# The ecological impacts of wild boar rooting in East Sussex

By

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# Abstract

The aim of the thesis was to assess the ecological impacts of wild boar rooting for up to three years on above and belowground community attributes and processes in semi-natural habitats in southern England. The research tested the hypothesis that wild boar are important allogenic ecosystem engineers.

Plant species richness, percentage cover and diversity were significantly greater from rooted than non-rooted treatments across woodland, grassland and woodland ride habitats. Abundance of various plant functional groups was differentially affected by rooting within different habitats. Protection from re-rooting by fencing had a significant positive impact on recovery of several plant functional groups and total plant cover.

Numbers of Hyacinthoides non-scripta individuals and flowering stems were significantly lower in rooted than non-rooted treatments in woodland, although the proportional change in H. non-scripta cover over three growing seasons was significantly greater in rooted than non-rooted treatments, illustrating substantial recovery. Rooting had no impact on H. non-scripta seed weight although seed viability and total numbers of seeds per capsule and per plant were significantly greater from rooted than non-rooted treatments.

Rooting significantly altered the viable seed bank in terms of overall abundance, species richness, diversity and functional group responsiveness measured by seedling emergence from soil of woodland and grassland habitats. It was suggested that the seed bank density was greater in rooted than non-rooted soil and, through altered soil properties, dormancy breaking mechanisms and germination were increased. It was suggested that emergence from the viable seed bank could largely contribute to the re-establishment of plants in rooted areas.

Leaf litter decomposition rate in woodland, soil NO3- concentration in woodland and grassland, soil NH4+ concentration in grassland were significantly greater from rooted than non-rooted soil. Further, belowground live plant biomass was significantly reduced in rooted than non-rooted soil in woodland and grassland. These belowground community attributes are fundamental determinants of productivity, performance and dynamics of the whole community.

It was concluded that the ecological impacts of rooting are patchy and fluctuate in distribution. Rooting transforms biotic and abiotic material from one physical state to another and fundamentally contributes to modifying the structure and dynamics of the whole community. It was concluded that wild boar are important allogenic ecosystem engineers.

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# Chapter 1

## General Introduction

### 1.1 Introduction

Extinct in Britain for at least three hundred years (Yalden, 1999; Goulding, 2003a), wild boar (*Sus scrofa*) have re-established a free-living population within the Weald of Kent and Sussex, in southern England. Once an integral feature of British woodlands, this former native animal has returned to a profoundly different environment. Around six and a half thousand years ago, Britain was covered by a vast expanse of wild woodland, and home to the wild boar (Rackham, 1997). By approximately two thousand years ago, little wild wood remained, and the majority of woodland present was actively managed as, for example, coppice or wood pasture (Rackham, 1997; UK Agriculture, 2006). With the advancement of civilisation, modern woodland has become extremely fragmented and is actively managed, covering only around 11% of England (Rackham, 1997). This novel situation has only previously occurred in Sweden where, as in Britain, a wild boar population that became extinct (around the turn of the 17th century) re-formed from captive escapees during the 1970's (Welander, 2000a). The environmental impact of the current population in southern England is visually dramatic although localised and seasonal (Goulding, 2003b; Wilson, 2005; pers. obs.), but ecological repercussions are likely to be significant. Although un-quantified, the overall proportion of ground disturbed by boar is likely to be small, but where it does occur, the impact can be severe. This novel situation now poses a very important and interesting question: what effects are these animals having on the ecology of semi-natural habitats in Britain?

## 1.2 Ecosystem Engineers

Animals that have large and widespread impacts on the environment could act as ecosystem engineers. Whilst direct ecological interactions such as competition and predation have been the subject of much research effort for over a century, the role of organisms in the creation, modification or maintenance of habitats had rarely been defined, identified or studied. Consequently Jones et al. (1994) proposed a concept of how species, by shaping their habitats to their own needs, change the availability of resources and thus dictate the fates of other species. They called this process ‘Ecosystem Engineering’ and the organisms responsible ‘Ecosystem Engineers’ (Jones et al., 1994; Lawton, 1994; Lawton and Jones, 1995; Gurney and Lawton, 1996; Jones et al., 1997; Alper, 1998; Thomas et al., 1999). Ecosystem engineers are defined as ‘organisms that directly or indirectly modulate availability of resources (including, in some cases themselves) to other species by causing physical state changes in biotic or abiotic materials. In so doing, they modify, maintain and/or create habitats (Jones et al., 1994). Ecosystem engineers have been shaping our ecosystems since life began. Jones et al. (1994) suggested ecosystem engineers might be so fundamental to ecology that all habitats on earth support them and are influenced by them.

