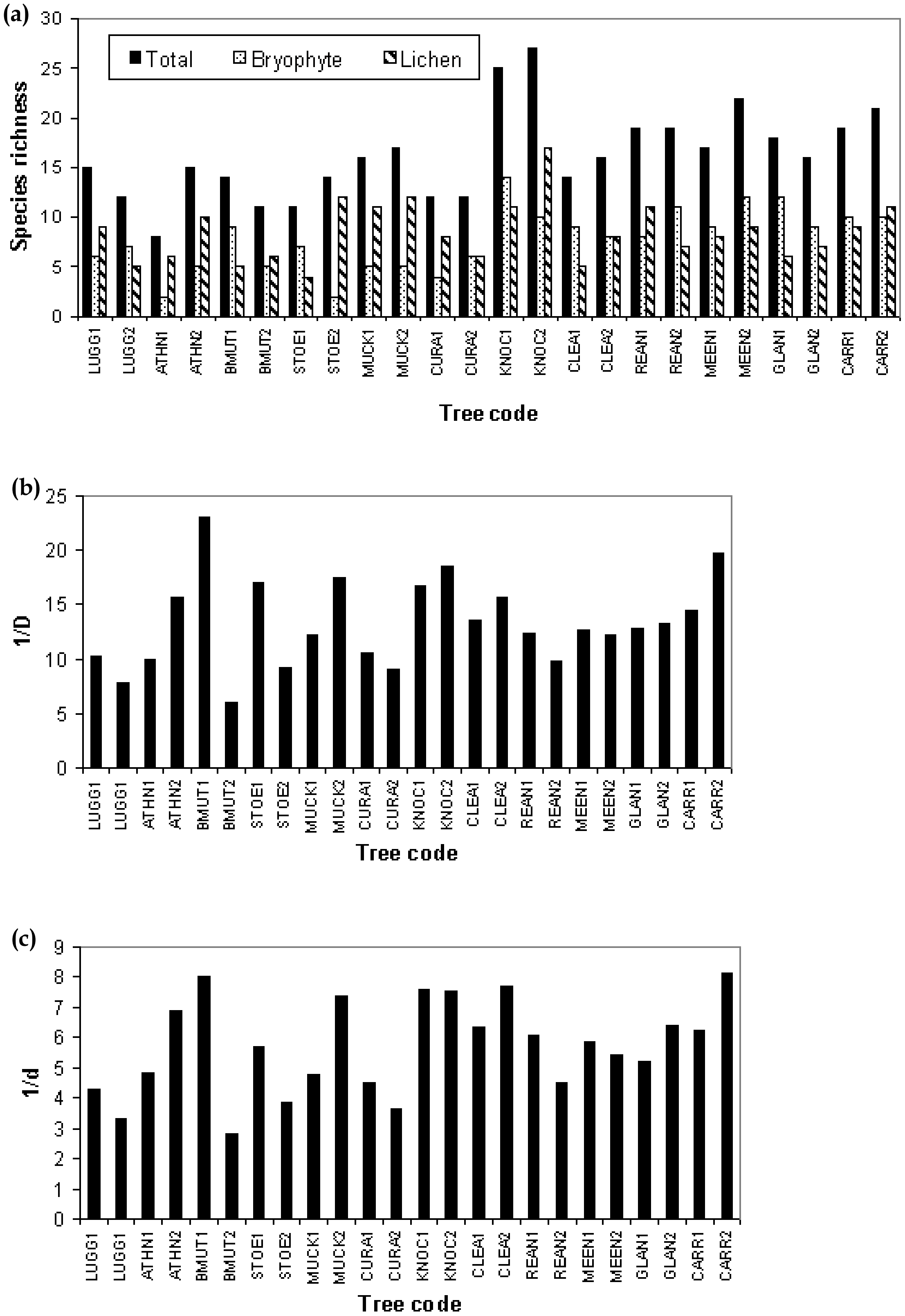

Figure 13. (a) Total, bryophyte and lichen species richness, (b) Simpson's Diversity and (c) Berger-Parker Evenness of the 24 trees studied

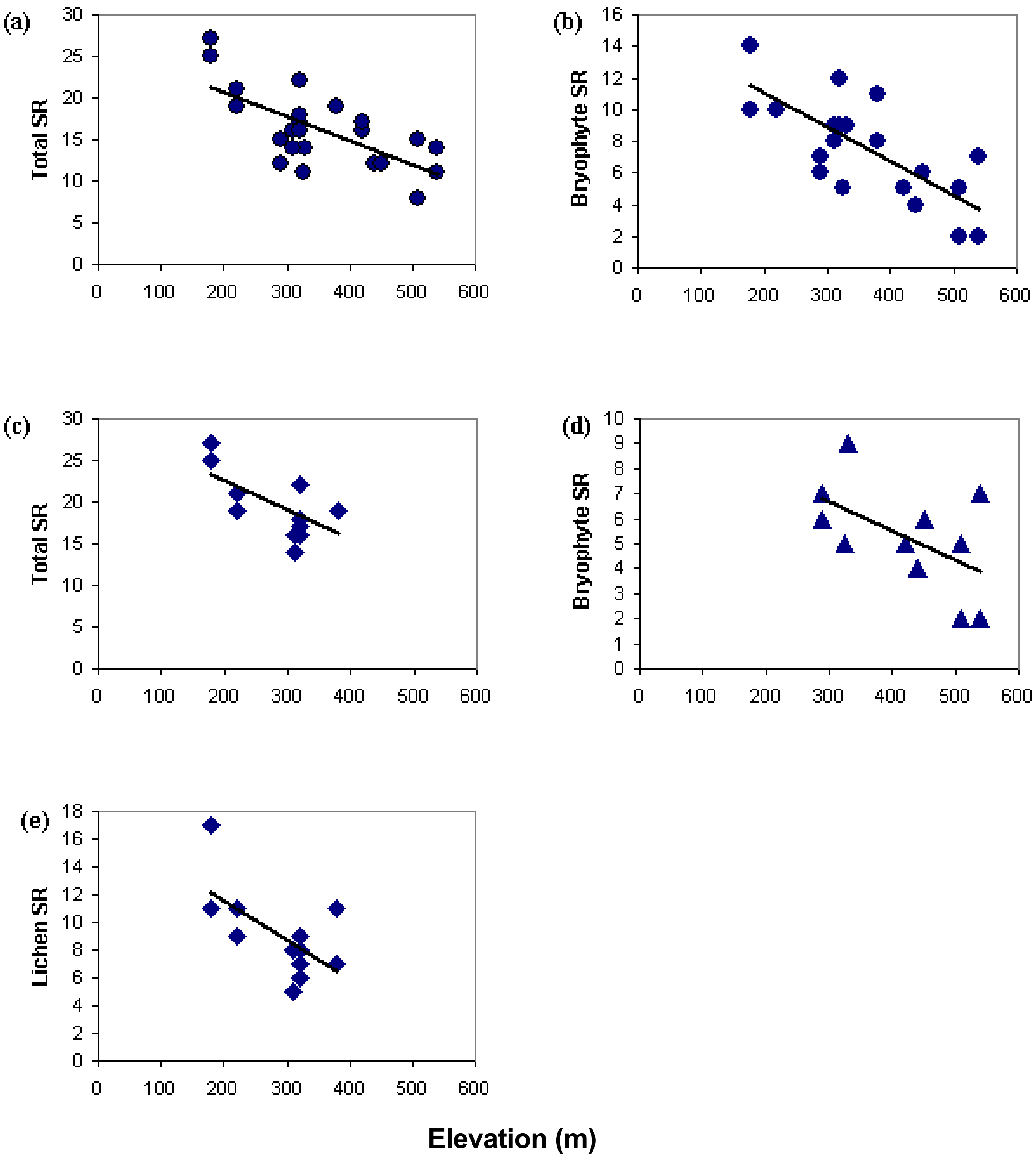

Figure 14. Linear regressions of epiphyte species richness on elevation. a) Elevation vs total species richness for all trees (r^2=0.459, p<0.001). b) Elevation vs bryophyte species richness for all trees (r^2=0.540, p<0.001). c) Elevation vs total species richness for the Cork cluster trees (r^2=0.426, p=0.021). d) Elevation vs bryophyte species richness for the Wicklow cluster trees (r^2=0.300 p= 0.065). (e) Elevation vs lichen species richness for the Cork cluster trees (r^2=0.426 p=0.033)

