ENHANCING WOODLAND PLANT COMMUNITIES ON COLLIERY SPOIL: AN EXAMINATION OF THE POTENTIAL OPPORTUNITIES FOR NATURE CONSERVATION

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by

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ABSTRACT

New woodlands that are established in areas with little remaining ancient woodland, or in relatively isolated locations, are unlikely to develop a full complement of the animal and plant species associated with older woodlands. In new woodlands, ecosystem development can best be promoted through the appropriate use of ecological restoration techniques. This thesis considers the range of techniques that may be used in ecological restoration, with particular emphasis on the restoration of woodland communities. Methods used in community restoration are described, as are previous approaches to woodland field layer enhancement/diversification. The suitability of colliery spoil as a medium for woodland establishment is considered, and the process of natural vegetation establishment on this substrate examined. Previous research indicates that where woodland develops on colliery spoil, either as a result of natural colonisation or as an end-use following reclamation, it is likely that the community is suitable for colonisation by a number of characteristic woodland plant species. This hypothesis was tested experimentally at two colliery waste sites, Bold Moss Tip and Gillars Green Wood, in St. Helens, Merseyside. Seed of herbaceous woodland species was introduced into the existing field layer of naturally-established woodland/scrub at both experimental sites. A number of woody species were also grown on colliery spoil, both in the field and glasshouse, and measures of plant performance for all species made in relation to experimental mineral nutrient addition.

At both experimental sites, low spoil pH was associated with relatively poor plant establishment. Species appeared to benefit to only a limited degree from the addition of mineral nutrients. A reduction in the number of some introduced species at Bold Moss was associated with probable competition from grasses and herbs. The relative abundance of field layer vegetation at Bold Moss was shown to be partly a function of canopy layer density.

A number of native broadleaved tree and shrub species were successfully grown on colliery spoil. In the field, growth of transplants was enhanced by addition of mineral nutrients on spoil only where pH remained unaltered. *Quercus robur* appeared to be particularly suited to growth on colliery spoil at Bold Moss, whilst *Sorbus aucuparia* was most suited to the low pH and light levels found at Gillars Green. Trees of three species grown in pots did show clear increases in growth rates in response to mineral nutrient additions over a period of two growing seasons.

It is concluded that, as a method for enhancing the herbaceous species component in the field layer of existing woodland and scrub vegetation on colliery spoil, seed introductions can provide an effective means of establishing populations of some herbaceous plant species. Where the manipulation of spoil pH is undesirable or impracticable, care must be taken to ensure that species used in such introductions are suitably adapted to the often extreme range of chemical characteristics associated with colliery spoil, most particularly low pH. Tree and shrub species should also be considered integral to any enhancement scheme. For nature conservation purposes, it would be preferable to work with natural processes of colonisation and successional change, so as to maintain local and regional landscape and ecological character. In this way community restoration may be used to effectively enhance biodiversity at local, regional, and national levels.

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1. 1 Introduction

Nature conservation in Britain has traditionally focused on the protection of the most ecologically valuable habitats. It is now widely accepted however, that this approach will be unable to ensure the long-term preservation of existing levels of biodiversity (Adams, 1996). Such an objective can only be met if efforts are made to enhance the ecological value of the wider landscape, particularly in areas with little remaining semi-natural habitat. Enhancement will involve a range of approaches, including the use of ecological restoration techniques for the establishment of functioning ecosystems with component animal and plant communities. Ecological restoration will be especially important in urban and industrial areas, where a variety of human activities have led to widespread ecological impoverishment.

This part of the thesis outlines some of the existing and potential problems associated with traditional approaches to nature conservation, and introduces the various methods used in ecological restoration. A more detailed examination of the ecological issues involved in the restoration of woodland communities, the main subject of this thesis, is also provided. Nomenclature follows Stace (1991) throughout.

1. 2 Current problems and solutions

Most forms of human land-use, including farming, forestry, mineral extraction and road building, result in a degree of ecosystem degradation. In many parts of the world this process has occured at such a scale that it has culminated in 'degraded landscapes' (Naveh, 1994). In Britain land-use changes have led to an increasing loss of natural variety, both of individual animal and plant species, and the habitats and landscapes in which they live. The last fifty years in particular has seen a reduction in the extent and 'quality' of many semi-natural habitat types; those areas of the landscape of greatest value to native wildlife (Rackham, 1986; Munton, 1983). In response to these losses, efforts have been made to protect and conserve some of the most valuable areas of semi-natural habitat, the cornerstone for much of this policy being the 1949 National Parks and Access to the Countryside Act. In an attempt to meet the nature conservation aims of the Act, a succession of government conservation agencies have concentrated on the identification, establishment, and protection of 'key areas' (e.g. Sites of Special Scientific Interest

[SSSI's] and National Nature Reserves [NNR's]) for nature conservation (Ratcliffe, 1977). More recently however, concern has been expressed at a number of existing and potential threats to the biological value of such sites (Adams, 1996), so much so that it is now recognised that conservation policies centred around the protection of key areas alone are unlikely to meet future nature conservation objectives.

1. 3 Weaknesses of the existing system

It is already accepted that many key areas have either received insufficient protection - damage to SSSI's has been well documented (Adams, 1993; Marren, 1993) - or suffered a loss in conservation value in the absence of appropriate management (HM Government, 1994). The site-based approach to nature conservation inherent in the selection of key areas is, in the long-term, also likely to impose a number of 'ecological constraints' (Adams, 1993) on the value of such sites. These arise as many sites are relatively isolated, the plant and animal communities within them consequently being more vulnerable than larger, more widely dispersed populations would be; both to a range of external threats (Janzen, 1986) that emanate from the surrounding landscape, and to the long-term and often unknown effect of isolation on genetic fitness and population viability (Gilpin and Soule, 1986).

A further threat, with the potential for initiating widespread changes in all ecological communities, comes from global climatic change. In the next century global warming is likely to have a significant affect on the structure and composition of semi-natural plant communities (Marrs, 1990), and on the distribution and population dynamics of invertebrate, bird, and mammal populations (Kruuk, 1990; Marquiss and Newton, 1990; Watt, Ward and Eversham, 1990). Whilst ecological systems are intrinsically dynamic, the migration responses of many groups - particularly in a highly fragmented landscape - may not be sufficient to keep apace of the 120-150 km per century northward shift of summer isotherms predicted for the midlatitudes of the northern hemisphere (Solomon and Cramer, 1993).

1. 4 New priorities for nature conservation

The challenges facing conservation policy makers and managers in the next century clearly demand more than the continued protection, management and monitoring of only the most valuable semi-natural habitats. In particular, much more emphasis needs to be placed on policies aimed at enhancing biodiversity within the wider landscape. Whilst well-defined areas of greatest biological quality have continued to be the focus for nature conservation policy and practice in Britain, less attention has been paid to the areas of greatest biological 'quantity', i.e. the landscape at large. As only around 9 per cent of the land area of Britain is currently protected as SSSI's under the 1949 Act (HM Government, 1994), the majority of Britain's biological resources are found in a landscape that often receives very little protection. The ecological value of this landscape is also far from resistant to detrimental change. The 1990 Countryside Survey for example, has shown that while the total area of most semi-natural habitat types has remained constant over the last ten to fifteen years, plant species diversity in the same habitats has generally declined (Barr *et al.*, 1993).

Action at a local level will be an important part of this process, as it is at this level that many important land-use planning decisions are made. Planning Policy Guidance Note 9 on Nature Conservation (PPG 9) (Department of the Environment, 1994) and Local Biodiversity Action Plans have already served to increase the value of nature conservation in the assessment of such planning decisions for many local authorities (Goode and Tyldesley, 1995). For many of the most impoverished areas of the country however, ecologically valuable landscapes will need to be 'created', mainly through use of appropriate ecological restoration techniques. Whilst ecological restoration is only one of a number of approaches required to meet the future needs of nature conservation in Britain, it is an integral part of the process, and has already provided a range of ecological (and social) benefits (Bradshaw, 1989).

1. 5 Ecological restoration

1. 5. 1 Principles of ecological restoration

Ecological restoration implies a requirement to ..."restore organisms and their interactions with one another and the physical environment" (Jackson, Lopoukhine and Hillyard, 1995). As stated by Bradshaw (in press) however, a number of words are in common usage in ecological restoration; especially restoration, rehabilitation, remediation and reclamation. The fact that each has a precise definition has tended to lead to some confusion in the practice of ecological restoration, particularly as each activity can be applied to ecosystems, habitats, communities and species. Throughout this thesis the term ecological restoration is used to encompass all actions aimed at bringing land ..."back to the original state... or to a healthy or vigorous state" (Bradshaw, in press). It is also used to include ecosystem restoration - the restoration of the fundamental processes by which ecosystems work (Bradshaw, in press) - habitat restoration - the restoration of the physical and biotic environment of a species or community - and community restoration - the restoration of a set of co-occuring and interacting species. Where these different forms of ecological restoration are considered, they are referred to in the text. Ecological restoration is distinctly different to creative conservation (though the two processes may use some of the same techniques), as creative conservation aims to establish relatively simple communities, without attempting to directly mimic semi-natural habitats. It may therefore lead to the creation of 'unnatural' plant communities, with little resemblance to National Vegetation Classification (NVC) community types (Scott and Luscombe, 1995).

The work in this thesis is specifically concerned with the ecological restoration of woodland plant communities, which is considered in more detail in section 1. 6. By means of a general introduction, the main techniques used in community restoration are firstly discussed below.

1. 5. 2 Community restoration techniques

The ecological restoration of communities can involve the use of any one of three well established techniques - community creation, community transplantation/translocation and community enhancement/diversification (Buckley, 1989a).

1. 5. 2. 1 Community creation

Community creation has been defined as the *de novo* construction of an ecological community (Buckley, 1989b). As practiced, community creation is not necessarily concerned with the establishment of diverse and ecologically complex communities, and can therefore overlap to some degree with creative conservation schemes. Vegetation establishment on a site can take a variety of forms, and is dependent upon limitations imposed by the physical nature of the site, and on its particular designated end-use. All schemes however, attempt to improve the quality of the site through the establishment of some form of vegetation cover.

Historically, a majority of community creation schemes have been undertaken on derelict industrial land (Bradshaw, 1989). The first requirement of such schemes is to overcome factors associated with the substrate that restrict ecosystem development (Bradshaw, 1983a; 1987a). Substrate amelioration is usually achieved through treatment to limit the effects of these factors (in particular nutrient deficiencies and toxicity), or by covering the substrate with imported or stored top soil (Bradshaw, 1983a). In this way the process of primary succession can be artificially assisted (Bradshaw, 1987b), or conditions modified to allow the introduction of suitable species.

For all sites where species introductions are necessary, land designated for amenity/recreational use has traditionally been sown with seed mixtures containing legumes and hard-wearing grasses (such as *Lolium spp.*), often in conjunction with the planting of trees and areas of woodland (Samuel, 1991). The creation of ponds and lakes, whilst a relatively expensive option, has proved to be a popular component of community creation schemes, as it has almost immediate value, both as a wildlife habitat and an educational resource (Bickmore and Larard, 1989). In the last decade particularly, community creation has become increasingly directed towards maximising the conservation value of sites (Bradshaw, 1989). With the use of diverse grass/herb seed mixtures and suitable management techniques for example, valuable grassland habitats have been created on a range of substrate types (Wells, 1983; Davis and Coppeard, 1989).

In some parts of Britain a more passive form of community creation may be appropriate. This strategy, termed 'habitat duplication', was advocated by the Nature Conservancy Council (NCC) in the late 1980's (Newbold, 1989), and involves encouraging the natural colonisation (secondary succession) of plant and animal species on land next or near to existing semi-natural habitats (particularly NNR's and SSSI's). Such an approach could be used to enhance the conservation value of large areas of the landscape, through providing networks of new habitats and conservation field margins to link up with existing areas of semi-natural habitat. Whilst this technique is suitable for many areas, its effectiveness is dependent upon the use of surviving remnants of a range of habitat types as sources of colonising plant and animal species. In some landscapes, particularly highly industrialised ones, such remnants are either relatively rare, or the desired habitat type is not represented.

1. 5. 2. 2 Community transplantation/translocation

Community transplantation/translocation involves the movement of communities and part of the soil profile from a donor to a receptor site (Buckley, 1989a), with the aim of maintaining the habitat in an unaltered form in its new location (Byrne, 1990). This technique requires the movement of as many habitat components as is practicable, usually including the underlying soil. Habitat transplantation/translocation is most often undertaken in an attempt to mitigate the damaging effects of housing, industrial and commercial development (Byrne, 1990), and is specifically aimed at conserving existing ecological value. Most transplantations/translocations fail to meet this objective however, as it is virtually impossible to transplant the whole habitat, particularly existing hydrological regimes.

The distribution of cut vegetation or topsoil on a prepared site is one of two main methods of community transplantation/translocation. This technique, called 'blading', has achieved some success, and is a relatively quick and cheap option (Byrne, 1990). Grassland creation has been been shown to be possible both through the strewing of hay collected from species rich grasslands (Jones, Trueman and Millet, 1995), and through direct soil removal and transfer (Helliwell, 1989). Heathland plant communities have also been successfully established on abandoned farmland, following the application of harvested heather shoots and stripped heathland topsoil (Pywell, Webb and Putwain, 1995).

The movement of turves from a donor to receptor site is the second, and at present more widely used, method of community transplantation/translocation (Byrne, 1990). Such movements do not guarantee that community structure will remain unchanged once the turves are relayed however, as sensitive animal and plant species may be subjected to lethal levels of stress (Bruns, 1988; Byrne, 1990). Most attempts at turf transplantation have been made using grassland habitats, as this community is both relatively easy to handle, and quick to recover from damage (Byrne, 1990). Transplantation of turves has proved suitable in heathland restoration projects too (Putwain and Rae, 1988), and has been shown to be more effective in recreating a mature heathland plant community than most other techniques (Pywell *et al.*, 1995).

As Byrne (1990) points out, a number of important factors must be considered before any community transplantation is attempted. Perhaps of greatest importance is the question of whether the development necessitating community transplantation is able to be relocated or modified to minimise damage in the first place. If this is not possible, the likely success (and cost) of the scheme must be compared with options based on other techniques of community restoration. This may be difficult to ascertain, particularly if little published research is available to provide guidance for the correct procedures to adopt in moving a particular community type (as is the case for woodland). Practical limitations of time, capital outlay, manpower, availability of donor material and the suitability of a receptor site must also be assessed, as must potential problems associated with aftercare and site management. All these considerations serve to make community transplantation/translocation techniques suitable for only a relatively limited range of situations. It must also be remembered that where turf transplantations and topsoil applications are used, an existing habitat has to be destroyed to provide the donor material. In addition, it is unlikely that this material will be of sufficient quantity for use in large-scale community restoration projects. In light of this, turf transplant techniques may best be used in conjunction with non-destructive methods of blading, in order to create small-scale 'refuge islands' (Pywell *et al.*, 1995), from which plant and animal species can colonise the surrounding area. This approach may also be used in community enhancement/diversification schemes.

1. 5. 2. 3 Community enhancement/diversification

Community enhancement/diversification aims to increase the 'ecological potential' of biologically degraded/impoverished habitats (Buckley, 1989a), mainly through assisting natural processes of vegetation establishment. It can involve the use of a range of techniques, from habitat management that manipulates the structure and composition of existing communities, to the (re)introduction of selected species.

Perhaps one of the most widely used techniques of community enhancement is that of coppice management in woodlands. Where levels of natural disturbance in a woodland are low, and there has been a history of such management in the past, coppicing can be reintroduced as a means of encouraging the development of relatively diverse herbaceous plant communities (Barkham, 1992). Rotational management (cutting and/or grazing) can similarly be of value when reintroducing management to neglected herb-rich grasslands, where it has the potential to conserve a more diverse community of invertebrates than a system of continuous management (Morris, 1991).

Such traditional forms of land management are used extensively by conservation managers in the management of many habitat types. Methods of community enhancement based upon species introductions are, however, less widely employed. These techniques are generally suitable for areas of recently established semi-natural habitat, such as old agricultural land, derelict industrial sites, and woodland plantations. They may include methods of turf transplantation and topsoil application in the establishment of small refuge islands, as well as individual species introductions. The introduction of plant species, both as seed and small plantlets, has been successfully used in this way to diversify species-poor grassland (e.g. Wells, Cox and Frost, 1989), and the herbaceous flora of recent woodland plantations (e.g. Francis, 1995).

1. 5. 3 Assessing the ecological feasibility of community restoration

Whatever technique is used, it is important to acknowledge from the outset that all community restoration projects fail to fully replicate the communities of established semi-natural habitat types (Newbold, 1989). The complex and often undetermined nature of community relationships in ecosystems dictates that the complete reassembly of any community is most probably ..."an imperfect and partially unobtainable objective" (Buckley, 1989). Given the length of time over which many semi-natural communities have been extant, it would clearly be impossible to determine, let alone reconstruct, the range of factors necessary for the development of complex biological communities. It is therefore important to ensure that the inherently creative approach of such schemes is never used to justify the destruction of existing semi-natural communities and habitats, whose value cannot be replaced (Newbold, 1989). When community restoration is used, schemes must be both adequately planned and implemented, with long-term monitoring forming part of the post-restoration site management plan. Only in this way can chosen community types be restored as accurately as is practicable.

Ultimately, the success of community restoration schemes is dependent upon a number of important factors. The starting point for any terrestrial plant community is the underlying soil (Bradshaw, 1987b). Past land-use will to a great extent determine the sort of plant community that can be established on the soils of a particular area, though remedial treatments can in some cases be used to alter resource availability and widen this choice. If active plant species introductions are used to change propagule availability on a site, the choice of species, the size of the introduced populations, and the order in which they are added to the community, are all likely to influence community structure (Gilpin, 1987a; Luken, 1990). It is also important to consider how sustainable communities will be in the medium to long-term. All management activities alter the rate and direction of plant community succession (Luken, 1990). In the absence of management, scrub and woodland communities may develop on many sites. Seral communities, such as heathland and grassland, therefore require continued management to ensure their survival.

Animals also have a great influence in modifying successional processes and community structure, though relatively little research is available to explain exactly what these influences might be. Most attempts at community restoration involve the establishment of vegetation on a site, and rely on natural colonisation for the establishment of the animal component of a community (Harper, 1987). As stated by Harper (1987) however, ..."plant communities are (to some extent) what animal communities make them"..., and it is therefore unlikely that restored habitats can ever be 'complete' without the range of animal species representative of those they try to reproduce.

Community transplantation/translocation could be said to be the exception to this rule, as it may lead to the establishment of a characteristic invertebrate fauna on a restored site. In the absence of more extensive and suitable habitats for later colonisation however, such refuge islands could not necessarily guarantee the long-term survival of invertebrate populations. Only repeated invasions from neighbouring, undisturbed (ie. high quality) habitats might ensure that invertebrate species eventually encounter the range of conditions (associated with vegetation structure and composition) required for continued survival. Where restored communities are isolated from remaining semi-natural vegetation, it is more likely that the invertebrate fauna will be characteristic of the surrounding landscape (Webb, 1989), and therefore very different to that found in existing semi-natural habitats.

Similarly, the establishment of vertebrate species populations will most likely be as a result of invasion from areas surrounding restored communities (Gilpin, 1987b), but only if these areas are themselves able to act as species reservoirs. This landscape matrix may therefore be seen as the dominant force in determining successional processes and community structure for animal species in many habitat islands, including newly restored ones (Samways, 1994).

These limitations do not mean that community restoration should not be attempted. With suitable planning, implementation and monitoring, such schemes can provide net ecological benefits (Bradshaw, 1989), particularly in biologically impoverished areas. Where schemes are undertaken as part of a much broader approach to ecological planning at the landscape scale, the benefits could be potentially greater. Ideally, community restoration should always be undertaken as part of such a process, and efforts made to ensure that schemes do not become merely ..."isolated exercises in technical proficiency" (Dunnett, 1995). In this way community restoration can be used to effectively enhance biodiversity at local, regional, and national levels.

1. 6 Restoration of woodland communities

1. 6. 1 Background

Woodland is a term used to describe any area of land whose vegetation is dominated by trees and shrubs, as distinct from scrub which is composed of sub-climax woody vegetation (Rodwell, 1991). Throughout this thesis the term woodland is used, with further qualification, to refer to woods which are composed of broadleaved tree species only. Where a particular broadleaved woodland type is considered separately, it is specifically identified with the relevant prefix.

As Peterken (1986) states, ..."the creation of new woodland is more than mere tree planting". This approach has however tended to be the one used by a wide range of public and private bodies in the creation of areas of woodland for amenity and conservation. New woodlands, with greater ecological value, can only be created through an understanding of the processes responsible for determining the structure and composition of our existing semi-natural woodlands. This process, in turn, must involve an examination of the history and ecology of the range of woodland types found in Britain today.

1. 6. 2 A history of Britain's woodland

Woodland cover in Britain was originally extensive, amounting to some 85 to 90 per cent of the total land area (Peterken, 1995). The diversity of woodland types in this original 'primaeval' woodland (Peterken, 1981), or 'wildwood' (Rackham, 1986), was related to a combination of natural variations in climatic and edaphic conditions and periodic disturbance events (Peterken and Mountford, 1995). All woodlands have been subsequently influenced by human exploitation and management (Fuller, 1995). This process started with the first clearance of Britain's wildwood by Neolithic farmers, at the beginning of the Sub-Boreal climatic period, 6000-7000 years ago (Rackham, 1992).

Although woodland clearance continued into the Middle Ages, many woods retained their value as sources of both a range of useful products (including timber, brushwood, and underwood), and services (such as areas for grazing and hunting game species). Some woods eventually acquired 'scarcity value' (Peterken, 1981). By the middle of the fourteenth century, these economic and social functions ensured the survival of most woodlands up to the 1840's (Rackham, 1986). From the beginning of the nineteenth century however, the fate of woodland became increasingly linked to the ..."boom and bust cycles of the modern economy" (Rackham, 1986). Agricultural and forestry practices were responsible for the damage or destruction of many areas of woodland, so much so, that total woodland cover stood at only 5 per cent of the land area of Britain by the beginning of the twentieth century (Department of the Environment, 1994).

Since that time there has been a dramatic increase in total woodland area, rising from 1.4 million hectares in 1947, to over 2.3 million hectares (10 per cent of the land area of Britain) in 1991 (Saunders, 1993). Most of this increase represents large-scale planting of non-native coniferous species (Watkins, 1993), both on previously open ground and in existing broadleaved woodland sites; a process known as 'conversion'. More recently, the establishment of the Broadleaves Policy (in October 1985) has placed greater emphasis on the need to protect existing broadleaved woodlands, and to increase their total area nationally through widespread planting schemes (Peterken, 1993a).

Both the historical origin of a particular area of woodland, and a measure of the degree of human influence on (or 'naturalness' of) stand composition and structure, can be used to classify all areas of woodland in Britain today as either ancient or recent, plantation or semi-natural (Peterken, 1981). A distinction can also be made between primary woodlands, fragments of original wildwood on soils undisturbed by cultivation, and secondary woodlands, occupying sites which have at some time been completely cleared of tree cover and subjected to other forms of land-use (Peterken, 1981). Ecological distinctions can be made between the various types of British semi-natural woodlands, through descriptions in National Vegetation Classification (NVC) floristic tables (Rodwell, 1991).

1. 6. 3 Woodland nature conservation in Britain

Peterken (1981) states that woodland nature conservation should aim to meet meet four main objectives :

1. The maintenance of self-perpetuating populations of native plant and animal species throughout their range.
2. The preservation of examples of all semi-natural woodland community types.
3. The maintenance of other features of interest, such as geological formations and physical artefacts, which are associated with woodlands.
4. The maintenance of an element of wilderness in the landscape.

Efforts to achieve these objectives have taken a variety of forms (see Peterken, 1977; 1981). Over the last twenty years however, particular emphasis has been placed on measures aimed at the protection and conservation of remaining semi-natural woodlands (Peterken, 1977).

Of all semi-natural woodlands in Britain today, those classed as ancient (ie. present from the year 1600 onwards [Peterken, 1981]) are of greatest value to nature conservation. Many have been almost continuously wooded for at least the last 400 years, and some throughout the long history of woodland clearance. Almost all have been managed in one form or another (Rackham, 1986), and this management (particularly as traditional coppice in native broadleaved woodlands) has greatly altered woodland structure and composition over time (Peterken, 1992). It has also facilitated the continuation of a high degree of 'past-naturalness' in many woodlands (Peterken, 1981). As a result of this continuity they tend to exhibit one or a combination of features that many recent and plantation woodlands lack, including the presence of rarer plant and animal species, more diverse biotic communities, and relatively undisturbed soil profiles (Peterken, 1981, 1983; Saunders, 1993).

Such ecologically important woodlands, whose area cannot by definition be increased (Peterken, 1993a), form only a small part of total woodland cover in Britain today. Data from the Ancient Woodland Inventory Project (Spencer and Kirby, 1992) show that ancient woodland covers 535,000 hectares, or around 1.5 per cent (Peterken, 1977), of the total land area of Britain. This represents 23 per cent of the total woodland area of England, 13 per cent that of Wales, and 7 per cent of Scotland (Fuller, 1995). The distribution of ancient woodland tends to be uneven in all three countries. In England for example, it is concentrated in the south-east, New Forest, Wye Valley, Chilterns, and southern Lake District (Saunders, 1993). Ancient woodlands also tend to be both small and isolated (Peterken, 1995), making the plant and animal communities within them susceptible to the effects of the ecological constraints and potential threats outlined in section 1. 3.

In the long-term, woodland nature conservation objectives can only realistically be met if attempts are made to increase the extent of native woodland in Britain. Such action could provide a number of ecological benefits, including an increase in woodland species population sizes and resilience, and the creation of more edge habitats and habitat mosaics (Peterken, 1995); particularly if the creation of new areas of woodland is undertaken with reference to, and the correct application of, island biogeographical theories (Spellerberg, 1995).

1. 6. 4 The creation of new native woodlands

A national tree planting target of around 33,000 hectares per year (Watkins, 1993) means that the total area of woodland in Britain will continue to expand in the future. The establishment of both Community Forests and the new National Forest, as well as the increased likelihood of planting on surplus agricultural land, means that much of this new woodland will be located in the lowlands of Britain. Such afforestation will provide a range of environmental and social benefits to large numbers of people (Watkins, 1993), and where adequately planned, net ecological gains through the creation of more complex plant communities on areas of derelict land and farmland. As (in comparison to upland forestry practice) it will be possible to employ a wide range of silvicultural systems and species in the establishment and management of lowland woodlands (Watkins, 1993), there is potential for creating woodlands with a range of structural characteristics and associated values for nature conservation.

One of the most important considerations in the establishment of new woodlands will be the methods by which this is to be achieved. Peterken (1993a) cites four possible sources for the development of new woodlands:

1. Natural succession on unwooded ground.
2. Planting appropriate native trees on bare ground.
3. Removing plantations of non-native tree species from ancient woods and allowing native trees to replace them.
4. Planting or allowing natural succession of native tree species within existing upland plantation forests.

It is important to ensure that these methods are used to create new native woodlands with high levels of naturalness. Peterken (1995) has provided a 'sliding scale' of definitions for what constitutes both native woodland and naturalness, as characterised by stand and species origins (shown in Table 1. 1). Based on these definitions the woodlands with greatest conservation value will be those with naturally regenerated stands comprised of locally native species and genotypes, those with the least being plantations of species that have been introduced to Britain (Peterken, 1995). The conservation value of new native woodlands has the potential for lying somewhere between these two extremes, and will depend upon whether locally or nationally native species, and natural regeneration or planting methods are used.

Table 1. 1 Native woodland: a sliding scale of definitions based on origins of species and stands. Taken from Peterken (1995).

	Stand naturally regenerated	**Stand planted**
Species and genotypes locally native	The strict definition, where regeneration is natural, and the site has not been disturbed.	A practical version of the strict definition, admitting planting as a means of restocking.
Species locally native	A practical and reasonably strict definition, acceptable for nature conservation.	A practical and reasonably strict definition, acceptable for nature conservation.
Species nationally but not locally native	A broad definition, acceptable for conservation in new native woodland.	The broadest meaningful definition, acceptable for amenity management.
Species introduced to Britain	The broadest meaningful definition, acceptable for amenity management.	–

New native woodlands will of course be established in order to meet a number of objectives, not just those of nature conservation, and this may determine the most appropriate method for woodland creation (Harmer and Kerr, 1995). Whilst natural colonisation is theoretically preferable to planting, as it results in structurally diverse woodland comprised of local genotypes (Rodwell and Patterson, 1994), planting is in many cases a more predictable method, which can provide an early visual impact (Harmer and Kerr, 1995), and may therefore be more appropriate for the creation of woodlands with amenity value. Wherever planting is used, it is important to determine both planting patterns and the choice of species to be used (Harmer and Kerr, 1995). Design prescriptions for new native woodlands (Rodwell and Patterson, 1994) have greatly aided this process, particularly as NVC woodland types have been used to provide guidance for the appropriate selection of species mixtures (Rodwell and Patterson, 1995). With the application of suitable design prescriptions and appropriate management, it may be possible to create new native woodlands with relatively high levels of 'future-naturalness' (Peterken, 1996).

The establishment of diverse woodland communities will however depend upon more than the establishment of the canopy and shrub layers alone. If the structure and composition of some new native woodlands is to mimic that of existing semi-natural woodlands, herbaceous species, invertebrates and fungi characteristic of these woodlands must also be represented (Spencer, 1995). With such an approach it may be possible to restore functioning woodland ecosystems, not just create areas of 'woodland', ie. areas of land covered by trees. For much of lowland Britain, this process will involve the establishment of broadleaved woodland habitats.

1. 6. 5 The ecology of woodland community restoration

In the absence of appropriate intervention, the 'maturation time' of new woodlands is likely to take many centuries (Spencer, 1995). A large body of research has established that this slow rate of development is associated with the effects of existing habitat fragmentation and isolation.

1. 6. 5. 1 Island biogeography theory and woodland fragmentation

Oceanic island biogeography theories are based on the general observation that an increase in island area is usually associated with an increase in species richness. In 'islands' of fragmented semi-natural habitat however, research has shown that such a relationship is far from straightforward, and may be subject to variations and exceptions, particularly when different taxonomic groups are considered (Spellerberg, 1995). With birds for example, altitude, diversity of tree species, productivity of the surrounding landscape, and species-specific social factors are all thought to influence the diversity and relative abundance of species found breeding in woodland (Fuller and Warren, 1995). Several authors (e.g. Rackham, 1980; Peterken and Game, 1984) have confirmed the species-area relationship for vascular plant species in woodland, though stand species are stated by Rackham (1992) to be less affected by woodland size than the number of herbaceous species. As with the bird fauna however, this relationship is in many cases more complicated, particularly as herbaceous species diversity in many woodlands is more likely to be related to the presence of other habitat types, such as base-rich streams, wet flushes and glades (Peterken, 1981; Peterken and Game, 1984).

For practical purposes, herbaceous community richness in recent woodlands is best described as a function of the proximity of a more diverse (usually ancient) woodland, and the various modes of species dispersal from these woods to a more recent site (Dzwonko, 1993). Recent woods are known to contain fewer species than ancient woods, as many typical woodland plants are thought to be either unable to colonise new sites, or excluded by more competitive species (Peterken and Game, 1984; Peterken, 1993b). Where recent woods are located next to ancient woods they are generally richer in herbaceous species than those that are more isolated (Peterken and Game, 1984; Dzwonko, 1993). Herbaceous community characteristics in such woods are derived from variations in soil conditions, light regimes and levels of competition (Dzwonko and Gawronski, 1994), whilst in isolated woodlands the herbaceous community will generally be characterised by those species that were present at the time of woodland establishment (Woodruffe-Peacock, 1918).

1. 6. 5. 2 Implications for new woodland development

New woodlands that are established in areas with little remaining ancient woodland, or in relatively isolated locations, are unlikely to develop a full complement of the animal and plant species associated with ancient woods. In many cases therefore, ecosystem development can best be promoted through the appropriate use of various ecological restoration techniques. Spencer (1995) divides such operations into two distinct classes, one involving the introduction/establishment of 'cornerstone' elements of woodland vegetation, soil organisms and fungi, the other the establishment of locally scarce species with higher conservation value. Of the two, the introduction of cornerstone species is least contentious, as it can be successfully used to create more diverse woodland communities, without compromising the 'meaning' associated with biological rarity (Spencer, 1995).

There is great variation in the levels of understanding as to the precise requirements of, and roles played by, particular groups of cornerstone species in woodland communities however, and this will influence the means by which such introductions are made. Where levels of knowledge are relatively low, as with soil organisms and fungi, success may best be achieved through inoculations from existing woodlands (Spencer, 1995). Where more is known about the ecological requirements of individual species, it may be possible to use more refined methods. All these operations will themselves ultimately depend upon an understanding of the ecological processes responsible for maintaining semi-natural woodland structure and composition over time, and the influence that various forms of woodland management may have on these processes. Using this knowledge it may be possible to either predict which species may be best suited to the range of environmental conditions present at a given site, or to determine the detailed management required to create these conditions.

1. 6. 5. 3 Ecology of the tree and shrub layer in unmanaged woodland

In unmanaged woodlands, the presence of an individual in the tree/shrub (stand) layer is often related both to differences in localised inter- and intraspecific performance (*sensu* Hutchings, 1986) (Crawley, 1986a), and to past opportunities for individual recruitment created by disturbance events (Jones, 1945; Silvertown and Doust, 1993). The degree to which species form mixtures or mosaics is partly derived from the scale at which these processes operate. Mosaic formation can also arise from local dominance by clonal tree species, such as *Populus tremula* L., *Prunus avium* L., and *Prunus spinosa* L., and as a consequence of the gregarious habit of certain 'crypto-clonal' species, e.g. *Tilia cordata* Miller and *Carpinus betulus* L. (Rackham, 1992). Many of the ecological interactions between stand species at different growth stages, and between seedlings of these species and the field layer are, however, largely undetermined. This serves to make any interpretation of stand layer characteristics in unmanaged woodland somewhat speculative, particularly as human influences have shaped most, if not all, of Britain's woodlands at some time in the past.

Despite this fact, it has been possible to use data collected from monitoring in a number of unmanaged woodlands to determine which factors are thought to most influence stand characteristics over time. One of these woodlands, Lady Park Wood in the Wye valley, has developed for the last fifty years in the absence of any form of human intervention (Peterken and Jones, 1987), and is one of Britain's most 'natural' woodland nature reserves. Long-term monitoring of old-growth stands has shown that the character of this woodland is greatly influenced by disturbance events (drought, chronic instability on steep slopes, disease and windthrow), and that each event affects different species and stands to varying degrees (Peterken and Jones, 1987; Peterken and Mountford, 1995). Thus, whilst the woodland tends towards *Fagus sylvatica* L. dominance during undisturbed periods, repeated interruptions allow the establishment of more opportunistic species such as *Fraxinus excelsior* L., *Betula spp.*, and *Populus tremula.* The proportion of these species present in any area at a particular time is dependent upon the nature and sequence of disturbances, and/or the chance of canopy openings coinciding with heavy mast years (Peterken and Mountford, 1995).

Whilst Lady Park Wood is, as Rackham (1992) states, perhaps atypical (much of the stand being dominated by *Fagus sylvatica,* a species particularly sensitive to 'catastrophe'), unpredictable disturbance events occur in all unmanaged woodlands at some time. As the ultimate outcome of succession in unmanaged woodland will always therefore be both unstable and unpredictable (Peterken and Jones, 1987), and stands are unlikely to form stable climax communities, the 'natural state' of any unmanaged woodland can at best be described as a particular 'set of probabilities' (Peterken and Mountford, 1995).

1. 6. 5. 4 Ecology of the field layer in unmanaged woodland

Many aspects of species and individual performance in the field layer are affected by forms of disturbance in the the overlying stand (Barkham, 1992; Evans and Barkham, 1992). Under a mature stand, where light levels are thought to be a major limiting factor for herbs (Packham and Cohn, 1990; Packham *et al.*, 1992), periodic tree fall and canopy gap formation will be an important influence. Tree fall is also associated with 'pit and mound' formation, which may in itself lead to localised diversification of the field layer (Evans and Barkham, 1992). Such relationships are far from straightforward however, and are greatly dependent upon the spatial and temporal distribution of such gaps within a woodland (Evans and Barkham, 1992). The degree to which stand species form mosaics may also influence field layer characteristics. More heterogeneous stands are generally characterised by diversity of crown density, foliage periodicity, leaf size, branch angle, and bark relief, all of which create a patchwork of microclimatic regimes that provide opportunities for a number of different species (Barkman, 1992). Differing rates of litter decomposition between species may produce heterogeneity in soil pH and nutrient status that can similarly affect field layer characteristics (Muys and Lust, 1993).

One of the main ecological distinctions between field layer species is related to the way in which they cope with the changing levels of photosynthetically active radiation (PAR) associated with seasonal leaf expansion and fall. Some species, such as *Anemone nemorosa* L. and *Hyacinthoides non-scripta* L., exhibit 'phenological escape' (Crawley, 1986a) through rapid growth and flowering in spring, before expansion of the overlying tree canopy. These species are thus able to survive in the field layer as 'shade-evaders' (Rackham, 1980). Leaf expansion at an earlier date than that for adult individuals similarly allows the saplings of a number of tree species, including *Acer pseudoplatanus* L. and *Fraxinus excelsior* , to exploit the higher light levels found before canopy expansion (Grime, Hodgson and Hunt, 1988). Other adaptations, including reduced respiration rates, increased photosynthetic rates per unit leaf area, and increased leaf area per unit weight, enable 'shade-tolerant' species (e.g. *Circaea lutetiana* L.) to maximise photosynthetic gain at much lower light levels (Crawley, 1986a). Species which are both spring- and summer-leafing (such as *Primula vulgaris* Hudson and *Primula elatior* L.) are thought to employ both shade mechanisms throughout the growing season (Rackham, 1980).

Reaction to shade is however only one of a number of important characteristics used to categorise woodland field layer species. Distinctions can also be based on differences in the means of reproduction, degree of seed dormancy, relative growth rate, competitive ability, the influence of soil factors, and susceptibility to grazing (Packham and Cohn, 1990). Collectively, these characteristics can be used to define the 'ecological range' of particular species, and, when used in combination with the practical application of C-R-S theory (Grime, 1979), to identify the 'ecological strategies' (*sensu* Grime *et al.*, 1988) of these species in relation to varying levels of stress, competition and disturbance (Packham and Cohn, 1990; Packham *et al.*, 1995). Species strategies can therefore be used as a means of interpreting spatial variation in field layer community measures such as diversity and evenness, though such interpretations must be made with caution, as the ecological range of some species (e.g. *Convallaria majalis* L.) may vary from one part of their range to another (Peterken, 1981).

Using this approach it is possible to identify a number of species which are likely to dominate field layer communities in unmanaged woodlands. These include competitive species such as *Holcus mollis* L., and a number of stress-tolerant competitors such as *Hyacinthoides non-scripta, Mercurialis perennis* L., and *Lamiastrum galeobdolon* Ehrend.

& Polatschek (Rackham, 1980; Packham and Cohn, 1990). These and many similar species persist due to the regular production of large numbers of seeds and/or rhizomatous clonal growth (Barkham, 1992). In contrast, truly ruderal species such as *Stellaria media* Villars tend to be confined to disturbed areas of woodland, such as tree fall gaps and woodland margins (Packham and Cohn, 1990).

As with the stand layer therefore, herbaceous community properties (such as diversity and evenness) are ultimately derived from both the sum of the interactions between individual plants and their surrounding environment (Crawley, 1986a), and the influence of past disturbance events (Barkham, 1992). Where plant communities exhibit large-scale uniformity in such properties, it is usually related to uniformity of substrate characteristics and/or disturbance events (Crawley, 1986a). These factors create field layer assemblage mosaics of varying sizes, in which one or a few species may dominate (Rackham, 1980). These species tend to be those with the most 'suitable' strategies. Whilst the abundance and/or biomass of certain species will change in response to varying levels of shade, species composition is thought to be a more permanent feature of many field layer communities (Rackham, 1980).

1. 6. 5. 5 The effects of management on woodland vegetation

Traditional woodland management may may involve one of three main systems - coppice, high forest and wood-pasture. All these forms of management produce frequencies and patterns of gap creation (or disturbance) that differ from those found in unmanaged stands. As a result, managed stands tend to be characterised by (Peterken, 1991):

1. Steady state gap creation rates (ranging from 10% per year in coppice woodland, to 0.8% per year in some high forest rotations).
2. Gaps that are created with a greater degree of predictability.
3. Gaps that are formed in related to stand age (older or larger trees being generally scarce within stands).
4. Larger and more regular gaps, containing low levels of dead wood.

In combination with tree planting or the natural regeneration that follows felling or coppicing, these factors serve to create high levels of even-agedness in managed woodland (Peterken, 1991). Many sites also contain stands in which the abundance of particular species has been greatly modified (Kirby and Patterson, 1992). Regular coppicing may, for example, discourage the growth of *Fagus sylvatica* , while at the same time favour species such as *Corylus avellana* L. and *Populus tremula* (Rackham, 1980). Whilst very few woodlands continue to be managed as wood-pasture, those surviving sites (e.g. Moccas Park in Hereford and Worcester, and Windsor Forest/Park in Berkshire) also exhibit greatly altered stands, many of which are dominated by large old individuals of *Fagus sylvatica* and *Quercus robur* L. (Harding and Rose, 1986).

Field layer responses to stand management involve the 'reorganisation' of existing vegetation and/or the establishment of new individuals (Evans and Barkham, 1992). The degree to which communities may change over time depends upon the nature of the canopy gaps created, and the density and degree of species mixing in the stand layer, both before and after each canopy opening (Fuller and Warren, 1995). In terms of a more detailed understanding of field layer responses, most has been gained through monitoring changes in the herbaceous communities of coppice woodlands (e.g. Rackham, 1980; Peterken, 1981; Barkham, 1992; Fuller and Warren, 1995). Following coppicing diversity tends to increase, as species like *Digitalis purpurea* L. and *Juncus conglomeratus* L. germinate and grow from buried seed. Perennials such as *Primula vulgaris* , *Hyacinthoides non-scripta* and *Anemone nemorosa* reach a peak of flowering and growth in the second spring following coppicing. As the 'shade phase' of the cycle returns once more, those species adapted to either tolerate or evade low light levels become increasingly dominant once more.

The response of field layer species in woodlands managed under other systems may, of course, show few similarities to the responses outlined above. A number of important differences are likely to arise from the variations in soil environment and faunal activity which are partly dependent on stand characteristics, and as a result of the effects of various grazing animals (Barkham, 1992). Continued coppicing may also create communities which contain a greater abundance of 'tough' generalist species than are found in woodlands under other management systems (Hambler and Speight, 1995), as many woodland 'specialists' may require relatively stable conditions to ensure survival (Packham and Cohn, 1990).

1. 6. 5. 6 Woodland community restoration in practice

All attempts at restoration of woodland communities will involve some degree of intelligent guess-work. This is only to be expected when woodland ecosystem functioning depends upon a large number of interacting factors, many of which are either unknown or unique to a particular woodland site. As the maturation time of community restoration projects may run into many decades or even centuries, it is likely that many woodlands will develop in ways that are far removed from the plans of their originators (Spencer, 1995). In light of these facts, the first rule of community restoration must be to ensure that long-term objectives for new native woodlands are precisely determined from the start of any project. Where nature conservation is the main objective, woodlands must be established in such a way so as to make them 'fashion resistant' and able to support representative species-rich animal and plant communities well into the future (Spencer, 1995). Site assessment and evaluation, vegetation establishment and management options, and current and future research needs, should also be an integral part of all community restoration projects (Ferris-Kaan, 1995).

Considering the techniques that may be used in woodland community restoration, transplantations of stand and field layers have seldom been attempted, though are thought to be technically and biologically possible (Down and Morton, 1989). In contrast, community creation and enhancement/diversification has been described in a number of published studies, the objective being to diversify woodland field layer communities in new urban woodlands on a range of substrate types (e.g. Francis, Morton and Boorman, 1992; Francis, 1993; Packham *et al.*, 1995). The guiding principle behind these studies has been to introduce species whose strategies best suit the range of environmental variables found at a site. Of all the variables that determine the range of conditions required for individual species establishment, level of shade has proved to be one of the most influential (Buckley and Knight, 1989). In the medium to long-term therefore, manipulation of the stand may provide a highly effective and practicable method by which such introductions can be achieved, particularly in woodlands on relatively fertile soils where, in the absence of adequate shade, competitive species tend to dominate (Packham *et al.*, 1995).

Where woodland is being created *de novo*, levels of shade may not suit woodland herbaceous species introductions until after canopy closure, particularly as before this time a number of competitive species may dominate the field layer. In such situations mulching, followed by immediate sowing of a selected and non-competitive herbaceous seed mix, may serve to minimise the establishment and spread of more competitive species (Anderson, 1995). A number of appropriate woodland species, which would be expected to increase in abundance as canopy closure occurs, could be included in the seed mixture.

