BIOFOREST

Biodiversity Assessment of Afforestation Sites

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Declaration

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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 objectives of this project were to assess biodiversity in a range of important habitat types which are frequently afforested, to develop methodologies and indicator species for the assessment of biodiversity and to assess the efficacy of the *Forest Biodiversity Guidelines* (Forest Service 2000c) and make recommendations for their improvement. The project involved a review of assessment practices in other countries (Gittings *et al.* 2004), a review of literature published on the relevant habitats and field surveys of habitats likely to be used for afforestation and of these habitats five years after afforestation with Sitka spruce (*Picea sitchensis*). A major database was developed during the project and all data from the project, including geographical information, are stored in that.

HABITATS REVIEW

A core principle of Sustainable Forest Management (SFM) is that forestry does not impact detrimentally on unforested habitats. Therefore, information on the biodiversity of habitats that are frequently subject to afforestation is required if Ireland's forests are to be managed sustainably. Biodiversity encompasses variation at three scales, genetic, species and ecosystem, and incorporates three elements, compositional diversity, structural diversity and functional diversity. Biodiversity is commonly used in a wider sense, incorporating both the variability of genes, species and ecosystems and also their conservation value, as determined by several criteria, including integrity and rarity at local, regional, national and international scales. It is in this wider sense that we use the term *biodiversity* throughout the report. Because of its scope, it is generally impossible to assess all facets of biodiversity, even in relatively small sites. For this reason, biodiversity research and management typically focuses on indicators of biodiversity. We reviewed the biodiversity of three types of habitats that are commonly afforested in Ireland, improved grasslands, wet grasslands and peatlands, and identified potential indicators of biodiversity that will be tested using field data later in this report.

The Irish habitat classification scheme developed by the Heritage Council (Fossitt 2000) provides the most current and widely used broad classification of habitats in Ireland. This level of classification is adequate for use when studying mobile, wide-ranging taxa, such as birds. However, the broad habitat types defined by Fossitt (2000) frequently combine distinctive plant communities that differ in ecology and biodiversity. The Braun-Blanquet system of phytosociology has often been used in the past by researchers in Ireland, and provides a more fine-scale system of classification. Another advantage of this system for our purposes is the use of character species to define and distinguish phytosociological associations (plant communities) and other levels in the classification hierarchy. Character species of plant communities of high biodiversity interest are well-suited to be potential indicators of biodiversity.

Climate, soils and human management determine the composition and abundance of species in grasslands. In general, the more intensive the management, the lower the biodiversity. Small pockets of semi-natural grassland are often found in a matrix of more intensive land-use, and are vulnerable to loss through agricultural intensification, dereliction or conversion to a different land-use, such as forestry. Various attempts have been made to estimate the cover of different grassland types in Ireland, but these are generally either inaccurate, out-of-date or localised. Irish grasslands are divided into three phytosociological classes comprising lowland pastures, upland acid grasslands and dry limestone grasslands. Lowland pastures are further subdivided into a group of dry semi-natural grasslands, improved grasslands and intermediates, and a group of nutrient-poor and base-rich wet grasslands. Improved grasslands are heavily grazed, are frequently cut for silage, usually receive high fertiliser and herbicide applications and are often reseeded. Such grasslands are generally species poor and are dominated by perennial ryegrass (*Lolium perenne*) and white clover (*Trifolium repens*), together with a limited number of agricultural weeds. With the exception of field-margin hedgerows, improved grasslands usually also support a poor bird fauna. In contrast, wet grasslands can be some of the most species-rich grassland communities in Ireland. Both nutrient-poor and base-rich wet grasslands are frequently dominated by rush (*Juncus*) species and often support a diverse assemblage of broadleaved herbs. However, species-poor intermediates between improved and wet grasslands can also be dominated by rushes and superficially resemble more high biodiversity types. Wet grasslands such as the Shannon callows can be important feeding and breeding grounds for wildfowl and waders.

Peatlands in Ireland include bogs, fens and wet heaths. Of these, the peatlands that appear to be most frequently afforested in Ireland are blanket bogs and wet heaths. Wet heaths occur on shallow peats or peaty podzols and are generally dominated by dwarf shrub vegetation, especially heather (ling, *Calluna vulgaris*) and cross-leaved heath (*Erica tetralix*). Wet heaths frequently occur in intimate mosaics with blanket bog. Blanket bogs can be divided into two types: lowland blanket bog, which occurs in oceanic climates in the west at elevations below about 150 m elevation, and upland blanket bog, which occurs in hilly or mountainous terrain throughout the country. Upland blanket bogs are characterised by an abundance of *Sphagnum* mosses, cottongrass (*Eriophorum*) species and dwarf shrubs, including heather, cross-leaved heath and bilberry (*Vaccinium myrtillus*). In contrast, lowland blanket bogs are more grassy in appearance, with black bog-rush (*Schoenus nigricans*) and purple moor-grass (*Molinia caerulea*) as among the most prominent species, and lower *Sphagnum* cover than in upland blanket bogs. Lowland blanket bogs also frequently include a variety of hydrological features, such as flushes, pools, streams and swallowholes; these can also be found in upland bogs, but are much less common. Blanket bogs and wet heaths support a number of birds of conservation concern, including Red Grouse, Lapwing, Golder Plover, Curlew and Greenland White-fronted Geese. Blanket bogs and wet heaths are important Irish habitats at the national and international levels. Active (i.e. peat-forming) blanket bogs are priority habitats for conservation under the EU Habitats Directive, and wet heaths are also a designated, though non-priority, habitat for conservation. Ireland contains approximately 8% of the world's blanket bogs, and therefore has an important international role in conserving these habitats.

STUDY SITES

Twenty-four pairs of planted and unplanted sites, ranging from 3.2 to 47.3 ha in area, were surveyed, with eight site pairs in each of three habitat types: improved grassland, wet grassland and peatland; an additional two unplanted wet grasslands and one unplanted improved grassland were also surveyed. Peatland sites included lowland and upland blanket bog and wet heath. The paired sites were matched for habitat, vegetation type, soil properties, and geographical location, and the planted sites were comprised of five-year old stands of Sitka spruce (*Picea sitchensis*). Management information on the sites was collected in the field and by correspondence with landowners.

VEGETATION

Our aims were to increase understanding of the biodiversity of plant communities commonly used for afforestation, develop methods and indicators for biodiversity assessment of afforestation sites and investigate the initial effects of afforestation on these plant communities. In each site, habitats were mapped according to the Irish habitat classification scheme (Fossitt 2000), and plant species were recorded using a simple ranked (DAFOR) scale of abundance. In addition to the main habitat types, supplementary and marginal habitats were surveyed, including hedgerows, scrub, streams, flushes, stone walls and earth banks. In the main habitat type of each site, three 100 m^2 plots were established in which the presence of each plant species was recorded. In each 100 m^2 plot, two 4 m^2 subplots were sampled in which the relative abundance of plant species was recorded to the nearest 5%. Species richness and diversity measures were calculated for the plot data. In addition to the species recording, data were also collected at all scales on vegetation structure, soil chemical and physical characteristics and other environmental factors. Community data were analysed using multivariate clustering and ordination techniques, and variation in biodiversity measures and structural and environmental variables among plant community groups were analysed using parametric and non-parametric statistics.

Total species richness and diversity was lower in sites pre-identified as improved grasslands than in peatlands or wet grasslands. Cluster analysis of the habitat data confirmed the pre-established habitat groups, and further subdivided improved grasslands and peatlands into subtypes. We also found that supplementary and marginal habitats can contribute substantially to the biodiversity of a site, through provision of habitat for species that would otherwise not occur in the main habitat matrix. Additional cluster analyses were carried out on 100 m^2 and 4 m^2 plot data. Although there was substantial variation among sampling scales in the assignment of sample units to clusters, certain patterns emerged from the data. In peatlands, the more intact lowland blanket bogs were distinguished at the larger scales from the remainder of the wet heaths and upland blanket bogs, which were on the whole more disturbed and of less biodiversity interest. Grasslands were generally divided into improved grasslands, semi-improved grasslands, nutrient-poor wet grasslands and base-rich wet grasslands. At the 4 m^2 plot scale, a semi-improved wet grassland grouping was recognised that supported lower biodiversity than the remainder of the semi-natural wet grasslands. These latter were recognised as potentially being of high biodiversity interest, although their value will depend to a great extent on the landscape context. A given semi-natural wet grassland may be of ecological importance in an agriculturally intensive landscape, whereas the same wet grassland may be of little interest in landscapes where communities of similar or higher quality are abundant.

Potential compositional indicators of biodiversity that were identified in the habitats review earlier in this report and also in previous BIOFOREST work were assessed for their value in discriminating between high and low biodiversity plant communities in the field survey. Indicator species for the different vegetation cluster groups were identified using quantitative methods, and potential compositional indicators of biodiversity were compared with these. Species identified as potential indicators that were confirmed by independent testing were then designated as a confirmed compositional indicators of biodiversity. Similarly, pre-identified potential structural and functional indicators were investigated to determine whether they were significantly associated with one or more high biodiversity plant community groups. In addition to confirmation of indicators, new provisional indicators arising from these data were identified. Indicators were developed separately for grasslands and peatlands. Guidelines are provided for the use of these indicators, particularly the spatial scales at which they are best applied.

Afforestation generally caused significant changes in species composition and abundance and decreases in the biodiversity of plant communities, based on comparisons of planted and unplanted site pairs. The changes were largely the result of exclusion of grazing livestock, site drainage for afforestation and changes in nutrient inputs. The relative abundance of species changed more than did plant species composition. Competitive grasses increased greatly in abundance to the detriment of ruderal and stress-tolerant species. In wet grasslands and peatlands, the proportion of species that are characteristic of wet habitats decreased. Forestry drains were found to provide a refuge for species not able to compete with vigorous grasses, but such refuges will most likely be temporary as most understorey species will be shaded out when the canopy closes in forests of densely shading species, such as Sitka spruce. The early stages of afforestation had no significant effect on the community composition of hedgerows surrounding planted grasslands.

We conclude that afforestation can have a detrimental effect on the biodiversity of semi-natural habitats. Such habitats should not be afforested unless they are common in the surrounding landscape. For some habitat types, notably wet grasslands, there is insufficient baseline information on their biodiversity and distribution to assist in landscape or regional-scale evaluations. *We recommend that a national survey and classification of Irish grasslands be undertaken as a matter of urgency.* The relevant authorities should also take care to ensure that piecemeal afforestation and agricultural intensification does not eliminate the biodiversity value of semi-natural landscapes. Largely intact peatlands with little or no turf cutting or with well-developed hydrological features such as flushes, streams or pools should not be afforested. On the other hand, afforestation of semi-improved and improved grassland habitats will most likely have a neutral or positive effect on biodiversity at the site scale and also at the landscape scale in areas where wooded habitats are uncommon.

HOVERFLIES

We used hoverflies as an indicator group to assess the biodiversity value of habitats that are typically used for afforestation in Ireland, to identify the indicators of biodiversity for these habitats, and to examine the changes in biodiversity that occur in these habitats in the initial period following afforestation. The pre-afforestation habitats we surveyed had low representation of the characteristic hoverfly fauna associated with wet grassland and specialist wetland habitats, although we recorded several rare species. The assemblages of open habitat-associated hoverflies differ between these three habitat types. Peatland sites have the lowest number of open habitat-associated species but the highest number of species restricted to semi-natural habitats. Peatland sites with acid fen habitat had higher numbers of wetland specialist species, and these included three rare species. In improved and wet grassland sites, grazing sensitive species were affected by grazing intensity, wet grassland specialist species richness was positively related to the frequency of tussocks and tall herbs, and woody vegetation species richness was positively related to the amount of broadleaved woody vegetation and the frequency of understorey vegetation. There were slight differences in the structure of the hoverfly assemblages between the planted and unplanted improved and wet grassland sites and there were higher numbers of species associated with woody vegetation in the planted sites. The differences in numbers of these species between the planted and unplanted sites were related to the differences in amount of broadleaved woody vegetation.

Based on our results, we make the following recommendations. Maintenance of surface water, *Salix* swamp and overmature hedgerow habitats with tall trees in landscapes subject to afforestation may be important for conservation of rare hoverfly species associated with these habitats. Afforestation should avoid peatland sites with extensive acid fen habitat as these are important for specialist wetland hoverfly species. Broadleaved woody vegetation should be promoted in young conifer plantations through the retention of existing vegetation and the planting of broadleaved trees as it encourages the development of hoverfly assemblages associated with woody vegetation habitats. Removal of hedges and scrub during afforestation should be avoided as it may cause a net decrease in hoverfly biodiversity.

SPIDERS

Spiders are an important component of terrestrial ecosystems as abundant predators and prey for higher organisms, but they also have a potential use as biodiversity indicators. The aim of this project was to assess spider diversity among habitats which are typically used for afforestation in Ireland, to identify habitat parameters which could potentially be used as indicators of biodiversity value and to assess the initial effects of afforestation on the spider fauna. The spiders were sampled using pitfall traps which were located within the major vegetation types present in each site as well as within supplementary habitat features which may add to biodiversity value of a site as a whole such as hedgerows, flushes and the edges of ditches and streams.

Each habitat supported distinct spider assemblages which reflect the major differences in both environmental conditions and management regime. The improved grasslands had low spider species richness and low variation in assemblage structure which is probably related to the intensive management of improved grasslands compared to the other habitat types, and so hedgerows may be an important aspect of the spider diversity within agricultural landscapes. The peatlands, and to a lesser extent wet grasslands, support a diverse and specialist spider fauna, including a number of rare species; this may be due to differences in soil moisture. Indicators of high biodiversity value identified included wet flushes in the peatlands and low grazing pressure in the wet grasslands, whereas cutover bogs had low biodiversity value. This study suggests that in terms of biodiversity value improved grassland is the preferable habitat for afforestation, because of the poor baseline spider diversity. However it may be unrealistic to expect land owners to afforest their most productive agricultural land, so the management and habitat indicators identified in this study may be of use for assessing habitat quality among the wet grassland and peatlands to allow sites with lower biodiversity value to be identified.

There were differences in the spider assemblages between the unplanted and planted sites across all of the habitats, though this difference was most notable in the peatlands. There were fewer habitat specialists and rare species in the planted sites, however several species associated with forested habitats were supported. The number of spider species associated with wet habitats was lower in the planted sites, which may be due site drainage in preparation for plantation establishment. In contrast, the planted improved grasslands were more species rich, and supported a greater number of spider species associated with low vegetation than the unplanted sites, which is probably related to a reduction in grazing pressure. The hedgerows did not differ notably in assemblage structure between the unplanted and planted sites, whereas areas of wet flush within the peatlands particularly differed, with the unplanted sites supporting a more unique and diverse spider fauna. This study suggests that even in the early stages of the forest cycle (five years) there is a change in the spider fauna, with the rare or specialist species being replaced by habitat generalists. It is also suggested that peatlands are particularly sensitive to afforestation, indicating that in terms of biodiversity loss, this habitat is the least suitable for afforestation.

BIRDS

We used mapping surveys to investigate the variation in bird assemblages between areas of unplanted peatland, wet grassland and improved grassland. We also used point counts to compare bird assemblages between these sites and sites that had been recently afforested and were closely matched in terms of pre-planting habitat. Environmental data on tree and shrub cover was also collected from each of these sites. Analysis of bird assemblages in unplanted sites separated them into five clusters – two peatland clusters, an improved grassland cluster, a wet grassland cluster and one mixed cluster that was intermediate between the other four clusters and included sites of all three habitat types. Sites in the two peatland clusters were less species rich than sites in the other three clusters, and supported lower overall densities of birds. This was, at least in part, due to the influence of hedges, which were lacking from peatland sites, but in grassland sites supported several bird species associated with tree and shrub cover that would otherwise have been very rare or absent. Hedges with high numbers of species and individual birds tended to be tall and wide with few gaps, a high density of mature standard trees, high plant species richness and high structural diversity of vegetation and microtopographical features such as ditches and banks. The three grassland clusters supported similar numbers of bird species, but the overall density of birds in the improved grassland cluster was very much lower than in the wet grassland cluster, with the mixed cluster intermediate between these two. This was partly due to the numbers of birds in the open land component of the sites and partly because of differences in cover of hedges (especially tall, wide hedges) and also non-hedge tree and shrub cover (all of these factors were lowest in improved grassland and highest in wet grassland).

Total shrub cover, bird species richness and total abundance of birds were significantly greater in planted sites than in the unplanted sites with which they were paired. The species composition of bird assemblages also differed among planted and unplanted sites. The bird species whose abundance responded most positively to afforestation during the first five years after planting were ground nesters, whose increased abundance appeared to be a response to increased availability of low-level cover. These species are relatively uncommon in the modern agricultural landscape, but the benefits of afforestation to these species forestry are short-term, lasting only until the tree canopy shades out the newly enriched shrub layer.

We recommend that afforestation should target habitats with relatively impoverished bird assemblages. By far the most abundant of these is improved grassland, but some areas of over-grazed wet grasslands and degraded peatlands can also be included. Afforestation of these habitats is especially likely to benefit birds if management of the forest is sympathetic to birds, encouraging broadleaved trees and shrubs, and creating structurally diverse open spaces. In particular, sufficient open space should be left around bird-rich hedges so that do not get shaded out during later stages of the forest cycle. Other open habitats generally cannot be afforested without negatively impacting on birds, particularly where elements of the bird community are of national or international conservation interest. However, even if these areas are not planted with trees, their value for birds may depend on their continuing to be grazed at low intensity, in order to allow the persistence of open habitat types.

SYNTHESIS AND RECOMMENDATIONS

We have developed a methodology for site assessment of biodiversity involving habitat mapping, recording of biodiversity indicators and evaluating the landscape context. We have identified two sets of confirmed indicators of grassland and peatland biodiversity for use in improved grassland, wet grassland and peatland habitats, and we have proposed an additional group of potential indicators that require further testing. These include compositional indicators (plant and bird species), structural indicators (e.g. shrub cover and grass cover) and functional indicators (e.g. grazing intensity, presence of fens or flushes). We also identify landscape-scale features that if present indicate that the wider-scale biodiversity of one or more species groups is likely to be high. Significant loss of these features from the landscape will probably have detrimental consequences for biodiversity. These indicators are suitable for use by non-specialists after training in their identification.

We have made six recommendations for improved planning and management of afforestation that will protect biodiversity of valuable open habitats at the site and landscape scales:

1. Foresters should compile annotated habitat maps of the entire site using the Irish habitat classification scheme (Fossitt 2000) and note the presence of indicators and other biodiversity features when conducting pre-afforestation site surveys.
2. Semi-natural habitats should not be afforested, unless there are mitigating circumstances.
3. Consider site biodiversity in context of the surrounding landscape prior to afforestation.
4. Retain hedgerows, scrub, wetland habitats and other marginal habitats and allow for adequate buffer zones.
5. Promote broadleaved woody vegetation in young conifer plantations.
6. Drains should not be eligible as part of an afforestation site Area of Biodiversity Enhancement.

Recommendations 1-3 and 6 should be incorporated into the *Forest Biodiversity Guidelines* (Forest Service 2000c), and recommendations 4 and 5 will require some modification and clarification of the *Guidelines* (Forest Service 2000c). Some of these support or extend recommendations made in other BIOFOREST reports (Gittings *et al*. 2004; Iremonger *et al*. 2006; Smith *et al*. 2005). We have also made three recommendations for further research that will provide direct benefits for afforestation planning in the context of biodiversity:

7. Testing and refinement of the indicators identified in this study and identification of indicators for other habitat types.
8. A comprehensive national survey and classification of grasslands.
9. Investigation of forestry and biodiversity at whole-farm and landscape scales.

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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 providing much-needed basic 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 (this project)**
- **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**

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, School of natural SciencesTrinity College, Dublin (TCD)**
- **Coillte Teoranta, The Irish Forestry Board (Coillte)**

The Coastal and Marine Resources Centre, 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 objectives of Project 3.1.1, the current project, were:

- Assess the biodiversity of frequently afforested habitats.
- Develop methodologies for biodiversity assessment and identify indicator species in these habitats.
- Assess the efficacy of the *Forest Biodiversity Guidelines* (Forest Service 2000c) and recommend improvements.

A number of work packages were outlined. The first of these was to carry out a review of methodologies used overseas for biodiversity assessment of potential forest sites. This was submitted in 2001 and accepted by COFORD and EPA in 2004 after some alterations (Gittings *et al*. 2004). The second was a review of literature on the biodiversity of habitat types that are typically used for afforestation in Ireland, and that is presented in the current document.

The start of fieldwork for this project was delayed because of an outbreak of Foot and Mouth Disease in Ireland in 2001. Virtually all afforestation in Ireland now takes place on privately owned land, and the outbreak precluded any work on this land category. In consultation with the Management Group and the Steering Group (see Appendix 1), the project was re-designed from a plan that would follow unafforested land through the first three years of afforestation to a plan that would mainly compare two sets of adjacent sites: one planted with forest five years previously and the other an unafforested control. In line with the other BIOFOREST projects there was a focus on Sitka spruce (*Picea sitchensis*) when selecting sites for study.

1.2 SEQUENCE OF PRESENTATION WITHIN THIS DOCUMENT

This document is divided into three main sections:

- The literature review.
- The main results sections, presented by taxonomic discipline.
- The conclusions and recommendations.

Within the second section, the results from the studies on the different taxa are presented mainly as stand-alone documents to streamline the production of publications from this report. Publication in the scientific literature, while not a condition of this Project, is very important as it assures the funding bodies and the stakeholders in Irish forestry that the research was of the highest international standard. The authors apologise for any repetition this generates between the chapters.

2 REVIEW OF THE BIODIVERSITY OF HABITAT TYPES USED FOR AFFORESTATION IN IRELAND

2.1 INTRODUCTION

2.1.1 Objectives

The objectives of this review are:

- To collate information from a wide range of different literature sources on the biodiversity of the habitat types pinpointed for this study: improved grasslands, wet grasslands and peatlands, and
- To use the information in the literature to identify species that may be used as indicators of biodiversity.

2.1.2 Structure of this chapter

We will first present a brief overview of the concept of biodiversity and the use of indicator species in Section 2.2. Grasslands are discussed in Section 2.3, where the ecology, distribution, vegetation and avifauna are reviewed. Peatlands are discussed in Section 2.4, where their ecology, origins, disturbance, distribution, vegetation and avifauna are reviewed. At the end of each of the habitat sections, we assess the conservation value of the habitat and identify potential indicators of biodiversity that will be tested using field data in later sections of the report. There is very little relevant literature on the Irish spider and hoverfly fauna associated with these habitats. Therefore, we have not included these groups in this review, but have made reference to the relevant literature in the spider and hoverly chapters.

2.1.3 Habitats

At present, little ecological information is collected prior to afforestation. Vegetation information collected by foresters on-site is limited to selection of one or more broad categories on Forest Service Form 1 (e.g. "grass", "grass-rush", etc.). Additionally, "important woodland and non-woodland habitats" are mapped and described as required under the *Forest Biodiversity Guidelines*, (Forest Service 2000c). but the habitats listed in the *Guidelines* do not conform to any published or frequently-used classification system. Therefore, it is difficult to determine what types of habitats are frequently subject to afforestation at present. For this reason, among others, Gittings *et al.* (2004). recommend that pre-afforestation habitat mapping and assessment should follow the Irish habitat classification system developed by the Heritage Council (Fossitt 2000).

Based on these limited data and consultations among the BIOFOREST Research Group, Steering Group and other external experts, we decided to focus this aspect of our work on three habitat groups: wet grassland, improved grassland and peatland. Peatlands were much afforested in the past by the State. Vast areas of virgin lowland and upland blanket bog were considered agriculturally worthless and their ecological value was not appreciated at the time. Although State afforestation of large peatland areas no longer happens, smaller scale afforestation still takes place by private landowners. We focused our review and survey of peatlands on lowland and upland blanket bog and wet heath. Wet heath frequently occurs in intimate mixture with bog vegetation, and is therefore difficult to separate from it. Grasslands are certainly the habitat group most frequently afforested at present. Wet grasslands probably represent a large proportion of those that are planted. Such habitats often represent the least agriculturally productive parts of a farm, and are therefore some of the most likely to be afforested. However, with improvements in the grants available to private landowners and changes in EU agricultural policy, some intensively managed, improved grassland is also being planted.

These three habitat groups will be considered in detail below. Related habitats will also be briefly discussed where appropriate. The main classification system used will be the Irish habitat classification developed by the Heritage Council (Fossitt 2000). However, we will also use extensively the Braun-Blanquet system of phytosociology to identify and discuss more fine-scale habitat/vegetation groupings. Where these smaller groupings are of biodiversity interest, their character and differential species will be highlighted as potential indicator species of biodiversity.

2.2 BIODIVERSITY AND INDICATORS

2.2.1 Biodiversity

2.2.1.1 Defining biodiversity

"Biodiversity" is a term that, while frequently used, is notably difficult to define succinctly. In essence, biodiversity encompasses all the variety of life (Gaston 1996). Perhaps the best and most commonly used definition is that proposed by the Convention on Biological Diversity: "'Biological diversity' means the variability among living organisms from all sources including, *inter alia*, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems" (UNEP 1992). In this definition, three aspects of biodiversity are evident: genetic diversity, species diversity, and ecosystem diversity.

At the most fundamental level is the diversity of genotypes within species. It is this genetic diversity that permits adaptation and evolution in the face of ecological change. To preserve the genetic diversity within species, it is necessary to conserve populations representative of this diversity (Mallet 1996). In this respect, isolated populations or populations at the limits of their species distributions may be particularly important. Conservation of species reaching the limits of their distributions in Ireland, such as arctic-alpine or Mediterranean plants, therefore acquires international significance.

Species diversity is probably the most commonly measured aspect of biodiversity. Assessing the biodiversity of species is easier than assessment of genetic diversity. The species diversity of a site consists of two attributes: species richness, the number of species present, and evenness, the relative abundances of the species (Begon *et al*. 1990; Gaston 1996). Traditionally, mathematical indices, such as Simpson's or Shannon's indices, have been constructed to take into account both these aspects of species diversity. Sites dominated by one or a small number of species are intuitively less diverse than sites where species abundances are more equably distributed. On the other hand, sites containing rare species are important for the conservation of biodiversity because loss of these species represents a reduction of species richness at the local, regional, and/or national level.

Ecosystem diversity has received little research attention, mainly because of a lack of clarity as to what biodiversity at scales above species actually means (Gaston 1996). Just as species richness and evenness can be identified as components of species diversity, however, ecosystem diversity can be said to include both the richness of different types of ecosystems and the evenness of their distribution. This then begs the question of what types of ecosystems exist. An ecosystem "comprises the biological community together with its physical environment" (Begon *et al*. 1990). For the purposes of this review, the Irish habitat types (Fossitt 2000), which are defined primarily on the basis of vegetation, but also by physical parameters (e.g. wet heath vs. blanket bog, intact vs. cutover bog), serve as a classification of ecosystems present in Ireland. The value of preserving a diversity of habitats has been enshrined in the EU Habitats Directive. Several Irish habitats are important at the European scale, including turloughs, machairs and raised bogs (European

Commission 1999). Other types, such as semi-natural woodlands, are relatively uncommon in Ireland and are therefore important for habitat diversity on the national scale.

At each of these three scales, biodiversity incorporates compositional elements (e.g. species or genes), structural elements (e.g. patterns of species abundance), and functional elements (e.g. ecosystem processes) (Noss 1990). The inclusion of the latter two elements implies that conservation of biodiversity must not simply attempt to retain native species and assemblages, but must also attempt to preserve the natural structure and functioning of these entities. In other words, habitats that have suffered human disturbance, such as drainage, burning, or soil disturbance, have experienced loss of biodiversity over and above any loss of species that may have occurred. Ecological linkage between compositional, structural, and functional elements also indicates that damage to one component will likely affect other components as well (Purvis & Hector 2000; Tilman 2000).

2.2.1.2 Biodiversity and conservation

Biodiversity, as discussed above, is an ecological concept and strictly speaking does not equate with conservation value. For example, the concept of biodiversity makes no distinction between native biodiversity and artificial diversity in the form of introduced species and altered ecosystems (Angermeier 1994). Also, the biodiversity of a site does not address such management concerns as levels of threat to the site or how the site fits into broader conservation goals. For these reasons, biodiversity *sensu stricto* should not be the sole criterion for determining the conservation value of habitats (Angermeier 1994; Gaston 1996; Puumalainen 2001).

Conservation practice in Ireland, as exemplified by the criteria used by the National Parks and Wildlife Service for selecting SAC sites, takes into account several factors including biodiversity (National Parks and Wildlife Service nd). SAC selection criteria include a comparative rating of the sites on the basis of :

- Representativity of the habitats in the site, or how well the site typifies the habitat type,
- The area of habitat on the site relative to the extent of the habitat in Ireland,
- The conservation status of the site with respect to the integrity of structure and function and restoration potential,
- The importance of Ireland for the habitat type in a European context,
- The presence of priority habitat types or species (i.e. those that are rare and in danger of disappearing in Europe),
- The presence of complexes of habitats and species listed in the EU Habitats Directive, and
- Additional criteria that may be site or habitat-specific.

The term “biodiversity” arose in the context of concerns about the destruction of natural habitats and the extinction of species on local and global scales (Gaston 1996). As such, use of the term in socio-political contexts is inextricably linked with the value of the natural world. In some cases, “biodiversity” is used as a synonym for nature conservation (Gaston 1996). For example, particular types or subtypes of habitat may be identified as important for biodiversity conservation because they contain high species diversity or because the habitat is rare or characteristic at a regional, national, or European level. It is this wider sense of the word “biodiversity”, incorporating both the variability of the natural world and its value, that is employed in the *Forest Biodiversity Guidelines* (Forest Service 2000c). When assessing the biodiversity of habitat types or subtypes in the context of afforestation, we will also consider the conservation value of these habitats. More generally, *our use of the term "biodiversity" in this report will mean both the variability of species and ecosystems and their conservation value*, in accordance with how "biodiversity" is used in management contexts.

2.2.2 Indicators

Because of its vast scope, it is generally impossible to evaluate and monitor all facets of biodiversity. For this reason, biodiversity research and management typically focuses on one or more indicators of biodiversity. Indicators can take many forms and be defined in many ways, depending, for example, on whether the indicator is to be used in shaping policy or management. In forest biodiversity research in Europe, indicators are often divided into structural, functional and compositional measures, corresponding to the three components of biodiversity discussed above (Ferris & Humphrey 1999; Hansson 2000; Puumalainen 2001). The use of individual species or groups of species, however, is the oldest and most often employed approach in forested and other habitats (Lindenmayer 1999; Lindenmayer *et al*. 2000; Simberloff 1998). Although more direct indicators of ecosystem structure and function are being developed, species can serve as indicators of these aspects of biodiversity as well as the compositional component. The use of lichens as indicators of air pollution is a well-known example (Richardson 1991).

When selecting one or more indicator species, it is important to specify exactly what the species are meant to indicate, for example species richness, the presence of certain rare species, or the lack of anthropogenic disturbance (Lindenmayer 1999). Once putative indicator species have been selected, rigorous validation should be conducted to ascertain that it actually indicates the attributes envisioned; however, such testing has seldom been conducted (Simberloff 1998; Simberloff 1999). Indicator development, therefore, remains a critical need in ecology and management (Noss 1999). An ideal indicator species would have the following attributes (compiled from Ferris & Humphrey 1999; Hansson 2000; Noss 1999):

1. Well-known taxonomy, distribution, and ecology, particularly with respect to the species and/or functions to be indicated,
2. Specialist species limited to the habitat in question,
3. Sensitive to ecosystem processes such that any changes are readily apparent,
4. Requires large areas to maintain viable populations,
5. Easily identifiable and readily sampled, and
6. Spatially, temporally and demographically balanced populations with poor dispersal abilities.

Certain species can be considered "*de facto*" indicators of biodiversity, in that they are particularly rare and may have legal protection. For plants in Ireland, there are four sources of information enumerating these species. Plants listed under the Flora (Protection) Order, 1999 are legally protected in the Republic of Ireland. Of these, three vascular plant species and two bryophyte species are also listed on Annex 2 of the Habitats Directive (92/43/EEC) as species of European interest whose conservation requires the designation of Special Areas of Conservation. *The Irish Red Data Book* (Curtis & McGough 1988) inventories rare and threatened vascular plants on the island as a whole and provides evaluations of their conservation status. Rare bryophytes in Ireland are identified by Holyoak (2003), and a Red List for bryophytes is currently being prepared.

A similar range of designations apply to fauna, although the coverage of species varies widely between groups. Most species of birds, most Irish mammals (except vermin), all the native herpetofauna, and three[1] species of invertebrates are protected under the Wildlife Acts, 1976 and 2000. The *Irish Red Data Book* (Whilde 1993) inventories the rare, threatened and internationally important non-marine vertebrate fauna and provides evaluations of their conservation status. However, the evaluation of the bird fauna in that work has now been superseded by the assessment of birds of conservation concern in Ireland (Newton *et al*. 1999). No Red Data Book has been published for any group of Irish invertebrates. However, the conservation status of the Irish hoverfly fauna has been assessed by Speight and Castella (2000). Eight Irish species of invertebrates, five fish species, and eight mammal species are listed on Annex 2 of the Habitats Directive (92/43/EEC) as species of community interest whose conservation requires the designation of Special Areas of Conservation, while 32 Irish bird species are listed on Annex 1 of the Birds Directive (79/409/EEC) as species whose conservation requires the designation of Special Protection Areas.

Perhaps the oldest formal use of indicator species is in the field of phytosociology. Phytosociology attempts to classify vegetation in a hierarchical scheme based on floristic composition, particularly the presence or absence of "character species." The fundamental unit in phytosociology is the "association", a plant community of definite floristic composition recurring in uniform environmental conditions (White & Doyle 1982). The phytosociological association is analogous to a taxonomic species, particularly in that an association is an abstract unit to which samples in the field are referred. Associations are grouped into alliances, which are in turn grouped into orders, which are combined at the highest level of the hierarchy into classes; associations may also be divided into subassociations. "Syntaxon" and "nodum" are non-specific terms used for phytosociological groupings, similar to the use of "taxon" in systematics. Each of these hierarchical levels is identified with one or more character species, i.e. species that are relatively faithful to the syntaxon and occur relatively frequently. Syntaxa (except classes) may also contain differential species, which aid in the discrimination between similar groups. It is important to note, however, that the association is not defined simply by character species, but by its complete species complement.

The Braun-Blanquet system of phytosociology has been the most commonly used system of habitat/vegetation classification in Ireland. One advantage of the approach is the fact that it is an international classification system, thus facilitating comparison among countries. However, the application of phytosociological associations defined in other parts of Europe to Irish vegetation presents difficulties. Because the Irish flora is relatively species-poor, many Irish species may occupy niches filled by other species in continental Europe. Adaptation of species to Irish conditions may also render useless some character species identified elsewhere. These arguments also apply to the adoption in Ireland of other types of indicator species selected in other countries, for example the ancient woodland indicator species identified in Lincolnshire by Peterken (1993). Local character species have been named for some associations in Ireland to circumvent these problems. A good example is *Schoenus nigricans*, which is a character species of Atlantic blanket bog in Ireland (White & Doyle 1982), but not in Britain, where *S. nigricans* is typical of minerotrophic fens (Rodwell 1991). Some associations in Ireland, however, have received very little or no study (White & Doyle 1982), and so the indicator value of character species identified in other countries is uncertain. Accordingly, White and Doyle (1982) have used the term "diagnostic species" for

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[1] The relevant legislation lists three species (S.I . No. 112 of 1990). However, one of these ("Freshwater pearl mussel") encompasses two taxa (*Margaritifera margaritifera* and *M. (m.) durrovensis*) that are listed separately in the Habitats Directive.

such character species, while suggesting that further work needs to be done to elucidate the value of such species in a phytosociological context.

In the following sections, we identify species that are potential indicators of biodiversity. Following Larsson (2001) and Noss (1990), we define *biodiversity indicator* as a "measurable surrogate" for biodiversity at a given level of ecological organisation– the ecosystem (and in some cases landscape) level in this study. Where appropriate, we also identify structural and functional features as potential indicators of biodiversity (Noss 1990; Larsson 2001). In later chapters, we test these potential indicators using field data and also identify new provisional indicators. Our procedure is as follows:

1. Identify grassland and peatland habitats of high biodiversity value from the literature,
2. Identify plant and bird species and structural and functional features characteristic of these habitats as *potential biodiversity indicators* (as invertebrate species are generally difficult for non-specialists to sample and identify, invertebrate species will not be used as biodiversity indicators),
3. Determine whether these potential indicators can effectively discriminate between high and low biodiversity assemblages of the taxonomic groups we surveyed in the field and identify these as *confirmed biodiversity indicators*, and
4. Identify additional *provisional biodiversity indicators* newly determined from the results of our field survey.

It should also be noted that our choice of taxonomic groups in this study, plants, spiders, hoverflies and birds, assumes that they are correlated with (indicate) biodiversity of other species groups not surveyed. The justification for this assumption is discussed at length by Smith *et al.* (2005).

2.3 GRASSLAND

2.3.1 Grassland Habitats and Ecology

A broad range of grassland types exist in Ireland, derived from different parent materials and soils of varying quality and subject to a variety of environmental variables. Irish grasslands were principally formed as a result of woodland clearance, and continued disturbance, primarily grazing, is necessary to keep them from reverting to scrub and later woodland (Mitchell & Ryan 1997; O'Sullivan 1968b). The type of grassland formed is primarily dependent upon climate and soils and strongly influenced by human management (Alard *et al*. 1994; O'Sullivan 1968b). To a large degree, increasing intensity of management corresponds with increasing poverty of species richness (O'Sullivan 1965; O'Sullivan 1968b). Improved grasslands are intensively managed pasture which consists of increased quality herbage and decreased plant species diversity (Alard *et al*. 1994; Fossitt 2000). They are typically reseeded at intervals, fertilised, drained, subjected to weed control using herbicides and grazed or cut at high stocking levels or frequencies. Conversely, semi-natural, unimproved grasslands have generally undergone low input management and in many cases contain a more diverse and representative Irish flora, and indeed some rare and endangered species (Curtis & McGough 1988). A management regime between intensive agricultural production and complete abandonment promotes higher biodiversity in semi-natural grasslands, such as annual cutting and/or low-intensity grazing (Byrne *et al*. 1997; Fossitt 2000).

Grasslands are grouped under the Irish habitat classification system into improved grassland (GA), semi-natural grassland (GS) and freshwater marsh (GM) (Fossitt 2000). Improved grasslands are further subdivided into **improved agricultural grassland** (GA1) and **amenity grassland** (GA2). Semi-natural grasslands are subdivided into **dry calcareous**

and neutral grassland (GS1), **dry meadows and grassy verges** (GS2), **dry-humid acid grassland** (GS3) and **wet grassland** (GS4). **Marsh** (GM1) is not divided into further categories. **Improved agricultural grassland** (GA1) are intensively managed, highly modified grasslands that receive heavy grazing or silage-making use. **Amenity grasslands** (GA2) include those highly managed grasslands not used for agriculture, including lawns, golf courses and playing pitches. As such grasslands are unlikely to be afforested and are in general ecologically equivalent to GA1 grasslands, they will not be considered further. **Dry calcareous and neutral grassland** (GS1) are unimproved or semi-improved dry grasslands used for low intensity grazing. They occur on calcareous or circumneutral well-drained soils in lowland areas. **Dry meadows and grassy verges** (GS2) with high cover of coarse, tussocky grasses are distinguished from the previous category by absence of grazing. Dry grassy areas, such as road verges and cemeteries, that are mown once or twice annually fall under this category. Unimproved or semi-improved grasslands on dry or damp acid soils are classified as **dry-humid acid grassland** (GS3). They most frequently occur in upland areas on podzols and peaty podzols, but also on sandy, acid soils in lowland areas. **Wet grasslands** (GS4) include semi-natural grasslands occurring on wet or waterlogged mineral or organic soils. These often intergrade with **marsh** (GM1), which is distinguished from the former on the absence of herbs characteristic of drier grassland and less than 50% cover of grasses and sedges.

Small, fragmented pockets of semi-natural grasslands are frequently found in a more improved matrix (Cooper *et al*. 1995), representing a substantial portion of the biodiversity of the wider landscape. Such fragments can easily be lost due to intensification, dereliction or conversion to a different land-use (Byrne 1996; Cooper *et al*. 2003; Cooper *et al*. 1995; O'Sullivan 1965; O'Sullivan 1968b). Intensification, through such management practices as reseeding, drainage, fertilisation and higher livestock densities, reduces diversity and increases the dominance of a few favoured species, such as *Lolium perenne* and *Trifolium repens*. Dereliction or excessive poaching by livestock can lead to invasion by bracken or scrub. One form of conversion is afforestation; farmers seeking to afforest part of their land are much more likely to select the low-productivity, high diversity pastures than improved pastures (Heritage Council 1999). Conversion to tillage, quarrying operations in eskers and moraines and dumping or small landfills are additional causes of semi-improved grassland loss (Byrne 1996; Cooper *et al*. 1995).

2.3.2 Distribution of Grasslands in Ireland

Attempts have been made to identify the cover of the various grassland types in the Republic of Ireland, including the CORINE project, but these were unsatisfactory in the provision of accurate estimates of cover for unimproved, semi-natural grassland types (Byrne 1996). Due to the mosaic nature of habitats in the landscape, it is very difficult to differentiate between cover of grassland and other habitats, particularly when considering habitat cover on a national or regional level. CORINE estimates of grassland cover are therefore presented as a broad classification including other habitats such as arable and woodland. These figures show "pastures" occupying 57.26% of land cover in the Republic of Ireland and "natural grassland" at approximately 3% (natural grasslands are low productivity, naturally seeded areas with some moor-type grasslands). "Agricultural land with natural vegetation" covers approximately 5% of the country and this category may include some semi-natural wet grassland but also woodland and scrubby habitat (Byrne 1996).

O'Sullivan (1982) estimated that 37.1% of the island of Ireland is improved or semi-improved grassland on well-drained, deep soils, equivalent to **improved agricultural grasslands** (GA1) and more fertile examples of **dry calcareous and neutral grassland** (GS1) in the Irish habitat classification scheme. These grasslands are generally found on high quality mineral soils throughout the country but are particularly common in the east and southeast. **Dry calcareous and neutral grasslands** on shallow limestone soils occupy a further 8.7% of Ireland, primarily in the midlands (O'Sullivan 1982). More species-rich, highly calcareous grassland types occur on eskers and limestone karst terrain, such as the Burren, occupying only 0.3% of the island. **Dry-humid acid grassland** (GS3) occurs in upland areas throughout Ireland, occupying roughly 2.5% of the land. It is often found in intimate mixes with **dry siliceous heath** (HH1) or blanket bog (PB2, PB3) (Fossitt 2000). Such grasslands have frequently been afforested by the State in the past, but O'Sullivan (1982) considered that most of their coverage had remained intact as unfenced commonage in 1982. **Wet grasslands** (GS4) and **marshes** (GM1) are also widely distributed, comprising about 19.5% of the land (O'Sullivan 1982). They are most widespread in the west and in drumlin regions, but also near lakeshores and rivers, such as the Shannon callows (Heery 1991), throughout the island. **Dry meadows and grassy verges** (GS2) traditionally developed in fields managed for hay-making. As such management has virtually ceased in the present time, dry meadows are quite rare; however, small examples of this habitat type can be found on roadsides, field margins, cemeteries and railway embankments scattered throughout the country (Fossitt 2000). The above estimates, however, are out-of-date; Byrne (1996) notes that "since the time of O'Sullivan, all semi-natural communities have probably declined in extent, due mainly to.. increased intensification, dereliction and conversion of grasslands to another land-use, such as commercial forestry."

Landscape ecological surveys in different regions of Northern Ireland from 1987-1992 show that grasslands overall occupied approximately 60.3% of the total land cover of Northern Ireland, but that unimproved, semi-natural grasslands occupied only 4.1% (Cooper & McCann 1994). While the exact figures will differ between the north and south of Ireland because of differences in geology and other factors, semi-natural grasslands probably occupy a similarly small proportion of total land area. A resurvey in 1998, using UK Broad Habitat categories, showed that improved grasslands had increased in area, largely through the intensification of other grassland types and conversion of former tillage (Cooper *et al.* 2003) (Table 1). "Neutral grasslands", a heterogeneous group including semi-improved agricultural land on wet, clay-rich soils, species-rich dry grassland and ruderal communities, suffered the greatest loss. "Fen, marsh and swamp", primarily "marshy grasslands" but also including fens and swamps, also declined significantly through agricultural intensification and afforestation. These land-use changes are almost certainly reflected in the Republic of Ireland.

Table 1. Estimate of grassland area (% of total land area) in 5 broad habitat categories in Northern Ireland in 1998 and % change from 1987-1992 baseline values. Data from Cooper *et al.* (2003). Also shown are the probable equivalent Heritage Council habitat classification codes (Fossitt 2000).

UK Broad Habitat	HC Habitat Code	1998 Cover (%)	Change (%)
Improved Grassland	GA1	42.0	+ 33
Neutral Grassland	GS1 + GS4? + others	18.7	- 32
Calcareous Grassland	GS1	0.1	- 7
Acid Grassland	GS3	2.1	- 8
Fen, Marsh & Swamp	GS4 + GM1 + others	3.9	- 19

2.3.3 Vegetation

2.3.3.1 Overview

Irish inland grasslands fall into three phytosociological classes: Molinio-Arrhenatheretea, the class of lowland meadows and pastures, Nardetea, predominantly upland acid grasslands and Festuco-Brometea, dry limestone grasslands (O'Sullivan 1982). Character species of the classes are given in Table 2, and correspondences between syntaxa, as described by O'Sullivan (1982), and the Irish habitat classification scheme (Fossitt 2000) are shown in Table 3.

Table 2. Character species of the Irish grassland classes Molinio-Arrhenatheretea, Nardetea and Festuco-Brometea (O'Sullivan 1982).

Class	Character Species	
Molinio-Arrhenatheretea	*Holcus lanatus* *Cerastium fontanum* *Plantago lanceolata* *Rumex acetosa* *Poa pratensis* *Lathyrus pratensis* *Prunella vulgaris* *Festuca pratensis*	*Trifolium pratense* *Ranunculus acris* *Festuca rubra* *Poa trivialis* *Cardamine pratensis* *Alopecurus pratensis* *Vicia cracca*
Nardetea	*Nardus stricta* *Luzula multiflora* *Veronica officinalis* *Lathyrus linifolius*	*Danthonia decumbens* *Carex pilulifera* *Festuca vivipara*
Festuco-Brometea	*Anthyllis vulneraria* *Blackstonia perfoliata* *Bromus erectus* *Carlina vulgaris* *Filipendula vulgaris* *Gentianella amarella* *Sesleria caerulea*	*Avenula pubescens* *Centaurea scabiosa* *Leontodon hispidus* *Homalothecium lutescens* *Koeleria macrantha* *Sanguisorba minor*

Table 3. Correspondence between the classification of grasslands under the Irish habitat classification (Fossitt 2000) and phytosociological syntaxa (O'Sullivan 1982).

Habitat	Order	Association
GA1 improved agricultural grassland	Arrhenatheretalia elatioris	Lolio-Cynosuretum
GA 2 amenity grassland	Arrhenatheretalia elatioris	Lolio-Cynosuretum
GS1 dry calcareous and neutral grassland	Arrhenatheretalia elatioris Brometalia erecti	Centaureo-Cynosuretum (several)
GS2 dry meadows and grassy verges	Arrhenatheretalia elatioris	Vicio-Arrhenatheretum
GS3 dry-humid acid grassland	Nardetalia	(several)
GS4 wet grassland	Molinietalia caeruleae	Junco acutiflori-Molinietum Senecioni-Juncetum acutiflori
GM1 marsh	Molinietalia caeruleae	Senecioni-Juncetum acutiflori (Filipendulion associations)

The Nardetea is represented by one Irish order, the Nardetalia, and while several schemes have been proposed for classifying lower levels of the hierarchy, none have been completely satisfactory (O'Sullivan 1982; White & Doyle 1982). Nardetalia grasslands are typically dominated by swards of *Agrostis capillaris, Festuca rubra* and *Anthoxanthum odoratum*, (O'Sullivan 1982) and presence of *Carex binervis* is characteristic in Ireland (White & Doyle 1982). Mosses, such as *Rhytidiadelphus squarrosus* and *Pseudoscleropodium purum*, are also often found (O'Sullivan 1982). Nardetalia grasslands typically occur on podzols or brown podzolics in cool, damp, upland situations. Changes in grazing regime can lead to invasion by bracken or gorse species.

Festuco-Brometea grasslands are represented in Ireland by the order Brometalia erecti and the alliance Mesobromion, further subdivided into three associations (O'Sullivan 1982). Brometalia grasslands are found in regions with shallow, limestone-derived soils and comprise the "calcareous" component of the Irish habitat **dry calcareous and neutral grassland** (Table 3). They can be very rich in species, supporting up to 45 per 4 m^2 plot (O'Sullivan 1982). Such grasslands are generally used for rough grazing; where grazing pressure is light, *Corylus avellana, Prunus spinosa* and *Crataegus monogyna* tend to spread.

Lowland grasslands of the Molinio-Arrhenatheretea in Ireland fall into two orders, four alliances and five associations (Figure 1). The character species of the two orders, Arrhenatheretalia elatioris and Molinietalia caeruleae are shown in Table 4. Many of the character species may actually be missing in Molinietalia communities. Arrhenatheretalia communities correspond with four of the improved and dry semi-natural grassland habitats in the Irish habitat classification (Table 3). They are subdivided into two alliances primarily differentiated by management, the Arrhenatherion elatioris and the Cynosurion cristati (O'Sullivan 1982). Grasslands in the Molinietalia caeruleae are also divided into two alliances, the Junco conglomerati-Molinion, occurring mainly on acid soils, and the Calthion palustris on limestone and shales (O'Sullivan 1982). Two additional alliances in the class have been recorded in Ireland: the Juncion acutiflori and the Filipendulion. The former is of doubtful status and limited distribution, whereas the latter is a tall-herb community of lake and river margins with less than 25% grass cover (O'Sullivan 1982). Molinietalia communities correspond with the Irish habitats **wet grassland** and **marsh** (Table 3). **Marsh** habitats would include Filipendulion communities and perhaps more herb-rich representatives of communities referable to the Calthion, which support 40-70% grass cover (O'Sullivan 1968a). More grass-rich Calthion communities and communities referable to the Junco acutiflori-Molinietum in the Juncion alliance would be classified as **wet grassland** habitats (Table 3).

Table 4. Character species for the orders of the class Molinio-Arrhenatheretea: Arrhenatheretalia elatioris and Molinietalia caeruleae.

Order	Character Species	
Arrhenatheretalia elatioris	*Bellis perennis*	*Taraxacum officinale*
	Dactylis glomerata	*Veronica chamaedrys*
	Trisetum flavescens	*Leucanthemum vulgaris*
Molinietalia caeruleae	*Juncus acutiflorus*	*Lythrum salicaria*
	Juncus effusus	*Lychnis flos-cuculi*
	Cirsium palustre	*Angelica sylvestris*
	Filipendula ulmaria	*Achillea ptarmica*
	Juncus conglomeratus	*Equisetum palustre*
	Senecio aquaticus	*Myosotis laxa*
	Lotus pedunculatus	*Deschampsia cespitosa*

2.3.3.2 *Dry meadows and grassy verges GS2*

The alliance Arrhenatherion elatioris includes oatgrass meadow communities on well-drained, deep and fertile soil (Figure 1). They correspond to the Irish habitat **dry meadows and grassy verges** (Table 3). They develop where grasslands are managed by hay-making or similar low-intensity usage, and are therefore very rare in modern farmland (O'Sullivan 1968b). Roadsides, railways, cemeteries and other places mown once yearly are typical situations in which the alliance can be found (O'Sullivan 1982). Tall grasses, including *Arrhenatherum elatius* and *Dactylis glomerata*, tall herbs, especially *Centaurea nigra*, and low climbers are prominent members of the community. Such communities support only an average of 23 species in a 4 m^2 plot (O'Sullivan 1982). The alliance is represented in Ireland by one association, formerly called the Centaureo-Arrhenatheretum, (O'Sullivan 1965) but perhaps more correctly named as the Vicio- Arrhenatheretum (O'Sullivan 1982). *Vicia sepium* is the lone differential species of the alliance, distinguishing it from the Cynosurion cristati.

2.3.3.3 *Improved agricultural grasslands GA1 and dry calcareous and neutral grassland GS1*

The alliance Cynosurion cristati includes at least moderately fertile grasslands on relatively well-drained loamy soils that receive more intensive management than the previous alliance. Differential species from the Arrhenatherion elatioris include *Lolium perenne, Cirsium arvense, Achillea millefolium* and *Odontites vernus*. Character species are *Cynosurus cristatus, Senecio jacobaea, Trifolium repens* and *Phleum pratense* (O'Sullivan 1982). The Lolio-Cynosuretum association represents the highly improved, species poor grasslands dominated by *Lolium* spp and *Trifolium repens* and corresponds with the Irish habitat type **improved agricultural grassland** (Table 3, Figure 1). Tall herbs are generally absent, and many of the typical agricultural weeds are rare or absent (O'Sullivan 1968b). In Limerick, O'Sullivan (1968b) recorded a mean of 24 species from 25 m^2 (i.e. 5 × 5 m) plots. This community is distinguished from the Centaureo-Cynosuretum by the absence of differential species of the latter association (O'Sullivan 1982). Three poorly distinguished subassociations have been recognised (O'Sullivan 1982).

Centaureo-Cynosuretum grasslands are of moderate fertility, with more diversity in grasses and weeds than the previous association, and include the less base-rich members of the Irish habitat **dry calcareous and neutral grassland** (Table 3, Figure 1). Differential species from the Lolio-Cynosuretum are: *Hypochaeris radicata, Carex flacca, Lotus corniculatus, Centaurea nigra, Luzula campestris* and *Rhytidiadelphus squarrosus (O'Sullivan 1982)*. Three subassociations that may be distinguished differ in soil depth, fertility and drainage (O'Sullivan 1968b; O'Sullivan 1982). The galietosum subassociation occurs on well-drained limestone soils; it is the most species-rich subassociation, averaging 37 species/25 m^2 plot in Limerick (O'Sullivan 1968b). The juncetosum subassociation is found on the more poorly drained soils of moderate fertility, and therefore often occurs in complexes with Molinetalia communities. Its differential species are some of those frequently occurring in Molinetalia grasslands, and *Juncus effusus* is often a prominent member. Intermediate grassland types are referable to a typicum subassociation (O'Sullivan 1982). Byrne (1996) estimates that dry galietosum grasslands are the most abundant semi-natural grassland type in Leinster, but that their extent is declining through intensification and other factors (Section 2.3.2).

More calcareous types of GS1 grassland are briefly described above (Section 2.3.3.1).

2.3.3.4 *Wet grasslands GS4*

In the Molinietalia, Junco conglomerati-Molinion grasslands (Figure 1) are found on poorly drained, less fertile gleys, peaty gleys and reclaimed blanket peats. In the west, they often occur as an ecotonal community between blanket bogs and more improved grassland types (O'Sullivan 1982). One association has been defined in Ireland, the Junco acutiflori-Molinietum. Character and differential species are the same for the association and alliance (Table 5) and are indicative of wet, acidic conditions. The dominant members of the community are the tall rush species *Juncus acutiflorus, J. effusus* and *J. conglomeratus*, and *Dactylorhiza* spp orchids are sometimes conspicuous. Hedgerows among fields of this grassland type are usually dominated by *Ulex europaeus* and *Salix* spp (O'Sullivan 1965). Junco-Molinietum communities are one of the most species rich grassland communities in Ireland, averaging 32 species/25 m^2 plot (O'Sullivan 1965) and 40 species/25 m^2 plot in Co. Limerick (O'Sullivan 1968b). Four subassociations have been proposed (O'Sullivan 1982):

1. *Trifolium repens* subassociation: widespread, includes regularly grazed grasslands that partially dry out in the summer. Differential species: *Trifolium repens. Cynosurus cristatus, Trifolium pratense, Lotus corniculatus, Briza media, Phleum pratense.*
2. *Agrostis canina* subassociation: comprises the least grazed, wettest, most oligotrophic sites. Differential species: *Agrostis canina* s.l., *Viola palustris, Pedicularis sylvatica, Carex viridula.*
3. *Plantago maritima* subassociation: occurs near the coast in Galway and Clare. Differential species: *Plantago maritima.*
4. *Salix repens* and *Lathyrus pratensis* subassociation: known from one site on the southeast coast. Differential species: *Salix repens* and *Lathyrus pratensis*.

Table 5. Character and differential species of alliances and associations of grasslands in the order Molinietalia caeruleae

Association	**Character Species**	**Differential Species**
Junco acutiflori-Molinietum	*Succisa pratensis* *Potentilla anglica* *Juncus conglomeratus* *Cirsium dissectum*	*Potentilla erecta* *Danthonia decumbens* *Nardus stricta* *Molinia caerulea* *Carex panicea* *Carex nigra* *Carex echinata* *Carex pulicaris* *Pseudoscleropodium purum* *Thuidium tamariscinum* *Hylocomium splendens*
Senecioni-Juncetum acutiflori	*Caltha palustris** *Bromus racemosus** *Crepis paludosa**	*Poa trivialis* *Senecio aquaticus* *Potentilla anserina* *Agrostis stolonifera* *Lolium perenne* *Carex hirta*

* "None of these is widespread in Irish wet grasslands. The other good continental Calthion species... are even rarer" (O'Sullivan 1982).

Calthion palustris grasslands are represented in Ireland by one poorly-defined association, the Senecioni-Juncetum acutiflori (Figure 1) (O'Sullivan 1982). They are found in wet meadows on moderately fertile gley, alluvial or relatively base-rich fen peat soils. Senecioni-Juncetum grasslands tend to occur on more continuously wet and base-rich soils than the Junco-Molinietum, such as those derived from limestone or shale, (White & Doyle 1982) but can also form mosaics with them (O'Sullivan 1965). Character species of the alliance in continental Europe are rare in Ireland, and so the Senecioni-Juncetum is probably better distinguished from the Junco-Molinietum by the association differential species (Table 5) (O'Sullivan 1982). Such grasslands are typically dominated by *Juncus effusus* or *Juncus acutiflorus* and may include other tall rush species. O'Sullivan (1965) has recorded an average of 24.8 species/4 m^2 plot in the community. Two subassociations have been distinguished: a slightly drier, more fertile *Trifolium repens* subassociation, and a less-managed *Ranunculus flammula* subassociation (O'Sullivan 1982). Differential species of the latter are *Ranunculus flammula, Agrostis canina, Hydrocotyle vulgaris* and *Viola palustris*. Eakin (1995) described two particularly species-rich communities in Fermanagh referable to the Senecioni-Juncetum. The first comprises "wet *Caltha* meadows", averaging 24.4 species/4 m^2 plot, that are characterised by *Lychnis flos-cuculi, Caltha palustris, Ranunculus flammula, Carex nigra* and *Festuca pratensis*. The second includes "species-rich wet meadows", supporting an average of 25.7 species/4 m^2 plot, and is characterised by *Senecio aquaticus, Prunella vulgaris, Luzula campestris, Lychnis flos-cuculi* and *Dactylorhiza fuchsii*.

Byrne (1996) included wet grasslands in her survey of semi-natural grasslands in Leinster. Her ordination and clustering of semi-natural grasslands produced one heterogeneous wet grassland group that included communities with affinities to the Centaureo-Cynosuretum juncetosum subassociation, communities with affinities to Senecioni-Juncetum grasslands and intermediate communities. "Overall the wet grassland group was poorly represented in Leinster. Its classification was unsatisfactory and further sampling is required to ascertain more fully its current status with respect to the different community types" (Byrne 1996).

2.3.4 Birds

2.3.4.1 Wet grasslands

Among the best examples of this habitat type in Ireland are the internationally important Shannon callows, 3400 ha of semi-natural grassland, which flood in winter and spring. The pastures and hay meadows which make up the callows are botanically rich and have never been ploughed or reseeded. They contain one of the three main concentrations of wet grassland bird species in Ireland and Britain. Because of their importance, their bird communities have been relatively well-studied and Irish literature relating to birds using this habitat type stems from research in this region (Hutchinson & O'Halloran 1994).

In particular, four wader species breeding on the Shannon callows: Snipe, Curlew, Lapwing and Redshank were studied in detail between 1987-89 (Herbert *et al*. 1990). In one of the survey years, 34 km^2 were searched for breeding waders and 1551 pairs of the four key species were found (Nairn *et al*. 1988). Over the 3 breeding seasons, the distribution of Snipe was influenced by wetness of the ground and their distribution was therefore sensitive to timing of flooding and rainfall, while the other three species did not respond to such changes. Curlew and Snipe selected hay meadow for nesting while Lapwing and Redshank nested preferentially in pasture. Lapwing were associated particularly with areas grazed by sheep and horses, where the sward was very low while Redshank used low intensity grazed areas close to muddy river banks and ditches suitable for feeding (Herbert *et al*. 1990). Other species associated with the grasslands in summer included Black-tailed Godwit, nesting in a modified fen which had not been grazed in living memory, Shoveler, Quail and Whinchat (5 pairs bred in areas with rank growth of *Molinia caerulea* and *Dactylis glomerata* (Nairn *et al*. 1988)).

The callows also provide ideal habitat for the globally threatened Corncrake. A doctoral study of the species included an investigation of the types of grassland selected for use by the species (Tyler 1996). The study used radio-tracking of individuals to investigate habitat use. Grassland types were based on species composition, grazing level and flooding. It was found that the species used a suite of different grassland types throughout the season. Extensive use was made of hay meadows (both dry and wet), grassy margins, and areas dominated by reed canary-grass (*Phalaris arundinacea*). Pasture and areas dominated by rushes and sedges were not selected. The vegetation types selected were those that exhibited the most growth and hence the most cover. Vegetation types lower than 20cm height were never selected. A study is currently being completed on the ecology of corncrakes in the Shannon Callows (Donaghy 2006).

In winter the Shannon callows are internationally important as feeding grounds for wildfowl and waders (Crowe 2005). For example, during the above-mentioned breeding wader survey large numbers of Whimbrel were observed feeding on spring migration during April and early May (Nairn *et al*. 1988). Also, migrating Greenland White-fronted Geese use wet callows associated with bog (see later section), feeding on *Agrostis stolonifera* and corms of *Ranunculus bulbosus* (H.G. Wilson in Cross 1990).

2.3.4.2 Improved grasslands

The literature on birds of improved grassland may conveniently be divided into that relating to: a) direct use of the grassland and b) hedgerow use.

2.3.4.2.1 Direct use of improved grassland

This is generally a species-poor habitat and species typical of other types of farmland may avoid it. For example, in an Irish survey of habitat use by Stonechat in Ireland, improved grassland held the lowest densities of the species of all habitat types in both winter and summer (Cummins 2001).

The only references to direct use of improved pasture by birds found in the course of this review concerned winter feeding by waders, in particular Black-tailed Godwits (Hutchinson & O'Halloran 1994), and wildfowl. A study of winter use of agricultural land by Brent Geese around Strangford Lough in Co. Down in the winters of 1992/3 and 1995/6 found that 89% of the fields used by the geese were of improved pasture, with new-sown pasture and winter cereals also used (Andrews *et al*. 1996). All of the fields used were very close to the sea and not separated from it by any visual barriers such as hedges, so this usage is limited to coastal areas. During a winter survey of swans in 1990/91, both Whooper and Bewick's Swans were found to make increasing use of improved pasture for feeding as winter progressed, while Mute Swans kept to inland water bodies (Rees *et al*. 1997). Black-tailed Godwits were shown to use the improved pastures and fields to supplement mid-winter feeding (Hutchinson & O'Halloran 1994).

2.3.4.2.2 Breeding community of hedgerows in improved pasture

Probably because many hedgerows have disappeared from the Irish landscape, the bird communities of Irish hedgerows have received some attention. Three studies of farmland bird communities were consulted for this review, two involving sampling of hedgerows in mixed agricultural or mainly pastoral landscapes in both winter and summer (Holt 1996; Moles & Breen 1995) and one using territory mapping methodology to look at the breeding community of all parts of the farmland (mainly improved grassland with tillage, scrub and hedgerow) in summer only (Lysaght 1989).

In total, 32 species were recorded in all parts of the 5 improved pasture plots surveyed in mid-west Ireland by Lysaght (1989), with 21 occurring in half or more of the plots. Wren was by far the most abundant species, with Robin, Blackbird, Dunnock and Chaffinch also dominating. Holt recorded only 25 species in summer, in hedgerows adjacent to Kilcolman Nature Reserve, Co. Cork. The first five common species were identical to those found by

Lysaght (1989). Moles and Breen (1995) also recorded 32 species during their summer surveys of hedgerows in mixed agricultural land.

Rarer breeding species in Lysaght's study could be divided into two categories: 1, those species occurring at low densities throughout the region (these included Treecreeper, Pied Wagtail, Goldfinch and Linnet) and 2, those species having breeding requirements seldom satisfied in farmland (e.g. Chiffchaff, Grey Wagtail, Snipe, Skylark). Moles and Breen (1995) observed that hedgerow requirements for breeding rare and common birds differed. Many common species of passerine responded to greater height and width of hedgerows while some uncommon species preferred hedgerows with trees (e.g. Goldfinch, Spotted Flycatcher and Chiffchaff). At Kilcolman, hedgerow volume and width were found to explain most variation in breeding bird communities (Holt 1996). Holt (1996) observed a very low proportion of migratory species in his summer hedgerow communities (around 4%). Less frequent summer (and autumn) species at his site included Treecreeper, Spotted Flycatcher, Sedge Warbler, Blackcap and Stonechat. Hedgerows were also found to be important breeding sites for Song Thrushes (Kelleher 2005; Kelleher & O'Halloran in press) and for Robins (Fennessy 2001).

2.3.4.2.3 *Non-breeding community of hedgerows in improved pasture*

In winter, Moles and Breen (1995) found the presence of wet ditches in association with hedgerows to be a major factor influencing bird densities. Of a total of 37 species recorded in winter, less frequent ones included Skylark, Grey Wagtail and Mistle Thrush. Holt (1996) also recorded more species (40) in winter than in summer and noted an increase in the proportion of migrants in the winter hedgerow community (approximately 20%). He found hedge height to be a more important factor in determining winter hedgerow communities than in summer. The commonest species in winter were Redwing, Chaffinch, Robin, Wren, Song Thrush, Fieldfare, Blackbird, Goldcrest, Dunnock and Blue tit. More rarely encountered species included Redpoll, Stonechat, Yellowhammer, Blackcap, Brambling, Siskin and Linnet.

2.3.5 Conservation

2.3.5.1 *Designated Habitats*

Calcareous grasslands in the Irish habitat type GS1 supporting high abundances or diversity of orchids correspond to the Habitats Directive Annex I priority habitat "semi-natural dry grasslands and scrubland facies on calcareous substrates (Festuco-Brometea) (*important orchid sites) 6210". GS1 grasslands with scattered juniper also correspond to the Annex I habitat "*Juniperus communis* formations on heaths or calcareous grasslands 5130". **Dry meadows and grassy verges** (GS2) correspond to the annexed habitat "lowland hay meadows (*Alopecurus pratensis, Sanguisorba officinalis*) 6510". Some **dry-humid acid grasslands** (GS3) may be referable to the Annex I priority habitat "species rich *Nardus* grasslands on siliceous substrates in mountain areas 6230". Junco-Molinion sites in the Irish habitat type **wet grasslands** (GS4) could be included in the annexed habitat "*Molinia* meadows on calcareous, peaty or clayey-silt-laden soils (Molinion caeruleae) 6410" if they are not species-poor or on degraded peaty soils (European Commission 1999; Fossitt 2000).

2.3.5.2 *Rare and Protected Species*

2.3.5.2.1 *Plants*

Several species of vascular plants which are rare or protected occur in wet grassland habitats, and possibly in improved grasslands. These are listed in Table 6 together with details of their habitat preferences, distribution, and conservation status.

Table 6. Protected or rare plant species that may occur in improved and wet grassland habitats in Ireland. Conservation Status indicates if the species is protected by the Flora (Protection) Order, 1999 in the Republic of Ireland or the Wildlife (NI) Order, 1985 in Northern Ireland. If not, Conservation Status indicates if it is listed as endangered, vulnerable or rare in the Irish Red Data Book (Curtis & McGough 1988).

Species	Common Name	Habitat Details[1]	Distribution[2]	Conservation Status
Bromus racemosus	Smooth brome	Water meadows, hayfields	Scattered throughout	Red Data: rare
Calamagrostis epigejos	Wood small-reed	Damp, rocky ground	West and Derry	1999 Flora Protection Order
Colchicum autumnale	Meadow saffron	Damp meadows, riverbanks	Nore valley	1999 Flora Protection Order
Hordeum secalinum	Meadow barley	Pastures on damp, heavy soils	Mainly south half near coast	1999 Flora Protection Order
Juncus compressus	Round-fruited rush	Damp, alluvial grassland	Meath, Roscommon, Longford	Red Data: rare
Mentha pulegium	Penny royal	Damp, sandy places	Mainly southwest	1999 Flora Protection Order and 1985 Wildlife NI Order
Oenanthe pimpinelloides	Corky-fruited dropwort	Flood meadows and pastures	Clare, Kerry	Red Data: rare[3]
Orchis morio	Green-winged Orchid	Meadows, pastures and sandhills	Mainly centre and east	1985 Wildlife NI Order
Poa palustris	Swamp meadow-grass	Wet grassland	North	Red Data: rare
Pseudorchis albida	Small white orchid	Upland pastures, wet grassland, heath	Scattered, mainly northern	1999 Flora Protection Order and 1985 Wildlife NI Order
Sanguisorba officinalis	Great burnet	Lake margins, dry banks	Mayo and northeast	1999 Flora Protection Order
Scrophularia umbrosa	Green figwort	Riverbanks and lakeshores	Dublin, Kildare, Derry, Limerick, Fermanagh	Red Data: vulnerable
Sibthorpia europea	Cornish moneywort	Wet grassland, streamsides	Kerry	Red Data: rare
Sisyrinchium bermudiana	Blue-eyed grass	Damp meadows, lakeshores	West and northeast	1985 Wildlife NI Order
Spiranthes romanzoffiana	Irish lady's tresses	Damp meadows, lakeshores, boggy ground	West and north	1999 Flora Protection Order and 1985 Wildlife NI Order
Trollius europaeus	Globe flower	Riverbanks and lakeshores	Leitrim, Donegal, Fermanagh	1999 Flora Protection Order and 1985 Wildlife NI Order
Viola persicifolia	Fen violet	Damp, periodically flooded grassland	Clare, Galway, Lough Erne/ Shannon basin	1985 Wildlife NI Order

[1] Details compiled from Curtis and McGough (1988). and Webb *et al.* (1996).

[2] Summarised from (Preston *et al.* 2002).

[3] Formerly protected in Republic of Ireland, possibly introduced.

2.3.5.2.2 Birds

Lowland wet grasslands, such as the Shannon callows, are a threatened habitat, and they attract high proportions of wader populations (Herbert *et al*. 1990). The Corncrake is a species of global conservation concern (Tucker & Heath 1994) and is listed on the Irish Red List for bird species of high conservation concern (Newton *et al*. 1999). Quail, Lapwing and Curlew also feature on the Irish Red List while Black-tailed Godwit, Snipe, Redshank and White-fronted Goose are all Amber listed (Newton *et al*. 1999).

Until now research into use of wet grasslands by birds has concentrated on an exceptional area of known international importance, where concentrations of species of conservation interest are unusually high. Little or no published information is available, however, about bird communities of more common wet grassland types, in other parts of the country, although these will presumably support some of the more widespread amber-listed species (e.g., Curlew and Snipe) as well as more generalist farmland amber-listed species (e.g., Skylark and Stonechat). Therefore, in considering the impact of potential afforestation on wet grassland birds, knowledge of the bird diversity of sites that might realistically be considered for planting is unavailable.

Only one of the species commonly recorded in improved grassland areas is on the Red List for species of high conservation concern in Ireland: the Yellowhammer (Newton *et al*. 1999). Although the Yellowhammer uses hedgerows in areas of predominantly improved pasture, a recent survey of breeding Yellowhammers in Co. Tipperary gives a more complete impression of their breeding habitat preferences (Collins 1999). During the survey, 154 2 x 2 km squares were searched for breeding Yellowhammers. 98% of pairs were found breeding in farmland of which 36% in unimproved grassland, 23% in mixed grass and tillage and 32% in pure tillage. At field level, fields grazed by cattle, cereal fields and hedgerows with trees were selected. Clearly, the use of improved grassland by this species is only part of the habitat requirement.

Some of the species more rarely recorded using farmland and hedgerow habitats in association with improved grassland are on the Amber list as species of medium conservation concern in Ireland: Whooper and Bewick's Swans, Brent Geese, Snipe, Skylark, Stonechat, Spotted Flycatcher, Redpoll.

The current use of intensively farmed areas should be considered in the context of the declines in seed-eating farmland species reported in a recent investigation for the Heritage Council (Taylor & O'Halloran 1999; Taylor & O'Halloran 2002). The following species have all undergone range declines: Corn Bunting (now extinct), Grey Partridge, Turtle Dove, Twite, Redpoll and Yellowhammer. The loss of the Corn Bunting from the Irish landscape has been a failure of conservation measures for birds of conservation concern (Taylor & O'Halloran 2002).

2.3.5.3 Biodiversity and Indicator Species of Grassland Habitats

Of the two grassland types on which the field survey was focused, improved grasslands almost never have any conservation value or special biodiversity interest. Characteristics of improved grasslands may serve as "negative indicators of biodiversity" (i.e. they indicate low biodiversity, highly modified grasslands): dominance by *Lolium perenne*, few broadleaved herbs apart from *Trifolium repens*, high intensity grazing or cutting, recent reseeding, regular fertilisation and weed control. However, a few specific areas of improved grassland (mainly in coastal locations) do provide winter feeding habitat for some waterbird species of conservation interest (see Section 2.3.4.2.1).

Some wet grasslands support high biodiversity communities. Junco acutiflori-Molinietum grasslands can be species-rich; character and differential species of the association can serve as potential indicator species of biodiversity (Table 5). As the *Agrostis canina* subassociation represents the least disturbed of these grasslands, its differential species are also potential biodiversity indicators (Section 2.3.3.4). Of the Senecioni-Juncetum acutiflori communities, the less disturbed *Ranunculus flammula* subassociation may be of biodiversity value. Its differential species, and the two sets of species given by Eakin, (1995) are possible indicators.

2.4 PEATLANDS

2.4.1 Peatland Habitats and Ecology

Peatlands are ecosystems that form peat, relatively deep accumulations of poorly decomposed organic matter. Peat-forming systems in Ireland include bogs, fens and wet heaths. A bog is a type of peat-forming habitat that receives virtually all its water and mineral nutrients from rainwater (i.e. ombrotrophic, "rain-fed"). In this, bogs differ from fens, which are minerotrophic peat-forming systems fed by ground or flowing surface water and are therefore more nutrient-rich. Bog development requires a wet climate with cool summers so that potential evapotranspiration and decomposition are low. Cool and waterlogged soils together with the extremely acidic conditions characteristic of bogs result in the accumulations of thick layers of peat. Although there are no plant species specifically confined to ombrotrophic bog conditions (Eber 1982), the vegetation of bogs is adapted to wet, acidic, and nutrient-poor conditions. Plant strategies for dealing with poor nutrient status include the unusual adaptation of insectivory by such plants as sundews (*Drosera* species) and butterworts (*Pinguicula* species). Ericaceous species are also common, particularly on hummocks, turf-cutting banks, and other better drained situations.

Compared with research on Irish bogs, wet heaths have received little research attention. Much of the recent research that has been conducted has been in the form unpublished undergraduate and postgraduate theses of the vegetation of a particular area (e.g. McKee 2000; Mhic Daeid 1976). Heath communities are open habitats where trees and large shrubs are infrequent or absent and are dominated by ericoid dwarf shrubs, especially *Calluna vulgaris* (Gimingham 1972). To qualify as heath under the Heritage Council's Irish habitat classification system, cover of dwarf shrubs must exceed 25% (Fossitt 2000). Wet heaths typically occur on shallow peats or peaty podzols. The reported depths of peat that will support wet heath vegetation vary. Fossitt (2000) reports "an average depth of 15-50 cm", whereas Conaghan (2001b) and Moore (1968) consider that wet heaths develop on peats of 25-100 cm depth. McKee (2000) has recorded wet heath vegetation in Connemara from peats over 1.5 m depth.

Irish peatland habitats are classified on the basis of morphology, vegetation composition, and anthropogenic disturbance. The first three categories, **raised bog** (PB1) and **upland** and **lowland blanket bog** (PB2 and PB3), correspond to undisturbed bogs. **Raised bogs** are characteristic of the central plain of Ireland. Peat depth may reach 15 m and create domes that rise up to 6 m above the level of the surrounding mineral soil (Bellamy 1986; Foss *et al.* 2001). The majority of peat in **raised bogs** is poorly humified, waterlogged *Sphagnum* peat. A "lagg zone" of fen vegetation may surround more intact bogs, which should be classified in the appropriate category under the **fens and flushes** (PF) habitat group (Fossitt 2000).

A blanket bog is in essence a mantle of peat cloaking the landscape, which unifies several peat-forming systems into an ecological complex. **Upland blanket bogs** occur on plateaus or gently sloping ground generally above 150 m elevation in hills and mountains (Fossitt 2000). Peat depth is relatively shallow, typically 1-2 m, but can be deeper in localised areas. **Lowland**, or Atlantic, **blanket bog** is confined to lower elevation areas of the Atlantic seaboard where annual precipitation exceeds 1250 mm (Fossitt 2000). The peat of lowland blanket bogs is usually between 2-6m in depth, cyperaceous, highly humified, and has an average pH of 4.2 (Doyle 1990). The pH of lowland blanket peat is usually slightly higher than upland blanket peat due to strong maritime influences resulting in increased presence of ions such as chlorine, magnesium and sodium (Doyle 1990; Walsh & Barry 1958). The topography underlying both types of blanket bogs, however, has a profound effect on bog drainage, peat depth, degree of peat humification, and therefore such dependent factors as pH and vegetation. Where peat is thin, such as on steeper slopes or near rock outcrops, **wet heath** (HH3) vegetation can be found in an intimate mosaic with blanket bog. **Wet heath** develops on waterlogged peaty soils and shallow peats, frequently on slopes too steep for development of blanket bog.

Cutover bog (PB4) applies to both raised and blanket bogs "where part of the original mass of peat has been removed through turf cutting or other forms of peat extraction," including industrial cutaway bog (Fossitt 2000). The classification category is something of a catch-all, since "if the regenerating habitats of cutover bog cover a sizeable area and can easily be fitted elsewhere in the classification, this should be done" (Fossitt 2000). **Wet** (and dry) **heath** vegetation frequently develops on the surface of **cutover bog** (Hammond *et al*. 1990; Moore 1960; Moore 1968). **Eroding blanket bog** (PB5) is another disturbed bog category to be used where erosion rather than extraction has exposed sizeable areas of bare peat. Although some erosion may be natural, including bog bursts, the majority has been caused by overgrazing and road construction. Like **cutover bog**, the **eroding blanket bog** habitat type can develop wet or dry heath vegetation over time (Cooper & Loftus 1998).

The level of the water table relative to the bog surface profoundly influences the lichen, bryophyte and vascular flora and the degree of competition for niches between species (Boatman 1961). Bogs have a highly complex microtopography comprised of hummocks, hollows, pools and lawns. These features are created and occupied by different floral assemblages (Eber 1982). Hummocks, for example, are comprised of *Sphagnum* species that are better able to retain water than those that occupy lawns or hollows. *Calluna vulgaris* and other dwarf shrubs commonly occupy these drier perches. Bogs can also contain internal drainage features known as soaks. Because the flowing water increases nutrient availability, soaks often permit the development of more nutrient demanding fen vegetation. In blanket bogs, flushes, swallow holes, and drainage channels which reach the mineral soil support richer rheotrophic fen communities. These areas of more nutrient-rich vegetation should be classified under the appropriate category of **fens and flushes** (PF) (Fossitt 2000).

2.4.2 Peatland Origins and Disturbance

Raised bogs began to develop about 10,000 years ago on the sites of shallow lakes of glacial origin (Mitchell & Ryan 1997). Over time, a lake developed reed swamp vegetation, which was replaced in turn by fen communities as peat accumulated in the basin under anaerobic conditions. Eventually, sufficient fen peats accumulated to isolate the fen vegetation from the mineral-rich groundwater. *Sphagnum* and other typical bog species tolerant of nutrient-poor conditions were then able to colonise and transform the fen to a raised bog. Climate and local topography determine the extent to which raised bogs are contained in the original lake basin or are able to grow over surrounding areas of mineral soil. Such expanding bogs can coalesce, forming "ridge raised bogs" (Bellamy 1986) and larger bog complexes, and can be seen as intermediate between "true" contained raised bogs and blanket bogs (Bellamy & Bellamy 1966; Cross 1990). In Ireland, such complexes are much more common than contained raised bogs (Cross 1990).