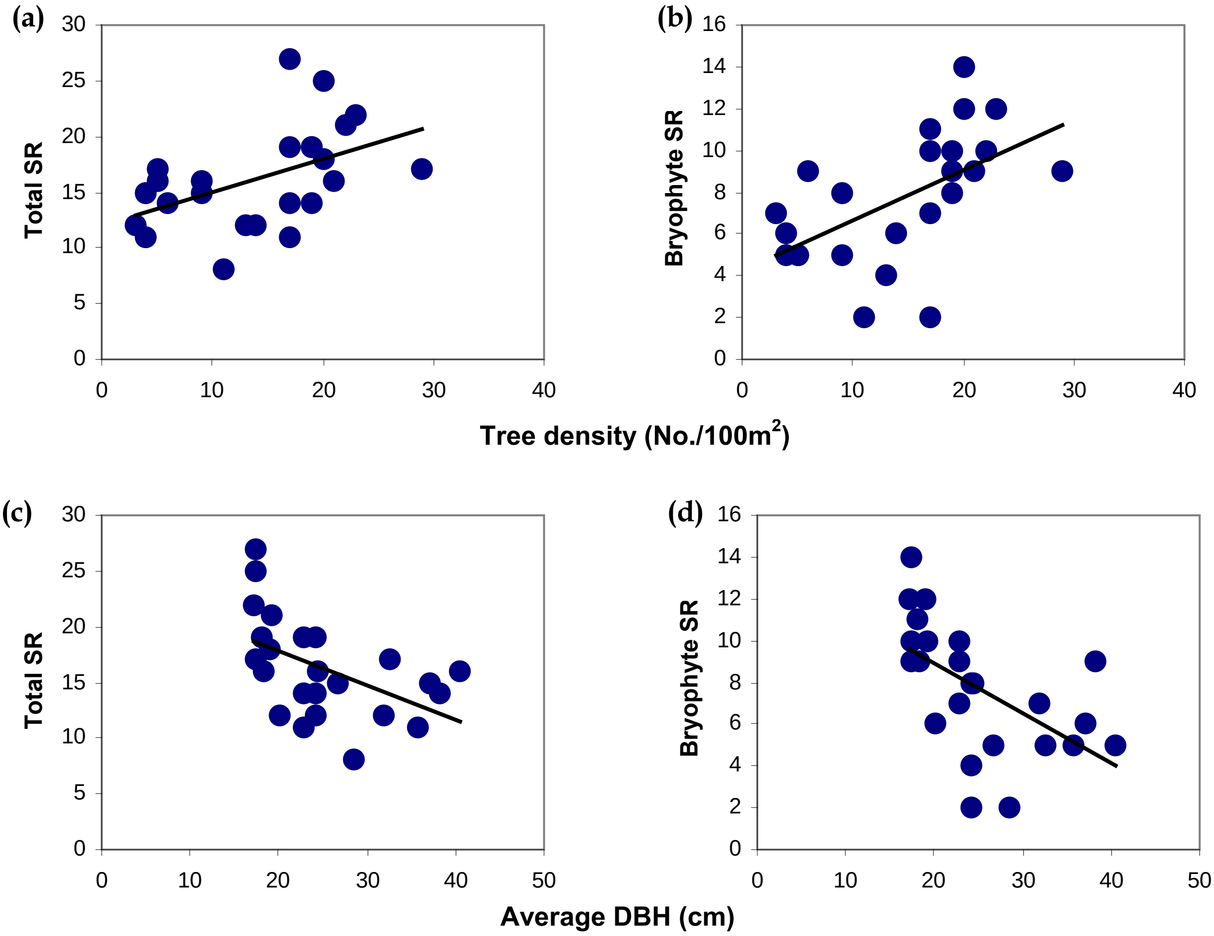

Figure 15. Linear regressions of epiphyte species richness on tree density and average basal area recorded in the 10m x 10m plots surrounding each tree. a) Tree density vs total species richness (r^2=0.459, p<0.001). b) Tree density vs bryophyte species richness (r^2=0.308, p=0.005). c) Average DBH vs total species richness (r^2=0.247, p=0.013). d) Average DBH vs bryophyte species richness (r^2=00.313, p=0.004).

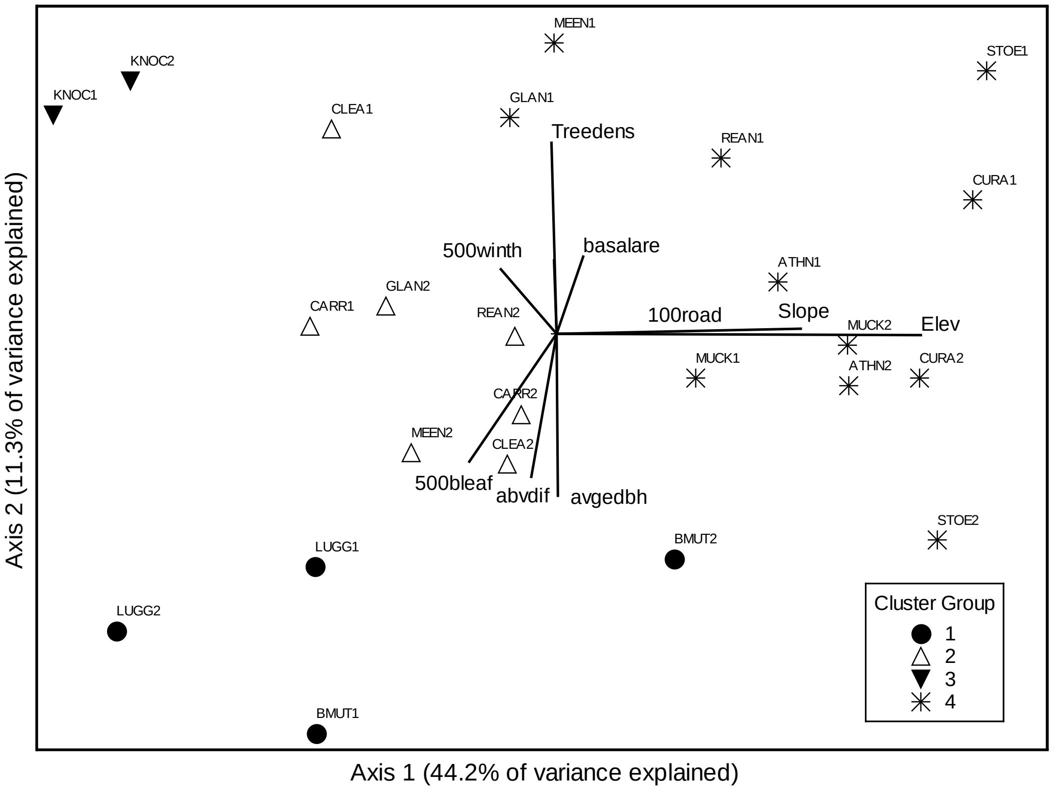

Figure 16. Joint plot of NMS ordination of species presence/absence data (3-D solution: stress = 17.34, p = 0.07). The point cloud has been rotated to maximise the variation explained by Elevation on dimension 1. Variables that have an r^2>0.15 for both axes are shown: *Elev* - Elevation of the sites, *Slope* – slope in degrees, *Treedens* - the number of trees in a 10m x 10m plot surrounding each tree, *avgedbh* – the average DBH of trees occurring in a 10m x 10m plot surrounding each tree, *500bleaf* – area of broadleaf scrub within 500m, *abvdif* – diffuse radiation incident (mols/m^2 *day) at the centre of the open space without effect of canopy, *500winth* – area of windthrow within 500m, *basalare* – the basal area in m^2 of trees in the 10m x 10m plot surrounding each tree.