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PART TWO: Experimental work

2. 0 BACKGROUND TO RESEARCH: ENHANCING WOODLAND PLANT COMMUNITIES ON COLLIERY SPOIL

2. 1 Introduction

When suitably planned and implemented, community restoration schemes can provide net ecological benefits, particularly in areas where existing semi-natural habitats are scarce. Such schemes are also likely to form an important element in meeting the future needs of nature conservation in Britain. At present a growing interest in ecological restoration has coincided with a number of policies aimed at the establishment of new native woodlands throughout Britain. Most of these policies will produce woodlands with little ecological resemblance to older semi-natural woodland communities.

This part of the thesis presents results from a number of ecological investigations into the potential for enhancing the diversity of plant communities associated with recently established woodland on derelict industrial land, both planted and naturally regenerated. In this case enhancement is taken to mean ..."to raise in degree...or to increase in value"... (Bradshaw, in press). Derelict land forms the main land-type available for the establishment of woodland in many parts of Britain, including Merseyside where the research was undertaken. Though the experimental work was confined to colliery waste sites, it was designed to provide information which could be applied to the enhancement/diversification of recent woodland plant communities growing on a range of substrates. Both herbaceous and woody species characteristic of local semi-natural woodlands were introduced to sites in order to determine the successional stage most likely to provide the range of environmental variables that maximise species establishment and survival.

2. 2 Establishing woodland on derelict industrial land

Derelict land is described as, ..."land so damaged by industrial or other development that it is incapable of beneficial use without treatment" (Department of the Environment, 1991). In 1993, the most recent year for which published data is available, approximately 0.3 per cent of the land area (39,600 ha) of England was classed as derelict (Department of the Environment, 1995). Though a significant proportion of derelict land is thought to suffer from some form of contamination, most can be reclaimed for a variety of soft and hard end uses (Department of the Environment, 1992). Derelict Land Grant (DLG) has traditionally been a major source of funding for reclamation to hard end uses. Since 1991, DLG has also been available for supporting projects aimed at general environmental improvements (Department of the Environment, 1991).

Even though derelict sites may contain naturally established plant and animal communities of great ecological interest and value (e.g. Gemmell, 1982; Falk, 1995), many are often perceived as being hazardous and/or unattractive. Landscapes dominated by such sites tend to limit the economic potential of a particular area, through reducing the desire for new housing and commercial development. Whilst landscape improvements will eventually be achieved through the natural establishment of some form of woodland (on all but the most extreme sites), this is likely to take many decades. Where woodland is actively planted however, more immediate landscape improvements can be achieved. Planting can also provide additional benefits, such as opportunities for timber production and recreation (Moffat and McNeill, 1994). As restoration to woodland is relatively inexpensive, with low establishment and maintenance costs, it has become an increasingly attractive option for ecological restoration schemes on a range of derelict land types (Moffat and McNeill, 1994). Whether established through natural colonisation or as a result of planting however, such woodland tends to be characterised by plant communities with little resemblance to those in older semi-natural woodlands, particularly as community development is often restricted by a number of stresses associated with the physical and chemical substrate characteristics of many derelict sites.

2. 3 Colliery spoil as a medium for woodland establishment

Of the total of 39,600 hectares of derelict land identified in 1993, the largest proportion (8,653 ha or c. 22%) was located in north west England. Approximately 675 hectares (c. 8%) of this total in the region was classified as colliery spoil, most of which was thought to justify some form of reclamation (Department of the Environment, 1995). In many parts of the north west, colliery spoil can dominate the landscape, and is regarded as an eyesore which local communities may have to live with for many years. However, such sites should also be viewed as a valuable resource, in that they often represent a significant proportion of land that is locally in non-agricultural, urban or industrial use. They therefore provide opportunities for enhancing the nature conservation value of the local landscape, particularly where existing semi-natural habitats are scarce.

Although suitable for the establishment of many native and naturalised plant species, colliery spoil is characterised by a number of physical and chemical factors that generally limit plant survival and growth. Whilst this is not necessarily detrimental to the total ecological value of a site (many invertebrates for example actually require open bare ground with sparse vegetation cover), some sites may either lack substantial vegetation cover, or be dominated by a few stress-tolerant species, some of which may subsequently inhibit the development of more diverse semi-natural habitats (Crawley, 1986a). For such sites with little existing ecological value, woodland community restoration is likely to provide net ecological gains. In these cases, the success of community restoration can be enhanced, either through manipulation of the physical and chemical characteristics of colliery spoil, or by using these characteristics as a 'framework' for guiding choices on plant species introductions. As this latter approach is the one examined in this thesis, these characteristics will now be considered in more detail.

2. 4 The physical and chemical characteristics of colliery spoil

Plant establishment and growth on colliery spoil is dependent upon species' tolerance to a number of often extreme physical and chemical factors associated with this substrate.

2. 4. 1 Acidity

Iron pyrites (FeS_2) is a constituent of most colliery waste heaps in the Lancashire coalfield (Kent, 1982). On exposure to air and water, iron pyrites produces a number of chemical compounds, one of which is sulphuric acid:

$$1. \quad 4FeS_2 + 15O_2 + 14H_2O \Leftrightarrow 4Fe(OH)_3 + 8H_2SO_4$$

This equation greatly simplifies a number of reactions which may also involve ferric ions as oxidising agents (Pulford, 1991). The first stage of iron pyrites oxidation usually involves the slow release of ferrous ions and elemental sulphur:

$$2. \quad 2FeS_2 + O_2 + 4H^+ \Leftrightarrow 2Fe^{2+} + 4S^0 + 2H_2O$$

Where oxygen is available and at pH values above 3.5, ferrous ions are oxidised to ferric ions, and iron oxide precipitated:

$$3. \quad 4Fe^{2+} + O_2 + 4H^+ \Leftrightarrow 4Fe^{3+} + 2H_2O$$

$$4. \quad Fe^{3+} + 3H_2O \Leftrightarrow Fe(OH)_3 + 3H^+$$

Elemental sulphur is also oxidised to sulphate ions:

$$5. \quad 2S^0 + 3O_2 + 2H_2O \Leftrightarrow 2SO_4^{2-} + 4H^+$$

Iron pyrites oxidation in this way is thought to be relatively slow, the rate being governed by the diffusion of oxygen into the spoil surface. It is the dominant route for oxidation where spoil pH is above 3.5, as ferric ions are insoluble and are unable to act as an oxidising agents. Where pH levels are below 3.5, ferric ions are soluble, though the stability of ferrous ions makes reaction 3 (above) slow. Under these acidic conditions the bacterium *Thiobacillus ferrooxidans* L. is thought to catalyse the rapid oxidation of ferrous ions (Pulford, 1991). Where spoil contains relatively high levels of iron pyrites the production of acids may continue for many years; tips that are initially neutral or slightly alkaline tend to become increasingly acidic over time (Davis *et al.*, 1992). However, if carbonate minerals such as ankerite and siderite are present, some of the acidity produced by pyrite weathering may be neutralised (Kent, 1982).

In reality, many sites are likely to be characterised by variability in pH between one area and another, and at different depths below the spoil surface (Kent, 1982). 'Favourable sites' (Williams and Chadwick, 1977) may provide suitable conditions for the establishment of plant species in areas where levels of acidity are relatively low. Where acidity is high, vegetation establishment may be prevented altogether. Vegetation growth on such areas will depend upon some form of amelioration, using techniques that either inhibit pyrite oxidation or interact with oxidation products. One of the most common methods for treating colliery spoil involves the addition of ground limestone. Organic materials such as sewage sludge and manure are also able to inhibit pyrite oxidation, as they can chelate ferrous and ferric ions (Pulford, 1991). Such materials offer additional benefits in that they are readily available materials which, unlike limestone, can be obtained without the destruction by quarrying, of valuable elements of the landscape.

In addition to being directly toxic to plants, high acidity is responsible for lowering the cation exchange capacity of spoil material (Davis *et al.*, 1992). This may allow elements such as aluminium (Al), manganese (Mn), copper (Cu), zinc (Zn) and iron (Fe) to come into solution at concentrations that are themselves phytotoxic. As spoil pH is likely to be at its lowest level in summer and autumn months (Williams and Chadwick, 1977), these toxic elements may be at their greatest concentrations at a time when many plants are attempting to become established on a site. Some species (particularly trees and shrubs) are however known to be able to tolerate such toxic elements, through interactions with mycorrhizal fungi (Wilkinson and Dickinson, 1995). Zinc tolerance for both *Betula pendula* Roth and *Betula pubescens* Ehrh. for example, is known to be increased in the presence of mycorrhizas of *Amanita muscaria* L. and *Paxillus involutus* Fr. (Atkinson, 1992).

2. 4. 2 Availability of plant nutrients

As Kent (1982) states, ..."it is only by the establishment of an active nutrient cycle that...vegetation can remain viable in the long term". Those nutrients of most importance to plant growth in terrestrial ecosystems and their main methods of input are shown in Table 2. 1. On colliery spoil, successional processes show many similarities with other primary successions, resource acquisition being particularly important for the successful development of plant communities (Marrs and Bradshaw, 1993). Colliery spoil is initially characterised by low levels of available nutrients; in the absence of human intervention, active nutrient cycles may take many years to develop.

Of the three main plant nutrients, nitrogen (N), phosphorus (P) and potassium (K), nitrogen is usually of primary importance in facilitating species colonisation and community development (Marrs and Bradshaw, 1993). Colliery spoil may contain significant amounts of fossil nitrogen, although this is unavailable to plants. However on weathering, ammonium ions are released. Although some grasses are known to be able to utilise ammonium as a nitrogen source (Richardson, in Williams, 1975), nitrogen in this form is unavailable to most plants without the subsequent conversion of ammonium to nitrate. This process is dependent upon the activities of nitrifying bacteria such as *Nitrosomonas* and *Nitrobacter.* Where pH is low, these soil micro-organisms may not be able to survive, and available nitrogen will continue to be in short supply (Kent, 1982). If nitrogen-fixing vascular species are unable to colonise such a site, atmospheric sources will be the main method of nitrogen input into the ecosystem. Rates may vary from <10 kg N ha^{-1} $year^{-1}$ in unpolluted areas, to >40 kg N ha^{-1} $year^{-1}$ in industrial/urban areas (Marrs and Bradshaw, 1993).

Phosphorus can also be a major limiting factor for plant community development on colliery spoil as it is generally available at only very low levels (Kent, 1982). As in most ecosystems, phosphorus is most likely to be present as the orthophosphate group, $PO4^{3-}$, in insoluble compounds of calcium, aluminium and iron phosphate (O'Neill, 1985). Release of phosphorus through natural chemical weathering in terrestrial ecosystems has been

Table 2. 1 Main methods of nutrient input into terrestrial ecosystems (taken from Rosen, 1990).

	N	K	Ca	Mg	Na	P	S	Cl
Wet deposition	+	+	+	+	+	-	+	+
Gaseous input	+	-	-	-	-	-	+	-
Particle input	+	?	?	+	+	?	+	+
N_2 -fixation	+	-	-	-	-	-	-	-
Mineral weathering	-	+	+	+	+	+	+	-

Where + = main method of input

? = probable method of input

estimated at 0.01 - 1 kg ha^{-1} $year^{-1}$, the rate depending upon a number of factors, including rock type, surface area of mineral particles per hectare, temperature, and the nature of the dissolved chemicals in soil water (Newman, 1995). However, as with the phosphorus in chemical fertilisers, phosphate ions released in spoil as a result of weathering are likely to be quickly immobilised by aluminium and iron oxides, particularly where pH is low (Fitter, 1974). Phosphorus inputs from the atmosphere are thought to range from 0.07 - 1.7 kg ha^{-1} $year^{-1}$ and may be significant for plant growth in certain cases (Newman, 1995). In addition, mycorrhizal associations are thought to enhance plant uptake of a number of inorganic nutrients, including phosphorus (Brundrett, 1991).

Colliery spoil contains significant amounts of shale which are partly composed of aluminosilicate minerals. As potassium is readily incorporated in the lattices of these clay minerals (particularly illite) this element is seldom deficient in colliery spoil (Kent, 1982).

2. 4. 3 Physical problems

Many of the physical properties of colliery spoil have been reviewed by Kent (1982). The proportion and distribution of fines, produced as a result of physical and chemical weathering of shales, is of particular importance, as it may greatly influence spoil water relations, rooting zone temperatures, cation exchange capacities and plant germination and establishment. Increased proportions of fines may also reduce the infiltration of rainfall, particularly on slopes, and create greater rates of erosion over time.

The generally dark colouration of colliery spoil can lead to high surface temperatures in the summer, especially in absence of adequate vegetation cover. These temperatures may be so great (>45°C) that heat injury and die-back severely limits the colonising ability of many plants, though deeply rooted species may be less affected. Whilst root extension is greatly dependent upon adequate nutrient supplies, spoil compaction is also influential. Since 1966 all colliery waste heaps have been compacted in thin layers, in an effort to reduce aeration and the risk of spontaneous spoil combustion. This has limited the ability of plants to penetrate the surface 'crust' of spoil (particularly when dry), and can create spoil heaps with generally poor infiltration capacities. Scarification or 'ripping' is often used in initial reclamation to overcome this problem.

2. 5 Implications for natural vegetation establishment

Naturally-occuring plant communities on colliery spoil heaps have been described by a number of authors (e.g. Brierley, 1956; Hall, 1957; Ash, Gemmell and Bradshaw, 1994). In the absence of human intervention, these communities are partly a function of the species composition of the local flora (Hall, 1957). However, even where colliery tips are in close proximity to semi-natural vegetation, many sites only develop a relatively restricted flora. This is partly due to the physical and chemical characteristics of colliery spoil, as described previously, to which few species may be adequately adapted. For those species that are adapted, an additional problem is that of immigration onto a site across an often 'hostile' landscape matrix. Successful establishment may also depend upon the probability of plant propagules finding conditions suitable for survival at different spatial and temporal scales.

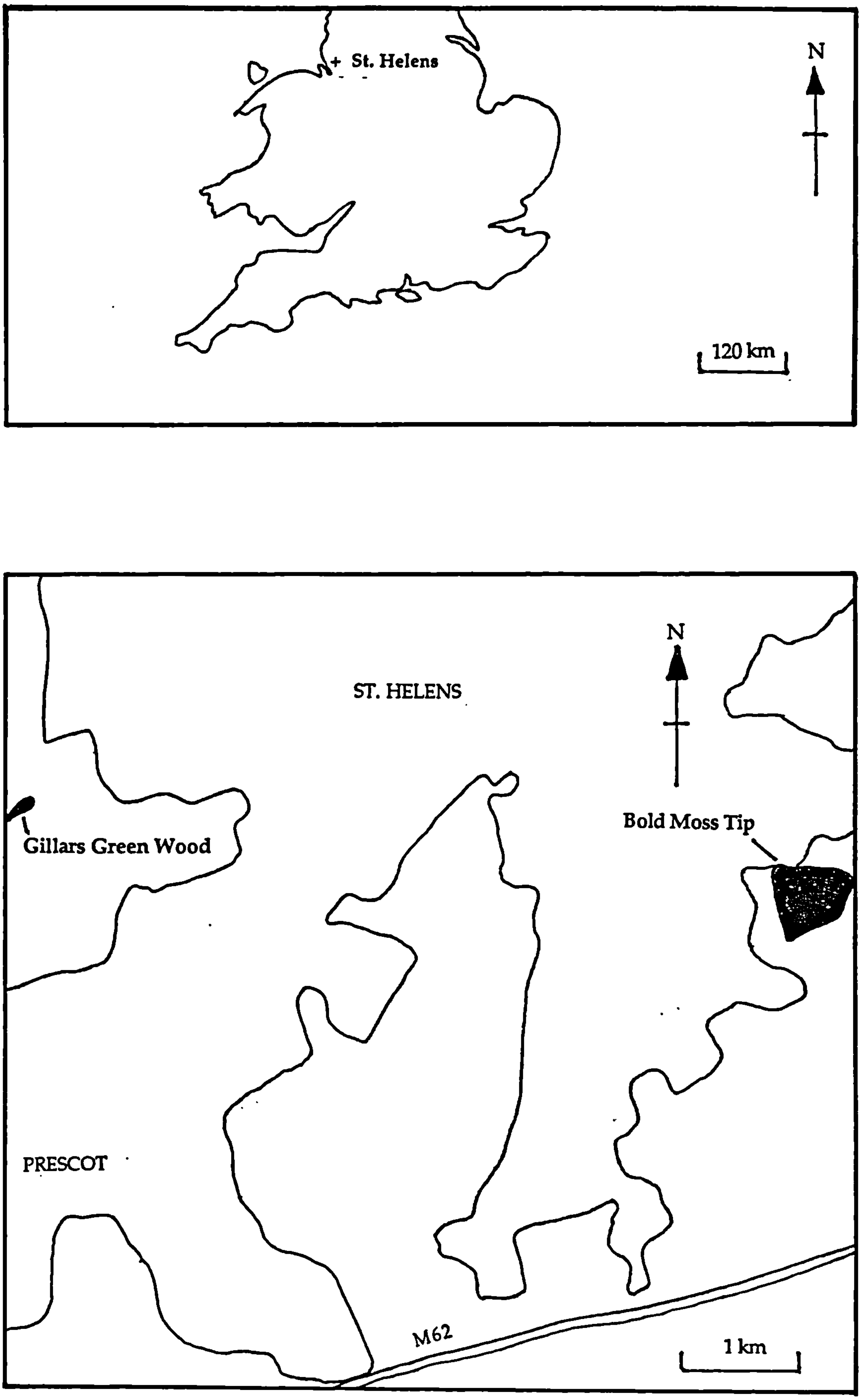
In general, plant species colonisation of colliery spoil sites does not follow a consistent pattern and there is no relationship between species richness and site age; variation in community composition is more closely related to variability in surrounding seed sources (Kent, 1982). On many sites trees such as *Betula* spp. and *Salix* spp. can successfully colonise bare spoil, although the pioneer flora may also comprise a high proportion of hemicryptophytes and therophytes, rosette species being particularly well represented at this pioneer stage (Down, 1973).

Though it is likely that plant community characteristics will change over time, introduction experiments have demonstrated that 'regeneration niches' (*sensu* Grubb, 1977) are available for appropriate species on a wide range of derelict industrial sites, including colliery spoil heaps (Ash *et al.*, 1994). Where woodland (ie. vegetation dominated by large woody species) develops on colliery spoil, either as a result of natural colonisation or as an end-use following reclamation, it is likely that the habitat is suitable for colonisation by a number of characteristic woodland plant species. However, the extent to which this may occur can only be determined through detailed experimental work.

2. 6 The study sites

Two colliery waste sites in Merseyside were chosen for experimental work. Both sites, Bold Moss Tip and Gillars Green Wood, lie within the Metropolitan Borough of St. Helens and the Lancashire Coalfield (Figure 2. 1), and form only a small part of the estimated 700 hectares of derelict land within the borough.

Figure 2. 1 Location of Bold Moss Tip and Gillars Green Wood, St. Helens, Merseyside.



2. 6. 1 Bold Moss Tip

Bold Moss Tip (SJ 544940) was started in 1955 when colliery spoil from Bold colliery was tipped onto what was then one of the few remaining raised bog habitats in the county of Lancashire. From 1959 onwards pulverised fuel ash (PFA) from a nearby power station was also tipped on the site by the Central Electricity Generating Board (CEGB). Today the site covers an area of 45 hectares, and is owned and managed by the Groundwork Trust, St. Helens.

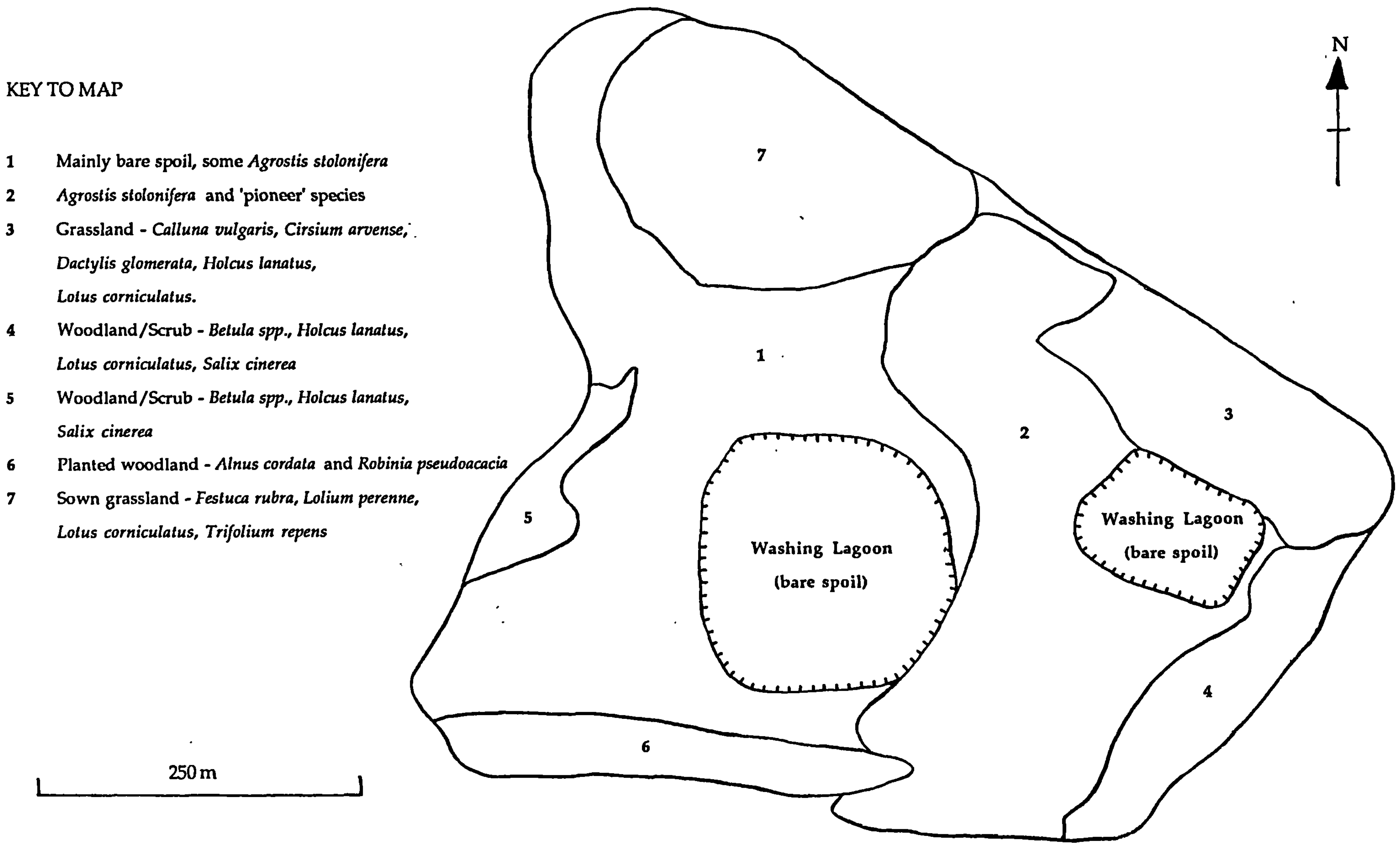
A great deal of information on the geology and ecology of the site has been determined through work undertaken by the Groundwork Trust (Groundwork Foundation, 1992). The site is morphologically diverse, two large washing lagoons being a major feature of the top of the tip. Investigations into tip stratigraphy have shown that it is comprised of mainly coarse particles of coal, siltstone, mudstone and shale, with significant quantities of PFA in places. Mean spoil pH is approximately 3.1, with a minimum of 2.3 for some areas during summer months. Where PFA is present at or near the spoil surface, pH values may be as high as 7.8, though these areas are relatively localised. Data for a range of chemical analyses of spoil material from Bold Moss Tip are given in Table 2. 2. Levels of trace elements are thought to be below phytotoxic concentrations over most of the site (Groundwork Foundation, 1992).

The ecological characteristics of Bold Moss Tip can be summarised by reference to the various plant communities that have become naturally established on the site. These have been broadly classified into community types (P. Gateley, unpublished report), as shown in Fig 2. 2. The two most common species on the tip are *Agrostis stolonifera* L. and *Salix cinerea* L.. Well developed scrub vegetation, mainly *Salix cinerea* with occasional *Betula pendula* and *Salix caprea* L., currently dominates the western and eastern slopes of the tip. Evidence from aerial photographs shows that most of this scrub has developed from the late 1970's onwards (Groundwork Trust St. Helens, unpublished report). Much of the top of the tip forms a plateau with little or no vegetation cover. An area of woodland on the southern slope is dominated by *Alnus cordata* Duby and *Robinia pseudoacacia* L., both of which were introduced to the site in an early attempt at land reclamation. Following treatment with crushed limestone, an area of grassland was sown on the northern side of the tip in the early 1990's.

Table 2. 2 Chemical characteristics of colliery spoil, Bold Moss Tip, St. Helens. Source: Groundwork Foundation (1992).

N. B. Values for metals equal total concentrations.

FACTOR MEASURED	MEAN	RANGE
Lime requirement (t/ha)	1.15	0-6.00
Calcium carbonate equivalent (t/ha)	70	0-300
Pyrite (%spoil content)	0.88	0.19-2.50
Arsenic (mg/kg)	38	11-143
Cadmium (mg/kg)	<1.00	<1.00-4.00
Copper (mg/kg)	64.70	33-232
Nickel (mg/kg)	28.80	12-49
Zinc (mg/kg)	81	23-654
Sodium (mg/kg)	470	300-1,400



KEY TO MAP

1 Mainly bare spoil, some *Agrostis stolonifera*
2 *Agrostis stolonifera* and 'pioneer' species
3 Grassland - *Calluna vulgaris, Cirsium arvense, Dactylis glomerata, Holcus lanatus, Lotus corniculatus.*
4 Woodland/Scrub - *Betula spp., Holcus lanatus, Lotus corniculatus, Salix cinerea*
5 Woodland/Scrub - *Betula spp., Holcus lanatus, Salix cinerea*
6 Planted woodland - *Alnus cordata* and *Robinia pseudoacacia*
7 Sown grassland - *Festuca rubra, Lolium perenne, Lotus corniculatus, Trifolium repens*

Figure 2. 2 Bold Moss Tip - Semi-natural community types. Source: P. Gateley (Unpublished Report).

2. 6. 2 Gillars Green Wood

Gillars Green Wood (SJ 476948) is located on the western side of St. Helens and covers an area of approximately 2.5 hectares. This small site was formed by a colliery that ceased coal production towards the beginning of this century. The spoil was tipped loosely in localised piles, forming a tip with steep sides and variable relief. At the western end, patches of spoil remain bare and poorly colonised by vegetation (Fig. 2. 3). The rest of the site is covered by well established woodland which shows some affinities with both *Quercus robur - Pteridium aquilinum - Rubus fruticosus* (W10) and *Quercus* spp. - *Betula* spp. - *Deschampsia flexuosa* (W16) woodland. Much of the field layer is dominated by *Pteridium aquilinum* Kuhn in more lightly shaded areas, and *Rubus fruticosus* L. agg. in areas with denser canopy cover. As *Pteridium aquilinum* is thought to spread mainly through vegetative expansion (Rodwell, 1991), it is likely that the site had existing semi-natural vegetation cover in the period preceding spoil dumping. The site is privately owned and unmanaged at present.

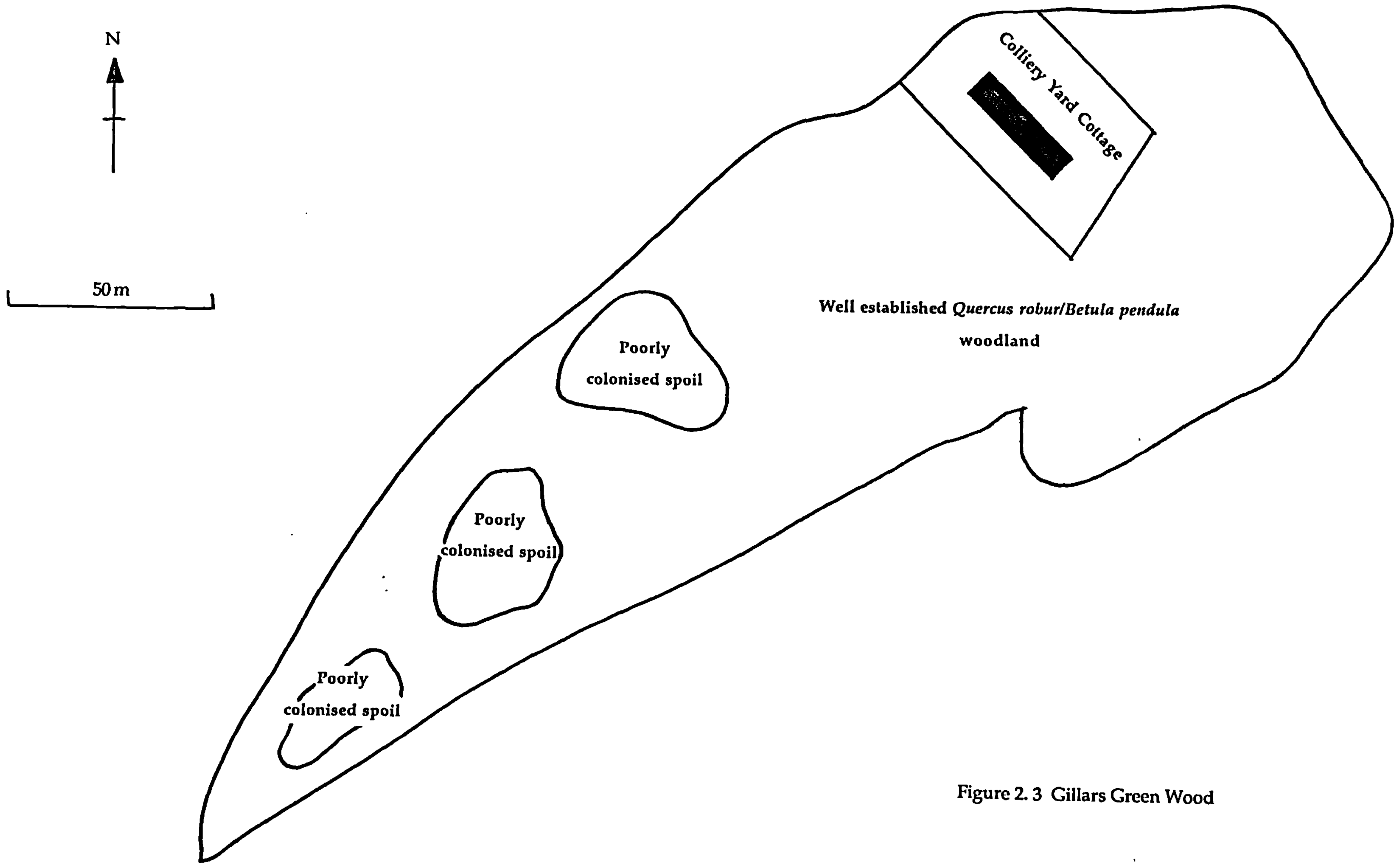


Figure 2. 3 Gillars Green Wood

3. 0 THE EFFECT OF MINERAL NUTRIENT ADDITION ON HERBACEOUS WOODLAND PLANT SPECIES INTRODUCED AS SEED INTO EXISTING VEGETATION ON COLLIERY SPOIL

3. 1 Introduction

Mineral nutrient availability is one of a number of 'supply factors' that has the potential to affect seedling establishment and subsequent plant growth (Harper, 1977). Through favouring the growth of those species with the adaptive features best suited to a given range of mineral nutrient concentrations, competition between plant populations may also be influenced (Berendse, 1994; Mamolos, Elisseou and Veresoglou, 1995). In general, the strength of these effects will differ from one location to another, as mineral nutrients are unevenly distributed within ecosystems. For example, spatial variation in nitrogen has been shown to occur at multiple scales, from successional sequences down to the level of individual plants (Gross, Pregitzer and Burton, 1995).

On many man-made wastes, mineral nutrient concentrations are of primary importance in determining the likelihood of successful plant colonisation and community development (Marrs and Bradshaw, 1993). Many naturally colonising plant species show adaptations (such as a reduced growth rate or 'scavenging'/nitrogen fixing abilities) that allow them to accumulate sufficient nutrients for growth over time, but for the development of later successional communities a greater nitrogen capital is required. On china clay waste, Marrs and Bradshaw (1993) have estimated that this is equal to 700 kg N ha^{-1} (in the soil) for the establishment of *Salix* scrub, 1200 kg N ha^{-1} for *Betula-Quercus* woodland.

Colliery spoil in particular is characterised by both N and P deficiencies that severely limit most plant growth. This often means that some form of amelioration by addition of mineral nutrients is required before vegetation can become established on a site. In grassland ecosystems, various legume species can be used to make N available to associated plant species (Jefferies, Bradshaw and Putwain, 1981). Past research has shown that application of chemical fertilisers may also be used to increase both plant yields (Fitter and Bradshaw, 1974; Bloomfield, Handley and Bradshaw, 1982) and vegetation cover (Ash, Gemmell and Bradshaw, 1994). Less however is known about the opportunities for enhancing plant community diversity in vegetation predominantly dominated by tree and shrub species on colliery spoil.

Field layer introductions of herbaceous plants into recent woodlands have been described by a number of authors (e.g. Cohn and Packham, 1993; Francis, 1993; Packham *et al.*, 1995). These studies have largely been undertaken on sites of relatively high nutrient status and with substantial seed banks. On these sites, successful introductions have tended to depend upon the use of either herbicides and/or mulching in the period preceding canopy closure, or on adequate shade levels once the canopy layer has fully developed. In contrast, mineral nutrient concentrations in colliery spoil are often extremely low, and seed banks poorly developed. Whilst it has been generally unnecessary in previous studies to examine detailed spatial variations in mineral nutrient concentrations within sites, field layer introductions on colliery spoil are likely to be significantly affected by differences in these concentrations.

This chapter examines the effects of fertiliser application on seedling establishment and survival, and the subsequent pattern of flowering and seed production, for a number of herbaceous woodland species introduced to existing woodland/scrub vegetation at Bold Moss Tip and Gillars Green Wood, St. Helens (see section 2. 6). No attempt was made to introduce woodland grass species to the experimental sites, as appropriate species could not readily be obtained. Experiments were devised to test the hypothesis that both a lack of seed immigrants into the study sites, and poor mineral nutrient supply, were responsible for limiting colonisation by herbaceous woodland species. Although light climate was not experimentally manipulated, an attempt was also made to correlate establishment and survival with the existing light climate (presented in Chapter 4). The results in this chapter are discussed with reference to techniques that may be used to diversify herbaceous woodland plant communities, both on colliery spoil and other substrate types.

3. 2 Methods

3. 2. 1 Choice of herbaceous woodland species for introduction

Woodland/woodland edge species associated with *Quercus robur* - *Pteridium aquilinum* - *Rubus fruticosus* (W10) and *Quercus* spp. - *Betula* spp. - *Deschampsia flexuosa* (W16) woodland in north-west England were identified from National Vegetation Classification floristic tables (Rodwell, 1991). W10/W16 woodland is found naturally in the region; two ancient woodland sites in St. Helens, Stanley Bank (SJ 537973) and Glasshouse Close (SJ 532974) woods, have stands which show strong affinities with both woodland types (Fee, 1993). From this initial list, a number of species were chosen to reflect a likely range of tolerance to low pH. The seeds were obtained commercially, and were derived from both regional and national sources. The final choice of fourteen species was based upon those that were readily available as seed:

Campanula latifolia L.	(Giant Bellflower)
Digitalis purpurea	(Foxglove)
Galium mollugo L.	(Hedge Bedstraw)
Geranium robertianum L.	(Herb Robert)
Geum urbanum L.	(Herb Bennet)
Hyacinthoides non-scripta	(Bluebell)
Hypericum pulchrum L.	(Slender St John's-wort)
Primula vulgaris	(Primrose)
Prunella vulgaris L.	(Selfheal)
Scrophularia nodosa L.	(Common Figwort)
Silene dioica (L.) Clairv.	(Red Campion)
Stachys sylvatica L.	(Hedge Woundwort)
Taraxacum sp. Wigg.	(Dandelion)
Teucrium scorodonia L.	(Wood Sage)

Once obtained, the germinability of all seeds was tested by placing a known number of seeds of each species (the precise number varied according to species) on damp filter paper within petri dishes. For those species with larger seeds, fifty seeds of each species were placed in each petri dish, for smaller-seeded species up to two hundred were placed in each dish. Four petri dishes (replicates) were used for each species. For those species thought to require stratification prior to germination, an equal number of seeds were first treated by placing them in a refrigerator at 2°C for approximately three months. At the end of this period all seeds, both treated and untreated, were placed in an incubator at 20°C. All seeds that germinated in the incubator were counted and removed from each petri dish, to provide a cumulative score of the proportion of germinable seed for each species.

3. 2. 2 The effect of mineral nutrient addition on the establishment and survival of plants introduced as seed

At both study sites existing areas of woodland/scrub were chosen for the experimental work. At Bold Moss, three 8 x 8 m plots were randomly located in each of four blocks within areas of *Salix/Betula* scrub, two blocks on the western and two on the eastern slopes of the site (see Figure 2. 2, section 2. 6, and Plates 3. 1 and 3. 2). In April 1993 the

Plate 3. 1 Bold Moss Tip, St. Helens - Eastern slope showing sparse *Betula/Salix* cover, January 1995.

Plate 3. 2 Bold Moss Tip, St. Helens - Looking north west from above the western slope, January 1995.

corners of all plots were marked with painted wooded pegs, and the existing herbaceous vegetation sprayed with 'Roundup', a glyphosate-based herbicide. Six samples of spoil were taken at random locations to a depth of approximately 10 cm within each plot, and air dried. The samples were then sieved through a 2 mm mesh to remove most fragments of shale and coal. Analytical techniques described in Allen (1989) (see Appendix 3. 1) were used to determine the pH, and extractable phosphorus (P), potassium (K), and calcium (Ca) concentration of each spoil sample. Analysis of nitrogen was not undertaken because significant amounts of fossil nitrogen are found in colliery spoil; this makes any accurate determination of nitrogen an extremely difficult procedure. Following herbicide treatment, a combination of fast and slow-release NPK fertilisers ('Kemira' and 'Enmag' respectively) were used to randomly apply one of the following fertiliser treatments to plots within each block, in April 1993: **F1** - zero fertiliser (control); **F2** - N at 3 g m^{-2}, P at 3.5 g m^{-2}, K at 3.7 g m^{-2}; **F3** - N at 7.5 g m^{-2}, P at 8.8 g m^{-2}, K at 9.3 g m^{-2}, (N.B. NPK concentrations for the F3 fertiliser treatment equalled 250% of those for the F2 fertiliser treatment). A schematic representation of blocks, plots and fertiliser treatments at Bold Moss is given in Figure 3. 1.

At Gillars Green, three 9 x 9 m plots (blocks) were randomly located under the canopy of *Quercus/Betula*. Each main plot was sub-divided into nine 3 x 3 m sub-plots. Five randomly located spoil samples were collected and analysed from all sub-plots, using the methods employed at Bold Moss. The same fertiliser treatments as used at Bold were applied to the sub-plots to give three replicates of each treatment per block (Figure 3. 2). No attempt was made to alter spoil pH within plots at either experimental site. This allowed the testing of the hypothesis that poor mineral nutrient supply was likely to be associated with reduced plant establishment and survival at both sites.

On 19 May 1993 the experimental herbaceous seed mix was applied to all plots at both experimental sites, at a density of 100 seeds m^{-2} for each species. *Digitalis purpurea* was omitted from the seed mixture at Gillars Green, as seedlings of this species were already established within the field layer. On 3 July 1993, four 0.5 m x 0.5 m permanent sampling areas (quadrats) were randomly located within each of the twelve plots at Bold Moss (random number tables were used to determine precise coordinates) to give sixteen per fertiliser treatment. Their position was marked by four iron bolts sunk into the ground, one at each corner. At Gillars Green three 0.5 m x 0.5 m sampling areas per sub-plot (twenty-seven per fertiliser treatment) were positioned in the same way on 5 July 1993. All sampling areas were subsequently relocated with the use of a metal detector.

Figure 3. 1 A schematic representation of blocks, plots and fertiliser treatments at Bold Moss Tip, St. Helens. N.B. Where **F1**= zero, **F2**= low and **F3**= high fertiliser application. An example of the location of the four sampling areas in each plot is also shown.

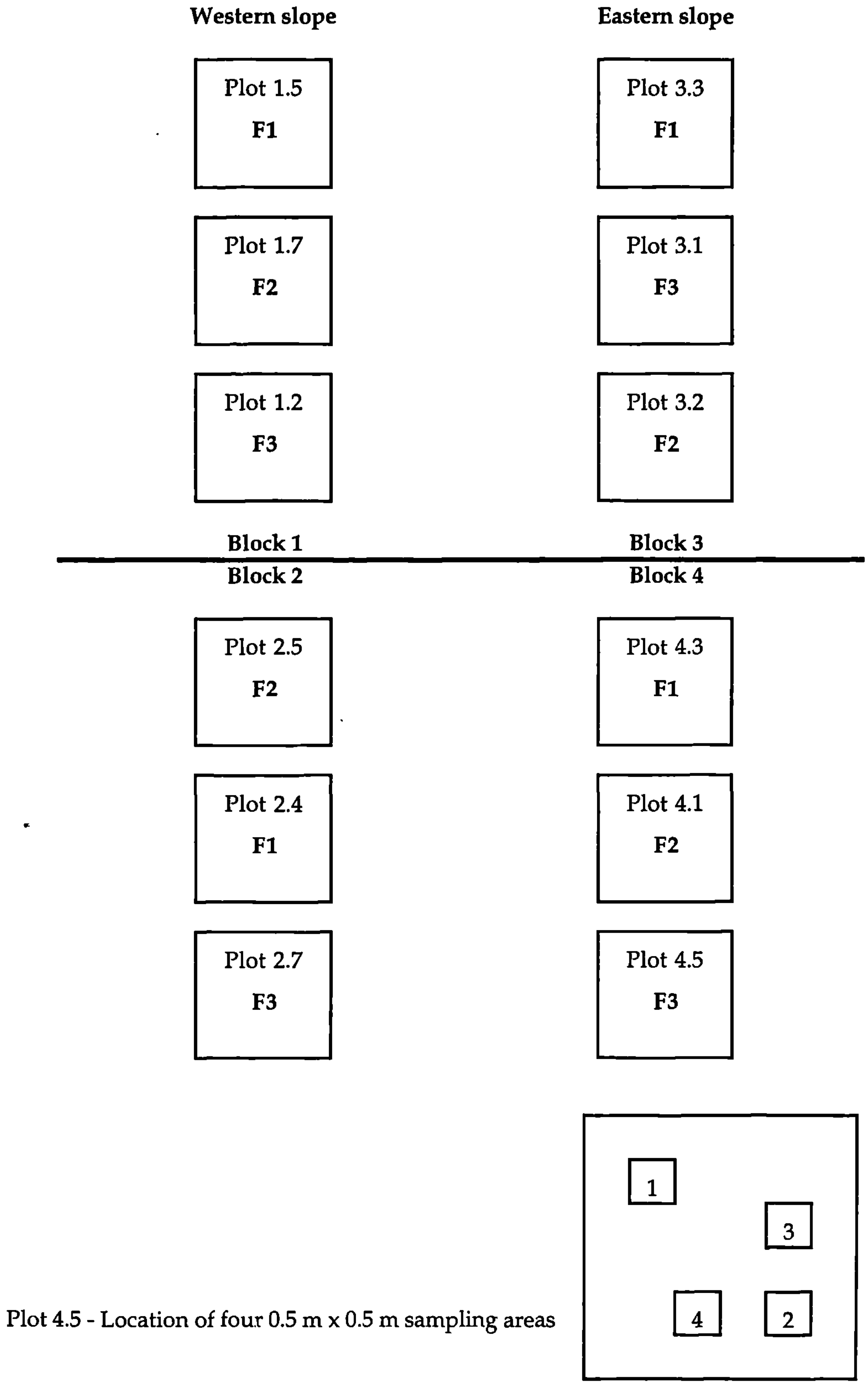


Figure 3. 2 A schematic representation of plots (blocks), sub plots and fertiliser treatments at Gillars Green Wood, St. Helens. N.B. Where F1= zero, F2= low and F3= high fertiliser application.

Plot/Block 1

F3	F3	F1
F1	F1	F3
F2	F2	F2

Plot/Block 2

F3	F2	F3
F2	F1	F1
F1	F2	F3

Plot/Block 3

F2	F3	F2
F1	F1	F1
F3	F3	F2

In each plot/sub-plot with fertiliser treatments F1 and F3 at both experimental sites, establishment and survival of individual plants of all introduced species was monitored within three permanent sampling areas (randomly chosen on the first census date at Bold) approximately every eight weeks. A full list of all census dates for both experimental sites is given in Appendix 3. 2. A metal-framed 0.5 m x 0.5 m quadrat with string sub-divisions was located on each sampling area with use of the buried markers. The location of individuals was then transferred onto data sheets showing a mapped representation of the divided quadrat. In the first few weeks following germination, cotyledon leaves of each species were identified using photographs of seedlings of each species, which had been grown in pots earlier in the year. For species with winter-green leaves, the death of individuals was determined on all census dates. For all other species, the census date following the last time an individual was recorded was taken to mark the time it left the population.