The origins and morphology of blanket bogs are more complex. Although the nuclei of some western blanket bogs began developing at around the same time as raised bogs, the major phase of blanket bog expansion apparently began approximately 4000 years ago (O'Connell 1990). During this phase, peat developed directly on mineral soils after forest clearance and climatic deterioration. Individual bogs were consolidated into a vast carpet of peat swathing the landscape. The periods of blanket bog initiation and growth vary markedly across a small area, probably because of differences in topography and perhaps Neolithic farming practices (O'Connell 1990).

Although considered part of the heathland group of habitats based on vegetation composition similarities, wet heath communities share their developmental history with bogs. The majority of Irish heathlands developed after forest clearance and climate change in prehistoric and early historic times. Paleoecological data indicate that extensive heathlands had developed between by around 3500 BC, although it is likely that forests later recovered in many such areas (Mitchell & Ryan 1997). Wet (and dry) heath vegetation also frequently develops on the surface of drained, cutover, or eroded bogs (Cooper & Loftus 1998; Moore 1960; Moore 1968). Similar transitions from drying bog to heath appear to have occurred during periods of warmer climate in the past. Paleoecological evidence suggests that transitions from bog to heath and back again may have occurred several times in the history of one site (Gimingham 1972).

Many peatlands have been modified by peat extraction for fuel. Where bog has been cut by hand, the remnant habitat is known as "cutover bog", whereas the term "cutaway bog" is applied to areas that have been denuded of peat by heavy machinery. Cutaway bog is more common in the midland raised bogs that are harvested mainly to feed peat-burning power plants. Bogs may also be altered by drainage, fertilisation and other agricultural or forestry management practices. By the mid-1990s, approximately 27% of the original blanket bog estate had been afforested (Foss 1998). Since the late 1990s, however, afforestation on peatlands has decreased (Conaghan 2001a) and is at present limited to the private sector. In particular, intact raised bogs are not afforested at present (Cross 1990). Accordingly, raised bog habitats will only be briefly considered in this review.

Peatlands, particularly wet heaths, are also disturbed by burning and grazing. Burning is sometimes carried out on wet heaths to encourage an early flush of growth for livestock, but is not normally seriously damaging (Conaghan 2001b). Controlled burns will normally kill only mature and degenerate phases of *Calluna*, leaving younger individuals to resprout vigorously from the base, but intense fires and burning in autumn are more likely to kill *Calluna* shrubs of all ages (Gimingham 1960) and risk setting peat on fire. Repeated burning can lead to dominance by *Molinia* and impede the regeneration of dwarf shrubs like *Calluna* and *Ulex gallii* (Conaghan 2001b). Accidental or arson-caused wild fires also occur in peatlands. Sheep are the most common livestock grazed on peatlands. Overgrazing on peatland commonage has become a recent problem attributable to the introduction of headage payments by the EU (Bleasdale & Sheehy Skeffington 1995; Conaghan 2001b). Overgrazing can drive vegetation change towards communities dominated by grasses, such as *Nardus stricta* and *Deschampsia flexuosa* (Bleasdale & Sheehy Skeffington 1995; McKee 2000). Trampling of bog or wet heath vegetation may also lead to exposure and erosion of peat.

As a result of exploitation and disturbance, only 21% of the original cover of blanket bog in the Republic of Ireland is considered of conservation interest (O'Connell 2002). Less is known about the extent of wet heaths of conservation value, but it is clear that there are many such sites that are not currently protected (Dwyer 2000).

2.4.3 The Distribution of Peatlands in Ireland

Blanket bog formerly encompassed approximately 775,000 ha in the Republic of Ireland (Hammond 1979). **Lowland blanket bog** is most abundant in the west and north-west coastal regions of Donegal, Mayo, Galway and to a lesser extent in Kerry. It usually occurs at elevations below 150 m and is subject to strong maritime influences. Atlantic blanket bog covered approximately 240,000 ha of Ireland (Hammond 1979). In comparison to Atlantic blanket bogs, **upland blanket bog** occurs at higher altitudes in both coastal and inland locations, mostly in the west of Ireland in Donegal, Leitrim, Fermanagh, Tyrone, Mayo, Galway, Clare and Kerry, but also in parts of the east in Dublin, Kildare and Wicklow and in the south in Waterford and Cork. In actuality, the distinction between lowland and upland raised bogs is not clear-cut. Schouten (1984) recognised a highland blanket bog, occurring in the hyper-oceanic west at 150 – 300 m elevation; bogs occurring above this elevation were classified as mountain blanket bogs. The vegetation of Schouten's (1984) highland blanket bog was of a transitional nature containing most species from lowland bogs, except for pool-dwelling species such as *Eriocaulon aquaticum* and *Lobelia dortmanna*, with the addition of *Empetrum nigrum, Vaccinium myrtillus* and *Diplophyllum albicans*, which are characteristic of mountain bogs. Upland blanket bogs are susceptible to erosion, especially when subjected to overgrazing. At present, there are few extensive areas of intact upland blanket bog that remain, with the best including parts of the Slieve Blooms, the Wicklow mountains, the Caherbarnagh/Mullaganish Mountains in Cork and Kerry and the Cuilcagh Mountains in Cavan and Fermanagh (Conaghan 2001a). Overall, estimates of the area of intact lowland and upland blanket bog remaining are unreliable and difficult to ascertain because of the problems in separating wet heath from blanket bog and the relatively recent damage from overgrazing by livestock (Conaghan 2001a).

The largest areas of **wet heaths** are to be found in the western half of Ireland (Conaghan 2001b). In the west, **wet heath** frequently occurs as a mosaic on shallower peat associated with blanket bog. Comparing her work with that of Moore (1960), McKee (2000) concluded that a greater diversity of wet heath types was present in Connemara than in the Wicklow Mountains. Natural **wet heaths** are relatively rare in the midlands, apart from heath-type vegetation regenerating on cutover bog.

Raised bogs are primarily found below 130 m elevation in the central plain of Ireland, in undulating topography overlain by glacial drift. Outliers occur in east Clare and north Limerick (an area north and south of the mouth of the Shannon) and the Lough Neagh and River Bann valley area in Northern Ireland (Cross 1990). Two types of **raised bogs** may be distinguished in Ireland on floristic and morphological grounds: True Midland raised bogs and Western raised bogs (Cross 1990; Schouten 1984). The True Midland type occurs in areas that receive less than 1000 mm/yr of rainfall, while the Western type is found in areas receiving 1000-1200 mm of rain annually, including Clare, Kerry, and parts of Galway, Mayo and Roscommon (Cross 1990). Western raised bogs have some floristic and morphological similarities to blanket bogs, and can therefore be considered intermediates between blanket and True Midland raised bogs. For example, Western raised bogs are perhaps more likely than the True Midland type to extend beyond their original lake basins (Schouten 1984). These divisions, however, are made largely for the sake of convenience, since the flora, morphology and chemistry of bogs forms a continuum from **lowland blanket bogs** in the oceanic west to True Midland **raised bog** in the drier centre and east.

2.4.4 Vegetation

2.4.4.1 Classification of Peatlands

Bogs and wet heaths are placed in the phytosociological class Oxycocco – Sphagnetea (Moore 1968; White & Doyle 1982). A synopsis of the phytosociological classification of Irish bog and wet heath vegetation is shown in Figure 2 and correspondence with Irish habitat types is given in Table 7. The fine-grained spatial scaling of differences in microtopography on the bog surface and the flora associated with different microtopographical features has caused difficulties with bog phytosociology (Schouten 1990). Phytosociology relies upon the recognition of a stand of vegetation distinguishable from adjacent stands on floristic or structural grounds. Because the surface of a bog is generally an intimate mosaic of hummocks, hollows, pools and lawns, "stand definition may become a matter of opinion, the more so since the vegetation contains a limited number of species" (Schouten 1990). The vegetation of a bog may therefore be considered as a complex of several stands or as one stand with an internal mosaic structure. The broad approach taken by the Irish habitat classification (Fossitt 2000) is similar to the latter, "single-stand" solution, whereas most phytosociological classifications employ the former, "several-stand" approach. The phytosociological associations discussed below therefore frequently apply to small areas of the overall bog complex, such as individual small pools.

Table 7. Correspondence between the classification of bogs and wet heaths under the Irish habitat classification (Fossitt 2000) and phytosociological syntaxa (White & Doyle 1982).

Habitat	Order	Association
PB1 raised bog	Eriophoro vaginati-Sphagnetalia papillosi*	Erico-Sphagnetum magellanici
PB2 upland blanket bog	Eriophoro vaginati-Sphagnetalia papillosi*	Vaccinio-Ericetum tetralicis
PB3 lowland blanket bog	Eriophoro vaginati-Sphagnetalia papillosi*	Pleurozio purpureae-Ericetum tetralicis
PB4 cutover bog	*Depends on vegetation present*	
PB5 eroding blanket bog	*Depends on vegetation present*	
HH3 wet heath	Sphagnetalia compacti	Narthecio-Ericetum tetralicis

* Vegetation of the wet hollows on bogs is referable to the Scheuchzerietalia palustris.

There are several character species of the Oxycocco – Sphagnetea, not all of which occur in all the bog and wet heath habitats in Ireland: *Vaccinium oxycoccus, Andromeda polifolia, Drosera rotundifolia, Eriophorum vaginatum, Sphagnum fuscum, S. magellanicum, S. capillifolium, S. tenellum, Pohlia nutans, Calypogeia muelleriana, Cephalozia connivens, Kurzia pauciflora,* and *Mylia anomala* (White & Doyle 1982). Within the class are three orders: (i) Sphagnetalia compacti, representing wet heaths, (ii) Eriophoro vaginati – Sphagnetalia papillosi, describing vegetation on deep peat in western Europe, and (iii) Scheuchzerietalia palustris, encompassing vegetation of wet hollows on bogs. Each of these is represented in Ireland by a single alliance (Figure 2).

Sphagnetalia compacti wet heaths belong to the alliance Ericion tetralicis in Ireland (Moore 1968; White & Doyle 1982). Order and alliance character species include: *Erica tetralix, Trichophorum caespitosum, Juncus squarrosus, Sphagnum compactum,* and *Sphagnum strictum.* Alliance differential species are *Potentilla erecta, Polygala serpyllifolia, Pedicularis sylvatica, Carex panicea* and *Succisa pratensis.*

In Ireland, the Eriophoro vaginati – Sphagnetalia papillosi is represented by the alliance, Calluno – Sphagnion papillosi. The character species of the order and alliance are: *Eriophorum vaginatum, Sphagnum papillosum, S. imbricatum, Odontoschisma sphagni, Narthecium ossifragum, Rhynchospora alba, Campylopus flexuosus, Cephalozia bicuspidata, Diplophyllum albicans, Mylia anomala, Cladonia portentosa,* and *C. uncialis.*

Scheuchzerietalia palustris is represented in Ireland by the alliance Rhynchosporion albae. The character species of these syntaxa are: *Carex limosa, Rhynchospora alba, R. fusca, Menyanthes trifoliata, Sphagnum cuspidatum, S. denticulatum, S. subsecundum* s.l., *S. pulchrum, Cladopodiella fluitans, Scheuchzeria palustris, Drosera intermedia,* and *Warnstorfia fluitans.* Note that *Scheuchzeria palustris* (Rannoch rush) is considered extinct in Ireland following destruction of its single site by peat extraction (Curtis & McGough 1988).

2.4.4.2 Raised Bog **PB1**

The vegetation of the flats and low hummocks of **raised bogs** are placed in the association Erico – Sphagnetum magellanici (Moore 1968; White & Doyle 1982) (Figure 2). Moore (1968) considered Irish raised bogs to belong to an extreme Atlantic race of European raised bogs. Because of the floristic similarities between raised and blanket bogs in Ireland, White and Doyle (1982) recommend two sets of differential species. The presence of *Andromeda polifolia* and *Vaccinium oxycoccus* distinguish raised bog from lowland blanket bog, and the occurrence of *Sphagnum imbricatum* s.l., *S. magellanicum,* and *S. fuscum* distinguish raised bog from upland blanket bog.

The distinction discussed above between Western and True Midland raised bogs has not been classified in phytosociological terms, but floristic differences have been noted. Western raised bogs can be distinguished from True Midland raised bogs by the presence of *Campylopus atrovirens* and *Pleurozia purpurea*, character species of **lowland blanket bog** (below), and the abundance of *Carex panicea* (Cross 1990; Schouten 1984). Schouten (1984) has also suggested that the co-occurrence of *Schoenus nigricans* with *Sphagnum* cover in the 10-50% range indicates that the bog is intermediate between raised and blanket types.

The vegetation of the wet hollows on raised bogs has been assigned to a separate order (Scheuchzerietalia palustris) from the more terrestrial vegetation of lawns and hummocks (Figure 2). The association Sphagno tenelli – Rhynchosporetum albae has the diagnostic species *Rhynchospora alba, R. fusca, Sphagnum cuspidatum*, and *S. tenellum*. The Scheuchzerietum association was recorded from a wet hollow on Pollagh Bog where *Scheuchzeria palustris* had its only Irish station before the bog was destroyed (White & Doyle 1982).

2.4.4.3 Upland Blanket Bog **PB2**

The most abundant species occurring in upland or montane blanket bogs include *Calluna vulgaris, Eriophorum angustifolium, E. vaginatum, Erica tetralix* and *Trichophorum caespitosum* (Conaghan 2001a). *Calluna* tends to gain dominance on shallower peats, whereas *Eriophorum angustifolium* becomes more abundant on deeper, wetter peats (Conaghan 2001a). Upland blanket bog vegetation is assigned to the association Vaccinio – Ericetum tetralicis (Moore 1968; White & Doyle 1982). Diagnostic species are *Vaccinium myrtillus, Empetrum nigrum* and *Eriophorum vaginatum*. *E. vaginatum* can also be found in lowland blanket bogs, but is less frequent than in the uplands (Conaghan 2001a). The presence of *Andromeda polifolia* and *Vaccinium oxycoccus* can help differentiate upland from lowland blanket bog vegetation (White & Doyle 1982), but they are only rarely encountered in upland blanket bogs in the west (Conaghan 2001a). Upland blanket bog generally supports greater abundances of dwarf shrubs than lowland blanket bog, including *Vaccinium myrtillus, Empetrum nigrum,* and also *Calluna.*

Where upland blanket bogs are wet and intact, the bryophyte layer is frequently well-developed. *Sphagnum capillifolium* is typically the dominant moss, with *S. papillosum, Hypnum jutlandicum* and *Racomitrium lanuginosum* frequently appearing (Conaghan 2001a). In western upland blanket bogs, however, *Sphagnum* species are much less abundant, usually achieving less than 10% cover (Schouten 1984). *Diplophyllum albicans* is the most commonly encountered liverwort, and is characteristic of upland blanket bog habitats (Conaghan 2001a; Schouten 1984). Overall, the liverwort flora can be quite rich in wet and undisturbed situations (Conaghan 2001a).

Moore (1962) recognised three subassociations of upland blanket bogs, described from the Dublin and Wicklow Mountains. The first was a *Juncus squarrosus* subassociation, differentiated by *Juncus squarrosus, Calypogeia muelleriana, Plagiothecium undulatum* and

Deschampsia flexuosa. The subassociation was usually found on peat approximately 1 m deep over schist or shale. A *Narthecium* subassociation, with differential species *Narthecium ossifragum, Cladonia uncialis, Cladonia sylvatica, Racomitrium lanuginosum* and *Andromeda polifolia*, typically occurred on gently sloping peats averaging 4 m over granite. The community was also characterised by the presence of small amounts of bare peat. The typical subassociation, characterised by the scarcity or absence of the above species, was found on peats of intermediate depth over granite. Mhic Daeid (1976) surveyed blanket bog vegetation on peat 1.5 - 2 m deep mainly on the Mangerton plateau in Killarney, and found that it corresponded with Moore's (1962) *Juncus squarrosus* association. The Killarney upland bogs, however, were also characterised by high constancy of a small form of *Melampyrum pratense*. Overall vegetation cover in all plots was 100%, *Trichophorum caespitosum* was uncommon and *Molinia caerulea* was absent. Mhic Daeid (1976) considered that Irish montane blanket bogs were generally heavily eroded and that "it is probable that the distribution of the various sub-associations and noda depends on the degree to which degeneration has proceeded in different areas."

Unlike lowland blanket bog (below), upland blanket bogs are generally characterised by large tracts of Vaccinio-Ericetum tetralicis vegetation, with few other vegetation types present. One significant exception is wet heath; upland blanket bog and wet heath vegetation frequently form mosaics and intergrade with each other, responding to small changes in slope, peat depth and degree of disturbance (Fossitt 2000). Accordingly, it can be difficult to determine where one habitat type begins and another ends. Dystrophic pools and swallow holes can also be found in some upland blanket bogs (Conaghan 2001a). Upland blanket bogs in Ireland have not been found to support well-defined Rhynchosporion communities of permanently waterlogged hollows on bogs (Conaghan 2001a).

2.4.4.4 *Lowland Blanket Bog* **PB3**

The vegetation of lowland (or Atlantic) blanket bog is "grassy" in general appearance, and is chiefly dominated by *Schoenus nigricans* and *Molinia caerulea* (Conaghan 2001a; Doyle 1990). Ericaceous shrubs, particularly *Calluna vulgaris* and *Erica tetralix*, are frequently present, but generally dwarfed. The relative abundances of these species is influenced by several factors, including peat depth, amount of lateral water movement and degree of disturbance (Conaghan 2001a). In comparison with other bog habitats, the *Sphagnum* component is poorly developed over most blanket bog areas (Doyle 1990), but moss cover nevertheless generally exceeds 30% (Conaghan 2001a).

Lowland blanket bogs in Ireland are referable to the association Pleurozio purpureae-Ericetum tetralicis (Doyle 1982; Doyle & Moore 1980; Moore 1968). Many but not all of the character species from the class Oxycocco-Sphagnetea occur in the Atlantic blanket bog complexes in the west and north-west of Ireland (Doyle 1982; Moore 1968). Notably absent from (or rare in) lowland blanket bog are *Vaccinium oxycoccus* and *Andromeda polifolia* (Doyle 1982; Doyle & Moore 1980; Moore 1968); the absence of these species can help distinguish lowland blanket bog from other bog habitats. The order character species which are widely found in Atlantic blanket bog complexes in the west of Ireland include *Sphagnum papillosum* and *Odontoschisma sphagni* (Doyle 1982). *Eriophorum vaginatum* is another order character species, but is generally confined to shallower or better-drained areas; the species is more common on upland blanket bog (Doyle 1982). Diagnostic species of the alliance (Calluno-Sphagnion papillosi) include: *Narthecium ossifragum, Rhynchospora alba, Campylopus flexuosus, Sphagnum imbricatum* s.l., *Cephalozia bicuspidata, Diplophyllum albicans, Mylia anomala, Cladonia portentosa* and *Cladonia uncialis* (White & Doyle 1982), all of which commonly occur on lowland blanket bog in Ireland (Doyle & Moore 1980).

The character species of the Pleurozio purpureae-Ericetum tetralicis association are *Schoenus nigricans, Pleurozia purpurea* and *Campylopus atrovirens* (Doyle 1982; Doyle & Moore 1980). *Schoenus nigricans* is a local character species of the association in the west of Ireland; it does not generally occur in lowland blanket bog in Scotland (Doyle 1982; Doyle & Moore 1980; Moore 1968). Differential species of this association, which aid in differentiating lowland blanket bog from other bog habitats, include: *Potentilla erecta, Polygala serpyllifolia, Pedicularis sylvatica* and *Pinguicula lusitanica* (Doyle 1982). The first three species are typical of wet heath; they may useful in distinguishing lowland blanket bogs from upland blanket bogs. Moore (1968) also proposed *Racomitrium lanuginosum* as a differential species, but although it is common in Atlantic blanket bog, it is also present in raised and montane blanket bogs in Ireland (Doyle 1982; Doyle & Moore 1980).

According to Doyle and Moore (1980) and Doyle (1982), there are five sub-associations of the Pleurozio-Ericetum, which occur under different conditions of peat depth, drainage and nutrient levels as influenced by water movement (Table 8). The typicum subassociation is defined negatively as possessing none of the differential species of the other subassociations. The Pleurozio-Ericetum typicum is characteristic of large areas of deep, level peat, where there is no additional input of nutrients from water draining in from surrounding peat. Hummocks created by such mosses as *Sphagnum capillifolium, S. imbricatum* s.l., *S. fuscum* and *Racomitrium lanuginosum* sometimes occur (Conaghan 2001a; Doyle 1982).

Table 8. Subassociations of the Pleurozio purpureae-Ericetum tetralicis association of lowland blanket bogs in Ireland (Doyle 1982).

Subassociation	Character and Differential Species	Conditions
typicum	–	Level peat > 4 m deep, hummocks
zygogonietosum	*Zygogonium ericetorum* *Drosera anglica* *Sphagnum magellanicum*	Permanently waterlogged depressions, bog pool edges
juncetosum	*Eleocharis multicaulis* *Carex panicea* *Juncus bulbosus*	Areas with moving surface water, disturbed areas
droseretosum	*Drosera intermedia* *Carex limosa* *Riccardia pinguis* *Rhynchospora fusca*	Slopes & hollows with lateral water movement in surface peat
scirpetosum	*Trichophorum caespitosum* *Leucobryum glaucum*	Shallow peat near drainage channels

The Pleurozio-Ericetum zygogonietosum sub-association is found predominantly near the Mayo and Galway coasts (Doyle 1982). It is characterised by dense, deep cover of up to sixty species of mucilaginous algae (known in aggregate as '*Zygogonium ericetorum*'). The algal mats may be 10 cm deep and often encase the bryophyte layer. *Drosera anglica* and *Sphagnum magellanicum* also serve as character species of the subassociation; the relative abundances of *Zygogonium* and the latter two species define three variant communities (Doyle 1982). The first variant lacks both *D. anglica* and *S. magellanicum* but has *Zygogonium* as the dominant differential species. This variant is found in waterlogged conditions in shallow depressions between low hummocks up to 10cm in height. The second variant in this subassociation has *D. anglica* and *Zygogonium* as differential species but lacks *S. magellanicum*. It is typically found at the edge of bog pools in waterlogged conditions where some nutrient enrichment from drainage from adjacent land is likely. The third variant contains *D. anglica, Zygogonium* and *S. magellanicum* and is found in very waterlogged depressions in the bog complex.

The Pleurozio-Ericetum juncetosum subassociation contains the character species *Eleocharis multicaulis, Carex panicea* and *Juncus bulbosus*. This subassociation occurs where there is movement of water over the surface of the peat, such as small surface runnels, or where grazing animals cause regular disturbances (Doyle 1982). The community is also found at the edges of shallow pools in the process of colonisation by terrestrial vegetation.

The Pleurozio-Ericetum droseretosum subassociation is differentiated by the presence *Drosera intermedia, Carex limosa, Aneura pinguis* and *Rhynchospora fusca*. This sub-association occurs on sloped areas of blanket bog where water movement occurs. Three variants have been described: 1) a typical variant with only *D. intermedia* present, 2) a *Rhynchospora fusca* variant frequently occurring in Kerry, with *R. fusca* and *Aneura pinguis* co-occurring with *D. intermedia*, and 3) a *Carex limosa* variant containing all four differential species, which is often found in Kerry and around Roundstone, Co. Galway (Doyle & Moore 1980; Mhic Daeid 1976).

The Pleurozio-Ericetum scirpetosum subassociation has the character species *Trichophorum caespitosum* (formerly *Scirpus caespitosus* ssp. *germanicus*, hence the subassociation name) and *Leucobryum glaucum*. This subassociation has been divided into two variants by Doyle (1982). The first variant is distinguished by the presence of *Zygogonium* and occurs on shallow, *level* peat near drainage channels. The second variant, distinguished by *Carex panicea*, is confined to areas of shallow, *sloping* peat adjacent to drainage channels (Doyle 1982). Such situations are exposed to considerable surface run-off during wet periods and desiccation during drier periods (Doyle & Moore 1980).

Unlike raised bogs, lowland blanket bogs do not support well-defined examples of the Sphagno tenelli-Rhynchosporetum albae association describing the vegetation of permanently waterlogged hollows on bogs, possibly because of less microtopographical variation on blanket bogs (White & Doyle 1982). However, Rhynchosporion vegetation has been recorded in lowland blanket bogs during conservation surveys (Conaghan 2001a). On lowland blanket bogs, the diagnostic species of the Sphagno-Rhynchosporetum association (and higher syntaxa) co-occur with those of the Pleurozio-Ericetum.

In addition to the bog communities of the Pleurozio-Ericetum association, several other communities can be found in lowland blanket bog habitats. Wet and dry heath can occur in blanket bog complexes, often at the edges of stream banks and on or around rock outcroppings (Doyle 1990; Fossitt 2000). Pools and lakes are frequent in lowland bogs; those contained entirely within the peat (i.e. not reaching sub-peat mineral soil or rock) are true "bog pools" and are classified as **dystrophic lakes (FL1)** (Fossitt 2000). The vegetation of bog pools is varied, and has been classified in several different phytosociological associations in the past. Doyle (1990) refers bog pool communities to the Isoeto-Lobelietum. Larger pools and lakes typically have a sparse vegetation, including such species as *Eriocaulon aquaticum, Menyanthes trifoliata, Eleocharis multicaulis,* and *Utricularia minor*. Other pools can support well-developed floating mats of aquatic *Sphagnum* species and the algal aggregate *Zygogonium ericetorum* (Doyle 1990).

The vegetation of flushes and other drainage features in lowland blanket bog is comprised of several different communities whose species composition is dependent on the rate at which water flows, water chemistry, especially base status, the shelter provided to plants by the feature, and the nature of the substrate, whether peat or mineral soil (Doyle 1990; Lockhart 1991). Doyle (1990) describes a number of vegetation communities of flushes and drainage channels on blanket bog. On areas of shallow peat associated with seepage areas and the heads of drainage channels, a *Myrica gale*–dominated shrub community often occurs, with *Calluna vulgaris* and *Erica tetralix* also present. Other common species include *Eriophorum vaginatum, Molinia caerulea, Hypnum jutlandicum* and *Pleurozium schreberi*. Below the *Myrica gale* community, in seepage areas and the upper reaches of drainage channels, vegetation dominated by *Eriophorum angustifolium* and *Sphagnum recurvum* s.l. can be found. Other species that may be present at low abundances include *Menyanthes trifoliata, Juncus bulbosus* and *Carex limosa*. This *Eriophorum-Sphagnum* flush community is probably best considered **poor fen/flush (PF2)**. In the upper parts of drainage channels where water flow is more substantial, another **poor fen/flush** community develops, dominated by *Juncus effusus* and *Sphagnum recurvum* s.l. Other frequently occurring species include the grasses *Agrostis stolonifera* and *Holcus lanatus, Galium saxatile* and *Polytrichum commune*. Lockhart (1991) describes these and other communities in more detail in his classification of 17 community types of lowland blanket bog flush vegetation in Galway and Mayo. Some of these community types are of restricted distribution and support rare vascular plants (e.g. *Saxifraga hirculus* and *Hammarbya paludosa*, c.f. Table 10) and bryophytes (e.g. *Leiocolea rutheana* and *Tomentypnum nitens*). Lockhart (1991) concludes that as blanket bog flushes encompass a diversity of community types, conservation goals should ensure that a representative range of types are protected and that adequate buffer zones are provided.

Other plant communities can be found associated with drainage features in lowland blanket bogs. In deeper, more sheltered channels experiencing significant water flow, *Juncus articulatus* is prominent in the vegetation, with *Ranunculus flammula, Hydrocotyle vulgaris, Epilobium palustre, Galium palustre, A. stolonifera* and *H. lanatus* usually present. In the lower reaches of drainage channels, where a mineral substratum underlies a slow-moving stream, Doyle (1990) has described two distinct vegetation types, both of which are species-poor. The first is dominated by *Carex paniculata*, with low abundances of other species such as *Potentilla palustris, Epilobium palustre, Juncus effusus, A. stolonifera* and *H. lanatus* between the sedge tussocks. The second community that occasionally develops is dominated by *Hippuris vulgaris*; *Ranunculus flammula*, *A. stolonifera, H. lanatus* and *Equisetum fluviatile* are the main associated species. If the stream is actively eroding the base of the channel, leaving a largely gravel substrate, these communities are probably best described as **eroding/upland rivers (FW1)**. Where the rivers meander, flood-plain banks can form, and are usually dominated by **dry-humid acid grassland (GS3)** or **wet grassland (GS4)**. Grazed examples of this vegetation are generally species-poor, but in Mayo a more species-rich variant occurs where the banks are occasionally flooded and grazing pressure is low (Conaghan 2001a). Where drainage channels are particularly deep- some may reach 4 m in depth- **scrub (WS1)** comprised of *Salix aurita* and *Salix cinerea* ssp. *oleifolia* can sometimes form. Often accompanying the willows are a number of acidophilous woodland species such as *Digitalis purpurea, Blechnum spicant, Rubus fruticosus* agg., *Hylocomium splendens* and *Thuidium tamariscinum.* A flora with similar woodland affinities can also develop in the shelter of swallow holes, entrances to subterranean drainage channels. Swallow holes are frequently fringed by *Calluna*-dominated heath vegetation, and the base is usually occupied by a *Juncus effusus* dominated **poor fen/flush (PF2)** community.

2.4.4.5 Wet Heaths HH3

Wet heaths are placed together with bog communities in the class Oxycocco – Sphagnetea, but are divided from them at the order level. **Wet heath** belongs to the Sphagnetalia compacti, whereas bogs are members of the orders Eriophoro vaginati – Sphagnetalia papillosi (blanket bogs and flats and hummocks of raised bogs) or Scheuchzerietalia palustris (wet hollows on bogs) (Moore 1968; White & Doyle 1982). Within the Sphagnetalia compacti, **wet heath** communities are allocated to the alliance Ericion tetralicis and the association Narthecio – Ericetum tetralicis (Figure 2) (Moore 1968; White & Doyle 1982). Many of the character and differential species at the alliance and association levels, such as

Polygala serpyllifolia, Pedicularis sylvatica, Carex panicea, Narthecium ossifragum, and *Succisa pratensis*, are also character species for different types of blanket bog vegetation. Character species at the order level may be of more diagnostic value: these are *Erica tetralix*, *Trichophorum caespitosum, Juncus squarrosus, Sphagnum compactum,* and *Sphagnum strictum*. Fossitt (2000) recommends distinguishing **wet heath** (HH3) from the dry heath types (HH1 and HH2) based on greater abundance of *Erica tetralix*, *Trichophorum caespitosum,* and *Molinia caerulea* in **wet heaths**. She also states that **wet heath** can best be separated from blanket bog habitats by the presence of *Juncus squarrosus* and *Carex binervis* and the absence of *Schoenus nigricans*.

Mhic Daeid (1976) assigns wet heath communities in the Killarney area from near sea level to 500 m elevation to an Ulicetosum gallii subassociation in which *Ulex gallii* and *Erica cinerea* are associated with *Calluna vulgaris* and *Erica tetralix*. She found the community on cutover blanket bog and shallower virgin peats with peat depths up to 1 m. In addition to the four dwarf shrub species, Mhic Daeid lists several species whose frequency of occurrence characterise the vegetation as a whole, and less frequent species that, taken as a group, are useful in differentiating between wet heaths and both drier heath associations and blanket bog (Table 9). Note that, contrary to Fossitt (2000), *Schoenus nigricans* is a component of wet heaths in Killarney. Additional species characterise wetter and drier variants of wet heath.

Table 9. Frequently occurring species on wet heath habitats (association Narthecio – Ericetum tetralicis, subassociation Ulicetosum gallii) in Killarney, Co. Kerry (Mhic Daeid 1976).

Frequently Occurring Species on Wet Heath in Killarney, Co. Kerry	
“Frequent”	
Calluna vulgaris	*Erica tetralix*
Erica cinerea	*Ulex gallii*
Trichophorum caespitosum	*Sphagnum compactum*
Racomitrium lanuginosum	*Pinguicula grandiflora*
Campylopus atrovirens	*Juncus squarrosus*
Sphagnum tenellum	*Sphagnum papillosum*
Sphagnum subnitens	*Cladonia portentosa*
Cladonia uncialis	*Eriophorum angustifolium*
Kurzia pauciflora	*Cephalozia connivens*
Breutelia chrysocoma	*Leucobryum glaucum*
Mylia taylorii	
“Less Frequent”	
Schoenus nigricans	*Myrica gale*
Sphagnum capillifolium	*Kurzia trichoclados*
Cladonia bellidiflora	*Lejeunea patens*
“Frequent in Wetter Variant”	
Nowellia curvifolia	*Pleurozia purpurea*
Narthecium ossifragum	
“Frequent in Drier Variant”	
Thuidium tamariscinum	*Agrostis canina*
Anthoxanthum odoratum	*Succisa pratensis*
Hypericum pulchrum	*Danthonia decumbens*
Festuca vivipara	*Pteridium aquilinum*
Galium saxatile	*Pseudoscleropodium purum*
Pedicularis sylvatica	

In the Twelve Bens region of Connemara, McKee (2000) recorded a relatively species-rich variant (mean of 22 species per quadrat) and a degraded, species-poor variant (mean of 15 species) of the Narthecio-Ericetum tetralicis. She found *Scapania gracilis* and the lichen *Cladonia portentosa* useful in characterising the species-rich communities. *Succisa pratensis* also occurred in this community, but not in blanket bog in the same area. The degraded variant of wet heath was characterised by larger areas of bare peat and differentiated from more intact heath by the rarity of *Polygala serpyllifolia, Succisa pratensis,* and *Cladonia portentosa*. *Molinia, Calluna, Trichophorum caespitosum*, and *Erica tetralix* were the dominant wet heath species in Connemara.

In their survey of the Dublin and north Wicklow Mountains, Pethybridge and Praeger (1905) did not explicitly identify wet heath, combining all *Calluna*-dominated communities into a "*Calluna* association". However, they describe a gradual transition from well-drained *Calluna* heath to *Trichophorum*-dominated blanket bog in which *Calluna* "becomes by degrees more stunted, and *Cyperaceae* gradually increase." When Moore (1960) resurveyed the same areas, he distinguished a "*Calluna-Juncus squarrosus-Nardus* wet heath" nodum on gentler slopes with poorer drainage, yet not waterlogged or flushed. Several species characteristic of wetter conditions, such as *Molinia, Narthecium ossifragum,* and *Erica tetralix*, were absent from "this rather loose unit," which Moore nevertheless considers partially corresponds to upland wet heath as described by Tansley (1939). Moore also identified a "*Calluna-Sphagnum-Molinia* flush" nodum occurring on peat up to 1 m depth. In this nodum, most species typical of the local blanket bogs also occurred with the addition of species not commonly occurring on bogs, such as *Carex echinata, Succisa pratensis, Juncus bulbosus, J. effusus,* and *J. acutiflorus*. Although Moore considers that the term "flush" is a better description for this community type, he notes that the species composition is similar to what Tansley (1939) calls wet heath.

Heather communities with either of the rare species *Erica ciliaris* or *Erica vagans* pose problems in classification. In the rest of Europe, these *Erica* species are usually found in dry to humid heaths belonging to the alliance Ulici-Ericion ciliaris (White & Doyle 1982). The lone Irish station of *Erica ciliaris* is a wet hollow at the edge of a blanket bog (Webb & Scannell 1983), which Conaghan (2001b) considers an example of wet heath. Similarly, the single station of *Erica vagans* where it is considered native, near Belcoo, Co. Fermanagh (Webb *et al.* 1996), is a nutrient-rich flush. These species might logically be considered species of wet heath rather than dry heath in Ireland.

2.4.5 Birds

Some information on the use of wet heaths and bogs by birds was included in a text entitled 'Birds of Raised Bogs' by H.J. Wilson, contained within a government-commissioned report on raised bog habitats in Ireland (Cross 1990). Some of the data used in the report was drawn from a log of observations made at Glenveagh National Park during the entire calendar year of 1980. The sampling was by no means systematic, but did allow a species list, with an estimated number of territories for certain species, to be drawn up (MacLochlainn 1984). Breeding Golden Plover (an estimated 2 territories) were observed using the 'upland plateau of peat' and the use of blanket bog by the species is also noted in the later report (Cross 1990). The plover requires large wet areas of short vegetation, usually created by fire, and since extensive burning occurs on blanket bog to allow spring grazing, the habitat is ideal (Cross 1990). Red Grouse is able to survive on blanket bog by maintaining larger territories than on its more optimal *Calluna vulgaris*-dominated habitat (Cross 1990). This species has been the subject of some research, due to its importance as a game species (Hutchinson 1989); when study plots in Mayo were fenced to prevent grazing, densities of grouse increased fivefold (Watson & O'Hare 1979). Recent studies by Murray and O'Halloran (2003) have shown declines in numbers and lower densities than previously thought in Co Mayo. Meadow Pipits were found to be the commonest species during a 1969/71 survey of a Mayo blanket bog (Watson & O'Hare 1979). Skylarks were numerous on flat bog and Snipe preferred well-grazed areas with rushes and bog myrtle (Watson & O'Hare 1979). Recent studies of Hen Harriers in Ireland have found that this species forages in a variety of bog and heath habitats, as well as rough pasture and young plantation forests, especially during the breeding season, and may also use these habitats for nesting (Norriss *et al*. 2002; O'Donoghue 2004). A list of species using cutaway bog given by H.J. Wilson (Cross 1990) is harder to interpret since birds observed in associated habitats such as pools, scrub and woodland are also included. However, the following species of conservation concern have been found breeding in or near Irish cutover bogs: Merlin, Red Grouse, Lapwing, Snipe, Curlew, Skylark, Grasshopper Warbler and Redpoll (Cross 1990). An interesting observation of Ringed Plovers using cutover bog as a nesting habitat was reported by Cooney (1998). He found 17 pairs in 1997 and 19 in 1998 nesting on bare peat or on peat with scant vegetation (usually marsh arrowgrass *Triglochin palustris*), a habitat not previously considered suitable for breeding Ringed Plover. In all cases the breeding sites were associated with newly flooded areas or lakes.

Greenland White-fronted Geese use blanket bog as a winter feeding habitat. In wetter areas of such bogs they feed on the swollen bases of cottongrass (*Eriophorum angustifolium*) and white-beaked sedge (*Rhynchospora alba*) (Cross 1990).

2.4.6 Conservation

2.4.6.1 Designated Habitats

Upland and lowland blanket bogs correspond with the EU Habitats Directive Annex I habitat "blanket bogs (* if active bog) 7130" (European Commission 1999; Fossitt 2000). If a significant area of vegetation is normally peat-forming, the blanket bog is "active" according to the Habitats Directive, and is therefore a priority habitat. Blanket bogs may support pockets of Rhynchosporion vegetation referable to the annexed habitat "depressions on peat substrates of the *Rhynchosporion* 7150" (European Commission 1999; Fossitt 2000), although such vegetation does not usually form distinct communities on Irish blanket bogs (White & Doyle 1982). "North Atlantic wet heaths with *Erica tetralix* 4010" comprise an Annex I habitat (European Commission 1999) corresponding to the wet heath (HH3) habitat type (Fossitt 2000).

2.4.6.2 Rare and Protected Species

Several species of vascular plants which are protected and/or rare occur in peatland habitats. These are listed in Table 10 together with details of their habitat preferences, distribution, and conservation status. Hen Harrier, Red Grouse, Lapwing and Curlew feature on the Irish Red List of birds of high conservation concern while Merlin, Golden Plover, Snipe, Skylark, Grasshopper Warbler, Redpoll and White-fronted Goose are all Amber listed (Newton *et al*. 1999). White-fronted Goose, Hen Harrier, Merlin and Golden Plover are listed on Annex 1 of the Birds Directive (92/43/EEC).

Table 10. Protected or rare plant species that may occur in wet heath or blanket bog habitats in Ireland. Conservation Status indicates if the species is protected by the Flora (Protection) Order, 1999 in the Republic of Ireland or the Wildlife (NI) Order, 1985 in Northern Ireland; if not, Conservation Status indicates if it is listed as endangered, vulnerable or rare in the Irish Red Data Book (Curtis & McGough 1988).

Species	Common Name	Habitat Details[1]	Distribution[2]	Conservation Status
Carex magellanica	Tall bog sedge	Upland blanket bogs and cliff ledges	Northeast	1985 Wildlife NI Order
Carex pauciflora	Few-flowered sedge	Upland blanket bogs	Antrim and Down	1985 Wildlife NI Order
Deschampsia setacea	Bog hair-grass	Wet bogs and lakesides	Connemara	1999 Flora Protection Order
Erica ciliaris	Dorset heath	Wet hollow at edge of blanket bog	One site near Roundstone, Co. Galway	Red Data: vulnerable[3]
Erica mackaiana	Mackay's heath	Drier parts of bogs, wet heath	Galway, Mayo, Donegal	Red Data: rare
Erica vagans	Cornish heath	Nutrient-rich flush in heath	One site near Belcoo, Co. Fermanagh	Red Data: vulnerable
Eriophorum gracile	Slender cottongrass	Wet bogs and lakesides	Mainly Connemara	1999 Flora Protection Order
Hammarbya paludosa	Bog orchid	Wet bogs	Scattered throughout	1999 Flora Protection Order and 1985 Wildlife NI Order
Lycopodiella inundata	Marsh clubmoss	Lake margins, wet bog, wet heath	Scattered, mostly west Galway	1999 Flora Protection Order and 1985 Wildlife NI Order
Orthilia secunda	Serrated wintergreen	Bogs and wet ledges	Fermanagh, Antrim, Derry; formerly Offaly	1985 Wildlife NI Order
Rubus chamaemorus	Cloudberry	Upland blanket bog	One site in Sperrin Mtns, Tyrone	1985 Wildlife NI Order
Saxifraga hirculus	Marsh saxifrage	Flushes in wet bogs	Mainly west Mayo and north	1999 Flora Protection Order and 1985 Wildlife NI Order

[1] Details compiled from Curtis and McGough (1988) , Webb *et al.* (1996) and Conaghan (2001b).

[2] Details from Preston *et al.* (2002).

[3] Formerly protected, possibly planted.

2.4.6.3 *Biodiversity and indicators of peatland habitats*

Blanket bogs are important Irish habitats at the national, international and sometimes local levels. Active blanket bogs are priority habitats for conservation under the EU Habitats Directive (Section 2.4.6.1). Ireland contains approximately 8% of the world's blanket bogs, and is therefore one of the most important centres for conservation of this global resource (Conaghan 2001a). Revegetating cutover or eroded bog can also support interesting and diverse plant communities. Wet heath habitats are also important for biodiversity conservation.

Species that are clear indicators of peatland biodiversity or habitats of particularly high conservation value have not been formally defined in Ireland. Therefore, only potential indicators can be proposed. The rare and protected species listed in the above section (plus others not listed, e.g. bryophytes) should be considered as *de facto* biodiversity indicators.

Species indicators of biodiversity are difficult to identify for wet heath habitats. Because the habitat is frequently ecotonal between bog and dry heath or grassland habitats, wet heaths are usually considered together with either or both of these habitats. The absence of focus on wet heaths in research and the regional nature of most studies contribute to the problems in identifying indicator species. McKee's (2000) character species for her species-rich subassociation of the Narthecio – Ericetum tetralicis in Connemara, *Cladonia portentosa, Succisa pratensis* and *Polygala serpyllifolia,* might have wider applicability as indicator species for high biodiversity examples of wet heath, but can also be found in a range of other plant communities. It is notable that her relevés have higher average species richness than sets of relevés recorded by Conaghan from wet heath near Galway city and near Birr, Co. Offaly (Conaghan 2001b). Mhic Daeid's (1976) Ulicetosum gallii subassociation of Narthecio – Ericetum tetralicis wet heath in Killarney appears more species-rich than Galway or Offaly heaths, although differences in relevé size make comparisons difficult. *Ulex gallii* may therefore serve as a potential indicator of biodiversity in wet heath.

For blanket bogs, it was not possible to identify particular species that are indicators of particular bog habitats of high biodiversity. Small scale variation in environmental conditions across a bog surface produces a mosaic of different vegetation communities. It is these small communities that are given subassociation status in phytosociological treatments, rather than the larger blanket bog complex. Blanket bogs containing a greater diversity of small vegetation communities, including non-bog habitats such as flushes and pools, have greater biodiversity than more homogeneous bogs (i.e. greater β or ecosystem diversity). Therefore, the presence of a number of species characteristic of different blanket bog subassociations (e.g. Table 8) or other habitat types may indicate high biodiversity.

In peatland habitats, disturbance frequently leads to an increase in plant species richness as new microhabitats are created, allowing colonisation by species that prefer drier, less acidic or more nutrient-rich conditions. Intact, active peatlands with a minimum of disturbance from grazing, trampling, peat-cutting and afforestation are therefore the goal of conservation, rather than increasing species richness. Compositional indicators of intact peatlands would include abundance of species preferring wet, nutrient-poor conditions and scarcity or absence of species indicative of well-drained, degraded or enriched conditions. Structural and functional indicators reflecting intact, relatively undisturbed conditions are probably the best indicators of peatland biodiversity. Structural indicators would include low abundance of bare or excessively trampled peat, relatively low cover of *Molinia caerulea* or *Calluna vulgaris* and heterogeneous microtopography. Functional indicators would include presence of pools, flushes, swallow holes and other hydrological features, signs of active peat formation, low levels of livestock grazing, low levels of peat cutting, lack of ongoing erosion, absence of recent fires and absence of human-made drains.

3 METHODS

3.1 STUDY DESIGN

From examination of data on applications for afforestation grants (Forest Service, unpublished data) and consultations with the Steering Group and external experts, we identified three broad habitat types that are among those typically used for afforestation in Ireland: peatlands, improved grassland and wet grassland (see Section 2.1.3). Our aim was to investigate the biodiversity of these habitats, and the initial effects of afforestation on this biodiversity. Ideally, these aims would be addressed by surveying sites before they were planted, and tracking them through different stages of the forest cycle. However, this approach was not practical, as the Forest Service was unable to disclose information about current planting grants, for reasons of confidentiality, and locating suitable afforestation sites without access to this information proved to be prohibitively time-consuming. An additional factor in this decision was the outbreak of foot-and-mouth disease in 2001, which precluded any surveying of farmland that year. Instead, we paired unplanted study sites with five-year old, first rotation plantations in which Sitka spruce (*Picea sitchensis*) was the main tree species planted, for which the pre-planting habitats matched those of the unplanted sites as closely as possible. We selected eight pairs of sites in each of the three above-mentioned habitat types. This approach had the obvious disadvantage that pre-planting differences in site ecology between unplanted and planted sites could not be ruled out or properly controlled for. However, an advantage of this approach over studying the same sites at different stages of the forest cycle is that planted and unplanted sites can be surveyed almost simultaneously, eliminating the risk of confounding differences due to forest growth stage with variation between years.

3.1.1 Site Selection

The criteria that we used to identify suitable sites were that they were representative examples of the relevant habitat type, and that the planted and unplanted sites were closely matched in terms of relevant environmental conditions such as soil type, drainage, slope, altitude, and proximity of other types of habitats such as forests and rivers. In the peatland category, we selected three lowland blanket bog sites, three upland blanket bog sites and two wet heath sites. Some of the bog sites also included areas of wet heath in a predominantly blanket bog matrix. In the wet grassland category, we did not select highly modified, species-poor sites dominated by *Holcus lanatus, Agrostis stolonifera* and *Juncus effusus* (referred to in the British NVC as MG10 rush-pasture (Rodwell 1992)). Wet grassland sites we selected included plant communities on a range of soils from base-rich limestone-derived soils to acidic soils derived from siliceous rock.

We initially identified the majority of candidate planted sites from the FIPS forest inventory, as areas for which planting grants had been approved (categorised as PGA in FIPS) approximately five years before the surveys were to take place. We refined this selection by examining digitised and hard-copy 1:40000 aerial photographs from series flown in 1995 and 2000, and a digitised landcover classification compiled by the Irish Forest Soils project (Loftus 2002). This enabled us to identify PGA areas that, before they were planted, appeared to conform to one of the three habitat types of interest to us. We also used the aerial photographs and landcover data to identify potential unplanted study sites from non-PGA areas of similar habitat within 5 km distance of the candidate planted sites. We identified other candidate sites by making enquiries of local and regional forest managers and forestry contractors. We then ground-truthed nearly 100 sites, of which we selected 24 pairs of planted and unplanted for our research. In addition to these paired sites, we also surveyed an additional three unplanted sites (one improved grassland and two wet grassland), which were afforested less than a year after we surveyed them. Relevant details and habitat characteristics of all the unplanted sites are summarised in Table 11, locations of all unplanted sites are shown in Figure 3, and site codes and names of the planted sites are given in Table 12.

Table 11. Unplanted site codes (by which they are referred to throughout the remainder of this document), names (these usually correspond to townland names), broad habitat types (PL Peatland, IG Improved grassland and WG Wet grassland), the year in which they were surveyed, and the relevant codes for more precise definitions of the most abundant habitats in each site. Irish habitat codes (Fossitt 2000) are explained in detail in the vegetation chapter (Section 4).

Code	Site name	Site type	Survey year	Irish habitat codes
Caru	Carnamoyle	PL	2004	PB2, PB4
Geau	Gearha	PL	2004	PB3, PB4, HH3, PF1
Incu	Derrybrien commonage	PL	2002	PF1, HH3
Sliu	Slievecorragh	PL	2002	HH3
Tieu	Tieveclougher	PL	2004	PB2, PB4
Togu	Tooreenagowan	PL	2004	PB2
Toou	Tooreenmore	PL	2004	PB3, PB4
Veeu	Ballyveeny	PL	2004	PB3, HH3
Agho	Aghoney	IG	2004	GA1, GS4
Balu	Ballynanoose	IG	2004	GA1, GS4
Bght	Ballybought	IG	2004	GA1
Cast	Castletown	IG	2004	GA1
Doon	Doon	IG	2004	GA1
Gary[1]	Garyandrew	IG	2002	GA1, ED3
Kilb	Kilbraugh	IG	2002	GA1, HD1
Kill	Kilcullen	IG	2004	GA1
Moan	Moanfune	IG	2002	GA1
Bool	Boolavaun	WG	2004	GS4
Clar	Clarbarracum	WG	2004	GS4
Clou	Cloonoughter	WG	2004	GS4
Coog[1]	Coolsnaghtig	WG	2002	GS4, HH1, GA1
Dong[1]	Donaghmore	WG	2002	GS4
Knaw	Knawhill	WG	2004	GS4
Mntp	Mountphillips	WG	2002	GS4
Moig	Moigh	WG	2004	GS4
Mull	Mullanmeen Under	WG	2002	GS4
Rauu	Curraun	WG	2004	GS4

[1] These sites were not paired with planted sites

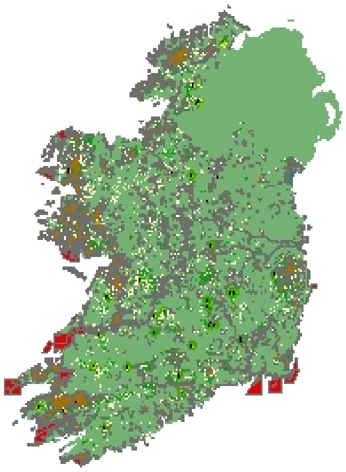
Table 12. Planted site codes (by which they are referred to throughout the remainder of this document), names (these usually correspond to townland names), broad habitat types (PL Peatland, IG Improved grassland and WG Wet grassland), the year in which they were surveyed, and the site codes and names of the unplanted site with which each was paired.

Code	Site name	Site type	Survey	Unplanted code	Unplanted name
Carp	Carnamoyle	PL	2004	Caru	Carnamoyle
Geap	Gearha	PL	2004	Geau	Gearha
Incp	Inchamore	PL	2002	Incu	Inchamore
Slip	Slievecorragh	PL	2002	Sliu	Slievecorragh
Tiep	Tieveclougher	PL	2004	Tieu	Tieveclougher
Togp	Tooreenagowan	PL	2004	Togu	Tooreenagowan
Toop	Tooreenmore	PL	2004	Toou	Tooreenmore
Veep	Ballyveeny	PL	2004	Veeu	Ballyveeny
Lead	Knocklead	IG	2004	Agho	Aghoney
Hanp	Ballycahan	IG	2004	Bght	Ballybought
Ratr	Rathreagh	IG	2004	Cast	Castletown
Clop	Cloonoughter	IG	2004	Clou	Cloonoughter
Flem	Flemingstown	IG	2004	Doon	Doon
Balb	Ballybeagh	IG	2002	Kilb	Kilbraugh
Muny	Mungmacody	IG	2004	Kill	Kilcullen
Kilm	Kilmacow	IG	2002	Moan	Moanfune
Balp	Ballynanoose	WG	2004	Balu	Ballynanoose
Garv	Garvoghil	WG	2004	Bool	Boolavaun
Glas	Garryglass	WG	2004	Clar	Clarbarracum
Glen	Glenfield North	WG	2004	Knaw	Knawhill
Cooa	Coolross	WG	2002	Mntp	Mountphillips
Curr	Curraghnaboul	WG	2004	Moig	Moigh
Gore	Gortnaree	WG	2002	Mull	Mullanmeen Under
Raup	Curraun	WG	2004	Rauu	Curraun

3.2 BIOFOREST GIS

3.2.1 Introduction

Geographical Information Systems (GIS) is a computer technology that links features commonly seen on maps (such as roads, town boundaries, water bodies) with related information not usually presented on maps, such as type of road surface, population, type of agriculture, type of vegetation, or water quality information. A GIS is a unique information system in which individual observations can be spatially referenced to each other.



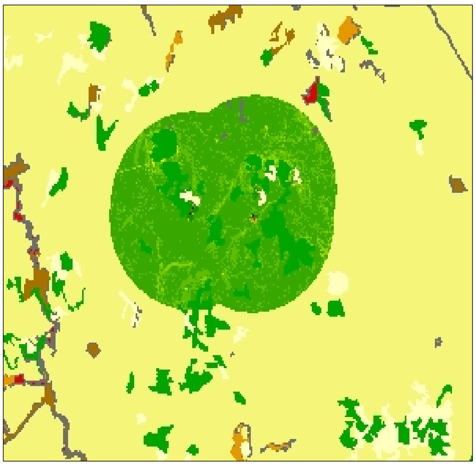
The BIOFOREST 3.1.1 GIS amalgamates botanical and zoological field data, along with existing base data from national organisations such as Coillte, Environmental Protection Agency, Ordnance Survey Ireland and the National Parks and Wildlife Service. The GIS allows access to tabular data, imagery and 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.

3.2.2 DATA TYPES

3.2.2.1 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);
- Designated Areas (Duchas - www.heritagedata.ie);
- Forestry Data i.e. Properties, Compartments, Stands, Old Woodland Database (Coillte licence agreement);
- CORINE land use change (EPA licence agreement).



3.2.2.2 BIOFOREST Survey Data

BIOFOREST 3.1.1 survey data include, amongst others:

- Vegetation data - species and structure;
- Bird species and behaviour data;
- Butterfly data;
- Spider species data;
- Hoverfly species data.

In addition, data on site management and site structure are integrated.

4 VEGETATION

4.1 INTRODUCTION

Sustainable Forest Management (SFM) has been adopted as a core principle of Irish forestry after the UN Convention on Biological Diversity in 1992 and ratification of the Lisbon Agreement in 1998. The goals of SFM are to manage forests in such a way that their multiple functions, economic, ecological and social, are not degraded and they do not damage other ecosystems. In Ireland, SFM principles are implemented by the Forest Service through several guidelines and regulations, including the *Irish National Forest Standard*, (Forest Service 2000e), Code of Best Forest Practice (Forest Service 2000b) and the *Forest Biodiversity Guidelines* (Forest Service 2000c).

Recent plans for afforestation call for an annual rate of 20,000 ha to be planted every year until 2030, resulting in an increase of forest cover from the current 10% to 17% (Department of Agriculture Food and Forestry 1996; Forest Service 2004a). Although current rates of afforestation (Forest Service 2004b) indicate that this target will not be met, they nevertheless suggest a substantial increase in forest cover. As present-day State afforestation by Coillte is virtually nil, afforestation is now being carried out on privately-owned land, primarily by farmers (Forest Service 2004b). It is likely that afforestation is mostly being carried out on the less agriculturally productive farmland, such as peatlands, heathlands and rough grassland (Heritage Council 1999). However, it is lower intensity agricultural land rather than improved grassland or tillage that can support plant communities of biodiversity interest.

In order to determine the positive and negative effects of afforestation on the biodiversity of a given site, it is necessary to know the biodiversity of the habitat to be afforested and also to understand the biodiversity of the forest that will replace it. The latter aspect of forestry and biodiversity was addressed by previous BIOFOREST work (Smith *et al.* 2005). This report is concerned with the biodiversity of habitats likely to be afforested and the initial changes in biodiversity caused by planting. Unfortunately, in Ireland, there is little information available for evaluation of sites that do not have designated conservation status, like NHAs, SACs and SPAs. This situation is likely to change with the establishment of a Biological Records Centre in the near future, but for the moment much of the information on Ireland's plant biodiversity is contained in inaccessible form as unpublished manuscripts, reports and theses, published papers in a range of journals and records held by the Botanical Society of the British Isles and diverse herbaria. Thus, the assessment of biodiversity in non-designated afforestation sites relies on the judgement of the foresters contracted to complete afforestation grant applications. Under guidance from the *Forest Biodiversity Guidelines*, (Forest Service 2000c) the forester must identify and map habitats of biodiversity importance. However, the guidance and training given to foresters (Gittings *et al.* 2004) and even baseline information on the biodiversity and distribution of some Irish plant communities (Section 2.3.2) is inadequate.

Given the current state of knowledge on the biodiversity of afforestation habitats, our research has the following objectives:

1. To assess the biodiversity value of habitats that are typically used for afforestation in Ireland,
2. To develop methods for site assessment prior to afforestation, including the identification of indicators of biodiversity, and
3. To examine the changes in biodiversity that occur in these habitats in the initial period following afforestation.

4.2 METHODS

4.2.1 Field recording

Vegetation and environmental data were collected in each of the site pairs shown in (Figure 3). Because one of the project aims was to develop methods for biodiversity assessment prior to afforestation, vegetation data were collected at three scales: the scale of the entire study site and two different plot scales. Field work was conducted during the summers of 2002 and 2004. The vast majority of taxa were identified to species, but some could only be identified to generic level. Microspecies were recorded as the aggregate species (e.g. *Rubus fruticosus* agg.). Hybrids were recorded separately where possible, particularly *Salix*, *Populus* and *Rosa* hybrids. Trees planted as part of the forestry crop were recorded separately from those originating from natural regeneration or older planting in hedgerows. Nomenclature follows Stace (1997) for vascular plants, Smith (2004) for mosses, Paton (1999) for liverworts and Purvis *et al.* (1992) for lichens. A full list of taxa recorded is given in Appendix 2.

At the site scale, habitats were mapped according to the Heritage Council habitat classification scheme (Fossitt 2000). According to this scheme, young plantations should be classified as conifer plantation (WD4), or as immature woodland (WS2) in the case of broadleaves or broadleaf/conifer mixes. For our purposes, however, we considered it more informative to classify planted sites according to the pre-afforestation habitat. Within each habitat, plant species were recorded on the DAFOR scale: D = dominant, A = abundant, F = frequent, O = occasional, R = rare. All vascular plant species were recorded. Terrestrial bryophytes and lichens were only recorded if they formed patches more than 100 cm^2 or were very frequent, but small in extent. The average height and percent cover of vegetation in each of several layers/ growth form categories was recorded: trees ≥ 5 m tall, saplings < 5 m tall, shrubs (including *Rubus* and subshrubs like *Calluna*) as defined by growth form (i.e. no height criterion), field layer (herbaceous vegetation, including ferns) and bryophyte layer (mosses, liverworts and lichens). Also recorded were the average percent cover of bare soil or rock, percent cover of leaf litter and percent cover of standing water.

The actual site area surveyed for plants differed in a few cases from the habitats surveyed under the hoverfly and bird components of the study because of time constraints and differing objectives. No longer than 2 hours was taken for the recording of any one habitat type within a site. Very small habitats, such as rocky outcrops or small stretches of treeline (WL2) in a site where hedgerows (WL1) are dominant, were not recorded separately from the main habitat type. In peatland sites, there were some inconsistencies in the way in which cutover bog was recorded. Where both cutover and intact bog were present in a site, cutover bog was usually recorded separately from intact bog, but in some cases the habitat types were combined for species and structure recording, although they were mapped separately. In afforested sites, it was sometimes difficult to determine whether a given area of peatland had been lightly cutover or was intact. In 2002, hedgerows associated with streams in grassland sites were not recorded separately from hedgerows without streams. In 2004, we decided to map and record hedgerows associated with streams separately to investigate whether there were important differences in biodiversity compared with "normal" hedgerows. Areas of gorse incursion into grassland were mapped, but not recorded separately if the flora under the gorse was largely similar to that of the wider grassland.

Within the main habitat type (improved grassland, wet grassland or peatland), three 100 m^2 plots were established adjacent to spider pitfall trap plots (Section 6.2.1) and at least 50 m distant from each other. Within each plot, the presence/absence of all vascular plant species and also bryophytes and lichens forming patches greater than 50cm^2 were recorded. Very frequent species forming smaller patches were also recorded, as above. Species that dominated the vegetation were noted, but no other abundance data were collected. The average height and percent cover of vegetation in each of the above layers/ growth form categories and the percent cover in each of the above ground cover classes was recorded. In addition, the following environmental and management data were collected: slope (degrees), aspect (degrees), soil drainage (3-point scale: poor, moderate, or well-drained), grazing intensity (ranked 0-3), recreational use (ranked 0-2) and the presence or absence of forestry drains. Aspect was transformed to a linear scale such that southwest aspect was assigned a value of 1, northeast aspect was assigned a value of 0 and northwest and southeast aspects had a value of 0.5:

$$\textbf{\textit{If}}\ 0 \le x \le 225 \rightarrow \qquad x' = \frac{|x - 45|}{180}$$

$$\textbf{\textit{If}}\ 226 \le x \le 360 \rightarrow \qquad x' = \frac{405 - x}{180}$$

where x = aspect and x' = transformed aspect. Other site management, particularly burning, other drainage, and turf-cutting, were noted. The elevation of each plot was estimated to the nearest 5 m by marking plot locations recorded with a GPS on digitised 1:50,000 Ordnance Survey Discovery Series maps.

Soil samples were collected from each 100 m^2 plot to a depth of 10 cm using a steel tube 3.5 cm in diameter or a trowel if use of the tube was unfeasible (e.g. very wet peats). Nine evenly-distributed subsamples were collected and bulked in the field. Soil pH was determined as soon as possible on field-moist samples, using a glass electrode on a 2:1 distilled water : soil suspension. Soil samples were then air-dried and sieved prior to chemical analyses by Coillte's soil laboratory in Newtownmountkennedy, Co. Wicklow. The laboratory determined percent loss-on-ignition and percentage of total nitrogen, phosphorus, potassium, calcium and magnesium according to international standards. Soil element percentages were converted to mg/L after calculation of soil bulk density following the method of Jeffrey (1970). Adjacent to one of the 100 m^2 plots in each site, a soil pit was dug, the soil profile was sketched and the soil type determined, according to the Irish classification (Gardiner & Radford 1980). For each horizon , colour, depth, stoniness and rooting depth were recorded, and the texture (e.g. clay loam, sandy silt loam) of each mineral horizon was determined by hand (Trudgill 1989). Depth and composition of the litter layer were recorded. Where soil varied significantly between plots within a site, additional soil profiles were recorded as necessary.

Within each 100 m^2 plot, two 4 m^2 plots were nested in the corners of the larger plot nearest to and furthest away from the spider pitfall traps. In afforested sites, plots were placed so that the 4 m^2 plots did not include any forestry drains. In these plots, the percentage cover of all vascular plants and all identifiable bryophytes and lichens was recorded to the nearest 5%. Species covering less than 5% were recorded in two classes: 3% (for cover of 1 – 5%) and 0.5% (for cover of < 1%). The average height and percent cover of vegetation in each of the layers/ growth form categories and the percent cover in each of the ground cover classes described under the site survey were recorded. In afforested sites, the height and diameter at breast height (dbh- 1.3 m) of each sapling occupying 5% or more of the plot were recorded (dbh was not recorded for trees < 2 m tall).

4.2.2 Data Analysis

The following biodiversity metrics were calculated for each 4 m^2 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, and hence low diversity) (Magurran 2004). In addition, all plant species were classified according to their ecological characteristics in several categories: woodland affinity (typical of open habitats, characteristic woodland species, species frequently found in both wooded and non-wooded habitats), moisture preference (dry, mesic, damp, wet), soil pH preference (acidophilic, neutral or broadly tolerant, basophilic) and native/alien status. The classifications of species were determined using habitat and autecological information contained in Webb *et al.* (1996), Clapham *et al.* (1987), Stace (1997), Fitter and Peat (1994), Jermy *et al.* (1982), Hubbard (1984), Paton (1999), Watson (1981), Smith (2004), Purvis *et al.* (1992) and Dobson (2000), as well as our own experience and judgement. 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 classifications are given in Appendix 2. The species richness of plants in these categories was calculated for each plot. The biodiversity metrics for the two 4 m^2 plots in each 100 m^2 plot were averaged to avoid pseudoreplication. Species richness in the above categories was also calculated for species presence/absence data collected over each 100 m^2 plot.

Vegetation composition of the unplanted sites was investigated using NMS ordination and flexible-beta cluster analysis on Sørensen distance measures (Legendre & Legendre 1998). For ordinations of habitat data collected using the DAFOR scale, the abundance ratings were converted to numerical rankings as follows: rare- 1, occasional- 2, frequent- 4, abundant- 8 and dominant- 16. Rankings followed this scale rather than a simple linear scale as the chosen scale provides a better reflection of true relative abundances. 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 200 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. Indicator species were identified for clusters using the indicator species analysis method of Dufrêne and Legendre (1997). Flexible-beta clustering was initially performed 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). Additional cluster analyses were also performed with $\beta = 0$ and $\beta = 0.5$. The topologies produced were then compared using the sum of significant indicator values, and the analysis producing the highest sum was selected (Dufrêne & Legendre 1997). Maximising the sum of significant indicator values maximises the variation in species distribution explained by a clustering solution. This procedure was also used to inform decisions on the number of clusters to select.

Differences between vegetation clusters in biodiversity metrics and structural and functional variables were tested using parametric and non-parametric statistics. Variables were inspected prior to testing to see if they met the assumptions of normality and homogeneity of variance required by parametric statistics. Data transformations or non-parametric statistics were used if required; the latter were always used for ordinal data. ANOVAs were used to test for differences among groups, followed by Tukey's HSD or Ryan's Q post-hoc tests, or Tamhane's T2 tests for variables with unequal error variances among groups (Day & Quinn 1989; Sokal & Rohlf 1995). Non-parametric tests used in lieu of ANOVA were the Mann-Whitney U test for two groups and the Kruskal-Wallis test for three or more groups. Differences in frequency of qualitative variables, such as soil type, among groups were tested using likelihood ratio χ^2 tests (or G-tests in Sokal and Rohlf (1995)).

Differences in biodiversity metrics between planted and unplanted site pairs were assessed using paired t-tests of data averaged at the site scale. Differences in plant species composition were investigated using blocked multiple-response permutation procedures (MRBP) with site pairs as blocks and using Euclidian distance measures. This method tests whether the mean distances among sample units are significantly less within pre-established groups (planted vs. unplanted, in this case) than would be established by chance. Effect size is summarised by the statistic A; A = 0 if the average within-group distance is equal to that expected by chance, A > 0 if average within-group distance is less than that expected by chance and A = 1 if the species composition in each sample unit is identical (McCune & Mefford 1997). In ecology, significant values of A are often less than 0.1, and A values exceeding 0.3 are considered fairly high. The above analyses were conducted separately for each habitat group (improved grasslands, wet grasslands and peatlands), and trees planted as part of the forestry crop were omitted from the datasets. Plant species composition in planted and unplanted sites was also investigated using NMS ordination.

Multivariate analyses were conducted using PC-Ord (McCune & Mefford 1997) and univariate analyses were performed with SPSS (SPSS 2001).

4.2.3 Biodiversity indicators

Two sets of biodiversity indicators were developed. The first set consists of potential indicator species identified by earlier BIOFOREST work, the habitats review (Section 2) and the BIOFOREST report on diversity over the forest cycle (Smith *et al*. 2005), and confirmed by independent research in this study. The biodiversity of each of the groups of sample units identified by cluster analysis was assessed. Potential compositional indicators that were also significant indicator species of one or more of the high biodiversity groups were considered confirmed indicators. Potential structural and functional indicators that were significantly associated with one or more of the high biodiversity vegetation clusters were also considered confirmed indicators.

The second set consists of a group of new provisional indicators suggested by the results of this study. Previously unspecified species that were significant indicator species of vegetation groups of biodiversity importance were considered provisional compositional indicators of biodiversity if they had a low indicator value for groups of little biodiversity interest, do not occur frequently in other habitats and are not overly difficult for non-specialists to identify. Structural and functional variables that had significant discriminatory power between high and low biodiversity vegetation clusters were also considered provisional indicators.

Indicators were developed separately for grasslands and peatlands.

4.3 RESULTS

4.3.1 Overview of habitat survey

No species in the Red Data Book for vascular plants (Curtis & McGough 1988) or bryophytes listed as rare in Holyoak (2003) were found in the habitat or plot surveys; no plant species recorded are listed in the Flora (Protection) Order, 1999. In the habitat survey, we recorded 531 taxa of vascular plants, bryophytes and lichens in 133 habitats in the 51 sites (Appendix 3). Habitat maps and full species lists with DAFOR ratings are given in the GIS database. The species composition and vegetation structure of the afforested improved grassland sites more closely corresponded with the habitat category dry meadows and grassy verges (GS2) than with improved agricultural grassland (GA1) (see Section 4.3.4). However, the latter classification was retained for ease of comparison with the unplanted sites.

In unplanted grassland sites, an average of 41-42% of the species recorded in the habitat survey were only found in non-grassland habitats, such as hedgerows, ditches, stone walls, scrub, etc (Table 13). In planted and unplanted grassland sites, 30-35% of species were found in both grassland and additional habitats. The proportion of species occurring only in the grassland habitats and not in the additional habitats was generally higher in the afforested sites than in the unplanted sites (Table 13), although these differences were not statistically significant.

Table 13. Percent (± standard error) of species occurring only in grassland habitats, only in additional (non-grassland) habitats and species common to both in planted and unplanted improved grassland and wet grassland sites. Differences between planted and unplanted sites within the same categories are not significant according to t-tests.

	Improved Grassland		Wet Grassland	
	Planted (n = 8)	**Unplanted** (n = 9)	**Planted** (n = 8)	**Unplanted** (n = 10)
Unique to additional habitat	37.9 ± 5.9	41.9 ± 4.2	33.5 ± 4.3	40.9 ± 4.0
Common to both	30.5 ± 1.8	31.4 ± 2.0	34.9 ± 2.9	33.5 ± 1.9
Unique to grassland habitat	32.5 ± 6.0	26.7 ± 4.3	31.5 ± 5.6	25.6 ± 3.0

In the two peatland site pairs containing flushes (GEAP/GEAU and INCP/INCU), 19.6% (± 5.7% se) of species were recorded only from flushes and not from the surrounding blanket bog or wet heath habitats. Two unplanted lowland blanket bog sites (TOOU and VEEU) included streams; in these sites, 30.2% and 27.5%, respectively, of species occurring in the streams and along the banks were not found in the wider blanket bog habitat.

4.3.2 Biodiversity of unplanted sites

4.3.2.1 Habitat groups

Vascular plant species richness at the 100 m^2 and 4 m^2 scales was significantly higher in unplanted wet grasslands than in unplanted improved grasslands or peatlands (Table 14). Bryophyte and lichen species richness was significantly different among the three habitat groups at both scales, highest in peatlands and lowest in improved grasslands. Total species richness was significantly lower in improved grassland 100 m^2 and 4 m^2 plots than in wet grasslands or peatlands. Simpson's diversity and Berger-Parker evenness in 4 m^2 plots also followed the same pattern.

Table 14. Means (± se) of vascular plant species richness (VSR), bryophyte and lichen species richness (BLSR), total species richness (TOTSR), Simpson's diversity index and Berger-Parker evenness index in 100 m^2 plots and 4 m^2 plots in unplanted sites in the three pre-defined habitat groups. N = number of sites. Means with the same letter superscript are not significantly different according to Ryan's Q multiple comparisons tests (VSR, TOTSR) or Tamhane's T2 tests for unequal variances (BLSR, Simpson's, Berger-Parker).

	Improved Grasslands (n = 9)	Wet Grasslands (n = 10)	Peatlands (n = 8)
100 m^2 Plots			
VSR	17.2 ± 1.7 ^{B}	31.1 ± 2.5 ^{A}	19.0 ± 1.8 ^{B}
BLSR	0.2 ± 0.1 ^{C}	2.7 ± 0.7 ^{B}	9.1 ± 0.6 ^{A}
TOTSR	17.4 ± 1.8 ^{B}	33.8 ± 2.7 ^{A}	28.1 ± 2.0 ^{A}
4 m^2 Plots			
VSR	10.6 ± 1.2 ^{B}	18.1 ± 1.2 ^{A}	12.2 ± 0.8 ^{B}
BLSR	0.5 ± 0.2 ^{C}	2.4 ± 0.5 ^{B}	6.9 ± 0.8 ^{A}
TOTSR	11.1 ± 1.3 ^{B}	20.5 ± 1.4 ^{A}	19.1 ± 0.9 ^{A}
Simpson's	0.61 ± 0.06 ^{B}	0.84 ± 0.01 ^{A}	0.83 ± 0.01 ^{A}
Berger-Parker	0.53 ± 0.05 ^{B}	0.29 ± 0.02 ^{A}	0.32 ± 0.02 ^{A}

The majority (67.6%) of plant species in improved grasslands were those preferring mesic conditions (Figure 4a). Species preferring damp conditions were the most common of the moisture groups in wet grasslands (39.7%), and species preferring wet habitats were the most common group in peatlands (34.4%). The majority of species in all three habitat groups were those preferring open conditions, whereas typical woodland plants made up less than 2% of the flora in any group (Figure 4b). Species often found in both wooded and unwooded habitats formed a lower proportion of the flora in improved grasslands than in wet grasslands or peatlands. Acidophilic species were much more common in peatlands (89.0%) than in wet grasslands or improved grasslands; calcicoles made up less than 6% of the flora in any habitat type (Figure 4c). Fewer peatland species were classified as competitors (22.8%) than in improved (70.3%) or wet (67.9%) grasslands (Figure 4d). Plants with a stress-tolerator strategy were less frequent in improved grasslands than in the other two habitat groups, whereas the majority of improved grassland plants (84.9%) adopted a ruderal strategy (at least in part, as a plant can exhibit a combination of strategies, for example competitive ruderals).

4.3.2.2 Habitat scale

An NMS ordination of the habitat scale DAFOR data for the unplanted sites, omitting linear habitats (e.g. hedgerows, streams), scrub and woodland, produced a significant two-dimensional solution (stress = 10.33, p = 0.005). The first dimension contrasted peatland sites with grasslands, and improved grasslands separated from wet grasslands along the second dimension (Figure 5). Cluster analyses were performed, and the solution using β = 0 and four final clusters produced the highest sum of significant indicator values. However, the fourth cluster identified had only one habitat (GEAU PF2), and so the second-best solution was chosen. This solution used β = -0.5 and identified three clusters, which corresponded exactly with the three pre-determined habitat groups, improved grassland, wet grassland and peatland (Figure 5). Ancillary habitat types tended to cluster together with the main habitat type present in each site. For example, the cluster analysis placed an area of bracken-dominated rough grassland (HD1) in KILB in the improved grassland group together with KILB GA1; however, the ordination indicates that there are notable differences in the vegetation composition of the two habitats (Figure 5). The sum of significant indicator values declined slightly at each step after the three-cluster stage in the β = -0.5 analysis until the six-cluster stage, when indicator values declined more steeply. Indicator values were then recalculated separately for peatland and grassland plots, so that the indicator species identified discriminate better between subgroups within these two easily distinguished broad habitat types.

The six-cluster solution separated improved grasslands into a fertile, highly improved Group I (top of Figure 5) and a more acidic, less fertile Group II below. Group I is referable to the phytosociological association Lolio-Cynosuretem, and Group II may have some affinities to the semi-natural dry to mesic grassland community Centaureo-Cynosuretum (Table 15; c.f. Section 2.3.3.3). Group III, containing wet grasslands, was not further divided between the three- and six-cluster stages. Indicator species of Group III include both Junco-Molinietum and Senecioni-Juncetum character or differential species. *Juncus conglomeratus, Carex panicea* and *Molinia caerulea* are character species of the Junco-Molinietum (Table 15), and *Senecio aquaticus* is a differential species of the Senecioni-Juncetum (see Table 5). Group III is a heterogeneous cluster, containing sites with very different species compositions. After a division based largely on abundance of *Agrostis stolonifera*, further subdivisions tended to separate individual sites from the main body. DONG and BOOL appeared to support a more distinctive flora than the other wet grassland sites, as these sites were the first and second to form their own clusters.

Peatlands were separated into a group of degraded, heavily grazed wet heath and upland bog sites (Group V, lower left in Figure 5), a group comprised mostly of oceanic heaths and flushes (Group IV, right in Figure 5) and a group of oceanic, wet bog sites (Group VI, upper left in Figure 5; Table 16).

Table 15. Significant indicator values for three grassland clusters identified by flexible-beta cluster analysis (β = -0.50) of habitat scale DAFOR data. Maximum indicator values for a group are shown in bold. P values were calculated using Monte Carlo randomisation tests (1000 randomised runs) and give the probability that the maximum indicator value is no greater than would be expected by chance.

Species	Group I n = 7	Group II n = 5	Group III n = 10	P
Poa annua	**68**	8	0	0.004
Urtica dioica	**57**	17	7	0.003
Stellaria media	**57**	0	0	0.009
Plantago major	**53**	15	13	0.016
Cirsium vulgare	**52**	22	5	0.036
Digitalis purpurea	0	**60**	0	0.008
Anthoxanthum odoratum	7	**60**	23	0.019
Plantago lanceolata	4	**58**	26	0.014
Bellis perennis	23	**58**	4	0.026
Cynosurus cristatus	8	**58**	23	0.028
Ulex europaeus	1	**57**	15	0.034
Dactylis glomerata	20	**55**	4	0.026
Trifolium repens	29	**54**	16	0.019
Pteridium aquilinum	0	**53**	1	0.031
Cerastium fontanum	23	**51**	24	0.005
Veronica chamaedrys	1	**48**	2	0.032
Juncus conglomeratus	0	0	**100**	0.001
Juncus acutiflorus	0	3	**91**	0.001
Filipendula ulmaria	1	2	**83**	0.001
Carex panicea	0	0	**70**	0.003
Dactylorhiza maculata / fuchsii *	0	0	**70**	0.004
Stellaria graminea	0	8	**64**	0.007
Galium palustre	2	2	**61**	0.008
Lychnis flos-cuculi	0	0	**60**	0.015
Agrostis canina	0	0	**60**	0.017
Molinia caerulea	0	0	**60**	0.017
Juncus effusus	1	30	**59**	0.025
Ranunculus flammula	0	9	**54**	0.02
Carex ovalis	0	3	**50**	0.03
Iris pseudacorus	0	4	**49**	0.028
Centaurea nigra	0	4	**47**	0.035
Senecio aquaticus	0	3	**42**	0.036

* Mostly *D. fuchsii*, but some non-flowering specimens could not be separated with certainty.

Table 16. Significant indicator values for three peatland clusters identified by flexible-beta cluster analysis (β = -0.50) of habitat scale DAFOR data. Maximum indicator values for a group are shown in bold. P values were calculated using Monte Carlo randomisation tests (1000 randomised runs) and give the probability that the maximum indicator value is no greater than would be expected by chance.