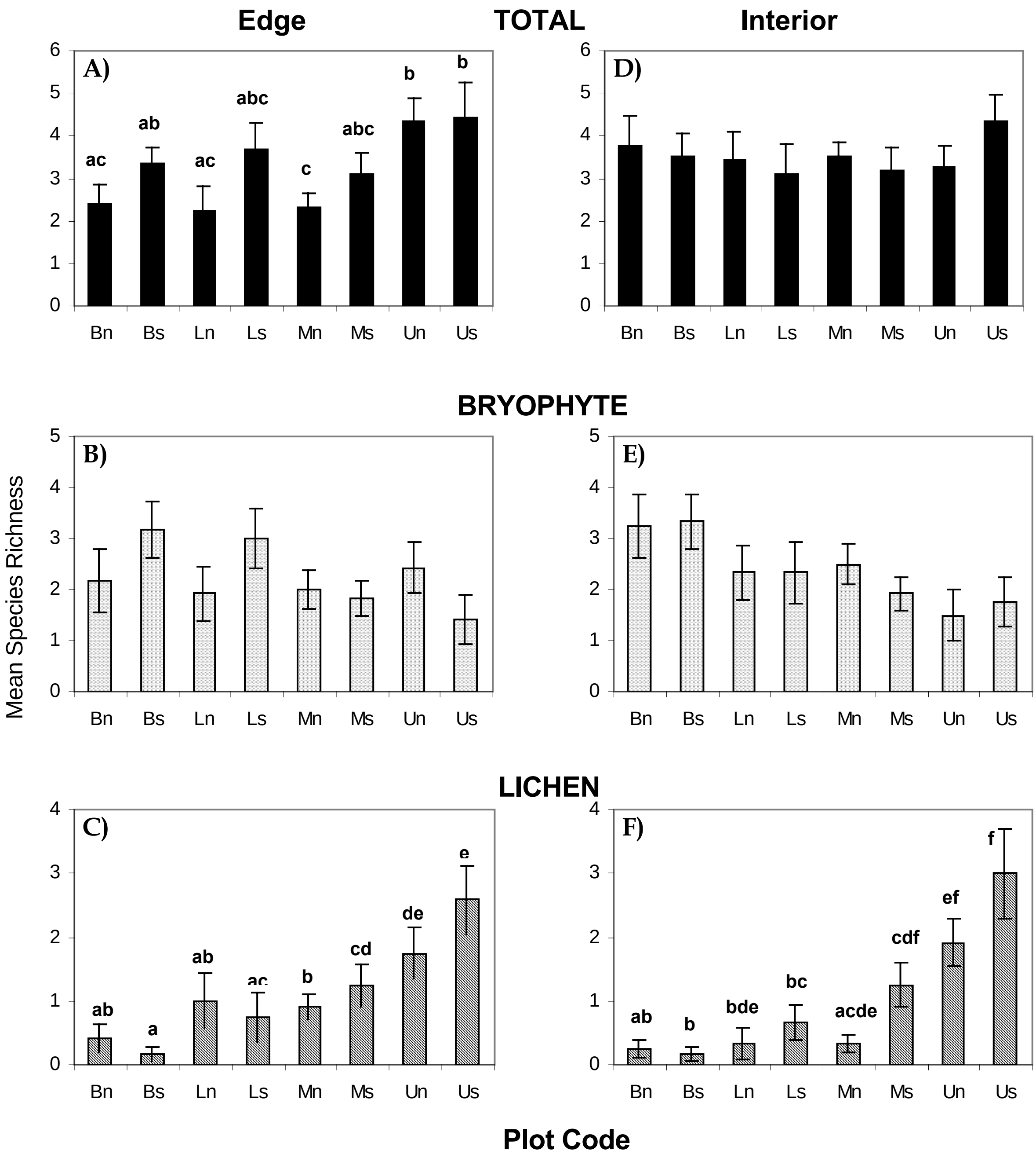

Figure 17. The mean (A),(D) total, (B),(E) bryophyte, and (C),(F) lichen species richness in the trunk plots for the open space edge (A,B,C) and forest interior (D,E,F) trees. (B = Tree Base, L = Lower, M = Middle, U = Upper, n = north aspect, s = south aspect). There is no significant difference between bars indicated by the same letter (a, b ,c, d, e or f).

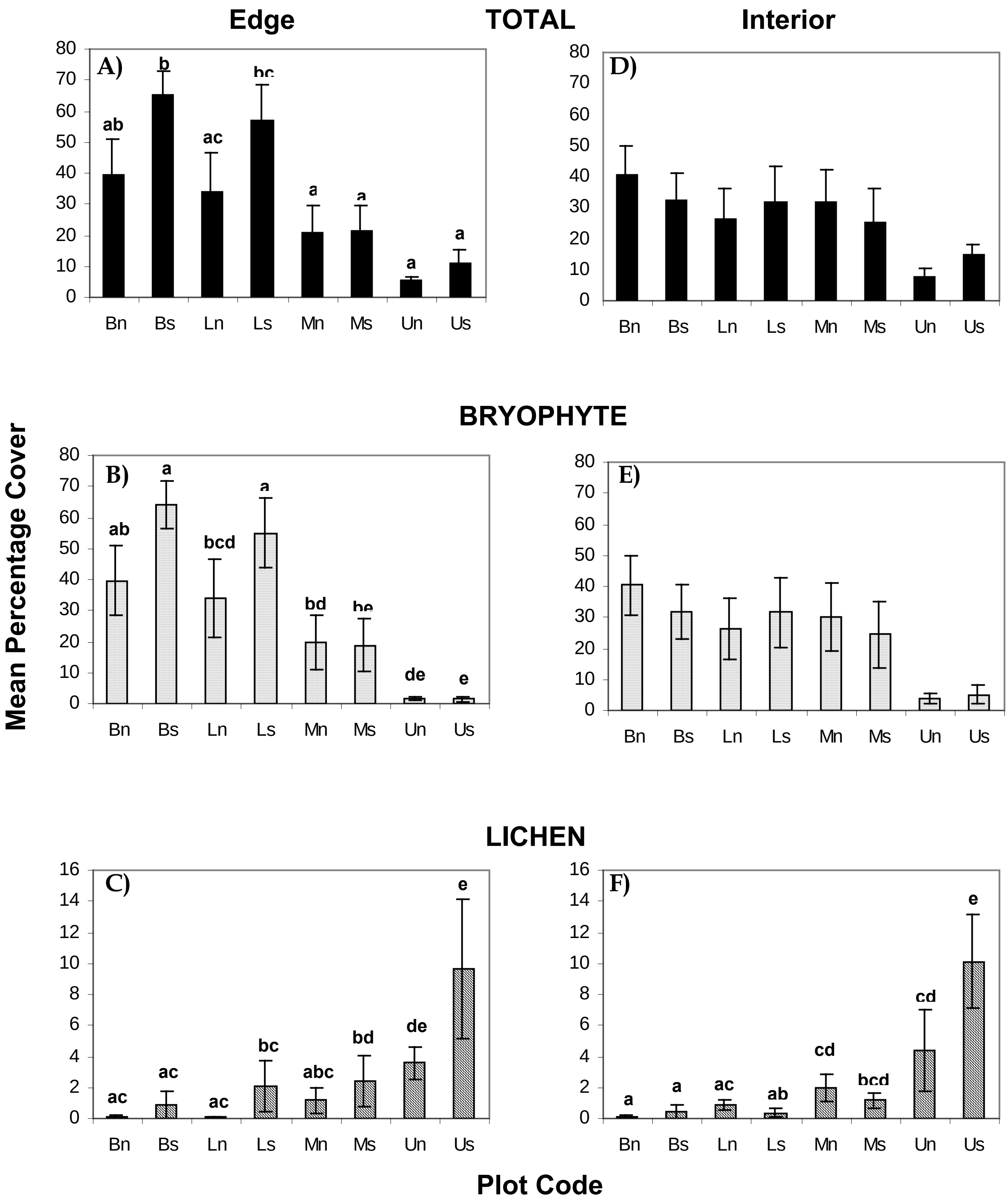

Figure 18. The mean (A), (D) total, (B), (E) bryophyte, and (C), (F) lichen percentage cover in the trunk plots for the open space edge (A, B, C) and forest interior (D, E, F) trees. (B = Tree Base, L = Lower, M = Middle, U = Upper, n = north aspect, s = south aspect). There is no significant difference between bars indicated by the same letter (a, b, c, d, or e)

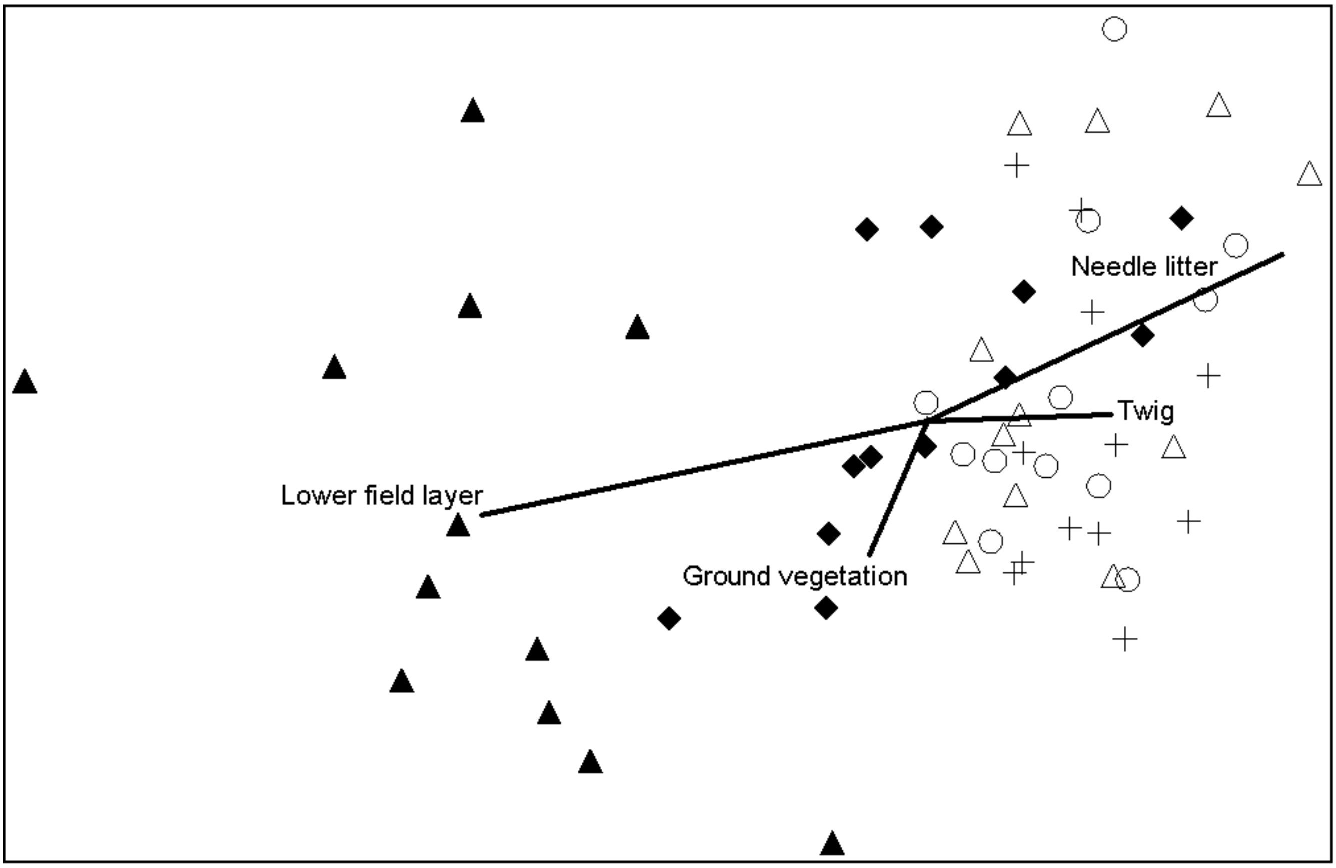

Figure 19. NMS ordination of spider assemblages (mean relative abundance per site) along the Open to Forest transect: % = Open (centre of the open space); ∀ = Open-boundary (2m into the open space from the boundary); + = Boundary (tree base);) = Forest-boundary (2m into the forest); 2 = Forest (5m into the forest). Final stress = 16.32; Final instability = 0.0005; Axis 1 r^2 = 0. 50; Axis 2 r^2 = 0.35. Cover of habitat variables that have a Pearson correlation (r) of >0.1 for both axes are shown.