At the same time as individual plants were plotted, the total number of plants of all introduced species in all sampling areas, across the complete range of fertiliser treatments (F1, F2 and F3), was recorded. This provided a count of all plants within fertiliser treatments on each census date. Monitoring was carried out from 11/12 July 1993, to 23 July 1995 at Bold Moss, and to 25 May 1995 at Gillars Green.

3. 2. 3 The effect of mineral nutrient addition on flowering and seed production in plants established from seed

Flowering was monitored in plants established from seed in plots at the Bold Moss site on three occasions during the summers of 1994 and 1995 - 27 May, 28 June and 1 September 1994; 18 May, 20 June and 28 July 1995. Twelve 0.5 x 0.5 m sampling areas were randomly located in each 8 x 8 m plot, and the number of individuals of all species recorded. The proportion of plants flowering in each sampling area was noted, as was the number of flowers per plant. Where an individual corolla had fallen from a plant, the presence of sepals, inflorescences, or infructescences (*sensu* Bell, 1991) were used to determine a cumulative total of flowers per plant at a given census date.

In 1994 and 1995, ripe closed fruits were collected from plants within plots whenever they were seen. The seeds within fruits were counted, slowly dried at 15°C and stored in paper bags until the autumn of the same year, at which time the seeds were placed on damp filter paper in petri dishes, and placed in an incubator at 20°C. Seeds that germinated were counted and removed from the petri dishes. These data were used to estimate mean annual seed return from plants, including the proportion of germinable seed, and to determine the effects of fertiliser treatments on these measures.

3.2.4 Data analysis

A simplified difference equation ($N_{t+1} = N_t + B - D$; where N_{t+1} equals the population number at time t, plus births B, minus deaths D) (Silvertown and Lovett Doust, 1993) was used to plot graphs of species' population flux; for the (net) total populations at the two experimental study sites, for all individuals in each experimental block, and (where numbers allowed) for all individuals in relation to two of the fertiliser treatments (F1 and F3). Graphs were constructed so as to present data on cumulative births/deaths and the net population size at each census date. The population density of introduced species was calculated as the mean number of seedlings/plants m^{-2} for all fertiliser treatments and blocks at each of the census dates. Where numbers allowed, probability values for the total number of individuals of each species within each treatment and block at both sites were calculated using a repeated measures analysis of variance (SAS, 1985). For each species, counts of individuals from all sampling areas (quadrats) in each plot/sub-plot were firstly pooled to give a total count per plot/sub-plot. All data for an individual census date were removed from the analysis if zero counts were recorded for all the pooled data (i.e. from each of the twelve plots at Bold Moss, or twenty-seven sub-plots at Gillars Green) on that date. The data for each species were subsequently transformed (Log_{10}+1) to stabilise sample variance prior to analysis.

Data for flowering and seed set were insufficient to allow any statistical analyses, therefore only numbers in relation to fertiliser treatment and block are presented.

3.3 Results

3.3.1 Seed germinability

The data for germinability of all species used in the herbaceous introductions are presented in Table 3. 1. Seeds were treated with a low temperature experience for all species except *Galium mollugo, Prunella vulgaris* and *Silene dioica*. Only *Scrophularia nodosa,* and to a lesser extent *Stachys sylvatica,* showed any notable increase in germinability following treatment, while *Digitalis purpurea, Hypericum pulchrum* and *Teucrium scorodonia* showed marked decreases. All seeds of *Campanula latifolia* and *Geum urbanum* failed to germinate. No reason for these decreases or failures could be

Table 3. 1 Germinability of seeds of herbaceous species used in the experiments at Bold Moss Tip and Gillars Green Wood, St. Helens.

SPECIES	*Campanula latifolia*	*Digitalis purpurea*	*Galium mollugo*	*Geranium robertianum*	*Geum urbanum*	*Hyacinthoides non-scripta*	*Hypericum pulchrum*
Proportion of untreated seeds germinated	0/200	207/250	411/800	98/200	0/200	0/200	166/200
Percentage of untreated seeds germinated	0	82.8	51.4	49.0	0	0	83.0
Proportion of treated seeds germinated	0/200	122/250	-	86/200	0/200	1/200	123/200
Percentage of treated seeds germinated	0	48.8	-	43.0	0	0.5	61.5

SPECIES	*Primula vulgaris*	*Prunella vulgaris*	*Scrophularia nodosa*	*Silene dioica*	*Stachys sylvatica*	*Taraxacum sp.*	*Teucrium scorodonia*
Proportion of untreated seeds germinated	7/200	63/200	42/200	63/200	1/200	2/200	134/200
Percentage of untreated seeds germinated	3.5	31.5	21.0	31.5	0.5	1.0	67.0
Proportion of treated seeds germinated	4/200	-	68/200	-	6/200	3/200	49/200
Percentage of treated seeds germinated	2.0	-	34.0	-	3.0	1.5	24.5

determined, and there was no evidence of fungal pathogens in the petri dishes during storage in the fridge. Data on germinability were too few to allow any statistical analysis of the results.

3. 3. 2 The effect of mineral nutrient addition on the establishment and survival of plants introduced as seed

3. 3. 2. 1 Spoil pH and chemical analysis

Results of the analysis of spoil pH and the spoil concentrations of P, K and Ca are shown for Bold Moss and Gillars Green in Tables 3. 2 and 3. 3 respectively. The substrate at Bold Moss showed a relatively wide variation in mean pH (pH 3.2 - pH 6.3) and P concentrations (0.26 - 23.18 mg kg^{-1}) both between plots, and between blocks. Similar variations were seen for mean K (2.40 - 27.62 mg kg^{-1}) and Ca (21.52 - 94.9 mg kg^{-1}) concentrations at Bold Moss. At Gillars Green mean pH values remained consistently low (<pH 4) within and between blocks, while mean P concentrations showed some variation within blocks, the maximum range being 3.10 - 14.64 mg kg^{-1}. Mean K concentrations at Gillars Green were relatively consistent between blocks (1.68 - 13.92 mg kg^{-1}). Concentrations of Ca at this site were similar in blocks 1 and 2 (see Figure 3. 2) (12.00 - 29.60 mg kg^{-1}), and slightly higher in block 3 (range 18.75 - 33.52 mg kg^{-1}).

The substrate at Bold Moss was expected to show a high level of variation in spoil pH and mineral nutrient concentrations as a result of the tipping of PFA on parts of the site. In comparison to other colliery tips in the region, P and Ca concentrations were found to be relatively high within some plots and blocks at Bold, and similar to those found in some podzol and regosol soils (Allen, 1989). At Gillars Green the generally lower mineral nutrient concentrations were closer to what would be expected for colliery spoil in the region (Ash, 1983).

3. 3. 2. 2 Population flux

At both study sites *Campanula latifolia* failed to germinate and *Taraxacum* spp. was already well established. These species are omitted from any further discussion. Insufficient numbers of individuals for two species (*Digitalis purpurea* and *Scrophularia nodosa*), meant that detailed analysis of data in the form of population flux graphs could not be plotted. Thus a detailed analysis is presented for ten species at Bold Moss, and four species at Gillars Green. Net population flux for the total species population across all

Table 3. 2 Mean pH and extractable mineral nutrient concentration - mg kg^{-1} air dry soil - (with sample standard errors) for herbaceous experimental plots, Bold Moss Tip, St. Helens.

PLOT NO	A1 P2	A1 P5	A1 P7	A2 P4	A2 P5	A2 P7	A3 P1	A3 P2	A3 P3	A4 P1	A4 P3	A4 P5
Mean pH	4.4	5.5	6.3	3.4	3.9	3.2	3.7	4.8	5.2	4.6	4.2	4.8
(±SE)	±0.45	±0.36	±0.07	±0.07	±0.34	±0.02	±0.17	±0.27	±0.27	±0.45	±0.18	±0.44
Mean P (mg/kg)	3.85	11.26	16.99	1.47	4.31	0.26	6.51	14.36	23.18	11.25	13.28	16.97
(±SE)	±1.66	±2.75	±2.22	±0.68	±1.93	±0.15	±1.83	±2.53	±2.29	±4.03	±2.25	±2.97
Mean K (mg/kg)	10.29	12.37	27.62	3.93	6.73	2.40	8.87	20.69	9.07	11.16	12.53	12.29
(±SE)	±2.28	±1.15	±6.80	±1.33	±1.96	±0.87	±3.42	±2.30	±4.03	±3.14	±3.10	±2.85
Mean Ca (mg/kg)	93.00	77.20	82.40	22.36	51.40	21.52	39.70	53.20	94.90	78.90	72.90	66.80
(±SE)	±29.60	±17.40	±22.20	±5.80	±11.00	±7.84	±17.60	±25.00	±36.40	±13.30	±16.20	±29.00

Table 3. 3 Mean pH and extractable mineral nutrient concentration - mg kg^{-1} air dry soil - (with sample standard errors) for herbaceous experimental sub-plots, Gillars Green Wood, St. Helens.

PLOT NO	A1	A2	A3	A4	A5	A6	A7	A8	A9
Mean pH	3.3	3.5	3.5	3.3	3.4	3.3	3.4	3.4	3.4
(±SE)	±0.03	±0.02	±0.04	±0.03	±0.04	±0.03	±0.02	±0.05	±0.02
Mean P (mg/kg)	14.64	8.65	7.76	3.10	3.45	3.67	3.98	3.98	3.76
(±SE)	±1.90	±2.83	±3.27	±1.22	±0.84	±1.03	±1.28	±1.31	±1.10
Mean K (mg/kg)	7.27	4.85	6.64	8.37	5.17	8.01	5.64	10.44	13.92
(±SE)	±1.46	±1.12	±0.86	±1.23	±2.10	±1.71	±2.22	±1.07	±1.09
Mean Ca (mg/kg)	20.34	13.56	14.04	15.64	3.52	3.55	2.83	2.27	5.79
(±SE)	±4.64	±1.18	±2.07	±1.69	±3.52	±3.55	±2.83	±2.27	±5.79

PLOT NO	B1	B2	B3	B4	B5	B6	B7	B8	B9
Mean pH	3.5	3.2	3.4	3.2	3.3	3.4	3.4	3.4	3.5
(±SE)	±0.03	±0.03	±0.02	±0.04	±0.03	±0.02	±0.03	±0.02	±0.02
Mean P (mg/kg)	3.06	6.29	2.49	9.65	4.63	5.24	12.38	11.01	10.00
(±SE)	±0.90	±1.73	±0.53	±2.26	±1.09	±1.53	±1.43	±1.38	±0.49
Mean K (mg/kg)	9.19	4.39	3.76	6.61	2.57	4.32	10.41	9.10	1.68
(±SE)	±1.96	±1.43	±1.45	±1.96	±1.16	±1.27	±1.08	±4.78	±1.04
Mean Ca (mg/kg)	15.92	13.62	12.00	29.6	20.8	22.86	20.19	19.61	26.40
(±SE)	±2.89	±1.97	±2.79	±14.60	±10.10	±6.54	±2.77	±8.17	±13.80

PLOT NO	C1	C2	C3	C4	C5	C6	C7	C8	C9
Mean pH	3.4	3.4	3.5	3.3	3.6	3.4	3.5	3.4	3.6
(±SE)	±0.05	±0.02	±0.04	±0.05	±0.03	±0.02	±0.07	±0.07	±0.06
Mean P (mg/kg)	8.72	9.87	11.32	8.41	1.80	10.84	9.97	4.05	2.79
(±SE)	±1.08	±0.66	±1.89	±1.13	±0.64	±2.26	±2.95	±1.26	±1.19
Mean K (mg/kg)	6.85	7.39	9.22	4.90	10.04	7.90	8.74	8.98	10.22
(±SE)	±0.65	±1.51	±1.09	±0.64	±0.64	±2.36	±3.19	±1.66	±1.54
Mean Ca (mg/kg)	31.40	22.01	20.69	28.27	27.25	18.75	33.52	26.46	29.29
(±SE)	±1.96	±3.56	±2.49	±7.44	±4.01	±5.60	±9.30	±4.20	±3.80

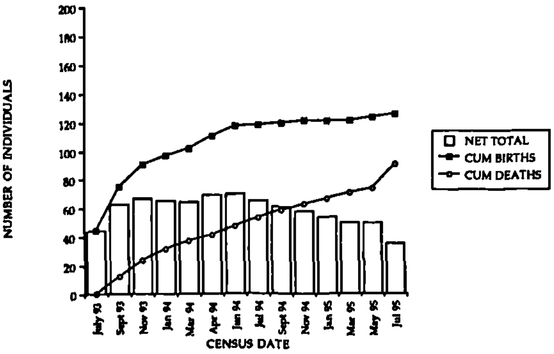
treatments and blocks, and for populations within treatments and blocks, are shown in Figures 3. 3 to 3. 5 for Bold Moss, and Figures 3. 6 to 3. 8 for Gillars Green.

BOLD MOSS TIP

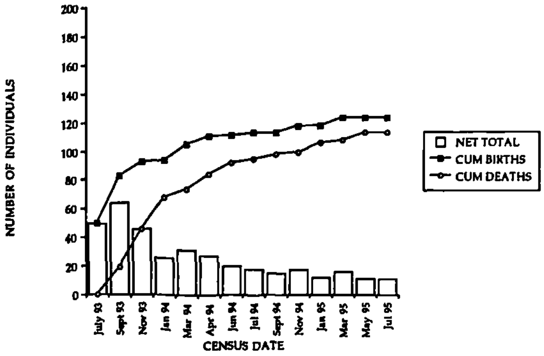
Data pooled from all of the sampling areas within the F1 and F3 fertiliser treatments (a total area of 6 m^2) at the Bold site (Figure 3. 3) show that *Hyacinthoides non-scripta* had the highest net population size of all sown species, exceeding 100 individuals in the spring of 1994 and 1995. *Galium mollugo, Geum urbanum, Prunella vulgaris* and *Teucrium scorodonia* all had a net population size of around 50 individuals for the majority of the monitoring period, as did *Primula vulgaris* following germination in the spring of 1994. *Geranium robertianum, Hypericum pulchrum, Silene dioica* and *Stachys sylvatica* had a lower net population size which did not exceed 20 individuals (in the total area of 6 m^2) by the summer of 1995. Results for some species showed little relationship with the seed germinability data (Table 3. 1). *Geum urbanum, Hyacinthoides non-scripta, Primula vulgaris* and *Stachys sylvatica* only germinated in any numbers in the field; germinability in all these species was most probably enhanced as a result of natural temperature variations during the winter and spring of 1993/1994.

The pooled data show that for most species, changes in cumulative births (recruitment) and deaths (mortality) continued to affect net population size throughout the monitoring period. Relatively high levels of recruitment were seen in seven species. Recruitment for *Galium mollugo* and *Geranium robertianum* reached a maximum in July/September 1993. Net population size for *Galium mollugo* reached a maximum in April/June 1994 and subsequently fell slowly up until July 1995; the net population size for for *Geranium robertianum* reached a maximum in September 1993 and fell sharply between November 1993 and January 1994. *Geum urbanum* and *Teucrium scorodonia* showed two distinct peaks in recruitment, during September/November 1993 and June 1994. The net population size of *Geum urbanum* subsequently fell steadily up until July 1995, while for *Teucrium scorodonia* only a small decrease was seen between May and July 1995. *Prunella vulgaris* showed a steady level of recruitment throughout the monitoring period; the net population size changed very little once it reached a maximum in June 1994. For *Hyacinthoides non-scripta* and *Primula vulgaris* peaks in recruitment were seen in March/April and June 1994 respectively. Net population size for *Hyacinthoides non-scripta* fell sharply between April and June 1994, and increased again in March 1995; much of this trend may be due to the fact that an exact determination of net population size between these two census dates was not possible for this species, as shoots (leaves) were only present above ground for a part of this period. Net population size for *Primula vulgaris* remained relatively constant

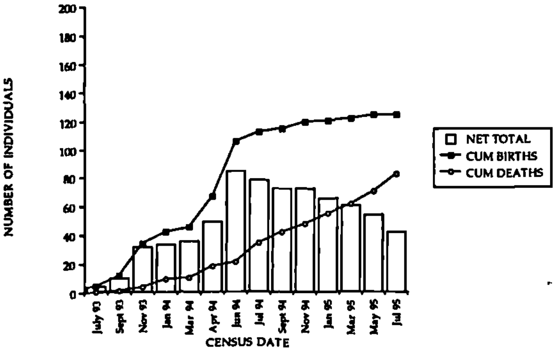
Figure 3. 3 Net population flux for introduced species, Bold Moss Tip, St. Helens - pooled data from F1 and F3 sampling areas (total area of 6 m^2). N.B. Where 'Net Total'= Net population size.



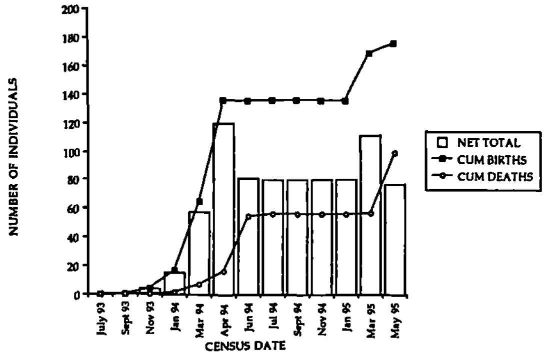
Galium mollugo



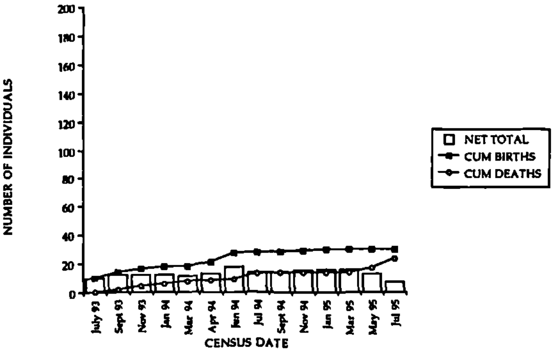
Geranium robertianum



Geum urbanum

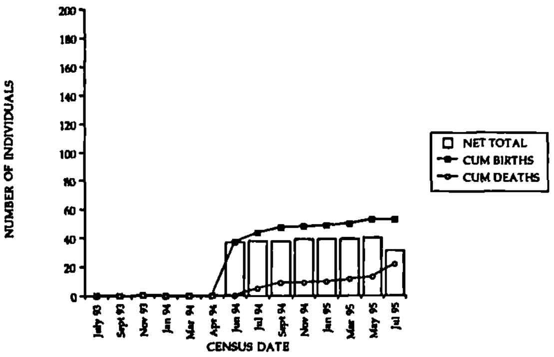


Hyacinthoides non-scripta

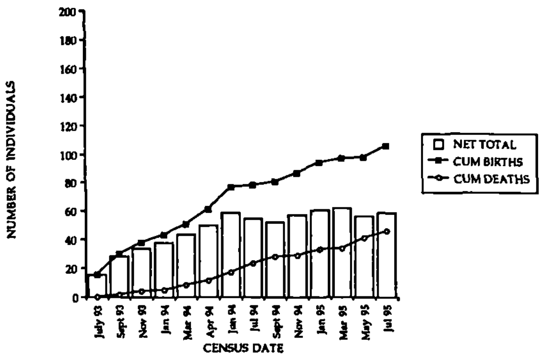


Hypericum pulchrum

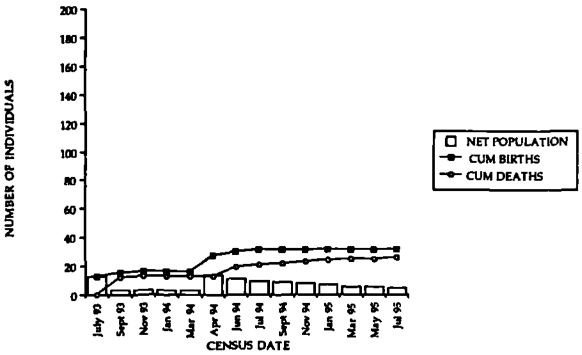
Figure 3. 3 Continued



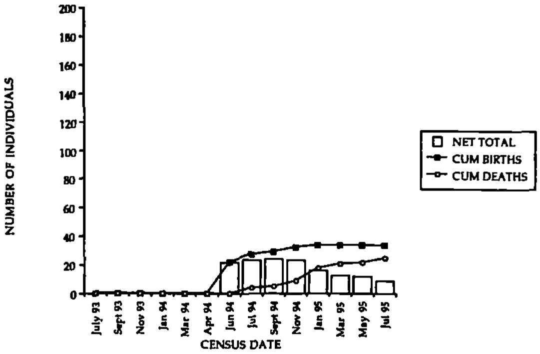
Primula vulgaris



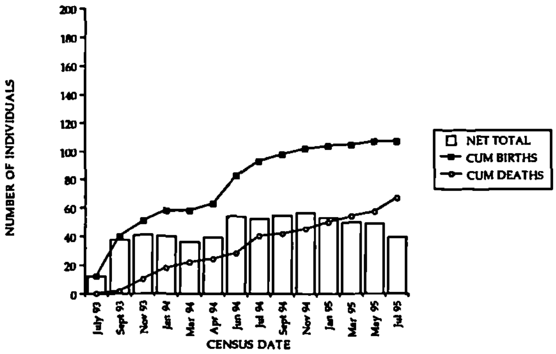
Prunella vulgaris



Silene dioica



Stachys sylvatica



Teucrium scorodonia

Figure 3. 4 Net population flux in relation to mineral nutrient addition for introduced species (total area of 3m^2 per fertiliser treatment), Bold MossTip, St. Helens. N.B. Where 'Net Total'= Net population size.

ZERO NUTRIENT INPUT (F1)

HIGH NUTRIENT INPUT (F3)

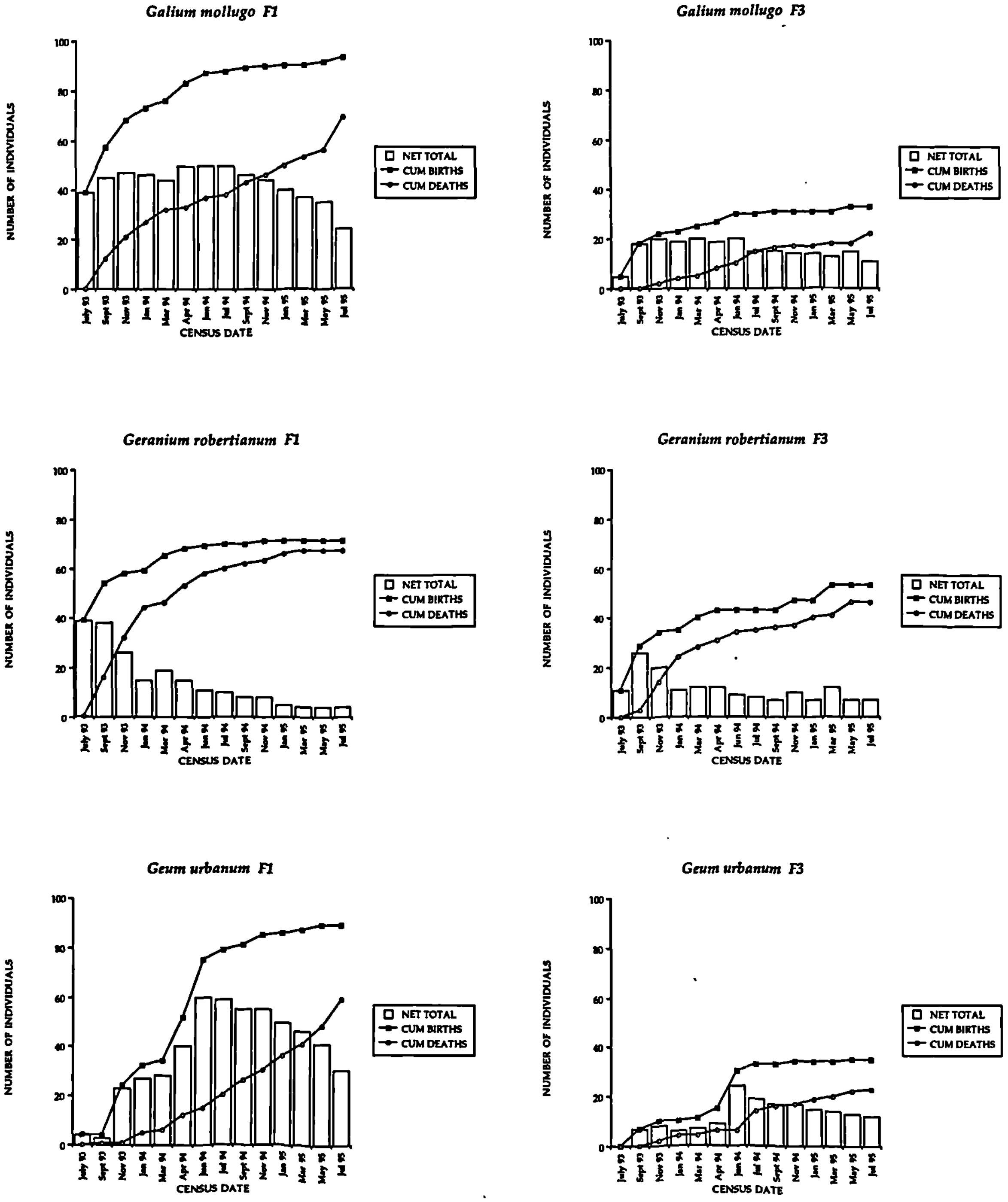
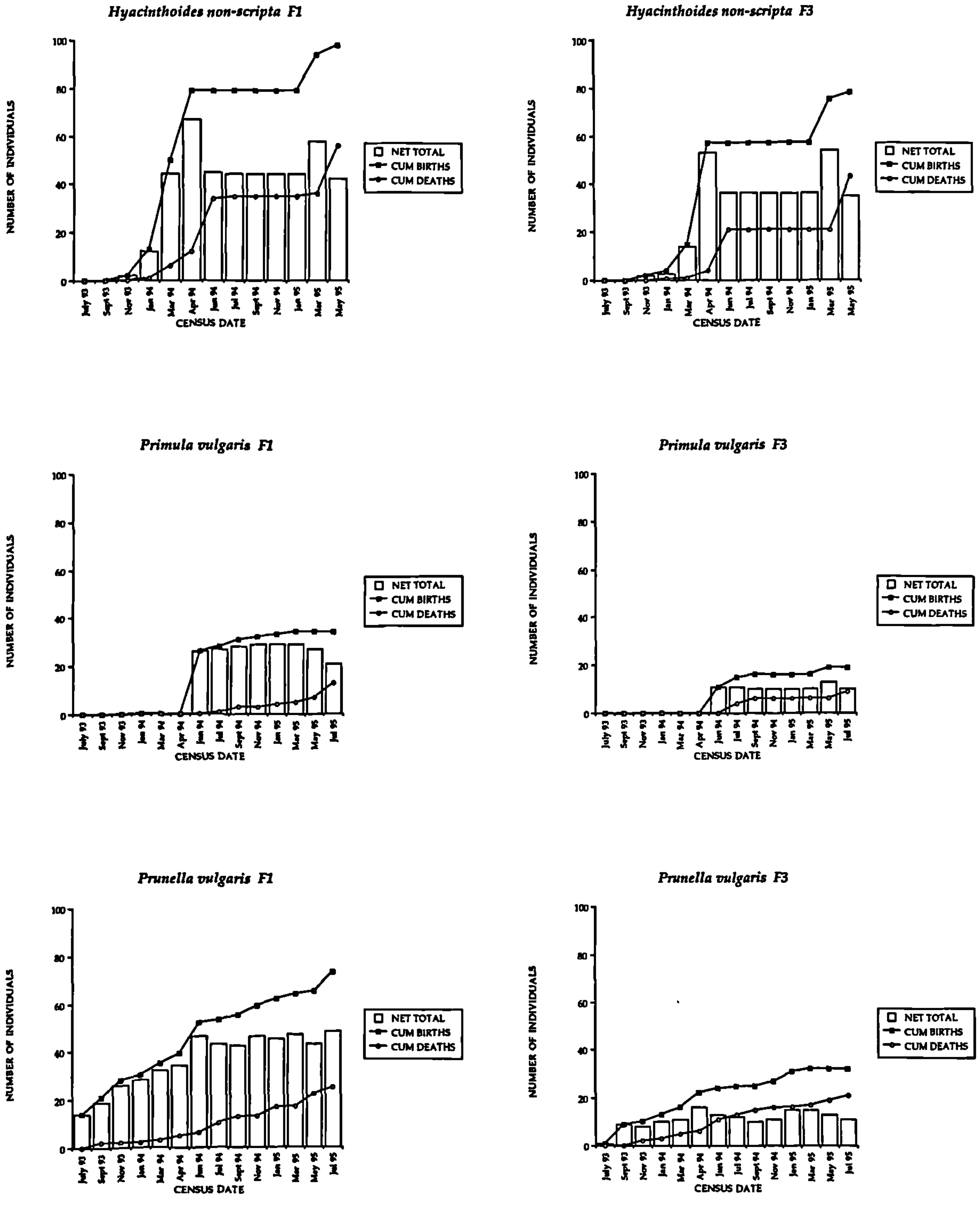


Figure 3. 4 Continued


Hyacinthoides non-scripta F1

Hyacinthoides non-scripta F3

Primula vulgaris F1

Primula vulgaris F3

Prunella vulgaris F1

Prunella vulgaris F3

Figure 3. 4 Continued

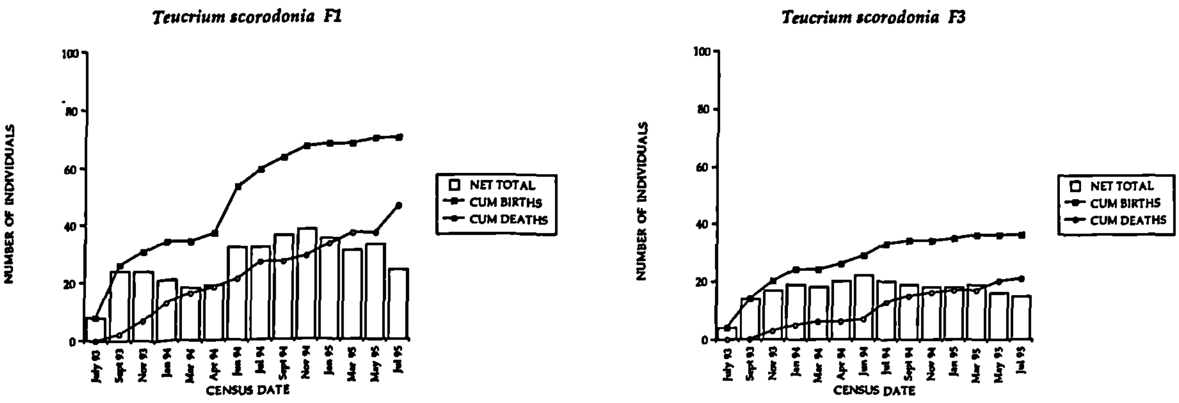
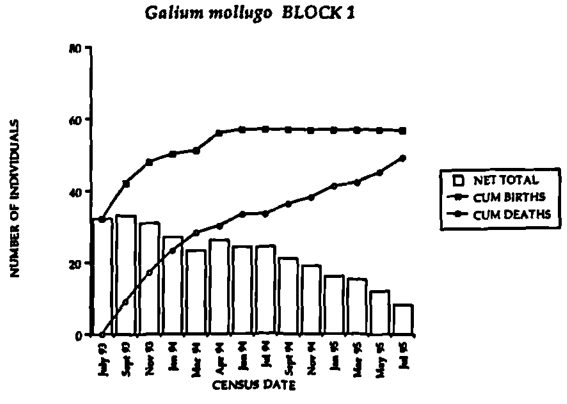
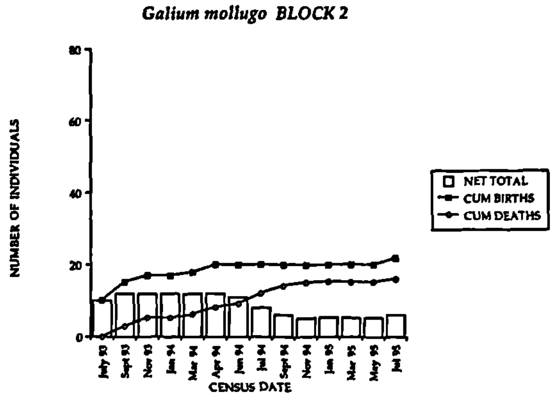
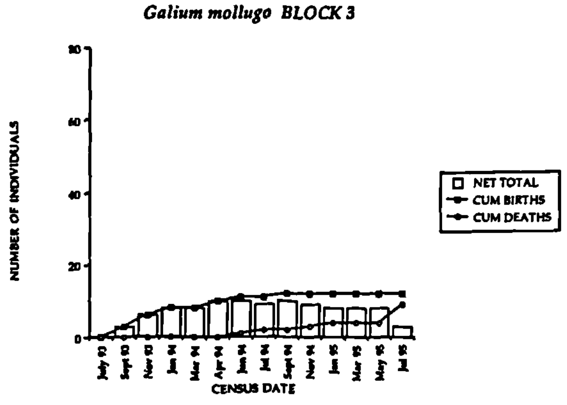
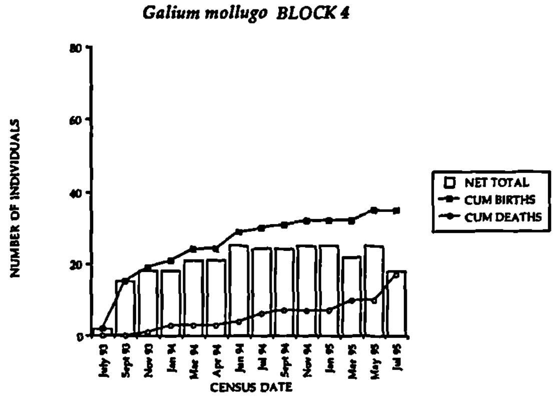


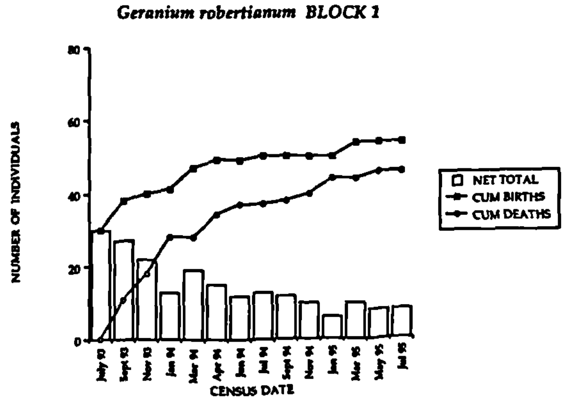
Figure 3. 5 Net population flux in relation to block for introduced species (total area of 1.5 m^2 per block), Bold Moss Tip, St. Helens. N.B. Where 'Net Total'= Net population size.


Galium mollugo BLOCK 1


Galium mollugo BLOCK 2


Galium mollugo BLOCK 3


Galium mollugo BLOCK 4


Geranium robertianum BLOCK 1

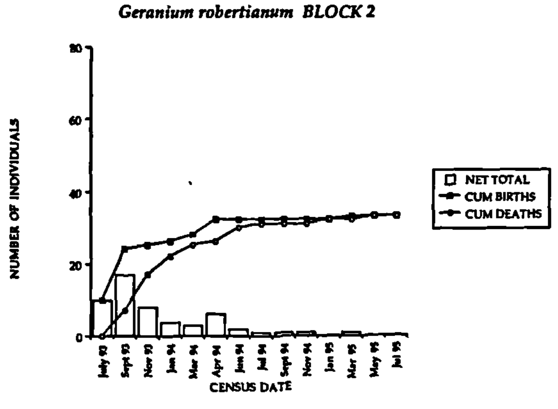
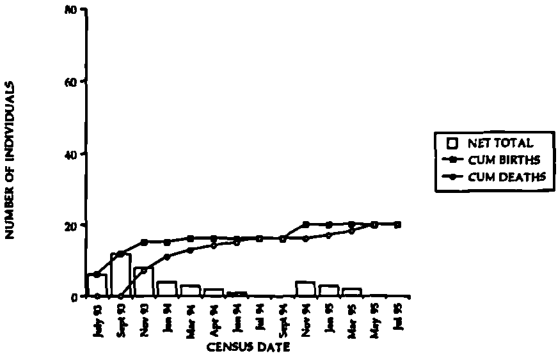
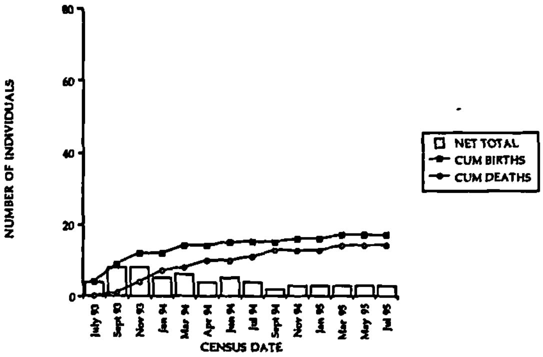

Geranium robertianum BLOCK 2

Figure 3. 5 Continued

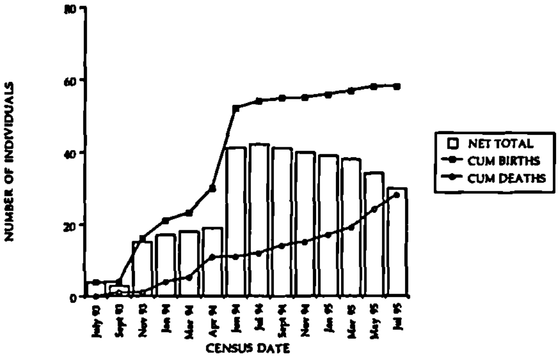
Geranium robertianum BLOCK 3



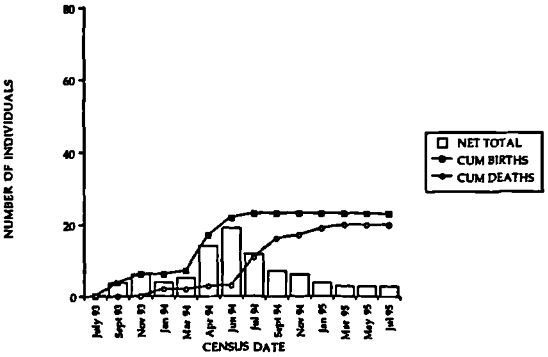
Geranium robertianum BLOCK 4



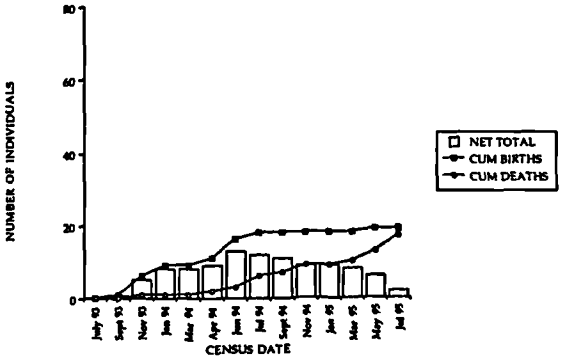
Geum urbanum BLOCK 1



Geum urbanum BLOCK 2



Geum urbanum BLOCK 3



Geum urbanum BLOCK 4

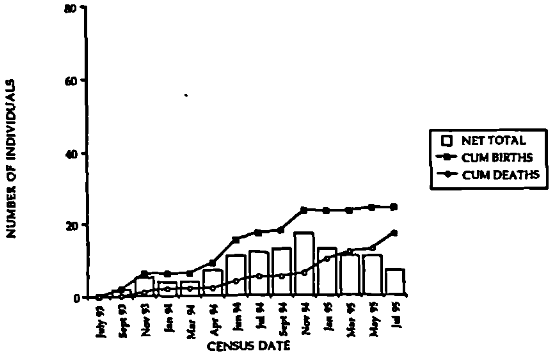
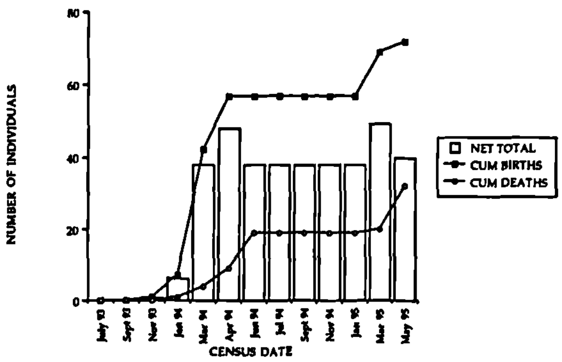
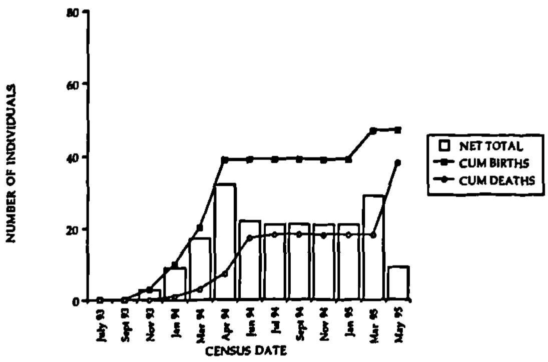
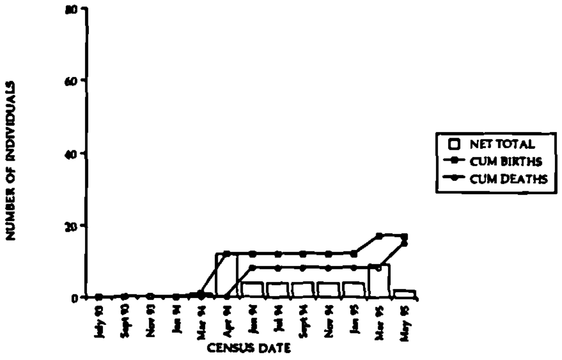
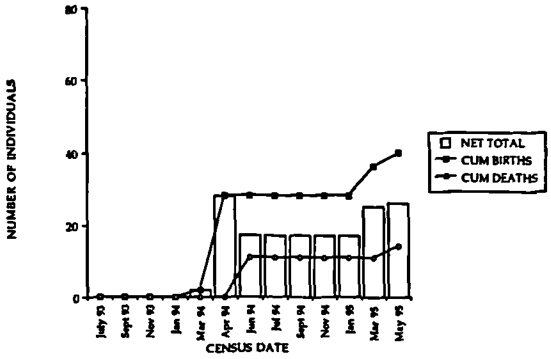


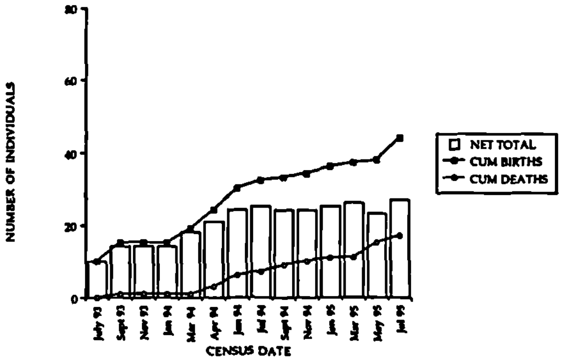
Figure 3. 5 Continued


Hyacinthoides non-scripta BLOCK 1


Hyacinthoides non-scripta BLOCK 2


Hyacinthoides non-scripta BLOCK 3


Hyacinthoides non-scripta BLOCK 4


Prunella vulgaris BLOCK 1

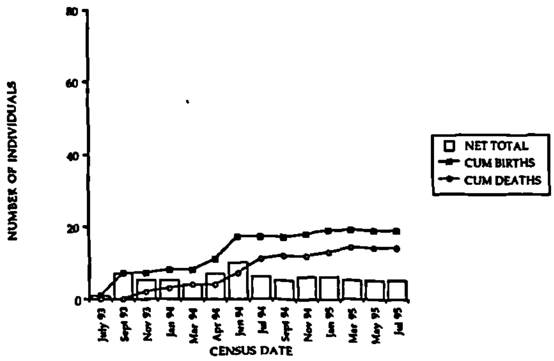
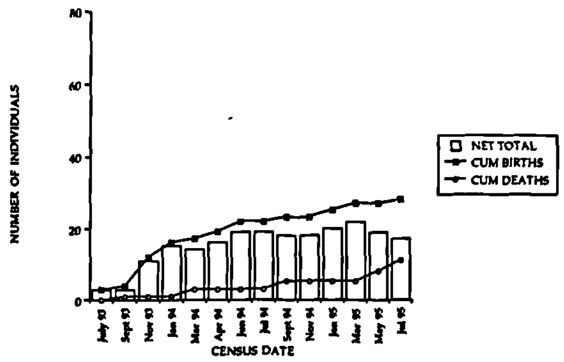
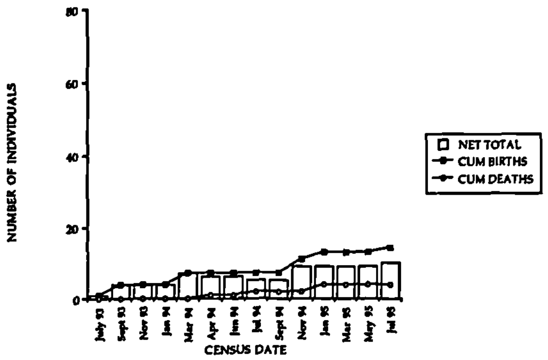

Prunella vulgaris BLOCK 2

Figure 3. 5 Continued

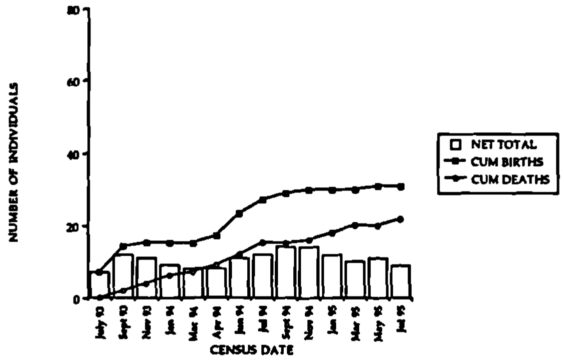
Prunella vulgaris BLOCK 3



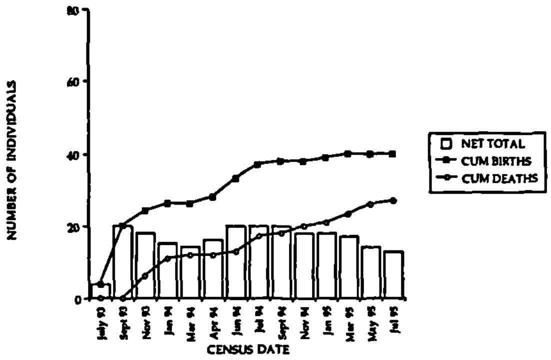
Prunella vulgaris BLOCK 4



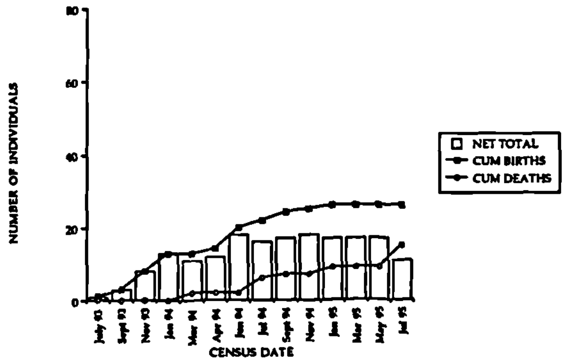
Teucrium scorodonia BLOCK 1



Teucrium scorodonia BLOCK 2



Teucrium scorodonia BLOCK 3



Teucrium scorodonia BLOCK 4

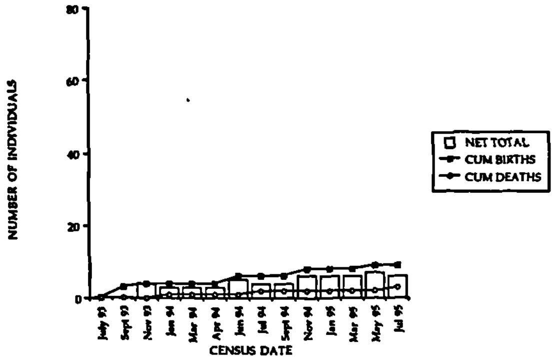
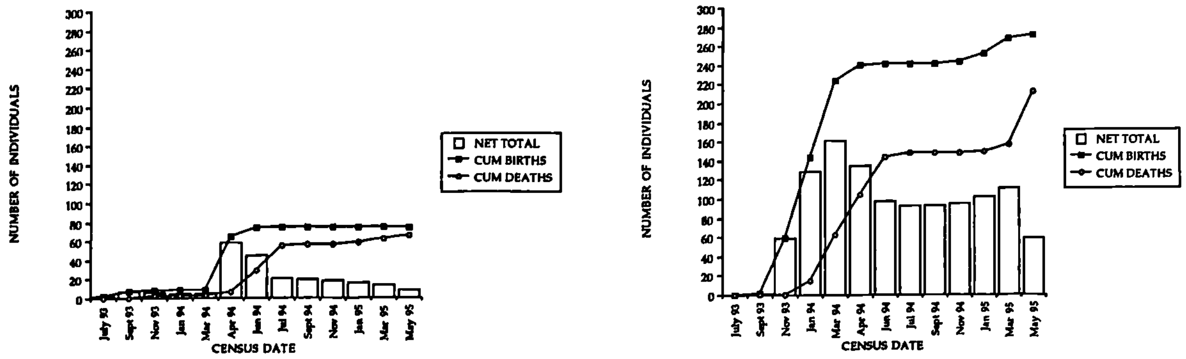
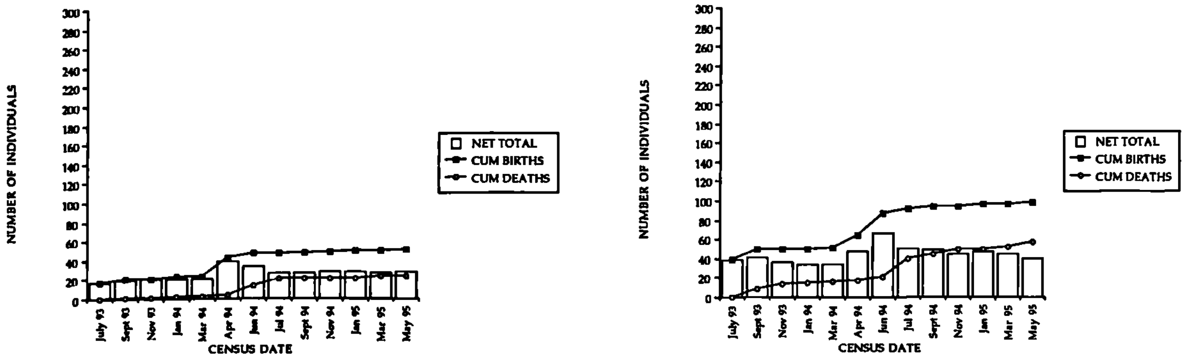


Figure 3. 6 Net population flux for introduced species, Gillars Green Wood, St. Helens. Pooled data from F1 and F3 sampling areas (total area of 13.5 m²). N.B. Where 'Net Total'= Net population size.