Species	**Group IV** n = 6	**Group V** n = 5	**GroupVI** n = 5	**P**
Carex viridula	**83**	0	0	0.002
Succisa pratensis	**79**	6	1	0.001
Cirsium palustre	**71**	9	1	0.009
Trifolium repens	**67**	2	2	0.018
Juncus conglomeratus	**67**	0	0	0.011
Myrica gale	**66**	0	20	0.019
Dactylorhiza maculata	**65**	3	8	0.011
Ranunculus flammula	**64**	2	2	0.016
Juncus acutiflorus	**63**	38	0	0.026
Salix aurita	**62**	0	10	0.015
Carex echinata	**59**	35	3	0.044
Lotus pedunculatus	**58**	0	3	0.047
Leucobryum glaucum	**56**	0	13	0.040
Carex panicea	**56**	25	19	0.002
Bellis perennis	**56**	0	3	0.049
Blechnum spicant	**54**	8	21	0.039
Sphagnum fallax	1	**94**	0	0.001
Rhytidiadelphus loreus	0	**80**	0	0.007
Polytrichum commune	4	**79**	4	0.006
Deschampsia flexuosa	0	**76**	1	0.008
Hylocomium splendens	16	**72**	1	0.006
Nardus stricta	7	**71**	4	0.030
Galium saxatile	18	**64**	0	0.016
Sphagnum palustre	18	**59**	8	0.009
Juncus squarrosus	21	**56**	14	0.009
Sphagnum cuspidatum	5	12	**64**	0.023
Drosera rotundifolia	19	6	**58**	0.044
Racomitrium lanuginosum	9	4	**57**	0.045
Campylopus atrovirens	0	3	**51**	0.047
Erica tetralix	25	23	**51**	0.019
Molinia caerulea	35	23	**42**	0.043

4.3.2.3 100 m^2 Plot Scale

The NMS ordination of the 100 m^2 plot presence/absence data yielded a significant two-dimensional solution (stress = 12.93, p = 0.01). The first dimension, which accounted for the great majority of the variation in the data, was a gradient from peatlands, through wet grasslands to improved grasslands (Figure 6). The second dimension reflects degree in grassland improvement ; species most strongly correlated with Dimension 2 are *Lolium perenne* (τ = -0.63), *Potentilla erecta* (τ = 0.52), *Rhytidiadelphus squarrosus* (τ = 0.50) and *Carex panicea* (τ = 0.48). The flexible-beta cluster analysis with β = -0.25 was better than those using β = -0.5 and β = 0, according to consistently higher sums of significant indicator values computed for all clustering stages with 2 - 10 groups. The 8-cluster stage had the highest sum of squared indicator values in the β = -0.25 analysis, and is shown in Figure 6. Indicator values were then recalculated separately for peatland and grassland plots, so that the indicator species identified discriminate better between subgroups within grasslands and peatlands.

Peatland plots were separated into two groups. Group 1 contains plots on wet, mainly lowland blanket bog. Indicator species of this group include several characteristic wet bog species (Table 17). Two of these species, *Schoenus nigricans* and *Pleurozia purpurea*, are character species of the Atlantic blanket bog association Pleurozio purpureae-Ericetum tetralicis (Doyle 1982; Doyle & Moore 1980), and Moore (1968) proposed *Racomitrium lanuginosum* as a differential species of the association. Group 2 plots included the more heavily grazed upland blanket bog and wet heath sites. Accordingly, indicator species of this group include those characteristic of wet heath and Moore's (1962) *Juncus squarrosus* subassociation of the Vaccinio – Ericetum tetralicis association of upland blanket bog, including *Juncus squarrosus*, *Deschampsia flexuosa* and *Galium saxatile* (Table 17). The presence of *Holcus lanatus* as a Group 2 indicator species also suggests conditions enriched beyond those normally encountered in bog or wet heath.

Cover of bryophyte layer vegetation, in this case *Sphagnum* and other typical bog mosses such as *Leucobryum glaucum* and *Campylopus atrovirens*, was significantly lower in Group 1 plots (21.8 ± 4.1%) than in Group 2 (55.8 ± 4.3%), according to a t-test ($t_{22} = 5.70$, $p < 0.0001$). Leaf litter from small shrubs, forbs and graminoids was significantly higher in Group 1 plots (13.9 ± 3.8%) compared with Group 1 plots (6.0 ± 1.0%), according to a Mann-Whitney U test (U = 35.5, n = 23, p = 0.034). Slope and soil P concentrations were significantly lower in Group 1 plots (1.7 ± 0.6° and 68 ± 8 mg/L P) than in Group 2 plots (5.1 ± 0.8° and 168 ± 17 mg/L P), according to a MANOVA ($F_{1,22} = 10.7$, $p = 0.003$ and $F_{1,22} = 25.0$, $p = 0.0001$). There were significant differences in grazing intensity recorded in the field, according to a Mann-Whitney U test (U = 40.0, p = 0.047). In Group 1, 45.5% of plots were moderately or heavily grazed, whereas 76.9% of Group 2 plots were moderately or heavily grazed. Other management practices did not sufficiently distinguish Group 1 from Group 2 plots.

Table 17. Significant indicator values for two peatland clusters identified by flexible-beta cluster analysis (β = -0.25) of 100 m^2 vegetation presence/absence data. Maximum indicator values for a group are shown in bold. Only species with an indicator value ≥ 25 are shown. P values were calculated using Monte Carlo randomisation tests (1000 randomised runs) and give the probability that the maximum indicator value is no greater than would be expected by chance.

Species	Group 1 n = 11	Group 2 n = 13	P
Odontoschisma sphagni	**73**	0	0.002
Drosera rotundifolia	**64**	11	0.013
Rhynchospora alba	**64**	0	0.002
Racomitrium lanuginosum	**57**	1	0.01
Pleurozia purpurea	**55**	0	0.005
Schoenus nigricans	**55**	0	0.006
Cladonia portentosa	**51**	3	0.038
Anthoxanthum odoratum	3	**85**	0.001
Rhytidiadelphus squarrosus	0	**77**	0.001
Hylocomium splendens	1	**76**	0.002
Polytrichum commune	0	**69**	0.001
Luzula multiflora	0	**69**	0.002
Juncus squarrosus	3	**62**	0.012
Deschampsia flexuosa	0	**62**	0.002
Sphagnum fallax	0	**62**	0.003
Carex echinata	7	**57**	0.035
Galium saxatile	0	**46**	0.012
Vaccinium myrtillus	0	**46**	0.012
Carex nigra	1	**46**	0.031
Holcus lanatus	0	**38**	0.029
Rhytidiadelphus loreus	0	**38**	0.032
Nardus stricta	0	**38**	0.044

The remaining six groups were comprised of grassland plots. One group contained only one plot, BGHTF3, an outlier that had only three species present: *Lolium perenne*, *Holcus lanatus* and *Dactylis glomerata* (Figure 6). Group 3 members were on average the most acidic of the grassland sites and tended to be poor in P, Ca and Mg and high in total number of species (Table 18). Indicator species include those characteristic of damp or wet, acidic conditions, such as *Agrostis capillaris, A. canina* s.l., *Lophocolea bidentata* and *Carex echinata* (Table 19). Several Group 3 indicator species are also character or differential species of the Junco acutiflori-Molinietum association of more oligotrophic wet grasslands (see Table Table 5). In addition, the presence of *Agrostis canina* s.l. and *Carex viridula* as significant indicator species suggests affinities to O'Sullivan's (1982) *Agrostis canina* subassociation representing the least grazed, most oligotrophic Junco-Molinietum grasslands. Shrub cover in Group 3 plots was low (2.3 ± 1.1%) but was higher on average than other groups (Kruskal-Wallis H = 16.5, df = 4, p = 0.003), particularly Groups 6 and 7 which had no shrub cover in any plot. Group 3 also had the highest mean cover of bryophyte layer vegetation (15.1 ± 4.0%) and leaf litter (10.4 ± 1.4%) of all groups; covers of these layers varied significantly among groups (respectively H = 22.0, df = 4, p = 0.0002 and H = 20.1, df = 4, p = 0.0005). There were also significant differences in field layer height among groups (Kruskal-Wallis H = 30.0, df = 4, p < 0.0001), with the highest field layer in Group 3 plots (82 ± 6 cm).

The three plots in Group 4 were all from KNAW, characterised by another set of species of acid, wet grassland, such as *Danthonia decumbens, Carex pilulifera, Molinia caerulea* and *Succisa pratensis* (Table 19). KNAW appears to be another example of Junco-Molinietum grassland, distinguished by some unusual species, such as *Corylus avellana* regeneration and *Euphrasia rostkoviana*, which were not found in any other plots. Species richness was high, but not significantly different from any other groups because of small sample size (Table 18). Soil pH in the plots was relatively low, and N, Ca and Mg concentrations were also generally lower than most other grassland groups, except for Group 3 (Table 18). Field layer height averaged 60 ± 6 cm tall.

Indicator species of Group 5, such as *Carex hirta, Lathyrus pratensis, Filipendula ulmaria* and *Iris pseudacorus* (Table 19), suggest a more base-rich type of wet grassland. *Carex hirta* as an indicator species, and high indicator values for other species, such as *Senecio aquatica,* suggest affinities to the Senecioni-Juncetum acutiflori association (see table Table 5). The Group 5 plots had soils with higher pH and nutrient concentrations, especially Ca, and were located in lower elevation sites than the previous wet grassland Groups 3 and 4 (Table 18). Species richness was significantly higher than that in Groups 6 or 7 (Table 18). Field layer averaged 69 ± 7 cm height.

Group 7 includes the most fertile (Table 18) and intensively managed of the grasslands plots, referable to the Lolio-Cynosuretum. This group was also the most species poor on average. *Urtica dioica* is an indicator of nutrient-rich conditions, and the remainder of the indicator species are common weeds of *Lolium perenne*-dominated improved pasture (Table 19). Soil N was generally low and some plots had extremely high Ca concentrations, although the majority did not differ much from the base-rich wet grasslands of Group 5 (Table 18). Field layer height was much lower than the above groups, averaging 23 ± 3 cm tall. Bryophyte layer cover was quite low, averaging only 0.2 ± 0.1%.

Group 6 includes plots pre-identified as both wet grasslands and improved grasslands. Agricultural weeds such as *Bellis perennis, Rumex obtusifolius* and *Cerastium fontanum* are indicator species of these semi-improved grasslands (Table 19). Species richness was significantly higher than in Group 7, but lower than that in Groups 3 and 5 (Table 18). Although indicator species and association character species do not match well (Section 2.3.3.3), Group 6 might best be referred to the Centaureo-Cynosuretum juncetosum, based on the relative (to Group 7) richness of weed species and the presence of *Alopecurus geniculatus*, a species of damp, disturbed ground. Group 6 plots were often located in hilly country, on higher elevation, more steeply sloping sites (Table 18). As with Group 7, mean field layer height was low (33 ± 5 cm tall).

Group 6 and 7 sites were generally on brown earths and brown podzolics, whereas the soils of Groups 3-5 were usually gleys. However, two Group 6 sites were also on gley soils, and there were no significant differences in soil type among groups, according to a likelihood ratio test (likelihood ratio $\chi^2 = 16.9$, df = 12, p = 0.19). There were significant differences in stoniness of the soil A-layer, according to a Kruskal-Wallis test (H = 10.4, df = 4, p = 0.009); Group 6 and 7 sites had median rankings of "frequent" stones, whereas median stoniness for Group 3 and 5 sites was "none". In general, grasslands in Groups 3-5 received less frequent inputs of fertiliser, whereas most Group 6 and 7 grasslands received one or more fertiliser applications per year. There were no significant differences among groups in grazing intensity as estimated in the field (Kruskal-Wallis H = 8.48, df = 4, p = 0.075). Note, however, that our estimations of grazing intensity, based on one site visit, are strongly influenced by timing of livestock rotation among fields.

Table 18. Means (± se) of environmental and soil variables and median soil pH in the five grassland groups formed by clustering of 100 m^2 species presence/absence data.

	Group 3 (n = 12)	Group 4 (n = 3)	Group 5 (n = 9)	Group 6 (n = 21)	Group 7 (n = 11)
Total SR	35.7 ± 2.3 ^{A}	30.7 ± 3.5 ^{ABC}	40.0 ± 3.5 ^{A}	21.1 ± 0.8 ^{B}	14.2 ± 1.2 ^{C}
Slope (°) [2]	6.1 ± 0.8	1.7 ± 0.7	1.4 ± 0.4	6.9 ± 1.1	4.9 ± 1.2
Transformed aspect [3]	0.29 ± 0.07 ^{AB}	0.14 ± 0.06 ^{A}	0.32 ± 0.07 ^{AB}	0.53 ± 0.05 ^{B}	0.33 ± 0.07 ^{AB}
Elevation (m) [2]	111 ± 10	125 ± 0	67 ± 7	191 ± 16	94 ± 10
pH [2]	5.49	5.68	5.82	6.18	6.67
LOI (%)	29.1 ± 5.2	18.7 ± 1.7	25.1 ± 4.1	17.3 ± 0.9	15.9 ± 2.3
N (mg/L) [2]	4617 ± 255	3912 ± 407	5191 ± 277	4092 ± 99	3650 ± 200
P (mg/L) [4]	455 ± 28 ^{AB}	428 ± 74 ^{A}	598 ± 42 ^{AB}	719 ± 35 ^{B}	713 ± 71 ^{B}
K (mg/L) [5]	4410 ± 594 ^{AB}	5608 ± 318 ^{AB}	3218 ± 501 ^{A}	6825 ± 448 ^{B}	5784 ± 670 ^{AB}
Ca (mg/L) [2]	1600 ± 245	849 ± 61	4553 ± 1528	2320 ± 188	12,348 ± 4528 [6]
Mg (mg/L) [7]	1332 ± 200	1566 ± 72	1738 ± 195	1902 ± 151	2744 ± 415

[1] Groups with different letter superscripts differ significantly according to Tamhane's T2 post-hoc tests for unequal variances following a significant ANOVA ($F_{4,51}$ = 30.3, $p < 0.0001$).

[2] Significant differences ($p \leq 0.001$) among groups according to a Kruskal-Wallis test.

[3] Groups with different letter superscripts differ significantly according to Tukey's HSD post-hoc tests following a significant ANOVA ($F_{4,51}$ = 4.01, p = 0.007) and MANOVA (Wilk's Λ = 0.316, $p \leq 0.0001$).

[4] Groups with different letter superscripts differ significantly according to Tukey's HSD post-hoc tests following a significant ANOVA ($F_{4,51}$ = 7.02, $p \leq 0.0001$) and MANOVA (Wilk's Λ = 0.316, $p \leq 0.0001$).

[5] Groups with different letter superscripts differ significantly according to Tukey's HSD post-hoc tests following a significant ANOVA ($F_{4,51}$ = 6.33, $p \leq 0.0001$) and MANOVA (Wilk's Λ = 0.316, $p \leq 0.0001$)

[6] Median Ca = 3171 mg/L.

[7] Significant differences among groups (Kruskal-Wallis H = 10.3, p = 0.03).

Table 19. Significant indicator values for five grassland clusters identified by flexible-beta cluster analysis (β = -0.25) of 100 m^2 vegetation presence/absence data. Maximum indicator values for a group are shown in bold. Only species with an indicator value ≥ 25 are shown; also omitted are species occurring only in 1 plot in Group 4. P values were calculated using Monte Carlo randomisation tests (1000 randomised runs) and give the probability that the maximum indicator value is no greater than would be expected by chance.

Species	Group 3 n = 10	Group 4 n = 3	Group 5 n = 9	Group 6 n = 21	Group 7 n = 11	P
Agrostis capillaris	**59**	0	3	6	0	0.001
Lophocolea bidentata	**57**	0	2	0	0	0.006
Agrostis canina s.l.	**54**	0	20	1	0	0.001
Calliergonella cuspidata	**52**	0	15	0	0	0.011
Carex echinata	**50**	0	0	0	0	0.001
Pellia epiphylla	**50**	0	0	0	0	0.005
Epilobium obscurum	**46**	0	9	2	0	0.016
Luzula multiflora	**45**	29	0	0	0	0.018
Cirsium dissectum	**40**	31	0	0	0	0.007
Rhytidiadelphus squarrosus	**35**	0	24	0	0	0.02
Juncus conglomeratus	**33**	7	18	0	0	0.022
Epilobium palustre	**33**	0	0	0	0	0.017
Carex binervis	**33**	0	0	0	0	0.019

Species	Group 3 n = 10	Group 4 n = 3	Group 5 n = 9	Group 6 n = 21	Group 7 n = 11	P
Juncus bulbosus	**33**	0	2	0	0	0.034
Senecio jacobaea	**29**	0	4	3	3	0.035
Pseudoscleropodium purum	**27**	0	8	0	0	0.048
Carex viridula	**25**	0	0	0	0	0.049
Corylus avellana	0	**67**	0	0	0	0.002
Euphrasia rostkoviana	0	**67**	0	0	0	0.002
Danthonia decumbens	1	**59**	0	0	0	0.001
Carex pilulifera	3	**53**	0	0	0	0.008
Lotus pedunculatus	9	**52**	10	0	0	0.007
Molinia caerulea	18	**52**	6	0	0	0.012
Succisa pratensis	16	**47**	14	0	0	0.014
Carex panicea	20	**45**	14	0	0	0.021
Prunella vulgaris	11	**45**	9	2	0	0.024
Deschampsia cespitosa	0	**44**	11	0	0	0.02
Potentilla erecta	37	**44**	5	0	0	0.025
Leontodon autumnalis	0	**41**	0	5	3	0.006
Angelica sylvestris	2	**38**	10	0	0	0.031
Senecio aquatica	0	**36**	25	0	0	0.046
Trifolium pratense	8	**33**	10	11	4	0.048
Stellaria graminea	0	0	**89**	0	0	0.001
Carex hirta	0	0	**73**	0	0	0.002
Lathyrus pratensis	0	8	**68**	0	0	0.001
Cirsium palustre	23	0	**51**	4	0	0.001
Centaurea nigra	0	0	**51**	0	0	0.003
Carex disticha	0	0	**47**	1	0	0.006
Filipendula ulmaria	6	6	**45**	1	0	0.017
Phleum pratense	0	0	**45**	0	8	0.002
Iris pseudacorus	0	0	**44**	0	0	0.01
Galium palustre	8	8	**42**	0	0	0.012
Festuca pratensis	0	0	**40**	0	0	0.025
Equisetum palustre	5	0	**32**	0	0	0.034
Lythrum salicaria	2	0	**27**	0	0	0.021
Hypericum tetrapterum	0	0	**26**	0	2	0.024
Lolium perenne	0	0	8	**41**	**41**	0.001
Bellis perennis	1	0	0	**39**	8	0.025
Rumex obtusifolius	0	0	1	**31**	13	0.042
Cerastium fontanum	17	0	24	**31**	12	0.001
Trifolium repens	14	0	24	**30**	16	0.001
Alopecurus geniculatus	0	0	0	**29**	0	0.04
Cirsium arvense	0	0	1	0	**69**	0.001
Dactylis glomerata	0	0	12	9	**41**	0.046
Elytrigia repens	0	0	6	0	**39**	0.043
Urtica dioica	0	0	0	0	**27**	0.019

4.3.2.4 4 m^2 Plot Scale

The NMS ordination of the 4 m^2 plot data (means of the two plots within the same 100 m^2) produced a significant three-dimensional solution (stress = 10.79, p = 0.01). The first dimension reflects a gradient from peatlands to grasslands (Figure 7). The species most strongly correlated with Dimension 1 include: *Holcus lanatus* (τ = 0.70), *Ranunculus repens* (τ = 0.60), *Trifolium repens* (τ = 0.50), *Molinia caerulea* (τ = -0.59), *Erica tetralix* (τ = -0.59) and *Calluna vulgaris* (τ = -0.58). Dimension 2 primarily reflects abundance of *Lolium perenne* (τ = -0.79) (Figure 7a); conversely, species preferring damp or wet, acidic conditions are positively correlated with Dimension 2, such as *Carex panicea* (τ = 0.58), *C. echinata* (τ = 0.55) and *Potentilla erecta* (τ = 0.52). Dimension 3 contrasts more nutrient-rich wet habitats, characterised by *Agrostis stolonifera* (τ = 0.59), *Holcus lanatus* (τ = 0.40) and *Carex hirta* (τ = 0.36), with acidic, nutrient-poor wet habitats with *Eriophorum angustifolium* (τ = -0.53), *E. vaginatum* (τ = -0.49) and *Calluna vulgaris* (τ = -0.43) (Figure 7b).

The flexible-beta cluster analysis of the 4 m^2 plot data with $\beta = -0.25$ was superior to the alternatives, according to the sums of significant indicator values. The 10-cluster solution had the highest sum of significant indicator values and is superimposed on the NMS ordination in Figure 7. Indicator values were then recalculated separately for peatland and grassland plots. One of the ten groups formed included one with a single member, TOGUF3, which was unusual in that it had much higher cover of *Calluna vulgaris* and *Narthecium ossifragum*, compared with other peatland plots, and an absence of *Molinia caerulea*. The rest of the peatland plots were separated into three groups. Group B contained a mixture of flushed plots and plots degraded by turf cutting or overgrazing characterised by *Carex panicea*, *C. echinata* and *Juncus bulbosus*, among others (Table 20). The soils in this group tended to be more mineral-rich (67.7 ± 12.9% loss-on-ignition) and have greater N concentrations (3653 ± 413 mg/L) than Groups A (94.2 ± 0.9% loi, 2922 ± 143 mg/L N) and C (93.3 ± 2.2% loi, 2798 ± 260 mg/L N). Group B plots also had a higher cover of bare peat (11.1 ± 3.3%) than the other groups (A: 4.7 ± 2.1%, C: 0.6 ± 0.3%). Group C plots were in degraded upland bogs and wet heaths, and were characterised by *Polytrichum commune, Hylocomium splendens Deschampsia flexuosa, Sphagnum fallax* and *S. palustre* (Table 20). Bryophyte layer cover was much higher in this group (64.4 ± 6.8%) than in Group A (21.0 ± 1.8%) or Group B (23.9 ± 4.1%). Group A contained the remainder of the peatland sites that did not fit into either of the two previous categories; they tended to be less disturbed than other peatland sites, and had some characteristic wet bog species as indicators (Table 20).

Table 20. Significant indicator values for three peatland clusters identified by flexible-beta cluster analysis (β = -0.25) of 4 m² vegetation abundance data. Maximum indicator values for a group are shown in bold. Only species with an indicator value ≥ 25 are shown. P values were calculated using Monte Carlo randomisation tests (1000 randomised runs) and give the probability that the maximum indicator value is no greater than would be expected by chance.

	A	**B**	**C**	
Species	n = 10	n = 7	n = 6	**P**
Molinia caerulea	**56**	27	14	0.001
Drosera rotundifolia	**53**	24	0	0.044
Odontoschisma sphagni	**52**	8	0	0.021
Schoenus nigricans	**50**	0	0	0.012
Carex panicea	5	**69**	9	0.005
Juncus bulbosus	1	**69**	0	0.007
Carex echinata	2	**65**	6	0.021
Sphagnum papillosum	7	**60**	0	0.013
Anagallis tenella	0	**43**	0	0.031
Polytrichum commune	0	0	**99**	0.001
Hylocomium splendens	0	3	**91**	0.001
Deschampsia flexuosa	2	0	**81**	0.001
Sphagnum fallax	0	3	**78**	0.002
Sphagnum palustre	4	1	**72**	0.004
Galium saxatile	0	0	**67**	0.003
Rhytidiadelphus loreus	0	0	**66**	0.005
Anthoxanthum odoratum	3	12	**66**	0.008
Eriophorum vaginatum	8	6	**66**	0.021
Plagiothecium undulatum	0	0	**65**	0.003
Luzula multiflora	5	0	**64**	0.003
Pleurozium schreberi	0	1	**62**	0.017
Potentilla erecta	18	19	**59**	0.013
Juncus squarrosus	3	2	**59**	0.015
Vaccinium myrtillus	1	0	**58**	0.005
Rhytidiadelphus squarrosus	0	8	**51**	0.04

Two wet grassland sites formed their own groups: the three plots in DONG formed Group E and the MULL plots formed Group F. DONG/Group E was a particularly species-rich and diverse site on high pH, calcium- and nitrogen-rich soils (Table 21). In contrast, K concentrations were lower on average than in other groups. The group is probably best referable to the Senecioni-Juncetum, judging by its base status and the high indicator value for *Potentilla anserina*, an association differential species (Table 22 and Table 5). The high indicator values for *Phleum pratense* and *Lotus corniculatus*, however, indicate some affinity with the *Trifolium repens* subassociation of the Junco-Molinietum (Section 2.3.3.4). Grazing pressure was generally low and the site had not been grazed during the summer prior to sampling.

MULL/Group F was also a relatively species-rich, diverse site (Table 21); it was moderately to heavily grazed and supported a higher proportion of species preferring wet conditions (23.8 ± 2.3%), relative to other wet grassland sites. Soil pH and K concentrations were relatively low, while soil N was high (Table 21). The MULL plots are an example of the wet, oligotrophic *Agrostis canina* subassociation of the Junco-Molinietum (Section 2.3.3.4). Field layer cover was lower in Group F plots (78.3 ± 3.0%) than in other groups, which averaged 91.7 - 98.3% cover (Kruskal-Wallis H = 18.7, df = 5, p = 0.002). On the other hand, bare ground cover was higher (4.4 ± 1.5%) than in other groups, which averaged 0.4 - 1.6% cover of bare ground ($H = 30.4$, df = 5, $p < 0.0001$). Bryophyte layer cover was also high (14.2 ± 3.0%) relative to Groups E (0.8 ± 0.5%), G (2.1 ± 1.4%) and I (0.2 ± 0.05%).

The majority of the wet grasslands clustered into Groups D and G. Group D had higher species richness (VSR 18.2 ± 0.9, BLSR 3.8 ± 0.5) than Group G (VSR 15.4 ± 1.2, BLSR 1.1 ± 0.4), but differences in total species richness were not significant (Table 21). Group D also had a greater proportion of acidophilic species than Group G (35.6 ± 3.7% and 25.0 ± 2.8%, respectively), including the indicator species *Cirsium dissectum, Molinia caerulea* and *Carex panicea* (Table 22), despite there being little difference in soil pH between the two groups (Table 21). The primary difference between the two grassland types appears to be related to soil fertility: soil P concentrations were lower in Group D than Group G (Table 21), and a higher percentage of Group D species had stress-tolerant strategies than Group G (68.2 ± 2.4% and 60.2 ± 2.4%). However, there were no clear differences between the groups (or Groups E and F) in frequency or amount of fertiliser application. The majority of cover in Group D plots is occupied by a mixture of species, usually including *Juncus acutiflorus, Anthoxanthum odoratum, Holcus lanatus* and *Agrostis stolonifera*; *Molinia caerulea* is prominent in most of the plots, but absent in some of the more base-rich plots, such as those in RAUU. In Group G plots some of the same species are abundant, but *Holcus lanatus* and *Agrostis stolonifera* are more important in the sward, *Molinia caerulea* is always absent and *Juncus effusus* is sometimes prominent. Bryophyte layer cover was also higher on average in Group D (15.4 ± 5.4%) than in Group G (2.1 ± 1.4%).

Improved grasslands split into two groups with low species richness and diversity, Group H and Group I. The latter group represents the highly improved, very low diversity (Table 21), *Lolium perenne*-dominated improved grasslands, whereas the former sites are semi-improved dry grasslands, with somewhat higher species richness provided by the consistent presence of some characteristic "weeds" (Table 21 and Table 22). Group H sites tended to be on steeper slopes in higher elevation areas than Group I sites (Table 21). Field layer height was lower in these groups (H: 17 ± 2 cm tall, I: 24 ± 3 cm tall) than in the wet grassland groups (56 - 77 cm tall) (Kruskal Wallis $H = 30.6$, df = 5, $p < 0.0001$).

Table 21. Means (± se) of environmental and soil variables and median soil pH in the six grassland groups formed by clustering of 4 m^2 species abundance data.

	Group D (n = 11)	**Group E (n = 3)**	**Group F (n = 3)**	**Group G (n = 14)**	**Group H (n = 6)**	**Group I (n = 20)**
TOTSR [1]	22.0 ± 1.1 ^{AB}	25.7 ± 2.6 ^{A}	26.2 ± 5.9 ^{A}	16.5 ± 1.3 ^{BC}	14.9 ± 1.8 ^{BC}	9.8 ± 0.8 ^{C}
Simpson's [2]	0.86 ± 0.01 ^{AB}	0.89 ± 0.01 ^{A}	0.83 ± 0.03 ^{AB}	0.82 ± 0.02 ^{B}	0.73 ± 0.04 ^{ABC}	0.57 ± 0.05 ^{C}
Slope (°) [3]	3.3 ± 0.8	0.7 ± 0.7	8.0 ± 2.5	3.9 ± 0.4	12.0 ± 2.3	5.3 ± 0.7
Transformed aspect	0.21 ± 0.05	0.45 ± 0.09	0.44 ± 0.12	0.42 ± 0.08	0.51 ± 0.12	0.43 ± 0.05
Elevation (m) [3]	94 ± 9	92 ± 2	145 ± 3	106 ± 16	267 ± 13	130 ± 13
pH [3]	5.72	6.18	5.61	5.46	6.26	6.44
LOI (%) [4]	26.1 ± 5.3	26.3 ± 5.8	27.7 ± 2.3	23.1 ± 3.2	16.8 ± 0.8	15.7 ± 1.4
N (mg/L) [3]	4225 ± 220	5794 ± 324	5623 ± 226	4547 ± 185	4061 ± 49	3739 ± 131
P (mg/L) [5]	449 ± 34 ^{A}	539 ± 27 ^{AB}	520 ± 58 ^{AB}	610 ± 36 ^{AB}	788 ± 82 ^{B}	742 ± 49 ^{AB}
K (mg/L) [6]	4365 ± 682^{AB}	2842 ± 356^{A}	3416 ± 578^{A}	5353 ± 616^{AB}	7495 ± 223^{B}	6152 ± 498^{AB}
Ca (mg/L) [7]	2065 ± 467	7595 ± 4380 [8]	2533 ± 425	1757 ± 250	2432 ± 167	9098 ± 2837 [9]
Mg (mg/L)	1389 ± 147	1816 ± 318	1830 ± 314	1689 ± 226	2163 ± 245	2346 ± 272

[1] Groups with different letter superscripts differ significantly according to Tukey's HSD post-hoc tests following a significant ANOVA ($F_{5,50}$ = 16.1, $p < 0.0001$) and MANOVA (Wilk's Λ = 0.346, $p < 0.0001$), with one outlying plot in Group I omitted.

[2] Groups with different letter subscripts differ significantly according to Tamhane's T2 post-hoc tests for unequal variances following a significant ANOVA ($F_{5,50}$ = 5.06, p = 0.001) and MANOVA (Wilk's Λ = 0.346, $p < 0.0001$), with one outlying plot in Group I omitted.

[3] Significant differences ($p \leq 0.001$) among groups according to a Kruskal-Wallis test.

[4] Significant differences among groups (Kruskal-Wallis H = 13.8, p = 0.017)

[5] Groups with different letter subscripts differ significantly according to Tukey's HSD post-hoc tests following a significant ANOVA ($F_{5,51}$ = 5.59, p = 0.0004) and MANOVA (Wilk's Λ = 0.495, p = 0.002).

[6] Groups with different letter subscripts differ significantly according to Tukey's HSD post-hoc tests following a significant ANOVA ($F_{5,51}$ = 3.77, p = 0.006) and MANOVA (Wilk's Λ = 0.495, p = 0.002).

[7] Significant differences among groups (Kruskal-Wallis H = 15.3, p = 0.009).

[8] Ca concentrations (mg/L) in plots: F1 = 2514, F2 = 16,316 and F3 = 3954.

[9] Median Ca = 2780 mg/L.

Table 22. **Significant indicator values for six grassland clusters identified by flexible-beta cluster analysis (β = -0.25) of 4 m^2 vegetation abundance data. Maximum indicator values for a group are shown in bold. Only species with an indicator value ≥ 25 are shown. P values were calculated using Monte Carlo randomisation tests (1000 randomised runs) and give the probability that the maximum indicator value is no greater than would be expected by chance.**

	D	**E**	**F**	**G**	**H**	**I**	
Species	n = 11	n = 3	n = 3	n = 14	n = 6	n = 20	**P**
Cirsium dissectum	**60**	0	0	1	0	0	0.005
Molinia caerulea	**58**	1	3	0	0	0	0.004
Carex panicea	**52**	15	8	0	0	0	0.013
Pseudoscleropodium purum	**40**	0	9	0	0	0	0.034
Thuidium tamariscinum	**36**	0	0	0	0	0	0.025
Iris pseudacorus	0	**97**	0	0	0	0	0.001
Phleum pratense	0	**86**	0	1	0	0	0.001
Potentilla anserina	3	**74**	0	0	0	2	0.002
Hypericum tetrapterum	0	**67**	0	0	0	0	0.007
Mentha aquatica	0	**66**	0	0	0	0	0.007
Lotus corniculatus	4	**61**	0	0	0	0	0.015
Filipendula ulmaria	17	**57**	0	1	0	0	0.006
Carex nigra	17	**57**	0	1	0	0	0.006
Carex flacca	2	**56**	1	0	0	0	0.023
Galium palustre	5	**50**	5	0	0	0	0.022
Centaurea nigra	2	**48**	3	2	0	0	0.035
Senecio jacobea	0	0	**98**	0	0	0	0.001
Pellia epiphylla	1	0	**89**	1	0	0	0.001
Epilobium obscurum	1	2	**77**	3	0	0	0.005
Juncus effusus	1	2	**77**	12	0	0	0.002
Carex viridula	0	0	**67**	0	0	0	0.007
Epilobium palustre	0	0	**66**	0	0	0	0.009
Juncus bulbosus	0	0	**65**	0	0	0	0.006
Calliergonella cuspidata	17	4	**64**	1	0	0	0.008
Cardamine sp	0	0	**57**	1	3	11	0.006
Carex echinata	6	0	**52**	0	0	0	0.014
Ranunculus repens	4	3	**47**	24	2	8	0.011
Ranunculus flammula	1	0	**46**	8	1	0	0.033
Agrostis canina s.l.	3	41	**45**	2	0	0	0.028
Agrostis stolonifera	11	0	0	**45**	0	25	0.009
Holcus lanatus	9	27	12	**28**	8	15	0.047
Cynosurus cristatus	0	11	4	5	**73**	0	0.001
Bellis perennis	0	0	0	2	**70**	4	0.003
Trifolium repens	1	3	15	10	**46**	14	0.029
Conopodium majus	0	0	0	0	**33**	0	0.014
Luzula campestris	0	0	0	0	**33**	0	0.024
Lolium perenne	0	0	0	2	35	**62**	0.001
Taraxacum officinale agg.	0	0	0	1	7	**57**	0.017

4.3.2.5 Biodiversity assessment of groups

Correspondence for a given plot among the three clustering topologies produced using data at different scales was generally good, but not exact (Appendix 4). Each clustering identified a group of improved grasslands and a group of semi-improved grasslands, although the membership of these groups varied depending on scale of analysis. The clustering of wet grasslands was more variable. The habitat-scale and 100 m^2 plot scale clusterings of peatlands generally distinguished oceanic, mainly lowland blanket bogs from wet heaths and most upland blanket bogs.

None of the grassland groups of the classification of habitat-scale data clearly represent a plant community of significant biodiversity. Groups I and II represent improved and semi-improved grasslands of little biodiversity interest. Group III is a heterogeneous grouping which includes all wet grasslands. At the 100 m^2 scale, Groups 3 and 4 represent the less modified, more oligotrophic wet grasslands. As semi-natural grasslands, they are of conservation value, particularly in landscapes where semi-natural habitats are uncommon. Group 5 represents the less modified, more base-rich wet grasslands. As with Groups 3 and 4, these semi-natural grasslands can be of conservation value. Groups 6 and 7 represent semi-improved and highly improved grasslands of no real biodiversity interest.

At the 4 m^2 plot scale, Groups E and F represent two species-rich, high diversity wet grassland sites, MULL and DONG. The community composition and species richness of the plots in these sites show that they are semi-natural wet grasslands of high biodiversity conservation value. Group D is a somewhat heterogeneous grouping of less fertile, semi-natural wet grasslands, containing both base-rich and base-poor plots. Species richness and diversity in these plots is relatively high, and therefore this group is also of biodiversity interest. In contrast, Group G plots tended to be of relatively low species richness, with *Holcus lanatus* and *Agrostis stolonifera* usually as the most abundant species. There are some similarities of this group with the British National Vegetation Classification type MG10, "*Holcus lanatus-Juncus effusus* rush pasture" which in turn is referable to the Centaureo-Cynosuretum juncetosum subassociation of semi-improved damp grasslands (Rodwell 1992). The composition and relatively low richness of Group G suggests that as a whole it is not of biodiversity interest, although some of the particular sites sampled by these plots may be. Groups H and I represent respectively semi-improved and highly improved grasslands, dominated by a lesser or greater degree by *Lolium perenne*. As such, they are of little biodiversity value.

Turning to peatlands, Group VI is comprised of five habitats from three sites: VEEU, lowland blanket bog and wet heath in Mayo, GEAU, intact and cutover lowland blanket bog in Kerry and TOGU, intact and cutover upland blanket bog in Kerry (Figure 5). Each of the sites contains an intact, wet part, of high biodiversity value (Section 2.4.6) and a part disturbed by heavy grazing or peat cutting. In our opinion, the intact bog sections of GEAU and VEEU were the highest quality peatland habitats surveyed. Both contained interesting hydrological features: a flush in GEAU and several well-developed pools and a stream in VEEU. This group has a marked oceanic character. The upland blanket bog site included (TOGU) was located in Kerry and supported some characteristic lowland oceanic bog species. It can probably be considered an oceanic highland bog *sensu* Schouten (1984), transitional between lowland and upland blanket bogs (Section 2.4.3). As a whole, Group VI is the habitat-scale grouping of most biodiversity interest.

In contrast, Group V contains heavily grazed upland bogs and wet heaths with some cutover areas; because these habitats are generally highly disturbed and modified, they are of less biodiversity value. Group IV was an ill-defined grouping with high and low biodiversity members, including wet heaths and flushes from two western sites (INCU and

GEAU), a largely cutover lowland blanket bog site (TOOU) and a dry heath habitat with some wet heath affinities (COOG). Of these, the poor flush in INCU had considerable biodiversity interest, despite being heavily grazed in parts, and had a rich and varied species composition reflecting wet, acidic but nutrient enriched conditions. (The presence of pre-famine lazy beds also provided some cultural interest.) The remainder of the habitats had less value; they were generally intermediate between Groups V and VI.

Group 1 of the 100 m^2 plot scale classification represents wet oceanic bogs, and includes all the plots located in the habitat-scale Group VI above, with the addition of the three plots from TOOU located in cutover lowland blanket bog (Appendix 4). Group 2 unified upland blanket bogs and heaths, but as many of the plots were highly disturbed by overgrazing, it does not represent a high biodiversity group even though some sites within the group may be of conservation value. In particular, the three plots in poor flush in INCU were included in Group 2 and are of biodiversity interest as discussed above.

The peatland Groups A, B and C formed by clustering of the 4 m^2 groups are not as coherent as the clusters generated using larger scale data (Appendix 4). The small scale of the 4 m^2 plots highlights the vegetation patterns in peatlands produced by small scale differences in topography and hydrology. In cutover peatlands, there are also small scale features, including turf-banks and disturbed but wet flats, that are highlighted by use of small plots. Therefore, the groups formed by these plots and their indicator species will not reflect the biodiversity value of the larger site as well as they do smaller-scale ecological differences.

4.3.3 Biodiversity indicators in unforested habitats

4.3.3.1 Confirmation of potential biodiversity indicators

4.3.3.1.1 Potential indicators

Potential biodiversity indicators have been identified from two sources: the review of grasslands and peatlands earlier in this report (Sections 2.3.5.3 and 2.4.6.3) and the indicators of biodiversity for pre-thicket plantations developed in the BIOFOREST report on diversity over the forest cycle (Smith *et al.* 2005). The latter set of indicators were not developed for any specified community or habitat type, but are generally associated with sites on more acidic, organic rich soils. Indicators can be divided into negative indicators of biodiversity, i.e. compositional, structural or functional variables that are associated with plant communities of low biodiversity value, and positive biodiversity indicators. Plant communities of low biodiversity value are those that support low diversity of plant species and/or are highly modified from semi-natural conditions. The potential positive biodiversity indicators are summarised in Table 23 below. Potential negative indicators of biodiversity identified by Smith *et al.* (2005) include high abundance of *Agrostis stolonifera* or *Dactylis glomerata* and high cover of graminoids. Potential negative indicators of biodiversity identified in the grasslands review (Section 2.3.5.3) include high cover of *Lolium perenne*, low cover of forbs, intensive grazing, recent reseeding, intensive fertilisation and intensive herbicide use.

Table 23. Potential compositional, structural and functional positive indicators of biodiversity identified by previous BIOFOREST work.

Pre-thicket plantations[1]

Compositional	*Structural*	*Functional*
Calluna vulgaris	High shrub cover	High LOI
Erica tetralix	High bryophyte cover	Gley or peat soil
Molinia caerulea		Low P
		Low Ca
		Low pH

Peatlands[2]

Compositional	*Structural*	*Functional*
Cladonia portentosa[3]	Low cover of bare peat	Presence of flushes
Succisa pratensis[3]	Low cover of *Molinia caerulea*	Presence of pools
Polygala serpyllifolia[3]	Low cover of *Calluna vulgaris*	Low grazing
Ulex gallii[3]		Little or no peat cutting
		Presence of swallow holes
		Absence of erosion
		Absence of fire
		Absence of drains

Grasslands[4]

Compositional

Agrostis canina
Caltha palustris
Carex echinata
Carex nigra
Carex panicea
Carex pulicaris
Carex viridula
Cirsium dissectum
Dactylorhiza fuchsii
Danthonia decumbens
Festuca pratensis
Hydrocotyle vulgaris
Hylocomium splendens
Juncus conglomeratus
Luzula campestris
Lychnis flos-cuculi
Molinia caerulea
Nardus stricta
Pedicularis sylvatica
Potentilla anglica
Potentilla erecta
Prunella vulgaris
Pseudoscleropodium purum
Ranunculus flammula
Senecio aquaticus
Succisa pratensis
Thuidium tamariscinum
Viola palustris

[1] Indicators identified by Smith *et al.* (2005).
[2] Indicators identified in Section 2.4.6.3.
[3] Wet heath only.
[4] Indicators identified in Section 2.3.5.3.

4.3.3.1.2 Grasslands

Several potential compositional indicators listed in Table 23 were independently identified as significant indicator species for one or more of the oligotrophic wet grassland groups, Groups 3 and 4 and Groups D and F, listed in Table 19 and Table 22, respectively. These confirmed compositional indicators of biodiversity are: *Agrostis canina* s.l., *Carex echinata, Carex panicea, Carex viridula, Cirsium dissectum, Danthonia decumbens, Juncus conglomeratus, Molinia caerulea, Potentilla erecta, Prunella vulgaris, Pseudoscleropodium purum, Ranunculus flammula, Senecio aquaticus, Succisa pratensis* and *Thuidium tamariscinum*. Only two compositional indicators were confirmed for base-rich grasslands in Group 5 and Group E, *Carex nigra* and *Festuca pratensis*. Several of the above species also had relatively high indicator values for both oligotrophic and base-rich wet grasslands, including *Agrostis canina* s.l., *Carex nigra, Carex panicea, Juncus conglomeratus* and *Senecio aquaticus*. The potential negative indicator, *Lolium perenne*, was an indicator species for the improved grassland Groups 6 and 7 and Groups I and H (Table 19 and Table 22, respectively), and is therefore confirmed as a negative biodiversity indicator. *Agrostis stolonifera* was a significant indicator for the low biodiversity wet grassland Group G (IndVal = 45), but also had some value as an indicator for the more diverse Group D (IndVal = 11). Similarly, *Dactylis glomerata* was a significant indicator species for the improved grassland Group 7 (IndVal = 41), but also occurred in wet grassland Group 5 (IndVal = 12). These two species could also serve as negative indicators of biodiversity if treated with some care.

Cover of graminoids, a potential negative structural indicator of biodiversity, was significantly higher in 4 m^2 plots in low-diversity Groups G and I (90.7 ± 3.3% and 88.1 ± 2.7%, respectively) than in the oligotrophic wet grassland Group D (73.1 ± 2.6%), but was not higher than in any of the other biodiverse wet grassland groups, according to Tamhane's T2 post-hoc tests for unequal error variances following a significant ANOVA ($F_{5,51} = 3.42$, $p = 0.01$). Graminoid (including rush) cover of less than 75% can be considered a confirmed indicator of biodiversity. Plots in the above low-diversity Groups G and I also supported significantly lower forb cover (16.0 ± 2.2% and 15.0 ± 2.3%, respectively) than the high biodiversity Groups D, E and F (29.6 - 45.5% forb cover), according to Ryan's Q post-hoc tests following a significant ANOVA ($F_{5,51} = 8.08$, $p < 0.0001$). Semi-improved grassland plots in Group H had significantly lower forb cover (20.9 ± 4.9%) than Group E plots. Thus it seems that forb cover of greater than 25% is a confirmed structural indicator of biodiversity. Shrub cover and bryophyte cover were higher in oligotrophic wet grassland Group 3 (Section 4.3.2.3) than in Groups 6 and 7, and can therefore serve as structural indicators of biodiversity, despite some high biodiversity groups having low shrub or bryophyte cover.

Among the potential functional indicators of biodiversity, soil type did not effectively discriminate between high and low biodiversity groups. The soils of improved and semi-improved grasslands in Groups H and I tended to have lower loss-on-ignition than wet grasslands (Table 21), but the low-diversity wet grassland Group G was not effectively distinguished from the remainder of the wet grasslands. Soil pH and Ca concentrations also did not discriminate well between high and low biodiversity grasslands, with some low diversity wet grasslands having low pH and Ca and high biodiversity base-rich wet grasslands having high pH and Ca (Table 18 and Table 21). Although grazing intensity, fertiliser and herbicide use were generally higher in improved grasslands, these factors did not consistently differ among groups and cannot be considered confirmed indicators of functional biodiversity in grasslands. The two sites that had been reseeded within the past 10 years were both assigned to improved grassland Groups 7 and I, and so recent reseeding can be considered a confirmed negative indicator; however, several other sites that had not been recently reseeded were also assigned to low-biodiversity groupings.

4.3.3.1.3 Peatlands

The peatlands of high biodiversity value that we surveyed were largely lowland blanket bogs or otherwise had oceanic characteristics (Section 4.3.2.5). Accordingly, the indicators discussed below and in Section 4.3.3.2.2 below are biased towards identifying lowland blanket bogs and may not serve as useful indicators of high-quality upland blanket bog in areas with a less oceanic climate, such as the midlands, Wicklow and eastern parts of Galway.

Molinia caerulea and *Erica tetralix*, potential compositional indicator species identified by Smith *et al.* (2005), are indicator species for the high biodiversity habitat-scale group of lowland blanket bogs, Group VI (Table 16). However, these species also have relatively high indicator values for Groups IV and V, and so cannot be considered as confirmed. Of the four species identified as potential indicators for biodiversity in wet heaths (Table 23), *Cladonia portentosa* was actually an indicator species for the 100 m^2 Group 1 of lowland blanket bogs and *Polygala serpyllifolia* and *Ulex gallii* were not indicator species for any group. *Succisa pratensis* was a significant indicator for Group IV of oceanic heaths and flushes, but this group contains sites of varying biodiversity.

There was no difference in shrub cover or cover of bare peat between the two 100 m^2 peatland groups. Group 2, of largely degraded upland blanket bogs and wet heaths, had significantly higher cover of bryophytes than Group 1, which appeared to be of higher biodiversity value (Section 4.3.2.3). Therefore, bryophyte cover is not a good indicator of biodiversity in peatlands. Similarly, low cover of *Molinia caerulea* or *Calluna vulgaris* were proposed as indicators of biodiversity in peatlands, but covers (mean of 4 m^2 plots) were actually significantly higher in Group 1 plots, according to Mann-Whitney U tests ($U = 32.0$, $p = 0.021$ and $U = 37.5$, $p = 0.048$).

Groups 1 and 2 were not distinguished by soil organic carbon or pH, but Group 1 plots had significantly lower average soil P (Section 4.3.2.3). Another confirmed potential functional indicator was grazing intensity, which was significantly lower in Group 1 than in Group 2 plots. The remaining potential functional indicators of biodiversity, presence of flushes and other hydrological features and absence (or low levels) of disturbance are measures of the integrity of peatlands and therefore reasons why certain peatlands are of high biodiversity value (Section 2.4.6.3). Therefore these factors are *de facto* indicators of biodiversity. The contribution of peatland flushes and streams to the number of plant species occurring in peatland ecosystems (Section 4.3.1) provides an example of the importance of hydrological features to peatland biodiversity. The flushes themselves, such as that in INCU, can also be of considerable biodiversity interest.

4.3.3.2 New provisional indicators

4.3.3.2.1 Grasslands

The indicator species of the highly improved habitat-scale Group I (Table 15) could be considered indicators of species poverty when taken in the context of grasslands where *Lolium perenne* is abundant.

Indicator species of Groups 3 and 4 (Table 19) that have not been pre-identified and confirmed in the above sections could be considered provisional biodiversity indicators for oligotrophic wet grasslands. Those that also have high indicator values for other groups, such as *Calliergonella cuspidata*, would not make good biodiversity indicators, as they are likely to be found in a broad range of wet grassland types. Other indicator species for these groups are characteristic of other grassland types, such as *Agrostis capillaris*, which is typical of dry-humid acid grassland (GS3) (Fossitt 2000). Eliminating these species and those more difficult for non-specialists to identify, the best provisional biodiversity indicator of oligotrophic wet grasslands is *Pellia epiphylla*. This is a large thallose liverwort; although it can be difficult to identify to species, it is relatively easy to identify to genus level. Other *Pellia* species are much less common, except in wet calcareous sites, and are not likely to indicate low biodiversity habitats. The best provisional biodiversity indicators for Group 5 include: *Carex hirta, Centaurea nigra, Iris pseudacorus, Lathyrus pratensis* and *Stellaria graminea*. These species indicate base-rich wet grasslands.

The indicator species of DONG / Group E that would be the best provisional biodiversity indicators for base-rich wet grasslands are: *Centaurea nigra, Hypericum tetrapterum, Iris pseudacorus* and *Mentha aquatica*. These species are easy to identify, particularly when in flower; other indicator species listed in Table 22 may also serve as biodiversity indicators. MULL / Group F indicator species reflect more oligotrophic conditions. The best provisional biodiversity indicators for this grassland type include: *Juncus bulbosus* and *Pellia epiphylla*. Other indicator species (Table 22) are more difficult for non-specialists or they occur frequently in other vegetation groups or plant communities. Because Group D is a somewhat heterogeneous grouping containing both base-rich and base-poor plots, there are few good indicator species for this group (Table 22). The best biodiversity indicators for this group, *Cirsium dissectum, Molinia caerulea* and *Carex panicea*, have been identified by the habitats review and confirmed in the section above.

Few new provisional structural and functional indicators of biodiversity were identified for grasslands. At both the 100 m^2 and 4 m^2 plot scales, field layer height over 50 cm tall had some power in discriminating between wet grasslands and improved grasslands (Sections 4.3.2.3 and 4.3.2.4). However, field layer height would not be able to differentiate between species-rich wet grasslands and species-poor grasslands with significant *Juncus effusus* incursions. It is also sensitive to recent grazing use or cutting for hay and silage, and is therefore not a good indicator. A better provisional indicator appears to be K concentrations in the soil, which were generally below 5000 mg/L in high biodiversity wet grasslands in contrast to higher concentrations in wet and improved grasslands of little biodiversity interest (Table 21).

4.3.3.2.2 Peatlands

As Group VI includes the most intact peatland sites (although with some cutover elements), the indicator species of Group VI in Table 16 could serve as provisional biodiversity indicators, if they were not associated mainly with the disturbed sections. *Sphagnum cuspidatum* and *Drosera rotundifolia* are typical wet bog species. *Racomitrium lanuginosum* and *Campylopus atrovirens* are also good provisional biodiversity indicators; the latter moss is a character species of the Pleurozio purpurea-Ericetum tetralicis association of Atlantic blanket bogs (Doyle 1982; Doyle & Moore 1980). Although three of these species are mosses, they are quite distinctive and not very difficult for non-specialists to identify. *Erica tetralix* and *Molinia caerulea* would not be good indicator species for reasons discussed in Section 4.3.3.1.3 above. As with the habitat-scale Group VI, 100 m^2 scale Group 1 indicator species that reflect wet, oligotrophic conditions may be provisional indicators of biodiversity. All indicator species of this group listed in Table 17 meet this criterion, with the exception of *Cladonia portentosa* which is frequently found on bogs negatively affected by drainage or peat cutting (personal observation). Most Group 1 indicator species prefer wet conditions and acid pH (Appendix 2), and all of them are diagnostic species for one or more levels in lowland blanket bog phytosociologal classification (Doyle 1982; Doyle & Moore 1980). *Odontoschisma sphagni* would not make a good indicator, however, as it is a minute liverwort. No additional structural or functional factors were found to consistently distinguish peatlands of high biodiversity from those of little biodiversity interest.

4.3.4 Effects of afforestation

4.3.4.1 Biodiversity metrics

There were no significant differences in mean vascular plant species richness in 100 m^2 plots between planted and unplanted sites, according to paired t-tests (Table 24). However, vascular plant species richness at the 4 m^2 plot scale was significantly higher in unplanted sites in all habitat groups. Bryophyte and lichen species richness in 100 m^2 plots was significantly higher in planted improved grasslands and peatlands than in unplanted sites, but was higher in unplanted than planted peatlands at the 4 m^2 plot scale. Simpson's diversity was lower in planted wet grassland and peatland 4 m^2 plots.

Table 24. Means (± se) of vascular plant species richness (VSR), bryophyte and lichen species richness (BLSR), Simpson's diversity index (Simp) and Berger-Parker evenness index (BP) in 100 m^2 plots and 4 m^2 plots in unplanted and planted sites in the three pre-defined habitat groups. N = 8 site pairs in each group. P values are the result of paired t-tests between site pairs; significant differences among planted and unplanted sites are indicated in bold type.

	Improved Grasslands			Wet Grasslands			Peatlands		
	UP	P	p	UP	P	p	UP	P	p
100 m^2									
VSR	17.1 ± 2.0	14.5 ± 1.8	0.297	28.9 ± 1.7	25.8 ± 2.3	0.321	19.0 ± 1.8	19.4 ± 1.5	0.764
BLSR	**0.2 ± 0.1**	**1.0 ± 0.3**	**0.036**	2.4 ± 0.6	4.1 ± 0.8	0.126	**9.1 ± 0.6**	**11.8 ± 0.9**	**0.016**
4 m^2									
VSR	**10.7 ± 1.3**	**7.3 ± 0.9**	**0.036**	**17.8 ± 1.2**	**11.1 ± 1.2**	**0.004**	**12.2 ± 0.8**	**8.8 ± 0.6**	**0.001**
BLSR	0.4 ± 0.2	0.5 ± 0.2	0.770	2.5 ± 0.6	1.5 ± 0.4	0.272	**6.9 ± 0.8**	**5.6 ± 0.7**	**0.017**
Simp	0.60 ± 0.06	0.63 ± 0.04	0.604	**0.84 ± 0.01**	**0.77 ± 0.03**	**0.038**	**0.83 ± 0.01**	**0.63 ± 0.04**	**0.002**
BP	0.55 ± 0.06	0.50 ± 0.04	0.511	0.29 ± 0.02	0.36 ± 0.03	0.081	**0.32 ± 0.02**	**0.54 ± 0.05**	**0.0009**

A significantly higher proportion of open habitat species were found in unplanted 100 m^2 plots than in planted plots in all three habitat groups (Figure 8a), according to paired Wilcoxon signed rank tests. Accordingly, a significantly higher proportion of species commonly occurring in both open and wooded habitats were found in planted sites. In 100 m^2 plots in peatlands and improved grasslands, a higher percentage of vascular plant species had competitor strategies in planted than in unplanted sites (Figure 8b). Plants with ruderal strategies comprised a higher proportion of the species in unplanted improved and wet grasslands than in planted sites. Stress-tolerators were proportionately more abundant in unplanted than in planted peatlands (Figure 8b), as were species preferring wet conditions (34% and 24%, respectively). Acidophilic plants made up 12% of the flora of planted improved grasslands: significantly more than the 4% of species in unplanted improved grasslands.

In the 4 m^2 plots, similar significant trends were found whereby open habitat species formed a greater percentage of the flora in unplanted sites, and species frequently found in both open and wooded habitats making up a greater proportion of the plants in planted sites. In peatland and wet grassland 4 m^2 plots, a significantly lower proportion of species in planted sites preferred wet habitats than was the case in unplanted sites (Figure 9). In peatlands, this is balanced by a higher percentage of species with affinities to damp and mesic conditions in planted sites compared with unplanted sites. As in 100 m^2 plots, planted improved grassland 4 m^2 plots had a significantly higher proportion of acidophilic species and a lower proportion of ruderal species than unplanted sites. Competitor species comprised a significantly higher percentage of the vascular flora of planted 4 m^2 peatland plots (35%) than of unplanted plots (24%).

4.3.4.2 Species composition

A two-dimensional NMS ordination (stress = 16.63, p = 0.005) of the 4 m^2 plot abundance data (with planted trees omitted), showed that species composition and abundances differed among planted and unplanted sites in the same habitat group (Figure 10). Ordinations of DAFOR data from grassland and peatland habitats represented in both paired sites and of

100 m^2 presence/absence data showed similar patterns. Improved grasslands showed the greatest shift in ordination space, reflecting substantial decreases in abundance of *Lolium perenne* and increases in competitive grasses, such as *Agrostis stolonifera* and *Elytrigia repens* (Figure 10). The position of planted wet grassland plots relative to unplanted plots from the same site pairs varied, depending on wet grassland type. In some planted wet grasslands, *Agrostis stolonifera* was much more abundant than in the corresponding unplanted sites (Figure 10). In peatland sites, the species with the greatest increase in planted sites was *Molinia caerulea*, which becomes dominant in many planted peatlands. Differences in vegetation composition and structure can also be seen in photographs taken in most site pairs that are available in the GIS database.

According to MRBPs of the 4 m^2 plot abundance data, there were significant differences in species composition and abundance between planted and unplanted sites within each of the three habitat groups. These differences were large in improved grasslands (A = 0.383, p = 0.003) and peatlands (A = 0.332, p = 0.003); the difference between planted and unplanted wet grasslands was not as large, but was nevertheless significant (A = 0.088, p = 0.006). MRBPs of the 100 m^2 plot presence/absence data also detected significant differences between planted and unplanted sites within the habitat groups. However, the differences were smaller than those found using abundance data, especially in peatlands (IG: A = 0.159, p = 0.003; PL: A = 0.083, p = 0.005; WG: A = 0.069, p = 0.005). Hedgerows, treelines and streams associated with hedgerows or treelines did not differ in composition between planted and unplanted sites, according to a MRBP (A = 0.007, p = 0.060).

4.4 DISCUSSION

4.4.1 Pre-afforestation site assessment

Previous BIOFOREST work has concluded that current strategic and site-based biodiversity assessments prior to afforestation are insufficient to prevent high biodiversity sites from being afforested (Gittings *et al*. 2004). Current barriers to effective on-site assessment of biodiversity include inadequacy of current habitat recording guidelines, insufficient baseline information on the biodiversity and distribution of some plant community types and the lack of methodologies to allow non-ecologists to evaluate site biodiversity.

The *Forest Biodiversity Guidelines* (Forest Service 2000c) stipulate that habitats should be mapped in afforestation grant applications and the areas set aside as retained habitats as part of the Area for Biodiversity Enhancement (ABE) should be marked and described. However, the habitats discussed in the *Guidelines* do not follow any published or commonly used habitat classification scheme, and no such scheme is recommended for usage. Gittings *et al.* (2004) recommend that habitat mapping should follow the Heritage Council's Irish habitat classification (Fossitt 2000), and we support this recommendation. Further guidance in habitat mapping is forthcoming in the form of additional guidelines from the Heritage Council (in prep). Habitat mapping is a first step towards biodiversity assessment and used alone may permit some high biodiversity sites to be identified and excluded from afforestation. However, in order for foresters to use this scheme when conducting their pre-afforestation surveys, they must be trained in habitat mapping and the Irish habitats classification scheme. Even with training, habitat classification and mapping is not always easy or foolproof, as British studies have found (Cherrill & McClean 1999; Stevens *et al.* 2004). A common cause of difficulties in habitat mapping includes the situation where a given habitat is in reality a mosaic of habitat types or is transitional between two (or more) habitat types. These problems should be addressed in training, and provision made for mapping mosaics and transitional habitats. The recording of additional or supplementary habitats, such as drains, hedgerows, pockets of scrub or woodland, stone walls or ruins and streams should not be neglected, as these habitats can support many additional species beyond those encountered in the "main" grassland or peatland habitat.

The broad categorisation used by the Irish habitat classification (Fossitt 2000) facilitates use by non-ecologists when compared with more finely divided classification schemes as the British NVC (e.g. Rodwell 1992) and Braun-Blanquet phytosociological classifications (White & Doyle 1982). This leads to a new problem, however, the combination within one habitat type of more than one subtype that may differ in biodiversity. In this study, wet grasslands (GS4) are the clearest example of a habitat type that includes distinguishable subtypes of high and low biodiversity value. Irish wet grasslands encompass sites in a wide ecological range of soil and climate conditions, species assemblages and management regimes (Section 2.3.3.4). Work by O'Sullivan (1965; 1968b; 1976; 1982) has clarified some of these distinctions, but his work is now out of date, and the abundance, distributions and conservation value of different wet grassland types are poorly known. Similar criticisms can be made of other Irish habitats described by Fossitt (2000), such as the amalgamation of dry circumneutral with dry calcareous semi-natural grasslands in the habitat type GS1. In contrast, the ecology, distribution and conservation value of peatland types is better understood, although distribution of wet heaths, which frequently occur in mosaics with other habitat types, is a notable exception. *We recommend that a national survey and classification of Irish grasslands be undertaken as a matter of priority.* Such a survey will benefit many other kinds of land-use planning in addition to afforestation. In the interim before the Irish habitat classification of grasslands is revised, development of a custom classification by the Forest Service may be necessary to facilitate biodiversity assessment prior to afforestation, as has been previously recommended by BIOFOREST (Gittings *et al*. 2004).

From the above discussion, it is clear that simply mapping GS4 habitats will not adequately assess afforestation site biodiversity, and therefore additional assessment methods are required. For this reason we have developed a number of compositional, structural and functional indicators of biodiversity that can be used to help determine whether a given site is potentially of high biodiversity interest and should not be afforested or has little value for biodiversity. It is important to note that the use of these indicators is not a substitute for a thorough survey and assessment by a trained ecologist and that erroneous conclusions about a site's conservation status are possible. However, they should be useful for initial evaluations by non-specialists, and additional help can be sought for borderline or unusual sites. One important consideration is the landscape context of the site. In an intensively agricultural landscape, a particular semi-natural habitat may have significant biodiversity value, whereas the same habitat may not be of much importance in less-intensive landscapes where similar habitats of equal or better quality are common.

The scale at which biodiversity and biodiversity indicators are assessed is important. This is demonstrated by the variation among sampling scales (habitat, 100 m^2 plot and 4 m^2 plot) in classifications of the same sampling unit. In peatlands, biodiversity should be primarily be assessed at the scale of the site, focusing on the confirmed functional indicators discussed in Section 4.3.3.1.3. Sites containing well-developed flushes, swallow holes, pools or other hydrological features should not be afforested. Similarly, intact bogs with little or no peat cutting should not be afforested. The provisional compositional indicators described in Section 4.3.3.2.2 should be looked for during the pre-afforestation site survey and habitat mapping. These indicator species should occur frequently in order to count as present. For example, if only two plants of *Drosera rotundifolia* were found in an entire site, then *Drosera rotundifolia* as a compositional indicator of biodiversity should not be counted. We suggest that during assessment a proposed afforestation site should be divided into imaginary 100 m^2 grid squares. A given compositional indicator should be found in many or most of these imaginary squares if it is to be counted as present at that site. If a more detailed investigation is required by an ecologist, he/she should record and compile species lists for additional habitats and features present, such as flushes and pools, and record species in the main habitat(s) in 100 m^2 plots. Plot recording should estimate plant abundances, either using the DAFOR scale or, preferably, recording cover to the nearest 10%, with additional cover categories for species with cover < 10%. In our experience, attempting to record cover more precisely in plots of this size would not yield estimates accurate enough to warrant the effort.

The scale at which biodiversity indicators should be assessed is smaller in grasslands, particularly for compositional indicators. The habitat-scale clustering of all wet grasslands into one group shows that wet grassland types cannot be adequately or efficiently distinguished by the use of a coarse recording scale such as DAFOR over large areas. This is probably because many wet grassland species can occur at low abundances somewhere within a site, thus increasing similarities among grassland types that may differ primarily in species relative abundance or evenness. Compositional and structural indicators (Sections 4.3.3.1.2 and 4.3.3.2.1) should be investigated at 100 m^2 or smaller scales. As with peatlands, we do not recommend that actual plots be used to search for and evaluate indicator species or structural characteristics; indicators frequently observed (using the above imaginary 100 m^2 grid guideline) during a walkover survey should count as present. If more detailed investigations are required, an ecologist should record percentage cover of species (to the nearest 5%, with smaller categories for species with cover < 5%) in a number of smaller plots, ranging from 4 m^2 to 25 m^2 depending on the fine-scale heterogeneity of the site. The plots should be placed to ensure that variation within the site, especially among fields, is adequately represented.

Whereas the confirmed indicators of biodiversity appear to be reasonably faithful and constant to sites of potentially high biodiversity value, further assessment of the provisional indicators is required. For example, *Centaurea nigra* and *Lathyrus pratensis* occur in other semi-natural grassland types in addition to wet grasslands, and may in fact be more characteristic of other grassland types (Fossitt 2000; O'Sullivan 1982). Others, for example *Iris pseudacorus*, may occur in certain semi-improved or improved grasslands and may not have the power to adequately distinguish between high and low biodiversity habitats. Biodiversity indicators for plant communities and other taxonomic groups are summarised in Section 8.3, where further guidance on their use is also provided. While we have gone some way to suggesting methods for biodiversity assessment prior to afforestation, further development and testing of methods usable by non-specialists is required. The British Forestry Commission provide a methodology for determining if a site is improved grassland and therefore eligible for the Better Land Supplement (Forestry Authority 1997). It is a useful method for identifying improved grassland, but not for distinguishing, for example, between semi-improved and unimproved wet grasslands; nevertheless, it may be a useful model on which a more comprehensive assessment protocol can be based.

4.4.2 Initial effects of afforestation

The substantial differences in species richness, relative abundance and composition observed between planted and unplanted sites were primarily due to relaxation of grazing, changes in nutrient management and drainage for afforestation. The effects of release from grazing were most dramatic in improved grasslands and included increases in vigorous, competitive grasses with corresponding decreases in less competitive ruderal species. Similar changes occurred in wet grasslands, but were less marked because of generally lower pre-afforestation grazing pressures. Elimination of grazing may also be partly responsible for decreases in the proportion of open habitat species and increases in species frequently found in both wooded and unwooded habitats. Ruderal species generally have open habitat affinities, and so their decrease from greater competition may be responsible for the observed shift in woodland preferences. The increase in wooded/unwooded habitat species may also be partly the result of colonisation from hedgerows, although plots were generally located away from hedgerows. In the field, we observed encroachment of hedgerow species, especially bramble and blackthorn, at the edges of hedgerows in planted sites, and these species were sometimes found to have regenerated by seed in the centre of afforested fields. Another possible source for wooded/unwooded habitat species, particularly bryophytes, is direct importation on nursery stock or tools or footwear used by the establishment foresters. In improved grasslands, the increase in acidophilic species may be partially due to decreases in fertiliser and liming applications, as such habitats are frequently not fertilised prior to afforestation. Differences in the proportion of acidophilic species may also reflect pre-afforestation differences between the planted and unplanted sites in soil characteristics.

Cessation of grazing by livestock may be partially responsible for the changes observed in peatlands, such as the large increase in abundance of *Molinia caerulea*. However, it appears that drainage and fertilisation of peatlands are more important factors. In addition to changes in hydrology, drainage promotes mineralisation of drier peat at the surface, increasing availability of nutrients, including N. It is standard practice to fertilise peatlands with phosphate on afforestation, as had been done at all sites for which we had such information. Beltman *et al.* (1996) have found that plant growth in Irish blanket bogs is limited by P or N+P, and other studies in bogs and wet heath in Ireland and the Netherlands have found that *Molinia* responds vigorously to increases in one or both of these nutrients (Roem *et al*. 2002; Tomassen *et al*. 2004). Drainage and fertilisation contribute to the decrease in stress-tolerant species in peatlands and species preferring wet conditions in peatlands and wet grasslands, and also to the increases in competitor species. Forestry drains were responsible for the discrepancies between plot scales in species richness differences between planted and unplanted peatlands and wet grasslands. Forestry drains were included in 100 m^2 plots, but not 4 m^2 plots and provided new, reduced-competition habitats for bryophytes and certain vascular plant species such as ferns and small sedges. This accounts for increases or lack of decreases in species richness at the 100 m^2 scale while species richness at the 4 m^2 decreases due to greater competitive pressures. A similar phenomenon was not observed in improved grassland sites because they were frequently not drained.

The MRBP test results for 4 m^2 and 100 m^2 plot data show that species relative abundances are more strongly affected than the presence/absence of species, at least at a larger scale. Although there have been substantial changes, the habitats have not radically changed to a completely different type. Sitka spruce saplings are still small, and the habitats are still grasslands and peatlands rather than woodlands. The trees have not yet begun to affect surrounding hedgerows and treelines. When the trees become more mature and form a closed canopy, the understorey vegetation will be almost completely eliminated if it is a pure Sitka spruce stand (Smith *et al*. 2005). The biodiversity benefit provided by forestry drains will then be largely eliminated, and side-shading will begin to affect the flora of adjoining hedgerows, treelines, stone walls and other marginal habitats if a sufficient setback has not been established around these features. Several sites were planted with a mixture of Sitka spruce and larch or lodgepole pine. Under a mixed canopy, some elements of the open habitat flora may survive, but research on mixed forests is needed before conclusions can be drawn.

4.5 CONCLUSIONS

We have found that, of the plant communities we surveyed, wet grasslands and peatlands generally support greater biodiversity than improved grasslands. Within wet grasslands, we have identified several subtypes, whose definition is dependent to some extent on the scale of the analysis. Wet grasslands can be divided in oligotrophic, base-rich and semi-improved subtypes, with some individual sites standing out as having distinctive plant communities. In our study, more or less undisturbed lowland blanket bog was highlighted as being the peatland type of greatest biodiversity interest. The upland blanket bogs and wet heaths we surveyed were more disturbed by turf cutting and overgrazing. Flush and stream habitats can substantially add to the biodiversity of peatland sites. Marginal and supplementary habitats, such as hedgerows, streams, scrub, stone walls and earth banks, can also increase the biodiversity of grassland sites by supporting species that otherwise would not persist in the grassland matrix.

Because of the high biodiversity value of undisturbed peatlands, we recommend that peatlands experiencing little or no turf cutting or including hydrological features such as flushes, pools and streams not be afforested. Additional confirmed and provisional indicators of biodiversity were developed and guidelines for their use were discussed. These indicators, however, are biased towards lowland blanket bog and may not be adequate to identify high quality wet heaths and upland blanket bog. We have also identified confirmed and provisional indicators of biodiversity for grassland communities, which will aid in distinguishing semi-natural wet grasslands from improved grasslands and semi-improved wet and dry grasslands. The biodiversity value of semi-natural habitats, especially grassland communities, is dependent on landscape context: a particular grassland may be of significant biodiversity interest in intensive agricultural landscapes, but of less value in landscapes where similar semi-natural grasslands are abundant. In order to elucidate this context, we recommend that a national survey and classification of Irish grasslands be carried out as a matter of urgency.

The initial effects of afforestation are largely the result of three factors: exclusion of grazing livestock, forestry drainage and changes in nutrient management. In general, ruderal and stress-tolerant species decline under competitive pressure from vigorous grasses and other competitor species. Forestry drains may provide a temporary habitat for less competitive species, but the overall of effect of drainage is to reduce the diversity of species dependent on wet conditions. The initial effect of afforestation on plant communities is to change the relative abundances of species, rather than causing a radical shift in species compositions. This shift will occur in crops of densely shading tree species when the forest canopy closes over.

Comparing the biodiversity of the plant communities we have surveyed with the biodiversity of conifer plantations investigated by earlier BIOFOREST work, we conclude that afforestation will have a detrimental effect on semi-natural habitats. Such habitats, including peatlands and semi-natural wet grasslands, should not be afforested, unless similar habitats are abundant in the landscape. Even then, the relevant authorities must take care that these habitats are not gradually eliminated by piecemeal afforestation and agricultural intensification. On the other hand, the effect of afforestation on improved and semi-improved grasslands will be neutral or positive, particularly in landscapes that contain little semi-natural woodland habitat.

5 HOVERFLIES

5.1 INTRODUCTION

The concept of Sustainable Forest Management (SFM) has been developed to ensure that forests are managed to maintain their biodiversity, productivity, regeneration capacity, and vitality. In Ireland, the SFM principles are being implemented in the development of both public and private-sector forestry. One of the six pan-European criteria for SFM adopted as part of the Irish National Forest Standard (Forest Service 2000e) is "Maintenance, conservation and appropriate enhancement of biological diversity in forest ecosystems." Adoption of this criterion requires consideration of the impact of afforestation on existing biodiversity.

The target rate for afforestation in Ireland over the next 25 years is 20,000 ha per year and aims to change the country's forest cover from its current level of approximately 10% to 30% (Department of Agriculture Food and Forestry 1996). Most future afforestation will be undertaken by the private sector (mainly farmers) under the Afforestation Grant and Premium Scheme (Forest Service 2000a). A recent study (Kearney 2001) suggests that, in north-western Ireland, afforestation grant schemes are primarily taken up by full-time farmers planting on peatland or on lands previously used for rough grazing. However, these are also lands which may be of significant conservation value – a value that would be compromised by commercial afforestation. Therefore, in accordance with the SFM principles, the importance of assessing the effects of afforestation on biodiversity takes on a new significance. Current legislation and procedures impose controls on afforestation within, or near to, sites with formal nature conservation designations, e.g. Natural Heritage Areas, Special Areas of Conservation and Special Protection Areas (Forest Service 2003). It is, however, widely acknowledged that there are non-designated sites of biodiversity importance: existing nature conservation designations do not include all sites of national importance (e.g. Dwyer 2000; Irish Peatland Conservation Council 2001) and are not intended to cover sites of regional or local importance.

In order to assess the effects of afforestation on the biodiversity of a particular habitat, it is necessary to first establish the biodiversity value of the habitat before afforestation, and then to consider the changes in biodiversity that result from afforestation. In the initial period after afforestation, before the canopy closes, the habitat will remain open. However, changes in the character of the habitat will occur. Depending on the nature of the pre-afforestation habitat, these changes may include: changes in the species composition and structure of the ground vegetation due to cessation of grazing, reduction or loss of wet habitat features due to drainage for forestry, and invasion by scrub. All these changes, along with the planting of the trees, may have effects on the biodiversity of the habitat.

In order to meet the requirements of SFM, information on the biodiversity of habitats that are typically used for afforestation in Ireland and on the effects of afforestation on these habitats is required. However, there have been no detailed studies of these topics. Therefore, 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 biodiversity value of habitats that are typically used for afforestation in Ireland.
2. To identify indicators of biodiversity for these habitats.
3. To examine the changes in biodiversity that occur in these habitats in the initial period following afforestation.

Hoverfly nomenclature follows Speight et al. (2004) and botanical nomenclature follows Stace (1997).

5.2 METHODS

5.2.1 Study Design

From examination of data on applications for afforestation grants (Forest Service, unpublished data), we identified three broad habitat types that are among those typically used for afforestation in Ireland: peatlands, improved grassland and wet grassland (see Table 11). For each of these three broad habitat types, we selected eight pairs of sites. Each pair included one five year old Sitka spruce (*Picea sitchensis*) plantation that had been planted on the relevant habitat type, and a nearby (usually within 1 km, maximum distance 5 km) unplanted site of the same habitat type. We selected sites by first identifying candidate sites from GIS data, and then ground-truthing the sites. The criteria that we used to identify suitable sites were that they were representative examples of the relevant habitat type and that the planted and unplanted sites were closely matched in terms of relevant environmental conditions such as soil type, drainage, slope, etc. We ground-truthed nearly 100 sites, of which we selected 24 for our research. In addition to these paired sites, we also surveyed an additional three unplanted sites (one improved grassland and two wet grassland). The habitat characteristics of all the unplanted sites are summarised in Table 11.

Table 25. Important open habitat components in the unplanted sites[1].

Site type	Habitat type (Fossitt 2000)	Habitat type (Speight *et al*. 2004)	Sites[1]
Peatland	Upland blanket bog (PB2)	Blanket bog (632)	Caru[2], Tiep[2] and Togu
	Lowland blanket bog (PB3)	Blanket bog (632)	Geau[2], Toou[2] and Veeu
	Wet heath (HH3)	Moorland (24)	Geau, Incu and Sliu
	Poor fen and flush (PF1)	Acid fen (612)	Geau and Incu
Improved grassland	Improved agricultural grassland (GA1)	Lowland improved grassland (2321)	All sites
	Improved agricultural grassland (GA1)	Intensive grassland (233)	Bght, Gary, Kill
	Wet grassland (GS4)	Humid eutrophic/mesotrophic (231131)	Agho, Balu
Wet grassland	Wet grassland (GS4)	Humid eutrophic/mesotrophic (231131)	All sites
	Wet grassland (GS4)	Humid oligotrophic (231132)	Bool, Knaw, and Mull
	Marsh (GM1)	Marsh	Mull

[1] note that there are discrepancies between the habitats recorded here and those recorded by the vegetation survey (see Section 4), due to differences in the areas surveyed.
note that sites can be listed under more than one category.
[2] sites with extensively cutover bog (PB4 in Fossitt 2000; 633 in Speight 2004).

We surveyed eight pairs of sites (four peatland and two each of improved and wet grassland) and the three unpaired sites in 2002. We surveyed the remaining sixteen pairs of sites in 2004. In the unplanted component of one of the paired improved grassland sites, cattle destroyed our Malaise traps. Therefore, this pair is excluded from our analyses.

5.2.2 Hoverfly sampling

We used Malaise traps to sample hoverflies. In each site we installed two Malaise traps. Following the standard sampling procedure recommended by Speight (2000), we placed the two traps close to each other (c. 10 m apart). In improved grassland and wet grassland sites, the traps were placed next to a hedgerow or treeline. In peatland sites, the traps were placed next to a linear surface water feature (i.e., a flush, brook or drainage ditch). Where possible, we selected a linear feature running east-west, so that the traps could be orientated with their collection bottles at their southern end. Where the linear feature was orientated north-south, we placed the traps with their collection bottles facing east. Before installing the traps in each pair of sites we closely examined both the planted and unplanted sites. We selected trap locations that were comparable between each of the pair, in terms of: the type and structure of the linear feature and the orientation of the linear feature. Where farm livestock were present, we used temporary electric fencing to protect the traps.

The Malaise traps were operated continuously from early/mid May to early/mid July (improved grassland and wet grassland sites) and to late August (peatland sites). The contents of the traps were collected approximately every three weeks. The early May to mid July sampling period covers the main activity period of most univoltine (one generation per year) species and the early generations of most polyvoltine (more than one generation per year) species. We decided to extend the trapping periods in the peatland sites because of the low catches in many of the traps. In joint analyses of the peatland sites with the other sites we only use data from the peatland sites for the first three sampling periods (i.e., comparable with the sampling periods for the other sites). All hoverflies caught in the Malaise traps were identified to species.

At two sites (one improved grassland and one wet grassland), damage to one of the Malaise traps resulted in the loss of the sample for one of the sampling periods. We extended the sampling period at these sites, and their pairs, for an extra three weeks. For pairwise comparisons of planted and unplanted sites that involved these sites, we substituted the data from the additional sampling period for the data from the period with the missing sample.

5.2.3 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 supplementary habitat types, in a 100 m radius around each malaise trap.

We recorded habitat structure in a 100 m radius around each malaise trap, using the categories defined in Table 26. 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 data analyses, these cover values were given values from 1 (rare) to 5 (dominant).

Table 26. 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 jacobea, 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).

We recorded cover of broadleaved and conifer tree and shrub vegetation separately. In planted sites, we also recorded cover of planted and naturally regenerated broadleaved tree and shrub vegetation separately, as well as their combined cover.

We recorded the vegetation structure parameters separately for the open (e.g., grassland) and woody habitat (e.g., hedges) components. We also recorded the frequency of the vegetation structure parameters separately for each discrete length of hedge or treeline and for each discrete patch of scrub. To obtain overall frequency values for these parameters for each site, we calculated the average frequency weighted by hedge/treeline length and scrub area. To do this, we converted scrub area to equivalent hedge length by dividing scrub area, in square metres, by five: i.e., assuming a hedge width of 5 m. We also recorded the width and cross-section profile of each discrete length of hedge or treeline, using the categories defined by Clements and Tofts (1992; see also Rich et al.2000) but with the addition of a category for hedge width greater than 5 m (score 5). We then obtained overall width and cross-section values for each site by the same weighted averaging method, with scrub being assigned a width score of 5 and a cross-section score of 4.