Two types of ecosystem engineer have been characterised. Autogenic engineers ‘change the environment via their own physical structure’ (living or dead tissue) that remain as part of the engineered environment. Allogenic engineers change the environment by transforming biotic or abiotic materials ‘from one physical state to another, via mechanical or other means and are not necessarily part of the permanent physical ecosystem structure’ (Jones et al., 1994, 1997).

Trees epitomise the autogenic engineer. The development of a forest results in physical structures that change the environment and modulate distribution and abundance of resources. Trees alter nutrient cycles, soil stability and hydrology, temperature, humidity, wind-speed and light levels. The many occupants of forests are largely dependent on physical conditions modulated by autogenic engineers and on resource flows that they influence (Lawton and Jones, 1995; Jones et al., 1997). A good example of an allogenic engineer is the beaver (Castor canadensis) (Lawton and Jones, 1995; Pollock et al., 1995; Nummi and Poysa, 1997; Alper, 1998; Wright et al., 2002; Wright et al., 2003; Baily et al., 2004). C. canadensis take materials from the environment such as trees and engineer them from one physical state (living trees) to another physical state (dead trees in a dam). The dam creates a pond, which has profound effects in modulating a whole series of resource flows used by other organisms. Dams alter biogeochemical cycles, nutrient retention, geomorphology, biodiversity, community dynamics and structural complexity of the environment. Both the biotic and abiotic influences of the C. canadensis dam are spatially and temporally extensive, sometimes lasting for centuries (Lawton and Jones, 1995; Pollock et al., 1995; Nummi and Poysa, 1997; Alper, 1998; Wright et al., 2002; Wright et al., 2003).

Ecosystem engineers can have positive and negative effects on the diversity and abundance of other species, ranging from extremely small to very large. A C. canadensis pond for example may create habitats for many species to live, but its transformation from a stream may equally remove habitats from many organisms such as stream dwelling organisms or species that lived in trees (Jones et al., 1997). The impact on the entire ecosystem can be extremely complex and only some species will benefit from the changes. However, on a larger temporal and spatial scale, the net effect of engineering may enhance species richness via a net increase in habitat diversity (Wright et al., 2002). The net effect of physical ecosystem engineers across a range of habitats is likely to increase species richness (Jones et al., 1997).

The size of the impact of the ecosystem engineer can vary enormously depending on the magnitude and types of changes that occur, the resources that are controlled, the number of species in the habitat that depend on these resources, and the extent to which these resources are adequate to support these species in the new habitat (Jones et al., 1997). Organisms that engineer rivers, streams, soil and sediments tend to have some of the largest impacts. Water, soil and sediments incorporate many resources such as nutrients and living space within one engineered context, thus modifying them has big effects. Therefore, ecosystem engineers can intrinsically create and shape environments and the energy that flows through them by providing the elemental resources, which underlie entire food webs.

## 1.3 Rooting

A previously unrecognised example of a possible important allogenic ecosystem engineer is the wild boar. Its rooting activities represent an important manifestation of ecosystem engineering. Rooting is the term given to the wild boar's foraging activity, which occurs within surface layers of soil. Visually analogous to ploughing, this 'rototiller-like' foraging activity (Ray, 1988; Moody and Jones, 2000) can be viewed as a severe form of soil disturbance. Rooting can result in the translocation of large amounts of earth. For example, in one year, in the eastern part of the Bialowieza Forest, 430m$^3$ of earth were translocated to different regions within mixed coniferous and deciduous forest (Falinski, 1986). Rooting fluctuates in area, depth and intensity and, because boar tend to root in patches, the effects on soil are likely to be heterogeneous (Welander, 2000b). Consequently, the structural complexity of the soil surface can be increased as rooting exposes a variety of substrates (such as humus, mineral soil, belowground plant biomass, rocks and stones) in a patchy manner (Milton *et al.*, 1997; Welander, 2000b). However, the structural complexity of the soil surface can also be decreased because like ploughing, rooting can destroy distinct soil horizons and can homogenise soil. Through mixing soil horizons, rooting can reduce vertical heterogeneity, over-turn leaf litter (reducing surface build up through incorporation into soil) and remove or redistribute vegetation in a patchy manner (Spatz and Mueller-Dombois, 1975; Bratton, 1975; Singer *et al.*, 1984).