Geum urbanum

Hyacinthoides non-scripta


Silene dioica

Teucrium scorodonia

Figure 3. 7 Net population flux in relation to mineral nutrient addition for introduced species (total area of 6.75 m^2 per fertiliser treatment), Gillars Green Wood, St. Helens. N.B. Where 'Net Total'= Net population size.

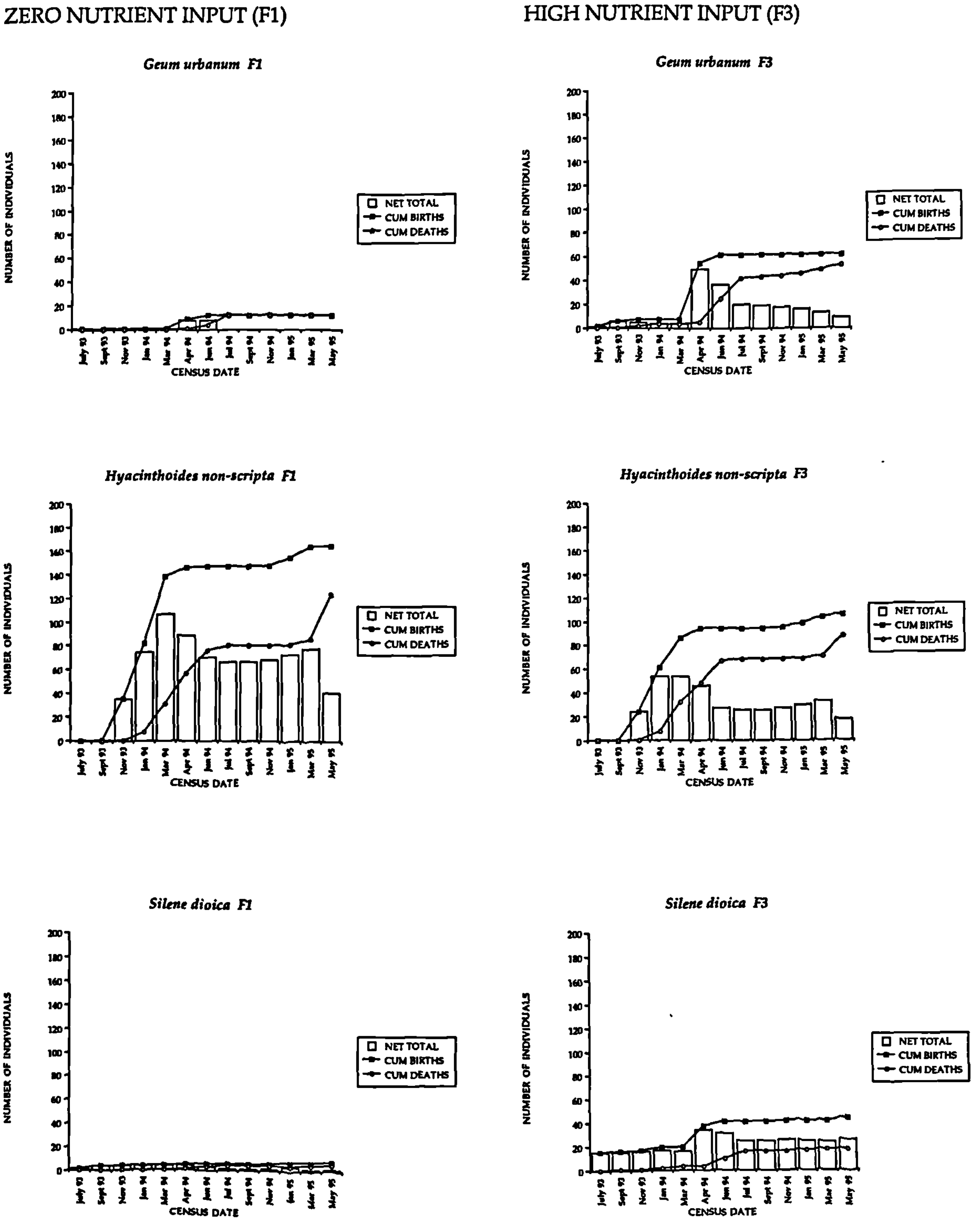
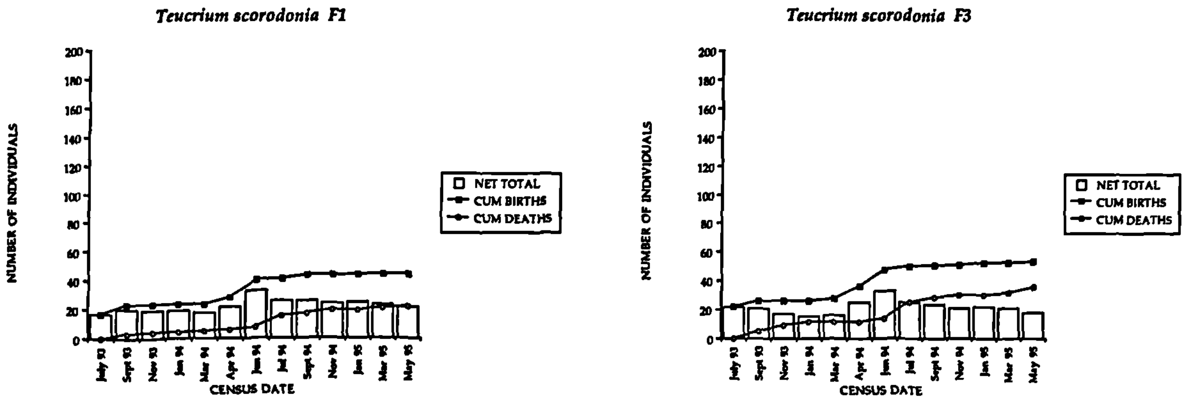


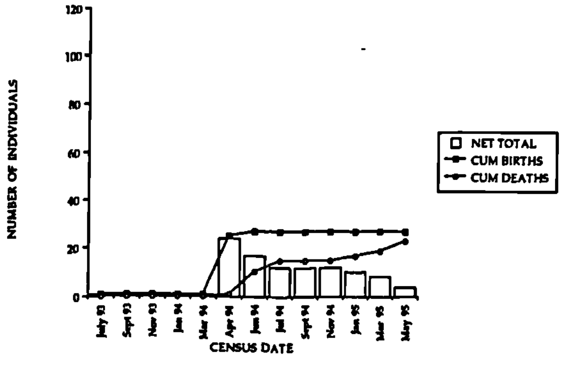
Figure 3. 7 Continued



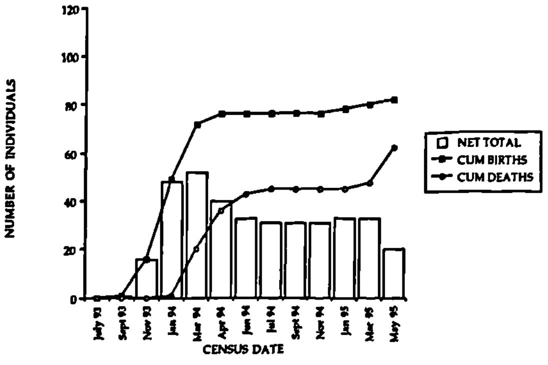
Teucrium scorodonia F1

Teucrium scorodonia F3

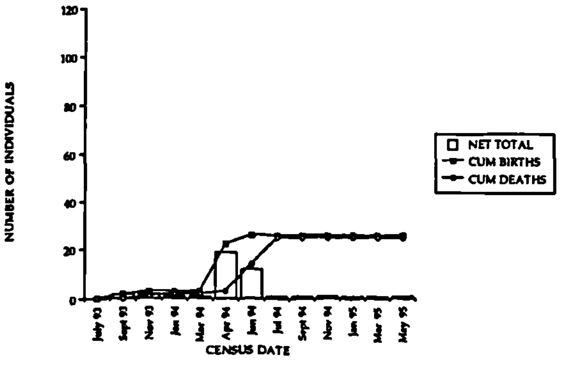
Figure 3. 8 Net population flux in relation to block for introduced species (total area of 4.5 m^2 per block), Gillars Green Wood, St. Helens. N.B. Where 'Net Total'= Net population size.



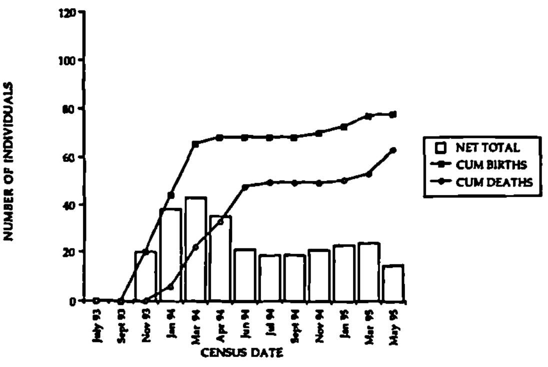
Geum urbanum BLOCK 1



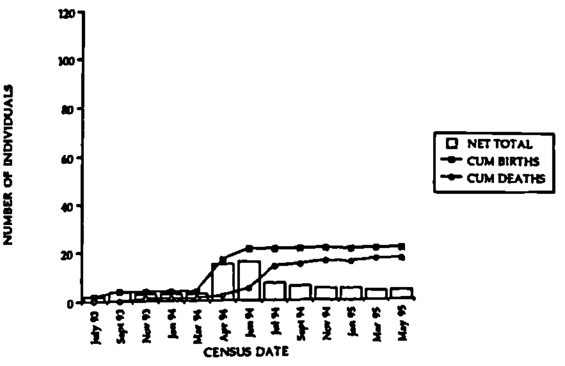
Hyacinthoides non-scripta BLOCK 1



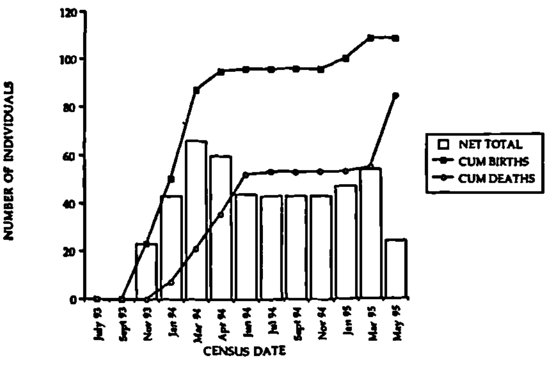
Geum urbanum BLOCK 2



Hyacinthoides non-scripta BLOCK 2



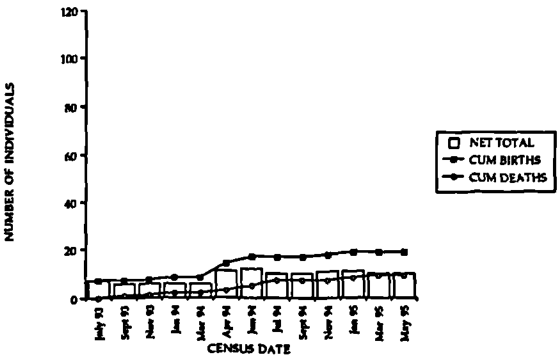
Geum urbanum BLOCK 3



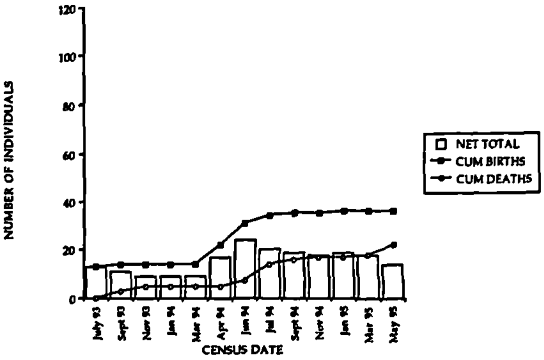
Hyacinthoides non-scripta BLOCK 3

Figure 3. 8 Continued

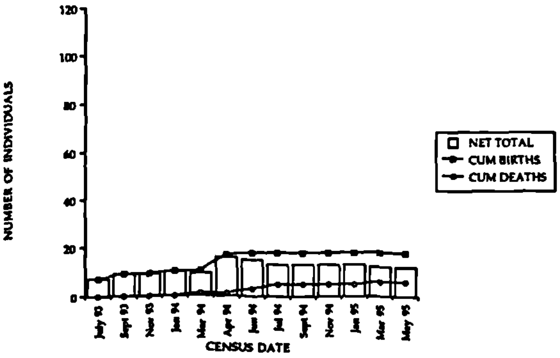
Silene dioica BLOCK 1



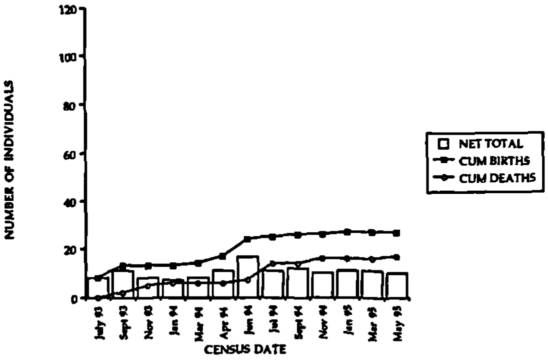
Teucrium scorodonia BLOCK 1



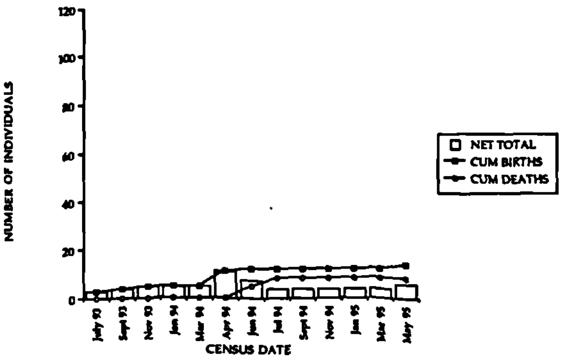
Silene dioica BLOCK 2



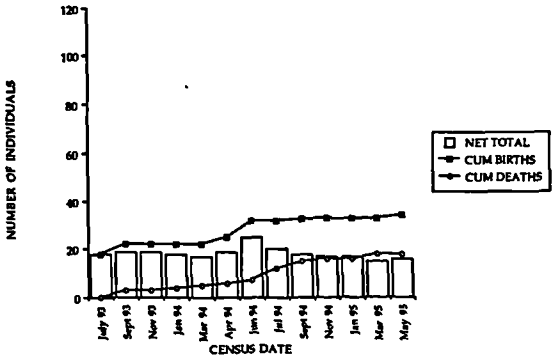
Teucrium scorodonia BLOCK 2



Silene dioica BLOCK 3



Teucrium scorodonia BLOCK 3



from June 1994 onwards, only falling slightly in July 1995. Three species, *Hypericum pulchrum* , *Silene dioica* and *Stachys sylvatica* had the lowest recruitment levels, and hence net population size, of all the introduced species. Recruitment for *Hypericum pulchrum* remained constant throughout the monitoring period, while for *Silene dioica* and *Stachys sylvatica* recruitment levels reached a maximum in July 1993/April 1994, and June 1994, respectively.

Mortality also varied greatly between species. In *Geranium robertianum, Hypericum pulchrum, Silene dioica* and *Stachys sylvatica,* mortality tracked recruitment quite closely; *Geranium robertianum* had the highest mortality in the winter of 1993/94; *Hypericum pulchrum* demonstrated relatively consistent mortality throughout the monitoring period; *Silene dioica* experienced a peak mortality in late summer 1993, and further losses from spring 1994 onwards; and *Stachys sylvatica* experienced consistent mortality from the winter of 1994/95 onwards. For all other species (except *Hyacinthoides non-scripta*) mortality was reasonably consistent throughout the monitoring period. No distinct seasonal change in mortality was observed for any species except *Hyacinthoides non-scripta,* which can only be monitored during the spring and early summer, and hence showed an unusual pattern of both recruitment and mortality.

Where population flux was compared between the two fertiliser treatments (in a total area of 3 m^2 for each fertiliser treatment) (Figure 3. 4), some differences occured. For all seven of the introduced species, net populations were at some time during the monitoring period higher in the F1 (control) fertiliser treatment than those in the F3 fertiliser treatment; for *Galium mollugo, Geum urbanum, Primula vulgaris, Prunella vulgaris* and *Teucrium scorodonia* this difference was pronounced, and is seen at all census dates; for *Geranium robertianum* , the net population was initially higher in the F1 fertiliser treatment, but became slightly lower than the net population in the F3 fertiliser treatment towards the end of the monitoring period; for *Hyacinthoides non-scripta* the higher net population in the F1 fertiliser treatment was only marginal at each census date. Patterns of relative recruitment and mortality were broadly similar to those in Figures 3. 3, and showed little variation between fertiliser treatments.

Comparisons of population flux between blocks for six of the sown species also showed some differences; four species, *Geranium robertianum, Geum urbanum, Hyacinthoides non-scripta* and *Prunella vulgaris,* showed consistently higher net population sizes in block 1; *Galium mollugo* had a higher population in both blocks 1 and 4, and *Teucrium scorodonia* in blocks 2 and 3. Patterns of relative recruitment and mortality were similar to those seen for each species in Figure 3. 3, though relative mortality for some species varied between blocks; for *Geranium robertianum, Geum urbanum, Hyacinthoides non-scripta* and *Prunella vulgaris,* levels of relative mortality were lowest in block 1; for *Galium mollugo* block 3, and for *Teucrium scorodonia* blocks 2 and 3.

GILLARS GREEN WOOD

Pooled data from all sampling areas within the F1 and F3 fertiliser treatments at Gillars Green (a total area of 13.5 m^2) show that *Hyacinthoides non-scripta* had the highest net population size of all sown species (Figure 3. 6), with a maximum in March 1994 of 160 individuals. In contrast, *Geum urbanum, Silene dioica* and *Teucrium scorodonia* had a relatively low net population size throughout the monitoring period, all showing a maximum population size of less than 80 individuals (within the total area of 13.5 m^2) in the spring and/or summer of 1994. The *Geum urbanum* population was characterised by high recruitment in the early spring of 1994, and an equally high mortality during the following summer, with a further slow increase in mortality and minimal recruitment for the rest of the monitoring period. This population fell to below 20 individuals by May 1995. *Hyacinthoides non-scripta, Silene dioica* and *Teucrium scorodonia* all showed peaks of recruitment in spring 1994, and lower relative mortality throughout the monitoring period. The highest absolute mortality occured in spring 1994/95 in the population of *Hyacinthoides non-scripta,* and in early summer 1994 in the populations of *Silene dioica* and *Teucrium scorodonia.* Population size for these three species remained above 20 individuals (within the total area of 13.5 m^2) up until May 1995.

In relation to fertiliser treatment (a total area of 6.75 m^2 for each fertiliser treatment), *Geum urbanum* and *Silene dioica* had a noticeably smaller net population size in the F1 fertiliser treatment, with low recruitment being matched by that of mortality. In the F1 fertiliser treatment, the small *Geum urbanum* population became extinct by July 1994, the *Silene dioica* population nearly so, being represented by only 2 individuals in May 1995. Net population size for *Hyacinthoides non-scripta* was greater with zero fertiliser; though patterns of relative recruitment and mortality were similar for the two treatments, relative recruitment was much higher in the F1 fertiliser treatment. *Teucrium scorodonia* showed no difference in either net population size or patterns of recruitment and mortality between fertiliser treatments.

Of the species previously considered, only *Geum urbanum* showed any clear difference in net population size between blocks; in block 2 the population nearly became extinct in July 1994 as a result of high relative mortality, though no clear cause for this mortality could be determined. No further recruitment was observed in this or any other block after this time. Patterns of recruitment and mortality were otherwise similar between all blocks for this and all other species. *Hyacinthoides non-scripta* showed little difference in net population size between blocks, though block 3 had a few more individuals at each census date. In populations of *Silene dioica* , relative mortality was lowest in block 2, the net population size being slightly larger than in the other two blocks.

3. 3. 2. 3 Population density and total numbers of individuals

BOLD MOSS TIP

Plant population densities in relation to fertiliser treatment and block are shown in Tables 3. 4 and 3. 5 respectively. Values are given for two census dates in the late spring/early summer of each year (1994/95), at a time when the population densities of all sown species were at their highest levels (in both years). This also allows for comparisons to be made between the two years, though, as there are no direct statistically significant effects of fertiliser treatment on population density (see below), these comparisons are of trends only.

In each year, *Hyacinthoides non-scripta* had the highest overall population densities, with >13 individuals m^{-2} in all fertiliser treatments. Differences between fertiliser treatments and census dates were slight and not statistically significant. Populations of *Galium mollugo, Geum urbanum, Prunella vulgaris* and *Stachys sylvatica* showed a progressive decrease in population density across fertiliser treatments, with the highest mean population density in the F1 treatment, and the lowest in the F3 treatment. Of these species, *Geum urbanum* showed variation in population density between census dates, with slightly fewer individuals being recorded in May 1995 in all fertiliser treatments. The *Stachys sylvatica* population had a higher mean density in June 1994 in the F1 treatment, and slightly higher values in May 1995 in the F2 and F3 treatments. The population density of *Geranium robertianum* was at its greatest in the F2 fertiliser treatment in June 1994, but by May 1995 there were few individuals present in each of the fertiliser treatments. The population densities of *Hypericum pulchrum* and *Teucrium scorodonia* were higher in the F1 fertiliser treatment, but with little difference in population density in the F2 and F3 treatments. The population density of *Silene dioica* was consistently low (<2 individuals m^{-2} by May 1995) in all fertiliser treatments. *Scrophularia nodosa* and *Digitalis purpurea* population densities were the lowest of all sown species, with overall mean values of <1 and <2 individuals m^{-2} respectively. For both species, population density was so low that no meaningful comparison of fertiliser treatments was possible.

Fertiliser treatment	F1		F2		F3	
Census date	**4.94**	**3.95**	**4.94**	**3.95**	**4.94**	**3.95**
Hyacinthoides non-scripta	18.75 ± 4.70	16.5 ± 4.80	13.75 ± 2.25	17.50 ± 3.50	16.25 ± 2.54	18.0 ± 3.37
Census date	**6.94**	**5.95**	**6.94**	**5.95**	**6.94**	**5.95**
Digitalis purpurea	-	-	0.5 ± 0.34	0.5 ± 0.34	1.5 ± 0.62	1.5 ± 0.62
Galium mollugo	10.25 ± 2.56	11.0 ± 2.59	9.75 ± 3.08	9.25 ± 2.63	5.50 ± 1.96	5.25 ± 2.06
Geranium robertianum	3.25 ± 1.22	1.50 ± 0.62	7.0 ± 2.05	2.75 ± 0.95	2.50 ± 1.26	1.75 ± 0.89
Geum urbanum	18.5 ± 4.63	12.5 ± 2.94	13.75 ± 5.15	12.0 ± 5.43	6.75 ± 2.21	4.25 ± 2.29
Hypericum pulchrum	2.25 ± 1.04	2.50 ± 0.96	1.25 ± 0.79	0.5 ± 0.5	1.25 ± 0.60	0.75 ± 0.40
Primula vulgaris	7.50 ± 3.50	8.50 ± 3.03	6.50 ± 3.18	8.75 ± 2.64	3.0 ± 1.57	5.0 ± 2.11
Prunella vulgaris	9.25 ± 3.96	10.25 ± 3.37	7.75 ± 2.77	7.0 ± 1.98	2.0 ± 0.82	2.75 ± 1.01
Scrophularia nodosa	0.25 ± 0.25	-	0.5 ± 0.5	0.25 ± 0.25	0.25 ± 0.25	0.5 ± 0.5
Silene dioica	2.25 ± 0.96	1.0 ± 0.45	1.75 ± 1.26	0.75 ± 0.40	1.75 ± 0.81	1.5 ± 0.81
Stachys sylvatica	6.0 ± 4.99	3.0 ± 2.24	0.5 ± 0.34	2.0 ± 1.03	-	0.25 ± 0.25
Teucrium scorodonia	9.75 ± 2.50	9.75 ± 2.58	5.75 ± 2.37	4.25 ± 1.57	7.0 ± 2.14	5.5 ± 2.09

Table 3. 4 Mean number of plants per square metre (with sample standard errors) in each fertiliser treatment, Bold Moss Tip, St. Helens. N.B. Where F1= zero, F2= low and F3= high fertiliser application.

Block number	1		2		3		4	
Census date	4.94	3.95	4.94	3.95	4.94	3.95	4.94	3.95
Hyacinthoides non-scripta	21.67 ± 6.06	26.67 ± 5.42	17.67 ± 3.25	18.33 ± 3.86	12.33 ± 1.87	10.33 ± 2.89	13.33 ± 2.62	14.0 ± 4.33
Census date	6.94	5.95	6.94	5.95	6.94	5.95	6.94	5.95
Digitalis purpurea	-	-	1.67 ± 0.59	2.0 ± 0.78	-	-	1.0 ± 0.72	0.67 ± 0.45
Galium mollugo	7.33 ± 3.07	5.33 ± 2.11	9.67 ± 4.10	7.67 ± 3.73	7.67 ± 2.48	9.67 ± 2.28	9.33 ± 2.22	11.33 ± 3.03
Geranium robertianum	6.0 ± 1.87	3.33 ± 1.19	5.0 ± 2.47	1.0 ± 0.72	3.67 ± 1.81	2.33 ± 1.15	2.33 ± 1.04	1.33 ± 0.57
Geum urbanum	17.33 ± 5.49	15.67 ± 3.86	17.33 ± 6.63	11.0 ± 7.22	9.0 ± 4.24	5.0 ± 2.32	8.33 ± 2.38	6.67 ± 2.33
Hypericum pulchrum	0.67 ± 0.45	-	2.67 ± 1.14	1.33 ± 0.75	2.0 ± 1.35	2.33 ± 1.15	1.0 ± 0.52	1.33 ± 0.75
Primula vulgaris	12.67 ± 4.40	13.67 ± 3.92	8.33 ± 4.16	7.67 ± 3.50	0.33 ± 0.33	3.0 ± 1.22	1.33 ± 0.57	5.33 ± 1.93
Prunella vulgaris	8.67 ± 4.85	8.33 ± 4.04	11.67 ± 3.46	5.67 ± 2.17	5.0 ± 2.66	8.33 ± 3.13	1.67 ± 0.77	6.33 ± 1.87
Scrophularia nodosa	0.33 ± 0.33	-	1.0 ± 0.72	1.0 ± 0.72	-	-	-	-
Silene dioica	3.67 ± 1.43	1.67 ± 1.04	3.67 ± 1.59	2.0 ± 0.60	-	0.67 ± 0.45	0.33 ± 0.33	-
Stachys sylvatica	8.33 ± 6.60	4.33 ± 2.93	0.33 ± 0.33	1.33 ± 1.33	-	1.0 ± 0.52	-	0.33 ± 0.33
Teucrium scorodonia	6.0 ± 2.39	5.0 ± 2.56	13.0 ± 3.34	10.0 ± 2.81	8.33 ± 2.44	8.0 ± 2.70	2.67 ± 1.73	3.0 ± 1.49

Table 3. 5 Mean number of plants per square metre (with sample standard errors) in each block, Bold Moss Tip, St. Helens.

The population density of *Primula vulgaris* was, at both census dates, lowest in the F3 treatment, with slightly higher densities being recorded in all fertiliser treatments in May 1995.

When the mean population density per species was examined in relation to block, some obvious trends occured. The population densities of *Hyacinthoides non-scripta, Geum urbanum, Geranium robertianum, Stachys sylvatica* and *Primula vulgaris* were greatest in blocks 1 and 2; in the case of *Stachys sylvatica* and *Primula vulgaris,* block 1 densities were especially high. In contrast, *Galium mollugo* density was lowest in block 1, the values slowly increasing from block 2 through to 4. The mean population density of *Silene dioica* tended to be greater on both census dates in blocks 1 and 2, as was the overall population density value for *Prunella vulgaris.* Population densities of *Scrophularia nodosa* and *Digitalis purpurea* were consistently low in all blocks, particularly blocks 1, 3 and 4. Though the population density of *Hypericum pulchrum* was much lower than that of *Teucrium scorodonia,* both species tended to have higher densities in blocks 2 and 3.

Probability values for the pooled total of individuals recorded from all four sampling areas in each plot, in relation to both fertiliser treatment and block, are shown for eleven species in Table 3. 6. No analysis was possible for *Scrophularia nodosa* due to insufficient data. Fertiliser treatment had no statistically significant effect on the numbers of individuals of all eleven species. However, numbers of *Digitalis purpurea* showed statistically significant differences between blocks, while numbers of *Galium mollugo, Geum urbanum, Primula vulgaris, Silene dioica* and *Prunella vulgaris* showed statistically significant block/time interaction effects.

GILLARS GREEN WOOD

Mean plant population densities for each species in relation to fertiliser treatment and block are shown in Tables 3. 7 and 3. 8 respectively. Figures are given for data collected on the same census dates as at the Bold site.

Galium mollugo, Geranium robertianum and *Stachys sylvatica* failed to become established at the site. *Hyacinthoides non-scripta* had the highest population density of all sown species, with values in the F1 and F2 fertiliser treatments being noticeably higher than in the F3 fertiliser treatment. The mean population densities of *Scrophularia nodosa, Primula vulgaris, Prunella vulgaris* and *Hypericum pulchrum* were very low in June 1994, and all species had died out by May 1995. The population densities of *Geum urbanum* and *Silene dioica* were greater in the F3 fertiliser treatment on both census dates,

Table 3. 6 Probability values for repeated measures analysis of variance, comparing the pooled number of plants in all sampling areas in each plot with fertiliser treatment and block, Bold Moss Tip, St. Helens. (* denotes P<0.05, ** P<0.01, *** P<0.001).

	SOURCE OF VARIANCE			
SPECIES	**Block**	**Fertiliser**	**Block*Time**	**Fertiliser* Time**
Digitalis purpurea	0.028*	0.115	0.692	0.965
Galium mollugo	0.865	0.485	0.0001***	0.553
Geranium robertianum	0.680	0.479	0.597	0.313
Geum urbanum	0.789	0.264	0.022*	0.158
Hyacinthoides non-scripta	0.082	0.613	0.146	0.725
Hypericum pulchrum	0.355	0.630	0.374	0.990
Primula vulgaris	0.141	0.313	0.048*	0.781
Prunella vulgaris	0.999	0.460	0.044*	0.928
Scrophularia nodosa	-	-	-	-
Silene dioica	0.090	0.551	0.008**	0.920
Stachys sylvatica	0.413	0.400	0.289	0.223
Teucrium scorodonia	0.151	0.415	0.237	0.937

Fertiliser treatment	F1		F2		F3	
Census date	4.94	3.95	4.94	3.95	4.94	3.95
Hyacinthoides non-scripta	11.56 ± 2.00	10.07 ± 2.35	10.52 ± 1.71	9.19 ± 1.78	6.22 ± 0.99	4.59 ± 0.85
Census date	6.94	5.95	6.94	5.95	6.94	5.95
Galium mollugo	-	-	-	-	-	-
Geranium robertianum	-	-	-	-	-	-
Geum urbanum	1.19 ± 0.76	-	1.04 ± 0.34	0.15 ± 0.15	5.04 ± 1.04	1.33 ± 0.64
Hypericum pulchrum	0.44 ± 0.25	-	0.59 ± 0.28	-	1.93 ± 0.69	-
Primula vulgaris	-	-	0.15 ± 0.15	-	0.44 ± 0.25	-
Prunella vulgaris	0.15 ± 0.15	-	0.15 ± 0.15	-	0.74 ± 0.37	-
Scrophularia nodosa	-	-	0.30 ± 0.21	-	0.74 ± 0.30	-
Silene dioica	0.44 ± 0.33	0.30 ± 0.21	2.52 ± 0.80	2.67 ± 0.83	4.44 ± 1.00	3.70 ± 0.93
Stachys sylvatica	-	-	-	-	-	-
Teucrium scorodonia	4.44 ± 0.89	3.11 ± 0.72	7.56 ± 1.82	5.33 ± 1.15	4.89 ± 0.96	2.67 ± 0.71

Table 3. 7 Mean number of plants per square metre (with sample standard errors) in each fertiliser treatment, Gillars Green Wood, St. Helens. N.B. Where **F1**= zero, **F2**= low and **F3**= high fertiliser application.

Block number	**1**		**2**		**3**	
Census date	**4.94**	**3.95**	**4.94**	**3.95**	**4.94**	**3.95**
Hyacinthoides non-scripta	7.26 ± 1.13	7.56 ± 1.47	8.89 ± 1.50	6.67 ± 1.46	12.15 ± 2.12	9.63 ± 2.37
Census date	**6.94**	**5.95**	**6.94**	**5.95**	**6.94**	**5.95**
Galium mollugo	-	-	-	-	-	-
Geranium robertianum	-	-	-	-	-	-
Geum urbanum	2.52 ± 0.86	0.74 ± 0.48	2.52 ± 0.77	0.15 ± 0.15	2.22 ± 0.91	0.59 ± 0.46
Hypericum pulchrum	1.33 ± 0.64	-	1.19 ± 0.42	-	0.44 ± 0.25	-
Primula vulgaris	0.30 ± 0.21	-	0.15 ± 0.15	-	0.15 ± 0.15	-
Prunella vulgaris	0.44 ± 0.33	-	0.44 ± 0.25	-	0.15 ± 0.15	-
Scrophularia nodosa	0.15 ± 0.15	-	0.30 ± 0.21	-	0.59 ± 0.28	-
Silene dioica	2.37 ± 0.89	2.37 ± 0.83	3.56 ± 0.96	2.96 ± 0.87	1.48 ± 0.53	1.33 ± 0.56
Stachys sylvatica	-	-	-	-	-	-
Teucrium scorodonia	4.59 ± 1.08	3.26 ± 0.91	8.00 ± 1.80	4.59 ± 1.14	4.30 ± 0.71	3.26 ± 0.57

Table 3. 8 Mean number of plants per square metre (with sample standard errors) in each block, Gillars Green Wood, St. Helens.

while the population density of *Teucrium scorodonia* was higher, also on both census dates, in the F2 fertiliser treatment.

Plate 3. 3 *Teucrium scorodonia* at Gillars Green Wood, St. Helens, summer 1995.

Few obvious differences in species population density between blocks were evident. Only the values for *Hyacinthoides non-scripta* and *Teucrium scorodonia* showed any influence, there being higher densities for each species in blocks 3 and 2 respectively. These differences were seen at both census dates. The population density of *Geum urbanum* was similar at both census dates in all blocks.

The probability values for the pooled total of individuals recorded from all three sampling areas in each sub-plot, in relation to both fertiliser treatment and block, are shown in Table 3. 9. Fertiliser treatments had a statistically significant effect on the total numbers of *Geum urbanum* and *Silene dioica,* increased fertiliser inputs resulting in higher numbers. Statistically significant treatment/time interaction effects were also seen for the same two species. No significant block effects were seen, though *Teucrium scorodonia* showed statistically significant block/time interaction effects.

Table 3. 9 Probability values for repeated measures analysis of variance, comparing the pooled number of plants in all sampling areas in each sub-plot with fertiliser treatment and block, Gillars Green Wood, St. Helens. (* denotes P<0.05, ** P<0.01, *** P<0.001).

	SOURCE OF VARIANCE			
SPECIES	**Block**	**Fertiliser**	**Block*Time**	**Fertiliser* Time**
Geum urbanum	0.630	0.005**	0.238	0.0001***
Hyacinthoides non-scripta	0.773	0.152	0.463	0.302
Silene dioica	0.385	0.028*	0.215	0.0001***
Teucrium scorodonia	0.660	0.384	0.001***	0.947

3. 3. 3 The effect of mineral nutrient addition on flowering and seed production in plants established from seed

The data for proportions of flowering plants at Bold Moss, expressed as the proportion of individuals that flowered in each sampling area, averaged over all fertiliser treatments and blocks, are shown for July in the two census years (1994/1995) in Table 3. 10. In both 1994 and 1995, only data collected in July were sufficient to allow some broad comparisons. Few sown species flowered during this month in either year. Of those that did, *Prunella vulgaris* and *Geranium robertianum* were the most numerous in both years, the proportion of flowering individuals rising by a factor of c. 2 in both these species between the two census years. *Galium mollugo* also showed an increase in the proportion of flowering individuals, though the rise for this species was by a factor of c. 10. *Geum urbanum* did not flower until 1995, while the figure for *Digitalis purpurea* represents a single individual flowering in July 1994. By spring 1996 *Primula vulgaris* had flowered in most plots, though no data were collected at this time.

Plate 3. 4 *Primula vulgaris* flowering at Bold Moss, spring 1996.

Differences in the mean proportion of flowering individuals in relation to fertiliser treatment and block in July 1995 are shown for four species in Figure 3. 9. Only *Geranium*

Table 3. 10 Proportion of flowering plants at Bold Moss Tip, St. Helens, expressed as the proportion of individuals that flowered in each sampling area, averaged over all fertiliser treatments and blocks, July 1994 and July 1995.

SPECIES/YEAR	**1994**	**1995**
Digitalis purpurea	0.042	0
(±SE)	(±0.042)	-
Galium mollugo	0.003	0.031
(±SE)	(±0.003)	(±0.014)
Geranium robertianum	0.035	0.082
(±SE)	(±0.018)	(±0.020)
Geum urbanum	0	0.020
(±SE)	-	(±0.012)
Prunella vulgaris	0.076	0.142
(±SE)	(±0.047)	(±0.040)

Figure 3. 9 Mean proportion of flowering individuals in a population in relation to fertiliser treatment and block (with sample standard errors), Bold Moss Tip, St. Helens, 28 July 1995. N.B. Where F1= zero, F2= low and F3= high fertiliser application.