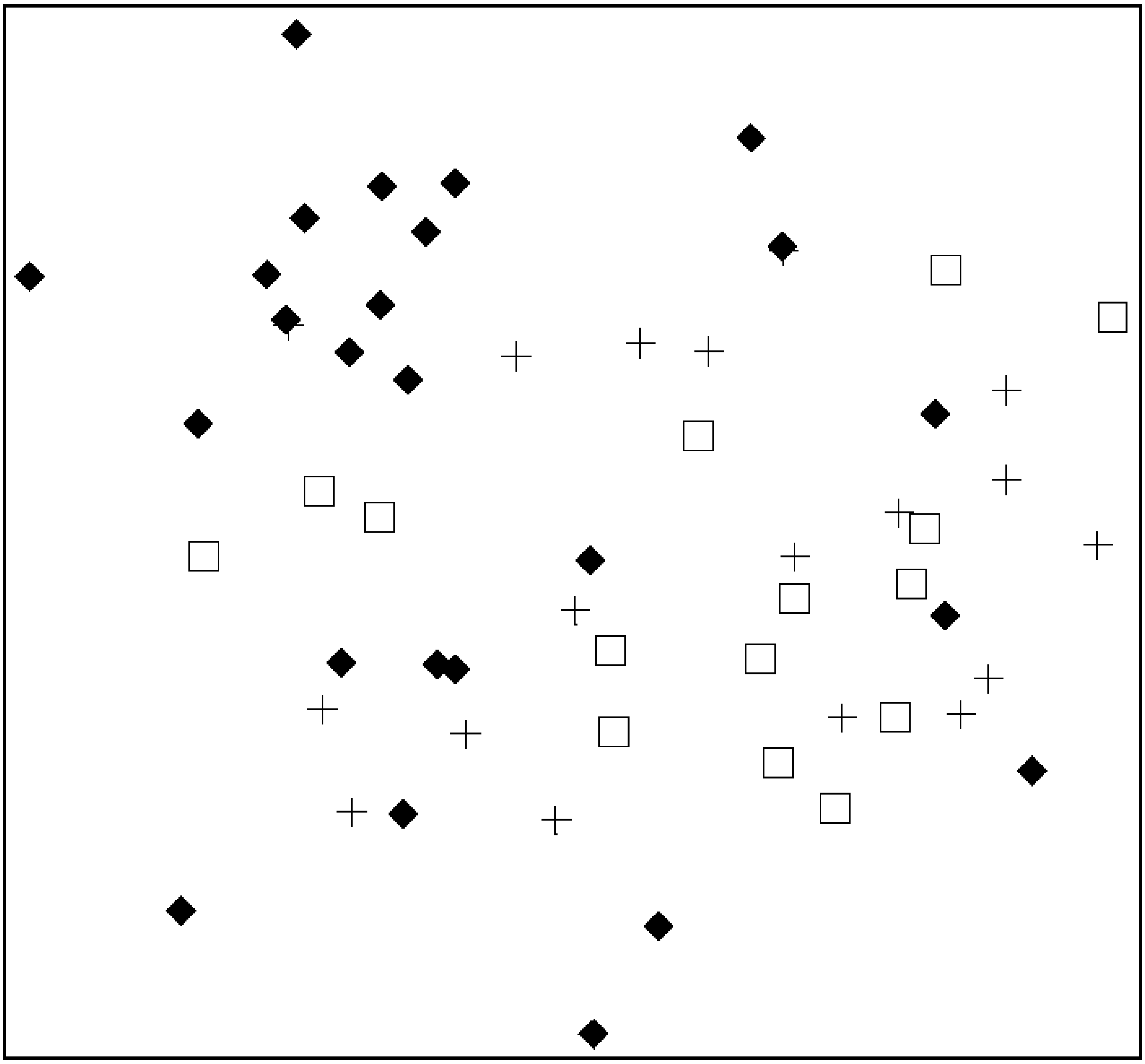

Figure 20. NMS ordination of spider assemblages (mean relative abundance per plot) among open space types: ∀ = Glades; 2 = Rides; ϶ = Roads. Final stress = 23.00; Final instability = 0.011; Axis 1 r^2 = 0. 40; Axis 2 r^2 = 0.25.

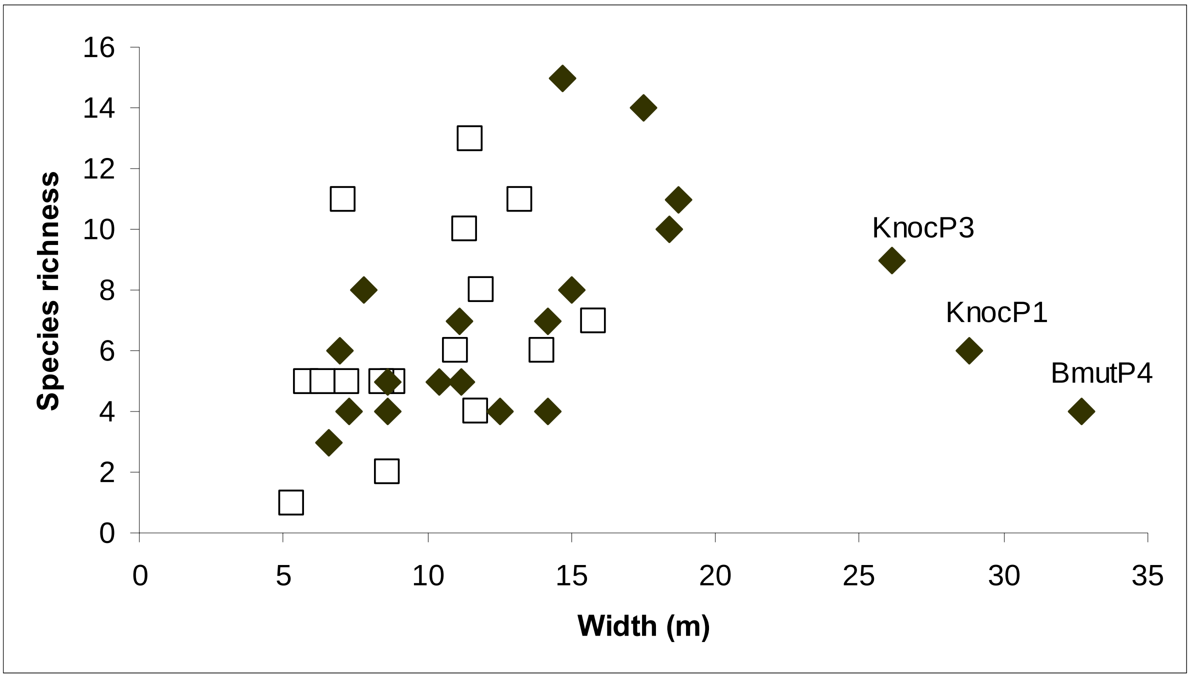

Figure 21. The relationship between open-associated species richness and ride width (∀) and road verge width (϶) with three outliers identified.