Evidence of rooting can be seen within different habitats in and around my study area in Beckley, East Sussex (see **section 1.4.2**). Although rooting is typically seen as small patches of overturned soil (~1m$^2$) (**Fig 1.1 a-c**) (Kotanen, 1995; Goulding, 2003b), these patches often overlap forming larger areas of rooted soil of up to a hectare (Kotanen, 1995) (**Fig 1.2 a-c**). The depth of rooting also varies. A typical rooting depth can be between 5-15cm (Kotanen, 1995; Goulding, 2003b; Mohr *et al.*, 2005) (**Fig 1.1 a-c**). However, occasionally rooting depths can be as great as 30cm or more (**Fig 1.3 a-b**). The location, depth, distribution and abundance of rooting is largely determined by boar population density, the proximity to cover, vegetation and soil type and food availability, farrowing activity, frequency of rocks, and soil-moisture levels (Belden and Pelton, 1975; Falinski, 1986; Hone, 1988). For example, Falinski (1986) found that rooting was shallowest in deciduous forest (8cm), coniferous and mixed forest (6-16cm), and deepest in grassland ecosystems (up to 22cm). Rooting can be distinguished into three general categories (Falinski, 1986). The first level involves cutting of the field layer along the animal’s path. The second type of rooting entails cutting and fragmentation of the field layer by tearing it up from the ground, translocation and returning. The third category is the mixing of the field layer fragments with the soil and litter, destruction of tree stems and up-rooting of small trees with shallow roots (Falinski, 1986).

Wild boar frequently re-root the same specific or general areas of woodland, grassland and woodland rides within their home range (Groot Bruinderink and Hazebroek, 1996; Goulding, 2003b) and therefore can be considered a major disturbance, one that English woodland has not seen for hundreds of years, since boar became extinct. However, pannage, the practise of driving domestic pigs into woods and wood pasture in autumn to fatten on acorns (or beech mast if any) before being slaughtered and salted down, is a similar disturbance regime that can still be seen the New Forest today (Rackham, 1997). The acorn crop cannot be relied upon as the crop size can vary substantially every year, and consequently, although once a common practice in woodland and wood pasture, pannaging was in considerable decline by 1086AD (Rackham, 1997).

However, domestic pig and wild boar husbandry is a traditional feature of natural or semi-natural woodland management throughout Europe today. Although this is not yet reflected in the UK, there is considerable interest in a more widespread practical application of pigs in the use of woodland management (Brownlow, 1994). Pigs are used for not only pannage, but also used as silvicultural tools for ground preparation, weeding and pest control, which enhances the regeneration and growth of young coniferous trees by removing competing vegetation from occupied areas and invertebrate pests (Brownlow, 1994). For example, herds of pigs are rotationally grazed, (particularly during the period of acorn production) over vast areas of southern Iberia; pigs are used not only to exploit the acorn crop and provide meat, but control scrub reinvasion and nutrient cycling, and have become part of the traditional landscape (Brownlow, 1994).

**a)**

**b)**

**c)**

**Fig 1.1 a-c:** Examples of rooting in relatively small isolated patches in three out of the four grassland sites within the study area. Rooting depth here is normal, between 5-15cm. **a)** = site **G4**, **b)** = site **G1**, **c)** = site **G4** (**Table 1.1**, **Fig 1.5**). Photos taken 2002.

**a)**

**b)**

**c)**

**Fig 1.2 a-c:** Examples of extensive rooting covering large areas. **a)** and **b)** = grassland site **G2**, **c)** = woodland ride site **R5** within the study area (**Table 1.1**, **Fig 1.5**). Photos taken 2002.

**Fig 1.3 a-b:** Examples of rooting at a greater than average depth (approx. 15-35cm) in **a)** = woodland ride site **R5** and **b)** = grassland site **G2** (**Table 1.1**, **Fig 1.5**). Photos taken 2003.