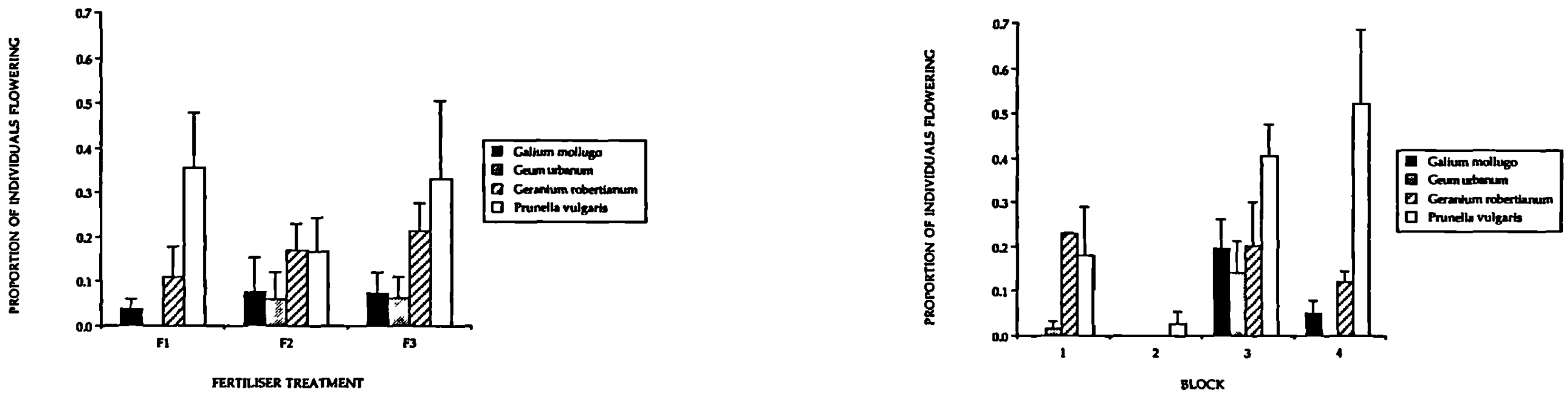
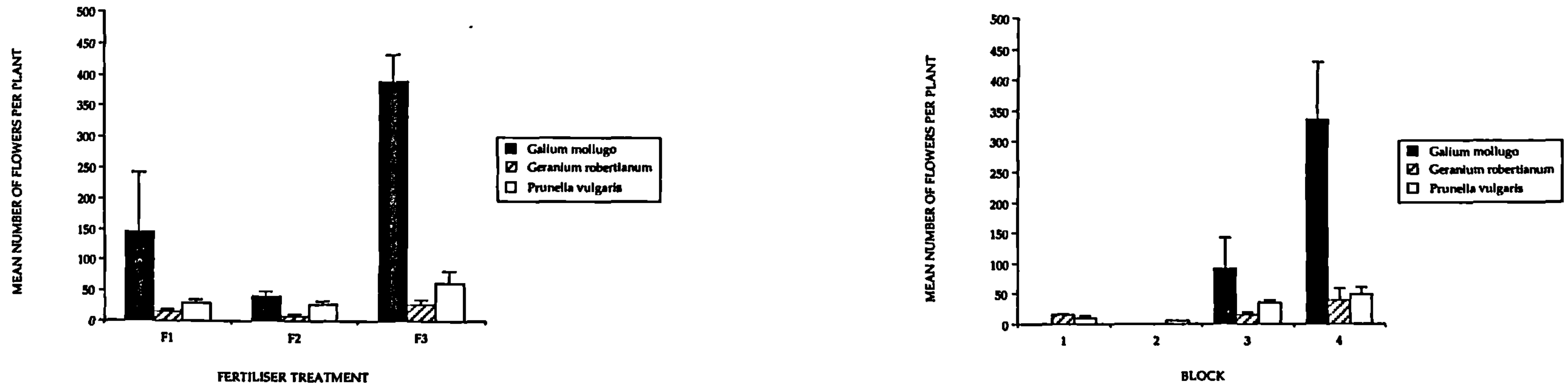


Figure 3. 10 Mean number of flowers per plant in relation to fertiliser treatment and block (with sample standard errors), Bold Moss Tip, St. Helens, 28 July 1995. N.B. Where F1= zero, F2= low and F3= high fertiliser application.



robertianum showed a progressive increase in the proportion of flowering individuals from the F1 to F3 fertiliser treatments. *Geum urbanum* was only found to flower in the F2 and F3 treatments, while the proportion for *Galium mollugo* was higher at the same two levels. *Prunella vulgaris* appeared to flower equally readily in all three fertiliser treatments. In relation to block, *Galium mollugo* only flowered in blocks 3 and 4. The highest proportion of flowering individuals for *Galium mollugo* and *Geum urbanum* was seen in block 3. Only *Prunella vulgaris* flowered in block 2, though the highest proportions for this species were seen in blocks 3 and 4. Figures for *Geranium robertianum* are similar for blocks 1, 3 and 4. Data for *Digitalis purpurea* could not be plotted due to insufficient numbers.

The mean number of flowers per plant in relation to fertiliser treatment and block, in July 1995, are shown for three species (*Galium mollugo, Geranium robertianum* and *Prunella vulgaris*) in Figure 3. 10. In each case, the mean number of flowers was highest in the F3 fertiliser treatment, though for *Geranium robertianum* and *Prunella vulgaris* this difference was only slight. The mean number of flowers per plant for all three species was highest in blocks 3 and 4, with the mean number for *Galium mollugo* being particularly high in block 4.

In both years seed was collected in any quantity only from plants of *Geranium robertianum* and *Prunella vulgaris*. The data for the number of seeds per fruit and the proportion of those that subsequently germinated were too small to allow any meaningful comparisons, though there were no obvious differences in relation to fertiliser treatment or block.

3. 4 Discussion

3. 4. 1 Population flux

Though all plant populations are dynamic over time, coexistence between plant species is a common feature of the majority of natural plant communities. Much of this coexistence is thought to arise from resource partitioning and differences in niche dimensions between species (Silvertown and Lovett Doust, 1993). In relation to nutrient supply, Tilman's resource ratio hypothesis (Tilman, 1982) states that a number of species can coexist on the same essential resources, as long as each species has different critical resource ratio requirements, and that the resources are distributed heterogeneously within a habitat (Silvertown and Lovett Doust, 1993). On industrial waste sites, deliberate species introductions have shown that communities are not saturated with species and that suitably adapted species can colonise and become permanent components of a community.

Nutrient availability also can be a major factor controlling plant colonisation (Ash *et al.*, 1994), and hence long-term community structure. Colliery spoil is severely deficient in two of the most important plant nutrients (N and P), and it was therefore expected that additions of these nutrients would lead to an increase in seedling establishment and adult survival for at least some of the species introduced to the two experimental sites.

For the species introduced at Bold Moss, population flux graphs in the two fertiliser treatments showed little difference in patterns of relative recruitment and mortality from those plotted for all individuals collectively. Overall, net population sizes varied greatly between species. Some of these differences may be related to variations in relative germinability between the sown species, though as the data in Table 3. 1 indicate this relationship is far from straightforward. Some species such as *Geum urbanum* and *Hyacinthoides non-scripta* only germinated to any extent in the field, where adequate levels of stratification could be attained. Others including *Digitalis purpurea* and *Hypericum pulchrum* germinated well in the laboratory but only formed a low net population size in the field. Successful germination of otherwise viable seed is known to be related to a number of factors including the length of time seed remains buried in the soil (Watkinson, 1986), pressure of seed predation, specific micro-environmental requirements (Silvertown and Lovett Doust, 1993), and seed orientation (Bosy and Aarssen, 1995). As such, the relatively low net populations of *Digitalis purpurea* and *Hypericum pulchrum* at Bold Moss may have arisen from enforced seed dormancy, a lack of available 'safe sites' (*sensu* Harper, 1977), or a combination of both these factors. Amounts of relative mortality were greatest for those species with the lowest net population size during the monitoring period (*Geranium robertianum, Hypericum pulchrum, Silene dioica* and *Stachys sylvatica,* see Figure 3. 3), with populations nearing extinction by July 1995. For these species, recruitment was tracked by relatively high mortality throughout the monitoring period. All of this evidence points to the possibility that the seed application rate of 100 seeds m^{-2}, used for all the experimental species at Bold Moss and Gillars Green, was too low to achieve adequate levels of establishment for some of the introduced species.

When relative recruitment and mortality were examined in relation to fertiliser treatment, variations between populations in the two fertiliser treatments (F1 and F3) were demonstrated. *Galium mollugo, Geum urbanum, Hyacinthoides non-scripta, Primula vulgaris, Prunella vulgaris* and *Teucrium scorodonia* were are all characterised by consistently higher net populations in the F1 treatment. Only *Geranium robertianum* had a greater population size in the F3 treatment, during the spring and summer of 1995, though this difference was very small. It was not possible to demonstrate a statistically significant effect of fertiliser treatment with a repeated measures analysis of variance for this or any other species at Bold Moss.

The response of species populations seems to be the converse of what would be expected at Bold Moss. Whilst plant population growth is regulated by the availability of a much wider range of resources (Watkinson, 1986), it seems unlikely that the distribution of any 'beneficial' resource coincides only with those plots that received the F1 fertiliser treatment, and that the influence of this resource is equally great with all sown species. This implies that the differences between populations for the majority of species are in some way related to the experimental fertiliser treatments. One hypothesis is that the addition of fertiliser in F3 treatment plots led to an increase in interspecific competition between herbaceous/grass species already established in these plots (which by the summer of 1993 had begun to grow again following glyphosate treatment earlier in the year) and herbaceous plants established from introduced seed. This hypothesis is not however supported by data in the population flux graphs (Figure 3. 4). In the case of species such as *Galium mollugo, Geranium robertianum* and *Prunella vulgaris,* net population size was generally lower in the F3 fertiliser treatment from the first census date onwards, at a time when the abundance of the existing field layer vegetation was relatively low in all plots. Also, patterns of population flux in the F1 and F3 fertiliser treatments tend to be similar throughout the study period, which would not be expected if plants established from introduced seed were subject to increasing levels of interspecific competition from summer 1993 onwards. The effects of plant competition on the introduced herbaceous species are examined in more detail in Chapter 4.

It must therefore be assumed that other environmental factors are exerting a consistent influence in plots with the same experimental fertiliser treatment, and that these effects are sufficient to cause differences in the population flux for the majority of introduced species. Of those spoil characteristics that were measured, the values for pH and P and Ca concentration offer a source of possible explanation; in three out of four pairs of plots, pH and P and Ca concentrations were higher in the F1 fertiliser treatment in comparison with F3; in the remaining two plots the difference in values was minimal. As much of the P in the spoil will be unavailable to plants, it seems likely that variation in spoil pH between plots is in some way associated with the net population size recorded at any one census date. Low substrate pH is known to inhibit the germination of some seeds, different species having different critical values below which germination may be severely limited. With *Primula vulgaris* for example, values below pH 4.5 have been shown to significantly reduce seed germination and subsequent plant establishment (Helliwell, 1980). Poor growth of *Primula vulgaris* has also been related to soil pH values of less than 4.7 (Helliwell, 1980). Since low pH is often associated with toxic levels of metal ions, seedling establishment for many of the introduced species may also have been reduced, and it is likely that seedlings may have emerged and subsequently died between census dates, and therefore went unrecorded. In support of this hypothesis it is interesting to note that of all sown species, *Hyacinthoides non-scripta* showed the least difference in population flux characteristics in the F1 and F3 fertiliser treatments (and hence at both the relatively high and low pH ranges); throughout its natural range in Britain, this species is more frequent and abundant on soils of low mineral nutrient status and relatively low pH (pH 3.5 - 4.5) (Grabham and Packham, 1983; Grime *et al.*, 1988), conditions very similar to those found at the Bold Moss site.

If net population size is to a large degree unrelated to the addition of major plant nutrients to colliery spoil at Bold Moss, variations in species population flux in relation to block may provide further evidence to relate spoil characteristics to the relative success and failure of establishment of the introduced species. Though patterns of relative recruitment and mortality within each block were similar for all species, population size between blocks was seen to differ for most of the introduced species. For all species except *Teucrium scorodonia* net population size was consistently higher in block 1; the mean pH for the three plots in this block was the highest for all blocks (pH 5.4), the mean values for plots in other blocks being pH 4.5 or less (pH 3.5 for plots in block 2, pH 4.5 for plots in blocks 3 and 4). Since the mean extractable P content of the spoil prior to fertiliser application for plots within blocks, was higher in blocks 3 and 4, it seems unlikely that this factor had any influence on net population size. Reasons for differences in net population size between blocks 2, 3 and 4 for the introduced species are more difficult to determine, as there appears to be little relationship between population size for each species and spoil chemical characteristics between blocks. It is likely that the observed differences in species population size are derived from variations in a range of other environmental factors (e. g. slope, aspect, levels of interspecific competition and light climate), each of which may act as the major determinant of net population size in any one plot and/or block. These factors may be particularly important for *Teucrium scorodonia* and *Hyacinthoides non-scripta*, the latter species having the highest net population size in a block (block 1) with a mean pH value of 5.4, well above the optimal range for this species (Grime *et al.*, 1988).

At Gillars Green, populations of *Hyacinthoides non-scripta*, *Silene dioica* and *Teucrium scorodonia* showed similar temporal patterns of recruitment and mortality to the populations at Bold Moss. Differences in the net population size for the introduced species (and for *Hyacinthoides non-scripta* in particular) between the two experimental sites were partly related to the larger number of sampling areas used for data collection at Gillars Green. The temporal pattern of population flux for *Geum urbanum* at Gillars Green showed some differences to that at Bold Moss, with relatively low levels of recruitment up until April 1994, and subsequently higher relative mortality. It is a species known to germinate most readily in spring however, and has been described as a 'semi-shade' species (Grime *et al.*, 1988). Since canopy cover is relatively dense at Gillars Green this may have reduced the survival of *Geum urbanum* seedlings, as observed from July 1994 onwards. Leaf expansion and subsequent canopy closure is likely to have been an important factor in the failure of many of the introduced plant species to become established on the site; even the recruitment of *Primula vulgaris* (which is classified as a shade-tolerant species) has been shown to be significantly reduced following closure of canopy gaps (Valverde and Silvertown, 1995). The extremely low pH values for all plots and blocks at Gillars Green (<pH 3.7) are also likely to have influenced the germination and establishment of many of the introduced species. For example, *Geranium robertianum* and *Prunella vulgaris* are known to be absent from soils of <pH 4.0, while *Hyacinthoides non-scripta* is particularly associated with soils of low pH (<4.5) (Grime *et al.*, 1988).

At Gillars Green, for two (*Geum urbanum* and *Silene dioica*) of the four introduced species for which population flux graphs were plotted, fertiliser application appeared to increase net population size. Both *Geum urbanum* and *Silene dioica* had a greater net population size in the F3 fertiliser treatment, while in the absence of fertiliser the *Geum urbanum* population became extinct by July 1994. For these species it appears that low levels of N and P may act as major limiting resources (*sensu* Fitter, 1986). Net population size for *Teucrium scorodonia* showed no obvious difference between fertiliser treatments, while the net population size of *Hyacinthoides non-scripta* showed a reversed trend to that seen for *Geum urbanum* and *Silene dioica*, with a higher net population size in the F1 fertiliser treatment. For both *Hyacinthoides non-scripta* and *Teucrium scorodonia* it is likely that (as at Bold Moss) a range of environmental factors are important determinants of plant establishment and survival, and that the availability of major plant nutrients is not the main limitation to seedling establishment and plant growth at the Gillars Green site. Unlike Bold Moss, no significant differences were demonstrated for mean pH, P and Ca values in the two fertiliser treatments. In naturally-established populations of *Teucrium scorodonia*, ecotypic differentiation can be made between populations on acidic and calcareous soils, and between plants in shaded and unshaded habitats (Grime *et al.*, 1988). For this species it may therefore be possible that the seed used in the introduction experiments came from an 'unsuitable' source; as recruitment levels were similar in the two fertiliser treatments, a lack of response to fertiliser application may be related to unknown physiological characteristics in established plants. The relatively small difference in net population size between blocks for all four species follows no obvious pattern, and is likely to arise from a range of species-specific environmental requirements within each block.

3. 4. 2 Population density and total numbers of individuals

Data for plant population density from both study sites generally showed similar patterns to those seen for species' population flux. At Bold Moss densities for most species were greatest in the F1 fertiliser treatment, and only *Scrophularia nodosa* and *Digitalis purpurea* showed a very slight relative increase in plant population density in the F3 fertiliser treatment by May 1995. These two species occured at very low population densities however, and any assumption that densities were increased by the application of fertiliser at the highest treatment rate must be viewed with caution. It is just as likely that observed differences in plant population density for these and all other introduced species arose as a result of variations in environmental factors, both within and between individual plots. Whilst many species at Bold showed a reduction in plant population density in all fertiliser treatments from the 1994 to 1995 census dates (as might be expected following progressive mortality of established plants throughout this period), data for *Galium mollugo, Hyacinthoides non-scripta, Primula vulgaris* and *Prunella vulgaris* showed either little change or a slight increase from one year to the next. This was almost certainly a result of continued germination of introduced seed throughout mid to late 1994 and early 1995. When data for plant densities in each block were examined for the same four species however, absolute plant densities and relative increase in density were seen to be highly variable. In addition, plant densities for all of these four species showed a marked reduction from one year to the next in at least one block. Variation in plant population density for most of the sown species appeared to be more strongly associated with blocks than with fertiliser treatments. This is shown by the noticeably higher population densities of *Hyacinthoides non-scripta, Geum urbanum, Geranium robertianum, Stachys sylvatica* and *Primula vulgaris* in blocks 1 and 2, and *Galium mollugo* in blocks 3 and 4.

In comparison to some other species, the relatively low plant population densities observed for *Scrophularia nodosa, Digitalis purpurea* and *Hypericum pulchrum* at Bold Moss are unlikely to have resulted from low seed germination, since these species readily germinated in the laboratory. Seedlings of all species were identified from the first emergence of cotyledon leaves, and therefore it is unlikely that seedlings of these species emerged *en mass* and subsequently died without being recorded. Spoil nutrient status may have influenced the population density of established plants of *Scrophularia nodosa* and *Digitalis purpurea* (densities were greater in the F2 and F3 fertiliser treatments, while in 1995 plants of these two species were only recorded in plots with these treatments), but this does not explain the relatively low population density of these species, or of *Hypericum pulchrum*. Whilst both *Hypericum pulchrum* and *Digitalis purpurea* are characteristic of acidic soils (Stace, 1991; Grime *et al.*, 1988) their population densities remained low in all blocks. It therefore seems likely that factors other than spoil pH are acting to inhibit the germination of these species. *Scrophularia nodosa* is most characteristic of soils with a pH of c.6 (Grime *et al.*, 1988), and is often associated with woodland and scrub on calcareous mull soils overlying sedimentary limestones, shales and clays (Rodwell, 1991). It is therefore unsurprising that plant population densities for this species remained low. As with the other two species however, population density must also have been reduced by the inhibition of seed germination, possibly as a result of low spoil pH in many plots.

The probability values for the pooled total of individuals recorded from all four sampling areas in each plot at Bold Moss, in relation to fertiliser treatment and block, support the view that the population density of the introduced species was more likely to vary in relation to block rather than to the different fertiliser treatments. *Digitalis purpurea* was the only species for which this difference was statistically significant across all census dates; reference to the plant population density values in relation to block indicates that the population density of *Digitalis purpurea* was highest in block 2. This corresponds to the block in which pH values within plots were, on average, lower than in other blocks. Statistically significant differences between blocks for five other species (*Galium mollugo, Geum urbanum, Primula vulgaris, Silene dioica* and *Prunella vulgaris*) were also found, but only for certain census dates during the monitoring period (time/block interaction effects). These differences most probably arose from unequal recruitment and/or mortality between blocks over specific periods of time during the monitoring period.

At Gillars Green plant population densities for *Geum urbanum* and *Silene dioica* were highest in the F3 fertiliser treatment. A statistically significant difference in the pooled total of individuals recorded from all three sampling areas in each sub-plot at Gillars Green also occured for the same two species in relation to fertiliser treatment. These results support evidence obtained from the population flux graphs that the addition of mineral nutrients may lead to an increase in the number of established plants of *Geum urbanum* and *Silene dioica*, but only where variation in spoil pH is minimal. Where spoil pH is consistently low and germination is not inhibited, plants are more likely to benefit from the addition of major nutrients, as all individuals will be equally affected by the stresses associated with high concentrations of hydrogen (H^+) ions in the spoil. However, where pH values vary significantly from one location to another (in this case at the between sub-plot scale), different individual plants are likely to be affected by a variety of stresses to varying degrees of severity. In addition to pH values affecting soil structure, coherence, swelling, porosity, aeration and water-holding capacity, it is known that at low pH (<pH 4) soluble aluminium (Al^{3+}) has the ability to interfere with plant nutrient uptake, and that some species are more competitive than others for these resources in the presence of high concentrations of Al^{3+} (Crawley, 1986b). In contrast to the Gillars Green site, the greater variability in spoil pH at Bold Moss may explain why the addition of fertiliser did not influence plant population density in some of the introduced species. The failure of many of the introduced species to either germinate or become established at Gillars Green is probably related to enforced seed dormancy or a lack of safe sites. In addition to the extremely acidic nature of the spoil, canopy shading was high during the summer months in all plots, and leaf litter accumulation in some sub-plots was sufficiently high to probably inhibit the germination and/or growth of individuals of some species.

The pooled total of individuals of *Teucrium scorodonia* recorded from all three sampling areas in each sub-plot at Gillars Green was shown to be statistically significantly different between blocks, but only for certain census dates (time/block interaction effect). The data for plant population density indicate that in both 1994 and 1995 the population density of *Teucrium scorodonia* was greatest in block 2. As pH, P and Ca values did not vary to much extent between plots, it is likely that these differences arose from variations in other environmental factors within each block, which were not measured in this research.

3. 4. 3 Flowering and seed production

Reproductive maturity in flowering plants is only reached when sufficient resources and meristematic tissues have been accumulated by individual plants during prior periods of growth (Silvertown and Doust, 1993). As a result, the onset of flowering in many plant species is often related to plant size or biomass rather than to plant age, and may be viewed partly as a function of plant resource acquisition (Watkinson, 1986). Resource deprivation therefore has the potential to prevent individual plants from reaching their 'reproductive threshold' (*sensu* Fitter, 1986). Resource deprivation may arise from a variety of sources, including low light intensity, reduced soil moisture and a lack of available plant nutrients. Once beyond their reproductive threshold, patterns of flowering for individual plants or species may be significantly affected by a number of additional factors. For example, physiological stress due to summer drought can depress flowering in the following year in some perennial plant species (Inghe and Tamm, 1988). Resource limitation may also influence the proportion of fertilised ovules developing into seeds on individual plants (Howe and Westley, 1986; Watkinson, 1986). As this proportion is a component of plant yield (Silvertown and Lovett Doust, 1993), it will be dependent partly upon the relative availability of major plant nutrients.

Only five of the introduced species at Bold Moss flowered in sufficient numbers to allow for comparisons of reproductive output between fertiliser treatments. The nature of the data in both years precluded the use of statistical comparisons between treatments. Of the five species, *Geranium robertianum* and *Prunella vulgaris* had the highest proportion of flowering plants for the site as a whole. Established plants of both these species possess a number of characteristics associated with ruderal plants, and would therefore be expected to produce flowers relatively early in their life-history (Grime *et al.*, 1988). Both *Galium mollugo* and *Geum urbanum* produced a higher proportion of flowering individuals in 1995, probably due to the higher levels of productivity occurring at this time. The low population size of *Digitalis purpurea* within plots did not allow for any meaningful treatment comparisons for this species.

The mean proportion of flowering individuals in relation to fertiliser treatment showed interesting trends for some species. *Galium mollugo* and *Geranium robertianum* had a higher proportion of flowering individuals within fertilised plots (the proportion for *Geranium robertianum* being greatest in the F3 fertiliser treatment), while *Geum urbanum* flowered only in plots which had received fertiliser. These trends indicate that a lack of major plant nutrients at the Bold Moss site may limit flowering in these species. *Prunella vulgaris* showed no clear relationship between flowering and fertiliser application. Differences related to block were also evident, most particularly the absence or lack of flowering individuals in block 2. This difference was not related to lower population density for these four species within this block, because plant population densities were similar to, or even higher than, those for the same species in other blocks. Plants within this block appeared to be unable to reach a size sufficient to promote flowering, probably due to resource limitations imposed by spoil chemical characteristics and/or the competitive effects of existing herbaceous and woody vegetation. Conversely, block 3 appeared to be the only block in which these resource limitations were insufficient to prevent flowering in all four species. Higher proportions of flowering individuals for

Galium mollugo and *Prunella vulgaris* in blocks 3 and 4, and for *Geum urbanum* in block 3, are probably related to the lower levels of tree/shrub foliage in these plots (see Chapter 4), and a related increase in available light in the field layer.

Data for the mean number of flowers per plant in relation to fertiliser treatment and block showed that higher numbers occured for all three species (*Galium mollugo, Geranium robertianum* and *Prunella vulgaris*) in the F3 fertiliser treatment, indicating a possible link between flower number and plant nutrient availability. Differences were also noticeable between blocks however, the mean number of flowers for all three species being higher in blocks 3 and 4, as was found for the proportion of flowering individuals in different blocks. As previously argued, it is likely that the generally higher levels of available light in the field layer is an important determinant of reproductive performance in plants of these species.

3. 5 Conclusions

The pH of colliery spoil at Bold Moss was found to be highly variable, with acid and base-rich areas being found throughout the tip, the latter probably relating to pockets of pulverised fuel ash deposition. In contrast, all spoil at Gillars Green was found to be consistently highly acidic, which is characteristic of a majority of the colliery tips in the region. On both sites herbaceous woodland species were introduced as seed to the existing field layer, and plants of some species successfully established throughout the course of a two year monitoring period. A much wider range of species became established at Bold Moss however, and only *Hyacinthoides non-scripta* was recruited in any numbers at Gillars Green. At both sites, high concentrations of H^+ ions in the spoil appeared to be associated with a reduction in seed germination and relatively poor plant establishment, though the role played by a range of other environmental factors should also be considered. The results indicate that both a lack of seed immigrants, and the extreme physical and chemical properties of colliery spoil, are responsible for limiting colonisation by many herbaceous woodland species at the two study sites.

The addition of mineral nutrients to colliery spoil at both sites resulted in varying plant performance. At Bold Moss species' population size and density were found to be unrelated to the addition of major plant nutrients, though an increase in the proportion of flowering plants and number of flowers per plant *did* show some association. At Gillars Green however the population density of established plants of two species, *Geum urbanum* and *Silene dioica*, increased in response to the application of mineral nutrients. Where spoil pH is uniformly low some species may therefore benefit to a limited degree from the addition of major plant nutrients. Where spoil pH varies as much as at Bold Moss however, it is likely that areas of extreme H^+ ion concentration will continue to limit plant growth to the extent that only those plants in more base-rich areas will benefit substantially from the input of additional mineral nutrients.

As a method for enhancing/diversifying the herbaceous species component in the field layer of existing woodland and scrub vegetation on colliery spoil, seed introductions can therefore prove an effective means of establishing populations of some herbaceous plant species. Where the manipulation of spoil pH is undesirable or impracticable, care must be taken to ensure that species used in such introductions are suitably adapted to the often extreme range of chemical characteristics associated with colliery spoil, most particularly low pH. Though this may limit the number of species that can be introduced, the vegetation communities that are created will in the long-term be more 'naturalistic' and therefore require fewer management inputs. This will also reduce the cost of such diversification schemes and make them potentially more attractive to those considering community restoration measures. On substrates with higher base concentrations the range of species that may be utilised can be much greater, though all should ideally be representative of the flora indigenous to the region.

The addition of major plant nutrients should usually be considered an integral part of any introduction process on derelict industrial and urban land, as existing concentrations may be insufficient for adequate plant establishment, survival and reproductive output. Where introductions are made into existing vegetation it must however be assumed that as nutrient levels are already sufficient for the growth of a range of species, the success or failure of plants established from seed may be equally related to a range of other environmental factors.

4. 0 THE EFFECT OF VEGETATION AND LIGHT CLIMATE ON HERBACEOUS WOODLAND PLANT SPECIES INTRODUCED AS SEED INTO EXISTING VEGETATION ON COLLIERY SPOIL

4. 1 Introduction

Of the range of possible interactions between individual plants, competition is characterised by net costs to both parties. Differences in competitive intensity between plants, defined as ..."the degree to which a plant is reduced by the presence of neighbours" (Belcher, Keddy and Twolan-Strutt, 1995), are themselves a function of varying demands for, and responses to, a number of intermediate factors, including soil nutrient concentrations, light levels and the action of herbivores (Silvertown and Lovett Doust, 1993). Competition for resources may occur both below and above ground, and the relative influence of root or shoot competition on measures of plant performance tend to be related to various habitat characteristics. In grassland for example, where shade is minimal, shoot competition may be relatively unimportant in comparison to competition for soil resources (Belcher, *et al.*, 1995), though competitive intensity may be altered by increased nutrient supply; favouring species or individual plants with morphological attributes that confer higher potential growth rates (Berendse, 1994). In practice, continued competition is unlikely to lead to community equilibrium, as species coexistence within plant communities is generally characterised by continual fluctuations in species abundance at different temporal and spatial scales. Local disturbance events are thought to be particularly important in this process (Grime, 1979; Bengtsson, Fagerstrom and Rydin, 1994).

In woodland habitats shade is one of the most important determinants of field layer community characteristics, constituting a major form of stress that may severely limit plant performance (Packham and Cohn, 1990; Valverde and Silvertown, 1995). Individual species show a variety of adaptations to shading (Packham and Cohn, 1990) which may confer a competitive advantage over neighbouring plants. This may allow adequate growth throughout the summer months, when solar radiation in the field layer can fall to 1-5% of that available above the canopy (Hicks and Chabot, 1985). In addition to diffuse light, sunflecks are known to have significant effects on seed germination, photosynthesis, and microsite distributions following leaf expansion, and may provide in excess of 50% of the daily photon flux density (PFD; μmol m^{-2} s^{-1}) in the field layer of temperate woodlands (Chazdon, 1988).

In Chapter 3 increased spoil mineral nutrient availability was shown to have a limited effect on plant species establishment and survival on two colliery waste tips, Bold Moss Tip and Gillars Green Wood. Much of the observed variation in establishment and survival was thought to be related to spoil chemical characteristics, in particular the very low spoil pH at both sites. Since species were introduced by means of seeding into existing areas of vegetation, it is possible that establishment and survival was also affected by factors related to the composition and structure of the existing vegetation within experimental plots. This chapter examines two of these factors, presence of existing vegetation and light climate, in order to determine how significant they may be in effecting the numbers of individuals of the introduced species within experimental plots. The results are discussed with reference to the findings in Chapter 3, in order to provide a guide to woodland herbaceous species introductions on similar sites.

4. 2 Methods

Only the experimental plots established at the Bold Moss site were used for investigations into the hypothesised effects of existing vegetation and light climate, since these plots were characterised by a wide variation in cover/density of both woodland/scrub and existing herbaceous plant cover. At Gillars Green all plots were situated under a tree canopy of relatively uniform cover/density with little existing vegetation in the field layer. A preliminary survey at this site showed that variation in light climate and cover of field layer vegetation between both sub-plots and sampling areas (quadrats) was insufficient to allow any meaningful comparisons to be made.

4. 2. 1 The effect of the relative abundance of existing field layer vegetation on the numbers of plants introduced as seed

Permanent sampling areas located within plots at the Bold Moss site (Chapter 3) were the sampling areas in which an assessment was made of the relative abundance of plant species other than those introduced as seed (i.e. existing field layer plants). Of the four sampling areas in each plot, three were randomly chosen for monitoring in July 1993 and July 1994. At both times a pin frame was used to determine the cover of all grasses, herbs, mosses and bare ground (vegetation classes) within sampling areas; ten pins were located across each sampling area at 100 mm intervals to give a total of fifty pins per sampling area. For the grasses and herbs a cover repetition method was used, the total number of touches of each species per pin being recorded, while the presence of either mosses or bare ground was recorded at the point of each pin. The total number of individuals (population density) of all introduced species was obtained for each sampling area for July 1993 and July 1994, from data gathered as part of the long-term monitoring of plant establishment and survival (see Chapter 3).

4. 2. 2 The effect of light climate on the numbers of plants introduced as seed

Relative amounts of woodland/scrub cover were assessed for all plots at Bold Moss during July 1995. Within each 8 x 8 m plot random number tables were used to determine precise coordinates for twenty points, and at each a pole 4 m in length was used to determine cover repetition for all woody species. The pole was graduated at 100 mm intervals to allow the measurement of cover at three height classes, 0-1 m, 1-2 m, and 2+ m. The total number (in July 1995), number of cumulative births and proportion of deaths (calculated as the number of deaths divided by the number of births) (to July 1995) of plants of all introduced species were utilised from permanent sampling areas within plots, using data gathered as part of the long-term monitoring of plant establishment and survival (Chapter 3). Data for cumulative births and proportion of deaths was only available for sampling areas within plots to which F1 and F3 fertiliser treatments had been applied.

In June 1995 hemispherical photograph analysis was undertaken to provide sampling area-specific data on light climate. Sixteen permanent sampling areas were randomly chosen from the total of forty-eight, eight from blocks 1 and 2 on the western slope of the tip, eight from blocks 3 and 4 on the eastern slope. A wooden platform 500 x 500 mm in size, with adjustable metal legs at each corner, was positioned over each of the sixteen sampling areas, using the metal bolt at the corner of each sampling area as a marker for the platform legs. A Canon EOS620 camera with a 15 mm fish-eye lens was placed in the centre of the platform, and a circular spirit level was placed on the lens cap and used to alter the length of the platform legs until the lens was horizontal. A compass bearing was taken from above the lens of the camera to determine magnetic north, and a 0.75 m yellow pole placed in the ground c. 1 m from the camera along this bearing, to act as a marker on the photographic image. Photographs were taken over a period of a week (14th - 20th June) using 35mm black and white film (Ilford TMX135-36 ISO100), either on days when direct sunshine was obscured by cloud cover or before sunrise. Data for the total number, number of cumulative births and proportion of deaths of all introduced species, in and up to May 1995, were obtained from data gathered as part of the long-term monitoring of plant establishment and survival (Chapter 3).

4. 2. 3 Data analysis

Cover pin data were used to calculate the mean number of touches per pin for each of the vegetation classes within each sampling area. The values from sampling areas in July 1993 were subtracted from those for the same sampling areas in July 1994, to provide a value for net increase/decrease in the relative abundance of vegetation classes within sampling areas between the two monitoring periods. Probability values for the net increase/decrease data in relation to fertiliser treatment and block were calculated using analysis of variance (Minitab, 1991). Where data were sufficient, Spearman's rank correlation coefficients were calculated between the total number of individuals (population density) of introduced species and the mean number of touches per pin for each of the vegetation classes in July 1993 and July 1994; both within each sampling area, and within each sampling area grouped by block.

Data from the pole survey were plotted as the mean number of touches of woody vegetation for each height class within experimental plots. These values were subsequently combined to provide an index of canopy cover for each plot. These data were used to calculate Spearman's rank correlation coefficients between the pooled values for total number, number of cumulative births and proportion of deaths of introduced species in sampling areas within plots, and the canopy index values. Since it was thought unlikely that canopy cover had changed significantly since summer 1994, Spearman's rank correlation coefficients were also calculated for the canopy indices and the mean of the mean number of touches per pin, for each of the vegetation classes within three sampling areas in each plot in July 1994.

Hemispherical photographs were scanned and stored as 1-bit pict file images on an Apple Macintosh LCIII computer. Images were analysed using SOLARCALC 6. 02 (Chazdon and Field, 1985) to provide data on canopy openness (percentage open sky) above each of the sixteen sampling areas. The number of minutes of direct PFD and the number and duration of sunflecks within each sampling area, for a suntrack across each image on the day the photographs were taken (in June 1995), were also obtained. These data were used to calculate Spearman's rank correlation coefficients between the values for total number, number of cumulative births and proportion of deaths of introduced species within sampling areas in May 1995, and the sampling area-specific hemispherical photograph data. As stated previously, since it was thought unlikely that canopy cover had changed significantly since summer 1994, the hemispherical photographs taken in June 1995 were used to determine Spearman's rank correlation coefficients between the photograph data, and the mean number of touches per pin for each of the vegetation classes within each sampling area in July 1994.

4. 3 Results

4. 3. 1 The effect of the relative abundance of existing field layer vegetation on the numbers of plants introduced as seed

Probability values for analysis of variance of the relative abundance of grasses, herbs, mosses and bare ground in sampling areas in relation to fertiliser treatment and block are shown in Table 4. 1. Fertiliser application to plots was found to have no statistically significant effect on the change in relative abundance of any of the vegetation classes within sampling areas between the two monitoring periods in July 1993 and July 1994. A statistically significant difference occurred between blocks for the relative abundance of grasses, herbs and bare ground. The net changes in vegetation classes for each block are shown in Figure 4. 1. Grasses increased in abundance in all blocks, most particularly in blocks 2 and 4; herbs increased slightly in blocks 1 and 2, but decreased in abundance to a greater extent in blocks 3 and 4. Bare ground abundance fell to the greatest extent in blocks 1 and 2, less so in block 4, and increased slightly in block 3. Mosses increased in abundance in blocks 1, 3, and 4, and decreased in block 2, though all changes were relatively small.

Spearman's rank correlation coefficients for the total number of plants of seven of the introduced species in relation to the relative abundance of vegetation classes in July 1993 and July 1994 are presented in Table 4. 2. In July 1993 and/or July 1994, numbers of *Galium mollugo, Geranium robertianum, Geum urbanum, Primula vulgaris and Prunella vulgaris* showed a negative correlation with increased abundance of grasses, though this was only statistically significant for *Galium mollugo* and *Geranium robertianum* in 1993. Conversely, weak positive correlations were seen for *Geranium robertianum* (in 1994 only), *Silene dioica* and *Teucrium scorodonia* (in both July 1993 and July 1994). All species showed a similar negative correlation with an increased abundance of herbs, though for *Silene dioica* this was only evident in July 1994. For *Teucrium scorodonia* this correlation was statistically significant in both years, for *Primula vulgaris* and *Silene dioica* in July 1994 only. The relative abundance of mosses was not seen to have any statistically significant effect on numbers of introduced species, though numbers of most were seen to be negatively correlated with an increase in their abundance; *Geranium robertianum* and *Primula vulgaris* numbers were the only exception, being positively correlated with increased abundance in July 1994. Most species demonstrated a strong positive correlation

Table 4. 1 Probability values for analysis of variance of abundance of grasses, herbs, mosses and bare ground in response to fertiliser and block, July 1993 to July 1994, Bold Moss Tip, St. Helens. (* denotes $P<0.05$, ** $P<0.01$).

	SOURCE OF VARIANCE	
MEASURED VARIABLE	**Fertiliser**	**Block**
Grasses	0.229	0.007**
Herbs	0.977	0.008**
Mosses	0.645	0.482
Bare Ground	1.000	0.003**

Figure 4. 1 Net change in mean number of touches per pin in sampling areas for grasses, herbs, mosses and bare ground between July 1993 and July 1994 in four blocks, Bold Moss Tip, St. Helens.

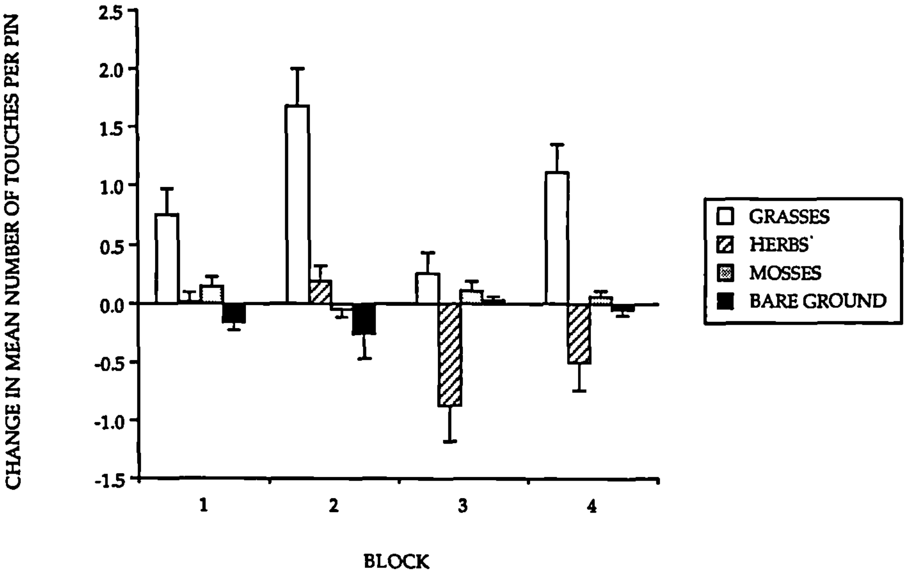


Table 4. 2 Spearman's Rank Correlation Coefficients: The total number of plants of introduced species in relation to the relative abundance of grasses, herbs, mosses, and bare ground in sampling areas, July 1993 and July 1994, Bold Moss Tip, St. Helens. (* denotes $P<0.05$, ** $P<0.01$). Missing values relate to species whose numbers were insufficient for inclusion in the analysis.

SPECIES	MEASURED VARIABLE							
	Grasses		Herbs		Mosses		Bare Ground	
	1993	1994	1993	1994	1993	1994	1993	1994
Galium mollugo	-0.327*	-0.220	-0.298	-0.038	-0.128	-0.311	0.358*	0.355*
Geranium robertianum	-0.439**	0.037	-0.166	-0.259	-0.117	0.093	0.335*	-0.079
Geum urbanum	-	-0.162	-	-0.315	-	-0.014	-	0.389*
Primula vulgaris	-	-0.047	-	-0.362*	-	0.014	-	0.232
Prunella vulgaris	-0.242	-0.172	-0.059	-0.319	-0.058	-0.046	0.211	0.413*
Silene dioica	0.165	0.049	0.302	-0.531**	-0.131	-0.119	-0.241	0.510**
Teucrium scorodonia	0.056	0.072	-0.664**	-0.485**	-0.006	-0.021	0.542**	0.425**

between the number of individuals and an increase in the abundance of bare ground within sampling areas; for *Galium mollugo* and *Teucrium scorodonia* this was statistically significant in both years, for *Geum urbanum, Prunella vulgaris* and *Silene dioica* in July 1994, and for *Geranium robertianum* in July 1993.

Correlations between the same variables within each block are shown for July 1993 and July 1994, in Tables 4. 3 and 4. 4 respectively. The strength and/or direction of correlations between the numbers of individuals of introduced species and the relative abundance of the four vegetation classes varied between blocks and years for most species. In July 1993 the numbers of *Galium mollugo, Geranium robertianum, Prunella vulgaris* and *Teucrium scorodonia* in block 1 showed a statistically significant negative correlation with increased abundance of herbs. In the case of *Teucrium scorodonia* this also occurred in block 3. In July 1994 these correlations remained negative for the same species and blocks, though they were no longer statistically significant. Figure 4. 2 presents graphs of the relative abundance of the four vegetation classes for each block. These show that there was no change in the relative abundance of herb cover in block 1 between July 1993 and July 1994. Also in block 1 in July 1993, statistically significant negative correlations were seen between the numbers of individuals of *Prunella vulgaris* and *Teucrium scorodonia* and an increased abundance of mosses. Statistically significant positive correlations occurred for numbers of the same two species in block 1 and an increase in the abundance of bare ground in sampling areas. In July 1994 statistically significant correlations relating numbers of individuals with the abundance of mosses and bare ground showed a number of differences from the previous pattern. The numbers of *Silene dioica* and *Teucrium scorodonia* were positively correlated with an increase in abundance of bare ground in block 1, the number of *Galium mollugo* was positively correlated with an increase in bare ground in block 4. The number of *Geum urbanum* showed a positive correlation with an increase in the abundance of mosses in block 4, and a negative correlation in block 3, and the number of *Prunella vulgaris* was positively correlated with increased mosses abundance in block 4. Reference to Figure 4. 2 shows that the relative abundance of mosses and bare ground in all blocks, varied very little between the two years. The most noticeable change was an increase in the abundance of grasses, particularly in blocks 1, 2 and 4, and a slight decrease in herbs abundance in blocks 3 and 4.

4. 3. 2 The effect of light climate on the numbers of plants introduced as seed

Graphs showing the height class distribution of woody species cover within plots at Bold Moss are shown in Figure 4. 3. Plots in blocks 1 and 2 were characterised by generally

Table 4. 3 Spearman's Rank Correlation Coefficients: The total number of plants of introduced species in relation to the relative abundance of grasses, herbs, mosses, and bare ground in each block, July 1993, Bold Moss Tip, St. Helens. (* denotes $P<0.05$, ** $P<0.01$). Missing values relate to species whose numbers were insufficient for inclusion in the analysis.

	MEASURED VARIABLE															
SPECIES	Grasses				Herbs				Mosses				Bare Ground			
	B1	B2	B3	B4	B1	B2	B3	B4	B1	B2	B3	B4	B1	B2	B3	B4
Galium mollugo	-0.586	0.149	-0.388	-0.456	-0.717*	-0.026	0.367	0.183	-0.448	-0.241	-0.068	0.139	0.536	0.166	-0.092	0.367
Geranium robertianum	-0.451	-0.070	-0.550	-0.621	-0.919**	-0.057	0.138	0.168	-0.521	-0.243	0.124	0.067	0.615	0.149	-0.187	0.677
Geum urbanum	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Primula vulgaris	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Prunella vulgaris	-0.456	0.137	-0.239	-0.311	-0.730*	-0.298	-0.020	0.104	-0.722*	0.390	0.538	-0.157	0.733*	-0.365	0.110	0.416
Silene dioica	-	0.104	-0.373	0.574	-	0.000	0.225	-0.406	-	-0.572	0.075	0.020	-	0.518	0.225	-0.562
Teucrium scorodonia	-0.518	0.086	0.518	-	-0.757*	-0.147	-0.728*	-	-0.745*	-0.052	0.311	-	0.760*	0.155	0.312	-

Table 4. 4 Spearman's Rank Correlation Coefficients: The total number of plants of introduced species in relation to the relative abundance of grasses, herbs, mosses, and bare ground in each block, July 1994, Bold Moss Tip, St. Helens. (* denotes P<0.05, ** P<0.01). Missing values relate to species whose numbers were insufficient for inclusion in the analysis.