In the unplanted improved and wet grassland sites we recorded grazing intensity, based on field observations and information provided by landowner. We used the following scale: 1 = sward height remains greater than 10 cm over most of the site throughout the year; 2 = sward height is seasonally reduced below 10 cm (e.g., aftermath grazing of hay meadows); 3 = sward height is regularly reduced below 10 cm throughout the year (e.g., rotationally grazed pastures). Sites which would have been classified in category 3, but which had intensively managed silage fields, were classified in category 3.

5.2.4 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). This database codes the degree of association of each species with each macrohabitat and microhabitat on a scale from zero (no association) to three (maximally preferred). 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. In this report, we describe species that are coded two or three for a particular habitat, as *typical* of that habitat, and species that are coded one for a particular habitat as *supplementary* for that habitat.

We defined species groups for a number of purposes, including broad groupings for analyses of assemblage structure, habitat-specific groups for analysing responses to habitat features, and species groups of particular conservation interest for identifying the most important sites and features for hoverfly biodiversity.

Our primary classification divided the recorded species into two groups, based on their predicted association with open space compared to tree and shrub dominated macrohabitats: open habitat-associated species and woody vegetation species. We defined open habitat-associated species as those that are coded to occur in the main open space macrohabitats that were present in the sites that we studied: unimproved humid grassland (23113), lowland improved grassland (2321), moor (24), intensive grassland (233), acid fen (612), blanket bog (632) and cutover bog (633). For our category of woody vegetation species, we first selected all the species that are coded to occur in the main woody vegetation macrohabitats that were present in the sites that we studied: atlantic thickets (122), *Salix* swamp (143), *Abies*/*Larix*/*Picea* plantations (saplings) (1812), scattered trees in open ground (19) and hedge (53). We then excluded species that are only supplementary for any of these habitats, and that are coded for the open space macrohabitats, because the primary associations of these species are generally with open space macrohabitats. We made one exception, *Cheilosia albipila*, because most Irish records of this species are from woodland habitats (Speight 2000a). We also excluded typical *Salix* swamp species, that are not typical of any of the other woody vegetation macrohabitats and that are coded for the open space macrohabitats; in our sites, *Salix* swamp was of very limited extent and these species would be more likely to be associated with the open space macrohabitats.

The habitat-specific species groups that we defined included species groups associated with the main macrohabitats that we studied (see above). We also defined two broader groups: surface water associated species and tree/tall shrub associated species. 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 defined tree/tall shrub associated species, from the microhabitats spreadsheet of the Syrph The Net database (Speight *et al*. 2004), as those that are coded for trees (gen.) category. However, we excluded species that are only coded for the low shrubs sub-category as these included species associated with dwarf shrubs such as *Calluna vulgaris*, and, in our sites, dwarf shrubs are more associated with the open habitats.

The species groups of particular conservation interest included anthropophobic (Boycott 1934; Speight & Castella 2001) species, wetland specialists, wet grassland specialists and scrub specialists. Our definitions of these categories are based upon the typical distribution of relevant habitat features in the Irish landscape. Anthropophobic species are dependent upon semi-natural habitats and will not persist in intensively farmed landscapes. We defined this category as including all species, except those predicted to occur in heavily-grazed improved grassland (23212), intensive grassland (233), and cultural macrohabitats (5) apart from orchards (54) and urban parks (55). In contrast to Speight and Castella (2001), 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. For our category of wetland specialists, we first selected species that are coded to occur in freshwater macrohabitats, but are not coded as typical of unimproved humid grassland (23113), lowland improved grassland (2321), moor (24), intensive grassland (233), blanket bog (632) and cutover bog (633). We then excluded from this group, species that are characteristic of anthropophilic habitats, based on the descriptions in Speight (2000a). The remaining wetland specialist species are generally associated with ground water fed supplementary habitats, such as brooks and flushes, or acid fen, and are, therefore, likely to be of localised occurrence, even in landscapes with extensive areas of semi-natural habitat. We defined wet grassland specialists as species that are typical of humid grassland but are not typical of lowland improved grassland. We defined scrub specialists as species included in the woody vegetation group that are not coded for hedge or hedge plus field margin (58). In intensively farmed landscapes, species in these two groups are likely to be of localised occurrence.

5.2.5 Data analysis

5.2.5.1 General methods

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

5.2.5.2 Biodiversity assessment of the habitat types

We used multivariate methods to examine the assemblages of the three habitat types, compared species richness of selected species groups between the habitat types with oneway Analysis of Variance (ANOVA) and Tukey's post-hoc tests and we examined the representation of characteristic species groups in the habitat types.

We examined the structure of open habitat-associated and woody vegetation separately, and carried out these analyses on the unplanted site datasets and on the planted and unplanted sites datasets. We did not examine woody vegetation associated hoverfly assemblages in peatland sites, because the unplanted peatland sites did not contain any significant components of woody vegetation habitats. We used global non-metric multidimensional scaling analysis (NMS), flexible-beta cluster analysis (with beta set at –0.25) and indicator species analysis (Dufrêne & Legendre 1997) on log (x+1) transformed abundance data. For the NMS and cluster analyses we used Sørensen (also known as Bray & Curtis) distance measures. The parameter set-up that we used for the NMS analyses is shown in Table 27. Where the NMS analysis produced a solution with more than two axes, the axes that explain the highest percent of variance in the distance matrix were used for graphical representation of the results. We examined the correlations of potentially relevant environmental variables with the ordination axes.

Table 27. 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

The ordination analyses showed a clear separation of the peatland sites from the improved and wet grassland sites, but did not show clear separation between the latter two habitat types. Therefore, for most of the further analyses (see Sections 5.2.5.3 and 5.2.5.4), we carried out separate analyses on the peatland sites and on the improved and wet grassland sites. The ordination analyses also identified three outlying improved and wet grassland sites (Coog, Kilb and Kill). These all had very low total catches of hoverflies compared to the other improved and wet grassland sites. Therefore, as the relationship between hoverfly fauna recorded in these sites and that recorded in the other improved and wet grassland sites may have been more influenced by the low number of hoverflies sampled (due to trapping conditions) rather than underlying habitat characteristics, we excluded these sites from further analyses of the improved and wet grassland sites. However, we included these sites in analyses comparing improved and wet grassland sites with peatland sites, as several of the peatland sites also had similarly low catches.

5.2.5.3 Identification of biodiversity indicators

We used a hypothesis testing approach to identify biodiversity indicators, by predicting possible functional relationships between the species richness of various species groups and relevant habitat parameters, and testing these relationships with Pearsons correlations (or Spearmans correlations for non parametric data). We used one-tailed tests because we were predicting specific responses. We examined three separate habitat components and their hoverfly fauna: the ground vegetation structure of the main habitat matrix (bog or grassland), the wet habitat features and the woody vegetation.

For the ground vegetation component, we predicted that grazing intensity would have a negative effect on the numbers of wet grassland specialists and grazing-sensitive species in improved and wet grassland sites. Therefore, we examined correlations between the species richness of these groups and the frequency of tall herb and tussock vegetation. We also compared the species richness of these groups between the different categories of grazing intensity using t-tests: we combined grazing categories 1 and 2 because of the small sample sizes in each of these groups.

For the wet habitat component, we predicted that the presence of wet habitat features would have a positive effect on the numbers of freshwater, wetland specialist and wet grassland specialist species. Therefore, we examined correlations between the species richness of these groups and the area of flushes (including acid fen), the length of brooks, the frequency of submerged sediment/debris, water-saturated ground and temporary pools, and with axes derived from NMS ordination of the wet habitat parameters.

For the woody vegetation component, we predicted that the amount and structure of such vegetation would have a positive effect on the numbers of woody vegetation, tree-tall shrub and scrub specialist species. We first investigated the relationships between the species richness of these groups and an index of woody vegetation cover. Where significant relationships occurred, we used the residuals from regression of the species richness with the index, to investigate correlations with the frequency of woody vegetation structure parameters and with hedge/treeline width and cross-section. We calculated the index of broadleaved woody vegetation cover as follows: I_u = hedge length in metres + area of scrub in square metres/5 (see Section 5.2.3).

5.2.5.4 *Comparison of planted and unplanted sites*

We used NMS ordinations to compare the assemblage structure of planted and unplanted sites (see Section 5.2.5.2). We used paired t-tests to compare species richness of planted and unplanted peatland sites. For the same comparison within the improved and wet grassland sites, we used a mixed-model ANOVA with HABITAT and PLANTED as fixed factors in a fully factorial model, and with the random factor SITE nested within HABITAT.

Where we had found significant relationships between species richness and habitat parameters in the unplanted sites, we used paired t-tests to examine whether changes in these habitat parameters between the planted and unplanted sites affected the species richness of the relevant species groups. To do this comparison for the woody vegetation habitat component, we first had to derive an index of broadleaved woody vegetation in the planted sites. Derivation of the index was more complex, compared to the unplanted sites, because non-hedge broadleaved woody vegetation cover did not tend to occur in discrete patches. To obtain the index (I_p) for these sites, we first derived an overall weighted cover for non-hedge broadleaved woody vegetation (BL) as: BL = (understorey frequency x 3) + (tall shrub frequency x 2) + (low shrub frequency). We then carried out separate multiple regressions of woody vegetation and tree and shrub associated hoverfly species richness on hedge length (HL) and BL. The ratio of the unstandardised Beta coefficients for BL and HL was used to convert BL cover values into hedge length equivalents. These hedge length equivalents were then summed with HL to give separate I_p values for the woody vegetation and tree and shrub associated hoverfly species groups.

We also examined whether the growth stage of the planted conifers contributed to the maintenance of woody vegetation hoverfly biodiversity. We derived an overall weighted cover of conifers (C), using the same method as for BL above. We then examined correlations of C with woody vegetation and tree and shrub associated hoverfly species richness, and with the residuals from the regression of woody vegetation and tree and shrub associated hoverfly species richness with I_p.

To investigate changes in species composition between planted and unplanted sites, we considered that it was necessary to carry out paired (by site) analyses on abundance data. The proximity of the planted and unplanted sites in most of the site pairs meant that presence-absence data would be confounded by movements of adults between the sites. Differences in species occurrences and abundances across site pairs meant that a method that factors out the variation between site pairs was required. It was also necessary to take account of possible differences in trapping efficiency between sites within a pair. We did not consider use of proportional abundances as appropriate because that method would mean that values for most species would be heavily influenced by abundances of a few common species. Instead, we used the standardised residuals from pairwise linear regressions between log (x + 0.5) species abundances units in the planted and unplanted component of each site pair (an adaptation of the method of Telfer *et al.* 2002). These regressions used the species abundances in unplanted sites as the independent variables and species abundances in the planted sites as the dependent variables, with species as the sample. In this way, we carried out a regression for each pair of sites. Taking the residuals from these regressions, allow us to use the overall relationship between abundances across all species within a site pair to correct for possible differences in trapping efficiency. For each species that occurred in more than five site pairs, we then calculated the mean of the standardised residuals across all the site pairs to give an index of relative abundance (IA). This index is positive if the species tends to be more abundant in planted sites and negative if the species is more abundant in unplanted sites. For each species, we then carried out a one-sample t-test to determine if IA was significantly different from zero. Because of the large number of t-tests carried out we calculated the Bonferroni-adjusted p-value to identify robust significant values. However, we also examined all species with t-tests that were significant at $p < 0.05$ to see whether the relationship was meaningful in terms of the ecological characteristics of the species involved.

5.3 RESULTS

5.3.1 Hoverfly fauna

We recorded a total of 98 species (see 192), of which 63 are associated with open habitats and 50 are associated with woody vegetation habitats. Five of the species that we recorded are not associated with any of the habitats included in our open habitat and woody vegetation categories: *Cheilosia variabilis*, *Didea fasciata*, *Parasyrphus vittiger*, *Xanthandrus comtus*, and *Xylota jakutorum*. We recorded four species that are considered to be threatened or probably threatened and another five species that are considered to be decreasing (Speight 2000; Speight *et al.* 2004, see Table 28). These species are referred to hereafter as *rare* species.

Table 28. Threatened and decreasing hoverflies recorded.

	Threat status[1]	Sites recorded from	Likely habitat source[3]
Anasimyia lunulata	Decreasing	Cast Veep	Unknown Pools in blanket bog
Brachyopa scutellaris	Decreasing	Rauu	*Fraxinus excelsior* treeline[4]
Chrysogaster virescens	Decreasing	Geau	*Salix* swamp
Epistrophe nitidicollis	Threatened[2]	Geau and Kill	*Salix* swamp/conifer plantation
Meligramma guttata	Threatened/Decreasing	Raup	*Salix* swamp/*Fraxinus excelsior* treeline[5]
Orthonevra geniculata	Decreasing	Clar Incu	Brook edge in *Salix* swamp Acid fen
Orthonevra nobilis	Decreasing	Geau	Acid fen
Parasyrphus nigritarsis	Threatened	Balp, Balu, Gore, Kill	*Salix* swamp
Xanthandrus comtus	Threatened	Balb	Unknown

[1] coded 2 or 3 for the relevant category in Speight *et al.* (2004).

[2] not listed as threatened or decreasing by Speight *et al.* (2004), but described as endangered by Speight (2000a).

[3] based on habitat associations described by Speight *et al.* (2004) and the available habitats within the vicinity of the site.

[4] species associated with treelines are not coded by Speight *et al.* (2004), but considered as potential source due to presence of overmature *Fraxinus excelsior* trees, and occurrence of other saproxylic hoverfly species.

[5] species associated with treelines are not coded by Speight *et al.* (2004) but this species is coded for scattered *Fraxinus* trees (1913).

5.3.2 Hoverfly biodiversity in pre-afforestation sites

5.3.2.1 Overall

The ordination of the open hoverfly assemblage in the unplanted sites is shown in Figure 1. The ordination generally differentiates between the three habitats, with the peatland sites being the most distinct.

The number of open habitat-associated, wet grassland specialist and woody vegetation species were significantly higher in wet grassland sites compared to peatland sites. However, peatland sites had the highest numbers of open habitat-associated anthropophobic species. There were no significant differences among habitat types in the numbers of anthropophobic species associated with wooded habitats (Table 29).

Table 29. Comparison of hoverfly species richness in pre-afforestation peatland, improved grassland and wet grassland sites. There is no significant difference between habitat types if designated with the same letter $^{(a \text{ or } b)}$.

Species group		Mean species richness/site (SD)			ANOVA	
		Peatland	Improved grassland	Wet grassland	$F_{2,23}$ =	p
Open habitat-associated species	All	13.1^{a} (46)	16.8^{ab} (5.3)	19.4^{b} (4.3)	3.94	0.034
	Anthropophobic	3.9^{a} (1.8)	1.6^{b} (1.4)	3.2^{ab} (1.1)	5.11	0.015
	Wetland specialists	2.5^{a} (2.3)	2.3^{a} (2.3)	3.7^{a} (2.2)	5.52	0.344
	Wet grassland specialists	2.8 $(1.0)^{a}$	3.6^{ab} (1.8)	4.9^{b} (1.7)	4.17	0.029
Woody vegetation species	All	7.5^{a} (3.4)	11.9^{ab} (4.6)	13.4^{b} (3.4)	5.57	0.011
	Anthropophobic	0.4^{a} (0.7)	0.5^{a} (0.8)	0.7^{a} (0.7)	0.40	0.675
	Scrub specialists	0.6^{a} (1.1)	1.0^{a} (1.3)	0.9^{a} (0.9)	0.26	0.772

The representation of the characteristic hoverfly fauna in the pre-afforestation peatland, improved grassland and wet grassland sites is shown in Table 30. Apart from the blanket bog fauna of peatland sites and the heavily-grazed improved grassland fauna of improved grassland sites, the average representation of the typical fauna was less than 50%. The high representation of the typical blanket bog fauna of peatland sites reflects the fact that there are only three species involved. The average representation of the supplementary fauna was always less than 35%. The average representation of the wetland specialist fauna in peatland sites was 6% (S.D. 6%). The average representation of the wet grassland specialist fauna was 20% (S.D. 10%) in improved grassland sites and 27% (S.D. 9%) in wet grassland sites.

Table 30. Representation of characteristic hoverfly fauna in pre-afforestation sites.

Site type[1]	Habitat association of fauna	Typical fauna[2]			Supplementary fauna[3]		
		No. of Irish species	Mean %[4]	Max %[4]	No. of Irish species	Mean %[4]	Max %[4]
P (n = 8)	Blanket bog	3	79 (25)	100%	17	34 (9)	59%
	Cutover bog	12	47 (13)	67%	26	27 (14)	38%
	Moor	9	38 (16)	56%	29	34 (10)	45%
	Humid grassland	36	30 (9)	42%	36	17 (8)	33%
IG (n = 8)	IG – lightly grazed	17	40 (10)	59%	17	27 (10)	41%
	IG – heavily grazed	9	54 (17)	89%	26	27 (10)	39%
	Humid grassland	33	30 (15)	45%	29	17 (15)	30%
	Hedge	19	28 (30)	47%	36	20 (17)	32%
WG (n = 10)	IG – lightly grazed	17	44 (9)	53%	17	30 (9)	52%
	Humid grassland	33	37 (37)	52%	29	17 (17)	27%
	Hedge	19	32 (11)	53%	36	28 (12)	32%

[1] P = peatland, IG = improved grassland, WG = wet grassland.

[2] coded 2 or 3 for relevant habitat type in Speight *et al.* (2004).

[3] coded 1 for relevant habitat type in Speight *et al.* (2004).

[4] percentage of the Irish fauna recorded per site.

5.3.2.2 *Peatland*

In the overall ordination of open habitat-associated species (Figure 11), the peatland sites sampled in 2004 (Tieu, Caru, Geau and Veeu) form a relatively tight cluster, while the sites sampled in 2002 are separated from the former but are quite widely scattered. A similar pattern is obtained from ordination of the peatland sites only. The total catches of hoverflies in three of the peatland sites (Incu, Togu and Toou) sampled in 2002 were very low: less than 100 across all five trapping periods, compared to more than 200 in all the other sites. Therefore, the wide scatter of the 2002 sites may indicate that the sampling level was not adequate to accurately characterise the hoverfly assemblages of these sites.

There was no relationship between species richness and wet habitat parameters. The highest numbers of wetland specialist species occurred at Geau (11). At all the other sites sampled in 2004 we recorded five or six wetland specialist species. Of the sites sampled in 2002, we recorded six specialist wetland species at Incu, and no more than three at the other sites. Strict wetland specialists (i.e., species not supplementary for any of the other relevant habitats) only occurred at two sites: *Chrysogaster virescens* and *Neoascia meticulosa* at Geau; and *Orthonevra geniculata* at Incu.

5.3.2.3 Improved and wet grassland

The ordination of the open habitat-associated species shows a broad separation between the two habitat types (Figure 11). Cluster analysis of the improved and wet grassland sites separated the sites into one large mixed group of improved and wet grassland sites (group A), another group of three wet grassland sites (group B), and three sites that are outliers. The outlier sites all had very low catches of hoverflies (less than 100), compared to the other sites (over 150), and were excluded from further analyses of the improved and wet grassland sites (see Section 5.2.5). All the indicator species are associated with the small group of wet grassland sites (group B), and are mainly species associated with surface water and/or oligotrophic habitats (Table 31). This group of sites also had significantly higher species richness than the other sites (Table 32). Across all improved and wet grassland sites, there was no relationship between the numbers of freshwater, wetland specialist and wet grassland specialists and wet habitat parameters. Sites with low grazing intensity had significantly higher numbers of grazing sensitive species: a mean of 3.43 (SD 1.5) compared to 1.9 (SD 0.6); one-tailed t-test assuming unequal variances, t = 2.5, p = 0.036, n = 15. The numbers of wet grassland specialists were positively correlated with the frequency of tussocks (r = 0.45, p = 0.046, n = 15) and almost significantly with the frequency of tall herbs (r = 0.41, p = 0.063, n = 15).

Table 31. Indicator species for the groups identified from the cluster analysis of open habitat-associated species in unplanted improved and wet grassland sites. Only species with a maximum IndVal of 25 or more are included. The max IndVal is indicated in bold.

	A: mixed group (n = 12)	B: species-rich wet grassland (n = 3)	P value of max IndVal
Cheilosia latifrons	0	**67**	0.024
Cheilosia pagana	4	**57**	0.046
Eristalis interrupta	1	**59**	0.070
Platycheirus clypeatus	42	**58**	0.079
Platycheirus occultus	4	**86**	0.001
Platycheirus perpallidus	1	**60**	0.052
Platycheirus rosarum	7	**83**	0.005
Platycheirus scambus	6	**86**	0.004
Sphaerophoria interrupta	0	**67**	0.026
Trichopsomyiaflavitarsis	6	**100**	0.001

Table 32. Comparison of hoverfly species richness in the groups identified from analysis of unplanted improved and wet grassland sites.

Habitat association of fauna	Mean species richness/site (SD)		t	p
	A: mixed group (n = 12)	B: species-rich wet grassland (n = 3)		
Open	18.3 (3.3)	23.7 (3.8)	2.48	0.028
Anthropophobic	2.4 (1.2)	4.3 (0.6)	2.53	0.024
Wet grassland specialists	4.3 (1.1)	6.7 (1.6)	3.13	0.008
Wetland specialists	3.2 (1.9)	5.0 (2.6)	1.37	0.193

The ordination of woody vegetation species was also strongly influenced by the three outlying sites (Coog, Kilb and Kill). When these sites were excluded, the ordination produced a single axis solution with high final stress (33.7), with sites arranged in general sequence of woody vegetation species richness. Numbers of woody vegetation associated species were correlated with the index (I_u) of broadleaved woody vegetation cover (r = 0.59, p = 0.010, n = 15) and numbers of tree/tall shrub species were almost significantly correlated with this index (r = 0.42, p = 0.059, n = 15). The residuals from the regression of woody vegetation species richness against I_u were correlated with the frequency of understorey vegetation (r = 0.61, p = 0.014, n = 15).

5.3.3 Comparison of planted and unplanted sites

5.3.3.1 Hoverfly assemblages

The ordination of the open habitat-associated species in the peatland sites does not show any separation between the planted and unplanted sites (Figure 12).

The ordinations of the open habitat-associated and woody vegetation species in the improved and wet grassland sites show a broad separation between the planted and unplanted sites (Figure 13 and Figure 14).

5.3.3.2 Species richness

Among the improved and wet grassland sites, the mixed model ANOVA indicated that there were significantly more woody vegetation and tree/tall shrub species in the planted compared to the unplanted sites. The interaction terms (HABITAT x PLANTED) were not significant for these analyses. There were no other significant differences in species richness between the planted and unplanted sites (Table 33).

Table 33. Comparison of species richness in unplanted and planted sites.

Site type	Species group	Mean species richness/site (SD)		Test statistic[1]	p
		Unplanted	Planted		
Peatland	Open	18.8 (6.2)	17.1 (6.3)	1.1	0.301
	Surface water	11.0 (4.8)	12.4 (4.4)	-1.7	0.133
	Wetland specialists	3.6 (2.5)	2.4 (2.3)	-1.0	0.312
	Woody vegetation	7.9 (3.8)	7.5 (3.4)	0.5	0.610
	Trees-tall shrubs	7.4 (3.5)	7.1 (4.5)	0.4	0.732
Improved and wet grassland	Open	19.3 (4.1)	20.7 (4.7)	0.8	0.400
	Surface water	13.0 (4.1)	13.2 (3.4)	0.0	1.000
	Woody vegetation	12.8 (3.7)	17.5 (3.5)	14.4	0.003
	Trees/tall shrubs	6.0 (2.4)	10.7 (2.8)	18.1	< 0.001

[1] t_7 for the peatland sites; $F_{1,11}$ for the improved and wet grassland sites.

In the planted improved and wet grassland sites, numbers of woody vegetation and tree-tall shrub associated species were positively related to the length of hedges and treelines and the weighted cover of other broadleaved woody vegetation (multiple regressions: woody vegetation species $F_{2,11}$ = 7.77, p =0.008; tree-tall shrub species $F_{2,11}$ = 5.27, p =0.025).

However, separate multiple regressions with the latter parameter replaced by planted broadleaved trees and naturally regenerated broadleaved woody vegetation were not significant. The differences in numbers of woody vegetation and tree/tall shrub associated species between the paired planted and unplanted sites were correlated with the differences in the indices of woody vegetation cover (woody vegetation species, $r = 0.66$, $p = 0.019$; tree-tall shrub species, $r = 0.69$, $p = 0.013$; $n = 12$). The growth stage of the planted conifers was not correlated with the species richness of these species groups.

5.3.3.3 *Species composition*

Nineteen species showed significant differences in abundance between the planted and unplanted sites, of which nine were more abundant in the planted sites and ten were more abundant in the unplanted sites (Table 34). With the Bonferroni correction, only six of the differences remained significant, but the preferences of most of the other species listed are ecologically meaningful (see Section 5.4.3). Wetland specialists were significantly more abundant in the unplanted sites: mean IA = -0.53 (SD 0.57), t_8 (H_0: IA = 0) = - 2.9, $p = 0.017$. However, the mean difference in abundance was not significant for open, surface water, woody vegetation and tree/tall shrub-associated species.

Table 34. Index of relative abundance (IA) between planted and unplanted sites.

Species	IA/site (SE)[1]	One-sample t-test (H_0: IA = 0)		
		t	df	p[2]
Melanostoma scalare	1.59 (0.13)	12.0	22	**< 0.001**
Platycheirus albimanus	1.18 (0.13)	9.1	22	**< 0.001**
Rhingia campestris	0.7 (0.14)	5.0	20	**< 0.001**
Helophilus pendulus	0.54 (0.18)	3.0	18	0.008
Melangyna lasiophthalma	0.48 (0.15)	3.1	17	0.006
Meliscaeva cinctella	0.44 (0.18)	2.4	11	0.033
Volucella bombylans	0.42 (0.17)	2.5	18	0.022
Leucozona lucorum	0.42 (0.13)	3.2	15	0.005
Platycheirus scutatus	0.4 (0.18)	2.3	20	0.034
Neoascia podagrica	-0.56 (0.25)	-2.2	13	0.047
Trichopsomyia flavitarsis	-0.71 (0.23)	-3.1	10	0.012
Platycheirus rosarum	-0.74 (0.23)	-3.2	12	0.008
Eupeodes corollae	-0.75 (0.24)	-3.1	9	0.013
Dasysyrphus albostriatus	-0.77 (0.2)	-3.8	4	0.019
Platycheirus perpallidus	-0.95 (0.31)	-3.1	4	0.037
Neoascia tenur	-1.06 (0.23)	-4.5	5	0.006
Cheilosia bergenstammi	-1.16 (0.2)	-5.8	8	**< 0.001**
Sphaerophoria fatarum	-1.49 (0.11)	-13.0	5	**< 0.001**
Lejogaster metallina	-1.64 (0.15)	-11.3	7	**< 0.001**

[1] IA is the mean standardised residual across all site pairs from pairwise regressions between log (x+0.5) species abundances in paired planted and unplanted sites. A positive value indicates that the species tends to be more abundant in planted sites, and a negative value indicates the opposite.

[2] p-values that are significant when alpha is adjusted by the Bonferroni correction are highlighted in bold.

5.4 DISCUSSION

5.4.1 Interpretation of trapping results

Two factors complicate the interpretation of the trapping results: variation in trapping efficiency between sites and movement of adult hoverflies away from their breeding habitat.

Variation in trapping efficiency between sites is always likely to occur to some extent but only becomes a significant issue when the variation is large. The potential variation in trapping efficiency can be assessed by examining the total catches of hoverflies from each site, although it is always possible that variation in total catches reflect absolute differences in abundance. In our dataset three of the unplanted improved and wet grassland sites had very low total catches: Coog, Kilb and Kill. In Kilb and Kill, the Malaise traps were located in very exposed positions, although the reason for the low catch at Coog is not so obvious. In addition three of the four peatland sites surveyed in 2002 had very low catches. All peatland sites are, by their nature, relatively exposed. The low catches in 2002 probably reflect the cool and wet weather during May-July 2002.

There is no simple way to allow for the effects of variation in trapping efficiency. Rarefaction methods are not appropriate because the hoverfly assemblages do not meet the assumptions required for the use of these methods (Krebs 1989). We used total catches to identify sites where low trapping efficiency may have occurred and took this into account in the interpretation of our results. For pairwise comparison of the species composition of planted and unplanted sites, we used residuals from the relationship of species relative abundances between the two sites as an index of each species' relative abundance. This method allows for potential variation in trapping efficiency between sites, without being biased by large abundances of a few common species.

Movement of adult hoverflies away from their breeding habitat is a well-known phenomenon. At many of our sites, we recorded wetland species where there was no wetland habitat within the immediate vicinity of the traps. Usually these sites were located on hillsides above valleys with riparian habitat. The dominant vegetation in wetland habitat (grasses, sedges and rushes) are plants with flowers that produce pollen but not nectar. As many hoverfly species require nectar as well as pollen (e.g., Gilbert 1981), adult hoverflies of certain species may need to move outside the wetland habitat to find food sources.

In the early summer of 2004, we also recorded many forest hoverflies in the unplanted peatland sites. The weather was very hot and dry at this time, so hoverflies may have been dispersing to find moist conditions, such as along the flushes and brooks where our traps were located.

5.4.2 Hoverfly biodiversity in pre-afforestation habitats

5.4.2.1 Biodiversity value

The pre-afforestation sites that we surveyed generally had a low representation of the characteristic hoverfly fauna associated with these habitats. However, even in a high quality example of a particular habitat, 100% representation of the predicted fauna would not be expected because: (i) not all species present will necessarily have been recorded; and (ii) the occurrence of species in particular sites will also depend upon stochastic processes. Therefore, to evaluate our results it is necessary to have some reference values for what proportion of the expected fauna a similar sampling program would record in a high quality site. A threshold of 50% has been proposed as indicating that a habitat is in "reasonable condition to support its associated biodiversity", and a threshold of 75% representation as indicating that the habitat is in "good condition" (Speight & Castella 2001), but it is not clear how rigorously these thresholds have been tested, and how they vary with levels of sampling intensity.

There are two available datasets that provide some comparative data (Table 35). The Glinny-Boulaling dataset is from a lowland farm in south Cork. This farm includes areas of wet grassland and remnant acid fen. A high representation of the characteristic fauna was recorded, but this resulted from an intensive survey (20 Malaise traps and additional sampling methods). The presence of remnant acid fen and woodland habitats was probably significant in enhancing the representation of wet grassland specialists and hedge species. The Cloheen Strand Intake dataset is from a coastal wet grassland site (with some marsh and swamp habitat) within the Clonakilty Bay candidate Special Area of Conservation in south Cork, and is based upon a similar sampling effort to that carried out in the present study (two Malaise traps for 54 days), although the sampling period (July-August) was different. This dataset shows that in a high quality wet grassland site, this level of sampling effort can produce a good representation of the characteristic humid grassland fauna. The representation of wet grassland specialists, compared to the levels achieved in the pre-afforestation wet grassland sites in the present study, is particularly notable. By contrast, the representation of the hedge fauna was more similar to the level in the pre-afforestation sites.

Table 35. Representation of characteristic hoverfly fauna in reference datasets.

Species group	Percentage of predicted fauna recorded:			
	Glinny-Boulaling[1]		Cloheen Strand Intake[2]	
	Typical	Supplementary	Typical	Supplementary
Humid grassland	73%	61%	60%	53%
Wet grassland specialists	67%		52%	
Hedge	89%	92%	47%	44%
Scrub specialists	12%		8%	

[1] Source: analysis of species list in Speight (2001).

[2] Source: T.Gittings, unpublished data.

Further reference datasets from high quality examples of semi-natural habitats are required to allow better evaluation of sample datasets. However, the above comparisons indicate that the pre-afforestation wet grassland sites that we surveyed were not of high biodiversity importance in terms of their representation of the characteristic humid grassland and wet grassland specialist faunas. In fact, while our sites had vegetation representative of the humid grassland category described by Speight *et al.* (2004), all the sites had been modified to some degree by drainage along field boundaries and most had had some input of artificial fertiliser. Therefore, under a strict interpretation of the definitions in Speight *et al.* (2004) these sites would be classified as lowland improved grassland. While a corollary of this interpretation would be that unimproved humid grassland does not exist to any significant extent in Ireland, the low representation of wet grassland specialists in our sites does provide support for this interpretation. In terms of their hoverfly fauna, our wet grassland sites were intermediate between the lowland improved grassland and unimproved humid grassland categories of Speight *et al.* (2004). A review of the Irish hoverfly fauna has shown that Ireland lacks a distinct fauna associated with unimproved well-drained grasslands (Speight 2004). Therefore, the wet grassland specialist fauna are largely species that were probably originally associated with wetland habitats, and the representation of wet grassland specialists in wet grassland sites probably reflects the availability of wetland habitat conditions (provided by supplementary habitats such as flushes and brooks or discrete habitats such as small patches of marsh). It would be interesting to know whether any sites with high representation of the wet grassland specialist fauna occur that do not have associated wetland habitats (unlike the two examples in Table 35).

In the peatland sites, we recorded higher levels of representation of the characteristic bog hoverfly fauna, but this reflects the depauperate nature of the typical Irish hoverfly fauna of these habitats. Also, as these habitats are generally widespread in the landscapes in which they occur, the representation of the typical fauna is probably not a very meaningful measure of their biodiversity value. The very low representation of specialist wetland species in these sites was probably a more accurate reflection of their biodiversity value.

Despite the low representation of characteristic fauna, we did record a number of rare species. These were mainly associated with two types of habitat features: surface water habitat features and *Salix* scrub, and it is notable that they were not associated with the main pre-afforestation habitat in the sites concerned (with the exception of *Anasimyia lunulata* at

Veeu). However, in several cases, the likely habitat origin for the species recorded was outside the site defined for our study. Therefore, these records are probably best interpreted as reflecting the occurrence of species of conservation importance in landscapes subject to afforestation, rather than in individual sites. In this context, our record of the threatened *Platycheirus amplus* from the BIOFOREST Growth Cycle Project (Gittings *et al*. 2005a; Smith *et al*. 2005) is another example of a rare species associated with surface water habitat features occurring in this type of landscape.

5.4.2.2 *Indicators of hoverfly biodiversity*

We found few relationships between hoverfly biodiversity and the habitat parameters that we recorded. This reflects the confounding effects of movement of adult hoverflies (see Section 5.4.1). For example, high numbers of species associated with wet habitat features occurred in some sites where there were no wet habitat features within 100 m of the traps: indeed the highest representation of wet grassland specialists occurred at one such site (RAUU). Riparian habitat occurred within several hundred metres of these sites suggesting that a landscape scale analysis might produce better relationships between hoverfly biodiversity and habitat parameters. Also, the cluster group of species-rich wet grassland sites were all sites located in landscapes with extensive areas of semi-natural habitat. The wet grassland sites in the other cluster group were generally located in landscapes more dominated by intensive agriculture. Therefore, our results suggest that hoverflies (sampled by Malaise trapping) may be a better indicator of biodiversity at a landscape scale compared to a site scale. However, other studies, have found better relationships between hoverfly biodiversity and habitat at a site scale. In the BIOFOREST open space project (Gittings *et al*. submitted), we found strong relationships between the biodiversity of wet habitat associated hoverflies in Malaise trap catches and the occurrence of wet habitat features within 100 m of these traps. Similarly, Malaise trap monitoring of hoverfly assemblages at Pollardstown Fen has been able to discriminate between limnocrene spring habitat and historically dewatered sites over scales of a few hundred metres (Gittings *et al*. 2005b). The difference with these studies may be, in the former case, the absence of better wetland habitat in close proximity to the sites, and, in the latter case, the sampling of the wetland habitat itself, rather than adjacent ecotonal or dry habitats.

We did, however, find some informative relationships between hoverfly biodiversity and habitat parameters. In the peatland sites, considering the two sampled years separately (because of apparent differences in the sampling efficiency between the two years; see Section 5.4.1), the two sites with acid fen habitat (Geau and Incu) had the highest number of wetland specialist species, and both sites had notable hoverfly species associated with the acid fen habitat. Other notable invertebrates were also recorded at these sites, including the Large Heath butterfly (*Coenonympha tullia* (Müller, 1764)) at both sites, the Keeled Skimmer dragonfly (*Orthetrum coerulescens* Fabricius 1798) at Incu, and the Narrow-bordered Bee Hawkmoth (*Hemaris tityrus* L.), a rare horsefly (*Hybomitra muehlfeldi* (Brauer, 1880)) and a spider new to Ireland (*Meioneta mollis*; see Section 6) at Geau[2]. While further data would be required to statistically show a relationship, our results support our subjective initial evaluation that the presence of extensive acid fen indicated that these sites were likely to support an interesting invertebrate fauna.

In the improved and wet grassland sites, the numbers of woody vegetation species were positively related to the index of woody vegetation cover and the frequency of understorey vegetation. Therefore, the amount and development of hedge and scrub vegetation appears

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[2] We also recorded another rare moth, the Red-tipped Clearwing (*Synanthedon formicaeformis* (Esper, 1783)), from Geau, although this species was more likely associated with nearby areas of wet woodland.

to be a good indicator of the biodiversity of the woody vegetation-associated hoverfly fauna. The numbers of scrub specialist species that we recorded were too small to detect any relationships with habitat parameters. However, as discussed above (see Section 5.4.2.1), we recorded several notable species associated with *Salix* swamp. This indicates the potential value of small patches of willows, particularly when they are located in riparian or swamp habitats. At one pair of sites, we recorded two rare species associated with *Fraxinus excelsior* treelines. This site had well-developed treelines with several overmature *Fraxinus excelsior* trees, indicating the potential value of well-developed treelines for maintaining a woodland component of the hoverfly fauna in open landscapes.

5.4.3 The initial effects of afforestation on hoverfly biodiversity

We did not find any differences in hoverfly assemblages or hoverfly species richness between planted and unplanted peatland sites. However, there were differences in assemblages of both open and woody vegetation associated species between planted and unplanted improved and wet grassland sites, and there was higher number of woody vegetation associated species in the planted sites.

The analyses of indices of relative abundance between planted and unplanted sites produced ecologically meaningful results. Across all sites, species that were more abundant in planted sites were either widespread generalist species that occur in a wide range of open and forest habitats (*Helophilus pendulis*, *Melanostoma scalare*, *Rhingia campestris*, and *Platycheirus albimanus*) or woody vegetation species (*Leucozona lucorum*, *Melangyna lasiophthalma*, *Meliscaeva cinctella*, *Platycheirus scutatus*, and *Volucella bombylans*). The association of the generalist species with the planted sites might reflect the greater availability of flowering plants (food sources for adults) due to relaxed grazing pressure and/or the more sheltered conditions in these sites. Species that were more abundant in the unplanted sites were all open habitat-associated (except *Dasysyrphus albostriatus*) and included four wetland specialists (*Neoascia tenur, Platycheirus perpallidus, Platycheirus rosarum,* and *Lejogaster metallina*), an additional wet grassland specialist (*Trichopsomyia flavitarsis*) and a species only associated with peatland habitats (*Sphaerophoria fatarum*). Overall, wetland specialists were significantly more abundant in unplanted compared to planted sites.

Our results, therefore, show within five years after afforestation an increase in the biodiversity of woody vegetation species and a decrease in the biodiversity of species associated with specialist open habitats. An increase in the biodiversity of woody vegetation species following afforestation may seem unremarkable, but it is notable that we were able to detect an effect at such an early stage. In the peatland sites, we did not detect this effect, probably because of the smaller size of the planted trees (due to slower growth) and the more open habitat. In the improved and wet grassland sites, the canopy had not closed (apart from in one site, Balb), but the trees had generally developed sufficiently to create more sheltered conditions, particularly alongside retained hedges and patches of scrub invasion. We found that the increase in species richness between paired planted and unplanted sites was related to the difference in amount of broadleaved woody vegetation. All three components of the latter (retained hedges, patches of scrub and planted broadleaved saplings) contributed to this effect. Therefore, retention of existing hedges and scrub and planting of broadleaved trees appear to promote the development of hoverfly assemblages associated with woody vegetation habitats in young conifer plantations. The regression line slopes suggest that where there is no difference in the amount of broadleaved woody vegetation between planted and unplanted sites, there will still be a higher number of woody vegetation associated species in the planted sites, probably due to the shelter effect discussed above. If conifer afforestation results in a decrease in the amount of broadleaved woody vegetation (e.g., through scrub clearance), the regressions suggest that this will cause a decline in the numbers of woody vegetation associated species. We have similarly found that, in mature plantations, broadleaved woody vegetation was associated with higher biodiversity of tree and shrub associated hoverflies (Gittings *et al.* submitted). In both cases, most of the hoverfly species involved have predatory larvae that feed on aphids in tree and shrub foliage. Therefore, broadleaved trees and shrubs appear to enhance the biodiversity of natural enemies of aphids and may have a potential role in controlling aphid outbreaks in commercial forestry plantations.

The decrease in the biodiversity of species associated with specialist open habitats probably mainly reflects the effects of drainage operations, as five of the six specialist species affected are associated with surface water habitats. The effects of forestry drainage on habitat conditions were most obvious at the two pairs of peatland sites with acid fen habitat (Geap/Geau and Incp/Incu). In these pairs, the planted and unplanted sites had similar topography with low hills separated by shallow valleys. The unplanted sites had extensive areas of acid fen habitat in the valleys, which were lacking from the planted sites (apart from some degraded remnants). It seems very likely that the planted sites would have had extensive acid fen habitat before afforestation, and that this habitat had been mainly lost due to forestry drainage. Therefore, while the initial habitat changes from afforestation may not affect the overall biodiversity of open habitat-associated hoverfly, these habitat changes can affect features of importance for species of particular conservation interest.

5.5 CONCLUSIONS

In open landscapes, hoverfly surveys using Malaise traps may be better suited to investigating relationships between hoverfly biodiversity and habitat at the landscape scale rather than at the site scale. The pre-afforestation habitats that we surveyed had low representation of the characteristic hoverfly fauna associated with wet grassland and specialist wetland habitats. Further reference datasets from high quality wet grassland and peatland sites would be useful to evaluate these results. However, these pre-afforestation habitats do not appear to be generally of high biodiversity importance in terms of their overall hoverfly assemblages. Maintenance of surface water, *Salix* swamp and overmature treeline habitats in landscapes subject to afforestation may be important for conservation of rare hoverfly species associated with these habitats. Afforestation should avoid peatland sites with extensive acid fen habitat as these are important for specialist wetland hoverfly species. Broadleaved woody vegetation should be promoted in young conifer plantations through the retention of existing vegetation and the planting of broadleaved trees as it encourages the development of hoverfly assemblages associated with woody vegetation habitats. Removal of hedges and scrub during afforestation should be avoided as it may cause a net decrease in hoverfly biodiversity.

6 SPIDERS

6.1 INTRODUCTION

In recent years increases in the total forest area across Europe can be largely accounted for by the afforestation of former agricultural land (MCPFE Liason Unit 2003). In Ireland, 10.2% of the total land area is currently under forestry (Forest Service 2004b), however the Irish government ultimately aims to achieve a forest cover of 17% (COFORD 2000). Although there has been a virtual cessation in state-owned afforestation in recent years, the growth of the private forest sector has continued with annual planting of 9600 ha per year, accounting for 99% of all Irish afforestation (Forest Service 2004b). The introduction of incentive schemes such as the Forest Farm Partnership, which provides farmers with annual premiums for establishing plantations on their land, has meant that 90% of the total afforestation is now accounted for by agricultural land owners (Teagasc 2005). Less productive agricultural land may be more readily selected by landowners for afforestation, however areas with lower productivity, usually those which are less intensively managed, are often those which contribute the most to biodiversity within the agricultural landscape (Cole *et al*. 2003; Downie *et al*. 1999). Major land-use changes (i.e. from agricultural land to forested land) are likely to result in major changes in flora and fauna, so there is a need to assess the biodiversity value of habitats which could potentially be used in afforestation. This will establish which habitats may be of less ecological or conservation importance, and hence should be more readily selected for afforestation.

During the forest plantation cycle, as the habitat changes from an open to a forested environment, the greatest changes in the flora and fauna occur when the canopy closes (Humphrey *et al*. 1999; Jukes *et al*. 2001; Oxbrough *et al*. 2005; Wallace & Good 1995). However, during the early stages of afforestation the silvicultural processes which take place (i.e. land preparation, chemical application, soil drainage) as well as the inevitable change in land-use that occurs (i.e. grazed to non grazed land) are also likely to influence to the organisms present. Previous research examining the initial affects of afforestation on habitats have documented changes in soil properties (Bellot *et al*. 2004; Farley & Kelly 2004), vegetation composition (Wulf 2004), and bird diversity (Allan *et al*. 1997). There has however been less investigation of these effects on invertebrates, despite their prevalence in terrestrial ecosystems and importance in food webs.

In order to evaluate the potential species loss or gain caused by afforestation it is first necessary to establish what species are present in a given habitat. The use of biodiversity indicators in habitat quality assessments have gained increasing importance in recent years (Duelli & Obrist 2003; Lindenmayer 1999; Paoletti 1999) with the recognition that for most groups of animals and plants the resources are not available to carry out complete inventories of the species present. Spiders are a large group of terrestrial predators which are primarily affected by changes in habitat structure (Uetz 1991). They can disperse aerially (Richter 1970) as well as over land, giving them the ability to colonise habitats relatively quickly compared to other groups of invertebrates with a more sessile nature. This suggests that any environmental changes which occur over a relatively short period of time, for instance the first few years after afforestation, may be reflected by changes in the spider fauna. Spiders have been successfully used as indicators of invertebrate diversity (Cardoso *et al*. 2004; Clausen 1986; Gravesen 2000; Marc *et al*. 1999), probably because of their predatory position in food webs and their relationship with vegetation structure, which can be linked to changes in environmental conditions.

With this in mind, we aim to investigate the diversity of ground dwelling spiders among several habitats typically used for afforestation in Ireland and to identify key features within these habitats which could potentially be used as indicators of their biodiversity value.

Furthermore we will examine the initial effects of afforestation on the ground dwelling spider fauna within three habitat types. This research will also provide valuable information on the distribution and ecology of spiders in several major Irish habitats which has been lacking in the past.

6.2 METHODOLOGY

6.2.1 Study areas

We used a paired sampling approach to survey ground dwelling spider assemblages in the following habitats: peatlands, improved grasslands, wet grasslands. These habitats were selected on the basis of recent afforestation trends in Ireland (Forest Service, unpublished data). Twenty four matched pairs of unplanted and planted sites (eight within each habitat) were selected on the basis of habitat, soil type, and geographical location. The site-pairs within each habitat type were widely distributed across Ireland (see Figure 3). Where possible the paired sites were adjacent to each other, although three of the pairs were separated by 1-5 km. The habitat type of the planted sites prior to afforestation was determined by consultation with land owners, foresters' records and the vegetation present at the site. The planted sites were comprised of five year old stands of Sitka spruce (*Picea sitchensis*), which is currently the most widely planted tree species in Ireland accounting for 65% of annual afforestation (Teagasc 2005). For a full list of site codes see Section 3.1.1.

The management regime varied among the habitat types: the unplanted improved grasslands were subject to heavy grazing and were usually fertilised at least once per year. The peatlands and wet grasslands were generally under low to heavy grazing pressure, however approximately half of the wet grasslands were also subject to annual silage cutting and fertilisation. In the planted sites the ground was generally prepared by mounding with drains established at frequent intervals, although drainage was much less frequent among the improved grasslands. Fertiliser was applied to most of the peatland and wet grassland planted sites though not the improved grasslands, and herbicide use was most frequent in the grassland sites in the years following planting. Mean tree height in the wet grasslands was 4.3m (±2.6SD), compared to 3.1m (±1.2SD) in the improved grasslands and just 1.6m (±0.7SD) in the peatlands.

The improved grasslands were generally on well drained brown earth or brown podzolic soils, ranging in elevation from 45-300m, and were heavily grazed. They were dominated by *Lolium perenne* but also often with some *Trifolium repens, Holcus lanatus* and *Cynosurus cristatus*. The wet grasslands were typically on moderately drained gley soils, ranging in elevation from 45-175m and were generally under low-moderate grazing pressure. *Juncus acutiflorus*, *Juncus effusus, Holcus lanatus* and *Agrostis stolonifera* were abundant in most sites although two sites had a high cover of *Molinea caerulea*. The peatlands were generally on poorly drained peat or peaty podzol soils which ranged in elevation from 20-250m with low-moderate grazing. Typical plant species included *Molinia caerulea, Calluna vulgaris, Eriophorum angustifolium* and *Eriophorum vaginatum* and mosses, especially *Sphagnum* species. For further details on the vegetation species present within each habitat see Section 4.

6.2.2 Spider sampling

Within each habitat type, sampling plots were established in areas of homogenous vegetation cover that encompassed the major types of vegetation cover present within the site; there were termed 'standard' plots. In addition to this, features within each habitat which may contribute to biodiversity of the site as a whole were also sampled: these included wet flushes, the edges of streams and ditches and hedgerows. These were termed 'supplementary' plots.

Spiders were sampled using pitfall traps which consisted of a plastic cup (7cm diameter by 9cm depth). A bulb corer was used to make a hole in the ground for the plastic cup, which was placed so that the rim of the cup was flush with the grounds' surface. In the sites which were heavily grazed (mostly improved grassland) a section of plastic piping (7cm diameter by 10cm depth), was inserted into the ground, and the plastic cup then inserted within this ring to protect it from trampling. Each plastic cup had two drainage slits cut 1cm from the rim of the cup and were filled to 1cm depth with ethylene glycol. Each sampling plot consisted of five pitfall traps which were arranged in a 4x4m grid, with one trap at each corner and one in the centre. However in the plots which sampled linear features (hedgerows, edges of ditches and streams) the pitfall traps were arranged in a line, each trap being 2m apart along the feature.

Within each site six plots were established, each plot separated by a minimum of 50m with three standard plots and three in supplementary habitats. In the grasslands all of the supplementary habitats sampled were hedgerows, whereas in the peatlands supplementary habitats were wet flushes, however in sites where these were not present linear features such as the edges of ditches and streams were sampled to adequately represent the diversity of microhabitats present. In two of the improved grassland planted sites there were no supplementary features present, so only three standard plots were established. In the wet grasslands and peatlands this gave a total of 96 plots, with 48 plots each in the unplanted and planted sites, whereas in the improved grasslands there were a total of 90 plots with 48 in the unplanted and 42 in the planted sites. The traps were active from May-July (63-65 days) and were changed three times during this period, approximately every 21 days. A large number of traps were lost through trampling in five of the sites so the pitfall traps were maintained for an extra 21 days in these pairs of sites. Due to the large number of sites and the intensity of fieldwork involving invertebrates the sampling was carried out over two field seasons in 2002 and 2004. Four pairs of peatlands were sampled in 2002 and four in 2004; two pairs of wet grasslands were sampled in 2002 and six in 2004; and two pairs of improved grasslands were sampled in 2002 and six in 2004.

Pitfall samples were stored in 70% alcohol and the spiders were sorted from the catch. Identification of spiders to species level was carried out using a x50 magnification microscope and nomenclature follows Roberts (1993). Juveniles were not identified due to the difficulty involved in assigning them to species. The lack of research carried out on spiders in Ireland means that it can be difficult to determine if species are genuinely rare or just under recorded. Therefore we used the *Provisional Atlas of British spiders* (Harvey *et al.* 2002) in conjunction with Irish records (Cawley 2001; Fahy & Gormally 2003; McFerran 1997; Nolan 2000a; Nolan 2000b; Nolan 2002a; Nolan 2002b; Smith 1999; Snazell & Jonsson 1999; van Helsdingen 1996b; van Helsdingen 1997) to determine species rarity. Species which occurred in less than five of the Irish counties and which are designated as either Nationally Scarce or are recorded as Red Data Book species (Bratton 1991) in Great Britain were considered to be rare. The species were assigned to habitat associations using the literature based on their preference for the following habitat and microhabitat characteristics: general habitat preference (open habitats, forested habitats or generalists), moisture preference (wet habitats, dry habitats or generalists) and vegetation preference (ground layer, low vegetation, bushes and trees or generalists).

6.2.3 Habitat variables

The percentage cover of vegetation was recorded in a 1m^2 quadrat surrounding each pitfall trap. The vegetation was classified into the following structural layers: ground vegetation (0-10cm), lower field layer (>10cm - 50cm) and upper field layer (>50cm – 200cm) and cover of other features such as deadwood, leaf litter and soil were also recorded. 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%). The main vegetation species present within each plot were also recorded and each plot was classified by habitat type according to the Irish habitat classification scheme (Fossitt 2000). Several of the flushes sampled were large enough to be designated as a poor fen and flush peatland habitat type, however as these areas were still not large enough to constitute a substantial area of the site they were still considered supplementary habitats. At two locations within each plot soil samples were taken using a bulb corer which extracted the top layer of substrate to a depth of 15cm. We then calculated organic content of the soil using the method outlined in Grimshaw (1989). Grazing intensity was estimated within each standard plot using the following scale: 0 = no grazing; 1 = light grazing; 2 = moderate grazing; 3 = heavy grazing (see Section 4.2.1).

6.2.4 Data Analysis

Preliminary analyses indicated that there were no major differences in species richness, abundance or assemblage structure across either the sampling years or the extra trapping period added due to trampling, so data from the two years were subsequently analysed together. Traps from the extra trapping period were used, as required, to replace traps lost during the first three sampling periods. If, after replacing lost traps, plots still had three or more traps lost (out of a possible 15), these plots were excluded from the analyses.

All variables were tested for normality and homogeneity of variance before the use of parametric statistics. ANOVA, paired t-tests and correlation analyses were carried out using SPSS (SPSS 2002) and multivariate analyses (NMS, MRBP, cluster analysis and Indicator Species Analysis) were carried out using PC-ORD (McCune & Mefford 1997). Constrained variables (percentage cover of habitat variables, Berger-Parker index) were arcsin transformed to meet the assumptions of parametric statistics.

6.2.4.1 Trends in spider diversity among the unplanted habitats

We used a mixed model ANOVA to identify trends in mean species richness, abundance and dominance per plot within each site with habitat type and plot type (standard/supplementary) as fixed factors and site as a random factor nested within habitat type. Dominance was calculated using the Berger-Parker index (Berger & Parker 1970), where d = Nmax/N (Nmax is the number of individuals in the most abundant species and N is the total number of individuals). The index ranges from 0-1, with one indicating the complete dominance of the most abundant species.

To examine general trends in spider assemblage structure within and among the habitat types we used global non-metric multi-dimensional scaling analysis (NMS). The following NMS parameter set-up was used: 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. Flexible-beta cluster analysis (with $\beta = -0.25$) and Indicator Species Analysis (Dufrêne & Legendre 1997) was used to examine these trends in more detail. Indicator Species Analysis involves combining the relative abundance and relative frequency of species within *a priori* groups to give an indicator value which is tested for significance with a Monte Carlo test. These analyses were carried out using relative abundance rather than absolute abundance as variation in vegetation cover among the habitat types may affect the efficiency of pitfall traps (Melbourne 1999).

To identify potential indicators of spider biodiversity within and among the habitat types we used Pearson's correlation analyses to investigate the relationship between habitat variables and species variables. For the analyses of habitat variables the appropriate median percentage cover value was substituted for the Braun-Blanquet value from each quadrat, and the mean value was calculated from the five quadrats within each plot. We used one-way ANOVA with Tukey's post-hoc tests to analyse trends among the spider assemblages in relation to grazing intensity and the habitat types according to Fossitt (2000).

6.2.4.2 *Initial effects of afforestation*

We used paired sample t-tests to examine the effect of afforestation on the spider assemblages of the standard and supplementary plots within each habitat type. The following response variables were tested: species richness, abundance, dominance and richness of the various habitat specialists. To examine the differences in spider assemblage structure among the unplanted and planted sites among the habitat types we used NMS, blocked multiple-response permutation procedures (MRBP) and Indicator Species Analysis. The NMS parameter set-up follows section 6.2.4.1. MRBP analysis tests the difference between pre-determined groups (site pairs) using the Euclidean distance measure. Differences are tested with the A statistic where A = > 0 if the average distance is lower than that expected by chance within each group, A = 0 if average distance is equal to that expected by chance within each group and A = 1 if the assemblages are the same within each group. The A statistic is tested for significance by comparing observed and expected values.

6.3 RESULTS

Over 14% of the traps were lost due to animal trampling. The majority of these were in the unplanted improved grasslands where nearly 27% of the traps were lost. With these plots excluded from the analyses this gave a total of 86 plots in the peatlands (43 planted and 43 unplanted), 70 in the improved grasslands (37 unplanted and 33 planted) and 90 in the wet grasslands (45 unplanted and 45 planted). For the paired site analyses this resulted in 6 paired peatlands, 7 paired wet grasslands and 5 paired improved grasslands.

There were 33157 individuals captured from 189 species: of these spiders 3448 were juveniles and so were excluded from the analyses. Within the peatland sites 8196 adults in 136 species were sampled, in the wet grasslands there were 5676 adults in 114 species and in the improved grasslands there were 4614 adults in 91 species. The most abundant species in the unplanted sites were *Pardosa pullata* (Clerck, 1757), *Pardosa amentata* (Clerck, 1757), *Silometopus elegans* (O. P.-Cambridge, 1872), *Oedothorax fuscus* (Blackwall, 1834) and *Pachygnatha degeeri* (Sundevall, 1830), each of these species constituting greater than 5% of the total adult catch within these sites. In the planted sites, *P. pullata* and *P. amentata* were the most abundant species, also constituting greater than 5% of the total adult catch each. A full list of the species sampled is given in Appendix 6. There were 42 species sampled that were associated with open habitats and 15 species associated with forested habitats; furthermore, two species were associated with dry habitats and 57 species associated with damp or wet habitats. The majority of species sampled were typical ground layer (110) species, although 30 species were associated with low vegetation and six species associated with trees and shrubs.

6.3.1 Trends in spider diversity among the unplanted habitats

6.3.1.1 *Patterns in species richness and abundance within and among the habitat types*

Across the habitat types species richness was lowest in the improved grasslands (Table 36), which supported significantly fewer species than the peatlands, though not the wet grasslands ($F_{2, 16} = 4.63$, $p = 0.03$). However the number of species supported between the standard and supplementary plots did not differ significantly ($F_{1, 16} = 0.95$, $p = 0.34$), and the interaction term between habitat type and plot type was not significant ($F_{2, 16} = 2.06$, $p = 0.15$). Abundance did not differ significantly among the habitat or plot types ($F_{2, 16} = 1.72$, $p = 0.21$ and $F_{1, 16} = 0.75$, $p = 0.40$ respectively), however the interaction between these was significant ($F_{2, 16} = 14.3$, $p = <0.001$) as abundance was greater in the supplementary plots than in standard plots within the improved grasslands and peatlands whereas in the wet grasslands it was greater in the standard plots. Dominance did not differ significantly among the habitat or plot types ($F_{2,16}$ = 0.66, p = 0.53 and $F_{1,16}$ = 1.47, p = 0.24 respectively), though there was a similar interaction between both factors as with abundance ($F_{2,16}$ = 3.28, p = 0.06).

There were no significant differences in the number of species associated with open or forested habitats among the habitat types ($F_{2,16}$ = 0.04, p = 0.96 and $F_{2,16}$ = 1.86, p = 0.17 respectively), however species richness of these groups did differ among the plot types significantly (Table 36). A significantly greater number of open associated species were supported in the standard plots ($F_{1,16}$ = 39.3, p = <0.001), although this difference was less notable in the peatland plots (interaction among habitat and plot type: $F_{2,16}$ = 3.31, p = 0.06). In contrast, there were significantly more forest associated species in the supplementary plots than in the standard plots ($F_{1,16}$ = 26.0, p = <0.001), again this difference being less noticeable in the peatlands (interaction term $F_{2,16}$ = 2.15, p = 0.13). The number of species associated with wet habitats did not differ significantly among the habitats types ($F_{2,16}$ = 2.19, p = 0.14), but there were significantly more of these species in the standard plots compared to the supplementary plots ($F_{1,16}$ = 20.6, p = <0.001). There was also a significant interaction between habitat and plot type ($F_{2,16}$ = 4.95, p = 0.02) which accounts for the similar numbers of wet habitat species found between the peatland standard and supplementary plots. Number of spider species which are associated with the ground layer increased from improved grassland, to wet grassland to the peatlands, which had significantly more of these species than either of the grassland habitats ($F_{2,16}$ = 9.80, p = 0.002), whereas similar numbers of these species were supported between the standard and supplementary plots ($F_{1,16}$ = 0.01, p = 0.93) and the interaction term was not significant ($F_{2,16}$ = 1.68, p = 0.22). Across the habitat and plot types there were no significant differences in the number of species supported which are associated with low vegetation ($F_{2,16}$ = 2.60, p = 0.11 and $F_{1,16}$ = 1.32, p = 0.27 respectively, interaction term $F_{2,16}$ = 1.60, p = 0.23).

Table 36. Mean (±SE) species richness, abundance, dominance and richness of habitat specialists per site among the habitats and plot types: standard and supplementary (Sup.).

	Improved grassland		Wet grassland		Peatland	
	Standard (n = 6)	Sup. (n = 6)	Standard (n = 8)	Sup. (n=8)	Standard (n = 5)	Sup. (n = 5)
Total species richness	16.9 (±1.8)	17.9 (±1.4)	23.0 (±2.5)	19.5 (±2.3)	26.8 (±1.8)	26.5 (±2.1)
Abundance	51 (±9.2)	174 (±44)	191 (±36)	71 (±15)	174 (±59)	225 (±54)
Berger-Parker	0.22 (±0.02)	0.32 (±0.05)	0.34 (±0.04)	0.28 (±0.04)	0.23 (±0.05)	0.30 (±0.05)
Open-associated species	7.6 (±0.7)	4.6 (±0.8)	7.7 (±0.8)	4.2 (±0.9)	6.8 (±0.9)	5.7 (±0.7)
Forest-associated species	0.5 (±0.2)	1.9 (±0.3)	0.5 (±0.1)	2.3 (±0.4)	0.5 (±0.2)	0.9 (±0.2)
Wet-associated species	6.8 (±0.6)	4.4 (±0.3)	8.9 (±.1)	4.8 (±1.2)	8.8 (±1.4)	8.5 (±1.0)
Ground layer-associated species	9.8 (±1.0)	11.4 (±1.0)	15.0 (±1.6)	13.6 (±1.5)	19.4 (±1.4)	19.4 (±1.6)
Low vegetation-associated species	1.6 (±0.3)	1.8 (±0.2)	2.7 (±0.6)	2.0 (±0.4)	3.1 (±0.4)	2.8 (±0.3)

6.3.1.2 Patterns in species assemblage structure within and among the habitat types

Preliminary analyses indicated that the spider assemblages in the peatlands were distinguished from those in the grasslands and therefore they were analysed separately. A

3-dimensional solution was recommended from the NMS ordination of the grasslands which accounted for 66% of the variation in the species data (Figure 15). Axis 1, which accounted for 26% of the variation, distinguished the spider assemblages by habitat type whereas Axis 2, which accounted for 20% of the variation, separated the spider assemblages of the standard and supplementary (hedgerow) plots. In general there was much greater variation in assemblage structure among the supplementary plots compared to the standard plots within both habitat types, with the standard plots distinguished much more clearly by habitat type across Axis 1. Among the standard plots, there was greater variation across Axis 1 in the wet grasslands compared to the improved grasslands, whereas the supplementary plots varied to a similar degree. Axis 3, which accounted for a further 20% of the variation in the species data, did not however, represent any trends in assemblage structure among the plot or habitat types.

Three ordination axes were recommended to best explain the trends in the spider species data among the peatland plots, which together accounted for 84% of the variation (Figure 16). Axis 1, which accounted for 47% of the variation, broadly distinguished the supplementary plots from the standard plots whereas Axis 2, which accounted for 18% of the variation, broadly distinguished the spider assemblages by habitat type. Axis 3 accounted for a further 19% of the variation in the species data and separated those linear supplementary plots with a high cover of upper field layer vegetation from those without. Across Axis 1 the linear supplementary plots (edges of streams and ditches) were separated from the standard plots and to a lesser degree from the supplementary plots in the flushes, the majority of which were in an intermediate position between the linear supplementary plots and the standard plots. All but one of the supplementary plots in the poor fen and flush habitat formed a distinct cluster of plots with the lowland blanket bog standard plots. These supplementary plots (IncuP1-3 and GeauP4 & 6) were in much larger areas of flush than the other supplementary habitats, which may explain their distinct position on the ordination. There were also several wet heath standard plots present in the cluster, both being from the same site as two poor fen and flush plots. Across Axis 2 the spider assemblages of the upland blanket bogs, wet heaths and to a lesser extent the lowland blanket bogs were distinguished from the cutover bogs. Cover of ground vegetation was associated with the wet heath and upland blanket bogs, whereas cover of lower field layer vegetation was associated with cutover bogs and linear supplementary plots in lowland blanket bogs (stream edges).

Cluster analysis revealed four main groups of spider assemblages which were separated by both habitat and plot type (Table 37). The cluster analysis initially separated the majority of the standard peatland plots along with some of the standard wet grassland plots (*Peatland-open* assemblage group) from the other habitat and plot types. Most of the standard plots from the improved grassland were then separated from the remaining plots (*Improved grassland-open* assemblage group). The next cluster of plots separated in the analysis consisted mainly of plots from the wet grassland, with both supplementary and standard plots present (*Wet grassland* assemblage group). The remaining plots were predominately supplementary plots from all three habitat types. Since these were mostly linear features the cluster was termed the *Linear* assemblage group.

Table 37. The number of plots within each habitat type and plot type in the spider assemblage groups revealed by cluster analysis (n = number of plots).

	Cluster group			
	Peatland-open (n = 42)	*Improved grassland-open* (n = 20)	*Wet grassland* (n = 16)	*Linear* (n = 44)
Improved grassland - Standard		15		
Improved grassland – Sup†		1	2	18
Wet grassland - Standard	7	3	8	3
Wet grassland –Sup.		1	6	13
Peatland- Standard	23			
Peatland – Sup. (stream edge)	1			5
Peatland – Sup. (ditch edge)	3			3
Peatland – Sup. (flush)	8			2

† Sup. denotes supplementary plots

6.3.1.3 *Common and rare species within and among the habitat types*

Indicator Species Analysis identified 23 significant indicator species among the assemblage groups (Table 38). These species are all relatively widespread in Ireland and the UK (Harvey *et al*. 2002; van Helsdingen 1996a), with the exception of *T. thorelli*, which, has a more local and patchy distribution (Harvey *et al*. 2002; van Helsdingen 1996a). There were five species associated with damp or wet habitats (*S. elegans, P. piraticus, A. elegans, A. olivacea, T. thorelli*) in the *Peatland-open* assemblage group. In the *Linear* assemblage group, where most of the plots were located in hedgerows, several of the indicator species are known to be typical of forested environments (*L. zimmermanni* and *M. fuscipes*). The *Improved grassland-open* assemblage group was characterised by species such as *O. fuscus*, *E. dentipalpis*, *E. atra* and *B. gracilis,* which are frequent aerial dispersers and often found in disturbed habitats (Harvey *et al*. 2002). Within the *Wet grassland* assemblage group there were only two indicator species identified; *P. amentata*, which is often found in undisturbed hay meadows (McFerran 1997) and *G. dentatum,* a species commonly found in wetland habitats (Cattin *et al*. 2003; Harvey *et al*. 2002).

There were several rare or notable species sampled within the peatland and wet grassland habitats for further details see Oxbrough (In review) however there were no rare species found within the improved grasslands. In particular, Geau, in west Kerry supported a number of rare species, including a new Irish record (*Meioneta mollis*): this site was predominately lowland blanket bog, however the wet flushes in which the supplementary plots were situated were sufficiently large for them to be classified as a poor fen and flush in the Irish habitat guidelines (Fossitt 2000). Several specimens of *M. mollis* were found within the poor fen and flush habitat in this site (GeauP3, 4 & 6) which may indicate that this species has a particular preference for wet habitats. In Britain this species is known from damp locations, and has been found in both grasslands and woodlands (Harvey *et al*. 2002). This site supported several other rare species including *Nigma puella* and *Zelotes lutetianus* which were sampled in the standard lowland blanket bog plots (GeauP5) and *Satilatlas britteni* which was sampled in the poor fen and flush (GeauP 3 & 6). *N. puella* is usually found in trees and bushes and so not commonly encountered through pitfall trapping, however *S. britenni* and *Z. lutetianus* are usually encountered in the ground layer in damp or wet locations (Harvey *et al*. 2002). Specimens of *S. britenni* were also sampled in a lowland blanket bog (VeeuP1,4,5 & 6) and an upland blanket bog (TieuP5) in north-west Ireland which may indicate that this species has a general preference for bog habitats. The rare species *Maro sublestus,* which is known to have a particular preference for wet or peaty habitats, was sampled in a supplementary stream plot in a lowland blanket bog (VeeuP6).

Table 38. Indicator Species Analysis among the assemblage groups. The maximum indicator value and associated significance (Monte Carlo test) are in bold type for each species.

	Percentage indicator value			
	Peatland-Open	*Linear*	*Improved grassland-Open*	*Wet grassland*
Silometopus elegans	**61*****	1	0	0
Pirata piraticus	**57*****	0	1	13
Pardosa pullata	**56*****	4	13	17
Agyneta olivacea	**44****	3	0	0
Lepthyphantes mengei	**42*****	6	0	3
Antistea elegans	**40*****	0	0	4
Ceratinella brevipes	**35****	14	0	3
Pardosa nigriceps	**35*****	4	3	3
Trichopterna thorelli	**33****	0	0	0
Trochosa terricola	**30****	4	1	11
Lepthyphantes zimmermanni	2	**50****	1	2
Monocephalus fuscipes	0	**39*****	0	14
Agyneta subtilis	10	**34****	0	2
Dismodicus bifrons	2	**34****	6	15
Maso sundevalli	3	**30****	1	4
Bathyphantes parvulus	4	**33****	1	11
Oedothorax fuscus	0	0	**89*****	3
Erigone dentipalpis	0	1	**88*****	0
Pardosa palustris	1	1	**64*****	4
Erigone atra	0	4	**76*****	6
Bathyphantes gracilis	4	16	**36****	18
Pardosa amentata	1	4	9	**77*****
Gnathonarium dentatum	1	0	0	**35*****

p = >0.05; ** p = >0.01; *** p = >0.001

In a wet heath site (IncuP3) the supplementary plots located in the flushes were large enough to be classified as poor fen and flush. In these plots several specimens of *Baryphyma gowerense* were sampled as well as in the standard plot of a wet grassland site (BoolP1 & 6). This species is classified as Insufficiently Known in the British Red Data Books (Bratton 1991), where it occurs mostly in coastal marshes, although it has been found in inland fens (Harvey *et al*. 2002), and the present study may indicate this species also has a preference for wetter habitats in Ireland. In the wet grassland hedgerows (MntpP 2 & 3, MoigP5) several specimens of *Saloca diceros* were sampled, this species known to be associated with wet habitats (Harvey *et al*. 2002). *Milleriana inerrans* was sampled in a wet grassland site (ClouP3): although this species has only previously been recorded from one location in Ireland, it has a scattered and patchy distribution in Britain where it is thought to be uncommon rather than rare. In Britain *M. inerrans* is found in many habitats, however it has a preference for newly disturbed habitats, which may explain its presence in the wet grassland site, which was heavily grazed.

6.3.1.4 Indicators of spider diversity

6.3.1.4.1 Habitat indicators

Trends in spider species variables within the peatlands were further examined using the Irish habitat classifications (Fossitt 2000) (Table 39). In the peatland standard plots, species richness and abundance was highest in the upland blanket bogs and lowest in the cutover bogs, whereas abundance was substantially lower in the cutover bogs compared to the other peatland habitats, although these differences were not significant. Dominance was greatest in the lowland blanket bogs, but not significantly so. The differences in the number of species associated with wet habitats and species associated with the ground layer were approaching significance being greatest in the upland blanket bogs and lowest in the cutover bogs. The number of species associated with low vegetation was lowest in the lowland blanket bogs but did not differ significantly among the peatland habitats.

Among the peatland supplementary plots total species richness and abundance was lowest in the cutover bogs and abundance was also relatively low in the wet heaths, however these differences were not significant. Dominance differed significantly among the peatland habitats being greater in the poor fen and flush and upland blanket bog than in the lowland blanket bog, although dominance was also relatively low in the cutover bog and wet heath supplementary plots. The number of species associated with wet habitats and the ground layer were highest in the upland and lowland blanket bogs, this difference approaching significance for ground layer species. The number of species associated with low vegetation did not differ significantly among the peatland habitats, though again was highest in the upland and lowland blanket bogs.

There were no significant differences in the species variables among the different peatland supplementary plot types (flushes, and the edges of ditches and streams).

Table 39. Mean (±SE) species richness, abundance, dominance and richness of habitat specialists among the peatland plots as classified by the Irish habitat categories (Fossitt 2000). One-way ANOVA and Tukey post hoc tests among the habitat types are shown within each plot type: Standard, df $=_{3,21}$); Supplementary df $=_{3,22}$).

	Cutover bog	Poor fen and flush	Lowland blanket bog	Upland blanket bog	Wet heath	ANOVA F
Standard plots	n = 3	-	n = 3	n = 9	n = 7	
Total species richness	18.0 (±1.2)	-	21.7 (±1.9)	27.9 (±2.3)	23. 5 (±2.6)	2.23
Abundance	49 (±12)	-	279 (±103)	217 (±42)	203 (±49)	1.84
Berger-Parker	0.24 (±0.03)	-	0.38 (±0.05)	0.28 (±0.04)	0.29 (±0.04)	2.14
Wet-associated species	3.3 (±1.2)	-	6.7 (±1.3)	8.1 (±1.0)	6.0 (±0.9)	2.70†
Ground layer-associated species	12.0 (±2.1)	-	16.7 (±2.4)	20.2 (±1.7)	16.3 (±1.7)	2.60†
Low vegetation- associated species	3.3 (±0.3)	-	1.7 (±0.7)	3.3 (±0.3)	3.3 (±0.4)	2.26
Supplementary plots	n = 7	n = 5	n = 3	n = 4	n = 4	
Total species richness	19.3 (±2.8)	23.4 (±5.3)	30.7 (±2.7)	28.5 (±1.0)	25.8 (±1.0)	1.76
Abundance	92 (±46)	136 (±60)	149 (±41)	278 (±95)	90 (±29)	1.59
Berger-Parker	0.21 (±0.02)	0.30[a] (±0.02)	0.15[b] (±0.01)	0.34[a] (±0.07)	0.17 (±0.02)	4.88**
Wet-associated species	4.4 (±1.3)	5.2 (±1.3)	8.0 (±0.6)	8.0 (±0.6)	5.8 (±0.3)	2.00
Ground layer-associated species	13.7 (±2.5)	14.2 (±3.0)	22.7 (±1.8)	21.3 (±0.9)	17.8 (±0.8)	2.71†
Low vegetation- associated species	2.4 (±0.3)	3.8 (±1.1)	2.3 (±0.3)	3.0 (±0.7)	3.5 (±0.5)	0.97

† Approaching significance, $p = 0.1 - 0.05$; * $p = >0.05$; ** $p = >0.01$.

[a] denotes value significantly greater than value marked with [b]

6.3.1.4.2 *Structural indicators*

Within the *Peatland-open* assemblage group, species richness, richness of species associated with wet habitats and species associated with the ground layer were significantly negatively correlated with cover of ground vegetation and positively correlated with cover of lower-field layer vegetation (Table 40), whereas abundance and dominance showed the opposite trend. In the *Linear* assemblage group cover of ground vegetation was not related to any of the species variables. Wet-associated species were significantly positively correlated with lower-field layer cover and negatively correlated with cover of deadwood, and total species richness and abundance were significantly negatively correlated with cover of the upper-field layer. In the *Improved grassland-open* assemblage group species associated with the ground layer were significantly positively correlated with cover of ground vegetation and negatively correlated with cover of lower field layer vegetation. In the Wet grassland assemblage group the species associated with the lower field layer vegetation showed the opposite trend. There were no significant relationships with open or forest associated species, or with soil organic content.

Table 40. Correlations (Pearson's r) between species variables and cover of habitat variables.

	Habitat variable			
Species variable	Ground vegetation	Lower-field layer	Upper-field layer	Deadwood
Peatland-open (n= 42)				
Species richness	-0.38*	0.32*	i.d	i.d
Abundance	0.42**	-0.56***	i.d	i.d
Berger-Parker	0.24	-0.39**	i.d	i.d
Wet-associated species	-0.32*	0.18	i.d	i.d
Ground layer associated species	-0.33*	0.28	i.d	i.d
Linear (n = 44)				
Species richness	-0.06	0.07	-0.45**	-0.09
Abundance	0.02	0.07	-0.50***	-0.14
Berger-Parker	-0.16	0.20	0.27	-0.01
Wet-associated species	-0.11	0.38*	-0.23	-0.30*
Improved grassland-open (n= 20)				
Ground layer species	0.46*	-0.39	i.d	i.d
Wet grassland (n = 16)				
Low vegetation associated sp	-0.59*	0.56*	i.d	i.d

* p = >0.05; ** p = >0.01; *** p = >0.001
i.d = Insufficient data

6.3.1.4.3 *Management indicators*

The species variables within each grazing category are shown in Table 41, however due to the large number of traps lost it was only possible to carry out this analysis within the wet grasslands. Grazing intensity generally had a negative effect on spiders with species richness, abundance and richness of the wet habitat specialists as well as number of species associated with ground layer and low vegetation being significantly lower in the moderate and heavily grazed sites compared to the ungrazed and lightly grazed plots. The number of species associated with open habitats generally declined over the grazing intensity gradient although this trend was not significant. The dominance index did not differ with grazing intensity.

Table 41. The mean (±SE) number of species, individuals, dominance and habitat specialist species among the standard plots in the wet grasslands divided into categories of grazing intensity.

	Grazing intensity				ANOVA F
	Ungrazed (n = 6)	Light (n = 6)	Moderate (n = 5)	Heavy (n = 4)	
Total species richness	28.2[a]	27.0[a]	16.6[b]	13.3[b]	
	(±1.7)	(±2.1)	(±2.4)	(±1.1)	13.0***
Abundance	221[a]	261 [a, c]	120[d]	63[b]	
	(±23)	(±43)	(±41)	(±46)	6.0**
Berger-Parker	0.33	0.43	0.32	0.30	
	(±0.06)	(±0.10)	(±0.04)	(±0.04)	1.0
Open-associated species	8.5	9.0	6.4	5.5	
	(±1.0)	(±0.7)	(±1.6)	(±0.5)	2.3
Wet-associated species	6.2[a]	5.8[a]	2.8[b]	2.3[b]	
	(±0.5)	(±0.8)	(±0.1)	(±0.8)	6.9**
Ground layer-associated species	19.0[a]	16.7[a]	11.2[b]	8.8[b]	
	(±1.0)	(±1.0)	(±1.5)	(±0.5)	17.5***
Low vegetation-associated species	3.3[a]	4.2[a]	1.4[b]	0.5[b]	
	(±0.8)	(±0.6)	(±0.4)	(±0.5)	6.7**

* p = ≥0.05; ** p = ≥0.01; *** p = ≥0.001

[a] denotes value significantly greater than value marked with [b]

[c] denotes value significantly greater than value marked with [d]

6.3.2 Initial effects of afforestation

6.3.2.1 The effects of afforestation on species richness and abundance

6.3.2.1.1 Standard plots

The mean number of species per standard plot within each habitat type is shown in Table 42. Total species richness did not differ significantly between the unplanted and planted peatland and wet grasslands; however in the improved grasslands mean species richness was significantly greater in the planted sites. Across the habitats total abundance and the number open-associated and wet habitat associated species was greater in the unplanted sites, though for abundance this difference was not significant among the improved grasslands. In contrast, the number of species associated with forested habitats was higher in the planted sites across the habitats, though not significantly so in the wet grasslands. The number of species associated with the ground layer did not differ significantly between the unplanted and planted peatlands and wet grasslands; however in the improved grassland the number of ground layer species supported was significantly greater in the planted sites. Similarly, the number of low vegetation species did not differ significantly among the unplanted and planted sites in the wet grasslands and peatlands, however there were significantly more of these species supported in the improved grasslands more planted sites.

Table 42. Mean (±SE) species richness, abundance and dominance per standard plot in the unplanted (UP) and planted (P) sites within each habitat type. Paired sample t-test statistics and associated significance between the unplanted and planted sites within each habitat are also shown.