Rooting causes direct and indirect impacts on plants at the community, functional group and individual species level, which could generate repercussions that spread throughout food webs. For example, rooting is known to directly increase (Bowman and McDonough, 1991; Kotanen, 1994, 1995; Welander, 1995; Onipchenko and Golikov, 1996; Milton et al., 1997; Arrington et al., 1999; Welander, 2000a) and decrease (Bratton, 1974, 1975; Hone, 1980; Howe et al., 1981; Alexiou, 1983; Singer et al., 1984; Ralph and Maxwell, 1984; Arrington et al., 1999) plant cover and species richness. Rooting also directly affects the relative abundance of functional groups such as annual forbs (Kotanen, 1994, 1995; Boeken et al., 1998; Welander, 2000a) and woody species (Andrezejewski and Jezierski, 1978; Lipscomb, 1989; Peart et al., 1994; Groot Bruinderink and Hazebroek, 1996). Some individual plant species may directly benefit from the disturbance whilst others, more sensitive to disturbance, could suffer a reduced population density (Kotanen, 1995), or localised extinction in more extreme cases.

Most of our knowledge of wild boar derives from studies of populations on the European continent and non-European countries. Little scientific literature currently exists on ecological impacts of rooting in Britain (Goulding, 2003a, 2003b). Among existing literature on wild boar populations abroad, contradictory research results relating to the effects of rooting on plant cover, species richness and functional group and individual species responsiveness make it difficult to predict the impacts of rooting on plants in Britain. Variation in climate, geology and ecosystem and community structure could lead to different responses of belowground soil attributes and above ground plant communities to rooting in different countries. It is therefore important to scientifically research the effects of rooting on plant re-establishment in Britain to gain a greater understanding into the effects of rooting on the above ground plant community under specific British climatic and environmental conditions.

Indirectly, rooting could affect the plant community via altering soil properties and processes. Properties include physical agents such as soil aeration, light levels and temperature (Singer et al., 1984; Kotanen, 1994), soil chemistry (Lacki and Lancia, 1983; Groot Bruinderink and Hazebroek, 1996; Mohr et al., 2005) and biological attributes such as the belowground invertebrate community and the seed bank (Vtorov, 1993; Kotanen, 1994; Mohr et al., 2005). Processes include soil decomposition and nitrogen transformation systems (Lacki and Lancia, 1983; Singer et al., 1984; Mohr et al., 2005). Significant change in the soil environment is likely to impact on the organisation and performance of both plant and animal elements of the community. Physical properties and soil processes underlie nutrient cycling and net primary productivity, which govern plant growth and thus the dynamics of the community. As with above ground impacts, little is known of the effects of rooting on belowground community attributes and processes, and that which does exist, is largely contradictory (Singer et al., 1984). For example, soil nutrient content has both been reported to have significantly increased in rooted areas (Lacki and Lancia, 1983; Singer et al., 1984; Kotanen, 1994), be unaffected by rooting (Groot Bruinderink and Hazebroek, 1996; Moody and Jones, 2000; Mohr et al., 2005) and decreased by the disturbance (Mohr et al., 2005). It is therefore necessary to carry out further work of this nature in Britain to gain an understanding into the effects of rooting on important belowground community attributes under specific British environmental conditions.

The potentially large and widespread impacts of rooting provide great scope for studying the potential role of wild boar as an allogenic ecosystem engineer (Jones et al., 1994; Brown, 1995; Gurney and Lawton, 1996; Jones et al., 1997; Alper, 1998; Wright et al., 2002). The direct and indirect impacts that rooting has on the physical, biological and chemical structures of soil, on cover, species richness and distribution of plants and associated fauna and the creation of heterogeneity (Welander, 2000b), may change the physical structure of the community and therefore modulate the availability of resources to other species. This complies with Jones et al. (1994) definition of allogenic engineering whereby rooting transforms living and non-living material from one physical state to another without being part of the permanent physical ecosystem structure. Since boar inhabited British woodlands for thousands of years, (Rackham, 1980; Marren, 1990) their engineering activities would probably have fundamentally helped shape past woodland structure. It is therefore proposed that wild boar, are important allogenic ecosystem engineers and their rooting is ecosystem engineering.