SPECIES	MEASURED VARIABLE															
	Grasses				Herbs				Mosses				Bare Ground			
	B1	B2	B3	B4	B1	B2	B3	B4	B1	B2	B3	B4	B1	B2	B3	B4
Galium mollugo	-0.382	0.332	-0.603	-0.778	-0.192	-0.215	0.585	-0.313	-0.490	-0.417	0.027	0.171	0.631	0.079	-0.443	0.821*
Geranium robertianum	-0.511	0.059	-0.228	0.639	-0.103	-0.065	0.068	-0.550	-0.026	-0.070	0.138	0.456	0.122	-0.020	-0.391	-0.431
Geum urbanum	-0.361	-0.254	-0.189	-0.523	0.008	0.026	0.488	-0.562	-0.485	0.158	-0.728*	0.853**	0.468	0.498	0.578	0.616
Primula vulgaris	-0.291	0.328	-0.621	-0.251	-0.153	0.165	0.000	-0.493	-0.196	-0.058	0.313	0.593	0,312	-0.106	-0.313	0.443
Prunella vulgaris	-0.446	0.035	-0.283	-0.505	-0.111	-0.009	0.256	-0.544	-0.368	-0.095	-0.028	0.798*	0.565	0.086	0.262	0.597
Silene dioica	-0.084	-0.367	-	-	-0.391	-0.120	-	-	-0.540	0.253	-	-	0.796*	0.651	-	-
Teucrium scorodonia	-0.321	0.051	-0.028	-0.274	-0.514	-0.034	-0.193	-0.275	-0.647	0.216	0.361	-0.411	0.767*	0.059	0.236	0.287

Figure 4. 2 Mean per block of the mean number of touches per pin for grasses, herbs, mosses and bare ground in each sampling area, July 1993 and July 1994, Bold Moss Tip, St. Helens.

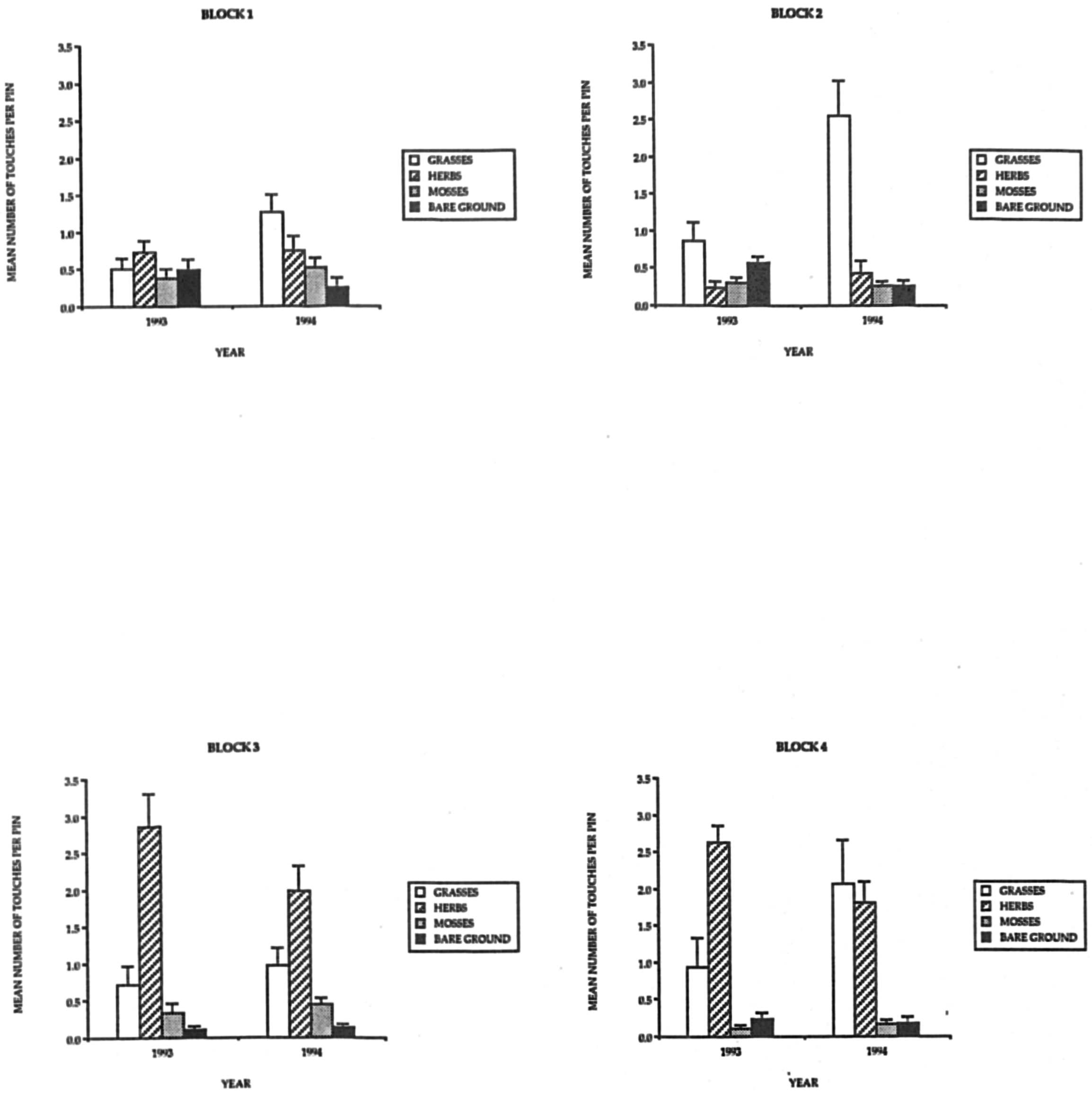


Figure 4. 3 Pole survey assessment of woodland/scrub cover in experimental plots, July 1995, Bold Moss Tip, St. Helens.

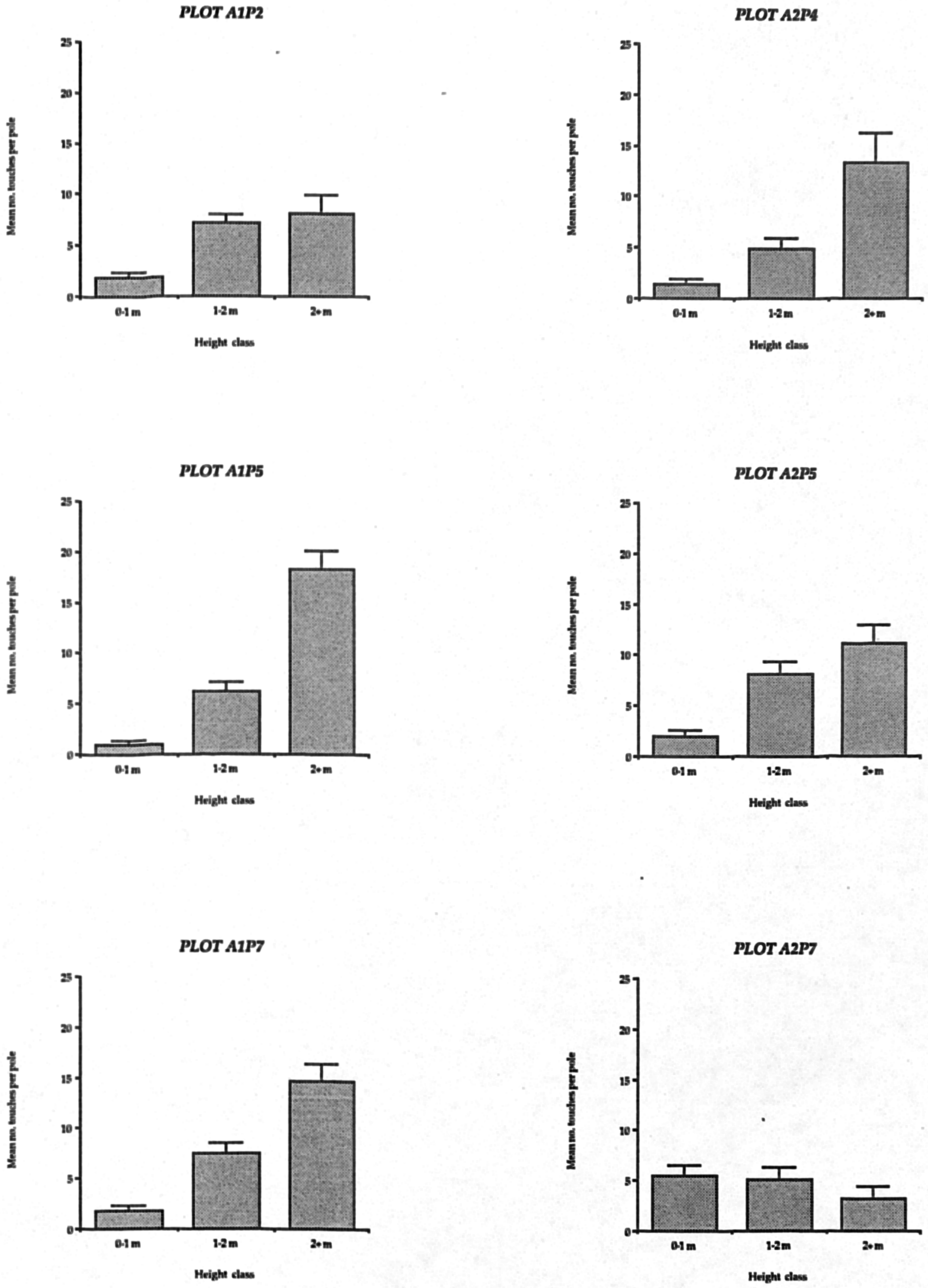
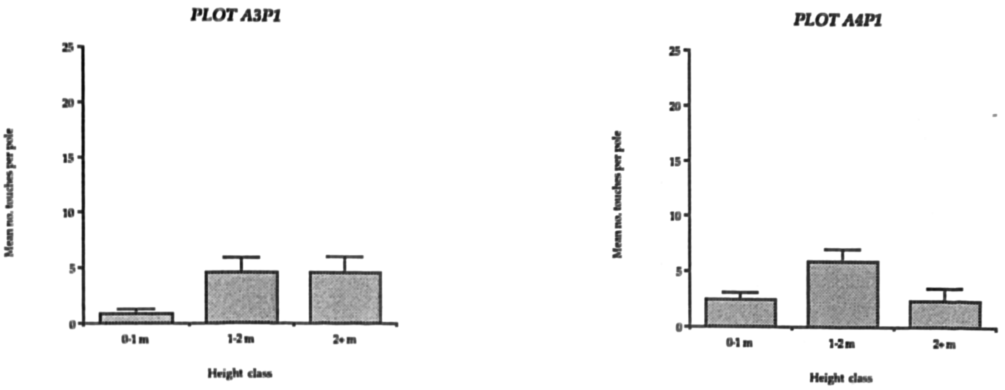
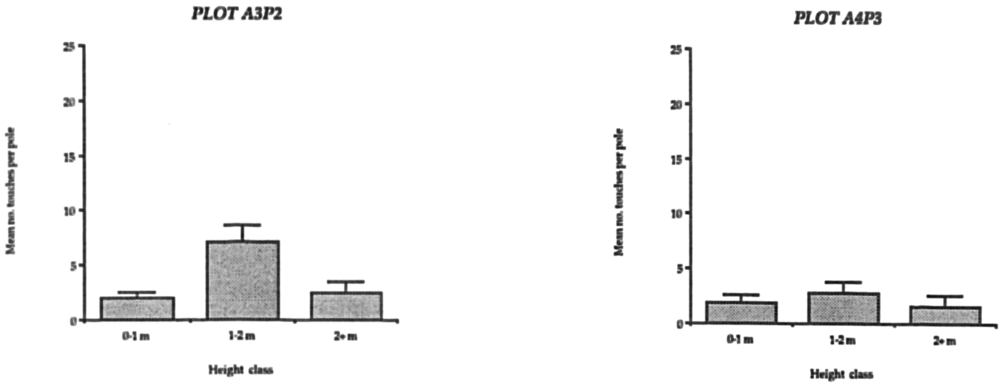


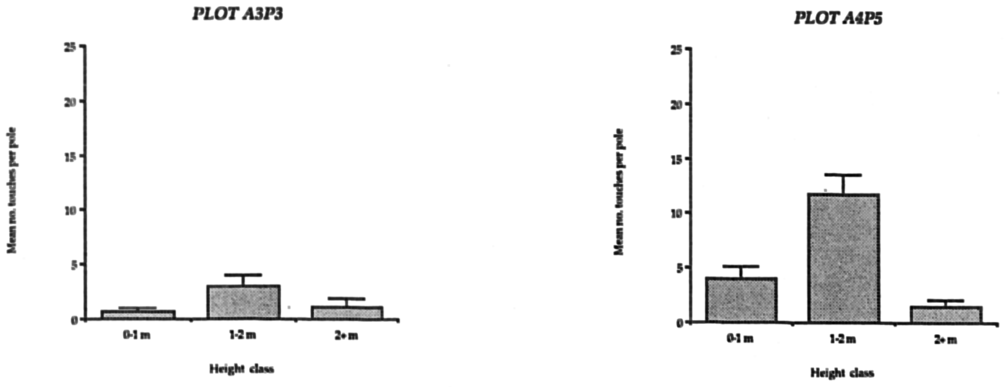
Figure 4. 3 Continued


PLOT A3P1

PLOT A4P1


PLOT A3P2

PLOT A4P3


PLOT A3P3

PLOT A4P5

higher canopy densities than in blocks 3 and 4, with a proportionately greater number of touches at the 2+ m height class. With the exception of plots A2P7 and A4P5, the effects of canopy shading in the field layer were therefore likely to have been greater for plots in blocks 1 and 2 than in blocks 3 and 4. No statistically significant correlations were found between the index of scrub cover and the total number of individuals, number of cumulative births, or proportion of deaths, of introduced species (see Table 4. 5). There was also no statistically significant difference between the canopy index and the relative abundance of vegetation classes within sampling areas (see Table 4. 6). There were some clear trends however; the total number of individuals of *Galium mollugo, Prunella vulgaris* and *Teucrium scorodonia* showed a negative correlation with increasing canopy cover, while *Geranium robertianum, Geum urbanum, Primula vulgaris* and *Silene dioica* were positively correlated. With the exception of *Geum urbanum* all species showed a positive correlation between the number of cumulative births within sampling areas and increased canopy cover, while the proportion of deaths was seen to decrease as canopy cover increased for four species, *Geranium robertianum, Primula vulgaris, Silene dioica* and (especially) *Geum urbanum*. There was a strong positive correlation between the proportion of deaths of *Teucrium scorodonia* and increased canopy cover within plots. Increased canopy cover was also associated with a reduction in the relative abundance of herbs and to a lesser extent mosses, and a slight increase in the abundance of grasses and bare ground.

Examples of hemispherical photographs taken at Bold Moss are shown in Plate 4. 1. The sampling area-specific data on light climate derived from the hemispherical photograph analyses are presented in Tables 4. 7 and 4. 8. Of the seven species for which data were analysed, four show statistically significant correlations between variables. The total number of *Geranium robertianum* within sampling areas was negatively correlated with an increase in the percentage of open sky and number of minutes of direct PFD. An increase in the number of sunflecks of more than 40 minutes duration was also correlated with a decrease in total numbers of individuals and cumulative births. The proportion of deaths within sampling areas was positively correlated with an increase in both the number of minutes of direct PFD and in the number of sunflecks of more than 40 minutes duration. The proportion of deaths for *Geum urbanum* was positively correlated with an increase in the number of minutes of direct PFD, and the total number of individuals was negatively correlated with an increase in the number of sunflecks of more than 40 minutes duration. The total number of *Primula vulgaris* was negatively correlated, while the proportion of deaths was positively correlated, with an increase in the number of minutes of direct PFD. Sunfleck length also appeared to be an important factor for *Primula vulgaris*, since the proportion of deaths was negatively associated with an increase in the number of flecks of

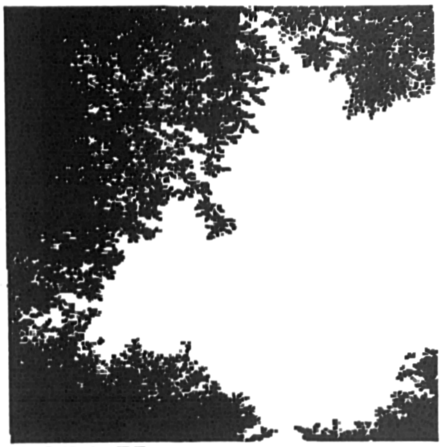
Table 4. 5 Spearman's Rank Correlation Coefficients: The total number, number of cumulative births and proportion of deaths of plants of introduced species in sampling areas in relation to the density of woodland/scrub cover, July 1995, Bold Moss Tip, St. Helens. (* denotes $P<0.05$, ** $P<0.01$).

SPECIES	MEASURED VARIABLE		
	Total Number	Cum. Births	Prop. Deaths
Galium mollugo	-0.222	0.143	0.000
Geranium robertianum	0.418	0.551	-0.241
Geum urbanum	0.400	0.000	-0.659
Primula vulgaris	0.509	0.386	-0.288
Prunella vulgaris	-0.021	0.108	0.036
Silene dioica	0.235	0.587	-0.037
Teucrium scorodonia	-0.204	0.381	0.595

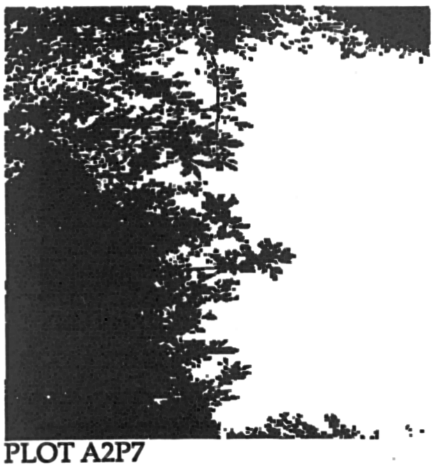
Table 4. 6 Spearman's Rank Correlation Coefficients: The relative abundance of grasses, herbs, mosses, and bare ground in sampling areas, July 1994, in relation to the density of woodland/scrub cover, July 1995, Bold Moss Tip, St. Helens. (* denotes $P<0.05$, ** $P<0.01$).

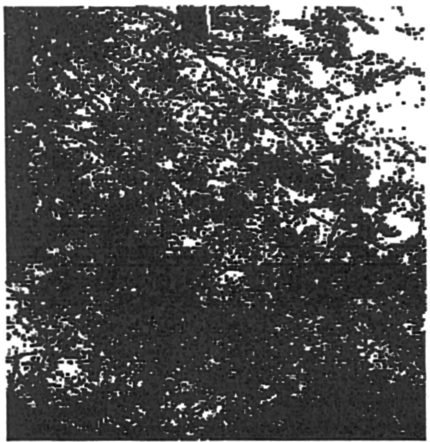
MEASURED VARIABLE			
Grasses	Herbs	Mosses	Bare Ground
0.252	-0.545	-0.018	0.112

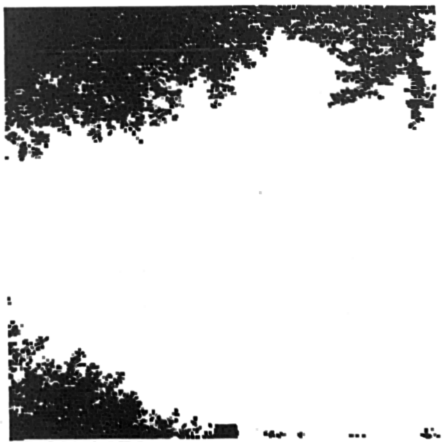
Plate 4. 1 A selection of hemispherical photographs taken at Bold Moss, June 1995.


PLOT A1P5


PLOT A4P3


PLOT A2P7


PLOT A2P4


PLOT A3P3


PLOT A3P3

Table 4. 7 Spearman's Rank Correlation Coefficients: The total number, number of cumulative births and proportion of deaths of introduced species in sampling areas in May 1995, in relation to hemispherical photo data, June 1995, Bold Moss Tip, St. Helens. (* denotes $P<0.05$, ** $P<0.01$).

T= Total number of plants, B= Cumulative births, D= Proportion of deaths.

	MEASURED VARIABLE														
	% Open Sky			Direct PFD			No. Sunflecks			Flecks <40			Flecks >40		
SPECIES	T	B	D	T	B	D	T	B	D	T	B	D	T	B	D
Galium mollugo	-0.108	-0.477	0.146	-0.014	-0.521	0.491	-0.003	-0.136	-0.295	0.035	0.017	-0.368	-0.182	-0.515	0.494
Geranium robertianum	-0.614*	-0.597	0.593	-0.585*	-0.624	0.710*	-0.055	-0.188	-0.046	0.101	0.068	-0.232	-0.729**	-0.723*	0.689*
Geum urbanum	-0.330	-0.390	0.475	-0.417	-0.519	0.740*	-0.128	-0.293	0.017	0.003	-0.052	-0.190	-0.567*	-0.625	0.625
Primula vulgaris	-0.497	-0.516	0.667	-0.556*	-0.518	0.821*	-0.095	-0.285	-0.500	-0.040	-0.231	-0.895**	-0.376	-0.264	0.730*
Prunella vulgaris	-0.144	-0.479	-0.217	-0.176	-0.359	-0.217	-0.296	-0.573	-0.173	-0.242	-0.463	-0.049	-0.191	-0.338	-0.119
Silene dioica	-0.601*	-0.701*	-0.034	-0.424	-0.575	0.135	-0.088	0.017	0.343	0.000	0.174	0.343	-0.583*	-0.717*	0.018
Teucrium scorodonia	-0.465	-0.686	-0.145	-0.168	-0.433	0.012	-0.268	0.162	0.273	-0.247	0.196	0.236	-0.207	-0.365	-0.086

Table 4. 8 Spearman's Rank Correlation Coefficients: The relative abundance of grasses, herbs, mosses, and bare ground in sampling areas, July 1994, in relation to hemispherical photo data, June 1995, Bold Moss Tip, St. Helens. (* denotes P<0.05, ** P<0.01).

	HEMISPHERICAL PHOTO DATA				
MEASURED VARIABLE	% Open Sky	Direct PFD	No. Sunflecks	Flecks <40	Flecks >40
Grasses	-0.137	-0.136	0.156	0.234	-0.302
Herbs	0.665*	0.627*	0.046	-0.147	0.673*
Mosses	0.227	0.639*	0.044	-0.127	0.555
Bare Ground	-0.719*	-0.961**	-0.190	0.053	-0.851**

less than 40 minutes duration, but positively correlated with an increase in the number of flecks of more than 40 minutes duration. For *Silene dioica* the total number of individuals and cumulative births were negatively correlated with an increase in both percentage open sky and an increase in the number of sunflecks of more than 40 minutes duration.

Within individual sampling areas, hemispherical photograph data also showed some statistically significant correlations with the relative abundance of some of the vegetation classes (Table 4. 8). The abundance of herbs was positively correlated with increases in percentage open sky, number of minutes of direct PFD and number of sunflecks of more than 40 minutes duration. The abundance of mosses was positively correlated with an increase in the number of minutes of direct PFD. Bare ground was negatively correlated with increases in percentage open sky, the number of minutes of direct PFD and the number of sunflecks of more than 40 minutes duration. The relative abundance of grasses was not statistically significantly correlated with any of the hemispherical photograph data, but was shown to be reduced by increased canopy openness.

4. 4 Discussion

4. 4. 1 The effect of the relative abundance of existing field layer vegetation on the numbers of plants introduced as seed

Given the generally low concentration of plant mineral nutrients in spoil at Bold Moss, it was expected that addition of compound fertiliser would cause changes in the structure and/or composition of the existing field layer communities within experimental plots. Other work has shown that the addition of mineral nutrients can affect both above ground biomass and the relative abundance of species within a relatively short period of time, since addition of N and P is likely to effect grasses and herbs to a different extent (e.g. Mamolos, Veresoglou and Barbayiannis, 1995). Where the abundance of existing vegetation was greatest, or showed a response to fertiliser addition, the number of plants of introduced species was expected to be reduced by increased competition from existing plant species, which would have higher efficiency of resource capture.

Fertiliser application was found to have no statistically significant effect on the relative abundance of any of the vegetation classes within sampling areas between the two monitoring periods, though statistically significant differences were found between blocks. It may be possible that too little time elapsed between the application of fertiliser in April 1993 and the second monitoring period in July 1994. However, this is unlikely as the slow-release component of the fertiliser mixture should have continued to supply nutrients for at least one complete growing season. An alternative hypothesis relates to the difficulty of predicting the likely effects of increased nutrient supply on plants, which is often far from straightforward, and involves a range of possible morphological, physiological and biochemical processes (Fitter, 1986). This may have meant that changes in vegetation composition and/or structure did occur in response to fertiliser addition, but were not measured and therefore not determined. This hypothesis was not tested any further. Statistically significant changes in relative abundance for some vegetation classes in different blocks may provide evidence for fertiliser application influencing vegetation abundance within blocks, as it is unlikely that existing mineral nutrient levels could support such a relatively rapid change from one year to the next. Plots within block 2 for example were characterised by low pH values and P concentrations, yet increases in the relative abundance of both grasses and herbs were greatest in this block. These increases most probably resulted from fertiliser additions to two of the three plots within this block.

Spearman's rank correlation coefficient values for seven of the introduced species indicated that the total number of most species was associated (to varying degrees) with the relative abundance of vegetation classes within sampling areas. The negative correlation between total numbers of many of the introduced species and increased abundance of grasses and herbs most probably resulted from a reduction in the number of safe sites for seed germination and establishment, as well as increased root/shoot competition within sampling areas. A majority of positive correlations between the number of individuals and an increase in the abundance of bare ground for all species at one or both of the monitoring periods further supports this hypothesis. The weakly positive correlation between total numbers of *Silene dioica* and *Teucrium scorodonia* and increased grasses abundance in both years is more difficult to interpret, as both species are known to be more abundant in habitats with intermediate to high amounts of bare ground (Grime *et al.*, 1988). Of the seven species examined however, both *Silene dioica* and *Teucrium scorodonia* are commonly found in soils of pH 5 or less (Grime *et al.*, 1988). At Bold Moss where most of the grass species in the experimental plots were either *Agrostis capillaris* L. or *Agrostis stolonifera*, it is possible that plants of these two species, and a majority of the grasses, were both associated with areas of spoil where pH values were relatively low.

Examination of correlation coefficient values for the same seven species in each block provided little additional evidence to help determine the possible relationships between total numbers of individuals of introduced species and the relative abundance of vegetation classes. A strong (and in July 1993 statistically significant) negative correlation between the numbers of *Galium mollugo, Geranium robertianum, Prunella vulgaris* and *Teucrium scorodonia* and increased herbs abundance occurred in block 1, despite much lower herb abundance than in blocks 3 and 4, while the strength and/or direction of correlations for most species were seen to vary between blocks and years. Given this level of variation for most species, and the fact that some correlations within blocks were inconsistent with those for all sampling areas at the same monitoring period (being of differing strengths or directions), it is likely that some of the correlations were derived from a range of other environmental factors, such as undetermined spoil chemical characteristics.

4. 4. 2 The effect of light climate on the numbers of plants introduced as seed

As spatial patterns of light in the woodland field layer are largely determined by the three-dimensional structure of the woodland canopy (Chazdon, 1988), indices of canopy cover were thought to be suitable for examining the relationship between the extent of tree and shrub cover within plots and levels of establishment and survival of the introduced species. Other research has shown that following introduction of either seeds or plants, some herbaceous woodland species are likely to benefit from relatively high amounts of canopy cover and the associated reduction in abundance of more competitive species (Francis, 1993). Whilst no statistically significant correlations were found between scrub cover indices and either the establishment and survival of the introduced species or relative abundance of vegetation classes within sampling areas, some clear trends were observed. Total numbers of *Galium mollugo, Prunella vulgaris* and *Teucrium scorodonia* were all correlated with increased canopy cover, as expected for species most commonly associated with open habitats and/or woodland margins and scrub (Grime *et al.*, 1988), while the number of those species characteristic of more shaded habitats, *Geranium robertianum, Geum urbanum, Primula vulgaris* and *Silene dioica,* was positively correlated with greater canopy cover. These habitat preferences appeared to be reflected in the correlation values for the proportion of deaths and canopy cover indices for most of the species. Though the number of cumulative births was positively correlated with increased canopy cover for all species except *Geum urbanum,* on which canopy cover had a neutral effect, proportion of deaths was positively correlated with increased canopy cover for both *Prunella vulgaris* and *Teucrium scorodonia,* and negatively correlated with increased canopy cover for the more shade-demanding species. Increased canopy cover was also correlated with a reduction in the relative abundance of herbs and mosses, and with an increase in the amount of grasses and bare ground.

Though the use of canopy indices proved useful in determining the broad relationships between canopy cover and the performance of some of the introduced species, more detailed information was required to allow comparisons between individual sampling areas. Hemispherical photograph analysis was used, as it is a technique that has proved valuable in the determination of relationships between light climate and the distribution and abundance of woodland plant and invertebrate species (e.g. Greatorex-Davies, Sparks and Hall, 1994; Valverde and Silvertown, 1995). Traditionally the analysis of hemispherical photographs has involved the estimation of amounts of direct and diffuse light under woodland canopies, and has utilised manual or semi-automated methods for calculating the percentage of shade or open sky for each photographic image (e.g. Barrie *et al.*, 1990). More recently, the development of the computer programme SOLARCALC (Chazdon and Field, 1985) has allowed the relatively rapid analysis of hemispherical photographs, and in addition to data for diffuse light, more detailed calculations for PFD levels and sunfleck variables within precisely located points on the woodland floor. Whilst the analyses of hemispherical photographs are unable to give precise predictions of sunfleck activity, they can, unlike sensor measurements, ..."provide a means of assessing light conditions over a relatively long period of time" (Chazdon, 1988). As leaf expansion in the woodland/scrub canopy within experimental plots had reached a maximum by June 1995, it was assumed that photographs taken at this time would be representative of woody species cover throughout the summer period. Additional photographs were taken from the same sampling areas within a number of plots during August 1995 to test this hypothesis, and analysis of the photographic images showed no significant differences between the two sets of data (see Appendix 4.1).

Total numbers of both *Geranium robertianum* and *Silene dioica* showed statistically significant negative correlations with an increase in the percentage of open sky above sampling areas. Both these species are particularly associated with habitats in which the vigour of potential dominant species is reduced by disturbance and/or shade (Grime *et al.*, 1988), and were probably most abundant where canopy shading was sufficient to reduce PFD (for *Geranium robertianum*) and the number of sunflecks of more than 40 minutes duration (for *Geranium robertianum* and *Silene dioica*) to levels that inhibited the growth of more competitive species. The correlation between an increase in the proportion of deaths of *Geranium robertianum* and greater levels of PFD and number of sunflecks of more than 40 minutes duration tends to support this hypothesis for this species. Cumulative births for both species were negatively correlated with either an increase in the percentage of open sky (*Silene dioica*) or an increase in the number of sunflecks of more than 40 minutes duration (both species), which may indicate an increased likelihood of germination inhibition and/or reduced seedling establishment with decreasing shade, though such an assumption cannot be tested without further investigations. The performance of *Primula vulgaris* was characterised by relationships with canopy characteristics similar to those for *Geranium robertianum*, with total numbers being negatively correlated, and relative mortality positively correlated, with an increase in direct PFD. This indicates that mortality response under certain canopy conditions may be an important determinant of population size for this species. Though tolerant of canopy shade, measures of reproductive output for *Primula vulgaris*, and seedling emergence in the field, have previously been shown to be enhanced by higher light levels (Valverde and Silvertown, 1995), which would appear to contradict this assumption. At Bold Moss the data appear to show that relative mortality may rise in response to increased light availability. The proportion of deaths was negatively correlated with the number of shorter sunflecks (<40 minutes) but positively correlated with the number of those of longer duration (>40 minutes). It therefore appears that *Primula vulgaris* has an optimal range of light climate conditions which ensure minimum mortality but adequate levels of growth and survival.

In contrast *Geum urbanum* is a species characteristic of 'semi-shade' (Grime *et al.*, 1988), and at Bold Moss appeared to be most abundant in the most shaded sampling areas. The total number of individuals within sampling areas was negatively correlated with an increase in the number of sunflecks of more than 40 minutes duration, and positively correlated with an increase in the number of sunflecks of less than 40 minutes duration. The correlation between an increase in proportion of deaths and greater PFD levels for this species indicates that whilst germination rates may be equal across a range of canopy shade, mortality will rise with increased canopy openness. Though, as with all correlation data, it is impossible to determine definite proof of cause and effect between these factors, such a relationship would not be unexpected for this species. Similarly, the relative abundance of vegetation classes within sampling areas also showed a number of expected associations with hemispherical photograph characteristics. The strong and statistically significant correlations between herbs and bare ground abundance, and the respective increase and decrease in the percentage of open sky above sampling areas, again indicates that canopy cover is likely to have influenced the performance of a number of the introduced species at Bold Moss, most particularly those with greater sensitivity to the effects of competition from potential dominant species within sampling areas.

4. 5 Conclusions

Successful introduction of herbaceous woodland species into existing vegetation at Bold Moss was dependent upon a range of environmental factors, any one of which may have either limited or facilitated the establishment of particular species. In examining the possible influence of plant competition, grasses and herbs limited the numbers of some introduced species. Conversely, areas of bare ground were associated with greater numbers of plants of most introduced species. Therefore, whilst bare ground was more likely to provide sites suitable for the germination and establishment of individuals of most species, some species appeared to be more sensitive to the effects of assumed competition from established vegetation on site. The relative abundance of existing vegetation and bare ground at Bold Moss was subsequently shown to be partly a function of the density of canopy cover from woody species within experimental plots. Canopy layer characteristics were therefore an important determinant of population size for many of the introduced species. By locally affecting levels of PFD and sunfleck number and duration in the field layer, canopy density and architecture may influence shoot competition at relatively small spatial scales, and therefore largely determine the extent to which plants can become successfully established. On colliery spoil however, extremely low pH may limit the number of suitable microhabitat patches still further (see Chapter 3).

At sites where the species richness of communities is increased by deliberate introduction of additional species, canopy layer characteristics should be used to determining an appropriate range of species for introduction to the field layer; *Galium mollugo, Prunella vulgaris, Teucrium scorodonia* are likely to have higher populations in areas of moderate canopy cover, while greater canopy cover will better suit species such as *Geranium robertianum, Geum urbanum, Primula vulgaris* and *Silene dioica*. Using those species that performed well at Bold Moss as a guide, other species (including woodland grasses and ferns) with similar ecological strategies (Grime *et al.*, 1988) could be identified and introduced to appropriate areas. If this procedure is undertaken over a number of growing seasons, it could be adapted to take account of any changes in vegetation characteristics in both the field and canopy layers. Given the high variability in spoil pH and amount of canopy cover at Bold Moss, the establishment of introduced plant species at this site will probably continue to be spatially highly variable. If however a wide range of potentially suitable species is used at the beginning of all introduction programmes, the successes and failures of the plants themselves will be able to help determine future site management options.

5. 0 ESTABLISHING WOODY SPECIES ON COLLIERY SPOIL

5. 1 Introduction

Spontaneous establishment of woody plant species is a characteristic of a range of derelict industrial sites (Prach and Pysek, 1994). Of all pioneer tree and shrub species on colliery waste in Britain, *Betula spp.* are often the most widespread and abundant, sometimes forming extensive stands after only a few decades (Hall, 1957). Both of the native *Betula* species are well adapted to tolerate the often extreme chemical and physical characteristics of colliery spoil, and can under certain conditions act as 'soil improvers'. On moorland soils for example they have been shown to convert mor to a mull humus type, as well as increase soil pH and exchangeable calcium and total phosphorus levels (Atkinson, 1992). However, whilst these species may help facilitate the long-term development of more diverse plant communities through improving spoil characteristics, when contrasted with stands in many semi-natural broadleaved woodlands, tree and shrub diversity on colliery waste sites remains low at all successional stages (Hall, 1957). As species heterogeneity in the canopy layer largely determines spatial and architectural (structural) diversity in woodland stands (Barkman, 1992), low specific diversity and evenness may in the long-term provide fewer niches for the maintenance of a range of associated plant and animal communities. Higher invertebrate diversities for example are known to be strongly associated with increases in structural diversity in woodland habitats (Southwood, Brown and Reader, 1979), as well as with the relative abundance of certain trees species (Kennedy and Southwood, 1984).

In Britain tree planting has been undertaken on colliery waste sites, though to date in Britain much of this planting has taken place on restored opencast sites, often for forestry purposes. Planting mixtures of acid-tolerant species, including a range of conifers, have been defined by the forestry profession for this purpose (Moffat and McNeill, 1994), though mixtures of purely native trees and shrubs have also been specified (Rodwell and Patterson, 1994). Tree establishment on colliery spoil is however sometimes severely limited by a number of 'particular problems' associated with this substrate (Moffat and McNeill, 1994); spoil acidity, salinity and infertility, as well as spoil compaction and vandalism may all serve to seriously affect tree survival and growth. When new woodland is planned for a site, and the production of commercially valuable timber is either impracticable (due to limitations related to site conditions) or undesirable (due to local planning policy), efforts should be made to establish ..."whole communities of native trees and shrubs appropriate to the site...with the aim of approaching the appearance and ecological integrity of semi-natural woodland" (Rodwell and Patterson, 1994). Where woodland is established in areas with little existing ecological value, these new woodlands are likely to provide both net ecological benefits (Watkins, 1993), and increased opportunities for recreation and amenity uses. In industrial and urban areas these benefits will be of particular importance.

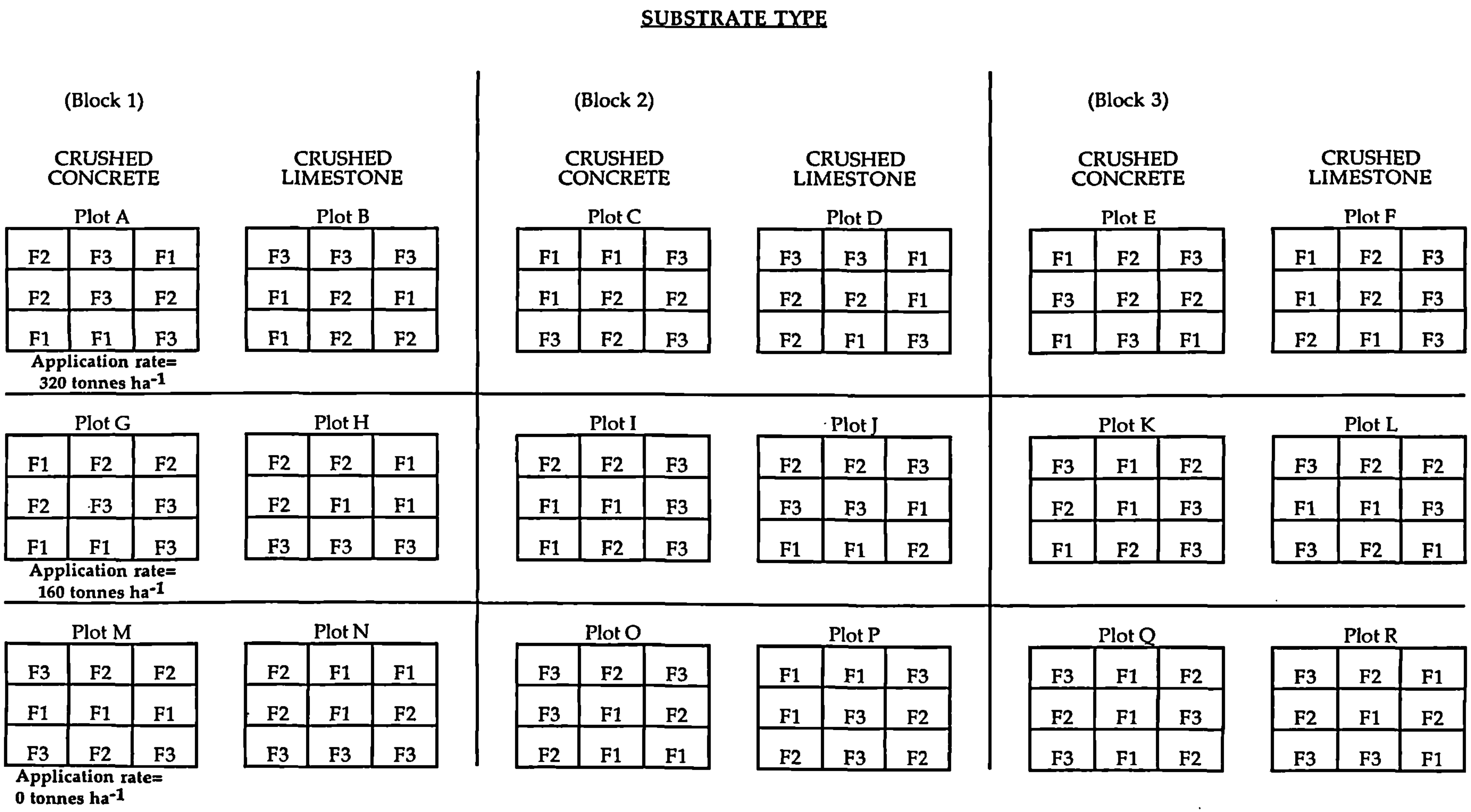
Current guide-lines on the choice of suitable native broadleaved tree and shrub species for planting on colliery spoil tend to rely on assessments of species' tolerance of spoil acidity (e.g. Moffat and McNeill, 1994; Rodwell and Patterson, 1994; Bradshaw, Hunt and Walmsley, 1995). Little detailed data is however readily available on specific measures of growth performance for many of the broadleaved tree and shrub species that might be planted on deep mined colliery spoil. This chapter examines the effects of altered mineral nutrient concentration on the growth of a number of tree and shrub species on colliery spoil, both in the field and a glasshouse. The effects of altered spoil pH on the growth of *Quercus robur* are also examined. The results are discussed with reference to the findings for the herbaceous plant species introductions in Chapters 3 and 4, to provide guide-lines for the diversification of woodland field and canopy layers on sites with similar characteristics.

5. 2 Methods

5. 2. 1 Field experiments

Both the Bold Moss and Gillars Green sites were used for experimental work. Tree and shrub species characteristic of semi-natural woodlands in north-west England were firstly identified from NVC floristic tables (Rodwell, 1991). At Bold Moss, plots already established by the St. Helens Groundwork Trust were used to determine tree growth in relation to variations in spoil pH and mineral nutrient availability. Eighteen 8 x 5.4 m plots were established on bare colliery spoil in 1990/91, in three rows of six plots. Within two of the rows of six plots, two substrate ameliorant treatments (either crushed concrete or crushed limestone) were systematically allocated to plots to give three plots per substrate treatment per row. Application rates were 160 and 320 tonnes ha^{-1} for both materials, with one rate of application being consistently applied across all the plots within a row (see Figure 5. 1). The remaining row of six plots was maintained as a control. Each 8 x 5.4 m plot was further sub-divided into nine 1.8 x 2.7 m sub-plots, and following the collection of five randomly located spoil samples, a combination of fast and slow-release NPK fertilisers was used to randomly apply one of the following treatments to each of three

Figure 5. 1 A schematic representation of *Quercus robur* experimental plots, sub-plots, blocks, and substrate/fertiliser treatments at Bold Moss Tip, St. Helens. N.B. Where F1= zero, F2= low and F3= high fertiliser application.



SUBSTRATE TYPE

(Block 1)		(Block 2)		(Block 3)	
CRUSHED CONCRETE	CRUSHED LIMESTONE	CRUSHED CONCRETE	CRUSHED LIMESTONE	CRUSHED CONCRETE	CRUSHED LIMESTONE
Plot A F2 F3 F1 F2 F3 F2 F1 F1 F3 Application rate= 320 tonnes ha^{-1}	Plot B F3 F3 F3 F1 F2 F1 F1 F2 F2	Plot C F1 F1 F3 F1 F2 F2 F3 F2 F3	Plot D F3 F3 F1 F2 F2 F1 F2 F1 F3	Plot E F1 F2 F3 F3 F2 F2 F1 F3 F1	Plot F F1 F2 F3 F1 F2 F3 F2 F1 F3
Plot G F1 F2 F2 F2 F3 F3 F1 F1 F3 Application rate= 160 tonnes ha^{-1}	Plot H F2 F2 F1 F2 F1 F1 F3 F3 F3	Plot I F2 F2 F3 F1 F1 F3 F1 F2 F3	Plot J F2 F2 F3 F3 F3 F1 F1 F1 F2	Plot K F3 F1 F2 F2 F1 F3 F1 F2 F3	Plot L F3 F2 F2 F1 F1 F3 F3 F2 F1
Plot M F3 F2 F2 F1 F1 F1 F3 F2 F3 Application rate= 0 tonnes ha^{-1}	Plot N F2 F1 F1 F2 F1 F2 F3 F3 F3	Plot O F3 F2 F3 F3 F1 F2 F2 F1 F1	Plot P F1 F1 F3 F1 F3 F2 F2 F3 F2	Plot Q F3 F1 F2 F2 F1 F3 F3 F1 F2	Plot R F3 F2 F1 F2 F1 F2 F3 F3 F1

sub-plots within all plots: F1 - zero fertiliser (control); F2 - N at 3 g m^{-2}, P at 3.5 g m^{-2}, K at 3.7 g m^{-2}; F3 - N at 7.5 g m^{-2}, P at 8.8 g m^{-2}, K at 9.3 g m^{-2}. In November 1992, six *Quercus robur* whips (mean height, 0.5 m) were planted within each sub-plot to give 54 trees per substrate/fertiliser treatment per row. Each tree was allocated an identification code, and in January 1993 the girth (at 80 mm height) and height (to the base of the leading bud) of each was measured. These measurements were repeated during November 1993 and November 1994, to provide data on growth increments for all trees over this two year period.