	Peatland			Wet grassland			Improved grassland		
	UP	P	t (df = 7)	UP	P	t (df = 7)	UP	P	t (df = 5)
Species richness	24.1 (±1.6)	23.0 (±1.6)	0.82	22.0 (±2.3)	20.0 (±1.1)	1.15	16.3 (±1.5)	20.9 (±0.9)	-3.63*
Total abundance	198 (±42)	91 (±17)	4.25**	182 (±32)	77 (±13)	4.41**	173 (±42)	99 (±19)	2.04†
Berger-Parker dominance index	0.32 (±0.09)	0.21 (±0.03)	4.01**	0.35 (±0.04)	0.26 (±0.03)	1.5	0.33 (±0.05)	0.31 (±0.05)	0.32
Open-associated species	6.6 (±0.6)	5.0 (±0.4)	2.71*	7.6 (±0.8)	5.5 (±0.5)	4.47**	7.5 (±0.7)	5.5 (±0.7)	2.48*
Forest-associated species	0.7 (±0.2)	1.3 (±0.2)	-2.60*	0.5 (±0.1)	1.1 (±0.4)	-1.77	0.5 (±0.2)	1.2 (±0.1)	-3.31*
Wet-associated species	8.0 (±0.9)	5.8 (±0.5)	2.16†	8.5 (±0.9)	6.4 (±1.0)	3.85**	6.8 (±0.3)	4.4 (±0.6)	3.31*
Ground-layer associated species	17.2 (±1.3)	15.8 (±1.4)	1.64	14.5 (±1.6)	13.9 (±0.8)	0.51	9.6 (±0.6)	13.4 (±1.1)	-2.98*
Low vegetation associated species	2.9 (±0.3)	2.5 (±0.5)	0.73	2.6 (±0.5)	2.5 (±0.4)	0.35	1.6 (±0.5)	2.6 (±0.3)	-3.00*

† Approaching significance, p = >0.1-0.05; * p = >0.05; ** p = >0.01

6.3.2.1.2 Supplementary plots

In the peatlands the number of species associated with wet habitats was significantly lower in the planted supplementary plots compared to the planted plots (Table 43), the number of ground layer species was also lower in the planted plots, however this difference was not significant. There were no significant differences in any of the measures of species richness, abundance or dominance between supplementary plots in the planted and the unplanted sites within the wet and improved grasslands.

Table 43. Mean (±SE) species richness, abundance and dominance per supplementary plot in the unplanted and planted sites within the peatlands. Paired sample t-test statistics and associated significance between the unplanted and planted sites within each habitat are also shown.

	Unplanted	Planted	t (df = 7)
Species richness	24.5	21.0	1.60
Total abundance	170	80	1.88
Berger-Parker dominance index	0.2	0.2	0.44
Open-associated species	6.1	5.4	0.85
Forest-associated species	0.5	1.2	-1.89
Wet-associated species	8.0	5.5	3.60**
Ground-layer associated species	17.0	14.5	1.95†
Low vegetation associated species	3.1	2.6	1.06

† Approaching significance, p = >0.1-0.05; * p = >0.05; ** p = >0.01

6.3.2.2 The affects of afforestation on spider assemblages

6.3.2.2.1 Standard plots

Preliminary analyses showed that the grassland and peatland spider assemblages were distinct, so these were subsequently analysed in separate ordinations. The NMS ordination of spider assemblages among the unplanted and planted grassland standard plots accounted for 84% of the variation in the data with three axes best explaining this variation (Figure 18a). Axis 1, which accounted for 31% of the variation, separated the unplanted from the planted plots. Axis 2, which represented 30% of the variation, distinguished the unplanted improved grasslands from both the unplanted and planted wet grassland plots. The improved grassland planted plots exhibited much greater variation spider assemblage structure compared to the unplanted plots, the planted plots being spread across Axis 2. The difference between the unplanted and planted spider assemblages was in the improved grasslands was significantly different (MRBP: A = 0.291, p = 0.008), whereas the difference between the assemblages of the unplanted and planted wet grasslands was less prominent and not statistically significant (MRBP: A = 0.045, p = 0.139). Axis 3, which accounted for a further 22% of the variation in the species data, represented a further separation of the unplanted and planted plots.

The NMS ordination of the peatland standard plots accounted for 86% of the variation in spider assemblage structure (Figure 18b) which was represented by three axes. Axis 1, which explained over 53% of the variation in the species data, distinguished the unplanted and planted plots which also differed significantly in assemblage structure (MRBP: A = 0.162, p = 0.004). The planted plots were associated with higher cover of upper and lower field layer whereas the unplanted plots were associated with higher cover of ground vegetation. There was some separation of the unplanted plots by habitat type with clusters of lower blanket bogs, wet heath and upland blanket bog/cutover bog plots. Across Axis 2 (which accounted for 17% of the variation) the planted plots exhibited greater variation than the unplanted plots. There was less coherence among the peatland habitat types and the wet heath plots were split into two groups by site. Axis 3, which accounted for 15% of the variation in the species data, further separated the planted plots, distinguishing several lowland and upland blanket bog plots (ToopP3 & 5, TogpP4) from the remaining plots.

The species with high indicator values in the unplanted sites (Table 44) are mostly ubiquitous species of open habitats (*P. pullata, P. palustris*) or found frequently in grasslands (*O. fuscus, E. atra, E. dentipalpis*), however *P. piraticus*, *S. elegans* and *W. vigilax* are all associated with wet habitats (Harvey *et al.* 2002). In the planted sites most of the species with high indicator values are generalist species which are commonly found broad range of open habitats, although *R. lividus* is a ubiquitous species which is found in both open and forested habitats. Notable exceptions however, include *O. gibbosus* which is associated with wet habitats, and *L. zimmermanni* and *M. fuscipes* which are associated with forests (Harvey *et al.* 2002).

Table 44. Indicator Species Analysis of the standard plots in the unplanted and planted sites within each habitat group. Only species with a significant maximum indicator value >50% are shown. Maximum indicator value and associated significance (Monte Carlo test) is indicated in bold type.

	Peatland		Wet grassland		Improved grassland	
	U	P	U	P	U	P
Pardosa pullata	**70*** **	15	-	-	-	-
Pirata piraticus (Clerck, 1757)	**67****	16	**71*** **	4	-	-
Silometopus. elegans	**58***	14	-	-	-	-
Walckenaeria vigilax (Blackwall, 1851)	**54****	3	-	-	-	-
Pocadicnemis pumila (Blackwall, 1841)	16	**83*** **	12	**66****	2	**75*** **
Lepthyphantes zimmermanni (Bertkau, 1890)	3	**73*** **	-	-	-	-
Oedothorax gibbosus (Blackwall, 1841)	10	**65*** **	7	**52***	-	-
Bathyphantes gracilis (Blackwall, 1841)	1	**54****	-	-	-	-
Robertus lividus (Blackwall, 1836)	5	**50****	2	**51****	0	**56****
Oedothorax. fuscus	-	-	**63*** **	0	**100*** **	0
Erigone atra (Blackwall, 1833)	-	-	**50****	5	**97*** **	1
Bathyphantes parvulus (Westring, 1851)	-	-	13	**64****	0	**79*** **
Lepthyphantes ericaeus (Blackwall, 1853)	-	-	10	56*	0	**62*** **
Erigone dentipalpis (Wider, 1843)	-	-	-	-	**100*** **	0
Pardosa palustris (Linnaeus, 1758)	-	-	-	-	**69*** **	0
Pocadicnemis juncea (Locket & Millidge, 1853)	-	-	-	-	2	**72****
Monocephalus fuscipes (Blackwall, 1836)	-	-	-	-	0	**62*** **
Walckenaeria acuminata (Blackwall,1833)	-	-	-	-	0	**56****

* p = ≥0.05; ** p = ≥0.01; ***≥0.001

There were five rare species found in the unplanted standard plots: *S. britteni*, *N. puella*, *Z. Lutetianus, M. inerrans* and *B. gowerense* (for details see section 6.3.1.3). These species were all sampled in the peatlands although *B. gowerense* was also found in the wet grasslands. Only one rare species was found solely in the planted sites, *Episinus truncatus* (Latrielle, 1809), which was sampled in an improved grassland site (KilmP3) and is usually associated with heathlands (Roberts 1993).

6.3.2.2.2 Supplementary plots

The NMS ordination of the spider assemblages among the supplementary plots in the unplanted and planted grasslands accounted for 70% of the variation in the species data with three axes best explaining this variation (Figure 19a). Axis 1, which accounted for 28% of the variation, separated the unplanted improved grassland plots from the planted plots to some degree, with the planted plots forming a relatively tight cluster. However the unplanted and planted wet grassland plots were not distinguished from each other. Similarly, the assemblages of the unplanted and planted supplementary plots did not differ significantly from each other: A = -0.020, p = 0.243 and A = 0.014, p = 0.170 in the improved grasslands and wet grasslands respectively. The remaining two axes each explained 21% of the variation among the spider assemblages representing some unknown variation in the supplementary plots (hedgerows) unrelated to habitat type or site.

The NMS ordination of the spider assemblages among the supplementary plots in the unplanted and planted peatlands accounted for 77% of the variation in the species data (Figure 19b). Three axes best explained the variation in the spider assemblages with Axis 1 accounting for 40%, Axis 2 20% and Axis 3 17%. Axis 1 distinguished the spider assemblages of the unplanted and planted supplementary plots, which differed significantly (MRBP: A = 0.143, p = 0.006). These differences were mainly driven by the distinct spider assemblages of the unplanted plots in the supplementary flushes from the poor fen and flush habitat, and to a lesser degree the supplementary flushes sampled in other habitat types; the unplanted upland blanket bogs also formed a distinct cluster of plots. The unplanted poor fen and flush and upland blanket bogs plots exhibited less variation in assemblage structure than the planted plots of the same habitat type, whereas the lowland blanket bog and wet heath planted and unplanted plots were not distinct in assemblage structure and exhibited similar degrees of variation. There were no supplementary plots sampled in the planted cutover bogs however the unplanted plots were widely scattered across both axes. The planted plots were associated with higher cover of upper and lower field layer vegetation whereas the unplanted plots were associated with cover of ground vegetation. Axis 2 may be reflecting differences in the litter layer among the plots, with high organic content representing high litter in the top stratum of the soil and high soil cover subsequently representing low cover of litter.

The indicator species identified in the supplementary plots among the unplanted and planted sites are shown in Table 45. The unplanted sites were characterized by generalist open species such as *P. pullata*, *E. atra* and *L. tenuis* as well as ubiquitous species such as *D. nigrum* and *L. robustum*, although *W. vigilax* is associated with a variety of wet habitats. In the planted sites all of the species with high indicator values were habitat generalists being commonly found in open and forested areas, with the exception of *B. parvulus* which is most commonly found in a range of grasslands types (Harvey *et al*. 2002).

Table 45. Indicator Species Analysis of the supplementary plots in the unplanted and planted sites within each habitat group. Only species with a significant maximum indicator value >50% are shown. Maximum indicator value and associated significance (Monte Carlo test) is indicated in bold type.

	Peatland		Wet grassland		Improved grassland	
	U	P	U	P	U	P
Pardosa pullata	**87*****	10	-	-	-	-
Walckenaeria. vigilax	**69*****	3	-	-	-	-
Pachygnatha degeeri	**50****	5	-	-	-	-
Pocadicnemis. pumila	22	**78****	-	-	-	-
Pocadicnemis. juncea	-	-	19	**54***	22	**61***
Lepthyphantes tenuis (Blackwall, 1852)	-	-	-	-	**75*****	8
Erigone atra	-	-	-	-	**65*****	1
Leptorhoptrum robustum (Westring, 1851)	-	-	-	-	**50***	3
Bathyphantes. parvulus	-	-	-	-	17	**73*****
Lepthyphantes. ericaeus	-	-	-	-	13	**66****
Walckenaeria. acuminata	-	-	-	-	5	**50***

* p = >0.05; ** p = >0.01; ***>0.001

There were four rare species found in the supplementary plots: *M. mollis, S. britteni, B. gowerense* and *M. sublestus* (see section 6.3.1.3 for details). *Saloca diceros* (O. P. –Cambridge, 1871) was found in both unplanted and planted (CoosP1) hedgerows in the wet grasslands.

6.4 DISCUSSION

This study used a paired-site approach which compared unplanted and planted habitats matched for environmental attributes and geographical location. Ideally, researchers should be able to survey a location both before and after the event being investigated (Before-After-Control-Impact design: Green 1979). Furthermore, problems involving spatial and temporal variation in such studies can be overcome with adequate temporal replication on multiple control sites (Underwood 1994). However for investigations involving land-use changes such as afforestation that take place over many years a sampling design which tracks a single or multiple sites over time is difficult to implement. Paired-site sampling designs have been successfully utilised in previous research (Barnett *et al*. 2004; Berger *et al*. 2002; Kladivko *et al*. 1997), and in the present study has allowed ground dwelling spider assemblages to be investigated during the early stages of afforestation over the course of one field season rather than over several years.

6.4.1 Spider diversity among the unplanted habitats

6.4.1.1 Spider diversity among the standard plots

The spider assemblages were differentiated among the habitats investigated, with the improved grasslands being particularly distinct from the peatland and wet grassland in terms of species composition, lower species richness and lack of rare species. This is consistent with other studies which compare intensively managed grassland with more semi-natural grasslands (Cole *et al*. 2003; Downie *et al*. 1999). The difference in spider species composition among the habitat types probably reflects differences in the management regime (i.e. grazing and mowing intensity, chemical application, management history) and environmental factors (i.e. soil type and moisture, and altitude) as well as cover of vegetation.

Among the habitat types, improved grassland had the most intensive management regime, being subject to relatively intensive grazing, but also periodic fertilisation and reseeding. Intensive grazing leads to the suppression of vegetation and there has been extensive research on the negative effect of this ground dwelling spider communities (Cole *et al*. 2003; Dennis *et al*. 2001; Dennis *et al*. 1998; Downie *et al*. 1999; Gibson *et al*. 1992). Habitat structure (and hence vegetation structure) is the primary factor influencing spider communities; for instance vegetation structure is architecturally important for web builders and aids the concealment of active hunters (see Uetz (1991) for a review). In the present study the improved grasslands exhibited little variation in assemblage structure and were characterised by ubiquitous, opportunistic species such as *E. atra*, *E. dentipalpis*, and *O. fuscus.* These species readily disperse by ballooning (Harvey *et al*. 2002; Weyman *et al*. 1995) and consequently are typically found in newly disturbed habitats such as intensively managed grasslands (Cole *et al*. 2003; Rushton & Eyre 1989).

The wet grasslands generally represented an intermediate spider assemblage between improved grassland and peatland (Figure 15). The wet grasslands were subject to varying levels of grazing intensity as well as periodic cutting for hay and occasional fertiliser application, so they may also represent an intermediate level of management intensity between the improved grassland and the peatlands. Schwab *et al.* (2002) found that spider species richness increased with decreasing management intensity of hay meadows (frequency of cutting and fertilisation). In contrast however, Cattin *et al.* (2003), found that mowing did not influence total richness or numbers of rare and specialised wetland spiders, although more generalist rare species were adversely affected. However they only examined low intensity mowing treatments. As seen in the present study there is a negative effect of moderate-heavy grazing pressure on species richness, abundance and richness of species associated with wet habitats. This agrees with the findings of Perner and Malt (2003) who showed that species associated with wet habitats decreased with increasing management intensity on arable land. One rare species was found in heavily grazed wet grasslands (*M. inerrans*), but this species is not rare in Britain, and as an aerial disperser, it may have a more common distribution in Ireland than current records indicate.

Among the habitat types, general differences in environmental conditions are likely to have a substantial effect on spider species composition. The habitat types represented a broad gradient in soil moisture from the improved grasslands on relatively dry soils to the peatlands on much wetter soils. Soil moisture has been found to positively influence spider density (Kajak *et al*. 2000), whereas Usher (1992) found spider assemblage structure was influenced by a wet-dry gradient. This may account for the higher number of specialist wetland species supported in the peatlands and to a lesser extent the wet grasslands in the present study, which included both common species (*S. elegans, P. piraticus, A. elegans, G. dentatum*) and rare species (*S. britenni, S. diceros, M. sublestus*). Furthermore, soil moisture may also indirectly affect the spider fauna through its influence on the vegetation species present (Cattin *et al*. 2003).

6.4.1.2 Spider diversity among the supplementary plots

The supplementary features sampled in the grasslands were hedgerows, and considering the influence of vegetation structure on ground dwelling spider communities it is unsurprising that the spider fauna differed among the standard and supplementary plots. In the hedgerows, the canopy creates very different environmental conditions for ground-dwelling spiders from those in the standard plots: for example ground and lower-field layer vegetation is suppressed (Pywell *et al*. 2005) and a more stable microclimate is created (Pollard 1968). This probably accounts for the presence of spiders associated with forests, being found in the hedgerows in this and a previous study (Toft & Lovei 2000).

The supplementary plots also exhibited greater variation in assemblage structure, within the habitat types, than the standard plots although dominance was higher in supplementary plots, thus indicating that they support a less diverse community. The hedgerows surveyed exhibited considerable variation in the plant species composition, which included hawthorn (*Crataegus monogyna*), willow (*Salix* sp.) and ash (*Fraxinus excelsior*), and structure including substantial variation in the understorey layer such as bramble cover (*Rubus fruticosus* agg). The width of the hedgerows also varied considerably within the grasslands, ranging from 1-15m (personal observation). Both of these factors have previously been found to affect the spider fauna of hedgerows (Maudsley *et al*. 2002; Toft & Lovei 2000). The substantial variability in hedgerow structure and species composition and few consistent differences among grassland types may explain why the supplementary plots are less distinguishable by habitat type than the standard plots. The hedgerow supplementary plots were characterised by more generalist species than the standard plots, and did not support any rare species. Similarly, Toft and Lovei (2000) also found that hedgerows support open generalist species rather than specialists.

In the peatlands the supplementary plots were comprised of the edges of ditches, streams and flushes. The spider assemblages present however did not form a distinct group from the other plots sampled within the peatlands. Rather, these supplementary plots were separated into two groups, most of the linear plots (edges of ditches and streams) were more similar to the hedgerow plots whereas most of the flushes were more similar to the peatland standard plots (Table 37). In this case, the spider fauna in supplementary peatland plots may be responding to variation in soil moisture. The hedgerows were generally situated on drier soils and ditches are likely to cause drainage of the immediate surrounding area which may also contribute to drier soil conditions. The edges of streams and in particular the ditches may have a more complex vegetation structure due to the protection from grazing afforded by steep banks. This may also result in the spider assemblages of some supplementary peatland plots being more similar to that of the hedgerows. Flushes by definition are directly influenced by ground water, in contrast, the banks of ditches and streams, though possibly affected by temporary flooding may otherwise remain relatively dry. Some spider species are sensitive to small differences in humidity (Norgaard 1951) which may be influenced by soil moisture. This could be especially important in the peatlands where there are fine-scale patterns in microtopography and moisture that correspond with vegetation zonation.

6.4.1.3 Indicators of biodiversity value

There was generally a greater variety of habitats within the peatlands than within grasslands. For instance, using the *Guide to Habitats in Ireland* (Fossitt 2000), the peatlands were classified into five different categories: upland blanket bogs, lowland blanket bogs, cutover blanket bogs, and poor fens and flushes (sampled as a supplementary habitat). Several of the supplementary flushes sampled were of sufficient size to be classified as poor fen and flush in the Irish habitat guidelines (Fossitt 2000). Although these plots did not have high overall richness of species or wet-associated species, a number of rare species were supported. Furthermore, along with the lowland blanket bogs they supported a distinct spider fauna from the other peatland plots. This suggests that the flushes, but more especially areas which are large enough to be designated as a poor fen and flush habitat, may be important indicators of peatland biodiversity value. In contrast, the cutover bogs relatively low species richness and lack of rare species suggests that they may be indicators of peatlands with low biodiversity value. The supplementary plots in the wet heaths also exhibited relatively low abundance and species richness of habitat specialists compared to the upland and lowland blanket bogs. This was due to the poor catches in the three supplementary ditches sampled within one site (Sliu), however these ditches had recently been cleared of vegetation so the poor catches were probably a result of this disturbance.

In the wet grasslands, moderate-high grazing intensity is an indicator of low overall spider diversity, and the number of wetland species, which may be specifically associated with this habitat, respond in a negative way to increased grazing pressure. Although it was not possible to analyse grazing effects in the improved grasslands or peatlands, the negative effects of grazing intensity on spider diversity across a range of habitats has been well documented (Dennis *et al*. 2001; Dennis *et al*. 1998; Gibson *et al*. 1992; McFerran *et al*. 1994). There was a positive influence of ground vegetation abundance on ground layer species in the improved grasslands, however the majority of these species were very common. Furthermore, the low biodiversity value of the improved grassland spider fauna in general may mean that variation in grazing regime or vegetation structure within this habitat may be of little consequence. In the peatlands cover of ground vegetation was negatively associated with total species richness and richness of wetland species whereas these species variables were positively associated with lower field layer cover. This is unlikely to be due to habitat differences (for example, there was generally higher bryophyte cover in the upland blanket bogs compared to the lowland blanket bogs which is a feature of the habitat rather than the grazing regime) as the relationship between species richness and vegetation cover was unrelated to habitat type. It may, however, be related to variation differences in grazing regime within the sites. For instance, Dennis *et al.* (1998) found that overall spider richness as well as the abundances of *L. mengei*, *A. olivacea* and *S. elegans* (common species in the peatlands in the present study) were significantly higher in tussocks compared to swards in upland grasslands which they related to protection from grazing. This suggests that information on the management of a site will be a more useful indicator of biodiversity value than a survey of the vegetation structure present.

Several species associated with forests occurred in high frequency and abundance in the supplementary hedgerow plots which probably benefit from the forest-type conditions created by the canopy; however hedgerows may not benefit other species groups. Indeed, in the present study upper-field layer cover was an indicator of low wetland species richness as well as low overall species richness and abundance in the supplementary linear plots. It is more likely that the richest hedgerows will be those which do not have such an extensive canopy cover and can support open species which are associated with the surrounding habitat as well as forest edge and forest species (Toft & Lovei 2000).

6.4.2 Initial effects of afforestation

6.4.2.1 Standard plots

Across all of the habitat types there were differences in the spider species supported between the unplanted and planted sites: species associated with the particular habitat characteristics of the unplanted site were replaced by generalist species. Across all of the habitats in the present study the number of spider species associated with forests was greater in the planted sites whereas the number of open associated species supported in these sites was lower. Furthermore, forest species increased in relative abundance to the point where two forest specialists, albeit forest generalists (*L. zimmermanni* and *M. fuscipes*), were identified as indicators of the planted peatland and improved grassland sites respectively. It is surprising that even at this early stage in the afforestation cycle forest specialists begin to colonise the plantations. Although the five year old spruce trees have not yet reached canopy closure, the trees, which are planted 2m apart and were generally 2-3m high (mean 2.2m ±0.9SD, range 0.1-4.3m) may create some of the conditions characteristic of forested environments such as protection from the wind and stable microclimates (Pollard 1968) which in turn may benefit forest specialists.

The unplanted sites were associated with higher cover of ground layer vegetation whereas the planted sites were associated with higher cover of lower and upper field layer vegetation. The improved grassland sites were much more intensively managed than the wet grasslands or the peatlands, with heavier grazing pressure. It is likely that the higher number of spider species supported in the planted improved grasslands is related to a reduction in grazing pressure, which has been found to directly influence spider diversity (Dennis *et al*. 2001) through the resulting increase in vegetation structure (Dennis *et al*. 1998). This can be seen in the present study where the unplanted improved grasslands were characterised by species such as *E. atra*, *E. dentipalpis* and *O. fuscus*, which are pioneer species frequently found dominating disturbed habitats (Cole *et al*. 2003). Furthermore, the planted sites had significantly higher numbers of species associated with low vegetation in the improved grasslands.

Prior to afforestation the unplanted sites are prepared for plantation establishment to encourage more suitable conditions for tree growth. This includes the establishment of drains, which on particularly wet sites are created at relatively frequent intervals: for instance the recommended spacing for mound drains is 8m (Forest Service 2003). Indeed, in the present study the majority of the sites had an extensive network drains established (personal observation). These drainage networks radically change the moisture content of the soil, especially in the peatlands and wet grasslands. Soil moisture has been found to determine spider distribution (Usher 1992) and may explain the lower number of both common and rare species associated with wet or damp conditions supported in the planted sites across all habitat types. In the peatlands however, the total number of species associated with wet habitats was not significantly lower in the planted sites, although total abundance was significantly lower which was largely accounted for by a reduction in the number of the Lycosid spiders *P. pullata* and *P. amentata*, both of which have a preference for wetter habitats (Harvey *et al*. 2002).

Although there was no difference in total species richness between the unplanted and planted sites in either the peatlands or wet grasslands there was a significant difference in the spider assemblages supported in the peatlands suggesting that the effects of afforestation on spiders are greater in the peatlands than the wet grasslands. All of the peatlands had fertiliser (phosphate) applied after afforestation, whereas the improved grasslands and approximately half of the wet grasslands did not. Furthermore, being generally the wettest sites they may have been subject to more intensive drainage. In the present study the unplanted peatland habitats support a more specialised fauna than the wet and improved grasslands. This suggests that spider species associated with peatlands are more likely to be affected by the habitat changes which occur after afforestation, for example dryer soils and growth of lower field layer vegetation. Vegetation structure changes dramatically after afforestation; unplanted bogs are dominated by a mixture of mosses, sedges, low herbs and some grasses and low ericaceous shrubs. After planting, purple moor-grass (*Molinia caerulea*), a coarse tussocky grass, was dominant in many of the peatland sites (Section 4). This may explain why species associated with low vegetation were significantly lower in the planted peatlands; these species may have been specialised to the particular vegetation structure present on the site prior to afforestation.

6.4.2.2 Supplementary plots

It is unsurprising that the hedgerows sampled in the grasslands did not differ to a great degree in either species richness or assemblage structure between the unplanted and planted sites. Whilst hedgerows are likely to be adversely affected by the effects of shading when the trees are more developed, at this early stage in the forest plantation cycle trees of 2-3m in height are unlikely to have a large impact. Furthermore, the Irish *Forest Biodiversity Guidelines* (Forest Service 2000c) recommend that hedgerows be regarded as areas for biodiversity enhancement within plantations, meaning that they should remain undisturbed during the afforestation process and a 3m buffer zone should be established around them (Forest Service 2000c) presumably to protect them from shading and disturbance by machinery. Indeed, the present study found that the hedgerows support a diverse spider fauna including forest and open specialist species, so the protection of these features during the afforestation process may be important for plantation biodiversity.

In the unplanted peatlands the supplementary flushes sampled, especially those larger areas which were designated as poor fen and flush habitats, supported a distinct spider fauna with several rare species compared to those in planted sites. The spider assemblages were probably affected by similar factors as the standard plots such as dryer soils caused by drainage and release from grazing pressure. The Irish *Forestry and Water Quality Guidelines* (Forest Service 2000d) stipulate that aquatic zones ('a permanent or seasonal river, stream or lake') which are marked on an Ordnance Survey six-inch maps should be protected during the afforestation process by a minimum buffer zone of 10m. These current guidelines are likely to exclude the supplementary flushes sampled within this survey because they are either not included in the definition or because of their small size. Indeed, in the present study neither of the larger supplementary flushes sampled which were designated as poor fen and flush habitats were marked on the Ordnance Survey six-inch maps. Furthermore these features are not regarded as areas for biodiversity enhancement within the Irish *Forest Biodiversity Guidelines* (Forest Service 2000c) and so are afforded little protection during the afforestation process.

6.4.3 Conclusions

The unplanted habitats surveyed support different ground dwelling spider assemblages which reflect the major differences in both environmental conditions and management regime. Although the hedgerows may not support the unique species found within the grasslands they are nonetheless an important part of the agricultural landscape. The present study investigated the ground dwelling spider fauna, however, the spider fauna in the higher vegetation layers of hedgerows is also important (Maudsley *et al*. 2002). Furthermore, the lack of diversity within improved grasslands in general, means that a large part of the spider diversity within this landscape may be accounted for by hedgerows.

Peatlands, and to a lesser extent wet grasslands, support a distinctive and diverse spider fauna, which suggests that in terms of biodiversity value, improved grassland is the preferable habitat for afforestation. However, it may be unrealistic to expect land owners to establish forest plantations solely on improved grassland, which is often the most fertile and productive agricultural land. This study suggests that management and habitat indicators have the potential to identify sites within each habitat type which have more or less biodiversity value for spiders. Therefore future research should focus on developing these indicators to be of use when assessing habitat quality in the afforestation site selection process, most especially with regard to assessing features within sites which may be of high biodiversity value, such as wet flushes. This way, if habitats such as wet grassland and peatland are considered for afforestation then sites with lower biodiversity value, such as those with heavier grazing or cutover bogs, can be identified.

During the forest cycle a fundamental change in the flora and fauna occurs at the time of canopy closure (French 2005; Humphrey *et al*. 1999; Jukes *et al*. 2001; Oxbrough *et al*. 2005; Wallace & Good 1995): indeed, prior to this the spider assemblages resemble that of the pre-planting habitat type (Oxbrough *et al*. 2005). In addition to this, however, the present study suggests that even in the early stages of the forest cycle (first five years) there is an impact on the spider fauna, with the rare or specialist species being replaced by habitat generalists. The improved grasslands did exhibit an initial increase in spider biodiversity after afforestation, but this is unlikely to persist after canopy closure (Oxbrough *et al*. 2005).

This study indicates that peatlands are the most sensitive of the habitats surveyed to afforestation, suggesting that in terms of biodiversity loss, this habitat is the least suitable for afforestation. In particular, small areas of wet flush within peatlands, which support distinct and rare species, should be protected during the afforestation process. Furthermore, the loss of specialist species across all of the habitats after afforestation indicates that retained areas which are selected for biodiversity enhancement when plantations are established will benefit from as little disturbance to the habitat and pre-planting management regime as possible.

7 BIRDS

7.1 INTRODUCTION

Large scale afforestation over the last half of the 20th century has seen forest cover recover from less than 1% 100 years ago (Mitchell 2000) to almost 10% (IPCC 2001). The Irish government aims to further increase forest cover to 17% by 2030 (Department of Agriculture Food and Forestry 1996). Although the natural climax vegetation of most of Ireland is forest, the new Irish forest estate is very different from native Irish woodlands, being composed largely of exotic, single species plantations. The large-scale conversion of open, agricultural and semi-natural habitats to forest plantations therefore constitutes a major ecological change in the Irish landscape, which has the potential to impact profoundly on Irish biodiversity. Existing legislation and official statutory guidelines safeguard against afforestation conflicting with conservation interests within or near any areas with a formal conservation designation such as Natural Heritage Areas, Special Protection Areas and Special Areas of Conservation (Forest Service 2000c). However, a stated objective of the National Forestry Standard, which sets out the criteria for Sustainable Forest Management, is to provide "evidence for the maintenance of habitats of significant conservation value within afforestation areas" (Forest Service 2000e). In order to effectively do this, it is necessary to have knowledge of two things:

1.the biodiversity in habitats and landscapes that are commonly afforested, and

2. how this biodiversity is affected by afforestation

Previous studies have found that birds are an appropriate taxon from which to select biodiversity indicators, especially in the context of forestry (Ferris & Humphrey 1999). Also, the conservation status of several Irish bird species of biodiversity concern is likely to be affected by widespread afforestation of the Irish countryside (O'Halloran *et al*. 1998). As concluded by the review of open habitat biodiversity in section 2.3.4, there is currently a dearth of information on the bird communities of the most commonly afforested habitats in Ireland. We address this knowledge gap by describing the bird assemblages of peatland, wet grassland and improved grassland sites. We investigate the environmental variation within these sites in order to determine what variables are responsible for the differences between bird assemblages. We also compare these bird assemblages with those of recently afforested sites, in order to ascertain the initial effects of afforestation on the birds of these open habitats. We compare the effectiveness of two alternative survey methodologies (point counts and mapping surveys) in estimating the abundance of individual bird species. Finally, we use the findings of this study and those of a previous study we conducted on breeding birds of different forest growth stages to evaluate the changes that will occur in bird assemblages of afforested habitats beyond five years after planting.

7.2 METHODS

7.2.1 Study site selection

We selected 51 sites comprising 3 individual unplanted sites and 24 pairs of unplanted and planted sites. The latter had been afforested with Sitka spruce five years before we surveyed them, and were paired with unplanted sites according to pre-planting habitat, which was judged to have been similar at the time that planted sites were afforested. There were eight paired sites in each of three categories: peatland, wet grassland and improved grassland. Two of the individual sites were wet grassland and the other was improved grassland. For a more detailed rationale and description of methods for site selection, see section 3.1.

7.2.2 General survey methods

Two different survey techniques were used to collect data from the study sites. Mapping surveys were conducted in unplanted sites, in order to make a complete census of each site. This was not possible in most of the planted sites, due to extensive cover of the planted conifer crop and the non-crop shrubs. Large areas of tree or shrub cover are better suited to surveying by point counts. We conducted point counts in both unplanted and planted sites, in order to be able to make an effective comparison between them.

Mapping and point count data were collected from each site over the course of two visits, one in May/early June and one in June/early July, in the summers of 2002 and 2004. All bird surveys were conducted between the hours of 0700 and 1800, in order to exclude periods in the early morning or evening when birds are known to be particularly active, and which are therefore not comparable with other times of the day. Bird surveying fieldwork was restricted to relatively fine days, i.e. surveys were not carried out in heavy or persistent rain, or in strong (greater than Beaufort Scale 4) winds. Clusters of birds of the same species were recorded as having a maximum number of two individuals, in order to reduce the influence of fledged family parties, frequently encountered during the second visit, on density estimates. Flying birds of the following mobile species and groups of species were excluded from analyses because their presence could not be assumed to indicate an association with the habitat at that location: Grey Heron, gulls, pigeons, Common Swift, wagtails, hirundines, corvids, Starling and finches (apart from Bullfinch). When not given in the text, scientific names of bird species are given in Appendix 8.

7.2.3 Mapping surveys

Each unplanted site was surveyed for birds using a mapping methodology similar to that described by Brown and Shepherd (1993). The surveyor covered the site on foot, making sure that all areas were approached to within a minimum distance of 50 m. For areas with shrub or tree cover this minimum distance was reduced to 20 m. The species, behaviour and position of all birds seen or heard was recorded on a 1:4000 map of the site. All birds detected within 15 m of the site boundaries were included in analyses.

A hedge survey was carried out in each unplanted site in the late summer and autumn of 2002 and 2004. Every hedge in each site was recorded using a standardised hedge-recording sheet, based on the protocol devised by Clements and Toft (1992). A sample data collection sheet is included in Appendix 7. The hedge was scored in the following categories: canopy height (1-4), canopy width (1-4), canopy structure (1-4), number of mature standard trees (1-4), number of young standard trees (trees were classed as young if they had clearly not attained the height or diameter typical of mature specimens of the appropriate species) (1-4), % gaps (1-4), number of connections to other hedges (1-4), number of connections to woodland or forest (1-2), presence and size of hedgebank (0-3), presence and structure of ditch vegetation (0-3), and presence of a grass verge on either side of the hedge (0-2). See Appendix 7 for more detail on how each of these indices was estimated. For analyses at the site scale, the size of each hedge was defined as the sum of its height, width and cross-sectional indices: 5 – 8 for small hedges, 8.5 - 9.5 for medium hedges and 10 – 12 for large hedges. In addition, all woody plant species contributing to hedge structure in the hedge canopy (the main body of the hedge) or as standard trees were identified to species or genus level. Willow (*Salix*) species/hybrids were identified only to genus level, but when two or more obviously different types of willow were present, this information was recorded. The position of all hedges on the site was recorded a 1930 series six-inch (scale 1:4000) Ordnance

Survey map. The same map was used to record the shape, size and position of any substantial areas of non-hedge shrub and tree cover. These areas were not surveyed in comparable detail to the hedges, but values for canopy cover were estimated for each, and they were assigned to the following categories: treelines (linear tree-features with a width of one tree), semi-natural woodland (two trees or more in width and greater than 2 m in height), shrub cover (2 m or less in height), pre-thicket forest plantation and closed canopy forest plantation. Areas occupied by other non-pasture habitat types, such as farmyards and gardens, were also mapped.

7.2.4 Points counts

Point counts (Bibby *et al*. 2000) were used to sample bird communities at all sites, planted and unplanted. Six points were situated in each site (giving a minimum of 48 point counts in each unplanted and planted habitat type), at a minimum of 100 m apart, and incorporating as wide a variety of internal and external field boundaries of the site as could be accommodated by the site's size and shape. Points were located in the field using a Garmin GPS 12, accurate to within approximately 5 m when satellite reception is unobstructed, and 1930 series six-inch (scale 1:4000) OS maps, which clearly show most current field boundaries. Point counts were conducted for 10 minutes, during which time the identity and distance from the observer of all birds detected visually and aurally within a radius of 50m were recorded. Timing of visits was varied such that all points that received one visit in the morning and one in the afternoon.

The following elements of vegetation structure were recorded within a radius of 50 m at each point: pre-thicket crop tree height (m); % pre-thicket crop canopy cover; other tree height (m); %other tree canopy cover; % shrub layer cover; % herb and moss cover; tree density (stems per m^2). Shrub layer was defined as all woody vegetation between 0.5 and 2 m in height, not including the commercial tree crop.

7.2.5 GIS analysis

All data from bird mapping surveys, hedge survey and point counts were entered onto computer and attached to digitised maps using Arcview GIS 3.2. This program was used to calculate lengths of hedges and areas of non-hedge features, and to assign birds recorded during mapping surveys to hedge, non-hedge feature and open pasture categories. For the purposes of calculating the proportion of each site's area that was covered by hedge vegetation, and for assigning birds to hedges, we took the width of each hedge to be 24m. This is substantially wider than even the widest hedge, but this width was necessary to take account of variability in location of plotted points in the GIS representing birds. Because the majority of hedges were between 2 and 4 m wide, this area is referred to as land within 10m of a hedge. To assign birds to individual hedges, the distance of each bird detection to the nearest hedge was calculated using an add-in script for ArcView: Distance and Azimuth Matrix version 2.1 (Jenness 2005). All birds within 12 m of a hedge were assigned to the nearest hedge. In four cases, birds within equal distance of two hedges were assigned to one of the two hedges at random.

7.2.6 Density estimation

We used distance sampling techniques, following the methodology of Buckland et al. (2001), to estimate species densities from the point count data in each of our study sites. Detection functions should be based on no fewer than 40 observations (Buckland *et al*. 2001), so all species for which there were fewer than 40 observations were grouped with species judged to be of similar detectability for the purposes of estimating detection functions (Table 46). These groups were based on aspects of species' behaviour that affect the probability of birds being seen and heard at different distances from the observer. Note that groups do *not* necessarily comprise ecologically similar species but, rather, species for which we judge that the relationship between probability of detection and distance is similar enough to justify an amalgamated detection function. We used Distance v. 3.5 (Buckland *et al*. 2001) to fit detection functions for combinations of these groups.

Table 46. Groupings of species for estimation of detection functions. The species in each group were treated as being of similar detectability, for the purposes of fitting detection functions in Distance 3.5 (Buckland *et al*. 2001). The size of each group is the combined number of detections of all species in that group. The species in the last row of the table are those for which there were sufficient a number of detections to fit individual detection functions (the number of detections for each species follows the species name, in parentheses).

Group	Size	Group members
1	104	Blackcap, Sedge Warbler, Whitethroat
2	54	Greenfinch, House Sparrow, Linnet, Swallow
3	59	Blue Tit, Stonechat, Whinchat
4	36	Grasshopper Warbler, Chiffchaff, Pheasant, Water Rail
5	50	Hooded Crow, Jackdaw, Kestrel, Magpie, Mistle Thrush, Rook, Starling, Wood Pigeon
6	59	Redpoll, Siskin
7	56	Bullfinch, Cuckoo, Great Tit, Grey Wagtail, Long-tailed Tit, Pied Wagtail, Spotted Flycatcher
Individual detection functions	n.a.	Blackbird (121), Chaffinch (114), Coal Tit (46), Dunnock (77), Goldcrest (75), Meadow Pipit (395), Reed Bunting (56), Robin (186), Skylark (46), Song Thrush (52), Willow Warbler (276), Wren (276)

We selected detection functions principally according to the Akaike information criterion (AIC), an index which combines a measure of goodness of fit with an assessment of a model's parsimony (Buckland *et al*. 2001). However, the form of the detection function at near-zero detection distances has a disproportionate effect on density values and the AIC does not take into account the increased importance of model fit in this critical region (Buckland *et al*. 2001). Only models that provided a close fit to observations from 0-15 m from the observer were therefore considered for selection. These detection functions were used to produce estimates of species densities for every study site.

It was assumed that mapping surveys recorded all birds on a site. Densities were therefore estimated as the mean number of all birds within 12 m of site boundaries recorded during both visits to a site, divided by the area of the site.

7.2.7 Statistical analysis

We analysed bird assemblages at three scales: that of the hedge, that of the site, and that of the site pair, in order to compare planted and unplanted sites. Analysis at the first two scales was carried out for unplanted sites only. In order to make some of the datasets more amenable to analysis, and to better understand the relationships between variables in these datasets, multivariate statistical techniques were used to summarise variation in hedge plant species composition, and bird species composition at both hedge and site scales. These techniques included global non-metric multidimensional scaling analysis (NMS), cluster and indicator analyses, and were carried out using PC-Ord (McCune & Mefford 1997). All other statistical analyses were carried out using SPSS 12.01 (SPSS 2003).

We used NMS to summarise site-level information on bird assemblages, rather than more conventional eigenanalysis techniques (e.g. PCA and DCA), because it has been found to be better at recovering complex gradients (Legendre & Legendre 1998). Ordination analyses were carried out on bird and habitat data across all sites, and on bird data within each bird habitat sub-group identified by cluster analysis (see below). Sørensen (or Bray & Curtis) distance measures were used. The parameter set-up used for the NMS analyses was as follows: 6 initial axes; 50 runs with real data; stability criterion 0.0005; 20 iterations to evaluate stability; maximum number of iterations 500; initial step down of 0.2; random starting coordinates; and 50 randomised runs for Monte Carlo testing. All species occurring in less than 5% of the sites in a particular ordination were excluded from that ordination. The relationships between ordination axes and environmental variables were investigated using correlation.

Flexible-beta cluster analysis (with β=-0.25) was carried out using PC-Ord (McCune & Mefford 1997) in order to identify clusters of sites with similar assemblages of species. Indicator species analysis according to the method described by Dufrêne and Legendre (1997) was used to identify the species that typified bird assemblages in these clusters. An indicator value I was calculated for each species a in a given group of sites g as the product of the relative mean density d of the species in the group (compared to the mean density of the species across all groups), and the proportion of sites n in the group where that species occurred:

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

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

In order to eliminate the effect of hedge length on bird species richness (R) when conducting analyses at the hedge level, R values were standardised for length of hedge (h) by taking residuals from an empirically-derived cubic/quadratic curvilinear regression of form $R = ((0.4357 + 0.0192h - 0.00002h^2)+(0.0914 + 0.026h - 0.00005h^2 + 0.00000004 h^3))/2$. These residuals were added to the predicted value of R for a hypothetical hedge of length 500m. Similarly, plant species richness values were standardised for length of hedge by taking residuals from an empirically-derived quadratic equation of form $R = 5.1459 + 0.0137h – 0.000004h^2$. Bird abundances were standardised for hedge length by dividing them by the length of hedge/100, so that they were expressed as densities per 100m.

When analysing the relationship between the abundance of an individual bird species and either species richness or total bird abundance, the values of the latter two variables were adjusted so that they did not include data from the individual bird species in question. Thus, when the relationship between abundance of a particular bird and overall abundance is described, we are referring to the relationship between abundance of that species and the combined abundance of all *other* species.

Before parametric statistical analyses (i.e. Pearson's correlations or t-tests) were carried out on a set of data, the data were examined to ensure that they conformed with the assumptions of the test. If they did not, either transformations were carried out in order to homogenise variances or normalise the distribution of the data, or non-parametric statistical analyses (i.e. Kendall's correlations or Mann-Whitney tests) were used. Before carrying out t-tests, Levene's test for equality of variance was performed. Only if the result of this test was not statistically significant was equality of variances assumed.

7.3 RESULTS

7.3.1 Hedges

Clusters analysis of hedge plant species data identifies four distinct clusters of hedges (Table 47). Cluster 1 is by far the largest, having more hedges than the other three clusters combined. Hedges in this cluster are typified by the presence of blackthorn, ash, dog rose and ivy, and the absence of typical peatland species heather, bog myrtle and cross-leaved heath, and also gorse. Hedges in cluster 2 are typified by the presence of hawthorn and gorse, and the absence of oak, birch and rhododendron. No plant species were identified as being typically present in cluster 3 hedges, but a low incidence of hawthorn, holly, ivy, blackthorn, rowan, honeysuckle, hazel, elder and sycamore was identified as being characteristic of this group. Although incidence of bramble is high in all clusters (the lowest frequency of occurrence for this species is in cluster 3, where it is found in 85% of hedges), it is identified as being typical of cluster 4, where it occurs in every hedge. Also typical of cluster 4 was the presence of rowan and holly, and the absence of ash and dog rose.

Hedges clusters also differ from one another in several measures of diversity and structure. Mean plant species richness values of the hedges in all four clusters are significantly different from one another, descending from the most species-rich hedges in cluster 1, through clusters 4 and 2, to the least speciose hedges, in cluster 3 (Table 48). Cluster 1 hedges also support more bird-species and higher total numbers of birds than the hedges in the other clusters. Cluster 1 hedges were, on average, the longest, tallest and widest hedges, with the greatest cross-sectional structural complexity (though the latter two measures of structure did not differ significantly between clusters). The shortest hedges, in both height and length, were those in cluster 3. Hedges in cluster 1 had the greatest mean densities of mature and young standard trees, the lowest proportion of gaps, and the highest number of connections with other hedges. They also had the highest scores for structures associated with the hedge, including height of hedge bank, vigour of ditch vegetation, and presence of a grassy verge. Cluster 3 hedges had the lowest mean scores for mature and young standard tree densities, for hedge connectivity and for hedge bank height, but the most gappy hedges were in cluster 2, and the least vegetated ditches were in cluster 4.

Table 47. Indicator species and their indicator values for clusters from cluster analysis of hedge plant presence/absence data. Only species with a maximum IndVal of 25 or more are included. The max IndVal is indicated in bold, and the p-value is the chance of obtaining such a high IndVal at random. Note that the last 18 indicators in the table are the absence of plant species, rather than their presence.

		Cluster 1 (n=198)	Cluster 2 (n=86)	Cluster 3 (n=40)	Cluster 4 (n=36)	p
presence of…	Scientific name					
Blackthorn	*Prunus spinosa*	**66**	5	0	0	0.001
Ash	*Fraxinus excelsior*	**56**	9	0	0	0.001
Rose	*Rosa sp*	**45**	0	1	0	0.001
Ivy	*Hedera helix*	**40**	4	0	16	0.001
Hawthorn	*Crategus monogyna*	34	**40**	0	13	0.001
Gorse	*Ulex europaeus*	7	**27**	22	18	0.006
Holly	*Ilex aquifolium*	12	0	0	**60**	0.001
Rowan	*Sorbus aucuparia*	1	2	0	**49**	0.001
Bramble	*Rubus fruticosus*	26	22	19	**27**	0.003
absence of…						
Heather	*Calluna vulgaris*	**26**	26	25	19	0.001
Bog myrtle	*Myrica gale*	**26**	26	23	23	0.001
Gorse	*Ulex europaeus*	**26**	1	4	7	0.001
Heath	*Erica* sp	**26**	26	26	21	0.002
Oak	*Quercus* sp	22	**27**	26	17	0.035
Birch	*Betula* sp	26	**27**	25	15	0.002
Rhododendron	*Rhododendron ponticum*	25	**26**	26	17	0.002
Hawthorn	*Crataegus monogyna*	0	0	**65**	12	0.001
Ivy	*Hedera helix*	2	22	**41**	10	0.001
Holly	*Ilex aquifolium*	13	33	**38**	0	0.001
Blackthorn	*Prunus spinosa*	1	20	**35**	33	0.001
Rowan	*Sorbus aucuparia*	28	24	**32**	4	0.001
Honeysuckle	*Lonicera periclymenum*	14	28	**29**	10	0.001
Hazel	*Corylus avellana*	15	24	**28**	22	0.004
Elder	*Sambucus nigra*	18	26	**28**	20	0.008
Sycamore	*Acer pseudoplatanus*	19	25	**26**	25	0.046
Ash	*Fraxinus excelsior*	1	16	32	**35**	0.001
Rose	*Rosa canina*	5	27	26	**29**	0.001

Table 48. Differences in plant and bird species richness, bird abundance, hedge length in m, and indices of hedge width, cross-sectional structure, density of mature and young standards, % gaps, connectivity to other hedges and to woodland, height of hedgebank, density of ditch vegetation and presence of a grassy verge. Means ± standard errors are given for each cluster, annotated with letters that the cluster is significantly different from all other clusters that do not share the same letter. F values and degrees of freedom within groups are adjusted according to Welch's robust test for equality of means, to allow for unequal homogeneity of variance between groups.

Hedge parameter	Cluster 1 (n=198)	Cluster 2 (n=86)	Cluster 3 (n=40)	Cluster 4 (n=36)	F $_{(\text{d.f1, d.f. 2})}$	p
Plant R	11.2±0.14^{a}	7.4±0.21^{b}	6.3±0.18^{c}	9.9±0.26^{d}	191.1$_{(3,110)}$	<0.0001
Bird R	6.5±0.17^{a}	5.1±0.21^{b}	6.2±0.28^{ac}	5.8±0.29^{abc}	9.1$_{(3,106)}$	<0.0001
Bird Abund	3.7±0.2^{a}	2.3±0.25^{b}	2.9±0.53^{ab}	3.4±0.42^{ab}	5.9$_{(3,98)}$	0.0009
Length	148.9±7.46^{a}	123.9±7.73^{ab}	67.3±7.1^{c}	104.4±8.23^{b}	22.1$_{(3,124)}$	<0.0001
Height	3±0.06^{a}	2.5±0.08^{b}	2.3±0.13^{b}	2.6±0.16^{ab}	16.2$_{(3,95)}$	<0.0001
Width	3.3±0.06	3.2±0.09	3.1±0.13	3.2±0.11	0.7$_{(3,100)}$	0.5570
CS	3.3±0.05	3.2±0.1	3.2±0.11	3±0.14	1.2$_{(3,93)}$	0.3025
Mature	1.3±0.11^{a}	0.6±0.13^{b}	0.2±0.12^{b}	0.5±0.18^{b}	13.6$_{(3,114)}$	<0.0001
Young	1.8±0.13^{a}	1.6±0.18^{ab}	0.6±0.2^{c}	0.9±0.22^{bc}	10.4$_{(3,108)}$	<0.0001
Gaps	2.7±0.06^{a}	2.1±0.12^{b}	2.4±0.17^{ab}	2.3±0.16^{ab}	7.4$_{(3,93)}$	0.0002
Hedge con	3.1±0.09^{a}	2.5±0.16^{b}	1.4±0.2^{c}	2.9±0.2^{ab}	20.9$_{(3,99)}$	<0.0001
Wood con	0.3±0.05	0.5±0.08	0.3±0.1	0.3±0.11	0.8$_{(3,101)}$	0.4729
Bank	1.9±0.07^{a}	1.7±0.11^{ab}	1.2±0.17^{b}	1.7±0.22^{ab}	6.5$_{(3,93)}$	0.0005
Ditch	2.1±0.08^{a}	1.3±0.14^{b}	1.5±0.19^{ab}	0.8±0.19^{b}	16.1$_{(3,98)}$	<0.0001
Verge	0.4±0.04^{a}	0.1±0.04^{b}	0.1±0.03^{b}	0.1±0.04^{b}	13$_{(3,133)}$	<0.0001

Several of the structural variables that are highest in cluster 1, as well as some that did not differ significantly between clusters, are also correlated with bird species richness and total abundance within 10m of hedges. In cluster 1, bird species richness is positively correlated with hedge height (Pearson's R = 0.47, n = 198, p < 0.0001) and width (Pearson's R = 0.31, n = 198, p < 0.0001), with density of mature standards (Pearson's R = 0.16, n = 198, p = 0.026) and plant species richness (Pearson's R = 0.19, n = 198, p = 0.006), and is negatively correlated with percentage gaps (Pearson's R = 0.16, n = 198, p = 0.020). In the same cluster, abundance of birds is also positively correlated with height (Pearson's R = 0.42, n = 198, p <0.0001) and width (Pearson's R = 0.29, n = 198, p <0.0001) of hedge, and negatively correlated with proportion of gap (Pearson's R = 0.23, n = 198, p = 0.001). Fewer structural variables are correlated with either bird species richness or abundance in the other clusters. In cluster 2, plant species richness is positively correlated with both bird species richness (Pearson's R = 0.42, n = 86, p < 0.0001) and abundance (Pearson's R = 0.38, n = 86, p = 0.0004). In cluster 3, bird species richness and abundance are positively correlated with both hedge height (Pearson's R = 0.41, n = 40, p = 0.009, and Pearson's R = 0.50, n = 40, p = 0.001, respectively) and hedge width (Pearson's R = 0.38, n = 40, p = 0.015, and Pearson's R = 0.43, n = 40, p =0.006, respectively). In cluster 4, hedge height is the only structural variable correlated with either bird species richness (Pearson's R = 0.35, n = 36, p = 0.039) or abundance (Pearson's R = 0.36, n = 36, p = 0.031). The only plant species whose presence was positively correlated with species richness and abundance of birds in more than one cluster was ivy: with species richness in cluster 1, Pearson's R = 0.20, n = 198, p = 0.004; with abundance in cluster 1, Pearson's R = 0.15, n = 198, p = 0.033; with species richness in cluster 4, Pearson's R = 0.45, n = 36, p = 0.006; and with abundance in cluster 4, Pearson's R = 0.46, n = 36, p = 0.004.

7.3.2 Habitats

A total of 46 bird species were recorded during mapping surveys (Appendix 8). The number of species recorded in a single site ranged between 5 and 28, and the estimated total density of birds in a site ranged from 0.3 to 10.7 birds/ha.

NMS ordination of bird density data from mapping surveys generated two axes, which together accounted for 94% of the variation in species densities between sites (Figure 21). Axis 1, which accounted for 66% of the variation in the original data, was strongly correlated with bird species richness and less strongly correlated with total abundance of birds. Peatland sites tended to have the lowest values, improved grassland sites had the highest values, and wet grassland sites were intermediate in value between the former two groupings. Axis 2, which accounted for 28% of the variation in the original data, was less strongly correlated with species richness than Axis 1, but was more strongly correlated with total bird abundance . Sites with relatively high axis scores generally had low shrub and tree cover. Non-peatland sites with high values tended to be more heavily grazed than sites with lower values.

Cluster analysis separates sites into five groups (Figure 21). Four of the clusters are comprised exclusively of one habitat type: sites in clusters 4 and 5 are peatland, sites in cluster 2 are wet grassland and cluster 1 is comprised of improved grassland sites. Cluster 3 includes sites of all three habitat types. Sites in cluster 3 are less typical representatives of their habitat types than those in the other four clusters. They all have higher Axis 2 values, were more heavily grazed, and have lower shrub and tree cover than sites in cluster 2, but are less agriculturally improved and of more upland character than sites in cluster 1. The only peatland site in cluster 3 is the most agriculturally improved of all the peatland sites, being the only one that was comprised wholly of enclosed land.

Indicator analysis of these five groups identified eight species (Blackbird, Blue Tit, Coal Tit, Goldcrest, Magpie, Robin, Wren and Willow Warbler) as being typical of cluster 2, and Meadow Pipit and Skylark as being typical of cluster 4 (Table 49). Although no species were identified as being typical of clusters 1, 3 or 5, cluster analysis of presence/absence data identified the absence of Skylark and Snipe as being characteristic of cluster 1.

There is substantial variation in size between the sites in different clusters, with sites in cluster 2 being the smallest, and sites in cluster 5 the largest (Figure 22). The proportions of cover types in sites also varies between clusters (Figure 23). Sites in cluster 2 have the lowest proportion of open land, with approximately two thirds of the site under hedge cover or other tree and shrub cover. Sites in clusters 3 and 1 have a progressively larger proportion of open land, a smaller area of land within 10 m of a hedge and almost no other tree and shrub cover. Sites in clusters 4 and 5 have no hedges and negligible other tree and shrub cover, so are effectively 100% open land. Species richness is not significantly different between any of the three predominantly grassland clusters, or between the two peatland clusters; but is much higher in the former than in the latter (Figure 24). Total bird abundance is significantly lower in clusters 1 and 5 than in clusters 2, 3 and 4, and highest of all in cluster in 2 (Figure 25). Only the sites in clusters 1-3 were on enclosed land, and therefore had hedges. In these sites, bird densities within 12m of hedges averaged about 10 birds/ha for cluster 2, just over 4 birds/ha for cluster 1, and an intermediate value for cluster 3 (Figure 26). Densities of birds in other tree and shrub cover descended progressively from 40 birds/ha in cluster 1 to just under 5 birds/ha in cluster 5 (Figure 27). However, as only sites in clusters 2 and 3 have more than 1% other tree and shrub cover (Figure 23), the potential for this variable to influence the bird assemblages in other site clusters is limited. Densities of birds in open land are generally lower than in the vicinity of hedges or other types of cover, ranging from a mean of less than 1 bird/ha in clusters 1 and 5, to between 3 and 5 times this density in clusters 2 (with the highest bird density in open pasture), 3 and 4 (Figure 28).

Table 49. Indicator species and their indicator values for clusters from bird mapping survey data. Only species with a maximum IndVal of 25 or more are included. The max IndVal is indicated in bold, and the p-value is the chance of obtaining such a high IndVal at random. Note that the last two indicators in the table, both for cluster 1, are the absence of Skylark and Snipe, rather than presence or abundance.

Species	Cluster 1 n = 7	Cluster 2 n = 7	Cluster 3 n = 6	Cluster 4 n = 5	Cluster 5 n = 2	P value
presence of…						
Blackbird	17	**56**	26	0	0	0.002
Blue Tit	26	**65**	6	0	0	0.001
Coal Tit	2	**62**	24	1	0	0.001
Goldcrest	14	**64**	16	1	0	0.006
Magpie	8	**52**	3	0	0	0.009
Robin	17	**52**	30	0	1	0.049
Wren	19	**56**	23	1	0	0.003
Willow Warbler	3	**71**	21	0	1	0.002
Meadow Pipit	2	11	31	**45**	9	0.021
Skylark	0	2	10	**54**	23	0.021
absence of…						
Skylark	**39**	27	6	0	0	0.002
Snipe	**32**	17	14	1	8	0.007

Across all sites, bird species richness and total abundance are correlated with several elements of shrub and tree cover (Table 50). Both bird species richness and total abundance are positively correlated with total length of large hedges, and bird species richness is also positively correlated with total length of medium hedges. Neither variable is correlated with length of small hedges. Total area of treelines and semi-natural woodland are also correlated with total bird abundance, though only area of treelines was also correlated with bird species richness. Neither bird species richness nor total abundance were correlated with non-hedge shrub cover, or young or mature plantation forest cover.

Table 50. Pearson's R correlation values between bird species richness (R) and total abundance values from bird mapping survey of unplanted sites, and 9 types of tree and shrub cover. Out of a total of 18 correlations, 8 were significant.

Variable	N	R Pearson's R	R p	abundance Pearson's R	abundance p
Small hedge	25	0.29	0.1662	0.02	0.9262
Medium hedge	25	0.55	0.0044	0.35	0.0892
Large hedge	25	0.61	0.0011	0.52	0.0075
Total hedge	27	0.74	<0.0005	0.54	0.0037
Shrub	24	0.27	0.1995	0.32	0.1290
Treeline	26	0.49	0.0116	0.40	0.0425
Woodland	25	0.31	0.1310	0.52	0.0076
Mature plantation	25	-0.01	0.9590	0.13	0.5507
Young plantation	24	-0.21	0.3365	-0.13	0.5368

Across all unplanted sites, the abundances of Blackbird, Blue Tit, Chiffchaff, Chaffinch, Coal Tit, Dunnock, Goldcrest, Robin, Song Thrush, Wren and Willow Warbler were strongly and positively correlated with overall bird abundance (Table 51). The abundances of Blackbird, Blue Tit, Chaffinch, Dunnock, Robin, Wren and Willow Warbler and more weakly correlated with species richness. Abundance of Skylark is strongly negatively correlated with bird species richness (Table 51).

Table 51. Pearson's R correlation values between bird species richness and total abundance values from bird mapping survey of 27 unplanted sites, and individual abundances of 21 species of birds. Out of a total of 92 correlations, 35 were significant.

Species	R Pearson's R	R p	abundance Pearson's R	abundance p
Blackbird	0.49	0.0099	0.86	<0.0001
Blackcap	0.11	0.5713	0.46	0.0152
Bullfinch	0.30	0.1305	0.64	0.0004
Blue Tit	0.48	0.0108	0.76	<0.0001
Chiffchaff	0.34	0.0858	0.71	<0.0001
Collared Dove	0.27	0.1746	0.53	0.0048
Chaffinch	0.58	0.0016	0.72	<0.0001
Coal Tit	0.36	0.0650	0.70	<0.0001
Dunnock	0.44	0.0233	0.68	0.0001
Goldcrest	0.36	0.0613	0.75	<0.0001
Greenfinch	0.30	0.1223	0.61	0.0008
Great Tit	0.34	0.0848	0.50	0.0081
Meadow Pipit	-0.37	0.0605	-0.36	0.0615
Pheasant	0.44	0.0205	0.57	0.0021
Robin	0.57	0.0021	0.85	<0.0001
Skylark	-0.62	0.0005	-0.37	0.0577
Spotted Flycatcher	0.28	0.1649	0.62	0.0006
Song Thrush	0.35	0.0700	0.70	<0.0001
Wood Pigeon	0.32	0.1071	0.52	0.0055
Wren	0.57	0.0021	0.87	<0.0001
Willow Warbler	0.42	0.0277	0.74	<0.0001

7.3.3 Afforestation

A total of 42 bird species were recorded during point counts. The number of species recorded in a single site ranged from 0 to 22, and the estimated total abundance of birds on a site ranged from 0 to 12.83 birds per ha. For sites that were surveyed by both methods, estimates of species richness were consistently lower than those derived from mapping surveys (in only one site were these two estimates the same, and in no sites were more species recorded during point counts than in the mapping survey). Density estimates from point count data tended to be higher than those derived from mapping surveys. Relative to estimates of density derived from mapping surveys, estimates derived from point counts tended to be highest in improved grassland sites and lowest in peatland sites.

NMS ordination of bird density data from point counts generated two axes, which together accounted for 90% of the variation in species densities between sites, separating them in a similar way to the ordination of mapping data (Figure 29). Again, axis 1, which accounted for 71.8% of the variation in the original data, is highly correlated with bird species richness, while Axis 2, which accounted for only 18.1% of the variation in the original data, is highly correlated with total bird abundance. Unplanted sites tended to have more extreme values for Axis 1 and lower values for Axis 2 than planted sites.

Of 18 bird species that are significantly correlated with plant species in the shrub layer, 15 are correlated with bramble cover, 14 are correlated with heather (*Calluna vulgaris*) cover, and 16 are correlated with hawthorn cover (Table 52). Of 15 that are correlated with species in the canopy layer, 12 are correlated with cover of ash and 9 are correlated with hawthorn cover (Table 53). The abundance of most of these bird species is positively related to bramble, hawthorn and ash cover, and negatively correlated with heather cover. However, the relationships between these elements of vegetation cover and abundances of Meadow Pipit, Redpoll, Skylark, and Stonechat are in the opposite directions.

Table 52. Kendall's correlation coefficients and significance levels for relationship between abundances of individual birds species and cover of three shrub species (less than or equal to 2 m) at the site level. N = 49. Out of 146 tests carried out, 53 significant results.

	Bramble		Heather		Hawthorn	
Species	τb	p	τb	p	τb	p
Blackbird	0.45	0.0000	-0.56	0.0000	0.65	0.0000
Blackcap	0.27	0.0196	-0.27	0.0380	0.28	0.0188
Chiffchaff	0.31	0.0086	-0.24	0.0662	0.38	0.0017
Chaffinch	0.17	0.1171	-0.46	0.0002	0.38	0.0006
Coal Tit	0.25	0.0229	-0.41	0.0013	0.34	0.0028
Dunnock	0.38	0.0004	-0.26	0.0326	0.35	0.0019
Goldcrest	0.35	0.0012	-0.33	0.0077	0.40	0.0003
Great Tit	0.15	0.2078	-0.28	0.0366	0.22	0.0709
Redpoll	-0.06	0.5720	0.24	0.0658	-0.24	0.0402
Mistle Thrush	0.27	0.0263	-0.18	0.1892	0.30	0.0151
Meadow Pipit	-0.29	0.0040	0.42	0.0002	-0.39	0.0002
Robin	0.48	0.0000	-0.55	0.0000	0.55	0.0000
Reed Bunting	0.25	0.0225	-0.19	0.1197	0.11	0.3127
Skylark	-0.12	0.2807	0.18	0.1580	-0.21	0.0649
Stonechat	-0.32	0.0047	0.44	0.0007	-0.33	0.0052
Song Thrush	0.38	0.0008	-0.39	0.0019	0.39	0.0007
Whitethroat	0.32	0.0040	-0.25	0.0455	0.24	0.0388
Wren	0.51	0.0000	-0.39	0.0007	0.46	0.0000
Willow Warbler	0.51	0.0000	-0.35	0.0023	0.46	0.0000
Abundance	0.48	0.0000	-0.44	0.0001	0.51	0.0000
R	0.46	0.0000	-0.49	0.0000	0.49	0.0000
Axis 1	-0.39	0.0001	0.53	0.0000	-0.53	0.0000
Axis 2	0.41	0.0000	-0.27	0.0151	0.41	0.0001

Table 53. Kendall's correlation coefficients and significance levels for relationship between abundances of individual birds species and cover of three canopy species (greater than 2 m) at the site level. N=48. Out of 100 tests carried out, 27 statistically significant results.

	Ash		Hawthorn	
Species	τb	p	τb	p
Blackbird	0.41	0.0003	0.41	0.0004
Blackcap	0.31	0.0133	0.34	0.0076
Chiffchaff	0.14	0.2582	0.18	0.1535
Chaffinch	0.37	0.0013	0.31	0.0083
Coal Tit	0.37	0.0020	0.27	0.0270
Dunnock	0.42	0.0004	0.16	0.1864
Goldcrest	0.19	0.1006	0.25	0.0328
Great Tit	0.29	0.0242	0.18	0.1791
Redpoll	-0.04	0.7748	-0.33	0.0088
Mistle Thrush	0.24	0.0658	0.04	0.7744
Meadow Pipit	-0.26	0.0209	-0.44	0.0001
Robin	0.45	0.0001	0.31	0.0070
Reed Bunting	0.15	0.1994	-0.17	0.1497
Skylark	-0.36	0.0030	-0.22	0.0784
Stonechat	-0.19	0.1296	-0.25	0.0466
Song Thrush	0.27	0.0270	0.09	0.4482
Whitethroat	0.13	0.2717	-0.01	0.9369
Wren	0.24	0.0303	0.14	0.2125
Willow Warbler	0.47	0.0000	0.09	0.4171
Abundance	0.47	0.0000	0.16	0.1406
R	0.38	0.0006	0.23	0.0427
Axis 1	-0.36	0.0009	-0.47	0.0000
Axis 2	0.39	0.0003	0.03	0.7775

Total shrub cover, bird species richness, total abundance of birds, and axis 2 scores (but not axis 1) were significantly greater in planted sites than in their unplanted pairs (Table 54). The abundances of eleven bird species also differed significantly between afforested and open sites within pairs. These birds were: Dunnock, Grasshopper Warbler, Meadow Pipit, Redpoll, Reed Bunting, Sedge Warbler, Skylark, Song Thrush, Whitethroat, Willow Warbler and Wren. All of these species, except Skylark, were more abundant in afforested sites.

The five species that show the greatest proportional difference in abundance between planted and unplanted sites are Grasshopper Warbler, Reed Bunting, Sedge Warbler, Whitethroat and Willow Warbler. The difference between planted and unplanted sites in combined abundance of these species is significantly greater in clusters 1 and 3 than in cluster 2 and clusters 4 and 5 combined. Kruskal Wallis H = 10.82, d.f. = 3, p = 0.01). There is a greater than three-fold increase in the combined abundance of these species, between sites with less than 1% bramble cover and sites with 2% bramble cover (Figure 30). In both cluster 1 and cluster 3, the unplanted sites have less than 1% bramble cover and the planted sites have more than 2% bramble cover (). The mean bramble cover in both unplanted and planted sites in cluster 2 is more than 2%, whereas both unplanted and planted sites in cluster 4 have an average of less than 1% bramble cover.

Table 54. Differences between unplanted and planted sites in ordination axis scores, bird species richness and abundance, total shrub cover and individual species abundances. Paired t-tests evaluate whether mean difference in values of the above variables is significantly different from zero. N = 23 for all groups (no birds were detected within 50 m of point counts in one of the unplanted sites – Togu - so it was not included in these paired t-tests).

	Mean planted - unplanted	s.e.	t	df	p
Axis 2	0.511	0.089	5.73	22	<0.0001
Abundance	1.907	0.553	3.45	22	0.0023
R	2.478	0.727	3.41	22	0.0025
Total shrub cover	4.681	1.288	3.64	22	0.0015
Sedge Warbler	0.164	0.043	3.82	22	0.0009
Whitethroat	0.185	0.060	3.06	22	0.0058
Dunnock	0.166	0.063	2.61	22	0.0158
Grasshopper Warbler	0.175	0.058	2.99	22	0.0068
Redpoll	0.206	0.071	2.91	22	0.0082
Meadow Pipit	0.269	0.105	2.58	22	0.0172
Reed Bunting	0.119	0.031	3.89	22	0.0008
Skylark	-0.063	0.022	-2.85	22	0.0094
Song Thrush	0.123	0.056	2.21	22	0.0376
Wren	0.163	0.064	2.54	22	0.0186
Willow Warbler	0.500	0.096	5.21	22	<0.0001

7.4 DISCUSSION

7.4.1 Birds and Hedges

Hedges that supported the highest densities of bird species and individuals were typified by blackthorn, ash, rose and ivy. These hedges tended to be tall and wide, have a high density of standard trees, have a low proportion of gaps and be well connected to other hedges. They also tended to be situated on high banks, and to have well-developed ditch vegetation and grassy verges. Another detailed (though smaller scale) study on the relationship between hedge bird assemblages and hedge characteristics, in southern England (Macdonald & Johnson 1995), also found that bird species richness was positively correlated with height, width, number of woody species, presence of a ditch and density of mature standard trees. Within this group (as well as within some of the other clusters) bird species richness and/or abundance was also related to hedge height, width, density of mature standards, plant species richness and occurrence of ivy, and negatively related to proportion of gaps. Common themes uniting some or all of these variables include structural diversity in and around the hedge, and the provision of cover. More structurally diverse hedges provide niches for a wider variety of species than smaller or more uniform hedges, and dense cover is required by some species for nesting, and preferred by many others for foraging and movement. Structurally diverse and species-rich field boundary vegetation can also have a positive effect on the food supply of both granivorous and insectivorous birds (Marshall & Moonen 2002). As well as influencing the birds occupying a hedge by providing food and cover, ditch vegetation may function as a surrogate of grazing intensity, which could influence other aspects of hedge structure and of vegetation cover in the rest of the adjoining area. Over-grazing has been identified as a causal factor in bird population declines (Newton 2004). As well as limiting shrub cover, high levels of grazing pressure can impact bird assemblages through its impact on the sward. While short, uniform swards are preferred by some open habitat specialists, increased defoliation will reduce the availability of invertebrate and plant food, while increased trampling by stock will result will destroy nests and young of ground nesting birds (Vickery *et al*. 2001).

Moles and Breen (1995) found that total area of field boundary was not a good predictor of species richness or abundance, but that variation in hedge structure correlated with variation in the abundance of common species, height and width being among the most important variables. In contrast, we found that hedge length was a good predictor of species richness at the site level. However, analysing large, medium and small hedges separately, we found that a relationship between site species richness and hedge length persisted only for medium and large hedges. It may be that the majority of hedges studied by Moles and Breen (1995) were of insufficient size for a relationship between species richness and total boundary length to be significant. Macdonald and Johnson (1995) also found that length of hedge had no bearing on the number of bird species in it, despite the fact that it was correlated with abundance, and many of the other hedge variables that we also found to be related to bird species richness. In contrast, we found that hedge length was strongly related to bird species richness at the level of the hedge. Studies should not, therefore, assume that there is no need to control for length when investigating the number of bird species supported by hedges.

When farmland sites are being planted with trees, it is likely that the bird assemblages of the resulting plantations will be diversified if hedges can be retained in the long-term. Although many of the bird species that use hedges will also be able to breed in conifer forest, others require low, dense shrub cover, often scarce in forest plantations, or show a preference for areas with broadleaved tree and shrub species (Iremonger *et al*. 2006). As well as providing habitat for birds directly, retained hedges might also act as sources from which shrubs and trees can colonise areas of the plantation that receive sufficient sunlight for an understorey to develop. In order to be successfully retained, a sufficient gap must left around hedges so that they do not become shaded out as the forest grows (Iremonger *et al*. 2006). If still larger open spaces can be left unplanted around some areas of hedge, this may increase both horizontal and vertical structural diversity around the hedge, which is likely to be of benefit to several different bird species.

7.4.2 Birds and Habitat

In sites belonging to all clusters, the densities attained by birds in areas of tree and shrub cover, and especially in hedges, were far higher than in areas of open pasture. Most of the bird species encountered in this survey were not specialists of open habitat. Moreover, forest specialists typically occur at higher densities in habitats with high levels of tree and shrub cover than grassland and peatland specialists do in open habitats (Lack 1933; Moss *et al*. 1979). However, open pasture accounted for over 50% of site area in all clusters bar cluster 2. Thus, although the density of birds in open peatland habitat in cluster 4 sites is not as high as in areas of hedge or other cover in cluster 1, it is over three times higher than the density of birds in open pasture in cluster 1, and cover of open land is sufficiently dominant in both of these clusters that cluster 4 has the higher overall bird abundance. Even a relatively small increase in the carrying capacity for birds of the open land element of afforestation sites for birds is therefore likely to have a large overall increase on site bird abundances, especially in sites with relatively low areas of hedge or other types of shrub and tree cover, and those with particularly low densities of birds in areas of pasture. Berg (2002) found that "residual habitats", such as shrubby areas and natural grassland, could have a disproportionate influence on the bird assemblages of farmland dominated landscapes in Sweden, despite occupying only about 10% of the land area.

Of the three clusters with hedges, the density of birds found within 10 m of hedges is highest in cluster 2, and lowest in cluster 1. This is probably due to differences in the structure of the hedges between each of these clusters. Whereas almost half of the hedges in cluster 1 were classed as small, less than a third of the hedges in cluster 3 and less than a fifth of those in cluster 1 fell into this category. The differences in the carrying capacity of hedge habitat between these sites were further increased by the total amount of hedge habitat available – over twice as much in cluster 2 and in cluster 1, with the area of hedge habitat in cluster 3 being intermediate. The density of birds in non-hedge tree and shrub cover was higher in cluster 1 than in any other cluster, and nearly 10 times as high as the density of birds within 10 m of cluster 1 hedges. However, the effect of this cover type on overall bird abundances in cluster 1 sites was negligible, due to the tiny proportion of site area it occupied. It is likely that the most proximate limit on density of birds in areas of intensive pasture, such as are typified by cluster 1 sites, is a lack of suitable cover for generalist and woodland birds. If the intensity of production were to be relaxed even slightly, allowing the dimensions of hedges to increase, or small patches of shrubs and trees to develop, this would probably have a noticeable and positive effect on the numbers of woodland and generalist birds in this type of farmland. The effect of afforestation would likely be a similar (if more dramatic) one in the long term, with additional short-term benefits to bird communities of open pasture during the tree establishment phase, resulting from a relaxation of grazing pressure.

Part of the reason that species richness is less variable among grassland sites than overall abundance is that species richness values at the site level were not standardised for site area. The lack of a significant relationship between species richness and site area was due to an inverse relationship between site size and bird densities. Sites in cluster 2, which generally had the most birds, were also the smallest sites; while by far the largest unplanted site (Togu) also had the lowest overall abundance of birds. It is likely that the lack of a significant relationship between species richness and site area is a consequence of this. However, the abundances of six of the nine bird species identified as being indicators of cluster 2 are also positively correlated with site species richness. This indicates that, if variation in species richness due to area were standardised, cluster 2 sites would be more species-rich than sites in other clusters. All of these species are relatively common in a typical Irish agricultural landscape and occur in a wide variety of habitats, though they reach their highest densities in habitats with at least an element of tree cover. With the exception of Blue Tit, all these species are typically abundant in conifer plantations. The abundances of both Meadow Pipit and Skylark, the two species identified as indicators of cluster 4, were negatively correlated with site species richness (the relationship for the latter species was far stronger than for the former). Elements of tree cover such as patches of woodland and large hedges, which serve to increase the suitability of sites to a wide range of generalist and woodland bird species, may have the opposite effect on open habitat specialists such as these. Sparks *et al.* (1996) caution that while large hedges are beneficial for many species (especially those typical of woodland habitats), their presence may be detrimental to open habitat species such as skylarks.

7.4.3 Surveying Methodologies

Densities derived from the point count data tend to be higher than those generated by the bird mapping surveys. This is probably because the point counts were biased towards collecting data from parts of the site with more birds (i.e. field boundaries). This was necessary in order to maximise the chances of detecting as many of the species present on the site as possible from just six counting positions. Even with the majority of points situated close to hedges and other types of shrub or tree cover, species richness values from point count data are generally lower than those derived from mapping surveys. This is because the probability of detecting species that are present at very low densities is usually small, especially considering that all birds more than 50 m from the observer were not recorded.

The discrepancy between density estimates from point counts and mapping surveys is greatest in improved grassland sites, which typically have a low density of relatively bird-rich field boundaries. Conversely, in peatland sites, which typically lack field boundaries, or have boundaries that do not support very high densities of birds in relation to the rest of the site, density estimates from point counts tended to be slightly lower than those derived from the mapping survey, possibly as a result of observer avoidance (the lack of cover in these sites made the observer more obvious to the birds around him). Density estimates from point counts and mapping surveys in wet grassland sites with small field sizes and relatively high cover of shrubs and trees tended to be more similar.

Territory mapping is recognised as generating much more accurate estimates of bird populations than distance sampling methodologies (Bibby *et al.* 2000). However, because multiple visits are required to confirm territories, it is much more time-consuming, and its application is therefore restricted to comparatively small areas. The mapping survey protocol described here is not as labour-intensive as territory mapping, the smaller number of visits allowing a much larger area to be covered during a given amount of time. Point counts have been recommended over line transects when surveying birds in 'patchy' environments, areas where movement of the observer is impeded by topography or dense vegetation, and habitats where a high percentage of bird registrations will be aural rather than visual (Bibby *et al.* 2000; Buckland *et al.* 2001). Some of the same advantages apply to a comparison with mapping surveys, though because mapping surveys are more flexible with regard to the route of the surveyor, and because detection functions are not an issue, they may be more suited than line transects to surveying in patchy environments and areas in which most detections are made aurally. In fact, in such environments, population estimates derived from mapping surveys may be more accurate than those derived from point counts, for either of two reasons. Firstly, in a heterogenous environment, the detectability of a bird will depend on its species as well as on the habitat it is occupying. However, unless the number of point counts conducted is very large, few bird species will be sampled frequently enough in all habitats they occur in to enable a separate detection function to be derived for each combination of species and habitat. Even in homogenous habitats, a rare species will often be detected too infrequently to derive a detection function from detections of that species alone. Species and habitats can be grouped, as they were in this study, but this inevitably decreases the fit of detection functions to the data, and introduces greater error to estimates of abundance. Secondly, when employing point counts in a patchy environment, the relative frequency of different habitat types must be taken into account; either by setting the number of counts conducted in each habitat type according to the frequency of that habitat in the study area, or by weighting the influence of individual points on estimates of total abundance. This may be relatively straightforward when patch size is large enough that each point count consists of only one habitat type, but when the area covered by a point count can incorporate several different habitat types, controlling for relative frequency of different habitats is more complex. We therefore recommend mapping surveys over point counts in patchy environments. Exceptions include situations where terrain and vegetation greatly restrict movement within the site.

7.4.4 Birds and Afforestation

Previous studies of breeding bird assemblages of farmland in Ireland have found the five commonest species to be Wren, Robin, Blackbird, Dunnock and Chaffinch (Lysaght 1989; Moles & Breen 1995). These studies were in sites at lower altitude, and in more intensively agricultural landscapes than were most of our study sites. Nevertheless, three of these five birds (Wren, Robin and Blackbird) are among the five commonest species found in both our planted sites and our unplanted sites, while Dunnock and Chaffinch are among the ten most abundant species. The other two most common species in our study were Willow Warbler and Meadow Pipit, reflecting the wider availability of scrub and more upland nature of our sites, compared to most lowland agricultural areas.