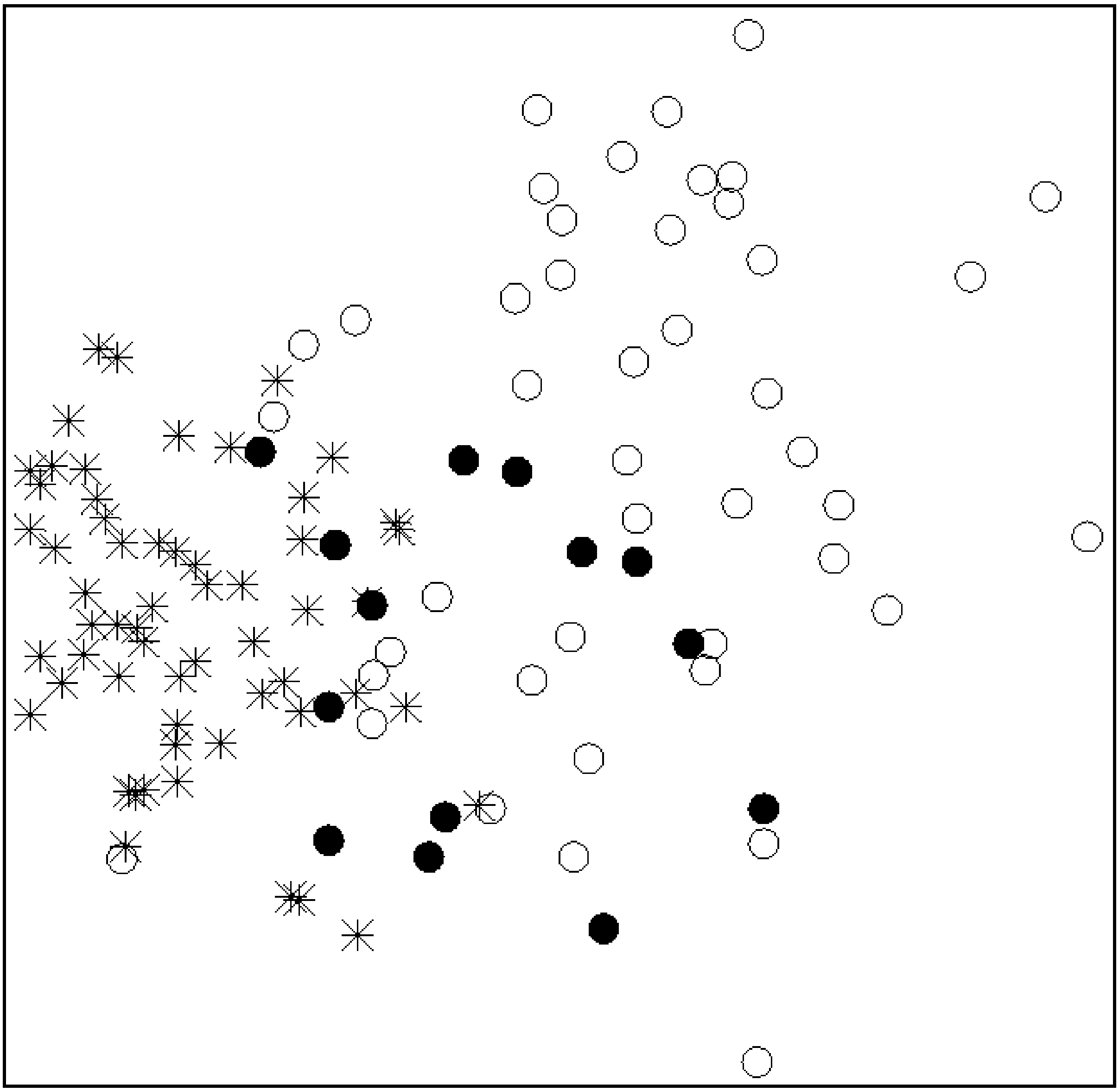

Figure 22. NMS ordination of spider assemblages (transect position per plot) in the two types of open space habitat and adjacent forest traps on the transect:) = Lower-field layer cover open space; # = Shrub/deciduous cover open space; ✳ = Forest traps. Final stress = 25.72; Final instability = 0.014; Axis 1 r^2 = 0. 38; Axis 2 r^2 = 0.22.

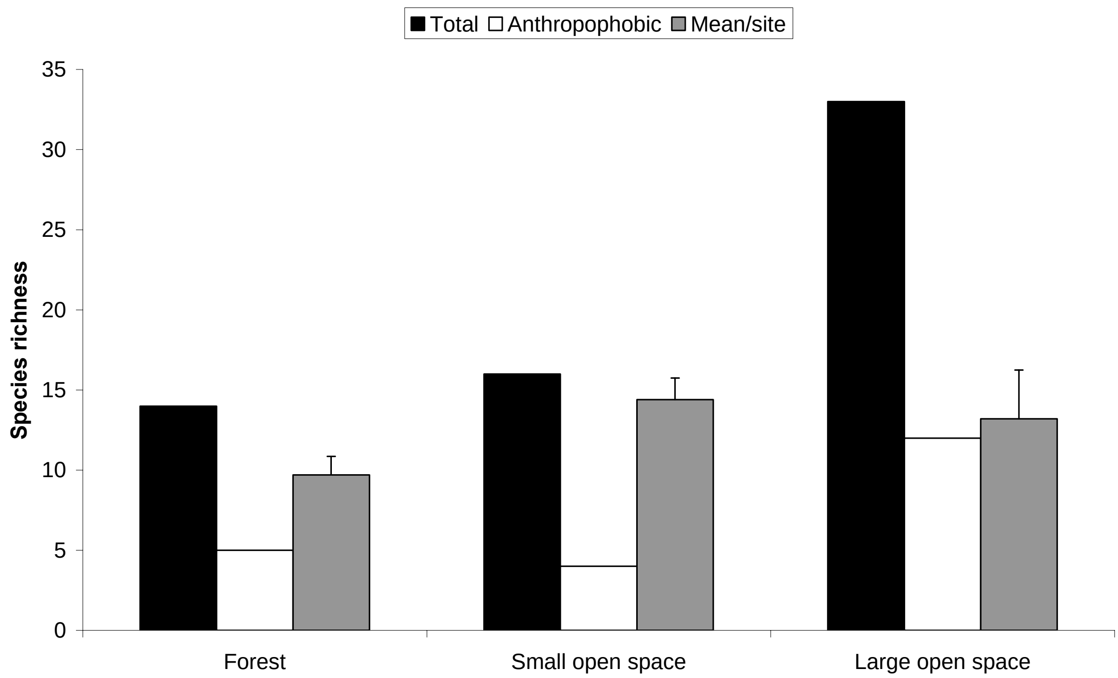

Figure 23. Open space associations of the recorded hoverfly fauna.

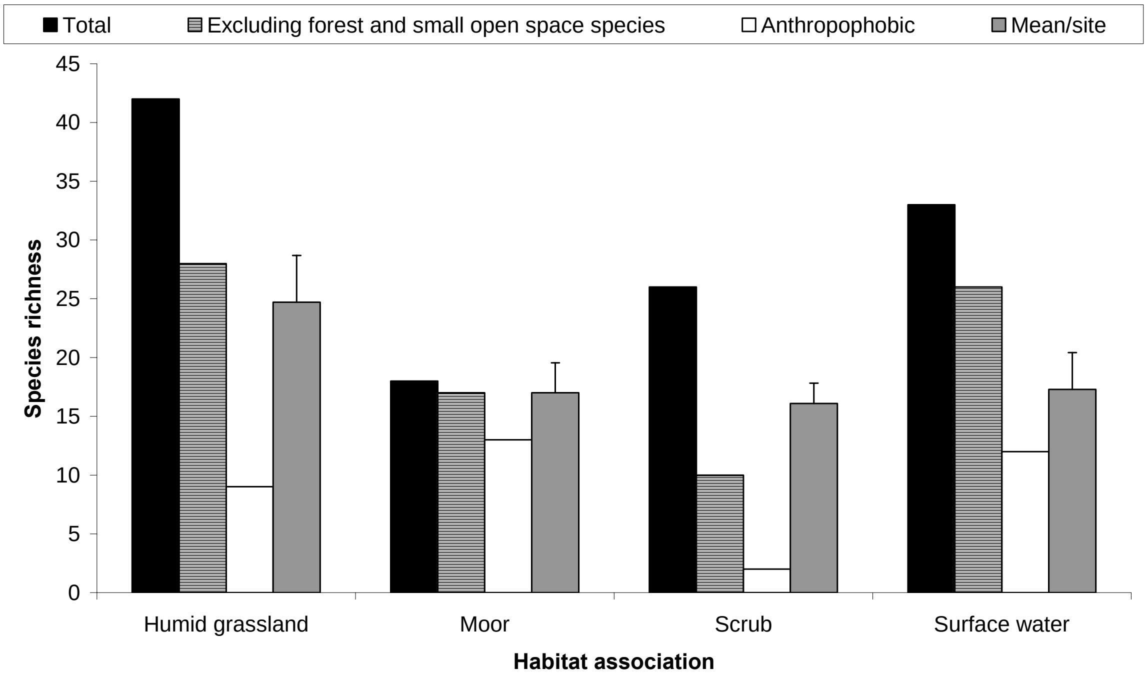

Figure 24. Macrohabitat associations of the recorded hoverfly fauna.