Spoil samples from the plots were air dried and then sieved through a 2 mm mesh to remove most large fragments of shale and coal. Analytical techniques described in Allen (1989) (see Appendix 3. 1) were used to determine the pH, and extractable P, K and Ca concentration of each spoil sample. Nitrogen concentration was omitted, because significant amounts of fossil nitrogen are found in colliery spoil; this makes any accurate determination of nitrogen an extremely difficult procedure.

At Gillars Green, the three 9 x 9 m plots established for the herbaceous woodland species introductions (see Chapter 3) were also used for woody species introductions. As no attempt was made to alter spoil pH within plots at this site, treatments consisted of fertiliser applications to sub-plots at the same rates for the tree plots at Bold Moss. Within each of the nine 3 x 3 m sub-plots per plot, six whips (mean height, 0.5 m) of *Acer campestre* L., *Frangula alnus* Miller and *Sorbus aucuparia* L. were planted in March 1993, to give a total of 54 trees of each species per fertiliser treatment. As at Bold Moss, each tree was allocated an identification code, and girth and height measurements taken during March 1993. These measurements were repeated during November 1993 and December 1994 to provide data on growth increments for all trees over this two year period.

5. 2. 2 Glasshouse experiments

Pot experiments were undertaken to examine the effects of fertiliser application on tree growth under controlled conditions. Colliery spoil from the Bold Moss site was collected during March 1994, then dried and sieved through a 5 mm mesh to remove most large fragments of shale and coal. Five samples of spoil were removed from the sieved spoil, and analysed to determine the pH values, and extractable P, K and Ca concentrations of the samples, using the methods previously described. The same volume of sieved spoil was placed into sixty pots (150 mm in diameter), and the pots placed into five blocks of twelve on benches in a glasshouse at the University of Liverpool. Seedlings of *Corylus* *avellana, Fagus sylvatica, Fraxinus excelsior* and *Pyrus pyraster* (L.) Burgsd. were grown from seed in February/March 1994, and grown individually for one month in a peat-based compost within 40 x 40 mm modules. In April 1994, four seedlings of each species were removed from the modules and randomly placed in three of the pots in each block. Each of the three fertiliser treatments used for the tree experiments at both Bold Moss and Gillars Green was in turn randomly applied to one of the three pots for each species per block. The pots were then randomly located within each of the five blocks, to give a total of 20 trees of each species per fertiliser treatment. All trees were planted in pots while at a similar height (c.70 mm) and with an equal number of leaves. In September 1994 the growth of all trees was assessed by measuring the height of individuals to the base of the leading bud, and by counting the total number of leaves per tree. For the 1995 growing season the pots were removed from the glasshouse and placed in a polythene tunnel at the University of Liverpool's Ness Botanical Gardens. The randomised block design was identical in this new location, and in December 1995 the heights of all the trees was measured to the base of the leading bud.

5. 2. 3 Data analysis

Data from both the Bold Moss and Gillars Green sites were analysed in the same way. For each individual tree, girth and height values from one monitoring period were subtracted from the values from the following monitoring period to provide measures of girth/height increase between growing seasons. Zero or negative values for girth/height increase were omitted from any further analyses. Probability values for girth/height increase of trees within treatments and blocks were calculated using a General Linear Model analysis of variance (Minitab, 1991). All data were first transformed (Log_{10}+1) to stabilise sample variance. At Bold Moss separate analyses were undertaken for data at each rate of substrate ameliorant application, as the experimental design precluded comparisons of tree performance between application rates. Separate analyses were carried out on data for each of the two growing seasons on both sites, as many trees were subject to vandalism (at Bold Moss) and/or grazing damage (at Bold Moss and Gillars Green) during the early part of 1993. All damaged trees were omitted from the analyses for the 1992-1993 growing season only.

Data obtained from the pot experiments were analysed using an analysis of variance (Minitab, 1991) to provide probability values for effects of fertiliser treatment and block on tree height/leaf number. All data were first transformed (Log_{10}+1) to stabilise sample variance. Leaf number was measured in 1994 only, as the 1995 measurements were undertaken during the winter.

5. 3 Results

5. 3. 1 Field experiments

Results from the spoil chemical analyses for the *Quercus robur* plots at the Bold Moss site are presented in Table 5. 1. At all three rates of spoil ameliorant application (substrate treatment), mean pH and mean extractable P, K and Ca concentrations were generally higher in plots to which crushed concrete had been added. For both substrate ameliorant types, mean pH for plots generally became progressively lower from the highest (320 t ha^{-1}) to the lowest (0 t ha^{-1}) substrate treatment, as did mean K concentration. Mean P and Ca concentrations were greatest in the 160 t ha^{-1} substrate treatment for plots of both substrate ameliorant types.

Probability values for analysis of variance comparing girth and height increases of *Quercus robur* with fertiliser treatment, substrate ameliorant type and block for the two growing seasons at the Bold Moss site, are shown for each substrate ameliorant application rate in Tables 5. 2 and 5. 3. In the 1992-1993 growing season, statistically significant differences were observed for increases in tree height in relation to fertiliser treatment, but only in the 160 t ha^{-1} substrate treatment. The graphs presented in Figure 5. 2 show that the mean tree height increase in this substrate treatment was greater in the F3 fertiliser treatment than in the other two fertiliser treatments, where the mean increases were similar. In contrast, mean tree height increase in the 0 t ha^{-1} substrate treatment was progressively greater from the F1 to F3 fertiliser treatments, while increases in the 320 t ha^{-1} treatment showed little variation. Mean girth increase showed similar, though weaker, trends in relation to fertiliser treatment for this same period. Reference to Figure 5. 3 shows that there was a slight difference in mean girth and height increase in relation to substrate ameliorant type during this period, increases being marginally higher for trees planted on crushed limestone. Mean height increase was also slightly greater on both substrates at the higher application rate of 320 t ha^{-1}.

In the 1993-1994 growing season, trees at Bold Moss showed a statistically significant difference in girth increase in relation to fertiliser treatment in both the 160 and 320 t ha^{-1} substrate treatments, and in height increase in relation to substrate ameliorant type (in the 160 t ha^{-1} treatment only) (Table 5. 3). Statistically significant differences were also

Table 5. 1 Mean pH and extractable mineral nutrient concentration - mg kg^{-1} air dry soil - (with sample standard errors) for *Quercus robur* experimental plots, Bold Moss Tip, St. Helens.

N.B. Where CC = Crushed concrete and LST = Crushed limestone.

PLOT	**A**	**B**	**C**	**D**	**E**	**F**	**G**	**H**	**I**	**J**	**K**	**L**	**M**	**N**	**O**	**P**	**Q**	**R**
Appl.^{n} rate	**320 t ha^{-1}**						**160 t ha^{-1}**						**0 t ha^{-1}**					
Substrate	**CC**	**LST**	**CC**	**LST**	**CC**	**LST**	**CC**	**LST**	**CC**	**LST**	**CC**	**LST**	**NO SUBSTRATE ADDITIONS**					
Mean pH	6.7	5.8	6.6	6.0	6.9	4.6	6.7	4.7	6.0	4.2	5.0	4.0	3.8	3.3	3.3	3.3	3.3	3.1
(SE)	±0.19	±0.68	±0.18	±0.55	±0.09	±0.56	±0.12	±0.59	±0.16	±0.45	±0.47	±0.46	±0.22	±0.06	±0.06	±0.03	±0.07	±0.04
Mean P (mg/kg)	2.68	0.94	1.92	0.08	5.73	0.64	7.75	1.77	0.65	0.95	3.59	2.17	1.28	0.46	0.40	1.68	0.58	1.45
(SE)	±1.15	±0.94	±1.22	±0.05	±2.58	±0.59	±3.98	±1.14	±0.28	±0.95	±0.89	±0.46	±1.28	±0.29	±0.26	±1.09	±0.39	±1.19
Mean K (mg/kg)	2.15	1.57	2.31	1.99	4.61	0.54	1.13	1.62	2.32	2.27	1.74	0.23	1.38	0.00	0.52	0.37	0.00	0.00
(SE)	±0.41	±0.67	±0.55	±0.24	±0.49	±0.48	±0.26	±1.01	±0.45	±0.58	±0.71	±0.23	±0.85	±0.00	±0.36	±0.33	±0.00	±0.00
Mean Ca (mg/kg)	226.6	80.7	178.2	146.4	146.4	85.8	255.6	147.7	191.5	114.5	168.6	57.4	46.1	36.44	48.31	17.56	41.67	28.5
(SE)	±11.0	±34.4	±48.8	±11.5	±60.9	±17.6	±22.3	±45.8	±27.3	±26.9	±16.6	±16.1	±16.2	±7.31	±16.5	±1.6	±6.17	±14.5

Table 5. 2 Probability values for analysis of variance comparing height and girth increase of *Quercus robur* with fertiliser treatment, substrate and block 1992-1993, Bold Moss Tip, St. Helens. (* denotes $P<0.05$, ** $P<0.01$, *** $P<0.001$).

		SOURCE OF VARIANCE			
GROWTH MEASURE	**RATE OF SUBSTRATE APPLICATION (t/ha^{-1})**	**Fertiliser**	**Substrate**	**Fertiliser*Substrate**	**Block**
HEIGHT INCREASE	0	0.427	-	-	0.655
	160	0.010**	0.065	0.311	0.162
	320	0.876	0.167	0.453	0.288
GIRTH INCREASE	0	0.374	-	-	0.108
	160	0.191	0.205	0.292	0.315
	320	0.457	0.468	0.309	0.253

Table 5. 3 Probability values for analysis of variance comparing height and girth increase of *Quercus robur* with fertiliser treatment, substrate and block 1993-1994, Bold Moss Tip, St. Helens. (* denotes $P<0.05$, ** $P<0.01$, ***$P<0.001$).

		SOURCE OF VARIANCE			
GROWTH MEASURE	RATE OF SUBSTRATE APPLICATION (T/Ha^{-1})	**Fertiliser**	**Substrate**	**Fertiliser*Substrate**	**Block**
HEIGHT INCREASE	0	0.374	-	-	0.002**
	160	0.565	0.026*	0.342	0.056
	320	0.118	0.546	0.884	0.240
GIRTH INCREASE	0	0.091	-	-	0.000***
	160	0.017*	0.298	0.187	0.043*
	320	0.000***	0.480	0.616	0.237

Figure 5. 2 Measures of tree growth in relation to fertiliser treatment at each substrate application rate (pooled data from crushed concrete and crushed limestone plots in each fertiliser treatment for each rate of substrate application), 1992 - 1993/1993 - 1994, Bold Moss Tip, St. Helens.

N.B. Fertiliser treatment (F LEVEL): 1= zero, 2= low and 3= high fertiliser treatment.
SUBSTRATE APPLICATION: 0= 0 t ha^{-1}, 160= 160 t ha^{-1} and 320= 320 t ha^{-1} (crushed concrete/limestone).

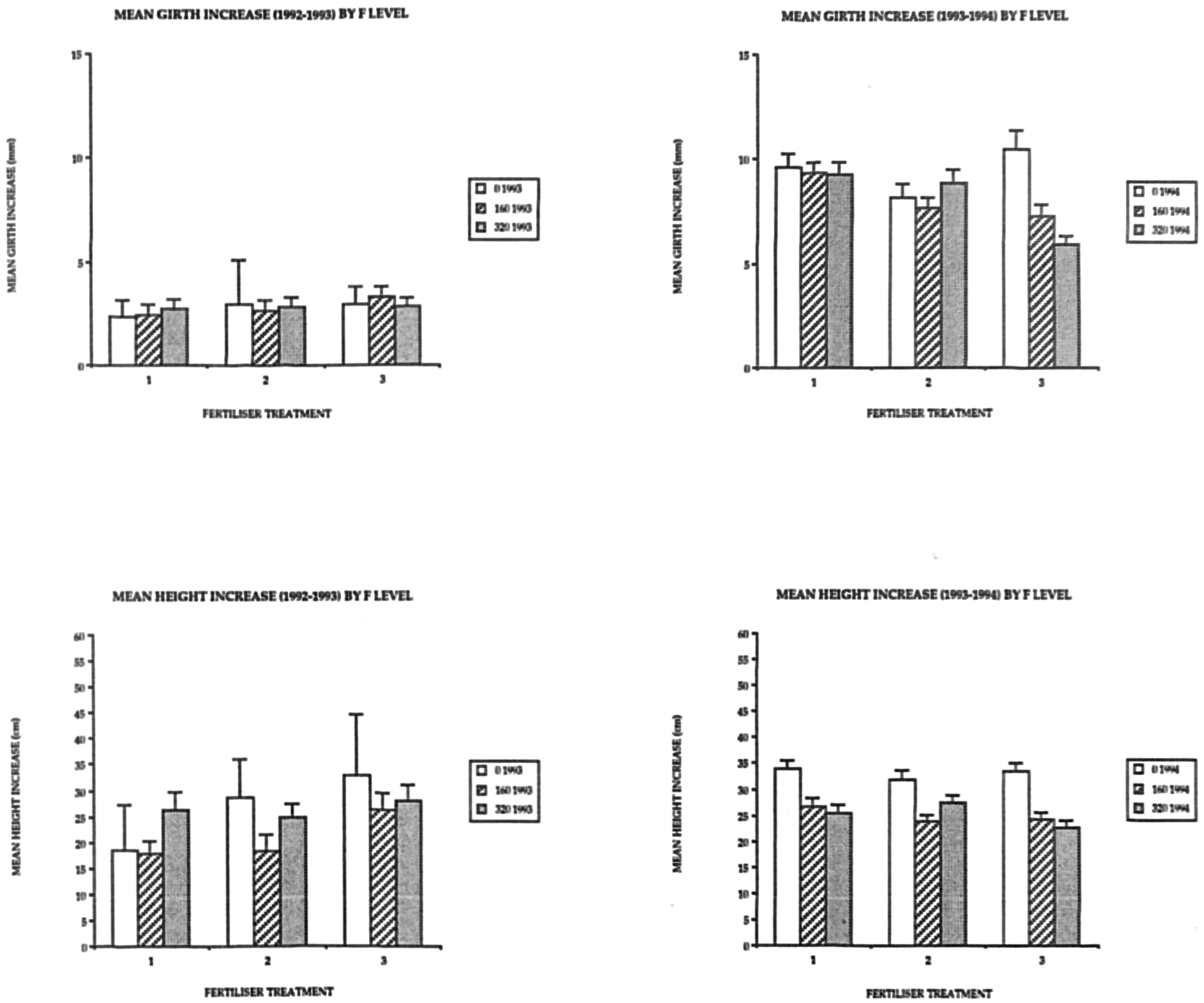
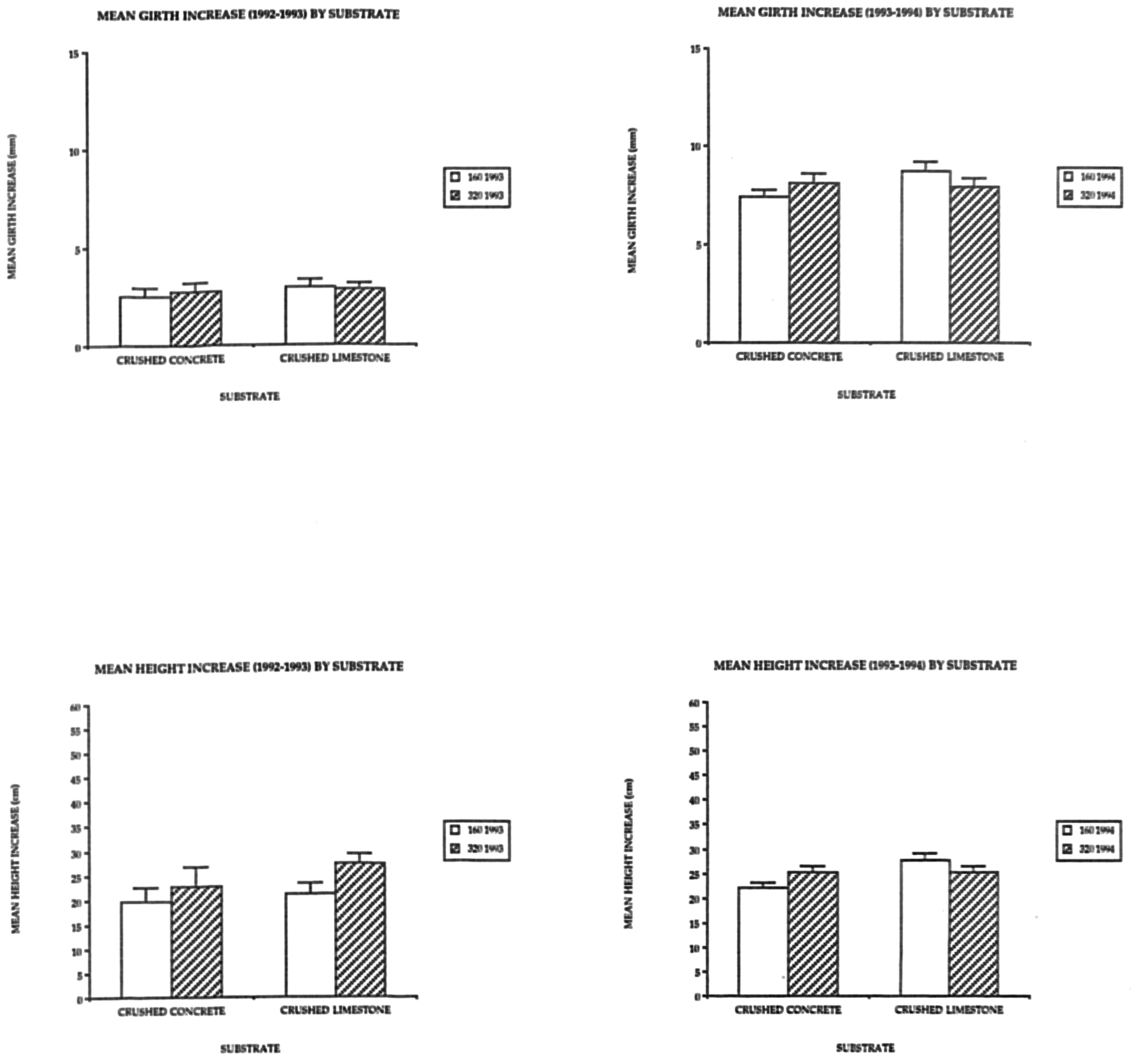


Figure 5. 3 Measures of tree growth in relation to substrate type and application rate, 1992-1993/1993-1994, Bold Moss Tip, St. Helens.



seen between blocks for girth and height increase in the 0 t ha^{-1} (control) substrate treatment, and for girth in the 160 t ha^{-1} treatment. Mean girth increase was lowest in the F3 fertiliser treatment in both the 160 and 320 t ha^{-1} substrate treatments during this period. They were only greater in the F3 fertiliser treatment in the 0 t ha^{-1} (control) substrate ameliorant treatment (Figure 5. 2). Mean height increase at all substrate ameliorant rates showed little variation in relation to amount of fertiliser application. Both mean girth and mean height increase were greater for trees growing on crushed limestone, but in the 160 t ha^{-1} substrate treatment only.

At Gillars Green, *Acer campestre* and *Sorbus aucuparia* showed statistically significant differences in girth increase in relation to fertiliser treatment during the 1993-1994 growing season (Table 5. 4). Height increase for *Acer campestre* was also significantly different in relation to fertiliser treatment, but only for the 1992-1993 growing season. Reference to Figure 5. 4 shows that for *Acer campestre* both mean girth increase during the 1993-1994 growing season, and mean height increase during the 1992-1993 season, was greater in each increasing fertiliser treatment. For *Sorbus aucuparia* during the following growing season, mean girth increase was higher in both the F2 and F3 fertiliser treatments, though the highest rate was seen in the F2 fertiliser treatment. Individuals of *Frangula alnus* achieved slight, though progressively greater, mean girth and height increases in each increasing fertiliser treatment during the 1992-1993 growing season, but by the following year these differences were no longer apparent.

Statistically significant differences in either girth or height increase in relation to block occurred for all species in at least one of the growing seasons. Height increase was significantly different between blocks for all three species during the 1992-1993 growing season, and for *Acer campestre* and *Sorbus aucuparia* in the following year. Girth increase was significantly different for *Acer campestre* in both years, and for *Sorbus aucuparia* in the 1992-1993 season. The graphs in Figure 5. 5 show that mean girth increase for all three species was greatest in block three during the 1992-1993 growing season. Mean height increase for *Acer campestre* and *Frangula alnus* was also greatest in block three during this period, though *Sorbus aucuparia* showed an opposite trend, with a greater increase in blocks one and two. By the end of the 1993-1994 growing season, both *Acer campestre* and *Frangula alnus* showed a progressively greater mean girth increase from block one to block three, while little variation was seen in mean height increase for these two species over the same period.

Table 5. 4 Probability values for analysis of variance of height and girth increase in response to fertiliser treatment and block of three tree species, 1992-1993/1993-1994, Gillars Green Wood, St. Helens. (* denotes P<0.05, ** P<0.01, *** P<0.001).

N.B. A = *Acer campestre*, F = *Frangula alnus*, S = *Sorbus aucuparia*.

		SOURCE OF VARIANCE					
		Fertiliser			**Block**		
GROWTH MEASURE	**SPECIES/YEAR**	**A**	**F**	**S**	**A**	**F**	**S**
HEIGHT INC.	1992-1993	0.047*	0.956	0.317	0.002**	0.004**	0.009**
	1993-1994	0.198	0.204	0.146	0.003**	0.511	0.035*
GIRTH INC.	1992-1993	0.164	0.686	0.423	0.006**	0.186	0.001***
	1993-1994	0.021*	0.531	0.020*	0.001***	0.230	0.061

Figure 5. 4 Measures of tree growth performance in relation to fertiliser treatment, 1992-1993/1993-1994, Gillars Green Wood, St. Helens.

N.B. Fertiliser treatment (F LEVEL): 1= zero, 2= low and 3= high fertiliser treatment.

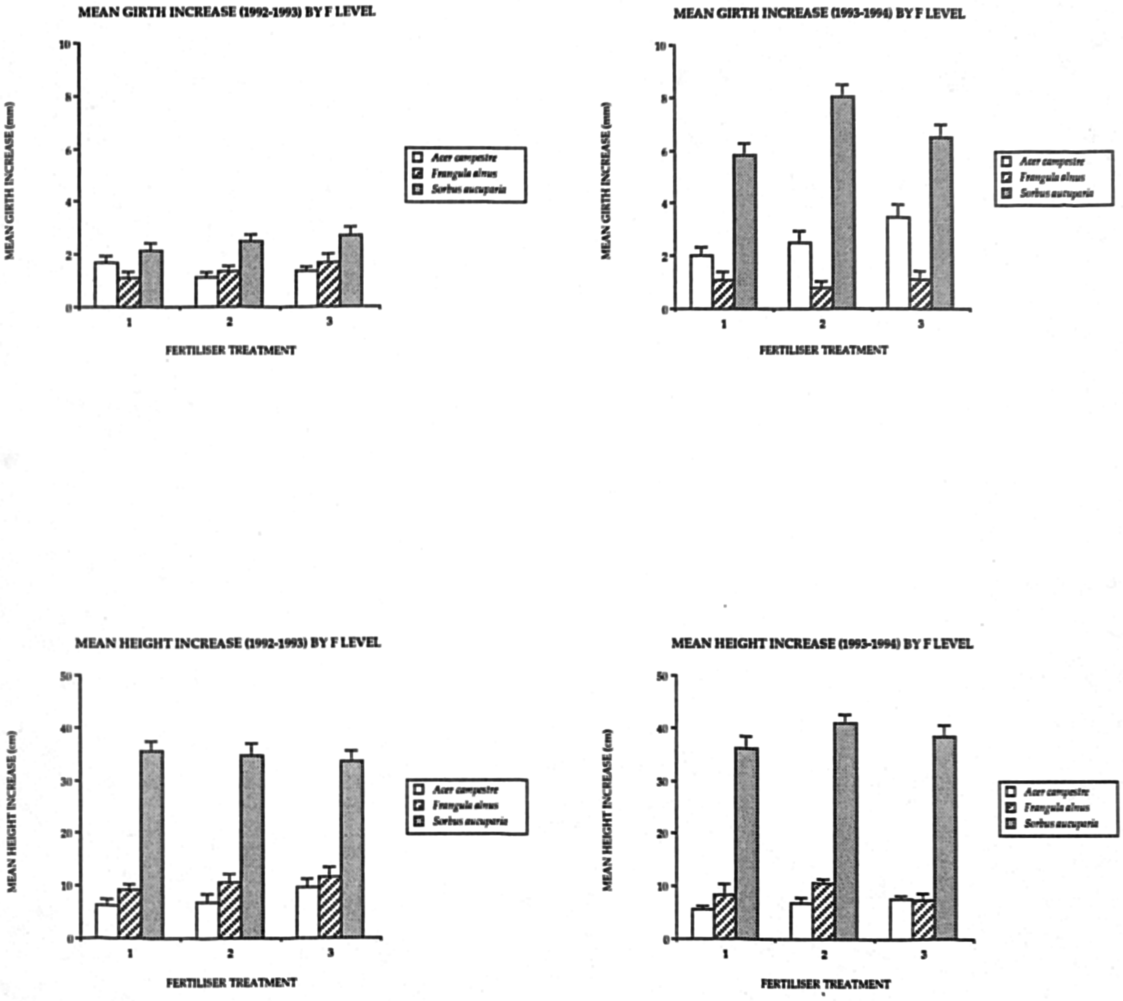
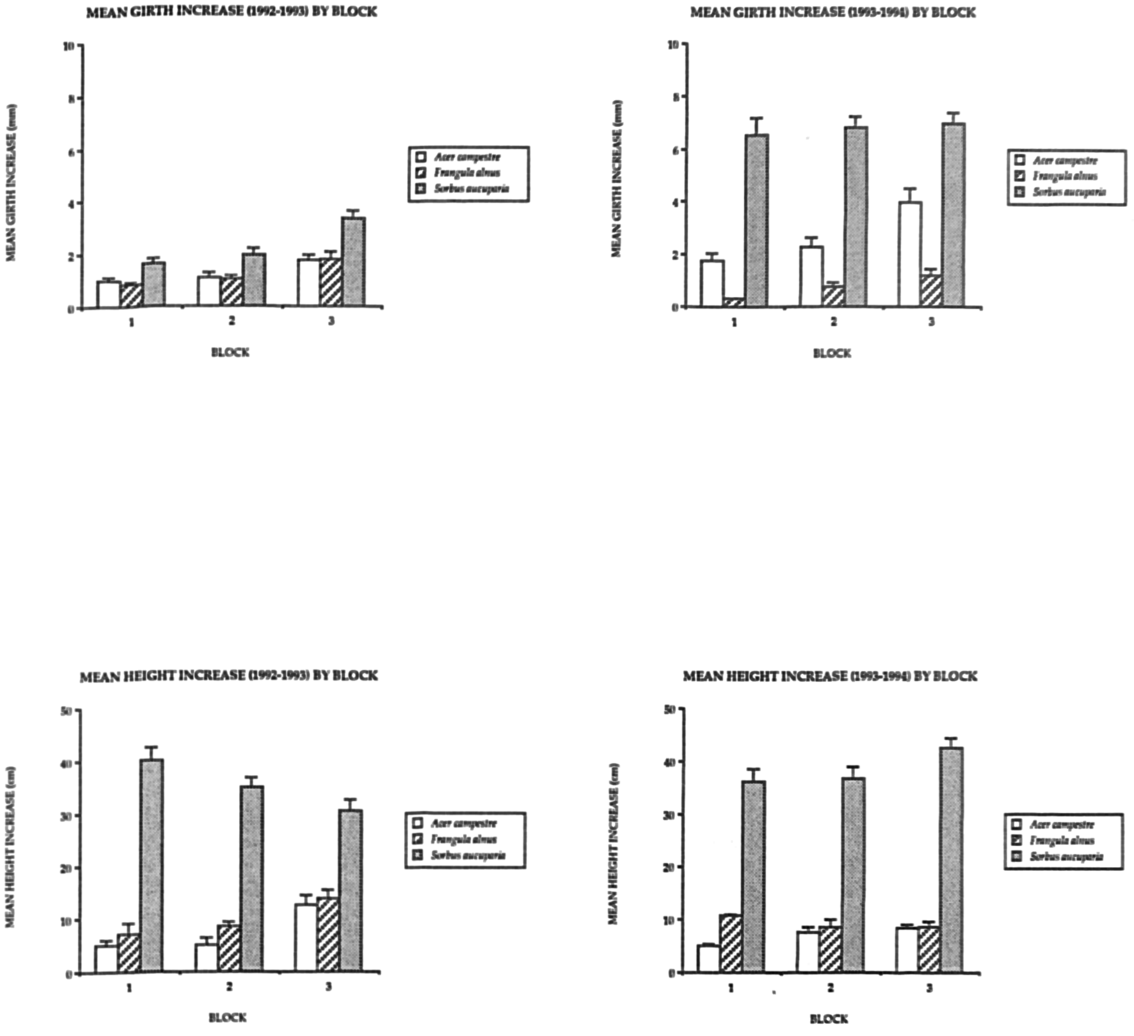


Figure 5. 5 Measures of tree growth performance in relation to block, 1992-1993/1993-1994, Gillars Green Wood, St. Helens.



5. 3. 2 Glasshouse experiments

The mean pH value for the spoil used in the pot experiments was pH 6.0 (± 0.02). The mean extractable mineral nutrient concentrations were, P 4.87 (±0.40) mg kg^{-1}, K 4.20 (±2.71) mg kg^{-1} and Ca 90.09 (±37.4) mg kg^{-1}. Probability values for the analysis of variance assessing the effect of fertiliser treatment and block on measures of tree performance are shown in Tables 5. 5 and 5. 6. In the 1993-1994 growing season (Table 5. 5), statistically significant differences occurred in both height and leaf number for *Fraxinus excelsior* and *Pyrus pyraster* in relation to fertiliser treatment. Leaf number was also statistically significantly different between blocks for two species (*Corylus avellana* and *Pyrus pyraster*) during this period. By the end of the 1994-1995 growing season tree height was significantly different between fertiliser treatments for individuals of *Corylus avellana, Fagus sylvatica* and *Fraxinus excelsior,* and there was a statistically significant difference in height between blocks for *Pyrus pyraster.* Reference to Figure 5. 6 shows that by the end of the 1993-1994 growing season, mean height and leaf number in *Fraxinus excelsior* and *Pyrus pyraster* were progressively greater in each increasing fertiliser treatment, while the values for the other two species showed little variation. By the end of the 1994-1995 growing season, the mean heights of *Corylus avellana, Fagus sylvatica* and *Fraxinus excelsior* were greatest in the F3 fertiliser treatment.

5. 4 Discussion

5. 4. 1 Field experiments

Growth of *Quercus robur* at Bold Moss was influenced by several different factors over the two growing seasons. In the first year (1992-1993) fertiliser application enhanced mean girth and height, particularly in the 0 and 160 t ha^{-1} substrate ameliorant treatments. By the end of the second year (1993-1994) however, these fertiliser effects had disappeared or in some cases had reversed. In particular, the mean girth increase of trees at the 160 and 320 t ha^{-1} substrate treatment was significantly lower in the F3 fertiliser treatment. This relationship is the opposite of that expected for trees growing on a site where low mineral nutrient availability is likely to limit tree growth, and this is difficult to explain. At both the 160 and 320 t ha^{-1} substrate treatments, crushed concrete/limestone addition did not increase spoil pH to levels > pH 7, that could have lead to increased fixation of the phosphate content of the added fertilisers (Brady, 1990). Neither were existing spoil P and K concentrations prior to the fertiliser application associated with any of the observed differences in girth increase. Whilst concentrations of both P and K were

Table 5. 5 Probability values for analysis of variance of the growth of four tree species grown in pots, in response to fertiliser treatment and block, 1993-1994. (* denotes $P<0.05$, ** $P<0.01$, *** $P<0.001$).

N.B. C = *Corylus avellana*, FG = *Fagus sylvatica*, FX = *Fraxinus excelsior*, P = *Pyrus pyraster*.

	SOURCE OF VARIANCE							
	Fertiliser				Block			
SPECIES	C	FG	FX	P	C	FG	FX	P
GROWTH MEASURE								
HEIGHT	0.110	0.851	0.001***	0.001***	0.108	0.268	0.903	0.803
LEAF NO.	0.196	0.757	0.001***	0.001***	0.039*	0.755	0.088	0.001***

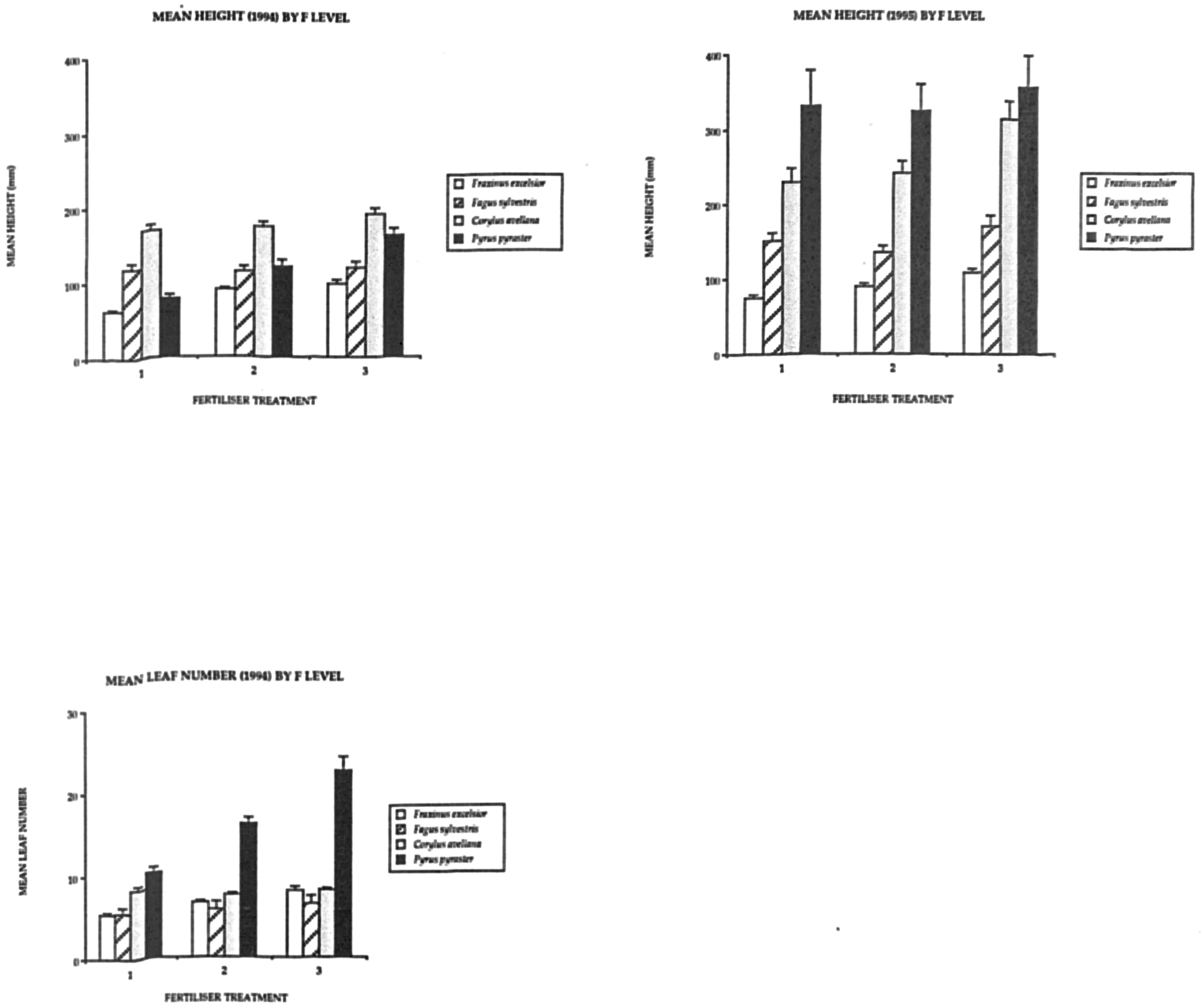
Table 5. 6 Probability values for analysis of variance of tree height for four tree species grown in pots, in response to fertiliser treatment and block, 1994-1995. (* denotes P<0.05, ** P<0.01, *** P<0.001).

N.B. C = *Corylus avellana*, FG = *Fagus sylvatica*, FX = *Fraxinus excelsior*, P = *Pyrus pyraster*.

	SOURCE OF VARIANCE							
	Fertiliser				**Block**			
SPECIES	**C**	**FG**	**FX**	**P**	**C**	**FG**	**FX**	**P**
HEIGHT	0.033*	0.040*	0.001***	0.884	0.141	0.060	0.993	0.017*

Figure 5. 6 Measures of growth performance of four tree species grown in pots in relation to fertiliser treatment, 1993-1994/1994-1995.

N.B. Fertiliser treatment (F LEVEL): 1= zero, 2= low and 3= high fertiliser treatment.



generally higher in plots to which crushed concrete had been added, trees in crushed concrete plots at both the 320 and 160 t ha^{-1} application rates performed less well than trees in crushed limestone plots. The generally higher pH values in plots to which crushed concrete had been added (at both application rates) in comparison with crushed limestone may have been due to the particle size distribution of the crushed limestone used, as this is known to partly determine the ability of this material to raise spoil pH and thus, in some situations, promote plant growth (Gemmell, 1981). This hypothesis is speculative and was not however tested any further.

Though found on soils with a wide range of pH values (Jones, 1959), *Quercus robur* most commonly grows on soils of pH 5 or less (Grime *et al.*, 1988), and is a species that may be expected to have a reduced growth rate in base-rich soils. At Bold Moss, the mean girth and height increase for trees on untreated spoil during two growing seasons was generally equal to, or greater than, those for trees planted in substrates with added crushed limestone or concrete. This indicates that the application of crushed concrete/limestone did not significantly enhance tree growth. Since the application of fertiliser to sub-plots on untreated spoil produced a greater mean height increase in the 1992-1993 growing season, and a greater mean girth increase in the 1993-1994 growing season, it appears that low mineral nutrient availability served to limit tree growth where spoil ameliorants were not added. Limitations on the analysis of the data obtained from the experiment, imposed by the original experimental design of the substrate ameliorant treatments, precluded any analyses of growth performance measures between amounts of added crushed concrete/limestone. This limitation restricted the interpretation of the experimental treatments on tree growth at Bold Moss.

At Gillars Green the three tree/shrub species chosen for planting were characterised by a range of habitat requirements, and it was expected that each species would respond differently to the fertiliser treatments, and also to the varying level of canopy shading. By the end of the 1992-1993 growing season, fertiliser application produced higher mean height increases for *Acer campestre* and *Frangula alnus*. This difference was only statistically significant for *Acer campestre* in the F3 fertiliser treatment. By the end of the 1993-1994 growing season however, only the growth of *Acer campestre* was enhanced by the application of fertiliser. In both growing seasons at Gillars Green, block effects exerted a much greater influence over relative growth, particularly for individuals of *Acer campestre* and *Sorbus aucuparia*. As spoil pH showed little variation between plots (blocks) (see Chapter 3), all being consistently below pH 4.0, spoil acidity could not account for the observed differences in tree growth between plots. Data on the availability of a number of potentially limiting resources (*sensu* Tilman, 1986) on the site also showed little variation between plots. For example, the mean of the mean P concentrations within plots ranged from only 5.82 (±1.29) mg kg^{-1} to 7.48 (±1.21) mg kg^{-1}, that of Ca from 16.18 (±0.80) mg kg^{-1} to 26.36 (±1.66) mg kg^{-1}, while figures for canopy shading above plots, determined through hemispherical photograph analysis (see Chapter 4), ranged from a mean of 12.33 (±1.45) % open sky in block 3 to a mean of 16.25 (±3.99) % open sky in block 2.

Of the three tree/shrub species used in the experimental work, *Frangula alnus* was probably the least tolerant of the high canopy shade within plots, as it is a species ..."strongly seral in habitat preferences, avoiding all but light shade" (Godwin, 1943). Both *Acer campestre* and *Sorbus aucuparia* are in contrast known to be highly tolerant of shade (Jones, 1945; Grime *et al.*, 1988), though *Acer campestre* is most commonly associated with soils of high base status (Jones, 1945; Rodwell, 1991). *Sorbus aucuparia* occurs most frequently on acidic soils (pH 3-5) (Grime *et al.*, 1988), and was therefore highly suitable for introduction to an established woodland on colliery spoil. It is interesting to note that of the three tree species, *Sorbus aucuparia* achieved the greatest growth (girth/height increase) in both growing seasons.

5. 4. 2 Glasshouse experiments

All of the four species grew successfully over the two year monitoring period. *Pyrus pyraster* was included in the experiment to determine the suitability of colliery spoil as a medium for growing species not usually included in standard woodland planting mixtures on such sites. It must be noted however, that it is equally likely that the seed obtained was that of the closely-related *Pyrus communis* L., as both species are often indistinguishable from one another (Stace, 1991).

In the absence of fertiliser application, mean height increase during 1994 was greatest for individuals of *Corylus avellana* and *Fagus sylvestris*, indicating higher natural relative growth rates for these two species during the early stages of establishment and growth. Increase in tree height and leaf number with successive fertiliser treatments for *Fraxinus excelsior* and *Pyrus pyraster* during this same period, probably reflected the natural association of these two species with relatively base rich, fertile soils (Wardle, 1961; Rodwell, 1991). By the end of 1995 however it was apparent that fertiliser application had caused a significant increase in tree height for *Corylus avellana, Fagus sylvatica* and *Fraxinus excelsior*, which suggests that low mineral nutrient availability may limit the growth of young trees on the Bold site. The significant block effects on *Pyrus pyraster* growth in both years cannot be explained by variation in spoil chemical characteristics within blocks, or by environmental gradients within the experimental areas, as they do not occur for the other species.

5. 5 Conclusions

Several native broadleaved tree species with a range of different ecological requirements were grown successfully on colliery spoil, at two sites in the St. Helens area, and in a glasshouse. At Bold Moss, the growth of *Quercus robur* transplants was not significantly enhanced by an increase in spoil pH using addition of crushed concrete/limestone, though the addition of mineral nutrients did increase tree growth on spoil where pH remained unaltered. *Quercus robur* is commonly associated with soils of low pH and fertility, and it appeared to be particularly suited to growth on colliery spoil at this site. Growth of the three tree/shrub species at Gillars Green also showed a varying response to fertiliser application, though *Sorbus aucuparia* appeared to be most suited to the low pH and light levels found at the site. Trees of three species grown in pots did show a clear increase in growth in response to fertiliser application over a period of two growing seasons. It is possible that adequate growth rates may be achieved at both Bold Moss and Gillars Green without the application of fertilisers. The mature woodland stand at the Gillars Green site, which may have originated as a result of natural seeding from a few surviving trees on the site, supports this conclusion. However, the long-term maintenance of woodland ecosystems on many colliery waste sites can probably only be achieved if there is sufficient nitrogen capital in the spoil. Aftercare for most tree species planted on such sites should therefore (in the short-term at least) include nitrogen and phosphorus applications, so as to achieve a sufficient 'working capital' for continued nutrient cycling and tree survival (Bloomfield *et al.*, 1982). Where the primary objective of woodland establishment is for nature conservation, it may be possible to accept rates of tree growth that would not be sufficient for timber/amenity uses, as long as careful monitoring of tree performance is undertaken on a regular basis.