## 1.4 The Study System

### *1.4.1* ***Wild Boar***

Wild boar are native on the European continent and live freely in large numbers within a great diversity of habitats (Kurz and Marchington, 1972; Dardaillon and Beugnon, 1987; Falinski, 1986; Meriggi and Sacchi, 1992, 2000). They are known to have adapted to living in swamp and marshland environments, mountains, coastal areas, deciduous and coniferous woodland, and can withstand extremely cold climates (Mauget, 1980; Falinski, 1986; Gerard *et al.*, 1991; Cargnelutti *et al.*, 1992; Boitani *et al.*, 1994; Spitz and Janeau, 1995; Russo *et al.*, 1997). Wild boar have also been successfully introduced into non-native countries such as dry arid and tropical regions of Australia and the USA (Bratton, 1975;

Singer *et al.*, 1981; Baber and Coblentz, 1986; Saunders and Kay, 1996; Caley, 1997; Gabor *et al.*, 1999). Radio tracking however reveals their high dependence on woodland habitats, which provide the greatest food availability and protective cover especially in highly disturbed environments (Kurz and Marchington, 1972; Singer *et al.*, 1981; Baber and Coblentz, 1986; Falinski, 1986; Caley, 1997; Goulding, 2003b).

Six thousand years ago, Britain was covered by extensive woodland (Ingrouille, 1995; Rackham, 1980; Marren, 1990). Approximately four million boar were thought to have inhabited these wildwoods of oak, ash, lime and hazel (Marren, 1990; Yalden, 1999; Goulding, 2003a). Habitat loss, over-hunting and finally absorption into domestic herds ultimately caused their extinction in Britain (Rackham, 1980; Goulding, 2003a). The exact date that wild boar became extinct in Britain is unclear due to subsequent attempts to reintroduce the animals from the continent (Goulding, 2003a). However, it is generally believed that free-living wild boar became extinct in England at around the turn of the 14th century and during the 16th century in Scotland (Yalden, 1999; Goulding, 2003a). By the end of the 17th century, all native and introduced wild boar in Britain had died out. Until recently, no free-living wild boar (native or introduced) had been present in Britain for the last 300 years (Goulding, 2003a).

The possible reintroduction of wild boar into Britain has been speculated on in recent years (Yalden, 1986; Howells and Edwards-Jones, 1997; Leaper *et al.*, 1999). However, a free-living population has become established in southern England during the past eighteen years and is thought to have originated from several accidental releases from domestic wild boar farms in Kent. The presence of wild boar in England has provoked considerable controversy (Goulding, 2003b), and probably the most controversial aspect of their presence for conservationists and farmers, is rooting. However, the more widespread issues of public safety and disease are of the greatest concern for the public and DEFRA (Department for Environment, Food, and Rural Affairs). The most prominent issue of public safety is the fear of wild boar attacks on people although no confirmed reports in the literature of wild boar making unprovoked attacks on people exist (Goulding and Roper, 2002). This is consistent with knowledge that wild boar are known to be shy, and typically avoid contact with people whenever possible (Genov, 1999; Goulding, 2003a; pers. obs.). The main concern relating to disease, focuses on the possibility that wild boar populations could act as a reservoir or means of spreading disease to livestock, predominantly domesticated pigs (Wilson, 2005). Those diseases of most concern are Classical Swine Fever, African Swine Fever, Foot and Mouth Disease, Aujeszky's Disease and Trichinosis (Wilson, 2005). Such concerns have led agriculturists and some landowners to class wild boar as economic pests and call for their eradication, whilst others have been advocating their deliberate reintroduction into this country (Leaper *et al.*, 1999).

The snout of wild boar is narrow, long and straight, well designed for foraging among surface layers of soils. Boars are generally larger than sows, and the adults have tusks that protrude from the side of the jaw, an adaptation also utilised whilst foraging (Genov, 1999; Goulding, 2003a) (**Fig 1.4 a-b**). Wild boar are omnivorous and consume a large variety of food depending on seasonal availability (Henry and Conely, 1972; Wood and Roark, 1980; Genov, 1981; Falinski, 1986; Dardaillon, 1987; Massei *et al.*, 1996). Their diet consists principally of above and belowground plant parts' that vary between approximately 90% (Henry and Conely, 1972; Genov, 1981) and 97% of total consumption (Massei *et al.*, 1996). The remaining portion chiefly consists of invertebrates and small vertebrates (Massei *et al.*, 1996).