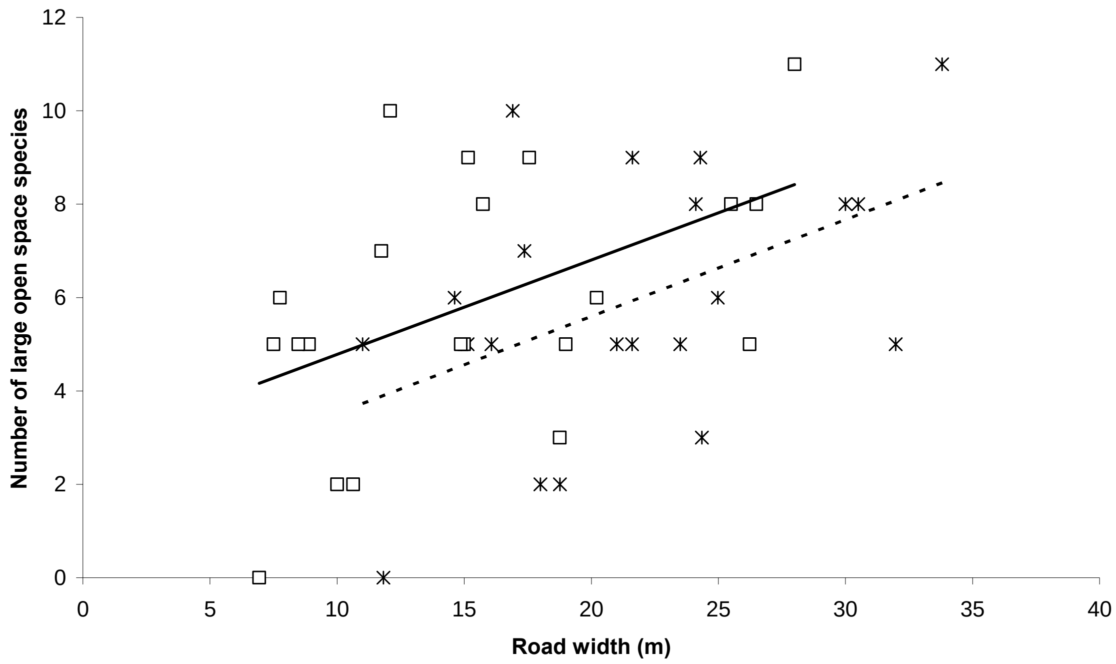

Figure 25. Relationship between forest road width over a 200 m length centred on the Malaise trap and species richness of hoverflies associated with large open spaces. Squares refer to average inter-canopy width (solid line shows linear regression of this relationship) and asterisks to inter-trunk width (dashed line shows linear regression).

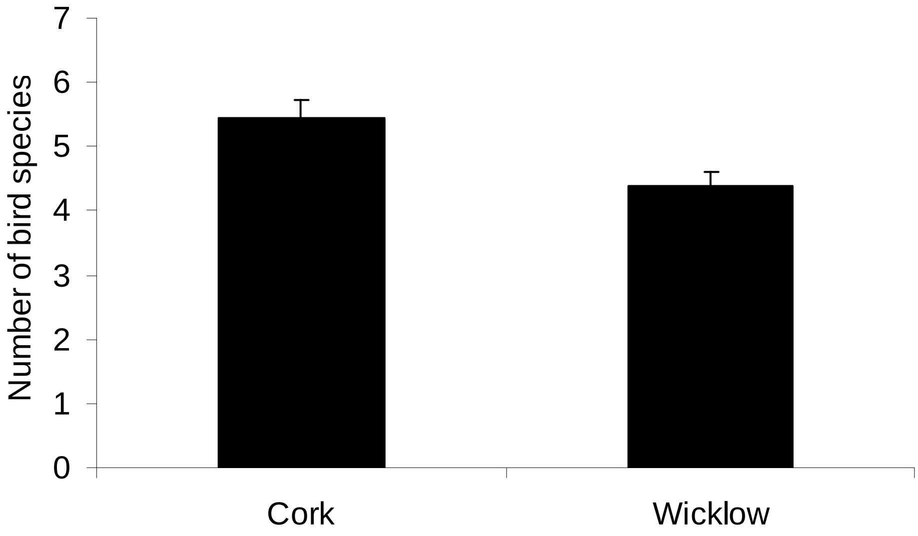

Figure 26. Mean number of bird species found along road sections of Cork and Wicklow sites (Nested ANOVA, F=4.75, d.f.=1,52, p=0.034). Bars indicate standard error.

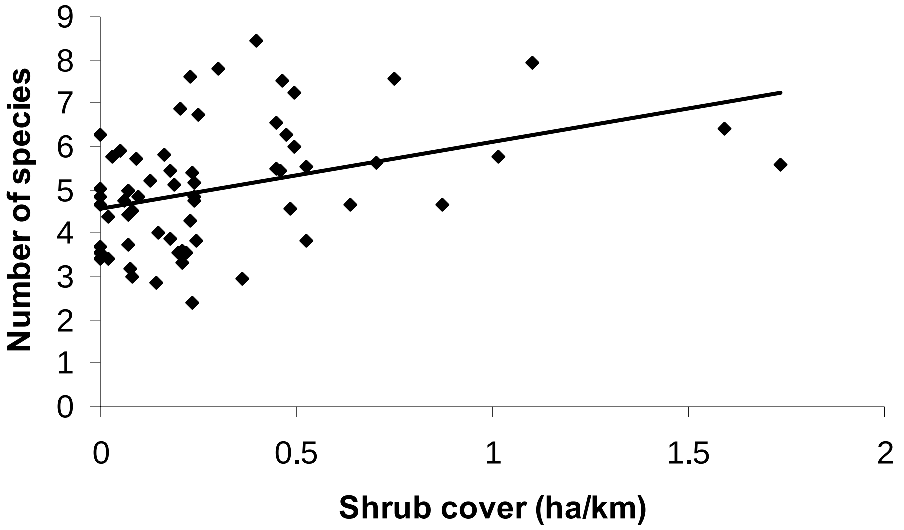

Figure 27. Relationship between number of bird species and area of shrub cover per km of road along sections of forest road in twelve forests (r=0.40, n=64, p=0.001).

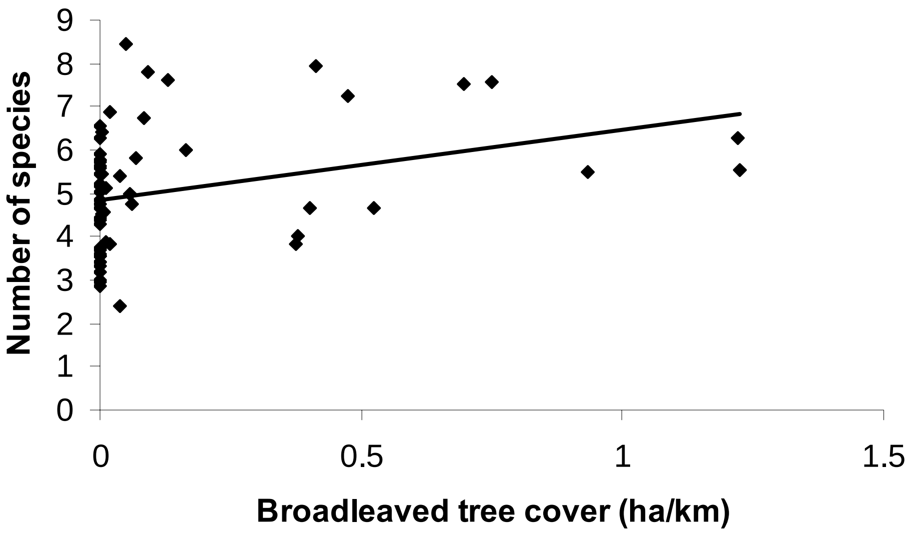

Figure 28. Relationship between number of bird species and area of broadleaved tree cover per km of road along sections of forest road in twelve forests (Kendall's τb, n=63, p=0.017).

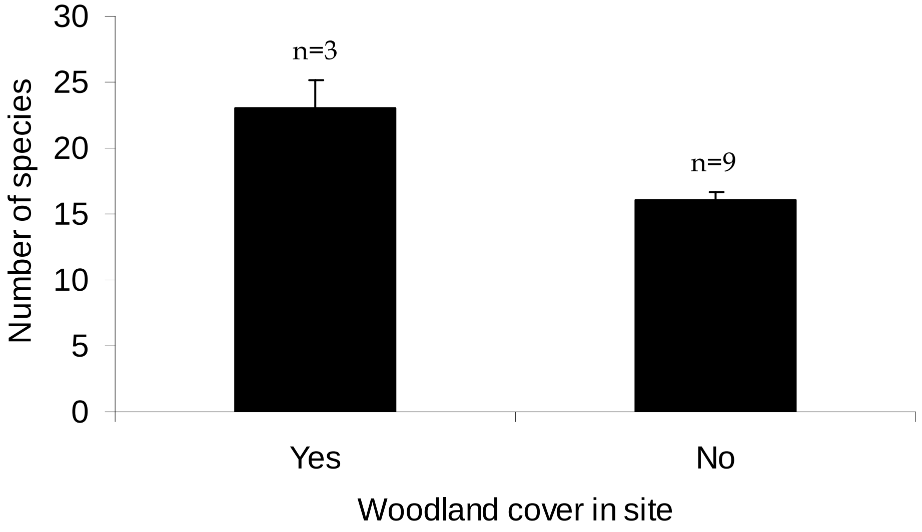

Figure 29. Number of bird species in sites with and without Woodland cover. Species richness is higher in sites containing an area of Woodland ($t=4.76$, d.f.$=10$, $p=0.001$). Bars indicate standard error.

Figure 30. In the three sites with Woodland cover, the number of bird species within 100m in areas with Woodland cover within 200m and areas with no Woodland cover within 200m. Areas with Woodland have a higher species richness ($t=2.14$, d.f.$=34$, $p=0.040$). Bars indicate standard error.