We found that post-afforestation sites held a significantly higher number and diversity of birds than do pre-planting open habitats. Moss *et al.* (1979) also found that conifer plantations support very high densities of songbirds. This is consistent with the response of most passerines to afforestation being a positive one, as the chance of a species being numerous enough to be sampled in a site would be increased. If this was the case, an increase in abundance would be expected across many species, but especially in the most numerous species, as these are more frequently sampled, and so estimation of their abundance is likely to be more accurate than for relatively scarce species. However, six of the ten species that were commonest in uplanted sites were not more numerous in planted than in unplanted sites. These include five of the six species that have been found by studies in Ireland and the UK to be the most numerous bird species in older conifer plantations (Moss *et al*. 1979; Wilson *et al*. In Press). This suggests that the increase in species richness and total abundance of birds in the first five years after afforestation is neither due to a general increase in abundance of all species, nor to elevated numbers of those species which are most common in later stages of the forest cycle.

Of the eleven bird species whose abundance was significantly different between unplanted and planted sites, only Skylark was less abundant in planted sites. Of all the species of open habitat recorded during this study, this is the most strongly associated with farmland. A study of declining birds in the UK showed that over 90% of skylarks were found in pastoral and arable land (Gregory & Baillie 1998), with the highest densities recorded in upland pasture. A study of the effects of afforestation on upland bird communities found that Skylark was the first species to disappear from sites after planting with trees, declining dramatically in density within the first few years after planting, and not persisting beyond the tree establishment stage (Moss *et al*. 1979). Newton *et al.* (1999) suggest that the decline of Skylarks in Ireland is probably linked to the intensification of agriculture, but afforestation of upland heath and grassland could also result in loss of habitat for this species.

Skylark is by no means the only Irish bird species for which unafforested open habitat is more or less requisite. Snipe, Water Rail and Wheatear were all recorded during mapping surveys. We did not record them from afforested sites, but neither did we record them during point counts of unplanted sites. Given the low numbers we recorded during mapping surveys, their absence from planted sites is not evidence of displacement by afforestation. However, all three species are strongly associated with open habitats (Snow & Perrins 1998), and it is unlikely that any of them make extensive use of even very young plantations. Other Irish species not recorded by us, which have been shown by studies in the UK to avoid even very young plantations, include Redshank *Tringa totanus*, Lapwing *Vanellus vanellus*, Golden Plover *Pluvialis apricaria*, Dunlin *Calidris alpina* and Red Grouse *Lagopus lagopus* (Hancock & Avery 1998; Lack 1933; Thompson *et al*. 1988). The effect of afforestation on open habitat birds is not necessarily restricted to the area planted with trees. Several studies have documented negative impacts of forest plantations on birds of adjoining habitats, either through avoidance by some bird species, or through increased rates of adult and nest predation (Stroud *et al*. 1990; Thompson *et al*. 1988). Other species make extensive use of young forest plantations but abandon them at or before canopy closure. These include Meadow Pipit, Grasshopper Warbler, Sedge Warbler, Whitethroat, Stonechat, Whinchat and Reed Bunting, all of which we recorded during this study, and Hen Harrier *Circus cyaneus*, which we did not.

Of the seventeen species that avoid forest habitat mentioned in the previous two paragraph, eleven are considered endangered or threatened in Ireland (Newton *et al*. 1999), and all but

Meadow Pipit are either relatively uncommon or declining breeders in the Irish agricultural landscape. Clearly, from an ornithological perspective, it would be better if areas where these species are most abundant are not afforested. Such areas will tend to be comprised either of peatland or of wet grassland habitats. The absence of Skylark and Snipe, which are two of the more widespread of the above species (though both are on the Irish Amber list), was found by this study to be an indicator for cluster 1, which was wholly comprised of improved grassland sites. This is not to say that all peatland and wet grassland sites will support sufficient numbers of these species to make a compelling reason to prevent them from being afforested. However, the bird assemblage of any peatland or wet grassland site proposed for afforestation should be examined to ensure that important open habitat species are not negatively affected. This could be done directly, by surveying the birds present on the site, or indirectly, by assessing the presence and prevalence of indicators.

The five species showing the greatest proportional difference in abundance between planted and unplanted sites (Grasshopper Warbler, Reed Bunting, Sedge Warbler, Whitethroat and Willow Warbler) are all, with the exception of Willow Warbler, relatively uncommon in the Irish agricultural landscape, and the Grasshopper Warbler is on the Irish Amber list of threatened bird species (Newton *et al.* 1999). An increase in abundance of this group of species will often reflect an increase bird diversity on at least a local scale. The difference in abundance of these species between planted and unplanted sites is greater in grassland sites with low or intermediate shrub and tree cover than in peatland sites or in wet grassland with high shrub and tree cover. One possible reason for this is bramble cover. The combined abundance of the bird species in question appears to be closely linked to bramble cover, being low in sites where bramble cover was less than 1%, and over three times higher in sites where bramble cover was more than 2%. This makes good sense, given that all five species typically nest in dense cover on or near the ground, and are associated with open habitats that have a high availability of low-lying shrub cover (Snow & Perrins 1998). Sparks *et al.* (1996) found that Dunnock and Whitethroat, two of the ten species whose abundances were higher in planted than in unplanted sites, prefer field boundaries with low to medium (2 m) bushes interspersed with rank vegetation – the kind of vegetation that will almost invariably incorporate a substantial element of bramble cover. In peatland sites, bramble cover in both planted and unplanted sites was generally too low to provide suitable habitat for these species. In wet grassland sites with a high availability of tree and shrub cover, bramble cover was greater than 2% in both unplanted and planted sites. Only in grassland sites with low or intermediate shrub and tree cover was there an increase in bramble cover from below 1% to more than 2% between unplanted and planted sites. It therefore seems likely that the positive effect of afforestation on the abundance of these birds is mediated through the availability of nesting cover, which in turn is made possible by a relaxation of grazing intensity in newly afforested sites.

However, these benefits will not be long-term, as cover of low shrubs will decrease rapidly as the plantation matures, practically disappearing from all areas apart from roadsides and other open spaces after the canopy closes. It is likely that the shrub layer will increase again at the start of the second rotation of the plantation. However, further research is required to confirm that the species that benefit from increased shrub layer in new plantations are similarly benefited by such habitat in young restock. Even if second-rotation plantations are as or more valuable to birds of open, scrubby habitats than newly afforested sites, these benefits will only apply for less than a third of the rotation length in a typical commercial crop of densely shading conifers (e.g. 35 – 45 years for Sitka spruce; Forest Service, 2004). The predominant change in bird community that results from afforestation is from an open habitat to a that of a forest. This change has been well-documented by an earlier part of this project (Smith *et al.* 2005; Wilson *et al.* In Press). Whether the impact of the plantation on birds is positive or negative will depend largely on the bird assemblage of the open habitat that the forest replaces. Peatland sites tend to have low bird diversity, but are occupied by specialist species such as Whinchat and Stonechat, and support high densities of Meadow Pipits and Skylarks, species which are absent from closed canopy plantations. Low productivity grassland sites with areas of shrub cover allowed to grow by low grazing pressure can support relatively high densities of the uncommon ground-nesters that this study has found to be the main beneficiaries of afforestation during the first five years, and which will largely disappear from a plantation as the canopy closes. The sites in which birds are likely to benefit most from afforestation, at least in the long term, are areas of high-productivity grassland with high intensity grazing and few areas of scrub outside of the hedgerows. Moreover, many of the bird species that inhabit the hedges in these sites will also be able to persist in the forests that replace them. Most of these species will be able to exist at even higher densities as they will no longer be restricted to the site's internal and external field boundaries. The value for birds of plantations in such sites can further be improved by leaving sufficient open space around existing hedges that they will not be shaded out as the plantation matures, ensuring continuity of dense shrub cover and increasing the presence of native broadleaved trees throughout the commercial forest cycle.

Under current patterns of afforestation, the grassland sites most likely to be planted are the less productive fields in the agricultural landscape (Kearney 2001). Many of these are sites with relatively high shrub and tree cover, which already have diverse assemblages of birds, and in which the overall impact of afforestation on the bird assemblage will probably be a negative one. If such sites are to be planted with trees, it may be possible to mitigate the effects of afforestation by leaving areas of these sites unplanted. However, if left ungrazed, the tree and shrub layers in unplanted areas will eventually develop to the extent that they form a closed canopy, and low-lying shrubs will disappear. In Swedish farmland, Berg (1997) suggests that areas of semi-natural grassland, many of which have been abandoned and will eventually turn into forest if left unmanaged, might be maintained by periodical grazing. If unplanted areas of forest continued to be grazed at low intensity, this might allow the persistence of low-lying shrubs and the ground-nesting bird species they support.

7.4.5 Indicators

As discussed above, retained hedges may greatly enhance the bird assemblage supported by a plantation. Any retained hedges are likely to be of some benefit to birds, but if not all hedges on a site can be retained then an effort should be made to identify the best hedges for birds and retain these. It would be relatively simple to survey the hedges on the site for birds prior to planting. However, if bird surveying is not an option, or the time of year is inappropriate to census birds directly (breeding bird surveys are best carried out in spring or early summer), some or all of the following variables may be useful as indicators of quality of hedge for birds: presence of blackthorn, ash, rose or ivy; absence of gorse; height and width both 2m or greater; at least 1 mature standard tree per 100 m; proportion of gaps 10% or less; and 8 or more plant species per 100m.

Among the unplanted study sites, high bird species richness was associated mostly with high numbers of species that were relatively common in the surrounding landscape. Moreover, most of the species found in these sites could also be accomodated in plantation forests, especially if the open spaces and external boundaries of these plantations were managed symptathetically to species that require dense shrub cover, and incorporated a substantial broadleaved element (Iremonger *et al.* 2006). Indicators of high bird species richness in these unplanted habitats should not, therefore, be used to inform decisions about whether a site is suitable for afforestation. However, birds should still be considered during biodiversity assessments for afforestation. Afforestation of the types of open habitats investigated during this study is a potential threat to two groups of birds. One of these is comprised of birds that inhabit low productivity wet grassland. This kind of habitat typifies low-intensity farming systems, which are often of considerable conservation importance in predominantly high-intensity agricultural landscapes (Bignal & McCracken 1996). Bird species of wet grassland, which include Grasshopper Warbler, Reed Bunting, Sedge Warbler and Whitethroat, are relatively uncommon over much of the modern Irish agricultural landscape. Our study showed that habitat changes during the first decade after afforestation tend to favour these species, but that after canopy closure there remains little suitable habitat for these birds in plantations, except along some external edges and in large open spaces. In sites where these species are currently rare or absent, afforestation is likely to be beneficial, especially if the establishment phase of second rotation plantations also constitutes suitable habitat for them. However, where these species are already abundant, the establishment of a forest on a site is likely to be to their detriment. If open habitats supporting these species are not frequent in the surrounding landscape, afforestation of such sites should be avoided. Other species of conservation concern that breed in wet grassland include Lapwing, Redshank, Curlew, Snipe and Skylark. Sites with high densities of the latter two species, or even low concentrations of any of the first three species, may be unsuitable for afforestation.

The other category of birds that should be considered in the context of afforestion is the peatland avifauna of Ireland. Few members of this group were encountered during this study, but the group includes several species of conservation interest. Breeding waders such as Golden Plover, Dunlin, Curlew and Snipe, raptors such as Merlin *Falco columbarius* and Hen Harrier and other species including Red Grouse, Skylark and Whinchat all breed on peatlands. They are all listed as being of either high or medium conservation concern by Newton *et al.* (1999) and, with the exception of Whinchat and Hen Harrier, their response to afforestation of suitable habitat is prompt and negative. The latter two species will breed preferentially in young plantations, but may still be negatively affected by afforestation, because they cease to use forests after canopy closure.

To describe these birds as potential indicators is perhaps misleading. It is possible to use some of these species to identify sites of conservation importance, but that is because these species are, themselves, of conservation interest. However, many of the above species are easy to detect and to identify, and so are therefore well-suited for inclusion in pre-afforestation biodiversity assessment. Other potential indicators of wet grassland sites with species of open, shrubby habitats are high shrub-cover and low grazing intensities.

7.5 CONCLUSIONS

It is likely that the main benefits to bird diversity resulting from afforestation of open habitats in Ireland, during the first five years after planting with trees, are due to the decrease in grazing pressure experienced by these sites. In particular, the increase in cover of low shrubs provides habitat for several ground nesting species that are relatively uncommon in the modern agricultural landscape. However, these benefits will be short-term, lasting only until the tree canopy shades out this newly enriched shrub layer.

This is not to say that new forest plantations cannot have a beneficial effect on bird diversity. The bird assemblages of many improved grasslands, over-grazed wet grasslands and some degraded peatland areas are sufficiently impoverished that their replacement with a generalist forest avifauna will, in many cases, constitute a positive effect on biodiversity. This is especially likely to be the case if the plantations in question are managed sympathetically to birds. In particular, sufficient open space should be left around bird-rich hedges so that do not get shaded out during later stages of the forest cycle.

However, other open habitats generally cannot not be afforested without negatively impacting on birds. When the open habitat birds in a site are of greater biodiversity value than the forest generalists that would replace them if the site was planted, alternative land uses to afforestation should be considered. This is especially the case where elements of the bird community are of national or international conservation interest. However, if left ungrazed, many unplanted habitats will eventually undergo succession to scrub and native woodland and end up under a closed canopy unsuitable for open habitat specialists. Such areas may need to be continued to be grazed at low intensity, in order to allow the persistence of open habitats and the bird species they support.

8 SYNTHESIS AND RECOMMENDATIONS

8.1 BIODIVERSITY OF FREQUENTLY AFFORESTED HABITATS

8.1.1 Importance of scale

Of the four taxonomic groups surveyed in this study, plants and spiders are better suited to assessment of biodiversity at the ecosystem or habitat scale, whereas the more mobile hoverflies and birds are appropriate for biodiversity assessment at the wider site (e.g. grassland plus hedgerow habitats) and landscape scales. The species composition and abundances of the hoverfly and bird assemblages in improved grasslands, wet grasslands and peatlands were influenced by such larger scale features as the abundance of additional habitat types in the surrounding area, e.g. hedgerows in grasslands and flushes in peatlands. The land-use intensity of the wider landscape matrix was also an important factor. For example, three wet grassland sites located in landscapes with extensive areas of semi-natural habitat had a distinct hoverfly assemblage compared with grasslands in more intensive agricultural settings (Sections 5.3.2.3 and 5.4.2.2).

These landscape-scale effects mean that it is more difficult to ascertain the value of a particular site for the biodiversity of hoverflies, birds or other animal groups with similar mobility than it is for more sedentary species. However, in the context of afforestation or other land-use change, the corollary is that changes to one site will influence the biodiversity of such species groups across the landscape. An example of this is the cumulative effect of afforestation projects on the abundance of Hen Harriers at a wide scale (Wilson *et al*. 2005). Land-use changes at the site scale will also affect the biodiversity of less mobile groups, such as plants and spiders, at the landscape scale, particularly if the habitat represented by a given site is uncommon in the surrounding area. As highlighted in earlier chapters, the biodiversity value of a particular wet grassland, for example, may be relatively low if it is located in an area where such wet grasslands are abundant, but it may be quite high if it is located in an area of intensive agriculture. Therefore, for all the reasons discussed above, the biodiversity value of a given site cannot be fully assessed outside of its landscape context.

8.1.2 Summary of unplanted habitat biodiversity

8.1.2.1 Surveyed habitats

The three habitat types surveyed, improved grasslands, wet grasslands and peatlands, each supported species assemblages that were more or less distinct from one another, depending on the taxonomic group and the scale of investigation. Vegetation and spiders were sampled in adjacent plots (with some exceptions and pitfall trap losses) and there was a significant degree of correspondence between assemblages of these groups identified by cluster analyses. All of the vegetation sample units classified in peatland clusters at different scales also belonged to the *Peatland- open* spider assemblage group. In grasslands, spider clusters were significantly associated with vegetation habitat-scale clusters (likelihood ratio $\chi^2 = 20.5$, df = 6, p = 0.002) and 4 m^2 plot clusters (likelihood ratio $\chi^2 = 25.4$, df = 9, p = 0.003), but the best correspondence was with the clusters formed from 100 m^2 plot presence/absence data (likelihood ratio $\chi^2 = 31.4$, df = 9, p = 0.0002) (Table 55). All but one of the plots in the *Improved grassland- open* spider group were also assigned to the semi-improved and highly improved vegetation Groups 6 and 7. Seven out of ten *Wet grassland* group spider plots and all wet grassland plots in the *Peatland- open* spider group were also part of the oligotrophic wet grassland vegetation Groups 3 and 4 and the more base-rich wet grassland vegetation Group 5. Two of the *Wet grassland* spider group plots assigned to vegetation Group 6 were actually in a site pre-identified as wet grassland habitat, but whose plots were reassigned to Group 6 by the cluster analysis on the basis of the semi-improved nature of the flora.

Table 55. Percentage of spider assemblage groups (rows) corresponding with 100 m^2 vegetation groups (columns). N = number of plots within the appropriate group. Vegetation Group 4 was combined with Group 3 as the former represented only one site. See Sections 6.4.2 and 4.3.2.3 for explanations and descriptions of clusters.

	Groups 3 & 4 (n = 12)	**Group 5 (n = 6)**	**Group 6 (n = 11)**	**Group 7 (n = 3)**
Improved grassland- open (n = 12)	0	8.3	66.7	25
Wet grassland (n = 10)	50	20	30	0
Wet grassland plots in the *Peatland-open* group (n = 7)	57.1	42.9	0	0
Linear (n = 3)	100	0	0	0

According to results from the vegetation and spider surveys, the biodiversity of improved grasslands was much lower than that of wet grasslands and peatlands. This is the result of more intensive management, including higher grazing pressure and greater use of fertilisers and herbicides, which lead to a simplified vegetation structure dominated by *Lolium perenne* and *Trifolium repens* with a lower representation of other grass species and agricultural weeds. In contrast, the species richness of birds in improved grasslands was relatively high, but appeared to be primarily associated with medium to large hedgerows, treelines and pockets of semi-natural woodland. Abundance of birds in open pasture parts of improved grassland sites was a third of that found in wet grasslands, and bird density per 100 m of hedgerow was also substantially lower in improved grasslands than in wet grasslands.

The wet grasslands surveyed included semi-natural oligotrophic and base-rich types of potentially high biodiversity value and also semi-improved wet grasslands of lower value. These wet grassland types are currently amalgamated into one category (GS4) in the Irish habitat classification scheme (Fossitt 2000). However, they are distinguishable on the basis of differences in floristic composition which arise primarily from differences in soil chemistry as influenced by past and present management. Grazing intensity is an important factor influencing spider biodiversity in wet grasslands, with lower total species richness in more heavily grazed sites and lower species richness of wet habitat, low vegetation and ground layer spiders. Grazing intensity also influenced hoverfly biodiversity: sites with low grazing intensity had significantly higher numbers of grazing sensitive species. As mentioned above, a species-rich assemblage of hoverflies was identified from a group of three wet grassland sites; the assemblage was distinct from the remainder of wet grassland and improved grassland sites. These three sites were included among the more botanically diverse wet grassland sites (c.f. Figure 11, Appendix 4, Section 4.3.2), but landscape-scale factors were also important. Most bird assemblages of wet grasslands were placed into a group inhabiting sites with numerous hedgerows and some limited scrub development; these sites supported high diversity and abundances of bird species, including some uncommon ground-nesters. Three wet grassland sites, however, were placed in an intermediate group along with two upland improved grasslands and one wet heath site. These wet grasslands had fewer hedgerows and also lower species richness and abundance of birds. The vegetation of most of the grassland sites in the intermediate bird assemblage category was semi-improved in character, however, one of the wet grassland sites (MULL) was notable for its plant biodiversity (Figure 21, Section 4.3.2.5, Appendix 4).

The species composition of peatlands was, unsurprisingly, more distinct from the two grassland types than was species composition between the grasslands. Peatlands as a whole supported significantly higher species richness of some groups (bryophytes and ground layer spiders) than the grasslands, and biodiversity measures for several other groups were comparable to those in wet grasslands and higher than improved grasslands. The species richness and abundance of birds, however, was lower in peatlands than in grasslands, probably because of the lack of tall shrub habitat, such as hedgerows, for cover and feeding and also observer (ornithologist) avoidance. On the other hand, peatlands supported distinct bird assemblages compared with grasslands, characterised by such open-country species as Skylarks and Meadow Pipits. Peatland sites with well-developed flush systems supported some rare species of invertebrates, and flushes also provided habitat for a number of plant species that would otherwise not occur in the main peatland matrix. Peatland sites disturbed by overgrazing and turf-cutting supported a different flora, with fewer characteristic wet bog species than intact blanket bog (although these results are confounded by an association between disturbance and upland blanket bog and wet heath, with lowland blanket bog being less disturbed). The total abundance and species richness of some groups of spiders was lower in cutover bog than in some undisturbed peatlands. The biodiversity value of intact peatlands, however, is greater than that afforded by species numbers alone. (In fact, disturbed peatlands often support higher numbers of plant species than intact ones, which illustrates the hazards in overreliance on species richness as a measure of biodiversity or conservation value.) Intact peatlands are increasingly uncommon at national and global scales, and Ireland has a particular international obligation for the conservation of undamaged blanket and raised bogs (Section 2.4.6).

The value of additional habitats that are associated with the main habitat matrix has also been shown by our research. These habitats can include the aforementioned flushes and streams in peatlands, and hedgerows, treelines, earth banks, stone walls and ruined buildings, streams, scrub and small pockets of woodland in grasslands. These habitats accounted for approximately 40% of the plant species found in grassland sites and supported distinct spider assemblages. Treelines and scrub provided habitat for some rare species of hoverfly, and hedgerows supported a rare spider species. In the grassland sites we surveyed, abundance and quality of hedgerows, treelines and scrub were major influences on the species richness of hoverflies associated with woody vegetation and also species richness, abundance and composition of birds. Bird species richness and abundance was positively related to the height and breadth of hedgerows and treelines, the density of mature standard trees and the abundance of medium to large hedgerows in a site. Additional factors of potential importance for birds include presence of ditches associated with hedgerows, pockets of native woodland, low abundance of gaps in hedgerows and the species composition of the hedgerow trees and shrubs.

8.1.2.2 Other habitats

Because of the resources available to us, we were only able to survey three broad habitat types which consultation and preliminary field work indicated are frequently afforested. There are other types of habitat that may also be afforested with some regularity. In particular, these include dry-humid acid grassland (GS3), dry calcareous and neutral grassland (GS1) and dry siliceous heath. Dry-humid acid grasslands are most commonly encountered on podzols or shallow siliceous soils in upland areas, and have undoubtedly been much afforested in the past by the State. Dry neutral grasslands are generally found in lowland situations on soils of moderate fertility and are the most frequent precursors of improved grassland; two of the sites surveyed in this study appeared to be transitional between these two types. Dry calcareous grasslands are a particularly species-rich grassland type occurring on shallow, limestone soils. Afforestation of dry calcareous and neutral grasslands is probably less common than dry-humid acid grasslands, although their fertility may make them candidates for broadleaf planting. All of these semi-natural grassland types are undoubtedly declining through a combination of land-use conversion, dereliction and scrub invasion, and intensification; however, the true extent and biodiversity value of these grasslands remains poorly known (Section 2.3.2). Dry siliceous heath occurs most frequently on podzols or peaty podzols in exposed situations, such as coastal areas and uplands. The most extensive areas are to be found in coastal counties and counties with sizeable upland areas; dry heath is relatively rare in the midlands, apart from heath-type vegetation regenerating on cutover bog (Conaghan 2001b). Dry siliceous heath frequently occurs in mosaic with other habitat types, including dry-humid acid grassland, upland blanket bog, wet heath and gorse scrub. Dry heath was probably much afforested in the past by the State, and substantial afforestation may still occur. Other threats to heathland biodiversity include overgrazing, repeated burning, dereliction and agricultural reclamation (Bleasdale & Sheehy Skeffington 1995; Conaghan 2001b).

The conclusions, recommendations and biodiversity indicators identified by this study should only be applied to the habitats we have surveyed. Extension of some findings to other habitat types may be appropriate if treated with caution. Clearly, better information on the types of habitats that are most frequently afforested is needed. Additional work identifying biodiversity indicators for other types of habitats may be required.

8.2 INITIAL EFFECTS OF AFFORESTATION

Compared with the changes that take place across the forest cycle from pre-thicket to closed-canopy to mature and overmature forests (Smith *et al*. 2005), the changes in species composition and abundance arising from the first five years of afforestation are not great. Nevertheless, there are detectable differences between planted and unplanted site pairs in all four taxonomic groups among one or more of the surveyed habitat types. For plants, species composition and abundance changed the most in improved grasslands and peatlands. For spiders, species composition change was the greatest in peatlands. There was little difference in hoverfly assemblages of planted and unplanted peatlands, whereas some differences could be observed in grasslands. Bird species composition also differed more between planted and unplanted grasslands than in peatlands.

In grasslands, reduction of grazing pressure, changes in nutrient inputs (fertiliser and slurry), drainage of wet grasslands and provision of shelter by growing conifer saplings were the primary factors responsible for ecological change. The most striking change was an increase in the abundance and stature of competitive grasses, such as *Agrostis stolonifera, Elytrigia repens* and *Dactylis glomerata*. In all groups, species characteristic of open conditions declined, while species typical of wooded habitats (spiders and hoverflies), species frequently found in both wooded and unwooded habitats (plants) or species associated with low shrubs (birds) increased. Growth of planted conifers and invasion by shrub species from the hedgerows fostered environmental conditions somewhat more similar to woodlands than the previously open grassland. These changes were not enough to encourage greater numbers of typical woodland birds, but did increase abundances of a group of five uncommon bird species that prefer nesting in dense cover near the ground. Increases in these bird species were strongest between unplanted and planted site pairs where the former site had little cover of low shrubs, particularly bramble. Similarly, site pairs that differed more in cover of woody vegetation exhibited greater differences in number of hoverfly species associated with woody vegetation and trees/tall shrubs, as compared to site pairs with similar woody vegetation cover. Site drainage led to decreases in plant, spider and hoverfly species associated with wet conditions. More generally, invertebrate species associated with particular open-habitat characteristics decreased and were replaced by habitat generalists.

Changes in species composition and abundance due to afforestation were similar in peatlands. *Molinia caerulea* became dominant in most sites as a result of drainage and phosphate fertilisation, but forestry drains also provided a refuge for bryophytes, ferns and other stress-tolerant species susceptible to increased competition. Tree growth was slower on peatland sites, resulting in less shelter and no real differences between planted and unplanted sites in open- and forest-associated hoverflies and fewer differences in bird assemblage composition. However, such differences were evident for spiders and plants, possibly due in part to the shelter provided by dense *Molinia* tussocks and the increased vigour of *Calluna vulgaris*.

Unlike changes in the main habitat matrix, there were no real differences in the plant or spider species composition of hedgerows and treelines in grasslands. This is unsurprising as the conifer saplings have not begun shading the hedgerows to any great degree. It is worth noting that these sites were established prior to the 3 m buffer on either side of hedgerows required by the *Forest Biodiversity Guidelines* (Forest Service 2000c), and that in most sites, conifers were planted right up to the hedgerow edge.

The differences in biodiversity observed in the first five years after afforestation represent the beginnings of the transition from open grassland and peatland habitats to mature forest. Previous BIOFOREST work (Smith *et al.* 2005) has found that in forests comprised of densely shading conifers, such as Sitka spruce, current management practices will lead to a closed canopy forest at mid-rotation prior to the commencement of thinning. The open habitat assemblages of fauna and flora will then be virtually annihilated. In ash forests, and perhaps forests comprised of other broadleaf and more lightly-shading conifer species, changes will be more gradual and less complete, but open habitat species communities will nevertheless be replaced by those more characteristic of forests. Therefore, the changes described above are of a temporary nature and cannot be maintained without changes to management later in the forest cycle, if at all.

8.3 INDICATORS FOR BIODIVERSITY ASSESSMENT

8.3.1 Overview

Because complete assessments of biodiversity are generally impossible in practical terms, even for relatively small areas, efficient evaluation of site biodiversity must rely on one or more indicators that have been shown (or are assumed to be) correlated with overall biodiversity. In this report, we have focused on studying the biodiversity of four taxonomic groups, plants, spiders, hoverflies and birds. These groups are taxonomically well-known and diverse in their ecological characteristics, such as trophic level and the scales at which they respond to the abiotic environment and to disturbance. By using these groups as indicators for overall biodiversity, we assume that they reflect the biodiversity of other groups that we did not have the resources to study, such as fungi, mites, molluscs, beetles, lepidoptera and protozoa. The extent to which our results and recommendations can be extended to apply to other taxonomic groups depends on the extent to which they are correlated with the groups we studied. For efficient site level assessment of biodiversity prior to afforestation, however, a full inventory of even one of the taxonomic groups we have studied is out of the question.

Therefore, there is a need for indicators of biodiversity that are easy to apply by non-specialists in site assessment prior to afforestation. In the preceding chapters, several biodiversity indicators for peatlands and grasslands were identified, and they are summarised in this section. It is important to reiterate that biodiversity does not necessarily equate with species richness (see Section 2.2.1). For example, in our study sites, high bird species richness was associated mostly with high numbers of species that were relatively common in the surrounding landscape. Moreover, most of the species found in these sites could also be accomodated in plantation forests, especially if the open spaces and external boundaries of these plantations were managed to promote species that require dense shrub cover (see Iremonger *et al*. 2006). For these reasons, species or other factors correlated with high bird species richness should not be used to inform decisions about whether a site is suitable for afforestation. Therefore, the indicators we have identified are factors that are associated with semi-natural or natural plant communities that have experienced little human modification and the invertebrate and bird assemblages of these habitats. We have also identified several bird species of conservation concern as *de facto* indicators of biodiversity: these species are themselves of conservation interest. Many of these bird species are easy to detect and to identify, and so are therefore well-suited for use in pre-afforestation biodiversity assessment. Red Data Book or legally protected plant species may also be considered *de facto* indicators of biodiversity, although we did not encounter any in our survey (Table 6 and Table 7).

We divide the indicators into three types, compositional, structural and functional (Section 2.2.1) and into two quality levels, firm and potential. Firm indicators include those that have been pre-identified, either through literature review or previous field research, and that have been tested and confirmed by this present research. Also included as firm indicators are those that have not been pre-identified, but are nevertheless supported by the present study and backed by research from other studies, and also birds of conservation concern. Firm indicators are not infallible, they simply have been independently identified by more than one source. Potential indicators are new indicators that have emerged from analysis of field data from the present study. Also included as potential indicators are some that would otherwise qualify as firm indicators, but about which we have reservations as to their ability to discriminate between high and low biodiversity sites. Potential indicators could be used to supplement firm indicators, but need to be verified using independent data before their status is confirmed. We also identify habitat features that are indicators of biodiversity at the landscape scale.

8.3.2 Firm biodiversity indicators

Firm indicators of biodiversity are listed in Table 56 below. Indicators should be assessed in a walkover survey of the site during the habitat mapping required for the site development assessment (Forest Service 2000c). Other indicators, such as low grazing intensity or soil nutrient status, can be assessed through discussion with the landowner or inspection of existing maps and records. See the relevant taxonomic group sections for further discussion of indicators and their development.

Plant species compositional indicators should occur frequently in order to qualify as "present" for biodiversity indicator purposes. To be considered "frequent", a site should be divided by an imaginary 10 × 10 m grid, and a species should occur in 50% or more of these grid squares (see Section 4.4.1 for further details). Another definition of "frequent" could be "everywhere you look, you see some" (Lowe 2004). Note that this does not imply that a species should be evenly distributed across a site.

In grasslands, Grasshopper Warbler, Reed Bunting, Sedge Warbler and Whitethroat are uncommon species associated with good shrub cover. Grasshopper Warbler has also undergone a significant population decline in the last 30 years. Skylark, Snipe and Stonechat, though widespread in peatland and (in the case of the first two species) grassland habitats, have also undergone significant populations declines during this time (Newton *et al*. 1999). Merlin and Hen Harrier are species of much higher conservation concern, requiring a far greater area of rough pasture and especially peatland than that of a typical afforestation site (the conservation of the latter species in relation to afforestation is discussed in greater detail in Wilson *et al.* (2005)). The presence of any of these species on a site is a potential indicator of biodiversity, though care must be taken to distinguish sites where habitat for these species is generally of low quality from sites whose value for these species constitutes a compelling reason to prevent afforestation. Site quality will be reflected in the individual densities and total number of the above species present on a site. Threshold values separating high and low quality sites will vary depending on species and habitats, and a more thorough analysis of the ecology of these species in the Irish landscape is needed before such values can be formalised. The availability of habitat for these species in the surrounding landscape should also be taken into account. Where suitable habitat for these species is abundant in the surrounding landscape, their presence does not constitute a conflict with afforestation. However, where the surrounding landscape is relatively hostile to these species, afforestation of a site may impact negatively on bird diversity on a local, regional or (in the case of Hen Harrier and Merlin) national and international scale. The other species in Table 56 are of sufficient conservation concern that their presence during the breeding season should flag a site as being of potential importance for birds

Vegetation structure should be assessed on a similar scale to plant species, using the same frequency criterion in order to count as “present”. Some caution may be required when using some of the vegetation structure indicators. For example, areas where bracken, docks or alien species are abundant should not be counted as qualifying under the “forb cover > 25%” indicator. Similarly, areas of gorse invasion should not count as “shrub cover > 5%”. Low graminoid cover should reflect relatively higher levels of other vegetation layers and not be merely the result of excessive poaching. The assessment of grazing intensity should be based upon the average grazing intensity over several years. The functional indicators for peatlands should be assessed at the whole-site scale.

In addition to these positive indicators, there are two firm *negative* indicators of biodiversity. These are: high cover of *Lolium perenne* and recent reseeding of pasture. Where these are present, they indicate low biodiversity, but their absence does not necessarily indicate high biodiversity potential.

Table 56. Firm indicators of biodiversity.

	Compositional[1]	Structural	Functional
Grasslands	*Agrostis canina* s.l	Bryophyte cover > 5%	Low grazing intensity
	Carex echinata	Forb[2] cover > 25%	
	Carex nigra	Graminoid cover < 75%	
	Carex panicea	Shrub cover > 5%	
	Carex viridula		
	Cirsium dissectum		
	Danthonia decumbens		
	Festuca pratensis		
	Juncus conglomeratus		
	Molinia caerulea		
	Potentilla erecta		
	Prunella vulgaris		
	Pseudoscleropodium purum		
	Ranunculus flammula		
	Senecio aquaticus		
	Succisa pratensis		
	Thuidium tamariscinum		
	Grasshopper Warbler		
	Reed Bunting		
	Sedge Warbler		
	Whitethroat		
	Lapwing		
	Redshank		
	Curlew		
	Snipe		
	Skylark		
Peatlands	Golden Plover		Presence of extensive flushes or fen habitat
	Dunlin		Presence of pools
	Curlew		Presence of swallow holes
	Snipe		Low grazing intensity
	Merlin		Little or no peat cutting
	Hen Harrier		Absence of erosion
	Red Grouse		Absence of fire
	Skylark		Absence of drains
	Whinchat		Total P < 100 mg/L

[1] High frequency (see text) of any plant species listed is a compositional indicator of biodiversity. See text for discussion of bird compositional indicators.

[2] Broadleaf herbaceous plants including ferns, but not grasses, sedges or rushes.

8.3.3 New potential biodiversity indicators

New potential indicators of biodiversity are given in Table 57. They should be assessed in the same way as firm indicators. As discussed above, they can be used to supplement a site assessment based on the firm indicators, for example as additional evidence in borderline cases. Some plant indicator species characteristic of improved grasslands, *Poa annua, Urtica dioica, Stellaria media, Plantago major* and *Cirsium vulgare*, may have potential value as *negative* indicators of biodiversity, but further research is needed before they are formally used.

Table 57. New potential indicators of biodiversity.

	Compositional	Structural	Functional
Grasslands	*Carex hirta*	High frequency of tussocks	Total K < 5000 mg/L
	Centaurea nigra	High cover of bramble	
	Hypericum tetrapterum	High cover of hawthorn	
	Iris pseudacorus		
	Juncus bulbosus		
	Lathyrus pratensis		
	Leontodon autumnalis		
	Mentha aquatica		
	Pellia epiphylla [1]		
	Stellaria graminea		
Peatlands	*Campylopus atrovirens*		
	Drosera rotundifolia		
	Pleurozia purpurea		
	Racomitrium lanuginosum		
	Rhynchospora alba		
	Schoenus nigricans		
	Sphagnum cuspidatum		

[1] Can be easily confused with other *Pellia* species, but see Section 4.3.3.2.1.

8.3.4 Landscape biodiversity indicators

The taxonomic group sections, particularly hoverflies and birds, have also identified indicators of biodiversity at the landscape scale (Table 58). These are features that, if present within a landscape, indicate that landscape-scale biodiversity of one or more species groups is likely to be high. Accordingly, complete loss of such features from the landscape will most likely have detrimental consequences for biodiversity. A site containing one or more of these indicators can be afforested without much risk, if the feature is left undisturbed and the plantation is set back an appropriate distance from it (see Recommendation 4 below). In fact, there may be opportunities for forestry to have a positive biodiversity effect by promoting conservation and management of some of these features as ABEs. However, caution should be excercised in the case of multiple afforestation projects over time in a single landscape. Excessive afforestation of sites with these indicators may eventually damage biodiversity at the wider scale, particularly where they are uncommon in the landscape (see Recommendation 3).

Table 58. Landscape scale structural indicators of biodiversity.

Salix swamp	treelines with overmature trees
scrub	surface water features (e.g. ponds, streams)
well-developed hedgerows	semi-natural woodland*

* Including very small pockets.

8.3.5 Biodiversity assessment

Biodiversity assessment prior to afforestation can be thought of as comprising a number of steps outlined in Figure 32. Biodiversity assessment should always begin with a habitat survey and mapping compiled by the forester contracted to prepare an application for an afforestation grant. This survey should map and briefly describe habitats according to the Irish habitat classification scheme (Fossitt 2000) and note the presence of the above biodiversity indicators (see Recommendation 1 below). Foresters will require training in order to conduct adequate habitat surveys (see recommendation in Gittings *et al*. 2004). The survey serves two funtions: 1) to determine whether or not a site or part of a site should be afforested on biodiversity grounds (see Recommendation 2), and 2) to identify habitats to be incorporated into the ABE. The survey will rapidly reveal if the site is obviously of low biodiversity value, such as improved grassland, arable land, cutaway or cutover bog with little opportunity for regeneration or other disturbed land. Such habitats can be afforested with little likelihood of biodiversity loss and probable biodiversity gains.

If the site is not clearly of low biodiversity value, then the indicators above should help decide whether or not it is of potentially high biodiversity. The indicators should be used in conjunction with each other: it would be misleading to characterise a site as having high biodiversity (or not) on the basis of just one or two indicators. We recommend as a general guideline the presence of at least four or more indicators in two or more groups (compositional, structural and functional) or four plant species indicators as a guideline for designating sites or parts of sites as potentially having high biodiversity. Unless similar habitats of comparable or higher biodiversity are abundant in the landscape, the site should not be afforested without a more detailed ecological assessment (not necessarily an EIA) and approval by a trained ecologist (see Gittings *et al*. 2004). In landscapes dominated by improved grassland, tillage, commercial forestry or other intensive land-uses, sites with two or more indicators present should also be referred to an ecologist for assessment prior to afforestation. The guidelines for the best number and combination of indicators in different situations should be tested by independent research using a different set of sites (see Recommendation 7).

Although the biodiversity indicators we have proposed represent a tool that can be easily applied by non-specialists, they are not infallible. High biodiversity peatland or wet grassland sites may have few or no indicators present. If a forester or forest inspector suspects that a particular site may be of biodiversity value, despite the absence of indicators, the site should be referred to an ecologist for a more detailed assessment. For example, if a site is heterogeneous and contains a variety of different semi-natural habitat types, a more thorough site assessment by an ecologist should be undertaken. In addition, if the site contains semi-natural habitats in excess of the 15% of the site that is to be designated as ABE, a decision to afforest should be carefully considered in the context of the habitats involved and the surrounding landscape matrix. Regardless of how abundant it is in the landscape, certain habitat types should never be afforested, such as priority habitats listed in the EU Habitats Directive (European Commission 1999), some of which are briefly discussed in Sections 2.3.5.1 and 2.4.6.1 (see Recommendation 2).

We re-emphasise that the indicators we have identified are only applicable to peatlands, improved grasslands and wet grasslands. Further biodiversity indicators should be developed for other habitat types (Section 8.1.2.2, and see Recommendation 7).

8.4 RECOMMENDATIONS

8.4.1 Management recommendations

The forest management and planning recommendations we have developed are briefly explained below and in further detail elsewhere in this report. These recommendations were developed in the context of afforestation with Sitka spruce and the habitats we have surveyed. They may not be entirely applicable where other habitats or tree species are concerned. We indicate whether modifications to the *Forest Biodiversity Guidelines* (Forest Service 2000c) (henceforth, *Guidelines*) are required or not. Further amendment to other documents, such as the *Code of Best Forest Practice* (Forest Service 2000b) may also be required.

Recommendation 1 Foresters should compile annotated habitat maps of the entire site using the Irish habitat classification scheme (Fossitt 2000) and note the presence of indicators and other biodiversity features when conducting pre-afforestation site surveys.

This recommendation should be incorporated into the *Guidelines*.

This recommendation is similar to one made earlier by BIOFOREST (Gittings *et al*. 2004), and is supported by the results of this study. The *Guidelines* stipulate that "local biodiversity factors (including habitats and species of particular interest" should be identified, described and located on a map and then incorporated into the site development plan. However, the list of habitats provided in the *Guidelines* is incomplete and does not agree with any established classification system. The Irish habitat classification scheme (Fossitt 2000) developed by the Heritage Council has become the standard classification used in Ireland, and therefore we recommend its use. The Heritage Council are also developing further guidance for habitat mapping (Heritage Council in prep), which should be available later this year. However, we have found that the Heritage Council classification scheme does not discriminate well between some habitat subtypes that differ in biodiversity, such as semi-natural oligotrophic wet grasslands and semi-improved, species-poor wet grasslands. Therefore, we have identified the indicators above to assist in identifying habitats of potentially high biodiversity; the presence of these should also be noted and mapped. In the absence of a revision of the Irish habitat classification, the development of an in-house modification of the classification scheme for use by foresters should be considered.

Recommendation 2 Semi-natural habitats should not be afforested, unless there are mitigating circumstances.

This recommendation should be incorporated into the *Guidelines*.

This recommendation is similar to one made earlier by BIOFOREST (Smith *et al*. 2005) and is supported by the results of this study. Semi-natural habitats are those that are unmanaged or managed at a low intensity and that support assemblages of native species that have originated and are maintained primarily by natural processes. We have found that semi-natural wet grasslands and peatlands often support species assemblages of high biodiversity value, particularly when compared with improved grasslands. At present, the *Guidelines* do not consider the choice of sites for afforestation or what types of sites should not be afforested. The *Guidelines* should be amended to recommend that semi-natural habitats should not be afforested, unless there are mitigating circumstances, such as when such habitats are common in the local area (c.f. Recommendation 3 below). Improved grasslands, arable land and other land highly modified or disturbed by human use should be selected over semi-natural habitats where possible. Semi-improved grasslands (grasslands that have been substantially altered by management from a semi-natural state but would not be considered improved by agricultural standards, e.g. see Section 4.3.2.5) usually can be afforested without major impacts on biodiversity. Peatland sites with well-developed flushes or other hydrological features or sites that have experienced little or no peat cutting should not be afforested. Priority habitats listed in the EU Habitats Directive (European Commission 1999) should not be afforested, regardless of whether they are part of a designated site or not. We provide indicators to help identify wet grasslands and peatlands of biodiversity value (Section 8.3), but research on other frequently afforested habitat types is needed (c.f. Recommendation 7).

Recommendation 3 Consider site biodiversity in context of the surrounding landscape prior to afforestation.

This recommendation should be incorporated into the *Guidelines*.

The biodiversity value of semi-natural habitats is greater in areas of intensive agriculture, forestry or other highly altered landscapes. Such sites may support a substantial number of the plant and animal species in the landscape. Therefore, they should not be afforested. On the other hand, in landscapes where particular semi-natural habitats are abundant, afforestation of some of these habitats should not have significant negative impacts on local biodiversity. However, foresters and forestry inspectors should be aware of the potentially negative consequences of the cumulative effects of individual afforestation projects on landscape biodiversity. Indicative Forestry Strategies (IFSs) developed by the Forest Service and local authorities should take account of this risk. Landscapes identified as being suitable or favourable for afforestation should be changed to a less-suitable or sensitive category when a pre-identified cumulative afforestation threshhold has been reached. A customised GIS system could be developed to implement this recommendation. Afforestation of improved grassland or arable land would generally have a positive effect on the biodiversity of the local area, particularly where wooded habitats are rare.

Recommendation 4 Retain hedgerows, scrub, wetland habitats and other marginal habitats and allow for adequate buffer zones.

Some modification of the *Guidelines* is needed.

Our research has demonstrated the biodiversity value at the site and landscape scales of marginal and additional habitats, such as hedgerows, scrub, streams, ponds, stone walls, earthbanks and others. Currently, the *Guidelines* list most of these and other habitats as eligible for inclusion into the ABE, but the habitats named in the *Guidelines* should be revised to take account of other semi-natural habitats described in Fossitt (2000). The *Guidelines* also recommend a 3 m protective zone or buffer around such habitats. However, previous BIOFOREST research on biodiversity within open spaces in mature forestry plantations has found that this would be insufficient to provide adequately lit conditions for linear features in mature forests of densely-shading conifers (Iremonger *et al*. 2006). In the previous research, we recommended a setback of 7 m on each side from hedgerows and other linear features such as streams and stone walls. Open space buffers around wetland or surface water features should follow those given in the *Forestry and Water Quality Guidelines* (Forest Service 2000d). Where broadleaves are planted adjacent to wooded habitats, including scrub, treelines or hedgerows, it may be appropriate to leave no buffer between the tree crop and the habitat. Where the area of marginal and additional habitats plus buffers exceeds the required 15% ABE area, the additional area of semi-natural habitat should normally not be removed or disturbed, and the decision whether or not to plant should be considered carefully.

Recommendation 5 Promote broadleaved woody vegetation in young conifer plantations.

Some modification of the *Guidelines* is needed.

Woody vegetation, including bramble, increases the structural diversity of young conifer plantations, which in turn can increase the diversity of birds and hoverflies associated with woody habitats. Pre-existing shrubs and saplings should be retained within conifer plantations and natural regeneration should be encouraged. This recommendation also supports the requirement to plant broadleaves as part of most grant-aided conifer plantations (Forest Service 2003).

Recommendation 6 Drains should not be eligible as part of an afforestation site Area of Biodiversity Enhancement.

This recommendation should be incorporated into the *Guidelines*.

Drains are currently eligible as part of the ABE for afforestation projects, being listed under "Ridelines, firebreaks and drains" in an undated Forest Service Information Note for use in conjunction with the *Guidelines*. Although forestry drains can provide suitable microhabitats for a number of plant species, most of these will be eliminated when the forest canopy closes over. Therefore, drains should not be considered eligible as ABE unless they are protected by a 7 m unplanted buffer on each side, as with other linear features (Recommendation 4).

8.4.2 Recommendations for further research

Recommendation 7 Testing and refinement of the indicators identified in this study and identification of indicators for other habitat types.

The use of biodiversity indicators has the potential to be an efficient methodology applicable by non-specialists for site assessment prior to afforestation. While the list of firm biodiversity indicators given above (Table 56) provides reasonably faithful and consistent characteristics of sites that are of potentially high biodiversity, further trials are needed using independent data to determine how many indicators in which categories best discriminate between high and low biodiversity sites. In addition, we were only able to concentrate on three broad habitat groups (improved grassland, wet grassland and peatlands). More indicators are needed which can identify potentially high biodiversity examples of other habitat types, such as dry-humid acid grassland and dry heath (Section 8.1.2.2).

Recommendation 8 A comprehensive national survey and classification of grasslands.

As has been emphasised at several points in this report, site biodiversity cannot be adequately assessed in isolation from information about the ecological characteristics, abundance and quality of similar habitats at the landscape, regional and national scales. Such information is notably sparse and incomplete for grasslands in Ireland. The classification of grasslands in the Irish scheme (Fossitt 2000) is inadequate to describe the true ecological amplitude of grassland types. Estimates of the frequency and cover and assessments of the biodiversity value of grassland types in Ireland are inaccurate or out-of-date. We therefore recommend that a comprehensive national survey and classification of grasslands be undertaken as a matter of priority. Such a survey would benefit many other types of land-use management, in addition to forestry.

Recommendation 9 Investigation of forestry and biodiversity at whole-farm and landscape scales.

Management of biodiversity in relation to forestry is important at the site scale and has been considered in this report and other work by BIOFOREST (Iremonger *et al*. 2006; Smith *et al*. 2005). Where possible we have also considered the relationships between forestry and biodiversity at the landscape scale, but we have not had the resources or remit to answer many important questions. We recommend that further research be conducted on forestry and biodiversity (a) at the scale of the landscape, and also (b) at the scale that is most relevant for many private land-owners, the scale of the whole farm. Important research questions include but are not limited to the following. What are the effects of afforestation on wider-scale biodiversity in landscapes of varying forest cover? What effects does forestry of different ages and species have on biodiversity at the farm and landscape scales? Can forests of native or non-native species act as corridors between various wooded and non-wooded semi-natural habitats? What factors influence the immigration of species into plantations from neighbouring hedgerows or other wooded habitats?

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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 Vascular plant, bryophyte and lichen species recorded

Plant species recorded in the habitat survey and vegetation plots. *Taxon* indicates: V- vascular plant, B- bryophyte, L- lichen. *Moisture* indicates moisture preferences: 1- dry, 2- mesic, 3- damp, 4- wet. *Alien* shows whether a plant is native to Ireland (0) or non-native (1). *Woodland* indicates: 1- species typical of open habitats, 2- species sometimes found in woodlands or species characteristic of woodland edges, 3- typical woodland species. *pH* indicates soil pH preferences: 1- acidophilic, 2- neutral or broadly tolerant, 3- basophilic. *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 (1997) for vascular plants, Smith (2004) for mosses, Paton (1999) for liverworts and Purvis *et al.* (1992) for lichens.

Species	Taxon	Moisture	Alien	Woodland	pH	C	S	R
Acer pseudoplatanus	V	2	1	3	2	1	0	0
Achillea millefolium	V	2	0	1	2	1	0	1
Achillea ptarmica	V	3	0	1	2	1	1	1
Aesculus hippocastanum	V	2	1	3	2	1	1	0
Agrimonia eupatoria	V	2	0	2	2	1	1	1
Agrostis canina sl. [1]	V	3	0	2	1	1	1	1
Agrostis canina ssp. canina	V	4	0	2	1	1	1	1
Agrostis capillaris	V	2	0	2	1	1	1	1
Agrostis gigantea	V	2	0	1	2	1	0	1
Agrostis stolonifera	V	3	0	2	2	1	0	1
Agrostis vinealis	V	2	0	2	1	1	1	1
Aira praecox	V	1	0	1	1	0	1	1
Ajuga reptans	V	3	0	3	2	1	1	1
Alchemilla filicaulis	V	2	0	2	3	0	1	0
Alnus glutinosa	V	4	0	3	2	1	1	0
Alopecurus geniculatus	V	4	0	1	2	1	0	1
Alopecurus pratensis	V	2	0	1	2	1	0	0
Anagallis arvensis	V	2	0	1	2	0	0	1
Anagallis tenella	V	3	0	1	1	0	1	1
Angelica sylvestris	V	3	0	2	2	1	0	0
Antennaria dioica	V	2	0	1	2	0	1	0
Anthoxanthum odoratum	V	2	0	2	1	0	1	1
Anthriscus sylvestris	V	2	0	2	2	1	0	1
Apium nodiflorum	V	4	0	1	2	1	0	1
Arctium minus	V	2	0	2	2	1	0	1
Arrhenatherum elatius	V	2	0	1	2	1	0	0
Arum maculatum	V	2	0	3	3	0	1	1
Asplenium adiantum-nigrum	V	2	0	2	2	0	1	0
Asplenium trichomanes	V	1	0	2	3	0	1	0
Athyrium filix-femina	V	3	0	3	2	1	0	0
Avena sativa	V	2	1	1	2	0	0	1
Barbarea vulgaris	V	3	0	2	2	0	0	1
Bellis perennis	V	2	0	1	2	0	0	1
Berula erecta	V	4	0	1	2	1	0	1
Betula pendula [2]	V	2	0	2	2	1	0	0
Betula pubescens	V	3	0	2	2	1	0	0
Blechnum spicant	V	2	0	3	1	0	1	0
Brachypodium sylvaticum	V	2	0	3	2	1	1	0

Species	Taxon	Moisture	Alien	Woodland	pH	C	S	R
Briza media	V	2	0	1	2	0	1	0
Bromus hordeaceus	V	2	0	1	2	0	0	1
Buxus sempervirens	V	2	1	3	3	1	1	0
Callitriche stagnalis	V	4	0	1	2	0	0	1
Calluna vulgaris	V	2	0	2	1	1	1	0
Caltha palustris	V	4	0	2	2	0	1	0
Calystegia sepium	V	2	0	2	2	1	0	0
Capsella bursa-pastoris	V	2	0	1	2	0	0	1
Cardamine flexuosa	V	3	0	2	2	0	1	1
Cardamine hirsuta	V	2	0	1	3	0	1	1
Cardamine pratensis	V	4	0	2	2	1	1	1
Cardamine species	V	.	0	.	.	.	1	1
Carex acutiformis	V	4	0	1	2	1	1	0
Carex binervis	V	2	0	1	1	0	1	0
Carex dioica	V	4	0	1	2	0	1	0
Carex disticha	V	4	0	1	2	1	1	0
Carex divulsa	V	2	0	2	2	0	1	0
Carex echinata	V	4	0	2	1	0	1	0
Carex flacca	V	3	0	1	3	0	1	0
Carex hirta	V	3	0	1	2	1	0	0
Carex laevigata	V	3	0	3	2	0	1	0
Carex limosa	V	4	0	1	1	0	1	0
Carex nigra	V	4	0	1	1	1	1	0
Carex otrubae	V	4	0	1	2	1	0	1
Carex ovalis	V	3	0	1	2	0	1	0
Carex panicea	V	4	0	1	1	0	1	0
Carex paniculata	V	4	0	2	2	1	1	0
Carex pendula	V	3	0	3	2	0	1	0
Carex pilulifera	V	2	0	1	1	0	1	0
Carex pulicaris	V	4	0	1	2	0	1	0
Carex remota	V	3	0	3	2	1	1	1
Carex rostrata	V	4	0	1	2	1	1	0
Carex sylvatica	V	2	0	3	3	0	1	0
Carex viridula	V	3	0	1	2	0	1	0
Centaurea nigra	V	2	0	1	2	1	1	1
Centaurium erythraea	V	1	0	1	2	0	1	1
Cerastium fontanum	V	2	0	1	2	0	0	1
Chamaecyparis lawsoniana	V	2	1	3	2	1	1	0
Chamerion angustifolium	V	2	0	2	2	1	0	0
Chenopodium album	V	2	0	1	2	0	0	1
Chrysoplenium oppositifolium	V	3	0	3	2	1	1	1
Cirsium arvense	V	2	0	1	2	1	0	0
Cirsium dissectum	V	4	0	1	1	1	1	0
Cirsium juvenile [3]	V	.	0	.	.	1	.	.
Cirsium palustre	V	3	0	2	2	1	1	1
Cirsium vulgare	V	2	0	1	2	1	0	1
Conium maculatum	V	3	1	1	2	1	0	1
Conopodium majus	V	2	0	3	2	0	1	1
Convolvulus arvensis	V	2	0	1	2	1	0	1
Corylus avellana	V	2	0	3	3	1	1	0
Crataegus monogyna	V	2	0	2	2	1	1	0
Crepis capillaris	V	1	0	1	2	0	1	1

Species	Taxon	Moisture	Alien	Woodland	pH	C	S	R
Crocosmia × crocosmiflora	V	2	1	2	2	1	1	0
Cynosurus cristatus	V	2	0	1	2	1	1	1
Cytisus scoparius	V	1	0	2	1	1	1	0
Dactylis glomerata	V	2	0	2	2	1	1	1
Dactylorhiza fuchsii	V	3	0	2	2	0	1	0
Dactylorhiza maculata ssp. ericetorum	V	3	0	1	1	0	1	0
Danthonia decumbens	V	2	0	1	1	0	1	0
Daucus carota	V	1	0	1	3	0	1	1
Deschampsia caespitosa	V	3	0	2	2	1	1	0
Deschampsia flexuosa	V	2	0	2	1	1	1	0
Digitalis purpurea	V	2	0	2	1	1	1	1
Drosera anglica	V	4	0	1	1	0	1	0
Drosera intermedia	V	4	0	1	1	0	1	0
Drosera rotundifolia	V	4	0	1	1	0	1	0
Dryopteris aemula	V	2	0	3	1	1	1	0
Dryopteris affinis	V	2	0	3	1	1	1	0
Dryopteris carthusiana	V	3	0	3	2	1	1	0
Dryopteris dilatata	V	2	0	3	1	1	1	0
Dryopteris filix-mas	V	2	0	3	1	1	1	0
Dryopteris juvenile [3]	V	2	0	3	.	1	1	0
Eleocharis multicaulis	V	4	0	1	1	0	1	0
Eleogiton fluitans	V	4	0	1	1	0	1	0
Elytrigia repens	V	2	0	1	2	1	0	0
Empetrum nigrum	V	3	0	1	1	1	1	0
Epilobium brunnescens	V	3	1	1	2	0	1	1
Epilobium ciliatum	V	2	1	1	2	1	0	1
Epilobium hirsutum	V	3	0	1	2	1	0	0
Epilobium montanum	V	2	0	2	2	1	1	1
Epilobium obscurum	V	4	0	1	2	1	1	1
Epilobium palustre	V	4	0	1	1	0	1	0
Epilobium parviflorum	V	4	0	1	2	1	1	1
Equisetum arvense	V	2	0	1	2	1	0	1
Equisetum fluviatile	V	4	0	1	2	1	0	1
Equisetum palustre	V	4	0	1	2	1	1	1
Equisetum sylvaticum	V	3	0	2	1	1	1	0
Equisetum telmateia	V	3	0	2	2	1	1	0
Erica cinerea	V	2	0	1	1	0	1	0
Erica tetralix	V	3	0	1	1	0	1	0
Eriophorum angustifolium	V	4	0	1	1	0	1	0
Eriophorum vaginatum	V	3	0	1	1	1	1	0
Euonymus europaeus	V	2	0	3	3	1	1	0
Eupatorium cannabinum	V	3	0	1	2	1	0	1
Euphorbia helioscopia	V	2	0	1	2	0	0	1
Euphorbia hyberna	V	3	0	3	2	1	1	0
Euphrasia anglica	V	2	0	1	2	0	1	1
Euphrasia rostkoviana	V	2	0	1	2	0	1	1
Fagus sylvatica	V	2	1	3	2	1	1	0
Fallopia convolvulus	V	2	0	1	2	0	0	1
Festuca ovina	V	2	0	1	2	0	1	0
Festuca pratensis	V	3	0	1	2	1	1	1
Festuca rubra	V	2	0	1	2	1	1	1
Festuca vivipara	V	2	0	1	1	0	1	0

Species	Taxon	Moisture	Alien	Woodland	pH	C	S	R
Filipendula ulmaria	V	3	0	2	2	1	0	0
Fragaria vesca	V	2	0	3	3	1	1	1
Fraxinus excelsior	V	2	0	3	3	1	0	0
Fumaria muralis	V	2	0	1	2	0	0	1
Galium aparine	V	2	0	2	3	1	0	1
Galium palustre	V	4	0	2	2	1	1	1
Galium saxatile	V	2	0	1	1	0	1	0
Galium uliginosum	V	4	0	1	2	0	1	0
Galium verum	V	1	0	1	2	1	1	1
Geranium dissectum	V	2	0	1	2	0	0	1
Geranium molle	V	1	0	1	2	0	0	1
Geranium pusillum	V	2	0	1	2	0	0	1
Geranium robertianum	V	2	0	2	2	1	1	1
Geum rivale	V	3	0	2	2	0	1	0
Geum urbanum	V	2	0	3	2	0	1	0
Glechoma hederacea	V	2	0	3	2	1	1	1
Glyceria fluitans	V	4	0	1	2	1	0	1
Glyceria notata	V	4	0	1	2	1	0	1
Glyceria × *pedicellata*	V	4	0	1	2	1	0	1
Gnaphalium uliginosum	V	3	0	1	2	0	0	1
Hedera helix	V	2	0	3	2	1	1	0
Heracleum sphondylium	V	2	0	2	2	1	0	1
Hieracium species	V	.	.	.	.	0	1	0
Holcus lanatus	V	3	0	1	2	1	1	1
Holcus mollis	V	2	0	2	1	1	0	0
Hyacinthoides non-scripta	V	2	0	3	2	1	1	1
Hydrocotyle vulgaris	V	4	0	1	2	1	1	1
Hypericum androsaemum	V	2	0	3	2	0	1	0
Hypericum elodes	V	4	0	1	1	0	1	1
Hypericum maculatum	V	2	0	2	2	1	0	1
Hypericum perforatum	V	1	0	1	3	1	0	1
Hypericum pulchrum	V	1	0	1	1	0	1	0
Hypericum tetrapterum	V	3	0	1	2	1	1	1
Hypochaeris radicata	V	2	0	1	2	1	1	1
Ilex aquifolium	V	2	0	3	1	1	1	0
Iris pseudacorus	V	4	0	2	2	1	1	0
Isolepis setacea	V	3	0	1	2	1	1	1
Jasione montana	V	1	0	1	1	0	1	0
Juncus acutiflorus	V	3	0	2	1	1	1	0
Juncus articulatus	V	3	0	1	2	1	1	1
Juncus bufonius	V	3	0	1	2	0	0	1
Juncus bulbosus	V	4	0	1	1	0	1	1
Juncus conglomeratus	V	3	0	1	1	1	1	0
Juncus effusus	V	3	0	1	1	1	0	0
Juncus inflexus	V	3	0	1	3	1	1	0
Juncus squarrosus	V	3	0	1	1	0	1	0
Knautia arvensis	V	1	0	1	2	1	1	1
Lamium purpureum	V	2	0	1	2	0	0	1
Lapsana communis	V	2	0	3	2	0	0	1
Larix decidua	V	2	1	3	2	1	1	0
Larix kaempferi [2]	V	2	1	3	2	1	1	0
Lathyrus linifolius	V	3	0	2	1	0	1	0

Species	Taxon	Moisture	Alien	Woodland	pH	C	S	R
Lathyrus pratensis	V	2	0	1	2	1	1	1
Lemna minor	V	4	0	1	2	1	0	1
Leontodon autumnalis	V	2	0	1	2	1	1	1
Lepidium heterophyllum	V	1	0	1	1	1	1	1
Leucanthemum vulgare	V	1	0	1	3	1	1	1
Ligustrum vulgare	V	2	1	2	3	1	1	0
Linum catharticum	V	2	0	1	2	0	1	1
Lolium perenne	V	2	0	1	2	1	0	1
Lonicera periclymenum	V	2	0	3	2	1	1	0
Lotus corniculatus	V	1	0	1	2	0	1	0
Lotus uliginosus	V	3	0	1	2	1	1	0
Luzula campestris	V	2	0	1	2	0	1	0
Luzula multiflora	V	2	0	2	1	0	1	0
Luzula species [4]	V	.	0	.	.	.	1	0
Luzula sylvatica	V	3	0	3	1	1	1	0
Lychnis flos-cuculi	V	3	0	1	2	1	1	1
Lysimachia nemorum	V	2	0	3	2	0	1	0
Lysimachia vulgaris	V	3	0	1	2	1	1	0
Lythrum portula	V	4	0	1	2	0	0	1
Lythrum salicaria	V	4	0	2	2	1	1	1
Malus sylvestris	V	2	0	3	2	1	1	0
Matricaria discoidea	V	2	0	1	2	0	0	1
Medicago lupulina	V	1	0	1	3	0	1	1
Mentha aquatica	V	4	0	2	2	1	0	0
Mentha arvensis	V	2	0	1	2	1	0	1
Menyanthes trifoliata	V	4	0	1	1	0	1	0
Molinia caerulea	V	3	0	2	1	1	1	0
Montia fontana	V	3	0	1	2	0	0	1
Myosotis discolor	V	2	0	1	2	0	1	1
Myosotis laxa	V	4	0	1	2	1	0	1
Myosotis secunda	V	4	0	1	1	1	1	1
Myrica gale	V	4	0	1	1	1	1	0
Myriophyllum alterniflorum	V	4	0	1	1	1	1	1
Nardus stricta	V	2	0	1	1	0	1	0
Narthecium ossifragum	V	4	0	1	1	0	1	0
Odontites vernus	V	2	0	1	2	0	0	1
Oenanthe crocata	V	4	0	1	2	1	0	1
Orchis mascula	V	2	0	3	2	0	1	0
Oreopteris limbosperma	V	3	0	2	1	1	1	0
Osmunda regalis	V	3	0	2	1	1	1	0
Oxalis acetosella	V	2	0	3	1	0	1	0
Pedicularis palustris	V	4	0	1	2	0	1	1
Pedicularis sylvatica	V	3	0	1	1	0	1	1
Persicaria hydropiper	V	3	0	1	2	0	0	1
Persicaria maculosa	V	3	0	1	2	0	0	1
Persicaria wallichii	V	2	1	1	2	1	0	1
Petasites hybridus	V	3	0	1	2	1	0	0
Phalaris arundinacea	V	4	0	1	2	1	0	0
Phleum pratense	V	2	0	1	3	1	1	1
Phragmites australis	V	4	0	1	2	1	0	0
Phyllitis scolopendrium	V	2	0	3	3	0	1	0
Picea abies	V	3	1	3	1	1	1	0

Species	**Taxon**	**Moisture**	**Alien**	**Woodland**	**pH**	**C**	**S**	**R**
Picea sitchensis	V	3	1	3	2	1	1	0
Pilosella officinarum	V	1	0	1	2	0	1	0
Pimpinella saxifraga	V	2	0	1	2	0	1	0
Pinguicula grandiflora	V	4	0	1	1	0	1	0
Pinguicula lusitanica	V	4	0	1	1	0	1	0
Pinguicula vulgaris	V	4	0	1	2	0	1	0
Pinus contorta	V	2	1	3	1	1	1	0
Pinus sylvestris	V	2	0	3	1	1	1	0
Plantago lanceolata	V	2	0	1	2	1	1	1
Plantago major	V	2	0	1	2	0	0	1
Platanthera bifolia	V	3	0	1	2	0	1	0
Poa annua	V	2	0	1	2	0	0	1
Poa humilis	V	3	0	1	2	1	1	1
Poa pratensis	V	2	0	1	2	1	1	1
Poa trivialis	V	3	0	2	2	1	0	1
Polygala serpyllifolia	V	2	0	1	1	0	1	0
Polygonum arenastrum	V	2	0	1	2	0	0	1
Polygonum aviculare	V	2	0	1	2	0	0	1
Polypodium interjectum	V	2	0	3	2	0	1	0
Polypodium vulgare	V	2	0	3	1	0	1	0
Polystichum setiferum	V	2	0	3	2	1	1	0
Populus × canadensis	V	2	1	2	2	1	0	0
Populus × canescens	V	2	1	2	2	1	0	0
Potamogeton polygonifolius	V	4	0	1	1	1	1	1
Potentilla anglica	V	1	0	1	2	1	1	1
Potentilla anserina	V	2	0	1	2	1	0	1
Potentilla erecta	V	2	0	2	1	0	1	0
Potentilla palustris	V	4	0	1	2	0	1	0
Potentilla reptans	V	1	0	1	3	1	0	1
Potentilla sterilis	V	2	0	3	2	0	1	0
Primula veris	V	2	0	1	3	0	1	0
Primula vulgaris	V	2	0	3	2	1	1	1
Prunella vulgaris	V	2	0	1	2	1	1	1
Prunus avium	V	2	0	3	2	1	1	0
Prunus cerasus	V	2	1	2	2	1	1	0
Prunus domestica	V	2	1	3	2	1	1	0
Prunus padus	V	3	0	3	2	1	1	0
Prunus spinosa	V	2	0	2	2	1	1	0
Pseudotsuga menziesii [2]	V	2	1	3	1	1	1	0
Pteridium aquilinum	V	2	0	2	1	1	0	0
Pulicaria dysenterica	V	3	0	1	2	1	1	0
Quercus juvenile [3]	V	2	0	3	.	1	1	0
Quercus petraea	V	2	0	3	1	1	1	0
Quercus robur	V	2	0	3	2	1	1	0
Ranunculus acris	V	2	0	1	2	1	1	1
Ranunculus bulbosus	V	1	0	1	3	0	1	1
Ranunculus ficaria	V	3	0	3	2	0	1	1
Ranunculus flammula	V	4	0	1	2	1	1	1
Ranunculus omiophyllus	V	4	0	1	2	0	1	1
Ranunculus repens	V	3	0	2	2	1	0	1
Rhinantus minor	V	2	0	1	2	0	1	1
Rhododendron ponticum	V	2	1	3	1	1	1	0

Species	Taxon	Moisture	Alien	Woodland	pH	C	S	R
Rhynchospora alba	V	4	0	1	1	0	1	1
Rorippa nasturtium-aquaticum	V	4	0	1	2	1	0	1
Rosa arvensis	V	2	0	2	2	1	1	0
Rosa canina	V	2	0	2	2	1	1	0
Rosa sherardii	V	2	0	2	2	1	1	0
Rosa species	V	2	0	2	2	1	1	0
Rosa × *dumalis*	V	2	0	2	2	1	1	0
Rosa × *pseudorusticana*	V	2	0	2	2	1	1	0
Rosa × *verticillacantha*	V	2	0	2	2	1	1	0
Rubus fruticosus agg.	V	2	0	2	2	1	1	0
Rubus idaeus	V	2	0	2	2	1	1	0
Rumex acetosa	V	2	0	1	2	1	1	1
Rumex acetosella	V	2	0	1	1	0	1	1
Rumex conglomeratus	V	3	0	1	2	1	0	1
Rumex crispus	V	2	0	1	2	1	0	1
Rumex obtusifolius	V	2	0	1	2	1	0	1
Rumex sanguineus	V	3	0	2	2	1	1	1
Rumex species	V	.	0	.	2	.	.	1
Sagina nodosa	V	3	0	1	2	0	1	1
Sagina procumbens	V	3	0	1	2	0	0	1
Salix × *multinervis*	V	3	0	2	1	1	1	0
Salix × *sericans*	V	3	1	2	2	1	0	0
Salix alba	V	3	1	2	2	1	0	0
Salix aurita	V	3	0	1	1	1	1	0
Salix caprea	V	2	0	2	2	1	0	0
Salix cinerea	V	3	0	2	2	1	0	0
Salix fragilis	V	3	1	2	2	1	0	0
Salix pentandra	V	3	0	1	2	1	1	0
Salix repens	V	3	0	1	2	1	1	0
Salix viminalis	V	3	1	1	2	1	1	0
Sambucus nigra	V	2	0	2	3	1	0	0
Sanicula europaea	V	2	0	3	3	0	1	0
Saxifraga spathularis	V	3	0	2	2	0	1	0
Schoenus nigricans	V	4	0	1	2	1	1	0
Scrophularia auriculata	V	3	0	1	2	1	0	1
Scrophularia nodosa	V	2	0	2	2	1	0	1
Sedum anglicum	V	1	0	1	1	0	1	0
Senecio aquaticus	V	3	0	1	2	0	0	1
Senecio jacobaea	V	2	0	1	3	0	0	1
Senecio vulgaris	V	2	0	1	2	0	0	1
Sinapis arvensis	V	2	0	1	3	0	0	1
Solanum dulcamara	V	2	0	2	2	1	1	1
Sonchus arvensis	V	3	0	1	2	1	0	1
Sonchus asper	V	2	0	1	2	0	0	1
Sonchus oleraceus	V	2	0	1	3	0	0	1
Sorbus aucuparia	V	2	0	2	1	1	1	0
Sparganium species	V	4	0	1	.	1	0	0
Spiraea salicifolia	V	2	1	2	2	1	1	0
Stachys palustris	V	3	0	1	2	1	0	1
Stachys sylvatica	V	2	0	3	2	1	0	0
Stellaria graminea	V	3	0	1	1	1	1	1
Stellaria holostea	V	2	0	2	2	1	1	1

Species	Taxon	Moisture	Alien	Woodland	pH	C	S	R
Stellaria media	V	2	0	1	2	0	0	1
Stellaria uliginosa	V	4	0	1	1	1	0	1
Succisa pratensis	V	3	0	1	2	0	1	0
Symphoricarpos albus	V	2	1	2	2	1	1	0
Taraxacum officinale agg.	V	2	0	1	2	0	0	1
Taxus baccata	V	2	0	3	2	1	1	0
Teucrium scorodonia	V	1	0	2	2	1	1	1
Torilis japonica	V	2	0	2	2	1	1	1
Trichophorum cespitosum	V	3	0	1	1	0	1	1
Trifolium campestre	V	1	0	1	2	0	1	1
Trifolium dubium	V	2	0	1	2	0	0	1
Trifolium medium	V	2	0	2	2	1	1	0
Trifolium pratense	V	2	0	1	2	1	1	1
Trifolium repens	V	2	0	1	2	1	1	1
Tripleurospermum inodorum	V	2	0	1	2	0	0	1
Trisetum flavescens	V	1	0	1	3	1	1	1
Typha latifolia	V	4	0	1	2	1	0	0
Ulex europaeus	V	2	0	1	2	1	1	0
Ulex gallii	V	2	0	1	1	1	1	0
Ulmus glabra	V	2	0	3	2	1	0	0
Ulmus minor	V	2	1	2	2	1	0	0
Ulmus × hollandica	V	2	1	2	2	1	0	0
Umbilicus rupestris	V	1	0	1	1	0	1	0
Urtica dioica	V	2	0	2	2	1	0	0
Vaccinium myrtillus	V	2	0	2	1	1	1	0
Vaccinium oxycoccos	V	4	0	1	1	0	1	0
Valeriana officinalis	V	3	0	2	2	1	1	1
Veronica anagallis-aquatica	V	4	0	1	2	1	1	1
Veronica arvensis	V	1	0	1	2	0	1	1
Veronica beccabunga	V	4	0	1	2	1	0	1
Veronica chamaedrys	V	2	0	2	2	0	1	0
Veronica filiformis	V	2	1	1	2	0	0	1
Veronica montana	V	3	0	3	2	0	1	0
Veronica officinalis	V	1	0	1	1	0	1	0
Veronica scutellata	V	4	0	1	1	0	1	0
Veronica serpyllifolia	V	2	0	1	2	0	0	1
Viburnum opulus	V	3	0	3	2	1	1	0
Vicia cracca	V	2	0	1	2	1	0	0
Vicia sepium	V	2	0	2	2	1	0	0
Viola arvensis	V	2	0	1	2	0	0	1
Viola palustris	V	4	0	2	1	0	1	0
Viola riviniana	V	2	0	2	2	0	1	0
Viola species [5]	V	.	0	.	.	0	.	.
Aneura pinguis	B	3	0	2	2	.	.	.
Atrichum undulatum	B	2	0	3	1	.	.	.
Aulacomnium palustre	B	4	0	1	1	.	.	.
Barbilophozia attenuata	B	2	0	2	1	.	.	.
Barbula recurvirostra	B	2	0	2	3	.	.	.
Barbula unguiculata	B	1	0	1	2	.	.	.
Brachythecium rutabulum	B	2	0	2	3	.	.	.
Breutelia chrysocoma	B	4	0	1	2	.	.	.
Bryum capillare	B	1	0	2	2	.	.	.

Species	**Taxon**	**Moisture**	**Alien**	**Woodland**	**pH**	**C**	**S**	**R**
Bryum pseudotriquetrum	B	4	0	1	2	.	.	.
Calliergonella cuspidata	B	3	0	2	3	.	.	.
Calliergonella lindbergii	B	3	0	2	2	.	.	.
Calypogeia arguta	B	2	0	3	2	.	.	.
Calypogeia fissa	B	3	0	2	1	.	.	.
Calypogeia muellerana	B	3	0	2	1	.	.	.
Campylium stellatum	B	4	0	1	3	.	.	.
Campylopus atrovirens	B	3	0	1	1	.	.	.
Campylopus brevipilus	B	3	0	1	1	.	.	.
Campylopus flexuosus	B	2	0	2	1	.	.	.
Campylopus fragilis	B	2	0	2	2	.	.	.
Campylopus introflexus	B	2	1	1	1	.	.	.
Campylopus pyriformis	B	2	0	2	1	.	.	.
Cephalozia bicuspidata	B	3	0	2	1	.	.	.
Ceratodon purpureus	B	1	0	1	1	.	.	.
Chiloscyphus species	B	4	0	2	2	.	.	.
Climacium dendroides	B	3	0	2	2	.	.	.
Dichodontium palustre	B	4	0	1	2	.	.	.
Dicranella cerviculata	B	3	0	1	1	.	.	.
Dicranella heteromalla	B	2	0	2	1	.	.	.
Dicranoweisia cirrata	B	1	0	2	1	.	.	.
Dicranum bonjeanii	B	3	0	2	2	.	.	.
Dicranum majus	B	2	0	3	1	.	.	.
Dicranum scoparium	B	2	0	2	1	.	.	.
Didymodon rigidulus	B	1	0	1	3	.	.	.
Didymodon spadiceus	B	3	0	1	3	.	.	.
Diplophyllum albicans	B	2	0	2	1	.	.	.
Ditrichum heteromallum	B	2	0	2	1	.	.	.
Drepanocladus aduncus	B	4	0	1	3	.	.	.
Eurhynchium striatum	B	2	0	3	3	.	.	.
Fissidens bryoides	B	2	0	2	1	.	.	.
Fissidens taxifolius	B	2	0	2	3	.	.	.
Fossombronia species	B	3	0	.	2	.	.	.
Homalothecium lutescens	B	1	0	1	3	.	.	.
Homalothecium sericeum	B	1	0	2	2	.	.	.
Hookeria lucens	B	3	0	2	1	.	.	.
Hylocomium splendens	B	2	0	2	1	.	.	.
Hyocomium armoricum	B	4	0	2	1	.	.	.
Hypnum andoi	B	1	0	3	1	.	.	.
Hypnum cupressiforme	B	2	0	2	1	.	.	.
Hypnum jutlandicum	B	2	0	2	1	.	.	.
Hypnum resupinatum	B	1	0	2	1	.	.	.
Isothecium alopecuroides	B	2	0	3	2	.	.	.
Isothecium myosuroides	B	1	0	3	1	.	.	.
Jungermannia atrovirens	B	3	0	2	3	.	.	.
Jungermannia gracillima	B	3	0	2	2	.	.	.
Jungermannia species	B	.	0	.	.	.	.	.
Kindbergia praelonga	B	2	0	3	2	.	.	.
Kurzia pauciflora	B	4	0	1	1	.	.	.
Leptodictyum riparium	B	4	0	2	3	.	.	.
Leucobryum glaucum	B	2	0	2	1	.	.	.
Loeskeobryum brevirostre	B	2	0	3	2	.	.	.