The experiment at Gillars Green indicates that the diversification of the woody plant community is possible at this site. Though, unlike Gillars Green, the experiment at Bold Moss was not undertaken within existing areas of semi-natural vegetation, it is likely that suitable woody species could be successfully introduced into the areas of *Salix/Betula* scrub. During the period of study, natural colonisation of woody species was observed on both sites; at Bold Moss seedlings/saplings of *Crataegus monogyna* Jacq., *Sorbus aucuparia* and *Sorbus intermedia agg.*, and at Gillars Green seedlings of *Ilex aquifolium* L. and *Sorbus aucuparia* (D. Fee, personal observation). All of these species are commonly dispersed by a number of native bird species (Snow and Snow, 1988). Despite the role of many bird and mammal species as vectors of seed dispersal for a number of woody plant species (Grime *et al.*, 1988), it is unlikely that woody species diversity will be ever be particularly high on either of the experimental sites. The reason is that the sites are relatively isolated within a landscape matrix of agriculture and urban/industrial development. At Gillars Green for example, which is estimated to be c. 80+ years old, there are currently no more than seven naturally-established woody species present on the site (D. Fee, personal observation). Species introductions will therefore provide the best means of diversifying the woody component of the plant communities on both study sites, though species choice must be undertaken with regard to a number of important ecological criteria (see Chapter 6). Introduction of trees and shrubs could utilise direct seeding methods, which are generally less costly than methods that involve tree/shrub transplants. However, though successfully used in other woodland establishment trials (e.g. Putwain, Evans and Kerry, 1988), this method proved unsuitable on the Bold Moss site, where, following the establishment of a field trial in which tree and shrub seeds were sown, seed predation pressure by small mammals and birds was found to be so high that very few seedlings emerged, and none became established. Rabbit-proof fencing and other forms of protection (e.g. tree shelters) may help overcome this problem, though on many urban sites this too would be impracticable, owing to the high level of vandalism that often follows the placement of any obviously 'artificial' structure on these sites (D. Fee, personal observation).

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PART THREE: Conclusions

6. 0 GENERAL DISCUSSION AND CONCLUSIONS

6. 1 Ecological restoration and nature conservation

Traditional approaches to nature conservation in Britain have in the past focused on the protection of the most ecologically valuable habitats (e.g. Ratcliffe, 1977). Intensive human land-use in the form of farming, forestry, mineral extraction and road building has however occurred at such a scale, that many urban and industrial areas have little existing ecological value. In these areas the most ecologically valuable sites are often on 'damaged land' (Department of the Environment, 1996), which in some cases may contain greater levels of biodiversity than an equal area of countryside (Gilbert, 1989). Intensive land-use may however culminate in 'degraded landscapes' (Naveh, 1994), in which levels of natural variety, both of individual animal and plant species, and the habitats and landscapes in which they live, are significantly reduced. In many of these areas, ecologically valuable landscapes can only be developed through the use of appropriate ecological restoration techniques. These techniques may involve the *de novo* creation of plant and animal communities, or the transplantation/translocation, and/or enhancement/diversification, of existing communities. Ecological restoration cannot fully replicate established semi-natural habitats however (Newbold, 1989), and should not therefore be used as a means of justifying their further destruction. Nor should ecological restoration be undertaken without considering the existing biological or earth science interest of a site, which may in some cases already be significant (Department of the Environment, 1996). Overall, ecological restoration projects must be adequately planned and implemented, with long-term monitoring forming part of the post-restoration site management plan. In this way ecological restoration may be used effectively to enhance biodiversity at local, regional, and national levels.

6. 2 Restoration of native woodland communities

Supported by a range of existing planting grants, a high level of public demand for more trees in the landscape (Tree Council, 1996), and the establishment of both Community Forests and a new National Forest, the area of native broadleaved woodland in Britain will continue to increase well into the new millenium. Nature conservation guide-lines for existing and newly planted woodlands are now generally well established within the forestry industry (e.g. Forestry Commission, 1990), and both establishment and management grants under the Woodland Grant Scheme (WGS) allow for multi-purpose woodland management objectives, including nature conservation. However, as Peterken

(1986) states, ..."the creation of new woodland is more than mere tree planting". Wherever trees are planted, the resulting woodland will have less nature conservation value than those with naturally regenerated stands, which will be comprised of locally native species and genotypes (Peterken, 1995). For nature conservation purposes natural colonisation is therefore theoretically preferable to planting nursery stock. Planting may be more appropriate for the creation of woodlands with high amenity value however, where an early visual impact is often desirable (Harmer and Kerr, 1995). In either case, the creation of diverse woodland communities is dependent upon more than the establishment of appropriate canopy and shrub layer species alone. If the structure and composition of new native woodlands is to mimic that of existing semi-natural woodlands, herbaceous species, invertebrates and fungi characteristic of these woodlands must also be represented (Spencer, 1995). With such an approach it may be possible to reconstruct functioning woodland ecosystems, rather than areas of land merely covered by trees.

The publication of recommendations for the creation of new native woodlands based upon both ecological and silvicultural principles (Rodwell and Patterson, 1994) has provided a stimulus for the creation of woodlands with significant ecological value. In adopting this approach the Forestry Authority has also recognised (in theory at least) that it may in some situations be desirable to consider the introduction of suitable herbaceous field layer species. This approach has been further advanced through the development of a protocol for the selection of plant species for introduction into existing woodland plant communities (Hodge, 1995). Diversification of field layer communities in recent woodlands has already been undertaken in a number of studies (e.g. Francis *et al.*, 1992; Francis, 1993; Packham *et al.*, 1995; Humphrey, 1996). Of all the variables that determine the range of conditions required for individual species establishment, levels of shade (and hence canopy density and structure) have generally been found to be one of the most influential (Francis, 1993).

6. 3 Enhancing woodland plant communities on colliery spoil

In 1993 colliery spoil heaps in north west England covered 675 ha of land; a large proportion of this land is situated in and around the town of St. Helens on Merseyside (Department of the Environment, 1995). Landscapes dominated by such sites tend to limit the economic potential of an area, through reducing the desire for new housing and commercial development. Many of these sites are an important resource however, often representing a significant proportion of land in non-agricultural, urban or industrial use. They therefore provide opportunities for nature conservation, ecological restoration, recreation and amenity, and (in some cases) timber production (Moffat and McNeill, 1994). Colliery spoil is however characterised by a number of physical and chemical factors that serve to severely limit plant survival and growth. Where this is the case, both survival and growth may be enhanced by manipulating these physical and chemical characteristics, through various forms of substrate treatment. Alternatively, existing site characteristics may be used as a framework for determining which plants would be most suitable for introduction to a given area.

Where woodland develops on colliery spoil, either naturally or as an end-use following land reclamation, it is likely that the established plant community will be suited to colonisation by additional woodland plant species. Natural immigration to sites by many species is however unlikely, due to the (often) large distances from suitable sources of plant propagules (Ash, 1983). The extent to which regeneration niches are available for herbaceous woodland species in established vegetation on colliery spoil was therefore examined through species introductions at two sites in St. Helens. Experiments were also undertaken to determine the suitability of colliery spoil for the growth of a number of native tree and shrub species.

6. 3. 1 Discussion and implications of the experimental herbaceous species introductions

The two sites used for species introductions were introduced in Chapter 2. At both sites a number of herbaceous woodland/woodland edge species characteristic of semi-natural woodland in the region (NVC W10/W16 type woodland) were introduced as seed into the field layer of existing woody vegetation. At Bold Moss the canopy layer consisted of *Salix/Betula* scrub that was approximately twenty years old and of varying density (an early successional woodland habitat), while the field layer was mainly composed of a small number of grass and ruderal species. The Gillars Green site was dominated by a dense canopy of *Quercus/Betula*, which had probably first started to colonise the site towards the beginning of this century (a late successional woodland habitat). The field layer at this site was generally sparse, with only *Digitalis purpurea* and *Rubus fruticosus* agg. in some areas.

Mineral nutrient additions were made to some areas of colliery spoil at both sites, to determine the effects of enhanced mineral nutrient supply on species establishment, survival, flowering and seed production. These effects were examined in Chapter 3. At both sites plants continued to grow throughout the course of a two year monitoring period, though fewer species became established at Gillars Green. Higher concentrations of H^+

ions in the spoil appeared to be associated with a reduction in seed germination and a relatively low percentage plant establishment (recruitment) for many species. At Gillars Green, where spoil pH was uniformly low, some species were found to benefit, to a limited degree, from the addition of plant mineral nutrients. Where spoil pH varied, as at Bold Moss, establishment and survival was primarily dependent upon H^+ ion concentration in local spoil patches. An increase in both the proportion of flowering plants and number of flowers per plant at Bold Moss did show some association with increased mineral nutrient levels.

These results indicate that though spoil acidity can severely restrict the number of plant species suitable for woodland community restoration on untreated colliery spoil, vacant regeneration niches are likely to ensure that seed introductions provide an effective means of diversifying field layer plant communities. Where manipulation of spoil pH is undesirable or impracticable, care must be taken to ensure that species used in such introductions are suitably adapted to the often extreme range of chemical characteristics associated with colliery spoil, most particularly low pH and spoil chemical characteristics associated with extreme acidity. Whilst addition of plant mineral nutrients may be required to ensure adequate establishment and percentage survival at some sites, introductions made into existing vegetation may be less dependent upon additional mineral nutrient inputs; though subsequent flowering and seed production may be more so. *Hyacinthoides non-scripta* for example was found to be particularly suited to existing spoil conditions at both sites, being a species that is naturally frequent and abundant on soils of low mineral nutrient status and low pH (Grabham and Packham, 1983; Grime *et al.*, 1988). In practice therefore, initial introductions using a wide range of species with different ecological strategies could be used to 'fine-tune' further introductions in following years, to establish a community of species best suited to specific site conditions.

Success or failure of herbaceous species introductions also will be largely dependent upon the effects of competition from existing vegetation and levels of available light in the field layer. In Chapter 4 these effects were examined in relation to establishment and survival in the introduced herbaceous species at the Bold Moss site. A greater abundance of grasses and herbs was found to be associated with a reduction in the number of some of the introduced species, while areas of bare ground were associated with a greater plant population density of most species. The relative abundance of existing vegetation and bare ground was subsequently shown to be partly a function of the density of canopy cover within plots. By affecting PFD levels and sunfleck number and duration in the field layer, canopy density and architecture was thought largely to determine the number of sites in which plants could become successfully established, though extremely low pH levels in some areas were thought to limit the number of suitable sites still further.

Canopy layer effects on measures of population performance for many of the introduced species at Bold indicate that these characteristics should be used to determine the appropriate range of species for introduction into the field layer within existing woodland/scrub communities at this site. Though many herbaceous woodland species require additional factors, such as freedom from grazing and other forms of disturbance for sustained growth (Packham and Cohn, 1990), distinct woodland field layer assemblages are only, in the long-term, likely to develop in areas of moderate to heavy shade (Francis, 1993). The successional stage at which introduced herbaceous species may become successfully established in existing woodland partly depends upon the particular ecological requirements of the species used, particularly in relation to levels of available light, and hence competition, in the field layer. Packham *et al.* (1995) estimate that conditions suitable for field layer introductions would be reached after six years if using *Quercus robur* planted at 1 - 1.5 m centres, by which time field layer species with higher relative growth rates would have become less competitive. In existing vegetation, where stand species composition and stem density cannot be significantly altered, a wide range of herbaceous species could be used at the beginning of an introduction programme, and the successes and failures of each species could be used to determine future site management options. If this procedure was undertaken over a number of growing seasons, it could be adapted to take account of changes in vegetation characteristics in both the field and canopy layers.

In summary, the natural establishment and growth of woody and herbaceous species on many colliery sites in Merseyside indicates that a number of existing communities may be suitable for field layer enhancement/diversification, without any need for major reclamation works being undertaken. Reclamation may in fact serve to reduce the existing ecological value of many woodland habitats, and involve unnecessary financial costs. However, many sites will require additions of calcium and other spoil ameliorants, especially where vegetation needs to be established on bare spoil. These works may also be important for maintaining a good working relationship with local communities, many of the residents of which often like to see 'something being done' with a site that has stood 'derelict' for a number of years (D. Fee, personal observation). Where field layer enhancement/diversification is appropriate, all species that are introduced should be native to the local area and obtained from legal sources. Though spoil conditions may limit the number of species that can be successfully introduced, the communities that are created will be more 'naturalistic' and require fewer management (and therefore financial) inputs. Herbicide treatments and mulching with leaf litter, wood chip or similar materials may benefit species establishment and survival in some situations, though these are more likely to be required on substrates with higher mineral nutrient levels (Packham *et al.*, 1995). Indeed, the characteristically low plant available N and P levels in colliery spoil should be regarded as a positive advantage in the creation of certain plant communities, since more competitive species (with higher relative growth rates) are less likely to become established at a density that would seriously reduce the effectiveness of plant species introductions. Field layer introductions of herbaceous woodland species on some colliery sites may also be possible at an earlier successional stage than would be successful on more fertile sites. This hypothesis however needs to be investigated further.

6. 3. 2 Discussion and implications of the experimental woody species introductions

A number of native broadleaved tree and shrub species were successfully grown on colliery spoil at Bold Moss and Gillars Green, and also in a glasshouse. The results were presented in Chapter 5. Growth of *Quercus robur* transplants was not significantly enhanced by an increase in spoil pH at Bold Moss, and the addition of plant mineral nutrients only increased tree growth on spoil where existing pH levels remained unchanged. At Gillars Green *Sorbus aucuparia* appeared to be most suited to the low pH and light quanta found at the site, achieving higher absolute growth rates than *Acer campestre* and *Frangula alnus* in both the 1993 and 1994 growing seasons. Only the growth of *Acer campestre* was clearly enhanced by the application of mineral nutrients to plots at this site. In contrast, trees of three species, *Corylus avellana, Fagus sylvatica* and *Fraxinus excelsior,* grown in pots demonstrated increased growth rates in response to mineral nutrient additions over a period of two growing seasons.

Results from the experimental work suggest that untreated colliery spoil provides a suitable growing medium for the diversification of the woody plant communities at the Bold Moss and Gillars Green sites. This hypothesis was supported by observations of natural colonisation by bird-dispersed propagules of woody species at both experimental sites; *Crataegus monogyna, Sorbus aucuparia* and *Sorbus intermedia* agg. at Bold Moss, and *Ilex aquifolium* and *Sorbus aucuparia* at Gillars Green (D. Fee, personal observation). It is however unlikely that woody species diversity will ever be particularly high at either site, as both are relative isolated from the few areas of diverse semi-natural woodland in the region, and are situated within a landscape matrix of predominantly agricultural and urban/industrial development. Canopy layer enhancement/diversification at both sites will therefore best be achieved by means of transplanting nursery stock or by direct seeding (in circumstances where this may be successful) and appropriate aftercare. It would be most practicable to use species that are known to be tolerant of existing site conditions, as this would help ensure adequate survival and growth rates, and reduce capital costs. In circumstances where the primary objective of woodland enhancement/diversification is for nature conservation, it may be possible to accept rates of tree growth that would not be sufficient for timber/amenity uses, thus requiring fewer mineral nutrient additions. Moderate to severe predation of unburied tree/shrub seed at Bold Moss (D. Fee, personal observation) indicates that direct seeding at this site is only likely to be effective following burial under an adequate layer of spoil. If this could be achieved, the resulting woodland stands would be characterised by a relatively high probability of future-naturalness (Peterken, 1996), particularly if they were allowed to develop with little management intervention. Fencing and other forms of protection may be required for tree transplants in certain locations at Bold Moss due to the likelihood of grazing damage (D. Fee, personal observation), though the potential for vandalism at this site may make this approach impracticable.

Whenever tree/shrub species are to be introduced to a site, the species chosen should be characteristic of semi-natural woodland in the local area, and (preferably) derived from local seed sources. Planting should not be undertaken in areas with existing ecological value in an attempt to 'speed-up' successional changes, as this may lead to an overall loss in the ecological value of a site; where woodland is being created *de novo*, as in areas of bare spoil, this proviso need not necessarily apply. If enhancement/diversification is undertaken, the overall aim should be to work with successional changes, and to make species introductions at suitable spatial and temporal scales. At Bold Moss woody species could be introduced to a number of existing canopy gaps within areas of *Quercus/Betula* scrub. At Gillars Green introductions of shade-tolerant species could be undertaken now; natural canopy gaps or gaps created by selective felling of stand trees will however be required before less shade-tolerant species can achieve adequate growth rates at this site.

6. 4 Further considerations

6. 4. 1 Developing an ecological rationale for woodland creation and enhancement/diversification in urban/industrial areas

Experimental work on the enhancement/diversification of field and canopy layers in recent woodland (including that presented in this thesis) has shown that, in the short-term at least, herbaceous and woody species may be successfully introduced into established woodlands on a number of substrates (e.g. Francis *et al.*, 1992; Packham *et al.*, 1995). Less is known about the feasibility of introducing herbaceous woodland species before canopy closure (i.e. immediately following tree planting or natural colonisation by woody species), and whilst with adequate planning and management this approach may prove successful (Anderson, 1995), little evidence is available to support this hypothesis. Decision support systems relating to site-specific techniques and methods may greatly aid the process of species introductions (e.g. Francis, 1993). Ideally however, site-specific schemes should be undertaken as part of a much wider strategic evaluation of nature conservation, recreation and amenity objectives for all semi-natural communities of ecological importance in a region (Department of the Environment, 1996). Such an approach requires the formation of an ecological rationale for woodland creation and enhancement/diversification, and should involve the consideration of a number of factors, including:

1. Where woodland is being created *de novo*, the suitability of this rather than another community type for a particular site.
2. The range of measures required to ensure that new woodland is established in areas that will maximise their nature conservation and/or recreation and amenity value.
3. An assessment of the most appropriate woodland structure and species composition ('woodland type') for a given site.
4. An assessment of whether field layer enhancement/diversification is required.
5. An assessment of the ecological value of enhanced/diversified new woodlands in comparison to older semi-natural woodlands in the region.

In addressing points one and two above, it is evident that consideration must be given to 'landscape-ecological planning' (Forman, 1995) at a number of different temporal and spatial scales, so as to maximise the ecological value of the wider landscape. For example, new woodlands should only be established in elements of the landscape that ensure no significant loss of regional or local landscape character (Bell, 1995). Also, where nature conservation is a priority in woodland establishment, principles of ecological landscape design should be applied to ensure that ecologically diverse and dynamic landscapes are maintained and/or created (Dunnett, 1995). The establishment of ecological buffer zones and wildlife corridors should also be considered in any strategic determination of new woodland locations, particularly where there is potential to minimise the deleterious effects of fragmentation in existing semi-natural woodlands (Spellerberg, 1995).

Woodland creation must be undertaken only where existing ecological value will not be compromised; derelict urban/industrial land for example can contain valuable plant and invertebrate communities (Falk, 1995; Shepherd, 1995) which may be adversely affected by new woodland establishment. There is no doubt however that new native woodlands are likely to provide a number of ecological benefits, including an increase in woodland plant/animal populations and population resilience, the establishment of edge and mosaic habitats, and the restoration of degraded soils to greater levels of biological productivity (Peterken, 1995). New woodlands can also provide a range of important social and economic benefits (Hodge, 1995), including:

1. An improvement of human 'well-being', providing contact with nature in an aesthetic setting that enhances the attractiveness of the urban environment, and improves the quality of life.
2. Screening and noise reduction, trapping of particulate atmospheric pollution, and provision of summer shade and shelter.
3. The creation of recreation and amenity areas for a wide range of age groups.
4. Economic benefits, including the attraction of commercial investment and increased residential property values.
5. Opportunities for timber production and income generation (on suitable sites).

The relative value of the ecological, social and economic benefits associated with new woodlands is likely to influence woodland type at any given site, as well as affect approaches to field layer enhancement/diversification. Where nature conservation is the priority, a relatively large number of appropriate species may be suitable for introduction; where recreation and amenity is of greater importance, a creative conservation approach using fewer 'uncontentious' (i.e. common) species (Scott, 1995) may be more suitable. Woodland creation and enhancement/diversification programmes should also be undertaken with reference to existing nature conservation policies of local, regional and national importance. Planning and land-development powers give particular significance to policies at the local authority level; PPG Note 9 on Nature Conservation, the Local Agenda 21 programme, and the development of Local Biodiversity Action Plans are of particular relevance to woodland creation and enhancement/diversification schemes throughout Britain.

Ideally, the techniques and methods employed in woodland creation and enhancement/diversification at any site should be determined through an assessment of the complete range of ecological, social, economic and policy considerations outlined above. One method of undertaking such a strategic evaluation is through the use of Geographical Information Systems (GIS), because they allow data from diverse sources to be captured, integrated, manipulated and analysed in relation to spatially referenced geographical locations (Lee, 1995). Though at present underutilised for studies in ecology and conservation (Haines-Young, personal communication), GIS have recently proved suitable in a number of ecological applications, including an ecological field survey of northern England, and a study of the conservation of heathland fragments in Dorset (Cherrill, McClean, Lane and Fuller, 1995; Veitch, Webb and Wyatt, 1995). Research on the use of GIS in determining multiple land-use potential for urban sites with existing nature conservation value has also been undertaken (Freeman, 1995).

One possible approach to the use of GIS in the strategic assessment of woodland creation and enhancement/diversification schemes is exemplified by the maps presented in Figures 6. 1 to 6. 3, all of which show an area of 20 km^2 of mixed urban/industrial and agricultural land located to the south and west of Bold Moss Tip in St. Helens. Each map was created using ARC/INFO (ESRI, 1993), a widely-used system which combines a cartographic data system (ARC) with a relational database (INFO) (Briggs and Mounsey, 1989), and takes the form of a cartographic representation of data sets thought likely to influence approaches to future woodland creation and enhancement/diversification in the area. All data were digitised from information readily available in map form, to show the geographical distribution of existing land-use (Figure 6. 1), policy proposals within the St. Helens Metropolitan Borough Council Unitary Development Plan (Figure 6. 2), and Mersey Forest planting classifications (Figure 6. 3). Initially it was intended that INFO would be used to perform further analyses of the data sets, by overlaying (combining) data and determining both the location of sites most suited to new woodland establishment, and the relative importance of conservation and/or recreation and amenity objectives for each site. It soon became apparent however that the area chosen was too small to produce results other than those that were most obviously expected; i.e. that those areas currently classed as derelict industrial land were most suitable for future woodland establishment, being in non-agricultural/urban/industrial use, in the Green Belt, and in areas of the Mersey Forest where >30% woodland cover is proposed. Nevertheless, it is likely that GIS would prove valuable in the strategic assessment of woodland creation and enhancement/diversification at a larger scale, as this would allow the assessment of a

Figure 6.1 Existing land use

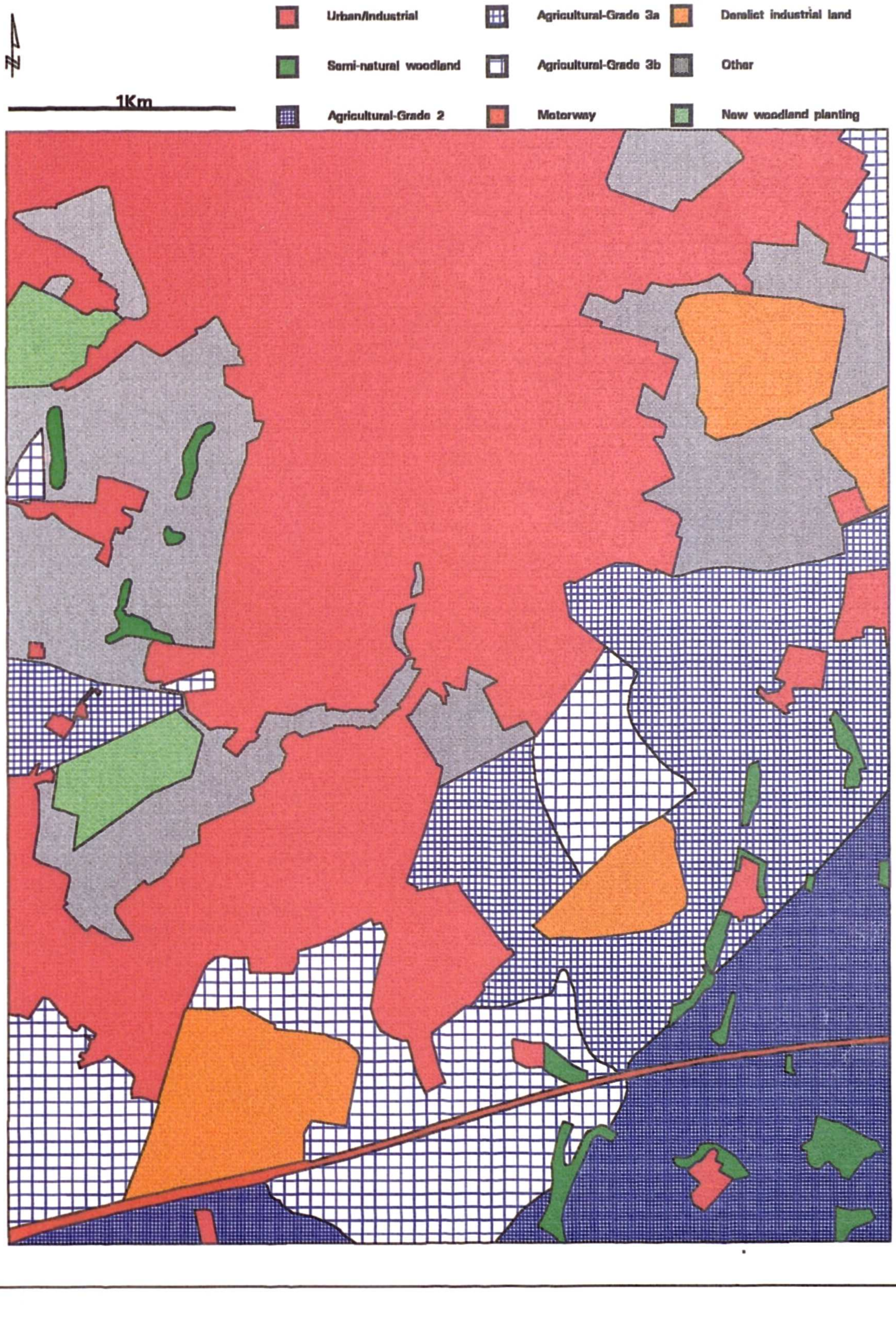


Figure 6.2 St.Helens MBC UDP policy proposals

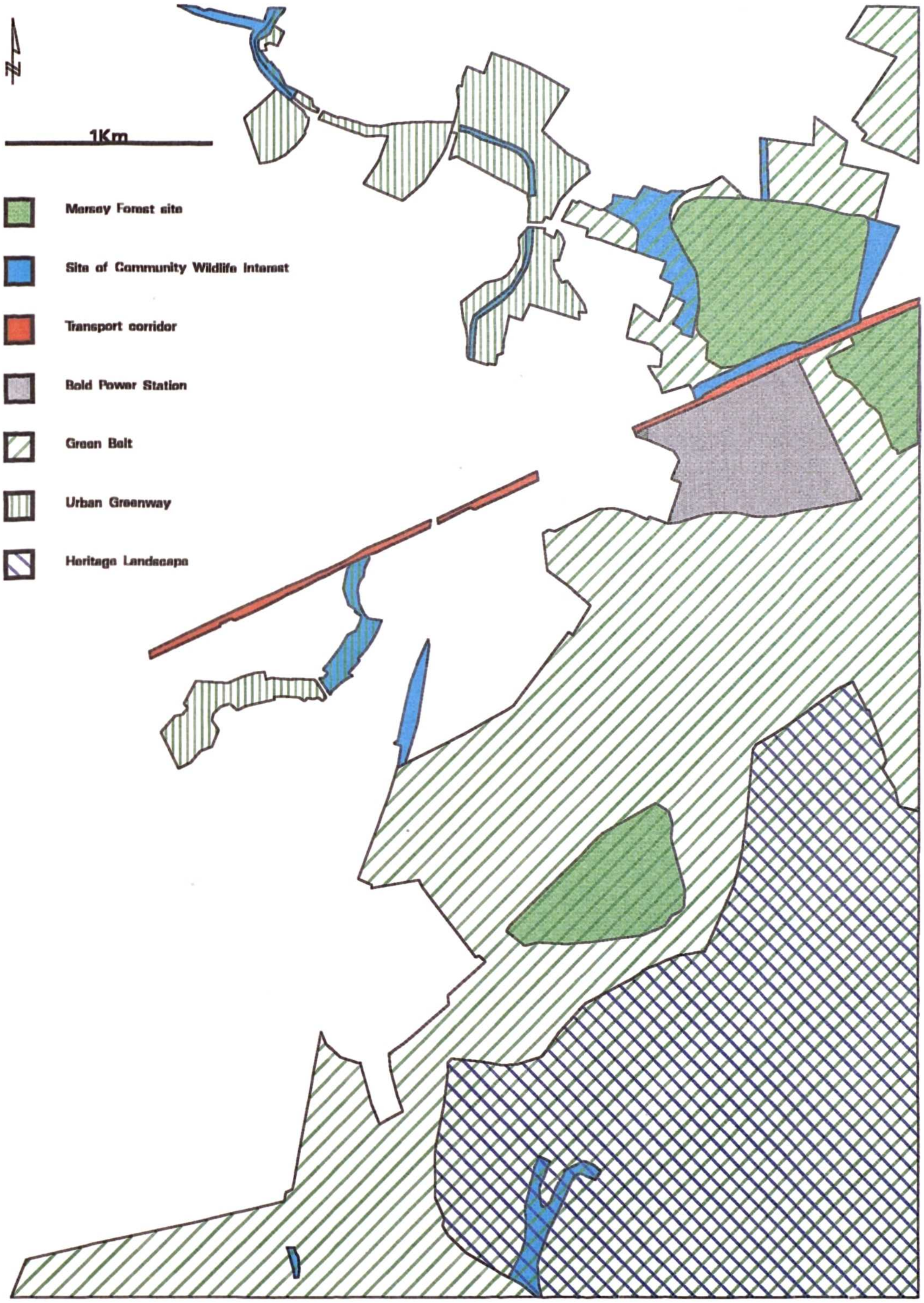
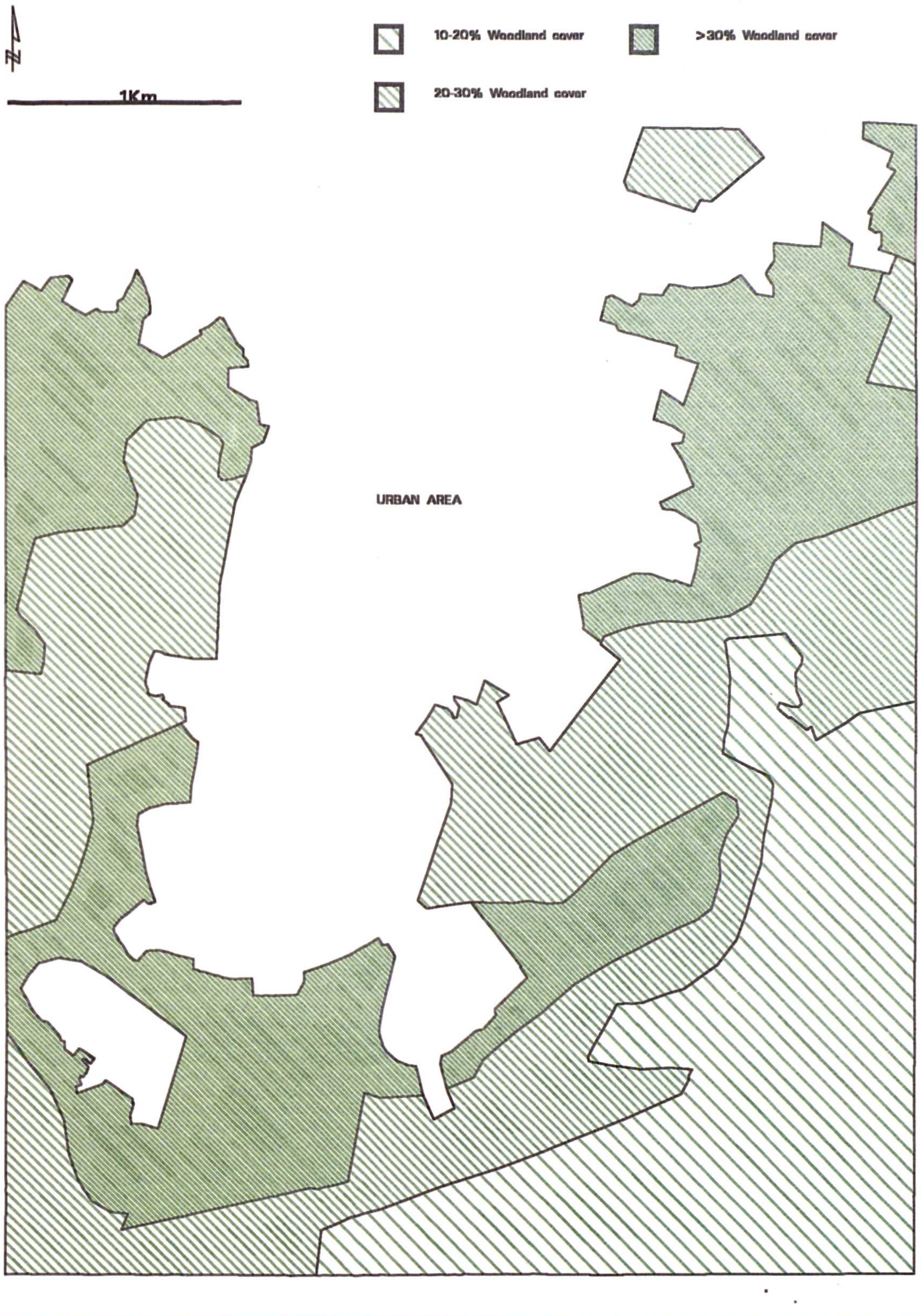


Figure 6.3 Mersey Forest planting classifications



much greater range of social, economic and ecological factors, which would in turn be more likely to show significant differences from one geographical area to another.

With adequate planning and appropriate long-term management and monitoring, woodland creation and enhancement/diversification techniques may be used to establish woodland plant communities of relatively high nature conservation value in many urban and industrial areas. For example, in the same area used for mapping the cartographic data in Figures 6. 1 to 6. 3, many of the existing semi-natural woodlands have a reduced nature conservation value which results from a combination of varying levels of human disturbance, isolation and extensive *Rhododendron* spp. cover. This makes new native woodlands potentially highly valuable in the local landscape. Work was undertaken in April/May 1996 - when the nature conservation value of fourteen semi-natural woodlands in the specified area was assessed by means of a scoring system based on criteria identified from Peterken (1981) and Kirby (1986) - to test the validity of this general proposition. Results from the survey (presented in Tables 6. 1 and 6. 2.) show that total nature conservation scores of existing semi-natural woodland in the area ranged from 188 to 22; many of the lower scoring woodlands being characterised by a combination of reduced plant species richness, small woodland size and high levels of *Rhododendron* spp. cover. Whilst it is not possible to place these scores in a broader context (the scoring system being specific to the characteristics of woodlands in the local area), they can be compared with predicted values for areas of new woodland in the same area. In comparison, estimated scores for new native woodlands established on the four areas of derelict industrial land in the area ranged from c. 109 to c. 267. These relatively high scores arise from the fact that all of the derelict sites are much larger than any existing woodland area, and could be allowed to develop a variety of natural structures from establishment onwards. Though this scoring exercise involves a degree of subjectivity, and is likely to apply insufficient weighting values to certain criteria, it does help demonstrate the high nature conservation value that could be derived from new woodlands in a landscape in which existing woodlands are few in number and/or of poor ecological value.

6. 4. 2 Historical, cultural and aesthetic factors

As stated by Mabey (1980), "woodlands are more...four-dimensional than perhaps any other kind of natural habitat". As such, approaches to woodland creation and enhancement/diversification should give some thought to the often considerable historical meaning that is associated with many existing semi-natural woodland habitats in Britain (Rackham, 1980). In addition to the range of issues that must be assessed in the

	CRITERIA CODE NUMBERS (see Table 6. 2)																	
Woodland Name	1	2	3	4	5	6	7	8	9	10	11	Positive Score	12	13	14	Negative Score	TOTAL	Rank Score
Plain	5	4	10	5	20	2	10	5	11	10	0	+82	-10	-6	-0	-16	66	12
Park Cottage	12	6	10	10	10	12	5	5	24	5	5	+104	-0	-6	-2	-8	96	6
Dog Kennel	11	6	10	5	10	14	5	5	27	5	0	+98	-70	-0	-2	-72	26	13
Ladies Walk	11	7	15	5	20	8	15	5	18	5	5	+114	-25	-2	-0	-27	87	8
Patch	7	4	10	10	10	10	10	5	18	5	5	+94	-10	-0	-0	-10	84	9
Old Hall	8	8	15	5	10	16	10	5	22	5	5	+109	-5	-10	-4	-19	90	7
Currant Dam	16	7	10	5	20	14	10	5	29	5	0	+121	-0	-10	-0	-10	111	4
Coney Green	34	18	15	10	20	40	10	5	27	5	0	+184	-0	-12	-0	-12	172	2
Lodge	14	4	15	5	20	18	10	5	26	5	0	+122	-10	-10	-0	-20	102	5
Duck	7	3	10	5	10	14	10	5	29	5	5	+103	-75	-6	-0	-81	22	14
Booth's	12	7	15	5	10	44	10	5	52	5	0	+165	-75	-6	-0	-81	84	9
Delph	76	16	20	5	20	14	15	5	23	5	5	+204	-0	-6	-10	-16	188	1
Round	33	9	15	5	10	4	10	5	17	5	5	+118	-30	-6	-10	-46	72	11
Dam	63	22	15	10	20	20	15	5	22	5	5	+202	-65	-8	-5	-78	124	3

Table 6. 1 An evaluation of the nature conservation value of existing semi-natural woodland in an area of St. Helens. Criteria are given in full in Table 6. 2.

Table 6. 2 Criteria used to assess the nature conservation value of each semi-natural woodland presented in Table 6. 1.

CODE NUMBER (from Table 6. 1)	**FULL DESCRIPTION OF CRITERIA**	**SCORING SYSTEM**
1	Herbaceous species richness	+1 point for each native/naturalised species (excluding *Rhododendron* spp.)
2	Woody species richness	+1 point for each native/naturalised species (excluding ornamentals)
3	Growth stages - seedling/sapling/young tree/mature tree/overmature tree	+5 points for each growth stage present
4	Growth forms - maiden/coppice stool/wavers/pollard/climber	+5 points for each growth form present
5	Vertical structure - shrub/canopy	+10 points for each layer in which vegetation present
6	Woodland size	+2 points for each 0.25 ha of area
7	Subsidiary habitats - pond/stream/open/rock/dead wood	+5 points for each subsidiary habitat present
8	Management	+5 points if no recent management obvious
9	Area/Perimeter ratio	Ratio value converted into equal score
10	Stand origin	+0 points if planted, +5 if planted/semi-natural, +10 if semi-natural
11	Adjacent land	+0 points if agricultural, +5 points if man-made non-agricultural, +10 points if semi-natural habitat
12	*Rhododendron* spp. cover	-1 point for each % cover value
13	Isolation	-2 points for each 100m in distance from centre of nearest neighbouring woodland
14	Disturbance	-0 points for no disturbance; -10/-5 points respectively for high and medium disturbance on public land, -4/-2 respectively on private land

ecological planning of new woodlands (see Ferris-Kaan, 1995), the concept of the 'conservation of meaning' (Rackham, 1991) must therefore also be ensured. This applies both to landscapes and species; for the latter the concept is best exemplified by the recent change in attitude towards the planting of *Tilia cordata* in Britain, discussed by Rackham (1991):

> "Tree-planting too seldom takes account of the meanings and values of trees. The small-leaved lime or pry is a relatively rare tree which hitherto has not usually been planted. It has now become a fashionable tree, part of the Standard Broadleaved Mixture. What good does that do? Although rare, pry is not threatened. Its meaning lies in being a rare and wonderful tree with a mysterious natural distribution. It is devalued by being made a common tree".

Similarly, trees and the landscapes in which they are located should not be viewed merely as 'artefacts' which can be controlled with 'technical solutions' to produce 'predictable outcomes' (Rackham, 1991), as this may lead to the dilution of existing vegetation patterns and meanings (Rackham, 1995); though in many urban/industrial areas with little existing natural vegetation cover this argument may be less relevant. However, the range of substrate characteristics that is often associated with derelict land in such areas may give rise to naturally-established and regionally distinct woodland plant communities of significant ecological value (Department of the Environment, 1996), the value and 'meaning' of which should not be compromised, but complemented, by new woodland establishment.

6. 5 Conclusions

Plant communities on colliery spoil in Merseyside are generally species-poor; site isolation makes colonisation by suitable plant propagules unlikely, while the often extreme range of spoil chemical and physical characteristics significantly reduces plant survival and growth. Naturally-established woodlands on colliery spoil in the region are characterised by field, shrub and canopy layers in which few typical woodland plant species are found. Further planting of large areas of colliery spoil in the region will give rise to woodlands with similar characteristics. As a whole however, these woodlands represent a significant proportion of all woodland in the region, and are therefore potentially highly valuable for nature conservation and recreation/amenity uses. Community enhancement/diversification techniques represent a means by which the nature conservation value of these woodlands can be increased, while simultaneously providing a more diverse landscape for public use and enjoyment.

Field layer enhancement/diversification can be achieved by the initial introduction of a number of herbaceous species best suited to existing site conditions, particularly spoil chemical characteristics and levels of canopy shade. Monitoring of successful species establishment and growth should be used to guide further introductions, using species with similar ecological characteristics. Field layer introductions should ideally be undertaken over a number of years, so as to accommodate any microclimatic changes that may arise from canopy layer development. Introductions before full canopy closure may depend to some degree on management inputs, particularly herbicide and/or mulch treatments, to reduce the competitive effects of existing species. Full canopy closure and higher levels of shade will better suit many typical woodland species however, and management inputs may be significantly reduced as a result.

Tree and shrub species should be considered integral to any enhancement/diversification scheme. New woodlands may be established by planting, using species that are characteristic of mature semi-natural woodland in the region. For nature conservation purposes however, it would be preferable to work with natural processes of colonisation and successional change, so as to maintain local and regional landscape/ecological character. As part of this process, species enhancement/diversification may be possible through introductions of transplants or seed at suitable successional stages.

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APPENDICIES

APPENDIX TO CHAPTER 3

A 3. 1 Soil analytical techniques

PHOSPHORUS

P was extracted by shaking 5 g of air-dry spoil in 125 ml of 2.5 % aqueous acetic acid solution for one hour on a table shaker. After standing for one hour, samples were filtered through Whatman No. 42 filter paper. A Guilford Colorimeter was used to determine the extractable P present as a phosphomolybdate complex reduced by ascorbic acid.

CALCIUM AND POTASSIUM CATIONS

Ca and K cations were extracted by shaking 5 g of air-dry spoil in 125 ml of 2.5 % aqueous acetic acid solution for one hour on a table shaker. After standing for one hour, samples were filtered through Whatman No. 42 filter paper. Ca was determined using a Varian-Techtron 1275 atomic absorption spectrophotometer. K was determined by flame emission, ionisation of K in the flame was suppressed by adding caesium chloride to the extracts.

pH

pH values were determined by half filling a 50 ml beaker with fresh spoil and adding deionised water to fill. The solution was stirred and left to stand for ten minutes. Electrodes were immersed in the solution and the pH read on a meter buffered at pH 4.

A 3. 2 Census dates of herbaceous woodland plant species monitoring at Bold Moss Tip and Gillars Green Wood, St. Helens.

BOLD MOSS TIP	GILLARS GREEN WOOD
11 July 1993	12 July 1993
5 September 1993	6 September 1993
6 November 1993	8 November 1993
11 January 1994	10 January 1994
10 March 1994	13 March 1994
22 April 1994	28 April 1994
10 June 1994	9 June 1994
30 July 1994	28 July 1994
22 September 1994	17 September 1994
14 November 1994	11 November 1994
24 January 1995	20 January 1995
27 March 1995	31 March 1995
18 May 1995	25 May 1995
23 July 1995	-

APPENDIX TO CHAPTER 4

A 4. 1 Hemispherical photograph data, Bold Moss Tip, St. Helens.

N.B. Missing values relate to sampling areas in which hemispherical photographs were not repeated in August 1995.

	MEASURED VARIABLE									
	% Open Sky		Direct PFD		No. Sunflecks		Flecks <40		Flecks >40	
Month	June 95	Aug 95	June 95	Aug 95	June 95	Aug 95	June 95	Aug 95	June 95	Aug 95
Plot/ Photo No										
AIP21	61%	-	208	-	4	-	3	-	1	-
A1P51	23%	-	36	-	8	-	8	-	0	-
A1P52	15%	12%	12	9	4	3	4	3	0	0
A1P54	16%	-	10	-	3	-	3	-	0	-
A1P72	34%	-	8	-	3	-	3	-	0	-
A2P43	17%	20%	66	54	17	14	17	12	0	2
A2P52	17%	21%	32	36	6	8	6	8	0	0
A2P71	55%	-	234	-	2	-	1	-	1	-
A3P13	76%	-	272	-	10	-	9	-	1	-
A3P22	76%	-	316	-	18	-	17	-	1	-
A3P32	89%	78%	288	283	4	4	3	3	1	1
A3P33	85%	-	420	-	5	-	3	-	2	-
A3P34	58%	-	230	-	5	-	4	-	1	-
A4P12	87%	79%	150	139	8	6	7	5	1	1
A4P31	91	-	272	-	6	-	5	-	1	-
A4P54	48	-	126	-	16	-	15	-	1	-

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