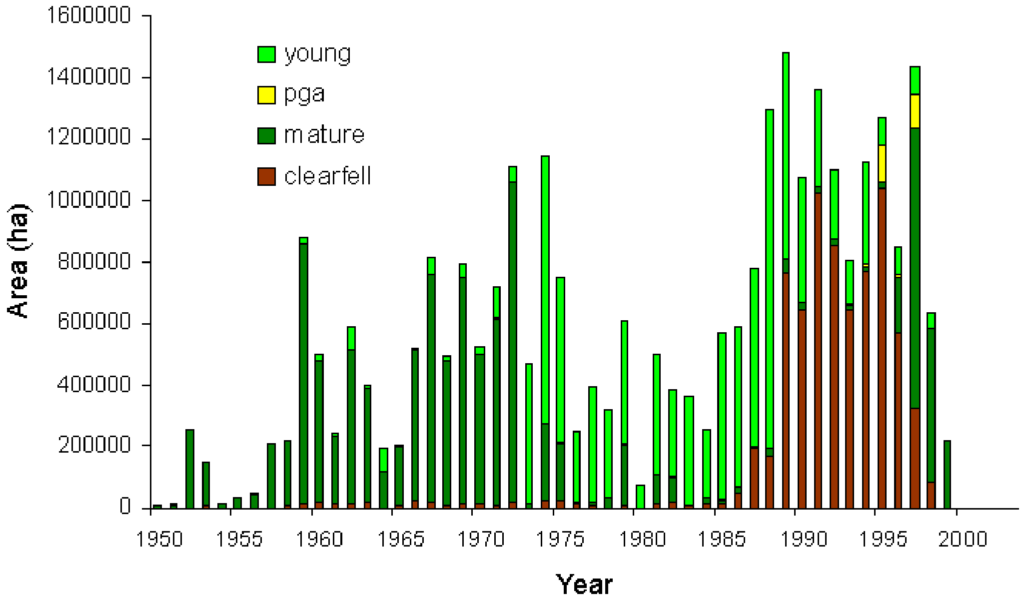

Figure 31. Total area of Coillte planting and proposed planting (PGA) within 500 m of 1156 randomly located points within Hen Harrier IAs. Planted areas are broken down into year of planting, and forestry types taken from the FIPS database.

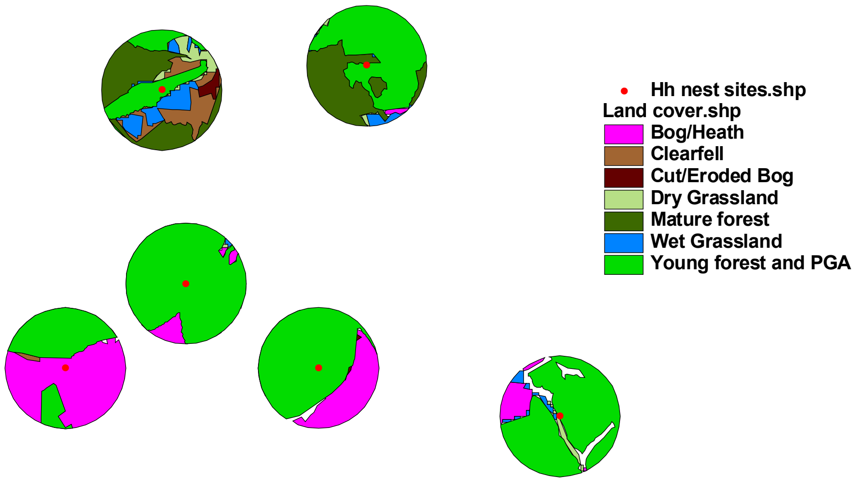

Figure 32. A screenshot of the GIS, showing a breakdown of landcover types within 500 m of six nests in the Mullaghareirk IA on the Cork/Kerry border.

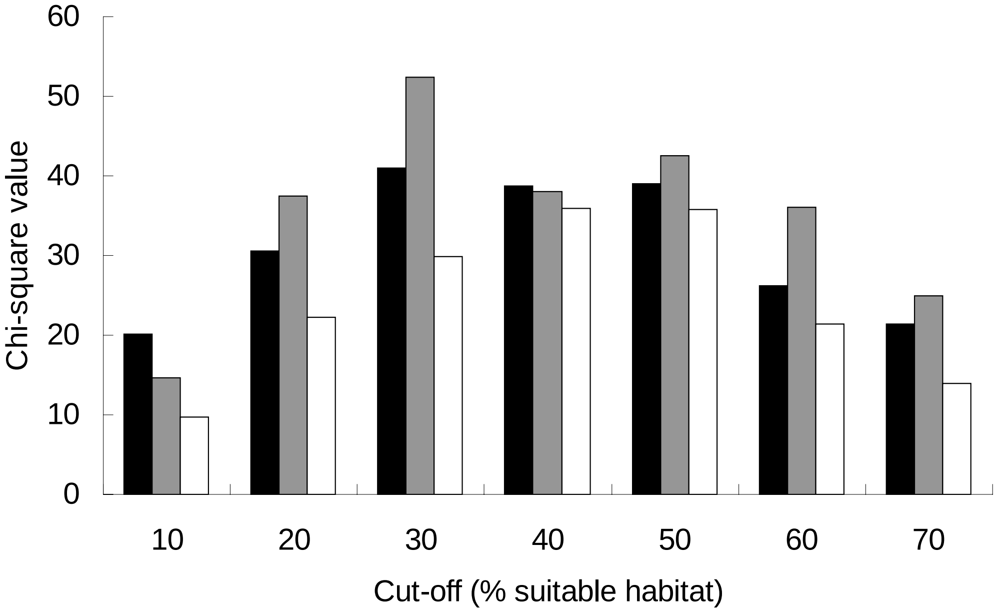

Figure 33. $\chi 2$ values for contingency tests of whether a significantly different proportion of points with a high percentage cover of suitable habitat is occupied by Hen Harriers, compared to points with a low percentage cover of suitable habitat. X-axis values indicate the threshold percentage value used to discriminate between high and low cover of suitable habitat. Suitable habitat was calculated at three different scales; 500m (black bars), 1 km (grey bars) and 2 km (white bars) from each point.

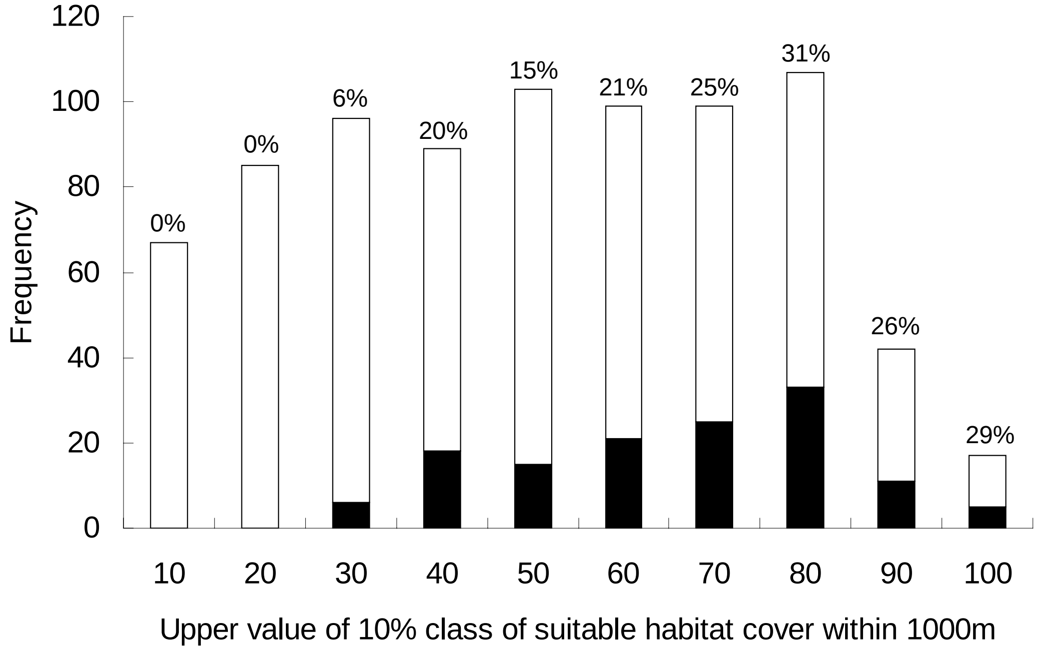

Figure 34. The frequency distribution of 804 points in the Hen Harrier IAs according to cover of suitable habitat for Hen Harriers within a radius of 1000m. The numbers of points within each 10 percentile group that were occupied by Hen Harriers during the 1998-2003 survey are represented by the black portions of the bars. The white section represents points situated more than 500m from the nearest Hen Harrier nest found during the survey. Each bar is annotated with the percentage of points occupied by Hen Harriers within that 10 percentile group. Habitat cover is taken from datasets last updated between 1997 and 1999.