Species	**Taxon**	**Moisture**	**Alien**	**Woodland**	**pH**	**C**	**S**	**R**
Lophocolea bidentata	B	3	0	2	2	.	.	.
Lophozia incisa	B	3	0	2	1	.	.	.
Lophozia ventricosa	B	2	0	2	1	.	.	.
Marchantia polymorpha ssp. ruderalis	B	2	0	2	2	.	.	.
Metzgeria furcata	B	1	0	3	2	.	.	.
Mnium hornum	B	2	0	3	1	.	.	.
Nardia scalaris	B	2	0	2	2	.	.	.
Neckera complanata	B	1	0	3	3	.	.	.
Odontoschisma denudatum	B	3	0	2	1	.	.	.
Odontoschisma sphagni	B	4	0	1	1	.	.	.
Oxyrrhynchium hyans	B	2	0	2	3	.	.	.
Pellia endiviifolia	B	3	0	2	3	.	.	.
Pellia epiphylla	B	3	0	2	1	.	.	.
Pellia neesiana	B	4	0	2	2	.	.	.
Philonotis calcarea	B	4	0	1	3	.	.	.
Philonotis fontana	B	4	0	1	2	.	.	.
Physcomitrium pyriforme	B	3	0	1	2	.	.	.
Plagiochila asplenioides	B	2	0	3	3	.	.	.
Plagiochila porelloides	B	2	0	2	2	.	.	.
Plagiomnium undulatum	B	2	0	3	3	.	.	.
Plagiothecium curvifolium	B	2	0	3	1	.	.	.
Plagiothecium denticulatum	B	2	0	3	1	.	.	.
Plagiothecium undulatum	B	2	0	2	1	.	.	.
Platyhypnidium riparioides	B	4	0	2	2	.	.	.
Pleuridium acuminatum	B	2	0	2	1	.	.	.
Pleurozia purpurea	B	3	0	1	1	.	.	.
Pleurozium schreberi	B	2	0	2	1	.	.	.
Pogonatum urnigerum	B	1	0	1	1	.	.	.
Polytrichastrum alpinum	B	2	0	1	1	.	.	.
Polytrichastrum formosum	B	2	0	3	1	.	.	.
Polytrichastrum longisetum	B	2	0	2	1	.	.	.
Polytrichum commune	B	4	0	2	1	.	.	.
Polytrichum juniperinum	B	2	0	1	1	.	.	.
Pseudephemerum nitidum	B	2	0	2	1	.	.	.
Pseudocrossidium revolutum	B	1	0	1	3	.	.	.
Pseudoscleropodium purum	B	2	0	2	2	.	.	.
Pseudotaxiphyllum elegans	B	2	0	3	1	.	.	.
Racomitrium fasciculare	B	1	0	1	1	.	.	.
Racomitrium lanuginosum	B	2	0	1	1	.	.	.
Rhizomnium punctatum	B	4	0	2	2	.	.	.
Rhynchostegiella pumila	B	2	0	3	2	.	.	.
Rhynchostegium confertum	B	2	0	2	3	.	.	.
Rhytidiadelphus loreus	B	3	0	2	1	.	.	.
Rhytidiadelphus squarrosus	B	2	0	2	2	.	.	.
Rhytidiadelphus triquetrus	B	2	0	2	3	.	.	.
Riccardia chamedryfolia	B	4	0	2	2	.	.	.
Riccardia latifrons	B	3	0	2	1	.	.	.
Riccardia multifida	B	4	0	2	2	.	.	.
Scapania gracilis	B	2	0	2	1	.	.	.
Scapania undulata	B	4	0	2	2	.	.	.
Scorpidium revolvens	B	4	0	1	3	.	.	.
Scorpidium scorpioides	B	4	0	1	3	.	.	.

Species	Taxon	Moisture	Alien	Woodland	pH	C	S	R
Sphagnum angustifolium	B	4	0	2	1	.	.	.
Sphagnum austinii	B	4	0	1	1	.	.	.
Sphagnum capillifolium	B	3	0	2	1	.	.	.
Sphagnum compactum	B	3	0	1	1	.	.	.
Sphagnum cuspidatum	B	4	0	1	1	.	.	.
Sphagnum denticulatum	B	4	0	2	1	.	.	.
Sphagnum fallax	B	4	0	2	1	.	.	.
Sphagnum inundatum	B	4	0	2	1	.	.	.
Sphagnum magellanicum	B	4	0	1	1	.	.	.
Sphagnum molle	B	3	0	1	1	.	.	.
Sphagnum palustre	B	4	0	2	1	.	.	.
Sphagnum papillosum	B	4	0	1	1	.	.	.
Sphagnum quinquefarium	B	2	0	3	1	.	.	.
Sphagnum squarrosum	B	4	0	2	2	.	.	.
Sphagnum subnitens	B	4	0	1	1	.	.	.
Sphagnum tenellum	B	4	0	1	1	.	.	.
Splachnum ampullaceum	B	3	0	1	1	.	.	.
Thamnobryum alopecurum	B	2	0	3	3	.	.	.
Thuidium tamariscinum	B	2	0	3	2	.	.	.
Tortula muralis	B	1	0	1	2	.	.	.
Tortula truncata	B	3	0	2	1	.	.	.
Ulota phyllantha	B	1	0	2	2	.	.	.
Cladonia cervicornis	L	1	0	1	2	.	.	.
Cladonia ciliata	L	2	0	1	1	.	.	.
Cladonia crispata	L	3	0	1	1	.	.	.
Cladonia fimbriata	L	2	0	1	2	.	.	.
Cladonia floerkeana	L	2	0	1	2	.	.	.
Cladonia furcata	L	2	0	2	1	.	.	.
Cladonia macilenta	L	2	0	2	1	.	.	.
Cladonia polydactyla	L	3	0	2	2	.	.	.
Cladonia portentosa	L	3	0	1	2	.	.	.
Cladonia species	L	.	0	.	.	.	.	.
Cladonia uncialis	L	3	0	1	1	.	.	.
Peltigera canina	L	2	0	1	3	.	.	.
Peltigera didactyla	L	1	0	1	2	.	.	.
Peltigera lactucifolia	L	3	0	2	1	.	.	.
Peltigera membranacea	L	2	0	2	2	.	.	.

[1] *Agrostis canina ssp. canina* and *Agrostis vinealis* not distinguished.
[2] Only recorded as a tree planted for forestry.
[3] Not identifiable to species because of very young developmental stage.
[4] *L. campestris / multiflora* not distinguishable.
[5] *V. riviniana / reichenbachiana* not distinguishable.

Appendix 3 Habitats Recorded in each site

Habitats recorded in each site according to the Heritage Council habitat classification scheme (Fossitt 2000). *Hab* indicates broad habitat group: IG- improved grassland, PL-peatland, WG- wet grassland. *Site* gives the four letter site code. *P/UP* shows whether the site is planted or unplanted. Site pairs are indicated by shading. Habitat codes are as follows:

BL1- stone walls and other stonework
BL2- earth banks
ED3- recolonising bare ground
FW1/2- eroding/upland and depositing/lowland rivers
FW4- drainage ditches
GA1- improved grassland
GS3- dry-humid acid grassland
GS4- wet grassland
HD1- dense bracken
HH1- dry siliceous heath
HH3- wet heath
PB2- upland blanket bog
PB3- lowland blanket bog
PB4- cutover bog
PF2- poor fen and flush
PF3- transition mire and quaking bog
WL1- hedgerow
WL2- treeline
WN2- oak-ash-hazel woodland
WN4- wet pedunculate oak-ash woodland
WS1- scrub

See Section 4.1 for details on habitat recording and mapping.

Hab	Site	P/UP	BL1	BL2	ED 3	FW1/2	FW4	GA1	GS3	GS4	HD1	HH1	HH3	PB2	PB3	PB4	PF2	PF3	WL1	WL2	WN 2	WN 4	WS1	Total
IG	LEAD	P						1											1					2
IG	AGHO	UP						1											1					2
IG	BALB	P		1				1																2
IG	KILB	UP						1			1								1					3
IG	BALP	P	1					1											1	1				4
IG	BALU	UP	1					1											1					3
IG	HANP	P						1											1					2
IG	BGHT	UP						1											1					2
IG	RATR	P						1											1				1	3
IG	CAST	UP		1				1											1					3
IG	FLEM	P	1					1											1					3
IG	DOON	UP	1					1											1					3
IG	MUNY	P						1											1					2
IG	KILL	UP						1											1					2
IG	KILM	P						1											1					2
IG	MOAF	UP						1											1					2
IG	GARY	UP		1	1[a]			1												1				4
PL	CARP	P												1		1								2
PL	CARU	UP												1		1								2
PL	GEAP	P											[b]		1[c]		1							2
PL	GEAU	UP											1		1[c]		1							3
PL	INCP	P											1				1	1						3
PL	INCU	UP											1				1							2
PL	SLIP	P							1				1											2
PL	SLIU	UP		1			1						1											3
PL	TIEP	P											1	1[c]										2
PL	TIEU	UP											1	1										2
PL	TOGP	P												1[d]										1
PL	TOGU	UP												1		1								2
PL	TOOP	P													1[c]									1
PL	TOOU	UP				1									1[c]									2

Hab	**Site**	**P/UP**	**BL1**	**BL2**	**ED3**	**FW1/2**	**FW4**	**GA1**	**GS3**	**GS4**	**HD1**	**HH1**	**HH3**	**PB2**	**PB3**	**PB4**	**PF2**	**PF3**	**WL1**	**WL2**	**WN2**	**WN4**	**WS1**	**Total**
PL	VEEP	P											1		1									2
PL	VEEU	UP				1							1		1									3
WG	GARV	P								1									1					2
WG	BOOL	UP		1						1									1					3
WG	GLAS	P				1				1										1				3
WG	CLAR	UP				1				1									1			1		4
WG	CLOP	P				1				1									1					3
WG	CLOU	UP				1				1									1					3
WG	COOA	P								1										1				2
WG	MNTP	UP								1										1				2
WG	CURR	P				1				1									1					3
WG	MOIG	UP				1				1									1					3
WG	GLEN	P				1				1									1					3
WG	KNAW	UP				1				1									1				1	4
WG	GORE	P		1			1			1														3
WG	MULL	UP				1				1										1				3
WG	RAUP	P								1									1	1				3
WG	RAUU	UP					1			1									1	1				4
WG	COOG	UP						1		1		1								1				4
WG	DONG	UP								1										1	1			3
Total			4	6	1	11	3	18	1	18	1	1	9	6	6	3	4	1	26	10	1	1	2	133

[a] Former cereal field left fallow in the year surveyed.

[b] Wet heath occurred with bog in matrix, not recorded separately.

[c] Includes cutover areas not separately recorded.

[d] Possibly includes cutover areas. These not recorded separately.

Appendix 4 Summary of vegetation clustering groups

The table below provides a synopsis of the clustering of vegetation plots described in Section 4.3.2. The *Plots* are 100 m^2 plots. *DAFOR* indicates the cluster groups of the habitats surveyed (using the DAFOR scale) in which the plots were located. *100 m^2* shows the group the plot was assigned to using 100 m^2 plot presence/absence data, and *4 m^2* gives the group the plot was assigned to using mean abundance data from two 4 m^2 plots nested within each larger plot.

Grasslands				**Peatlands**			
Plot	**DAFOR**	**100 m^2**	**4 m^2**	**Plot**	**DAFOR**	**100 m^2**	**4 m^2**
GaryF1	I	6	I	ToouF1	IV	1	A
GaryF2	I	6	I	ToouF2	IV	1	B
GaryF3	I	6	I	ToouF3	IV	1	A
BghtF1	I	7	I	GeauF3	IV	2	A
BghtF2	I	7	I	IncuF1	IV	2	B
CastF1	I	7	I	IncuF2	IV	2	B
CastF2	I	7	I	IncuF3	IV	2	B
CastF3	I	7	I	CaruF1	V	2	C
KillF1	I	7	I	CaruF2	V	2	C
KillF2	I	7	I	CaruF3	V	2	C
KillF3	I	7	I	SliuF1	V	2	C
MoafF1	I	7	I	SliuF2	V	2	C
MoafF2	I	7	I	SliuF3	V	2	A
MoafF3	I	7	I	TieuF1	V	2	B
BghtF3	I	*	I	TieuF2	V	2	B
AghoF1	II	6	H	TieuF3	V	2	C
AghoF2	II	6	H	GeauF1	VI	1	A
AghoF3	II	6	H	GeauF2	VI	1	B
BaluF1	II	6	G	ToguF1	VI	1	A
BaluF2	II	6	I	ToguF2	VI	1	A
BaluF3	II	6	I	ToguF3	VI	1	*
DoonF1	II	6	I	VeeuF1	VI	1	A
DoonF2	II	6	I	VeeuF2	VI	1	A
DoonF3	II	6	I	VeeuF3	VI	1	A
KilbF1	II	6	H				
KilbF2	II	6	H				
KilbF3	II	6	H				
BoolF1	III	3	D				
BoolF2	III	3	D				
BoolF3	III	3	D				
CoogF1	III	3	D				
CoogF2	III	3	G				
CoogF3	III	3	G				
MntpF1	III	3	D				
MntpF2	III	3	G				
MntpF3	III	3	G				
MullF1	III	3	F				
MullF2	III	3	F				
MullF3	III	3	F				

Grasslands				Peatlands			
Plot	**DAFOR**	**100 m^2**	**4 m^2**	**Plot**	**DAFOR**	**100 m^2**	**4 m^2**
KnawF1	III	4	D				
KnawF2	III	4	D				
KnawF3	III	4	D				
DongF1	III	5	E				
DongF2	III	5	E				
DongF3	III	5	E				
MoigF1	III	5	G				
MoigF2	III	5	G				
MoigF3	III	5	G				
RauuF1	III	5	D				
RauuF2	III	5	D				
RauuF3	III	5	D				
ClarF1	III	6	G				
ClarF2	III	6	G				
ClarF3	III	6	G				
ClouF1	III	6	G				
ClouF2	III	6	G				
ClouF3	III	6	G				

* Plots assigned to their own cluster groups.

Appendix 5 Hoverfly species recorded

	Number of sites	Habitat associations							
		Open	Surface water	Woody vegetation	Tree-tall shrub	Anthropophobic	Wetland specialists	Wet grassland specialists	Scrub specialists
Anasimyia lineata (Fabricius), 1787	1	√	√			√	√	√	
Anasimyia lunulata (Meigen), 1822	2	√	√			√	√	√	
Baccha elongata (Fabricius), 1775	28			√		√			
Brachyopa scutellaris Robineau-Desvoidy, 1843	1			√					√
Cheilosia albipila Meigen, 1838	2	√		√				√	√
Cheilosia albitarsis (Meigen), 1822	29	√		√		√			
Cheilosia antiqua (Meigen), 1822	3	√		√				√	
Cheilosia bergenstammi Becker, 1894	12	√		√		√			
Cheilosia illustrata (Harris), 1780	1	√		√		√			
Cheilosia latifrons (Zetterstedt), 1843	2	√						√	
Cheilosia pagana (Meigen), 1822	14	√		√					
Cheilosia variabilis (Panzer), 1798	2								
Cheilosia vernalis (Fallen), 1817	2	√		√		√			
Chrysogaster virescens Loew, 1854	1		√	√			√		√
Chrysotoxum bicinctum (L.), 1758	25	√		√		√			
Chrysotoxum fasciatum (Muller), 1764	3	√						√	
Criorhina berberina (Fabricius), 1805	4			√					√
Dasysyrphus albostriatus (Fallen), 1817	6			√		√			
Dasysyrphus venustus (Meigen), 1822	2			√		√			
Didea fasciata Macquart, 1834	1								
Epistrophe eligans (Harris), 1780	11			√					
Epistrophe nitidicollis (Meigen), 1822	2			√					√
Episyrphus balteatus (DeGeer), 1776	20	√		√		√			
Eristalinus sepulchralis (L.),	3	√	√					√	

	Number of sites	Habitat associations							
		Open	Surface water	Woody vegetation	Tree-tall shrub	Anthropophobic	Wetland specialists	Wet grassland specialists	Scrub specialists
1758									
Eristalis abusiva Collin, 1931	4	√	√			√	√	√	
Eristalis arbustorum (L.), 1758	5	√	√					√	
Eristalis interrupta (Poda), 1761	7	√	√				√	√	
Eristalis intricaria (L.), 1758	12	√	√			√	√	√	
Eristalis lineata (Harris), 1776	6	√	√			√	√	√	
Eristalis pertinax (Scopoli), 1763	21	√	√					√	
Eristalis tenax (L.), 1758	2	√	√					√	
Eumerus strigatus (Fallen), 1817	11	√		√					
Eupeodes bucculatus (Rondani), 1857	4			√		√			√
Eupeodes corollae (Fabricius), 1794	12	√		√					
Eupeodes latifasciatus (Macquart), 1829	26	√	√			√		√	
Eupeodes luniger (Meigen), 1822	8	√		√				√	
Ferdinandea cuprea (Scopoli), 1763	9			√					√
Helophilus hybridus Loew, 1846	3	√	√			√		√	
Helophilus pendulus (L.), 1758	36	√	√					√	
Lejogaster metallina (Fabricius), 1781	10	√	√				√	√	
Leucozona lucorum (L.), 1758	28			√					
Melangyna arctica (Zetterstedt), 1838	2			√					√
Melangyna lasiophthalma (Zetterstedt), 1843	25			√		√			
Melangyna umbellatarum (Fabricius), 1794	1		√	√			√		√
Melanogaster hirtella (Loew), 1843	21	√	√				√	√	
Melanostoma mellinum (L.), 1758	48	√		√		√			
Melanostoma scalare (Fabricius), 1794	49	√		√		√			
Meligramma guttata (Fallen), 1817	1			√					√
Meliscaeva auricollis (Meigen), 1822	17	√		√		√			
Meliscaeva cinctella	18			√		√			

	Number of sites	Habitat associations							
		Open	Surface water	Woody vegetation	Tree-tall shrub	Anthropophobic	Wetland specialists	Wet grassland specialists	Scrub specialists
(Zetterstedt), 1843									
Myathropa florea (L.), 1758	2		√	√		√			
Neoascia meticulosa (Scopoli), 1763	2	√	√			√	√		
Neoascia podagrica (Fabricius), 1775	21	√	√					√	
Neoascia tenur (Harris), 1780	7	√	√				√	√	
Orthonevra geniculata (Meigen), 1830	2	√	√			√	√		
Orthonevra nobilis (Fallen), 1817	1	√	√				√	√	
Paragus haemorrhous Meigen, 1822	4	√				√			
Parasyrphus nigritarsis (Zetterstedt), 1843	4			√					√
Parasyrphus punctulatus (Verrall), 1873	12			√					√
Parasyrphus vittiger (Zetterstedt), 1843	1					√			
Pipiza austriaca Meigen, 1822	7			√		√			
Pipiza noctiluca L, 1758	1			√		√			
Platycheirus albimanus (Fabricius), 1781	48	√	√	√		√			
Platycheirus ambiguus (Fallen), 1817	3			√		√			
Platycheirus angustatus (Zetterstedt), 1843	33	√	√					√	
Platycheirus clypeatus (Meigen), 1822	46	√	√			√			
Platycheirus fulviventris (Macquart), 1829	1	√	√				√	√	
Platycheirus granditarsus (Forster), 1771	40	√	√					√	
Platycheirus manicatus (Meigen), 1822	11	√	√					√	
Platycheirus nielseni Vockeroth, 1990	7	√				√			
Platycheirus occultus Goeldlin, Maibach & Speight, 1990	32	√	√			√		√	
Platycheirus peltatus (Meigen), 1822	10	√	√			√	√	√	
Platycheirus perpallidus Verrall, 1901	7	√	√			√	√		
Platycheirus ramsarensis Goeldlin, Maibach & Speight, 1990	5	√	√			√	√		
Platycheirus rosarum	19	√	√				√	√	

	Number of sites	Habitat associations							
		Open	Surface water	Woody vegetation	Tree-tall shrub	Anthropophobic	Wetland specialists	Wet grassland specialists	Scrub specialists
(Fabricius), 1787									
Platycheirus scambus (Staeger), 1843	22	√	√			√	√		
Platycheirus scutatus (Meigen), 1822	41			√		√			
Rhingia campestris Meigen, 1822	42	√	√					√	
Riponnensia splendens (Meigen), 1822	3		√	√		√	√		
Scaeva selenitica (Meigen), 1822	2			√					√
Sericomyia lappona (L.), 1758	5	√	√					√	
Sericomyia silentis (Harris), 1776	35	√	√					√	
Sphaerophoria fatarum Goeldlin, 1989	7	√				√			
Sphaerophoria interrupta (Fabricius), 1805	7	√	√	√		√			
Sphaerophoria philantha (Meigen), 1822	7	√	√					√	
Sphegina clunipes (Fallen), 1816	8		√	√					√
Syritta pipiens (L.), 1758	9	√	√					√	
Syrphus ribesii (L.), 1758	28	√		√		√			
Syrphus torvus Osten-Sacken, 1875	18			√		√			
Syrphus vitripennis Meigen, 1822	11	√		√				√	
Trichopsomyia flavitarsis (Meigen), 1822	16	√	√					√	
Tropidia scita (Harris), 1780	2	√						√	
Volucella bombylans (L.), 1758	35	√		√				√	
Volucella pellucens (L.), 1758	7			√		√			
Xanthandrus comtus (Harris), 1780	1					√			
Xylota jakutorum Bagatshanova, 1980	3					√			
Xylota segnis (L.), 1758	25			√		√			
Xylota sylvarum (L.), 1758	5			√					√

Appendix 6 The number of individual spiders sampled within each habitat type, site type (unplanted and planted) and plot type (O = open, L = linear), n = number of plots.

	Peatlands				Wet grasslands				Improved grasslands				Total
	Unplanted		Planted		Unplanted		Planted		Unplanted		Planted		
	O (n=35)	L (n=13)	O (n=29)	L (n=18)	O (n=24)	L (n=24)	O (n=24)	L (n=24)	O (n=24)	L (n=24)	O (n=24)	L (n=18)	
Agelena labyrinthica (Clerck, 1757)	0	0	0	0	1	0	0	0	0	0	0	0	1
Agroeca proxima (O.P.-Cambridge, 1871)	1	0	0	2	1	0	0	0	0	0	0	0	3
Agyneta cauta (O.P.-Cambridge, 1902)	10	0	26	0	0	0	0	0	0	0	0	0	26
Agyneta conigera (O.P.-Cambridge, 1863)	3	0	5	2	0	2	2	2	0	3	0	2	18
Agyneta decora (O.P.-Cambridge, 1871)	66	45	45	15	5	0	0	0	12	4	1	0	82
Agyneta olivacea (Emerton, 1882)	293	176	178	146	2	1	0	1	0	0	1	0	329
Agyneta ramosa (Jackson, 1912)	4	9	5	6	8	29	4	20	0	4	1	8	85
Agyneta subtilis (O.P.-Cambridge, 1863)	63	49	64	35	6	48	8	57	0	21	0	8	247
Allomengea vidua (Koch, 1879)	0	0	0	0	0	0	1	0	0	0	0	0	1
Alopecosa pulverulenta (Clerck, 1757)	61	7	0	3	14	1	4	4	3	1	9	3	42
Antistea elegans (Blackwall, 1841)	116	10	18	4	38	7	6	8	0	0	0	0	81
Aphileta misera (O.P.-Cambridge, 1882)	0	1	0	0	0	0	0	0	0	0	0	0	0
Araeoncus crassiceps (Westring, 1861)	18	0	0	0	0	0	0	0	0	0	0	0	0
Arctosa leopardus (Sundevall, 1843)	95	1	0	1	0	0	0	0	0	0	0	0	1
Asthenargus paganus (Simon, 1884)	0	0	0	0	0	0	1	0	0	2	0	1	4
Baryphyma gowerense (Locket, 1965)	1	0	2	1	4	0	0	0	0	0	0	0	7
Baryphyma trifons (O.P.-Cambridge, 1863)	3	1	2	5	14	3	17	4	1	1	9	0	56
Bathyphantes approximatus (O.P.-Cambridge, 1871)	0	0	0	0	8	2	4	0	0	0	6	1	21
Bathyphantes gracilis (Blackwall, 1841)	12	57	96	94	188	27	73	35	197	50	172	41	973
Bathyphantes nigrinus (Westring, 1851)	2	0	0	2	5	29	14	26	1	8	38	14	137
Bathyphantes parvulus (Westring, 1851)	17	37	35	19	39	20	103	62	7	52	192	158	687
Bathyphantes setiger (O.P.-Cambridge, 1894)	1	0	0	0	0	0	0	0	0	0	0	0	0
Bolyphantes luteolus (Blackwall, 1833)	0	0	1	0	0	0	0	0	0	0	0	0	1
Centromerita bicolor (Blackwall, 1833)	0	0	0	0	0	0	0	0	0	0	1	0	1
Centromerita concinna (Thorell, 1875)	15	2	2	0	3	1	0	0	0	1	0	0	7
Centromerus sylvaticus (Blackwall, 1841)	0	0	0	0	1	0	0	0	0	0	0	1	2

Centromerus dilutus (O.P.-Cambridge, 1875)	3	0	6	3	0	0	2	0	1	1	1	0	14
Ceratinella brevipes (Westring, 1851)	107	66	110	70	14	5	10	11	1	6	7	8	242
Ceratinella brevis (Wider, 1834)	32	0	13	15	1	3	0	2	0	7	1	0	42
Ceratinella scabrosa (O.P.-Cambridge, 1871)	0	0	1	0	7	85	5	65	2	47	7	32	251
Clubiona comta (C.L. Koch, 1839)	0	0	0	0	0	5	0	4	0	4	0	0	13
Clubiona diversa (O.P.-Cambridge, 1862)	3	0	0	1	0	0	0	0	0	0	0	0	1
Clubiona lutescens (Westring, 1851)	0	0	0	0	0	0	0	0	0	0	0	1	1
Clubiona neglecta (O.P.-Cambridge, 1862)	0	0	0	0	0	0	0	0	1	0	0	0	1
Clubiona reclusa (O.P.-Cambridge, 1863)	2	0	0	3	1	5	1	3	0	3	8	2	26
Clubiona stagnatilis (Kulczynski, 1897)	2	0	1	0	0	0	0	0	0	0	0	0	1
Clubiona trivialis (C.L.Koch, 1843)	1	0	1	0	0	0	0	0	0	0	0	0	1
Cnephalocotes obscurus (Simon, 1884)	19	6	4	0	2	0	0	0	0	0	0	0	6
Cryphoeca sylvicola (C.L.Koch, 1834)	0	0	0	1	0	1	0	2	0	1	2	1	8
Dicymbium nigrum (Blackwall, 1834)	13	26	5	2	55	8	21	14	32	4	38	7	186
Dicymbium tibiale (Blackwall, 1836)	2	0	4	4	4	0	2	0	3	2	3	0	22
Diplocephalus cristatus (Blackwall, 1833)	1	0	0	0	0	0	0	0	0	0	0	0	0
Diplocephalus latifrons (O.P.-Cambridge, 1863)	1	0	0	0	0	4	1	10	1	3	1	6	26
Diplocephalus permixtus (O.P.-Cambridge, 1871)	12	4	3	2	39	11	4	9	4	0	1	2	75
Diplocephalus picinus (Blackwall, 1831)	1	0	0	0	2	0	0	0	1	1	0	0	4
Diplostylor concolor (Wider, 1834)	0	0	0	0	0	15	0	5	0	9	8	18	55
Dismodicus bifrons (Blackwall, 1841)	22	7	28	12	7	35	13	34	17	30	37	18	231
Dolomedes fimbriatus (Clerck, 1757)	5	0	2	3	0	0	8	0	0	0	0	0	13
Drassodes cupreus (Blackwall, 1834)	18	0	0	1	0	0	0	0	0	0	0	0	1
Drassodes lapidosus (Walckenaer, 1802)	1	1	1	0	0	0	0	0	1	0	0	0	2
Drepanotylus uncatus (O.P.-Cambridge, 1873)	1	0	0	0	0	0	0	0	0	0	0	0	0
Dysdera crocata (C. L. Koch, 1838	0	0	0	0	0	0	0	0	0	2	0	0	2
Enoplognatha ovata (Clerck, 1757)	0	2	5	0	0	3	4	2	0	8	3	3	28
Enoplognatha thoracia (Hahn, 1833)	1	0	0	0	0	0	0	0	0	0	0	0	0
Episinus angulatus (Blackwall, 1836)	0	0	1	0	0	0	0	0	0	1	0	0	2
Episinus truncatus (Latrielle, 1809)	0	0	0	0	0	0	0	0	0	0	1	0	1
Erigone atra (Blackwall, 1833)	7	0	1	0	115	8	23	1	539	62	15	3	767
Erigone dentipalpis (Wider, 1843)	10	0	0	0	24	0	0	0	364	12	1	0	401
Erigone longipalpis (Sundevall, 1830)	0	0	0	0	0	0	0	0	2	1	0	0	3
Erigonella hiemalis (Blackwall, 1841)	1	0	0	0	0	3	1	0	1	3	5	4	17
Erigonella ignobilis (O.P.-Cambridge, 1871)	1	0	0	1	0	0	0	0	1	0	0	0	2
Ero cambridgei (Kulczynski, 1911)	4	0	9	3	5	2	2	1	1	1	2	0	26

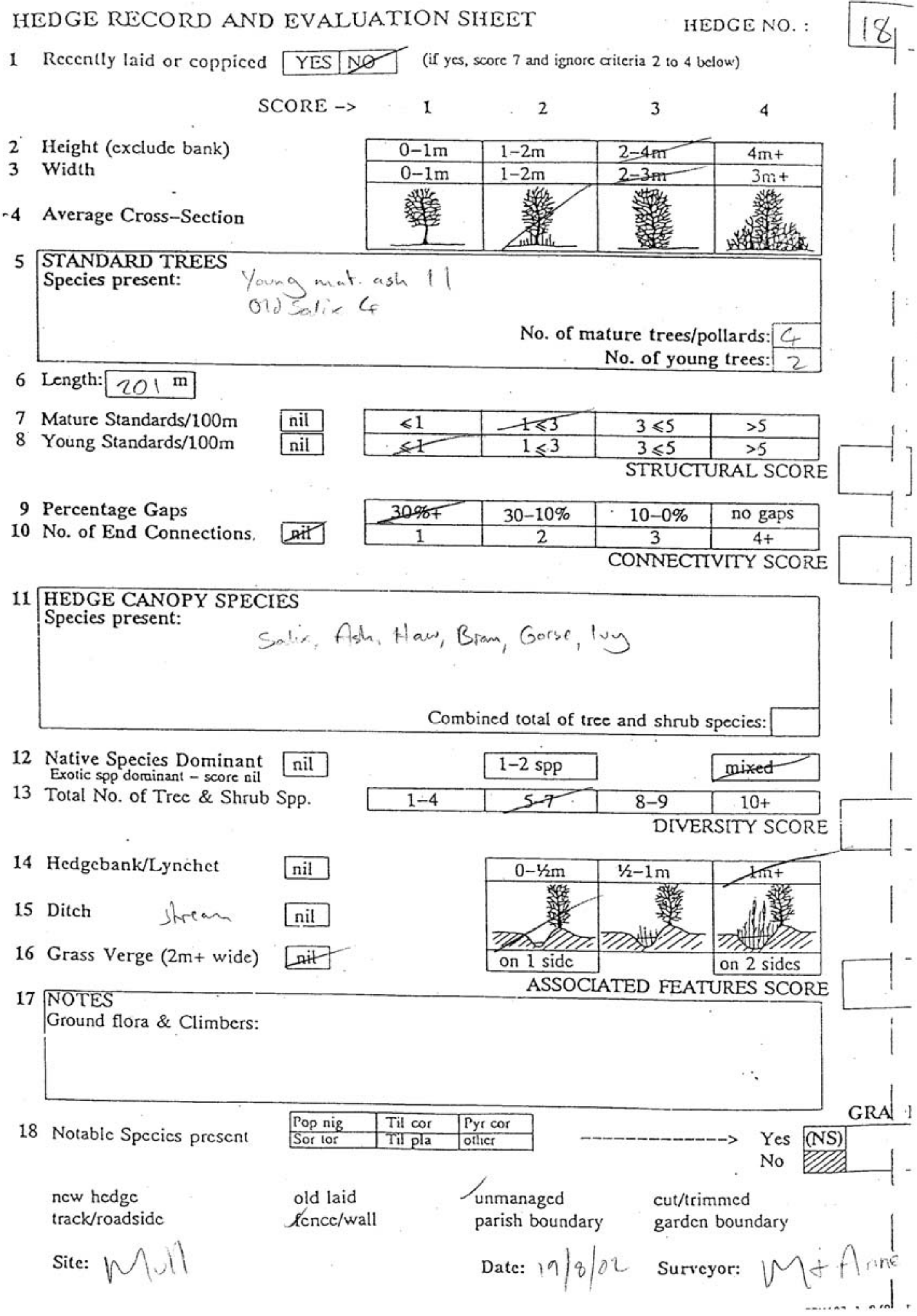
Ero furcata (Villers, 1789)	0	0	0	0	0	0	0	0	0	0	0	1	1
Euryopsis flavomaculata (C. L. Koch, 1836)	35	3	0	0	0	0	0	0	0	0	0	0	0
Gnathonarium dentatum (Wider, 1834)	0	1	2	1	18	2	18	1	1	2	1	0	46
Gonatium rubens (Blackwall, 1833)	2	0	5	1	0	0	0	0	0	2	1	0	9
Gongylidiellum vivum (O.P.-Cambridge, 1875)	20	19	16	12	26	11	21	20	27	19	62	21	235
Gongylidiellum latebricola (O.P.-Cambridge, 1871)	6	1	13	1	0	0	0	0	0	0	0	0	14
Gongylidum rufipes (Linnaeus, 1758)	1	0	0	0	0	5	0	8	0	4	1	1	19
Hahnia montana (Blackwall, 1841)	2	0	0	0	0	0	0	0	0	0	0	0	0
Hahnia nava (Blackwall, 1841)	0	0	0	0	1	0	0	0	0	0	0	0	1
Haplodrassus signifier (C.L. Koch, 1839)	5	4	6	1	0	0	0	0	0	0	0	0	7
Hilaira excisa (O. P.-Cambridge, 1870)	0	0	0	1	0	0	0	0	0	0	0	0	1
Hypomma bituberculatum (Wider, 1834)	20	14	12	11	14	8	9	2	8	4	7	3	78
Hypomma cornutum (Blackwall, 1833)	1	0	0	1	0	0	1	0	0	0	0	0	2
Hyposinga pygmaea (Sundevall, 1831)	3	0	1	0	0	0	0	0	0	0	0	0	1
Hypselistes jacksoni (O.P.-Cambridge, 1902)	9	1	2	1	0	0	0	0	0	0	0	0	3
Kaestneria dorsalis (Wider, 1834)	0	0	0	0	0	0	0	0	0	0	0	1	1
Kaestneria pullata (O.P.-Cambridge, 1863)	13	3	4	3	0	2	2	0	0	0	3	3	17
Lepthyphantes alacris (Blackwall, 1853)	0	1	27	9	0	7	2	5	1	5	0	13	69
Lepthyphantes angulatus (O. P. -Cambridge, 1871)	0	0	0	0	0	0	6	0	0	0	0	0	6
Lepthyphantes cristatus (Menge, 1866)	0	0	2	1	1	0	4	5	1	2	6	4	26
Lepthyphantes ericaeus (Blackwall, 1853)	36	32	71	60	21	27	28	29	1	19	66	37	359
Lepthyphantes flavipes (Blackwall, 1854)	0	0	3	0	0	8	0	15	1	3	0	3	33
Lepthyphantes mengei (Kulczynski, 1887)	81	31	98	85	39	16	20	6	0	6	4	2	276
Lepthyphantes minutus (Blackwall, 1833)	0	0	0	1	0	0	0	0	0	0	0	0	1
Lepthyphantes obscurus (Blackwall, 1841)	0	1	4	0	0	10	0	1	0	2	0	0	17
Lepthyphantes pallidus (O.P.-Cambridge, 1871)	0	0	0	0	0	1	1	0	0	0	2	2	6
Lepthyphantes tenebricola (Wider, 1834)	0	0	0	0	1	4	2	13	0	7	1	4	32
Lepthyphantes tenuis (Blackwall, 1852)	19	36	24	12	145	30	70	22	123	65	113	15	619
Lepthyphantes zimmermanni (Bertkau, 1890)	10	6	93	32	14	39	22	55	5	34	10	18	322
Leptorhoptrum robustum (Westring, 1851)	0	0	0	1	27	11	8	3	42	31	18	6	147
Lophomma punctatum (Blackwall, 1841)	2	11	8	10	12	1	6	3	0	0	21	1	62
Maro minutus (O.P.-Cambridge, 1906)	7	1	2	2	0	7	0	3	0	0	1	3	18
Maro sublestus (Falconer, 1915)	0	2	0	0	0	0	0	0	0	0	0	0	0
Maso sundervalli (Westring, 1851)	26	22	43	25	6	69	15	50	1	27	2	17	255
Meioneta beata (O.P.-Cambridge, 1906)	15	0	0	0	0	0	0	0	0	0	0	0	0
Meioneta mollis (O.P.-Cambridge, 1871)	19	0	0	0	0	0	0	0	0	0	0	0	0

Meioneta saxatilis (Blackwall, 1844)	23	14	3	1	1	0	0	2	6	13	4	4	34
Meta mengei (Blackwall, 1869)	0	0	0	0	0	1	0	3	0	0	3	0	7
Meta merianae (Scopli, 1763)	0	0	0	0	1	0	0	0	0	0	0	0	1
Meta *segmentata* (Clerck, 1757)	0	0	0	0	0	3	0	2	0	2	1	0	8
Metopobactrus prominulus (O.P.-Cambridge, 1872)	21	3	21	8	0	0	0	11	2	5	55	13	115
Micrargus herbigradus (Blackwall, 1854)	12	6	13	6	3	0	6	7	0	0	2	1	38
Micrargus subaequalis (Westring, 1851)	8	0	2	0	0	4	0	2	10	7	9	5	39
Microlinyphia pusilla (Sundevall, 1830)	5	2	3	7	1	2	0	0	0	0	0	0	13
Microneta viaria (Blackwall, 1841)	0	0	0	0	0	0	0	3	0	0	0	0	3
Milleriana inerrans (O.P.-Cambridge, 1885)	0	0	0	0	1	0	0	0	0	0	0	0	1
Minyriolus pusillus (Wider, 1834)	1	0	0	0	0	8	0	0	0	0	0	1	9
Monocephalus fuscipes (Blackwall, 1836)	5	5	3	5	19	74	22	168	1	58	44	100	494
Neon reticulatus (Blackwall, 1853)	5	0	7	0	0	0	0	0	0	0	0	0	7
Nereine clathrata (Sundevall, 1830)	0	1	2	5	9	24	16	31	0	17	4	23	131
Neriene Montana (Clerck, 1757)	0	0	0	0	1	1	0	0	0	0	0	0	2
Neriene peltata (Wider, 1834)	0	0	0	0	0	3	1	2	0	0	0	0	6
Nigma puella (Simon, 1870)	1	0	0	0	0	0	0	0	0	0	0	0	0
Oedothorax fuscus (Blackwall, 1834)	29	12	2	0	285	25	4	1	802	17	4	0	1140
Oedothorax gibbosus (Blackwall, 1841)	59	54	126	152	26	4	45	18	26	2	60	17	476
Oedothorax retusus (Blackwall, 1851)	1	23	1	2	38	11	9	6	120	35	36	9	267
Oxyptila trux (Blackwall, 1846)	152	29	20	13	18	13	14	14	22	41	10	5	170
Pachygnatha clercki (Sundevall, 1823)	34	10	23	6	50	10	15	3	1	1	8	4	121
Pachygnatha degeeri (Sundevall, 1830)	526	33	29	6	129	10	24	1	353	8	113	38	711
Pardosa agricola (Thorell, 1856)	0	0	0	0	1	0	0	0	0	0	0	0	1
Pardosa amentata (Clerck, 1757)	12	49	0	1	1183	338	404	132	163	95	351	83	2750
Pardosa nigriceps (Thorell, 1856)	168	28	33	5	36	5	18	6	10	11	30	22	176
Pardosa palustris (Linnaeus, 1758)	4	0	0	0	123	25	17	2	163	9	0	0	339
Pardosa pullata (Clerck, 1757)	1628	188	230	67	731	51	360	49	253	112	392	262	2507
Pelecopsis mengei (Simon, 1884)	1	2	0	0	40	0	1	0	0	0	0	0	41
Pelecopsis nemoralis (Blackwall, 1841)	0	0	0	0	0	0	0	1	0	0	0	0	1
Pelecopsis parallela (Wider, 1834)	4	0	2	0	0	0	0	0	0	0	0	0	2
Pepnocranium ludicrum (O.P.-Cambridge, 1861)	23	10	21	2	0	0	0	0	1	0	0	0	24
Pholcomma gibbum (Westring, 1851)	0	0	2	3	0	8	1	5	0	0	0	2	21
Pirata hygrophilus (Thorell, 1872)	19	1	8	3	0	0	0	0	0	0	0	0	11
Pirata latitans (Blackwall, 1841)	0	0	1	0	2	0	0	0	0	0	0	0	3
Pirata piraticus (Clerck, 1757)	472	108	75	28	281	36	16	4	3	1	8	3	455

Pirata uliginosus (Thorell, 1856)	71	3	96	46	1	0	0	0	1	1	4	0	149
Pisaura mirabilis (Clerck, 1757)	0	0	0	0	0	0	1	0	0	1	0	0	2
Pocadicnemis juncea (Locket & Millidge, 1853)	13	19	42	35	83	31	90	84	26	105	197	210	903
Pocadicnemis pumila (Blackwall, 1841)	111	54	269	274	40	42	75	51	9	8	58	10	836
Poeciloneta globosa (Blackwall, 1841)	0	1	1	0	0	1	1	1	0	0	0	0	4
Porrhomma egeria (Simon, 1884)	0	0	0	0	0	1	0	0	0	0	0	0	1
Porrhomma pygmaeum (Blackwall, 1834)	0	0	1	0	0	1	0	1	0	0	0	0	3
Robertus arundineti (O.P.-Cambridge, 1871)	3	2	2	4	0	0	0	0	0	0	0	0	6
Robertus lividus (Blackwall, 1836)	28	21	46	30	3	4	23	6	2	4	43	34	195
Robertus neglectus (O.P.-Cambridge, 1871)	0	0	0	0	0	0	0	0	0	1	0	0	1
Saaristoa abnormis (Blackwall, 1841)	5	17	29	12	0	3	8	8	0	0	11	2	73
Saaristoa firma (O.P.-Cambridge, 1905)	0	1	1	0	0	0	0	0	0	0	2	1	4
Saloca diceros (O.P.-Cambridge, 1871)	0	0	0	0	0	6	0	6	0	0	0	0	12
Satilatlas britteni (Jackson, 1913)	79	1	0	0	0	0	0	0	0	0	0	0	0
Savignya frontata (Blackwall, 1833)	0	0	0	0	4	0	0	0	16	1	1	0	22
Scotina gracilipes (Blackwall, 1859)	1	0	0	0	0	0	0	0	0	0	0	0	0
Segestria senoculata (Linnaeus, 1758)	0	0	0	0	1	1	0	4	0	4	1	1	12
Silometopus elegans (O.P.-Cambridge, 1872)	749	491	214	108	13	1	6	1	10	0	3	2	358
Tallusia experta (O.P.-Cambridge, 1871)	0	1	0	0	7	1	2	1	0	0	1	0	12
Tapinoba longidens (Wider, 1834)	1	0	0	0	0	0	0	0	0	0	0	0	0
Tapinocyba insecta (L. Koch, 1869)	0	0	0	0	1	0	0	0	0	0	0	0	1
Tapinocyba pallens (O.P.-Cambridge, 1872)	2	0	0	2	0	0	0	0	0	0	0	0	2
Taranucnus setosus (Simon, 1884)	1	0	16	11	7	2	12	7	1	1	12	7	76
Tetragnatha montana (Simon, 1874)	0	0	0	0	0	1	0	0	0	0	0	0	1
Textrix denticulata (Olivier, 1789)	0	0	0	0	0	1	0	1	0	0	0	0	2
Theonoe minutissima (O.P.-Cambridge, 1879)	0	1	6	4	0	0	2	0	0	0	0	0	12
Theridion bimaculatum (Linnaeus, 1767)	1	0	3	2	0	0	0	0	0	0	0	0	5
Theridion instabile (O.P.-Cambridge, 1870)	2	0	0	0	0	1	1	2	0	0	0	3	7
Theridion pallens (Blackwall, 1834)	0	0	1	0	0	0	0	0	0	0	0	0	1
Theridiosoma gemnosum (Koch, 1877)	0	0	0	0	0	0	0	1	0	0	0	0	1
Tibellus maritimus (Menge, 1875)	1	1	0	0	0	0	0	0	0	0	0	0	0
Tibellus oblongus (Walckenaer, 1802)	2	0	1	0	0	0	0	0	0	0	0	0	1
Tiso vegans (Blackwall, 1834)	37	47	1	0	1	0	0	1	3	0	6	0	12
Trichopterna thorelli (Westring, 1861)	165	1	2	0	0	0	0	0	4	0	0	0	6
Trochosa ruricola (De Geer, 1778)	8	0	0	0	0	1	0	0	16	0	5	1	23
Trochosa spinipalpis (O.P.-Cambridge, 1895)	6	0	1	0	59	16	26	4	1	1	3	0	111

Trochosa terricola (Thorell, 1836)	69	43	39	9	32	3	28	7	2	4	6	1	131
Troxochrus scabriculus (Westring, 1851)	0	1	0	0	0	0	0	0	14	0	0	4	18
Walckenaeria acuminata (Blackwall,1833)	5	12	6	7	6	4	11	8	0	6	30	17	95
Walckenaeria antica (Wider, 1834)	7	3	1	0	0	0	0	1	0	0	0	0	2
Walckenaeria atrobtibialis (O. P.-Cambridge, 1878)	33	8	58	11	0	0	0	0	0	0	2	0	71
Walckenaeria clavicornis (Emerton, 1882)	1	0	0	0	0	0	0	0	0	0	0	0	0
Walckenaeria cuspidata (Blackwall, 1833)	8	2	3	6	2	4	0	1	0	0	0	0	16
Walckenaeria dysderoides (Wider, 1843)	0	1	0	0	0	0	0	0	1	0	0	0	1
Walckenaeria kochi (O. P.- Cambridge, 1872)	0	0	0	1	0	0	0	1	0	0	0	0	2
Walckenaeria nodosa (O.P.-Cambridge, 1873)	2	1	0	0	0	0	0	0	0	0	0	0	0
Walckenaeria nudipalpis (Westring, 1851)	8	1	8	1	6	1	0	1	0	0	1	0	18
Walckenaeria unicornis (O.P.-Cambridge, 1861)	3	4	1	2	2	9	7	4	0	3	2	2	32
Walckenaeria vigilax (Blackwall, 1851)	89	52	10	9	42	16	43	12	2	1	7	2	144
Xysticus cristatus (Clerck, 1757)	12	3	0	0	16	2	4	0	2	2	3	2	31
Xysticus erraticus (Blackwall, 1834)	1	0	0	0	0	0	0	0	0	0	0	0	0
Xysticus ulmi (Hahn, 1831)	0	0	0	1	0	0	0	0	0	0	0	0	1
Zelotes latrielli (Simon, 1878)	2	0	2	0	0	1	0	0	0	0	0	0	3
Zelotes lutetianus (L. Koch, 1866)	1	0	0	0	0	0	0	0	0	0	0	0	0
Zora spinimana (Sundevall, 1833)	7	0	9	8	1	0	0	0	0	0	1	0	19
Immature	812	180	598	328	308	140	225	133	268	115	218	108	3433
Total Individuals	6942	2246	3204	1958	4552	1571	2107	1443	3718	1279	2641	1496	21528

Appendix 7 Sample hedge survey sheet



HEDGE RECORD AND EVALUATION SHEET

HEDGE NO.: 18

1 Recently laid or coppiced YES [✓ NO] (if yes, score 7 and ignore criteria 2 to 4 below)

	SCORE ->	1	2	3	4
2	Height (exclude bank)	0–1m	1–2m	[✓ 2–4m]	4m+
3	Width	0–1m	1–2m	[✓ 2–3m]	3m+
4	Average Cross-Section		✓		

5 **STANDARD TREES**
Species present: Young mat. ash 11
Old Salix 4
No. of mature trees/pollards: 4
No. of young trees: 2

6 Length: 201 m

7	Mature Standards/100m	nil	≤1	[✓ 1≤3]	3≤5	>5
8	Young Standards/100m	nil	[✓ ≤1]	1≤3	3≤5	>5

STRUCTURAL SCORE

9	Percentage Gaps		[✓ 30%+]	30–10%	10–0%	no gaps
10	No. of End Connections	[✓ nil]	1	2	3	4+

CONNECTIVITY SCORE

11 **HEDGE CANOPY SPECIES**
Species present: Salix, Ash, Haw, Bram, Gorse, Ivy
Combined total of tree and shrub species:

12	Native Species Dominant Exotic spp dominant – score nil	nil		1–2 spp		[✓ mixed]
13	Total No. of Tree & Shrub Spp.		1–4	[✓ 5–7]	8–9	10+

DIVERSITY SCORE

14	Hedgebank/Lynchet	nil	0–½m	½–1m	[✓ 1m+]
15	Ditch stream	nil			
16	Grass Verge (2m+ wide)	[✓ nil]	on 1 side		on 2 sides

ASSOCIATED FEATURES SCORE

17 **NOTES**
Ground flora & Climbers:

18 Notable Species present

Pop nig	Til cor	Pyr cor
Sor tor	Til pla	other

-------------------> Yes (NS)
No ▨

GRA

new hedge
track/roadside

old laid
[✓ fence/wall]

[✓ unmanaged]
parish boundary

cut/trimmed
garden boundary

Site: Mull

Date: 19/8/02 Surveyor: M + Anne

Appendix 8 Mean unplanted site densities of birds by site cluster. Densities are estimated from mapping survey data, except for Kestrel, Water Rail and Whinchat (these three species were not detected during mapping surveys, so site densities are estimated from point count data).

Common name	Scientific name	Cluster1	Cluster2	Cluster3	Cluster4	Cluster5
Blackbird	*Turdus merula*	0.22	0.72	0.33	0.01	0.01
Blackcap	*Sylvia atricapilla*	0.01	0.10	0.01		
Bullfinch	*Pyrrhula pyrrhula*	0.03	0.08	0.03		
Blue Tit	*Parus caeruleus*	0.10	0.26	0.03		
Chiffchaff	*Phylloscopus collybita*	0.02	0.11	0.02		
Collared Dove	*Streptopelia decaocto*	0.01	0.01			
Chaffinch	*Fringilla coelebs*	0.17	0.41	0.25	0.01	
Coal Tit	*Parus ater*	0.02	0.25	0.11	0.01	
Curlew	*Numenius arquatus*	0.01			0.01	
Dunnock	*Prunella modularis*	0.13	0.24	0.11		
Goldcrest	*Regulus regulus*	0.07	0.30	0.08	0.02	
Grasshopper Warbler	*Locustella naevia*		0.01	0.01	0.01	
Grey Wagtail	*Motacilla cinerea*	0.01		0.01		
Greenfinch	*Carduelis chloris*	0.01	0.03			
Great Tit	*Parus major*	0.07	0.14	0.05		
Hooded Crow	*Corvus corone*		0.02	0.02	0.01	
House Sparrow	*Passer domesticus*	0.01		0.00		
Jackdaw	*Corvus monedula*	0.01	0.01	0.01		
Kestrel	*Falco tinnunculus*		*0.02*	*0.02*		
Linnet	*Carduelis cannabina*	0.04	0.03	0.02	0.01	
Redpoll	*Carduelis flammea*	0.02	0.04	0.02	0.01	
Long-tailed Tit	*Aegithalos caudatus*	0.02	0.06	0.01		
Lapwing	*Vanellus vanellus*	0.01				
Mistle Thrush	*Turdus viscivorus*	0.01	0.01	0.03	0.01	
Magpie	*Pica pica*	0.03	0.10	0.01		
Meadow Pipit	*Anthus pratensis*	0.07	0.39	0.96	1.40	0.29
Pheasant	*Phasianus colchicus*	0.01	0.05	0.01		
Pied Wagtail	*Motacilla alba*	0.01	0.01	0.01		
Robin	*Erithacus rubecula*	0.31	0.93	0.53	0.01	0.01
Reed Bunting	*Emberiza schoeniclus*	0.01	0.08	0.11	0.02	0.01
Raven	*Corvus corax*		0.01		0.01	
Rook	*Corvus frugilegus*	0.07	0.06	0.02		
Skylark	*Alauda arvensis*	0.01	0.03	0.08	0.28	0.12
Stonechat	*Saxicola torquata*	0.01	0.01	0.01	0.07	
Spotted Flycatcher	*Muscicapa striata*		0.02			
Starling	*Sturnus vulgaris*	0.03		0.03	0.01	
Sparrowhawk	*Accipiter nisus*	0.01		0.01		
Siskin	*Carduelis spinus*			0.01	0.01	
Swallow	*Hirundo rustica*	0.04	0.05	0.01	0.01	
Snipe	*Gallinago gallinago*		0.02	0.03	0.05	0.01
Song Thrush	*Turdus philomelos*	0.06	0.17	0.09		
Sedge Warbler	*Acrocephalus schoenobaenus*	0.01	0.04	0.03	0.01	
Water Rail	*Rallus aquaticus*			*0.12*		
Wheatear	*Oenanthe oenanthe*			0.02		0.01
Whinchat	*Saxicola rubecula*				*0.03*	
Whitethroat	*Sylvia communis*	0.01	0.08	0.02		
Wood Pigeon	*Columba palumbus*	0.17	0.18	0.03		
Wren	*Troglodytes troglodytes*	0.34	1.04	0.42	0.02	0.01
Willow Warbler	*Phylloscopus trochilus*	0.07	0.86	0.25	0.01	0.01

FIGURES

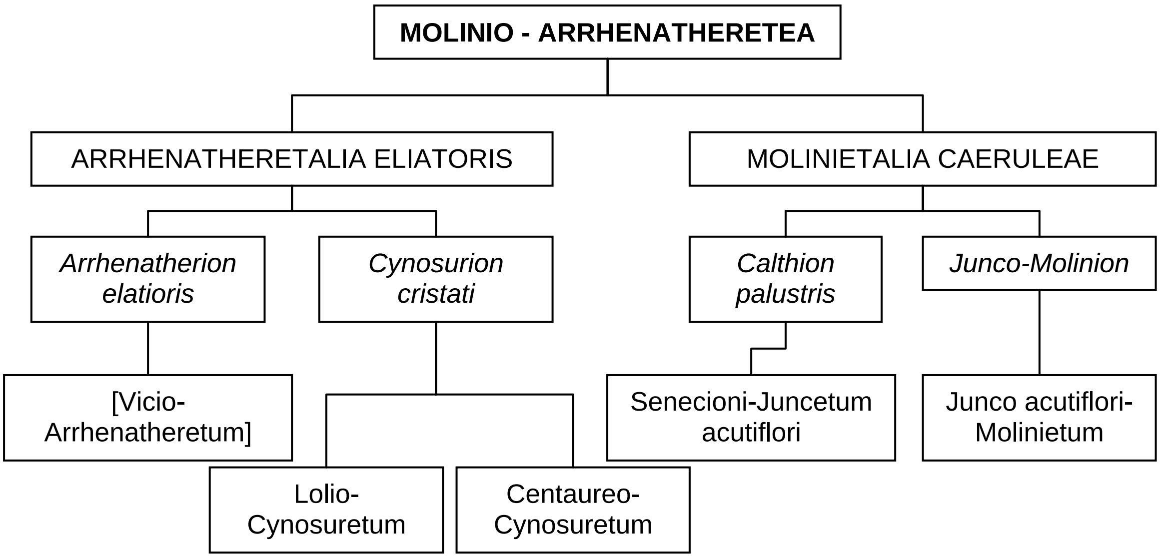


Figure 1. Phytosociological synopsis of grassland vegetation in the class Molinio - Arrhenatheretea in Ireland (after O'Sullivan 1982). Lower hierarchical levels are: 2nd- order, 3rd- alliance and 4th- association.

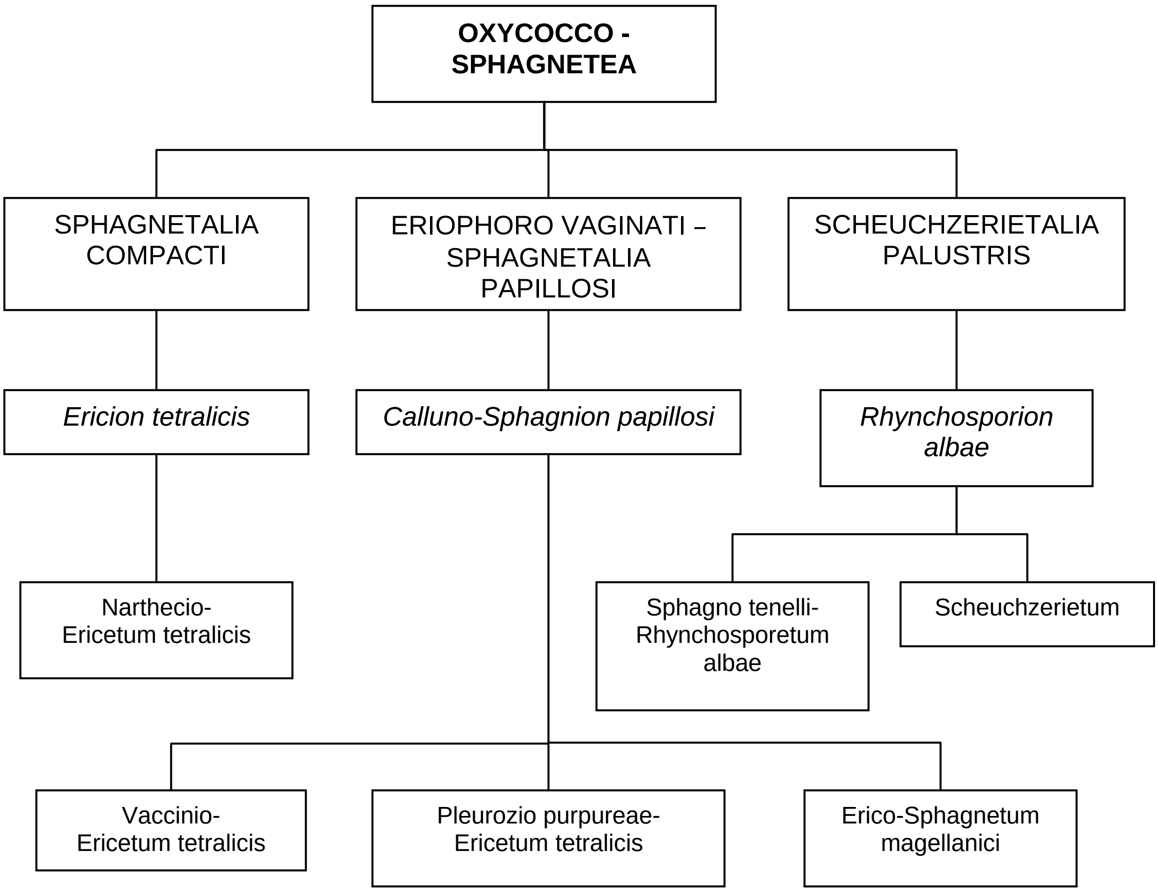


Figure 2. Phytosociological synopsis of peatland vegetation in the class Oxycocco-Sphagnetea in Ireland (after White & Doyle 1982). Lower hierarchical levels are: 2nd- order, 3rd- alliance and 4th- association.

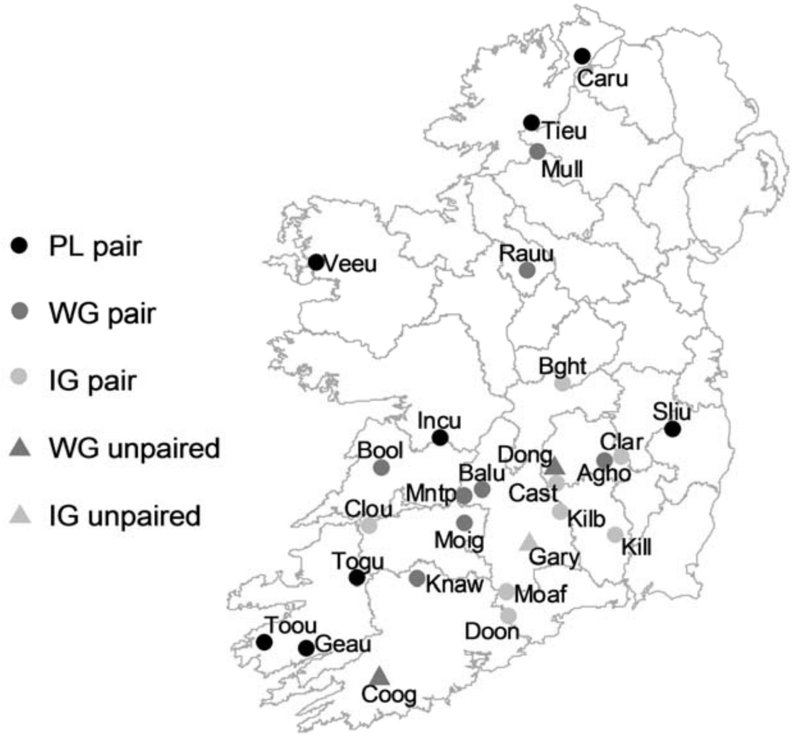


Figure 3. Study site locations. PL = Peatland, WG = Wet grassland, IG = Improved grassland. For paired, only the site code of the unplanted site is shown.

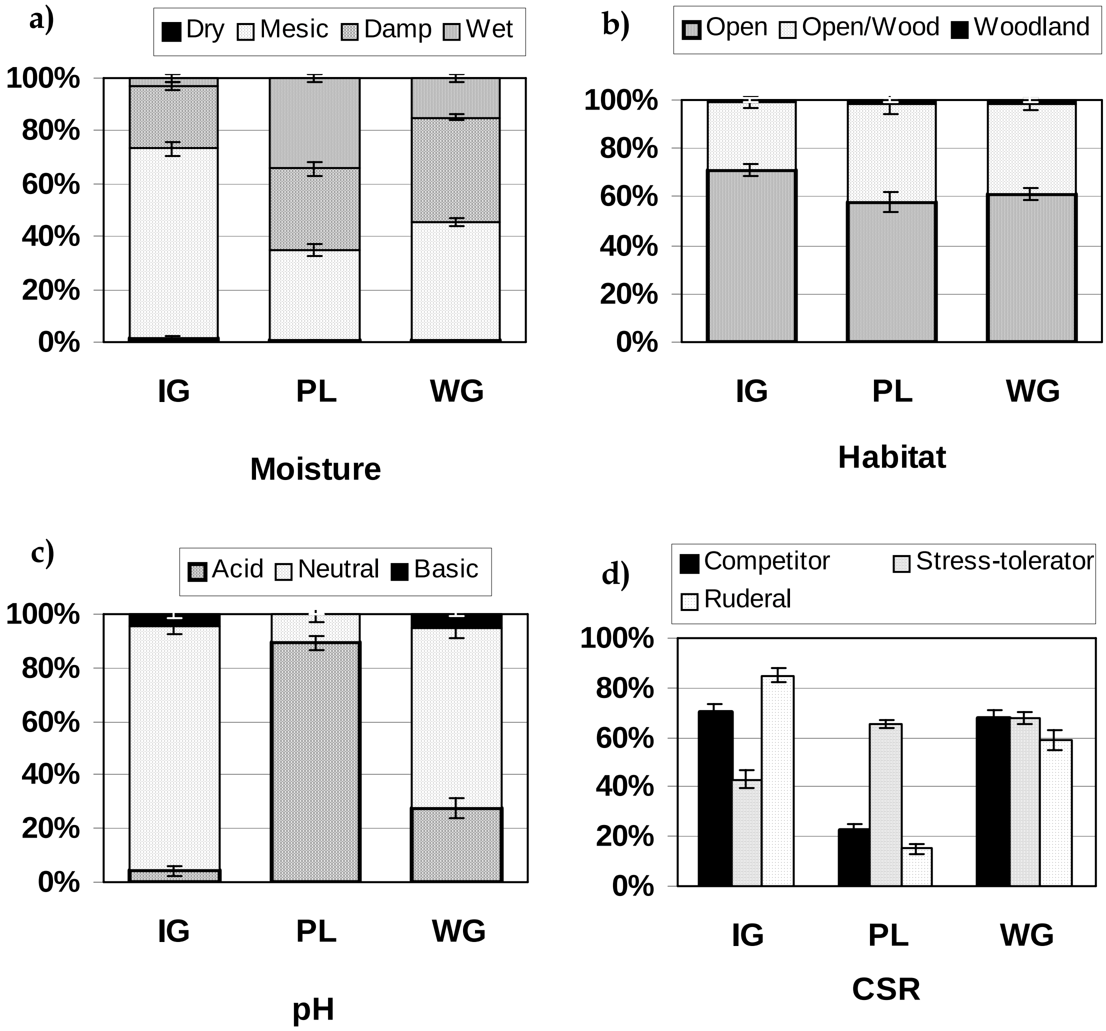


Figure 4. Ecological preferences and strategies of plant species in 100 m^2 plots in the three pre-defined habitat groups, improved grassland (IG), peatlands (PL) and wet grasslands (WG): a) moisture preferences, b) affinity to open or woodland habitats, c) pH preferences, d) CSR strategy (note that a species can have more than one strategy). Percentages are site averages with error bars at the top of each category showing standard error.

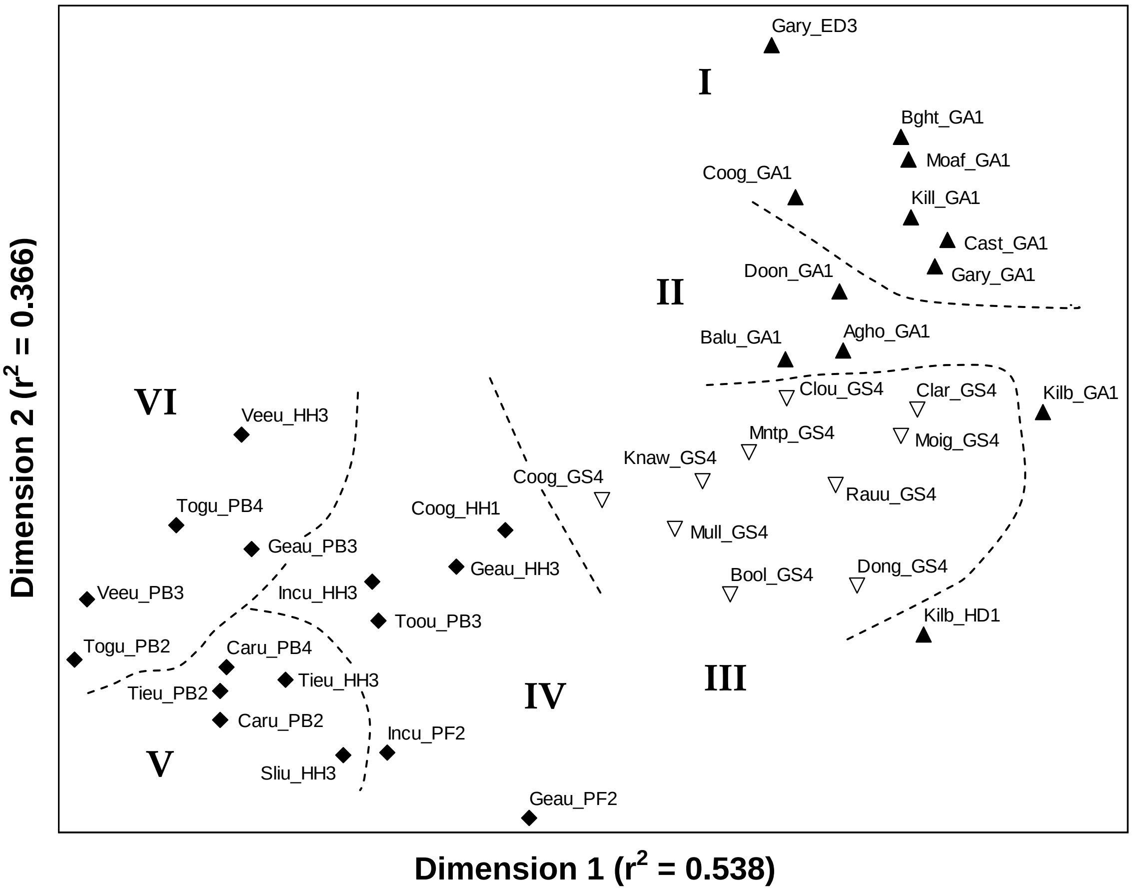


Figure 5. NMS ordination and flexible-beta clustering of unplanted site habitat survey (DAFOR scale) data. Three-cluster solution is indicated by different symbols. Dashed lines separate groups formed at the six-cluster stage, which are also identified by Roman numerals. Sites are indicated by site code and habitat code (after Fossitt 2000). Habitat code definitions are given in Appendix 3.

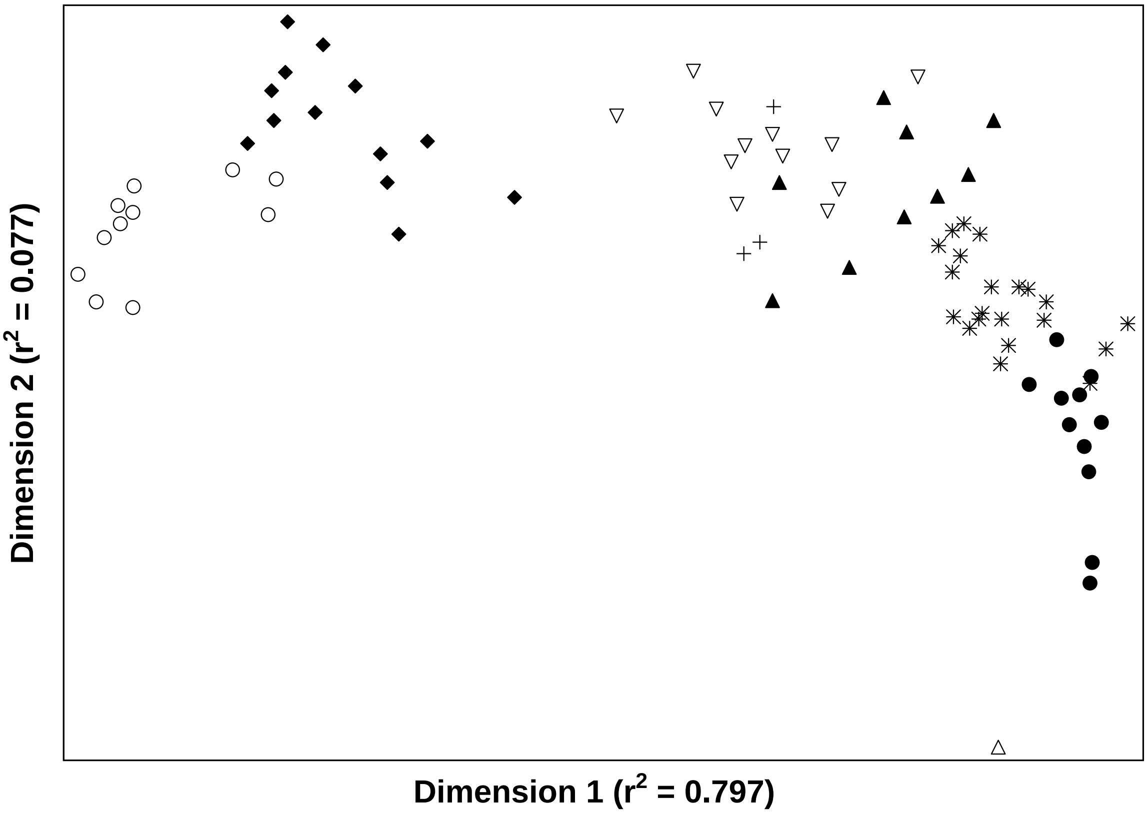


Figure 6. NMS ordination of unplanted 100 m^2 vegetation plot presence/absence data. Symbols indicate results of a flexible-beta ($\beta = -0.25$) clustering into 8 groups: ○ = Group 1, ◆ = Group 2, ▽ = Group 3, + = Group 4, ▲ = Group 5, ✳ = Group 6, ● = Group 7, △ = BGHTF3.

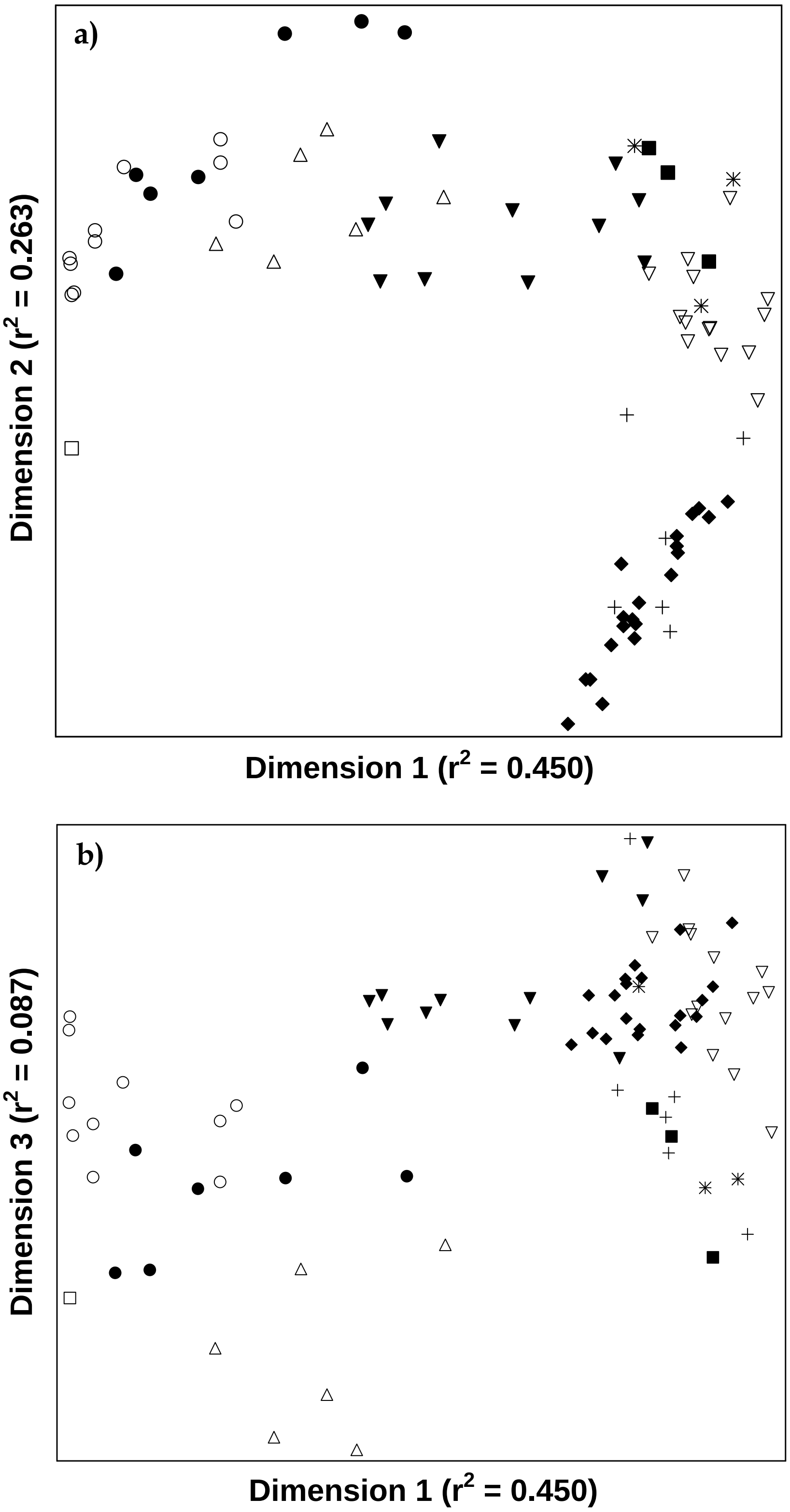


Figure 7. NMS ordination of unplanted 4 m^2 vegetation plot abundance data: a) dimensions 1 & 2, b) dimensions 1 & 3. Symbols indicate results of a flexible-beta (β = -0.25) clustering into 10 groups: ○ = Group A, ● = Group B, △ = Group C, ▼ = Group D, ✳ = Group E, ■ = Group F, ▽ = Group G, + = Group H, ◆ = Group I and □ = TOGUF3.

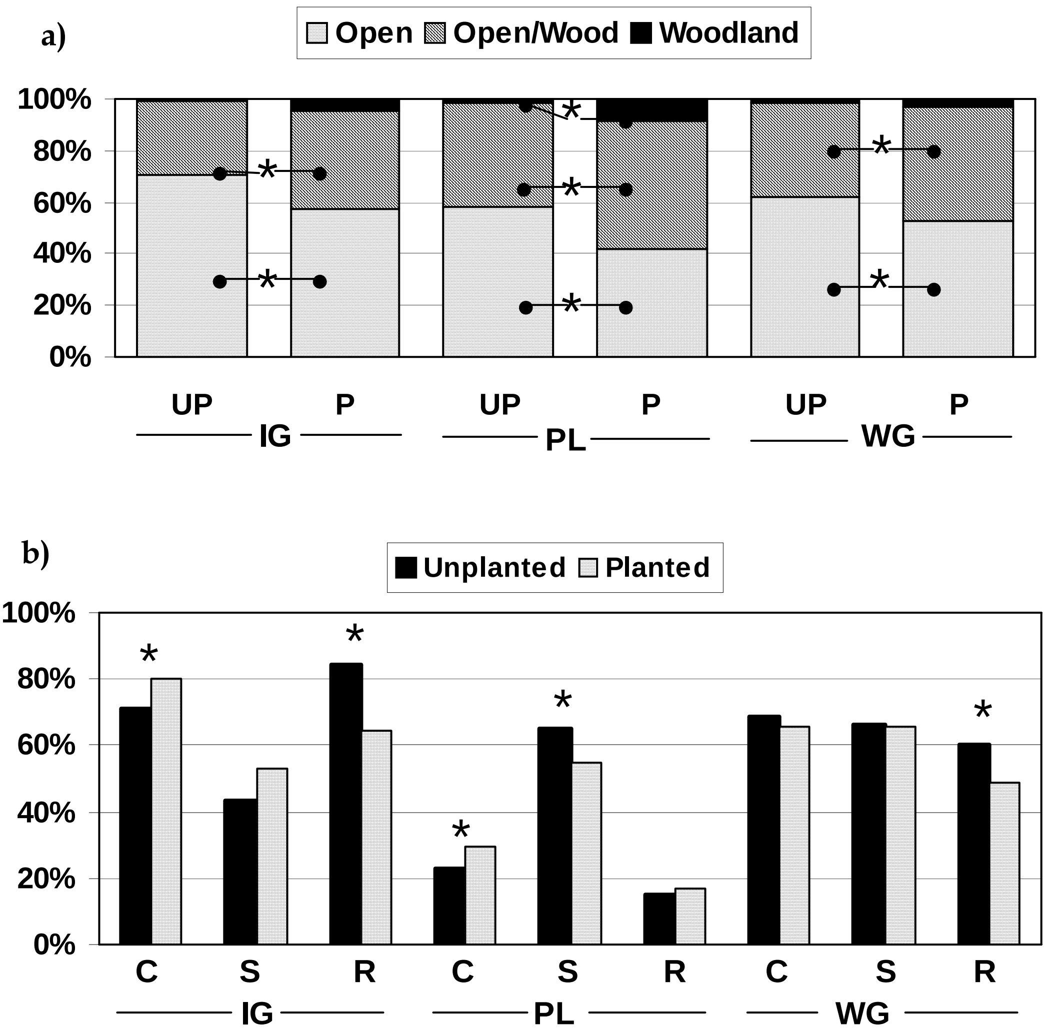


Figure 8. Mean percentage of species in 100 m^2 plots in unplanted (UP) and planted (P) improved grasslands (IG), peatlands (PL) and wet grasslands (WG): a) species with affinities to open habitats, open or woodland habitats or woodland; b) vascular plants classified as competitors (C), stress-tolerators (S) and/or ruderals (R). Asterisks indicate significant ($p \leq 0.05$) differences between unplanted and planted sites for a species class within a given habitat group, according to paired Wilcoxon signed rank tests.

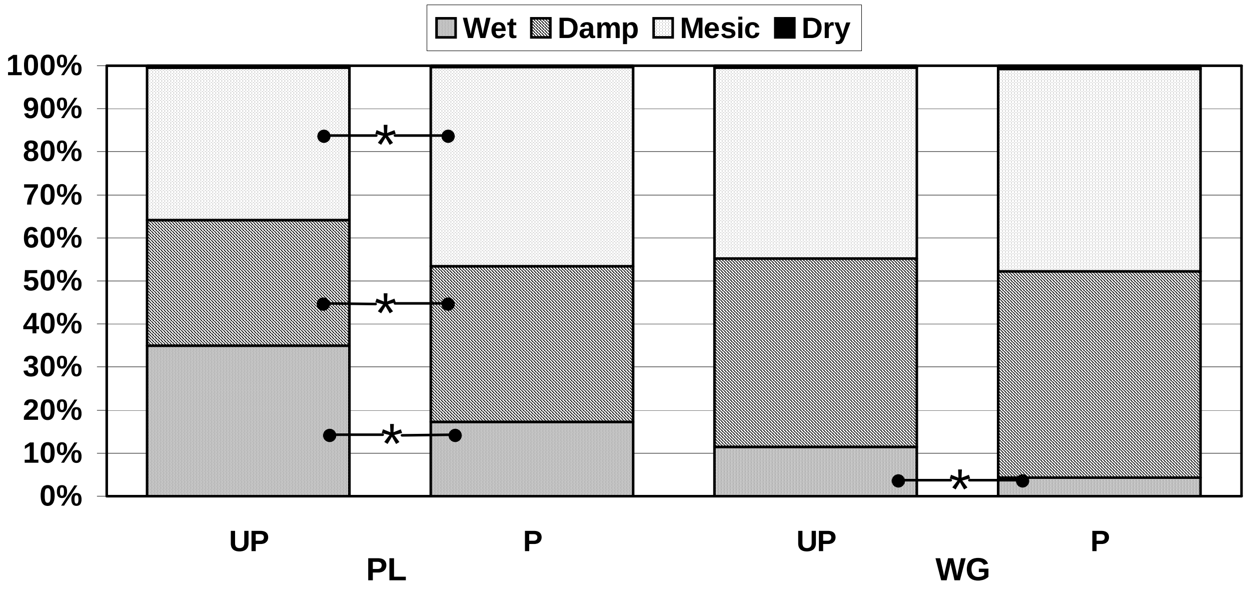


Figure 9. Mean percentage of plant species with affinities to wet, damp, mesic or dry conditions in 4 m² plots in unplanted (UP) and planted (P) peatlands (PL) and wet grasslands (WG). Asterisks indicate significant differences between unplanted and planted sites for a given moisture class within the same habitat group, according to significant pairwise Wilcoxon signed rank tests.