**Fig 1.4 a-b:** Wild boar (*Sus scrofa*). **a)** A typical adult male showing the strong snout and tusks, which are adapted for rooting. **b)** An adult sow with her piglets in woodland within the study area.

Wild boar are primarily nocturnal (Boitani *et al.*, 1994) and spend daylight hours resting, hidden in dense vegetation (Abaiger *et al.*, 1994). Boar typically spend between four and eight hours feeding at night (Spitz, 1986) within their home range, predominantly rooting. Home range size varies depending on food availability and habitat type (Falinski, 1986), sex of the animal, human disturbance and population density. The potentially large area encompassed by a boar's home range, illustrates how extensive the effects of rooting can be. For example, Singer *et al.* (1981) recorded a mean seasonal home range of male wild boar as 3.5km$^2$ and 3.1km$^2$ for females in Tennessee, USA. However, during a year with poor beech mast, home ranges of some animals increased to 10.7km$^2$, implying that home range size varies inversely with resource abundance and density (Caley, 1997). Russo *et al.* (1997) recorded small home range sizes between 0.029-1.081km$^2$ in the hunt-free Maremma National Park, central Italy. Saunders and Kay (1996) determined larger home ranges of 35km$^2$ for males and 11.1km$^2$ for females in a more highly disturbed area in New South Wales, Australia. Dardaillon and Beugnon (1987) reported home range sizes of between 20-68km$^2$ in Camargen, south-east France indicating the ability of boar to travel very large distances, particularly in disturbed environments.

### *1.4.2 Study area*

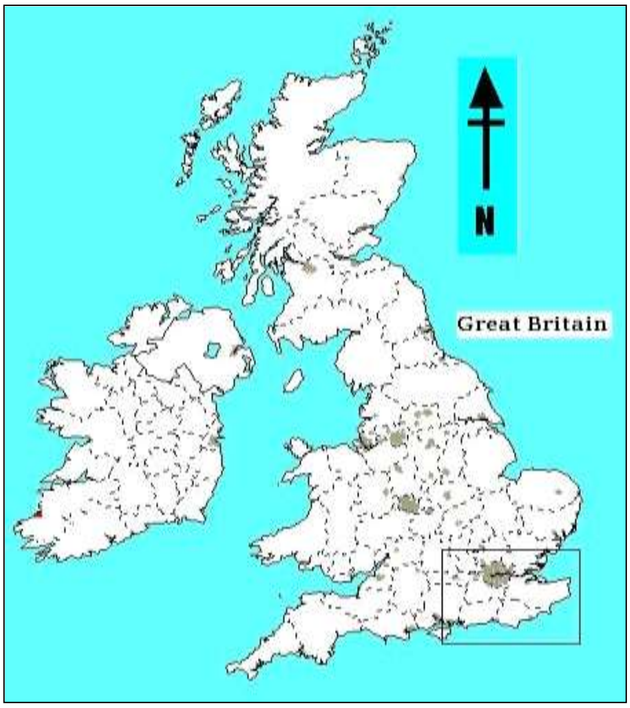
The region of East Sussex surrounding the study area is predominantly arable and grazing farmland interspersed with villages and woodland. The study area is within and around Bixley and Beckley woods, East Sussex (**Fig 1.5 a-d**); owned and managed by the Forestry Commission and Sussex Wildlife Trust and comprises conifer plantations, oak and mixed deciduous and sweet chestnut coppice (**Fig 1.6 b**, **Table 1.1**). Woodland rides (**Fig 1.6 a**) and agricultural and semi-natural grassland (**Fig 1.6 c**) are also present within the study area and used for my research (**Table 1.1**). Differences exist between sites within habitats in terms of predominant sub-classification of habitat type, management and dominant vegetation (**Table 1.1**).

The area represents the centre of the East Sussex population of wild boar and is where all previous research has been carried out on this population of animals (Goulding, 2003b). It is the largest out of three current breeding populations in England, with an estimated population density of around 200 animals, covering an approximate range of 175km$^2$ (Wilson, 2005). In contrast, the second largest population in west Dorset is estimated to be around 30 animals, covering an approximate range of 27km$^2$ (Wilson, 2005). The size and range of the smallest population within the Forest of Dean is unknown, although an alleged illegal release of at least 25-30 animals is suspected to have increased the population in spite of several being shot (Wilson, 2005).