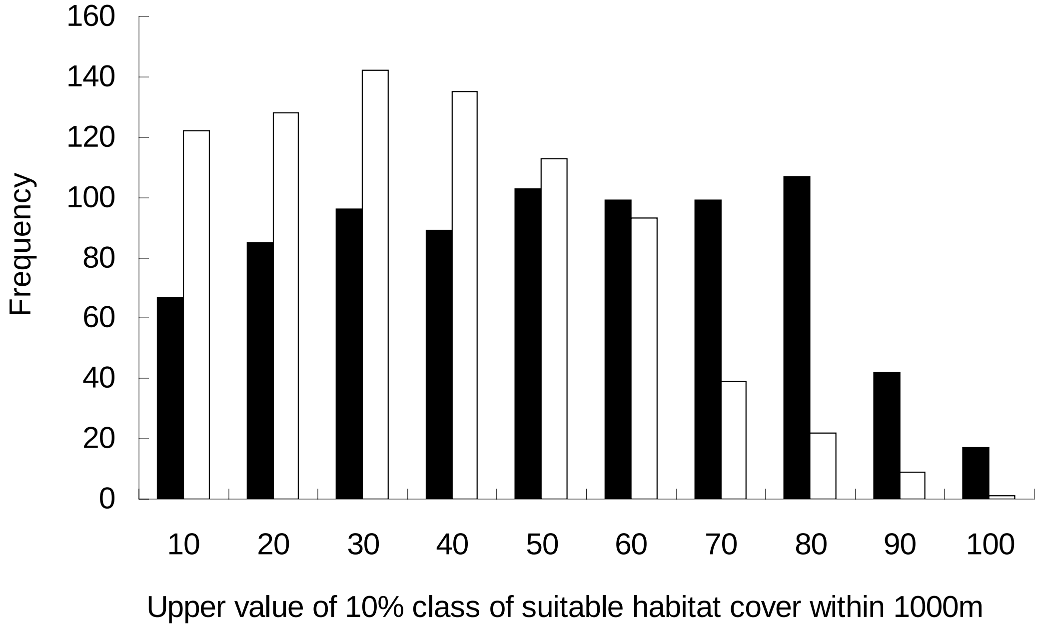

Figure 35. The frequency distribution of 804 points in the Hen Harrier IAs, according to cover of suitable habitat for Hen Harriers within a radius of 1000m, for the periods 1999 (black bars) and 2015 (white bars). Habitat cover for both periods is based on datasets updated between 1997 and 1999, from which 2015 habitat cover was extrapolated according to projected maturation and felling of forested areas.

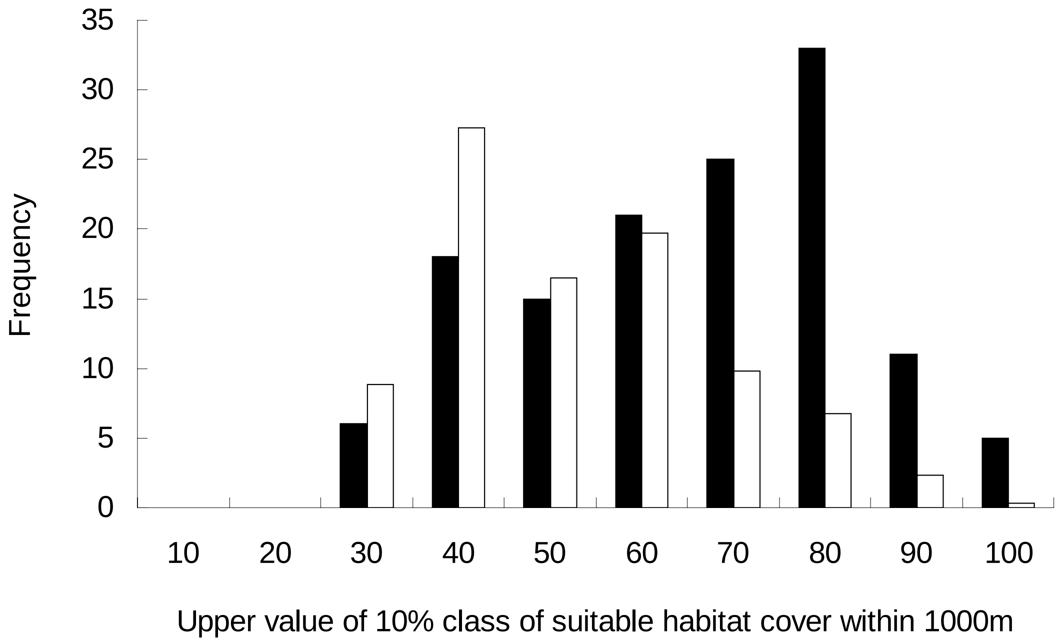

Figure 36. The frequency distribution of 134 points in the Hen Harrier IAs occupied by Hen Harriers during the 1998-2003 survey (black bars), and the estimated distribution in 2015 (white bars). The 2015 estimate is based on the proportion of points within each 10 percentile of suitable cover that was occupied by Hen Harriers.

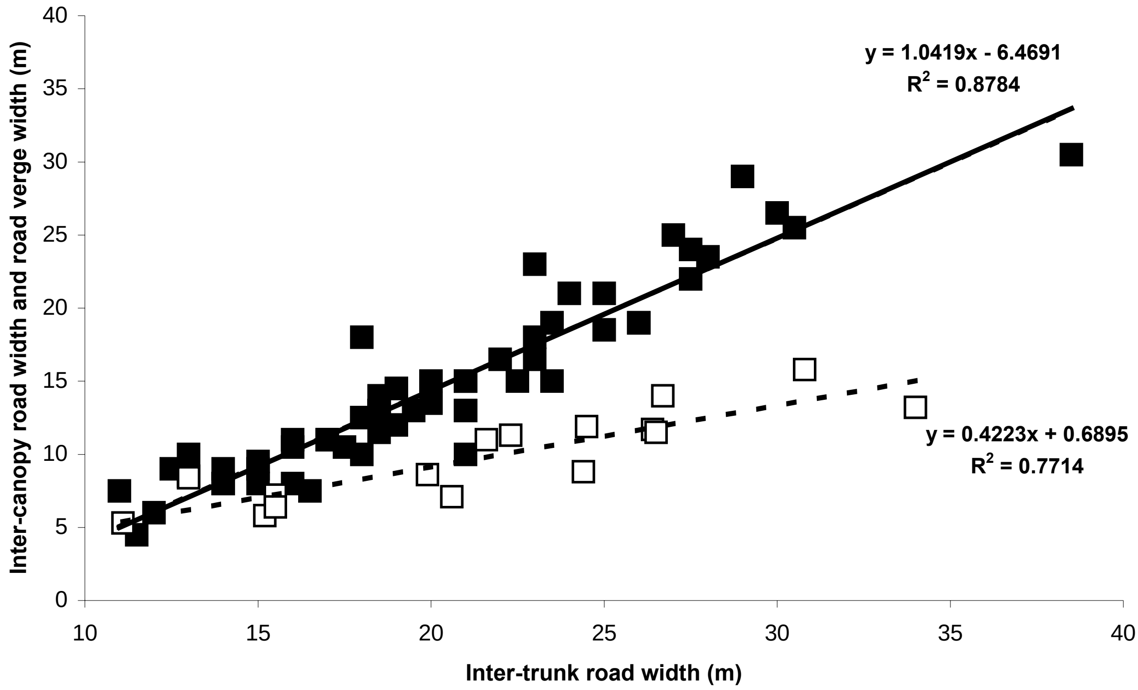

Figure 37. Relationship between inter-trunk forest road width and inter-canopy forest road width (solid squares) and road verge width (open squares). The inter-trunk vs. inter-canopy relationship is based on data from 63 forest road sections measured as part of the habitat recording for the bird survey (see Section 8.2.2). The inter-trunk vs. road verge relationship is based on data from 16 forest road plots measured as part of the habitat recording for the spider survey (see Section 6.2.4 On average, the standard inter-trunk forest road width of 15 m corresponds to an inter-canopy width of 9.2 m and a road verge width of 7.0 m.

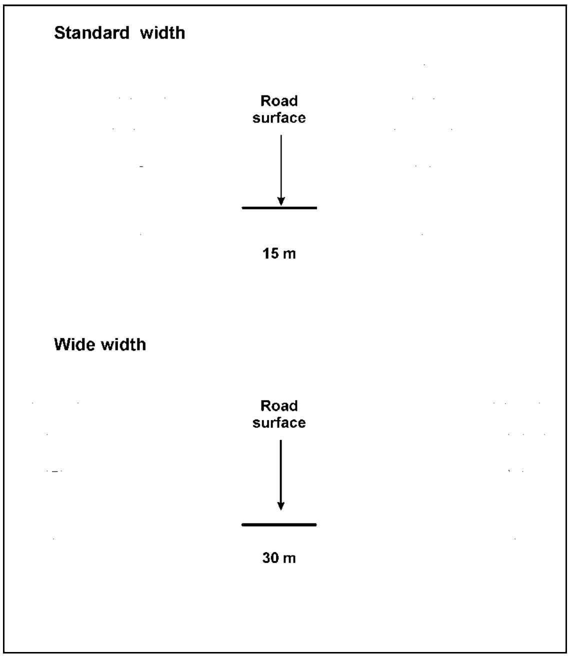

Figure 38. Diagram of road widths experimental treatments.