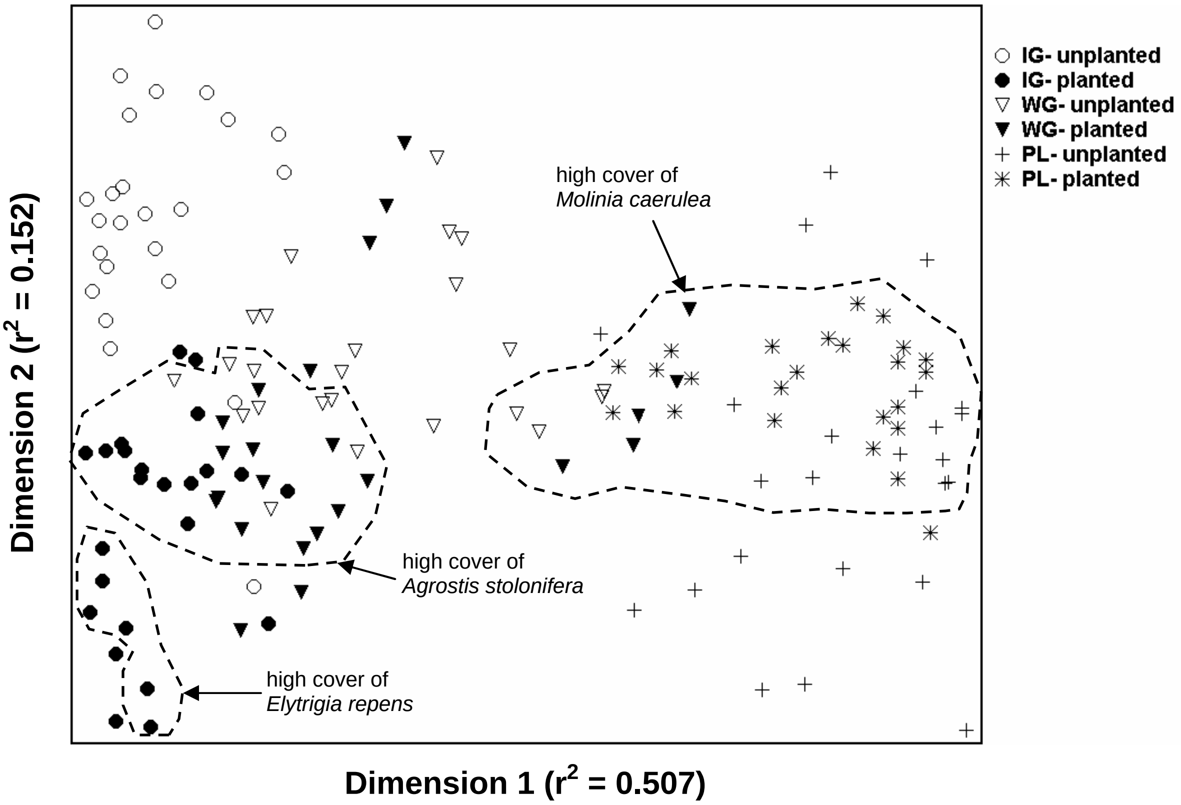


Figure 10. NMS ordination of planted and unplanted 4 m² vegetation plot abundance data. Symbols indicate habitat group and afforestation status.

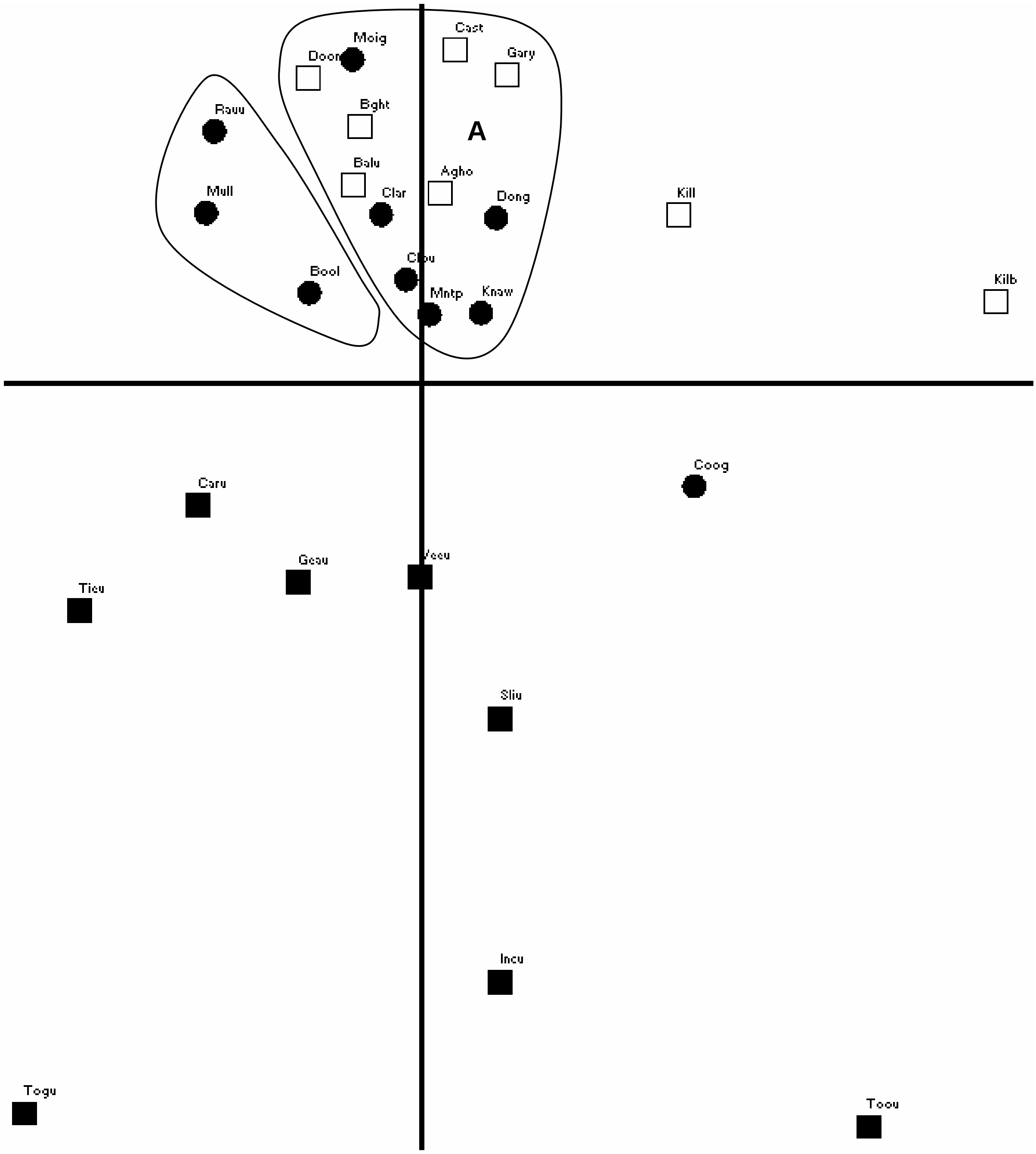


Figure 11. NMS ordination of open habitat-associated hoverfly species across all unplanted sites. The NMS ordination produced a two axis solution. The stress was 14.4 and the final instability was 0.00426. The r^2 values are axis 1 0.36, and axis 2 0.53. Peatland sites are represented by closed squares, improved grassland sites by open squares and wet grassland sites by circles. Groups identified from cluster analysis of the improved and wet grassland sites are outlined.

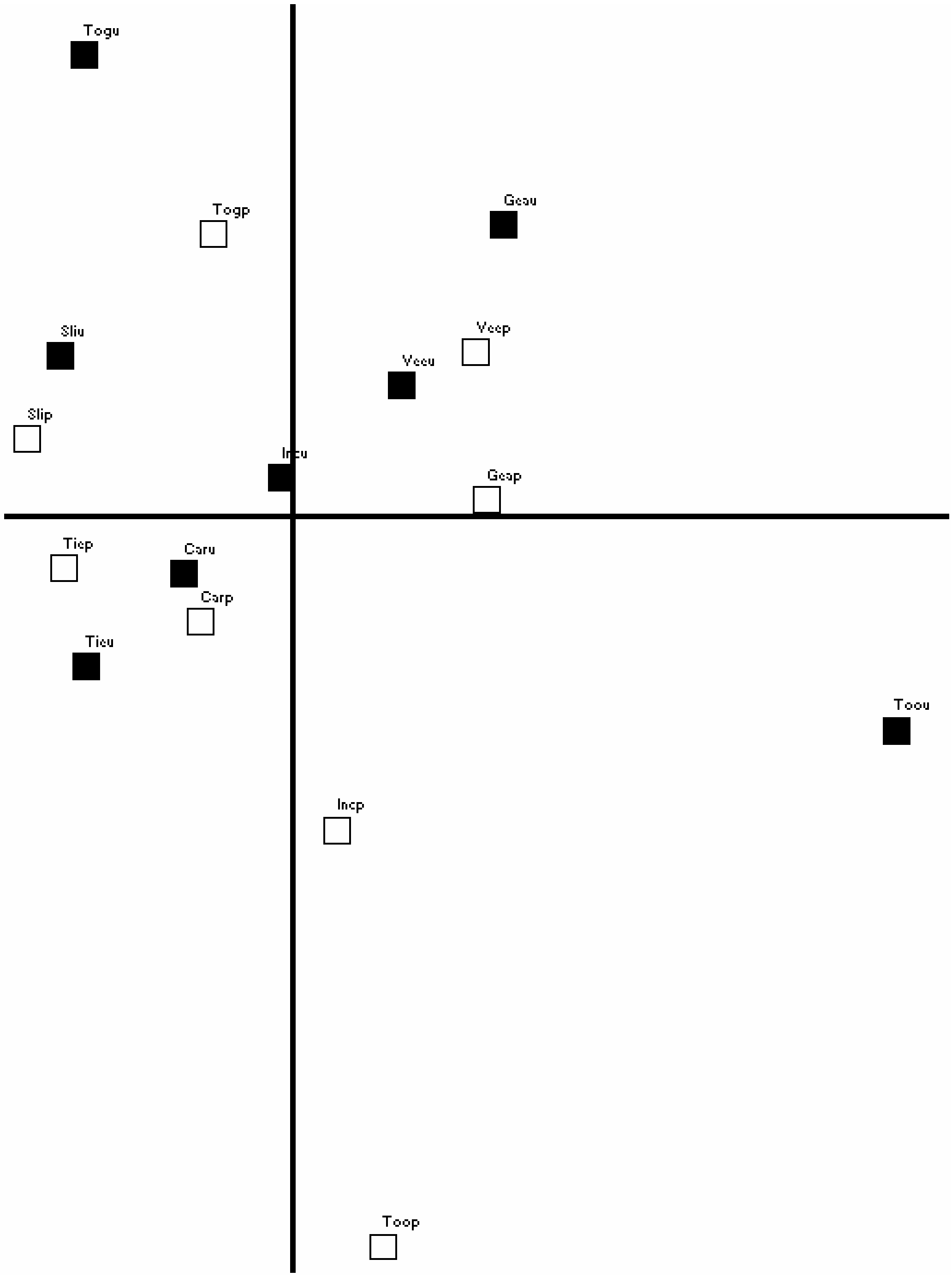


Figure 12. NMS ordination of open habitat-associated hoverfly species across planted and unplanted peatland sites. The NMS ordination produced a two-axis solution. The stress was 23.5 and the final instability was 0.00091. The r^2 values are axis 1 0.16, and axis 2 0.67. Unplanted sites are represented by filled symbols and planted sites by open symbols

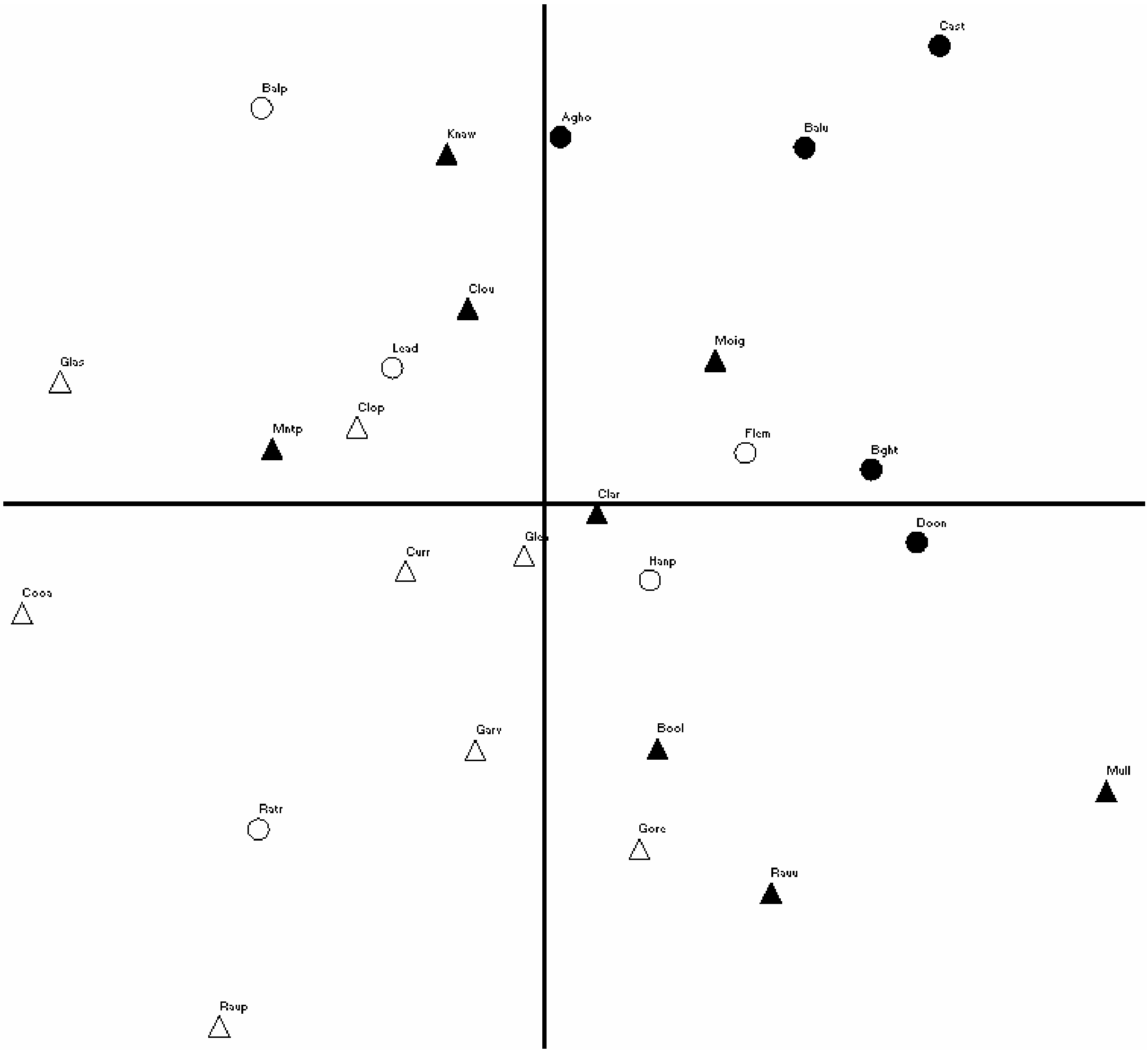


Figure 13. NMS ordination of open habitat-associated hoverfly species across planted and unplanted improved and wet grassland sites. The NMS ordination produced a two-axis solution. The stress was 19.7 and the final instability was 0.00136. The r^2 values are axis 1 0.37, and axis 2 0.37. Improved grassland sites are represented by circles and wet grassland sites by triangles. Unplanted sites are represented by filled symbols and planted sites by open symbols.

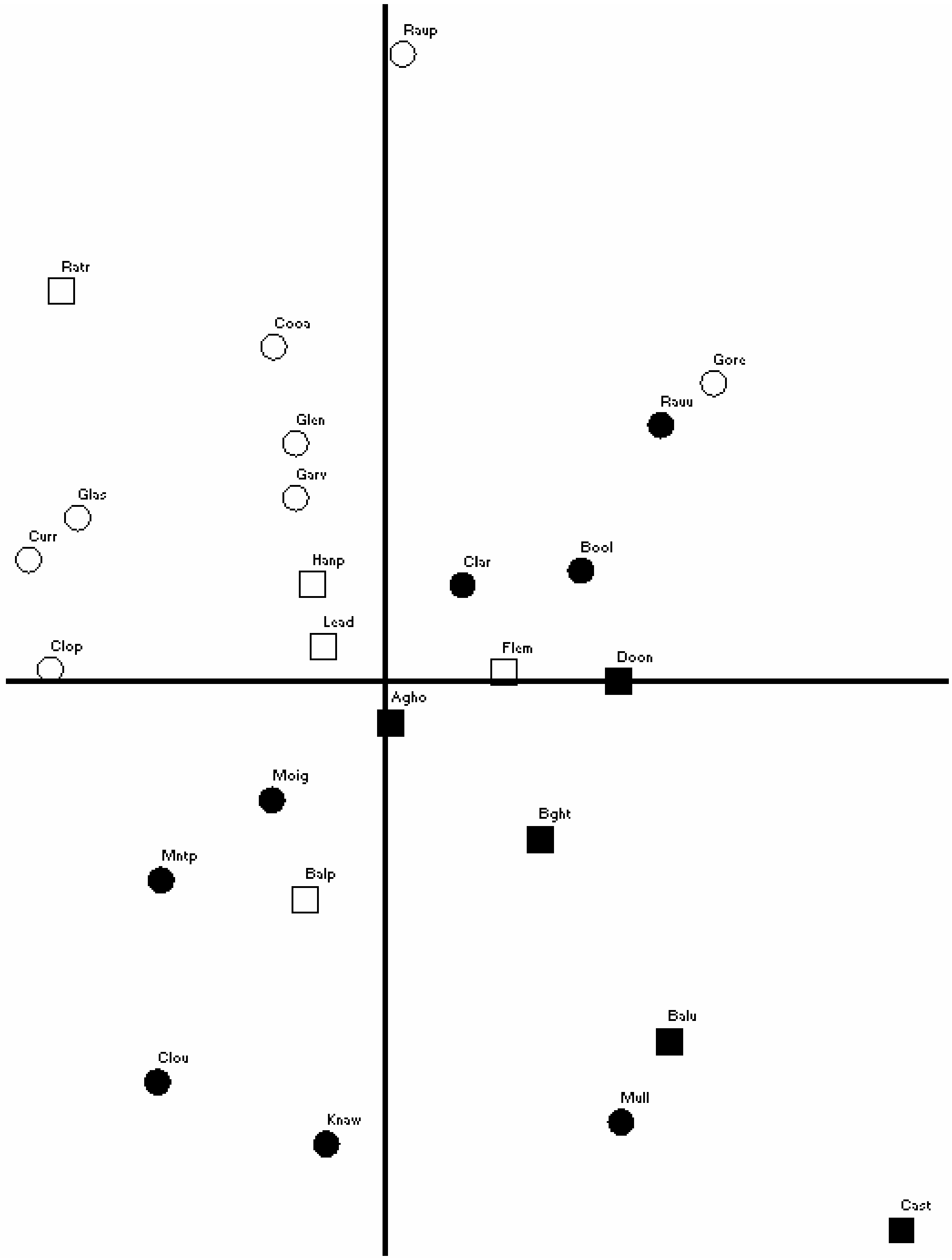


Figure 14. NMS ordination of woody vegetation-associated hoverfly species across planted and unplanted improved and wet grassland sites. The NMS ordination produced a three-axis solution, of which axes 1 and 3 are shown in this plot. The stress was 13.6 and the final instability was 0.00498. The r^2 values are axis 1 0.21, axis 2 0.14, and axis 3 0.47. Improved grassland sites are represented by squares and wet grassland sites by circles. Unplanted sites are represented by filled symbols and planted sites by open symbols.

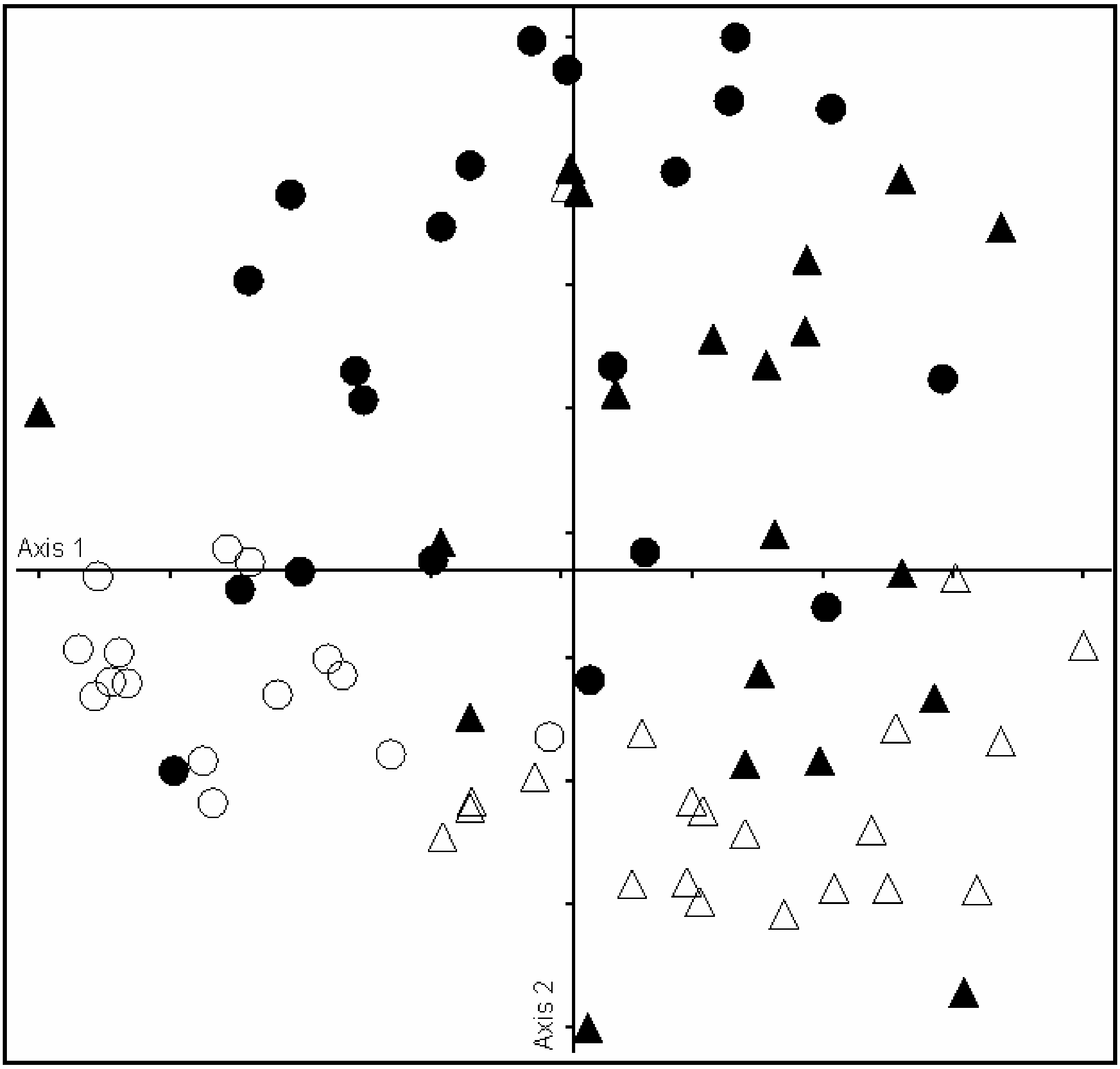


Figure 15. NMS ordination of spider assemblages among the grassland plots: + = Wet grassland-standard; % = Wet grassland-supplementary;) = Improved grassland-standard; # = Improved grassland-supplementary. Final stress = 14.01; Final instability = 0.001; Axis 1 r^2 = 0.26; Axis 2 r^2 = 0.20.

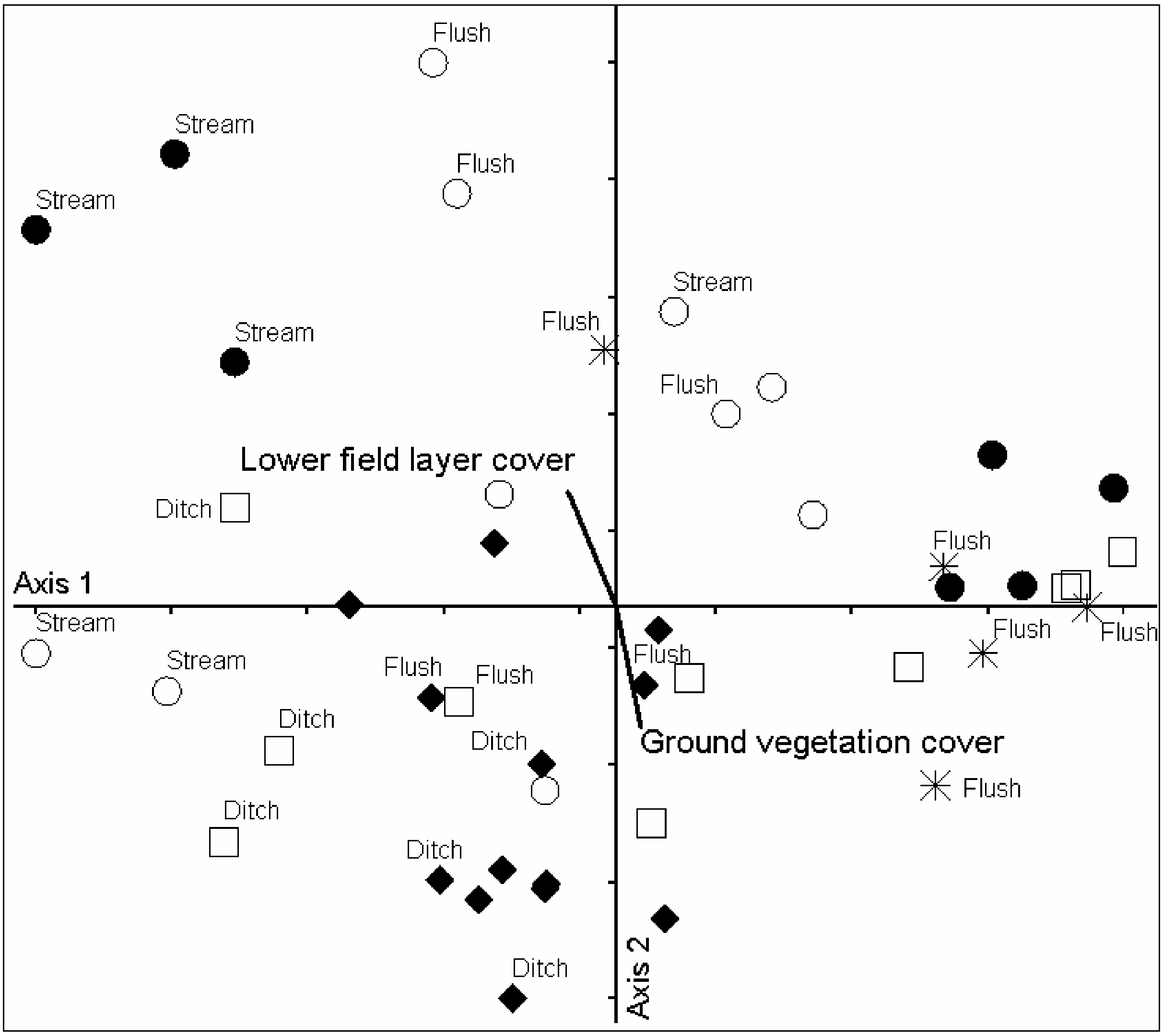


Figure 16. Joint biplot (NMS) of the spider assemblages among the peatland plots with the Irish habitat classifications (Fossitt 2000): ∀ = Upland blanket bog; # = Lowland blanket bog; э = Wet heath;) = Cutover bog; ✳ = Poor fen and flush. Text adjacent to plot symbol denotes the type of supplementary plot sampled: Stream = edge of streams; Ditch = edge of ditches; Flush. Standard plots are without text. Final stress = 13.07; Final instability = 0.0004; Axis 1 r^2 = 0.47; Axis 2 r^2 = 0.18.

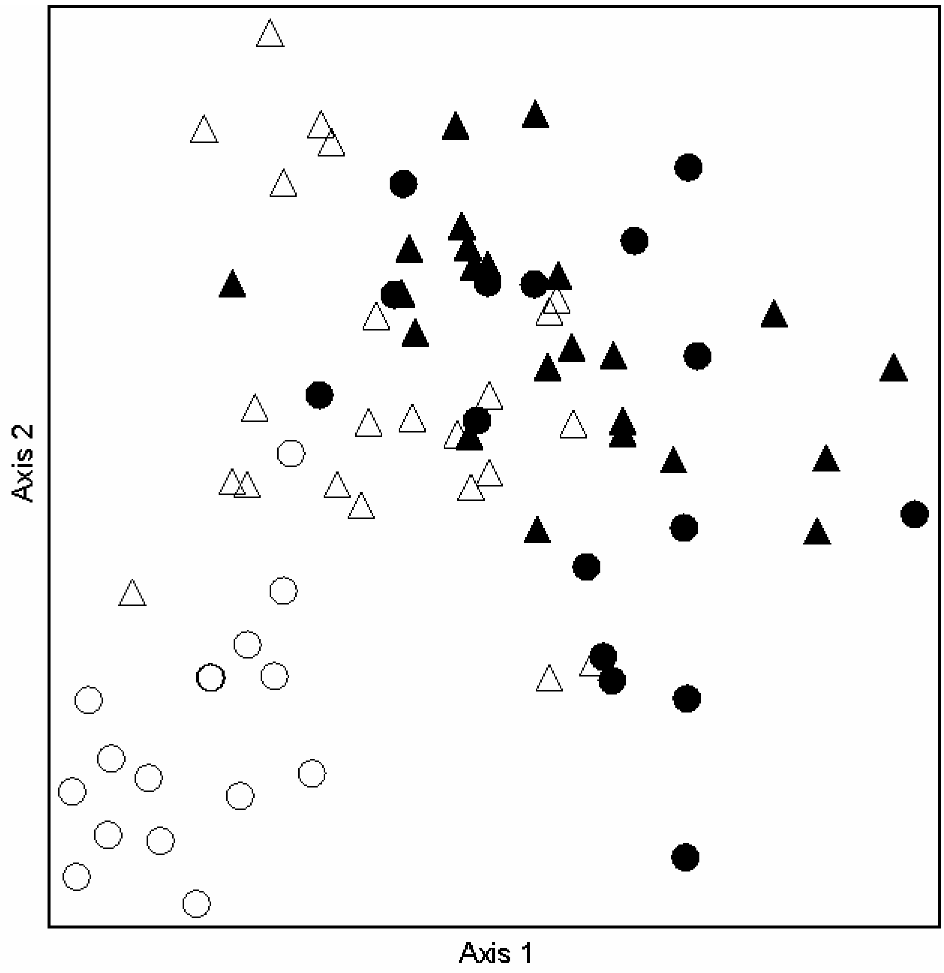


Figure 17. NMS ordination of the spider assemblages among the unplanted and planted standard plots in the grasslands:) = improved grassland unplanted; # = improved grassland planted; += wet grassland unplanted; % = wet grassland planted. Final stress for a 3-dimensional solution = 13.54; Final instability = 0.0004.

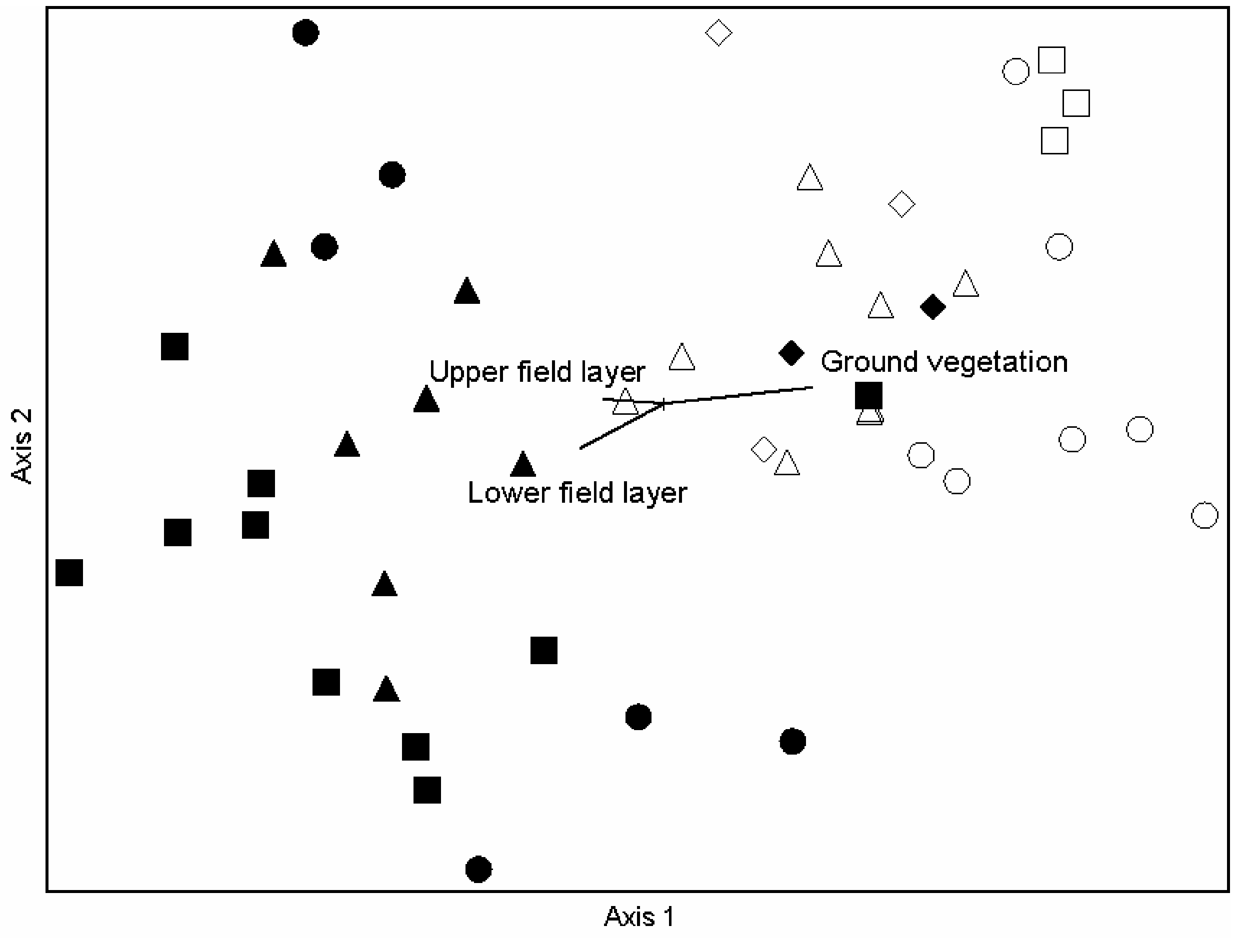


Figure 18. NMS ordination of the spider assemblages among the unplanted and planted standard plots in the peatlands by Irish habitat classification:) = Wet heath unplanted; # = wet heath planted; += upland blanket bog unplanted; % = upland blanket bog planted; ∍ = lowland blanket bog unplanted; ! = lowland blanket bog planted; (= cutover bog unplanted; ∀ = cutover bog planted. Habitat variables with a Pearson correlation r^2 value with the axes > 0.1 are shown. Final stress for a 3-dimensional solution = 12.83; Final instability = 0.0004

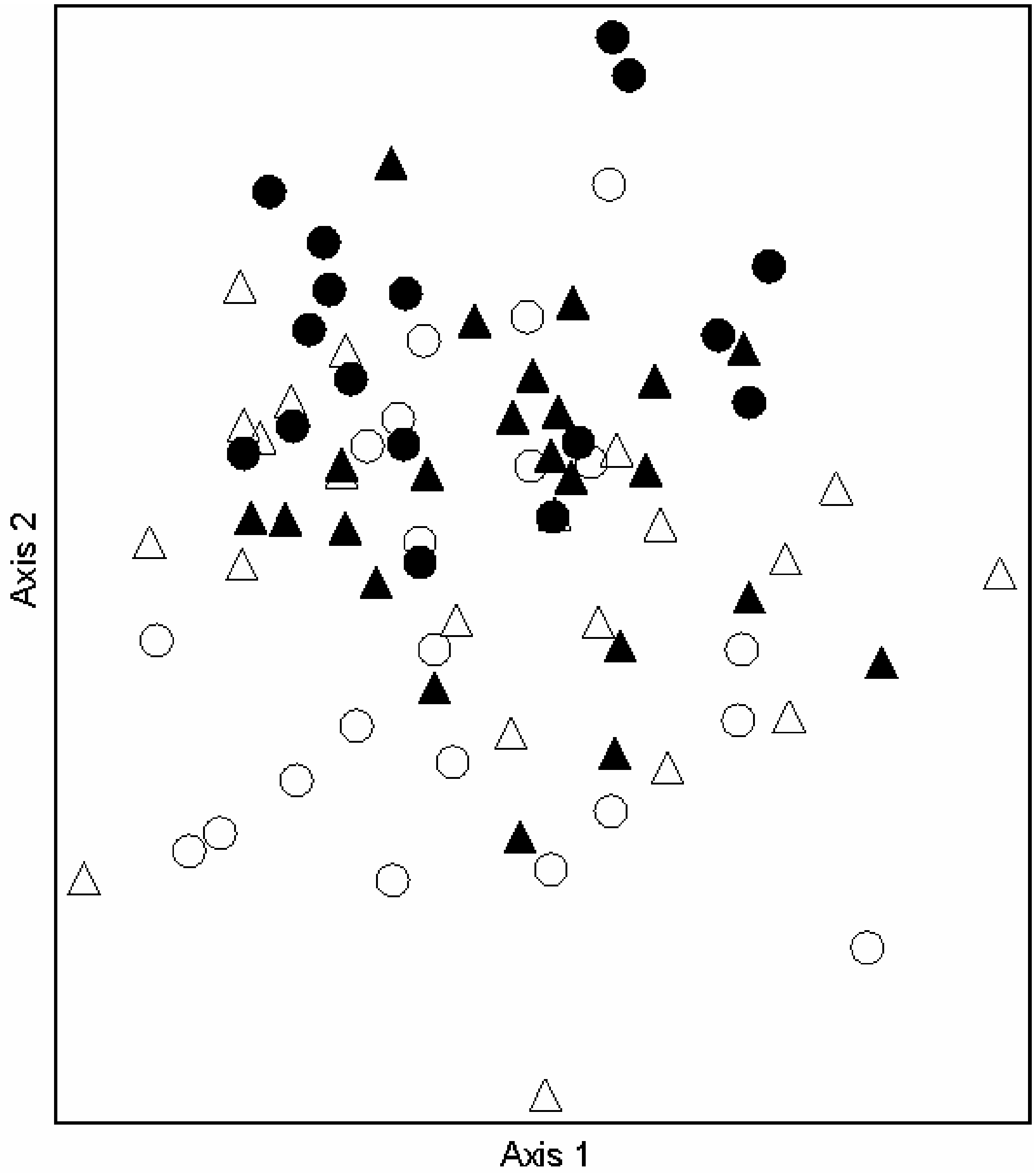


Figure 19. NMS ordination of the spider assemblages among the unplanted and planted supplementary plots in the grasslands:) = improved grassland unplanted; # = improved grassland planted; += wet grassland unplanted; % = wet grassland planted. Final stress for a 3-dimensional solution = 19.37; Final instability = 0.0008.

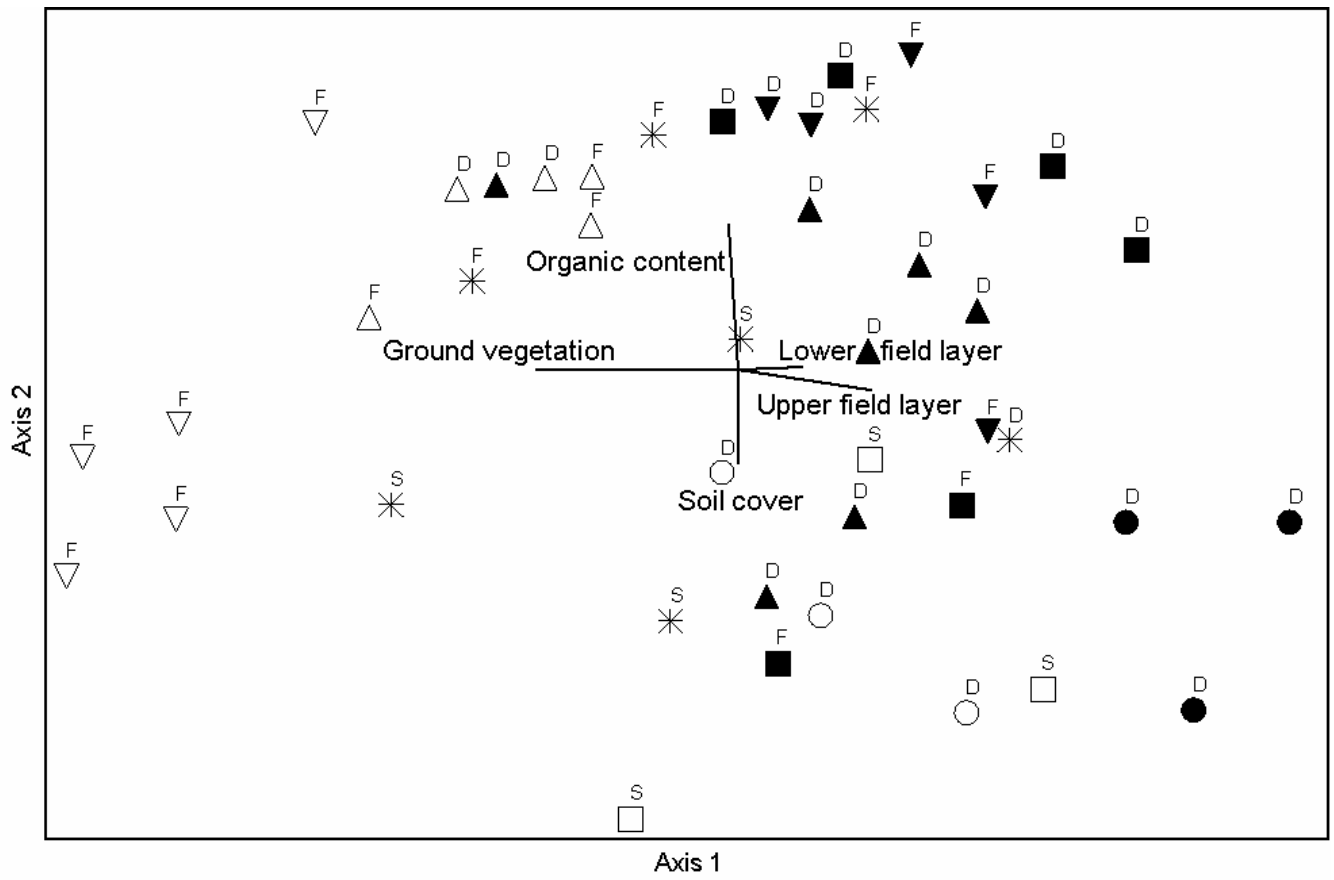


Figure 20. NMS ordination of the spider assemblages among the unplanted and planted supplementary plots in the peatlands by Irish habitat classification:) = Wet heath unplanted; # = wet heath planted; += upland blanket bog unplanted; % = upland blanket bog planted; э = lowland blanket bog unplanted; ! = lowland blanket bog planted; 👍 = poor fen and flush unplanted; & poor fen and flush planted; ✳ = cutover bog unplanted. Letter above plot symbol denotes supplementary plot type: F = flush; S = edge of stream; D = edge of ditch. Habitat variables with a Pearson correlation r^2 value with the axes > 0.1 are shown. Final stress for a 3-dimensional solution = 15.67; Final instability = 0.0004.

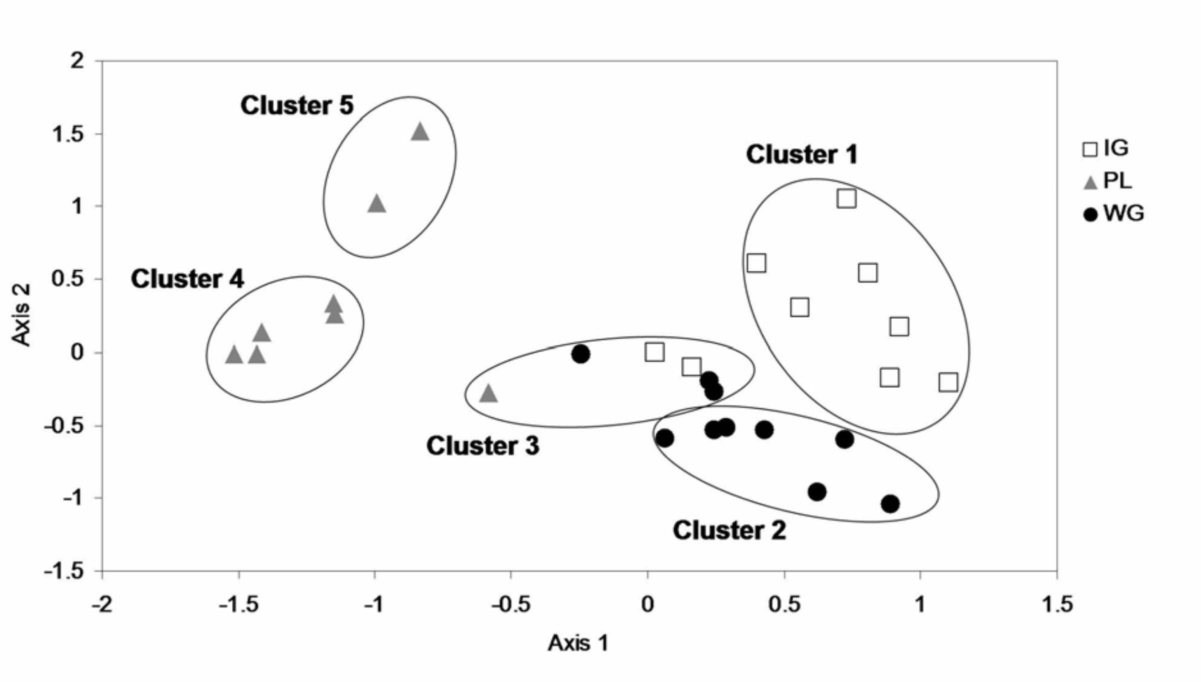


Figure 21. NMS ordination of bird density data from mapping surveys of 27 unplanted sites. Final stress for 2-dimensional solution = 9.046, p = 0.0196 (from 50 Monte Carlo runs). IG = Improved grassland sites, PL = Peatland sites, WG = Wet grassland sites.

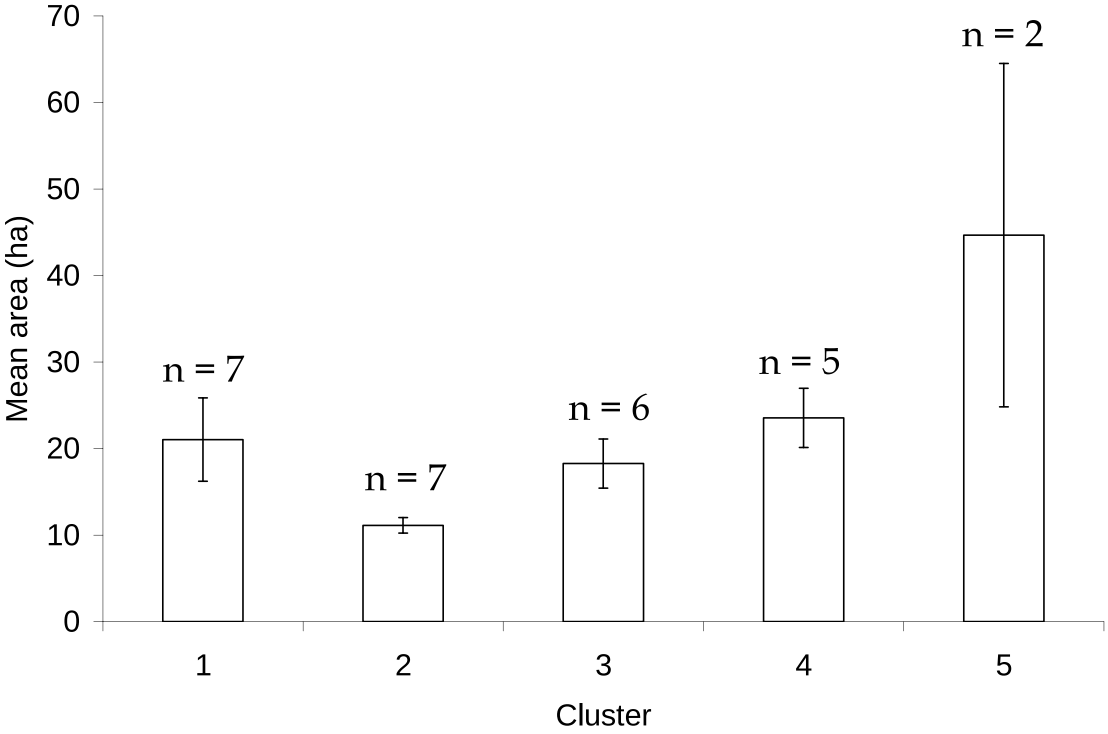


Figure 22. Mean site area in each of the five bird species clusters. Number of sites in each cluster is shown above each bar. Sites in cluster 2 are significantly smaller than sites in cluster 5 ($F_{4,22}$ = 3.70, p = 0.019).

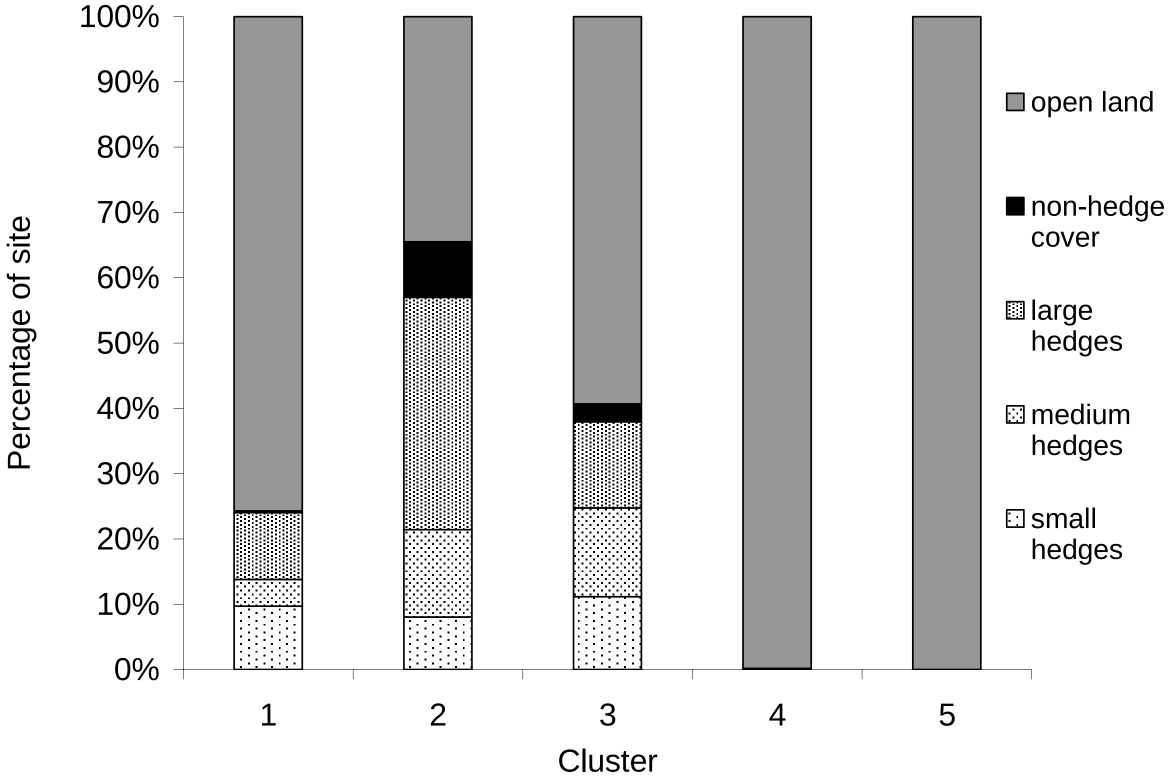


Figure 23. Proportions of cover types in the five bird species clusters. The values for hedge cover shown are the proportions of sites in each cluster within 10m of each of the three hedge categories.

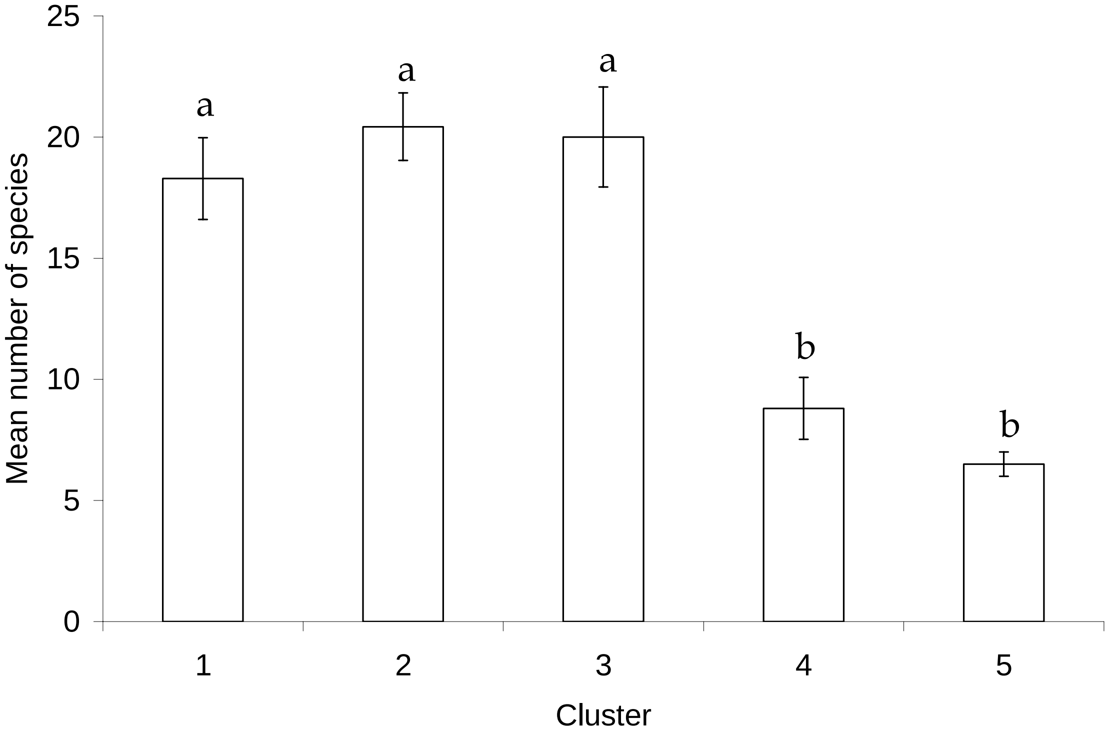


Figure 24. Mean species richness of bird species clusters. Species richness of clusters which share the same letter is not significantly different ($F_{4,22} = 10.6$, $p < 0.0005$).

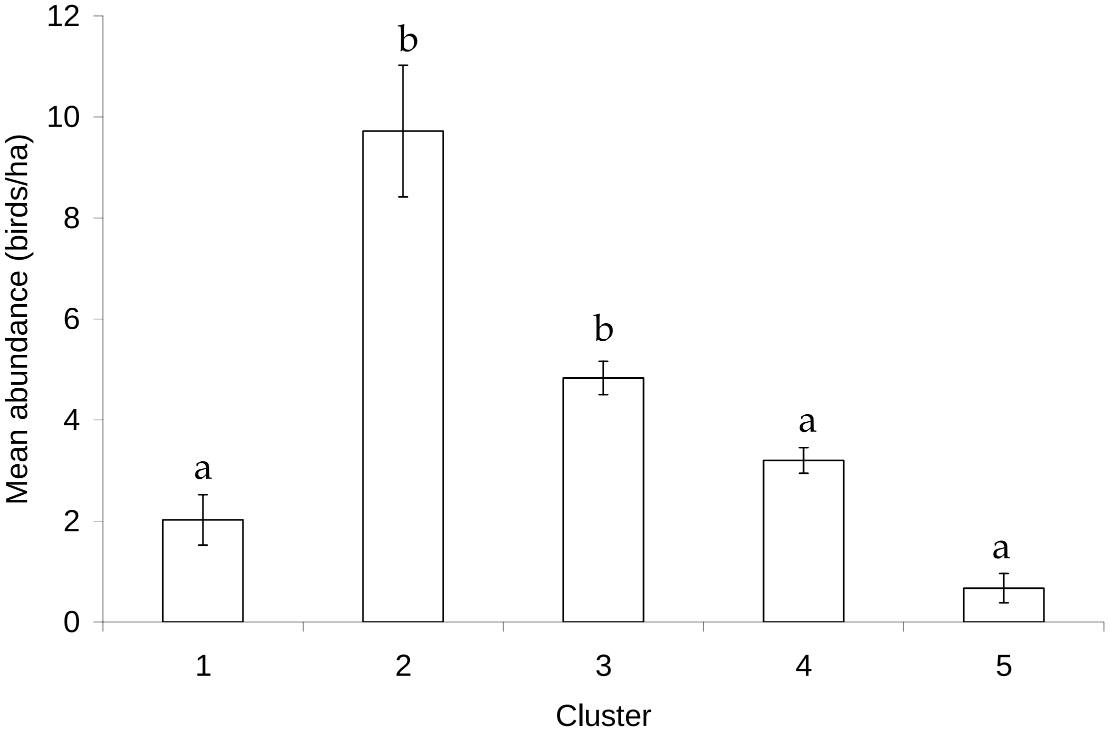


Figure 25. Mean total abundance of birds per ha in each of the 5 cluster groups. Mean abundance of clusters which share the same letter is not differ significantly different ($F_{4,22} = 15.9$, $p < 0.0005$).

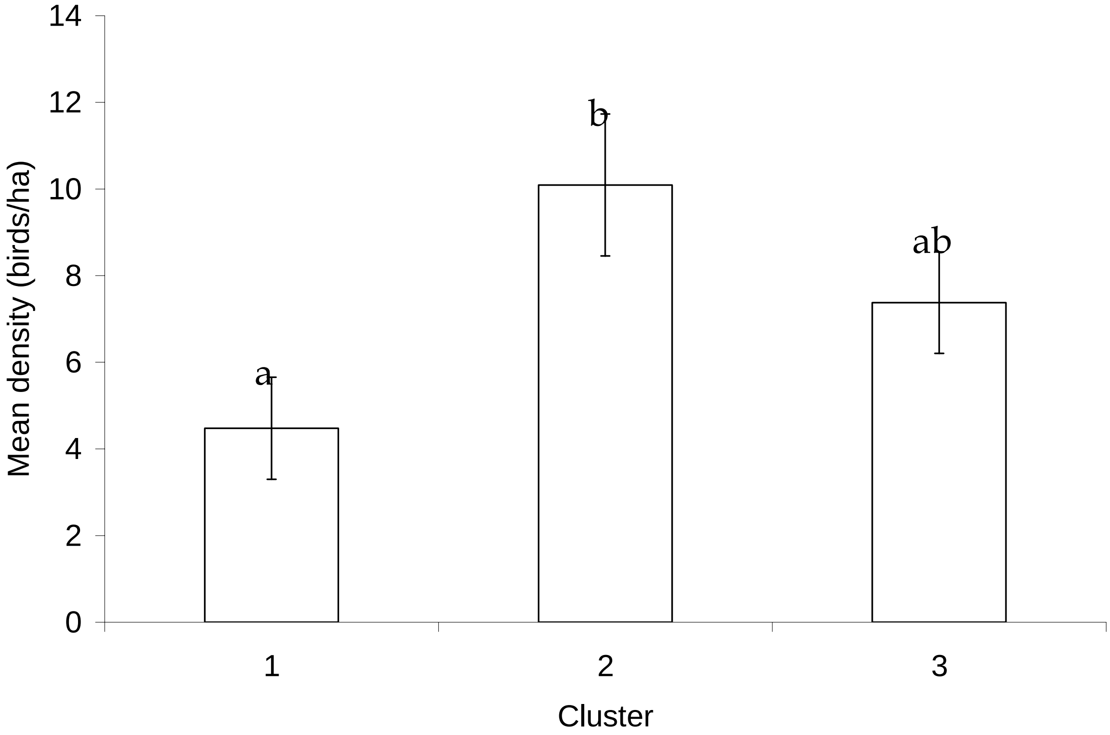


Figure 26. Mean density of birds within 10 m of hedges in bird species clusters 1, 2 and 3 (the only clusters whose sites had hedges). Mean abundance of clusters which share the same letter is not differ significantly different ($F_{2, 17} = 4.9$, p = 0.021).

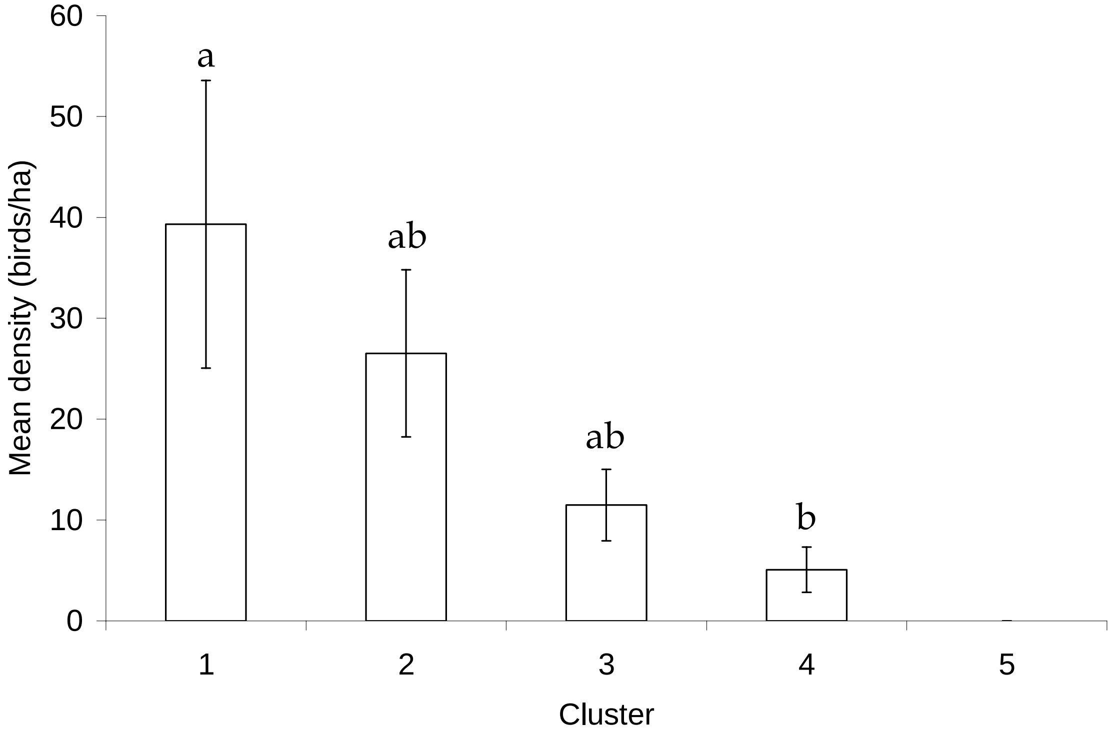


Figure 27. Mean density of birds in areas of non-hedge tree and shrub cover in the five bird species clusters (cluster 5 has no areas of non-hedge tree and shrub cover). Mean abundance of clusters which share the same letter is not differ significantly different ($F_{3, 17}$ = 3.3, p = 0.046).

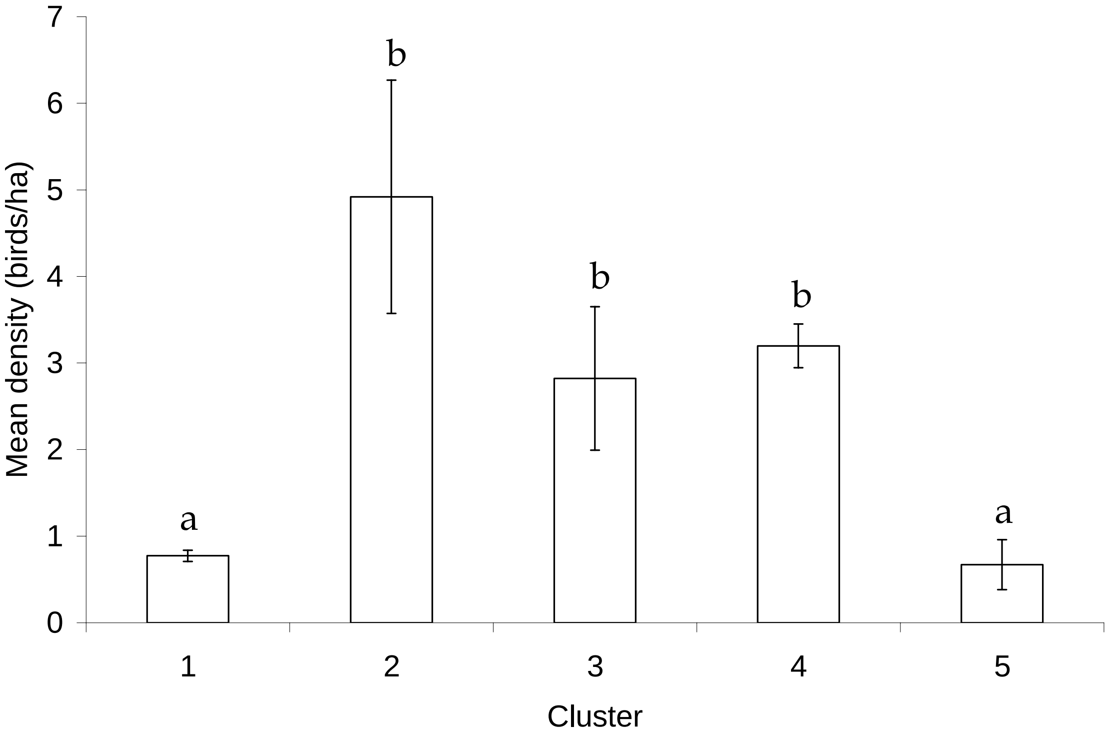


Figure 28. Mean density of birds in areas of open land in each of the five bird species clusters. Mean abundance of clusters which share the same letter is not differ significantly different ($F_{4, 22}$ = 15.92, $p < 0.0005$).

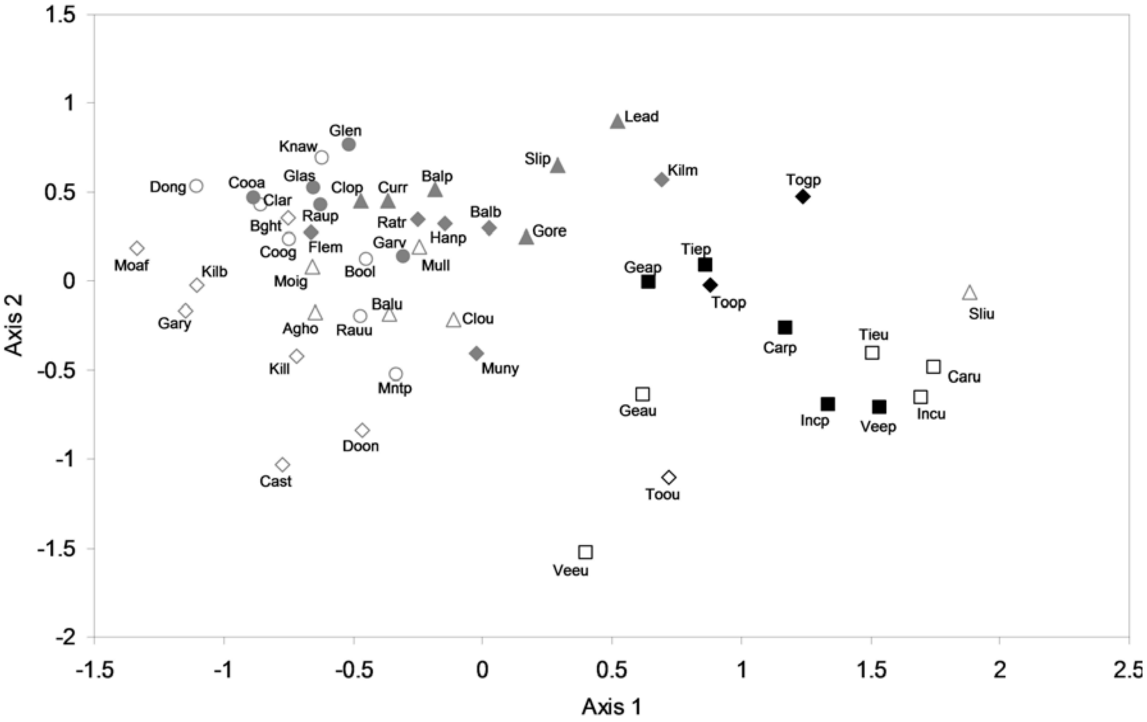


Figure 29. NMS ordination of bird density data from point counts of 49 planted and unplanted sites. Final stress for 2-dimensional solution = 14.251, p = 0.0196 (from 50 Monte Carlo runs). Sites are coded as follows: ◇ = Cluster 1 unplanted, ◆ = Cluster 1 planted, ○ = Cluster 2 unplanted, ● = Cluster 2 planted, △ = Cluster 3 unplanted, ▲ = Cluster 3 planted, □ = Cluster 4 unplanted, ■= Cluster 4 planted, ◇ = Cluster 1 unplanted, ◆ = Cluster 1 planted.

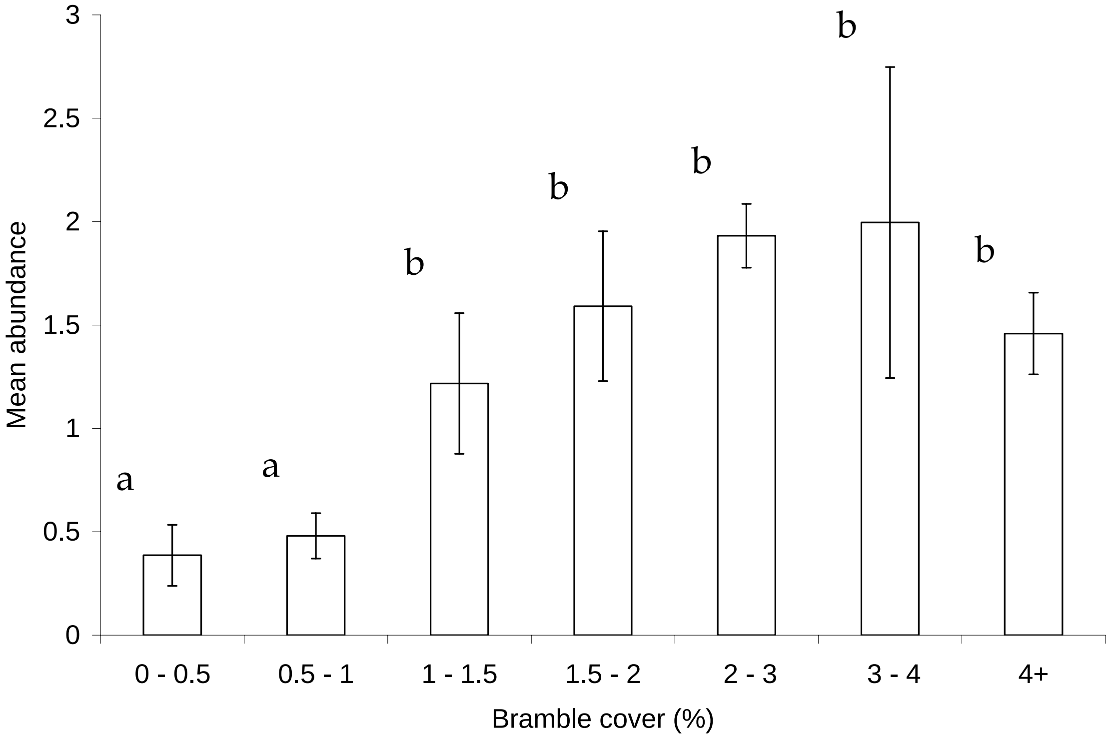


Figure 30. Mean combined abundance (birds/ha) of Grasshopper Warbler, Reed Bunting, Sedge Warbler, Whitethroat and Willow Warbler, in sites with different levels of bramble cover. The difference in abundance between the seven groups is significant (Kruskal Wallis H = 23.73, d.f. = 6, p = 0.0006).

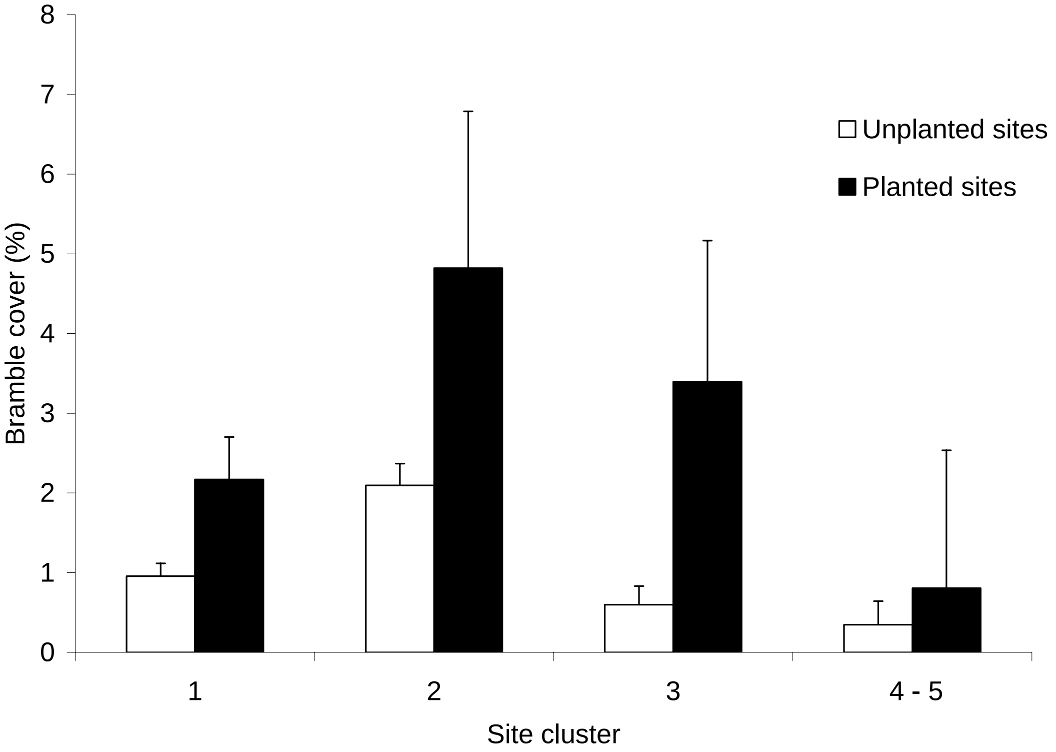


Figure 31. Mean bramble cover of unplanted and planted sites in the different site clusters.

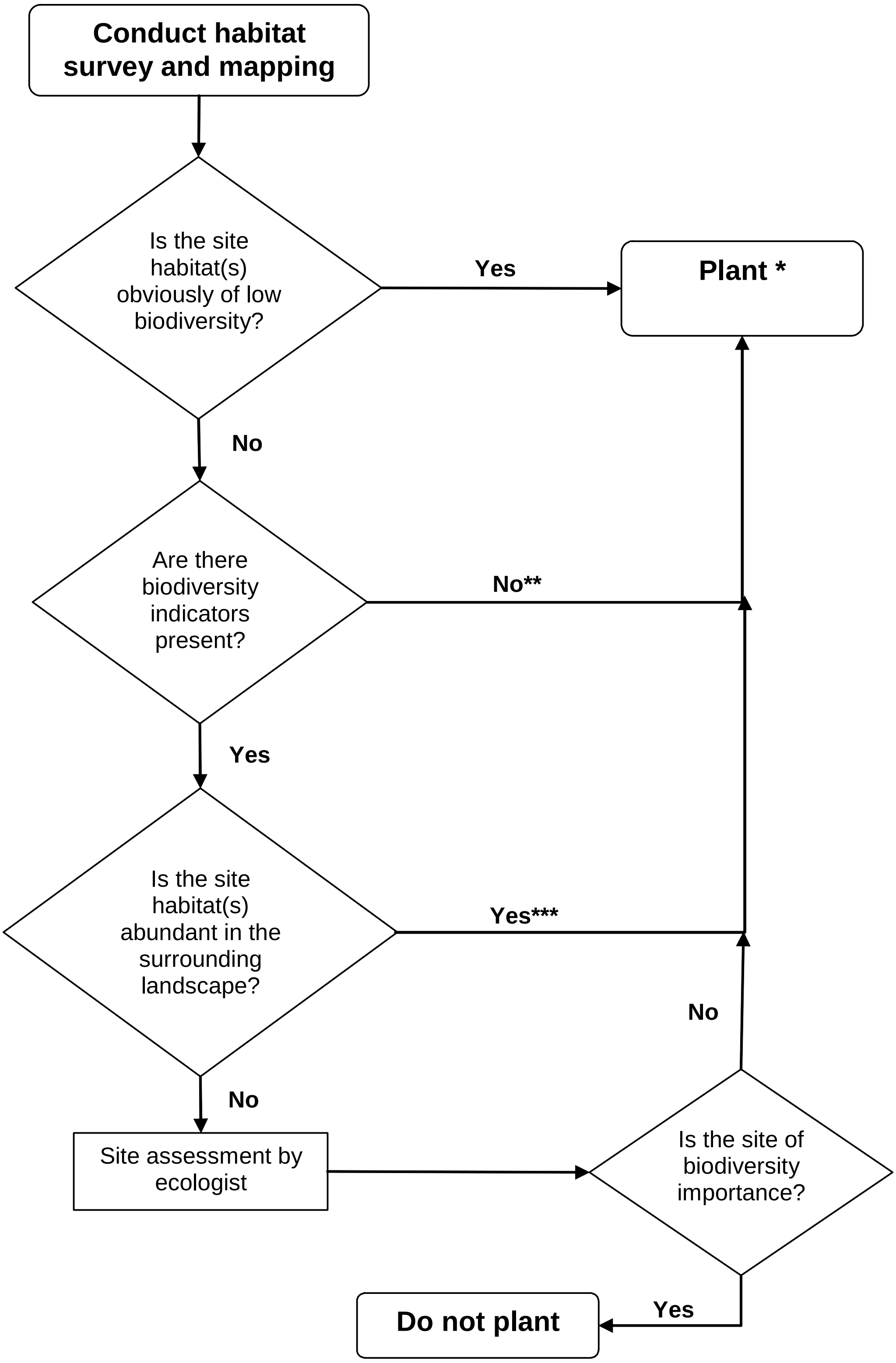


Figure 32. Flow chart outlining the stages in biodiversity assessment prior to afforestation. *Assuming that other criteria (e.g. landscape, water quality) have been met. ** Sites with no biodiversity indicators present may still have high biodiversity and should not be afforested (see Section 8.3.5). *** Some habitat types should never be planted (see Section 8.3.5).