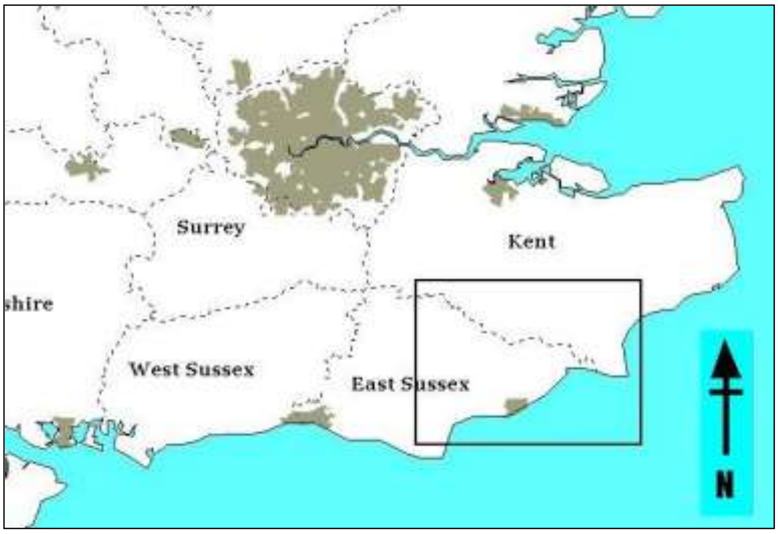
Within the study area, individual study sites were selected based on spatial distinction of districts across the three different habitat types coupled with the presence of rooting. The distribution of rooting within the study area, although un-quantified, varies considerably within and between sites. Some sites, such as a relatively small field could be extensively rooted throughout (**Fig 1.2**), whereas in contrast, a neighbouring site could be completely non-rooted. More typically, rooting occurs in patches over many sites with a relatively small coverage (**Fig 1.1**) relative to the size of the site. Overall, in the study area, in relation to the current population density and available space, the ratio of rooted to non-rooted ground is likely to be small. This suggests that the severity of rooting as a disturbance is regional and localised, which should be considered when viewing overall impacts of rooting within relatively large areas.

**Fig 1.5 a-d:** Four tiers of maps showing the approximate location of the study area in relation to **a** = Great Britain, **b** = South East England and **c** = Kent and East Sussex. **d** = the study area showing individual study sites (1:25 000, Ordnance Survey, 1998). **G1-G4** = Grassland sites 1-4, **W1-W5** = Woodland sites 1-5, **R1-R5** = Woodland Ride sites 1-5, See **Table 1.1**.

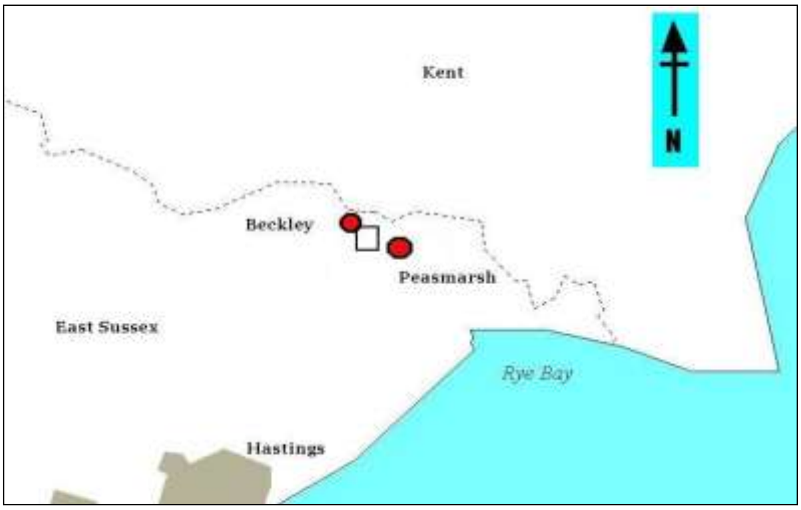
**a)**



**b)**



c)


Kent
N
Beckley
Peasmarsh
East Sussex
Rye Bay
Hastings

d)

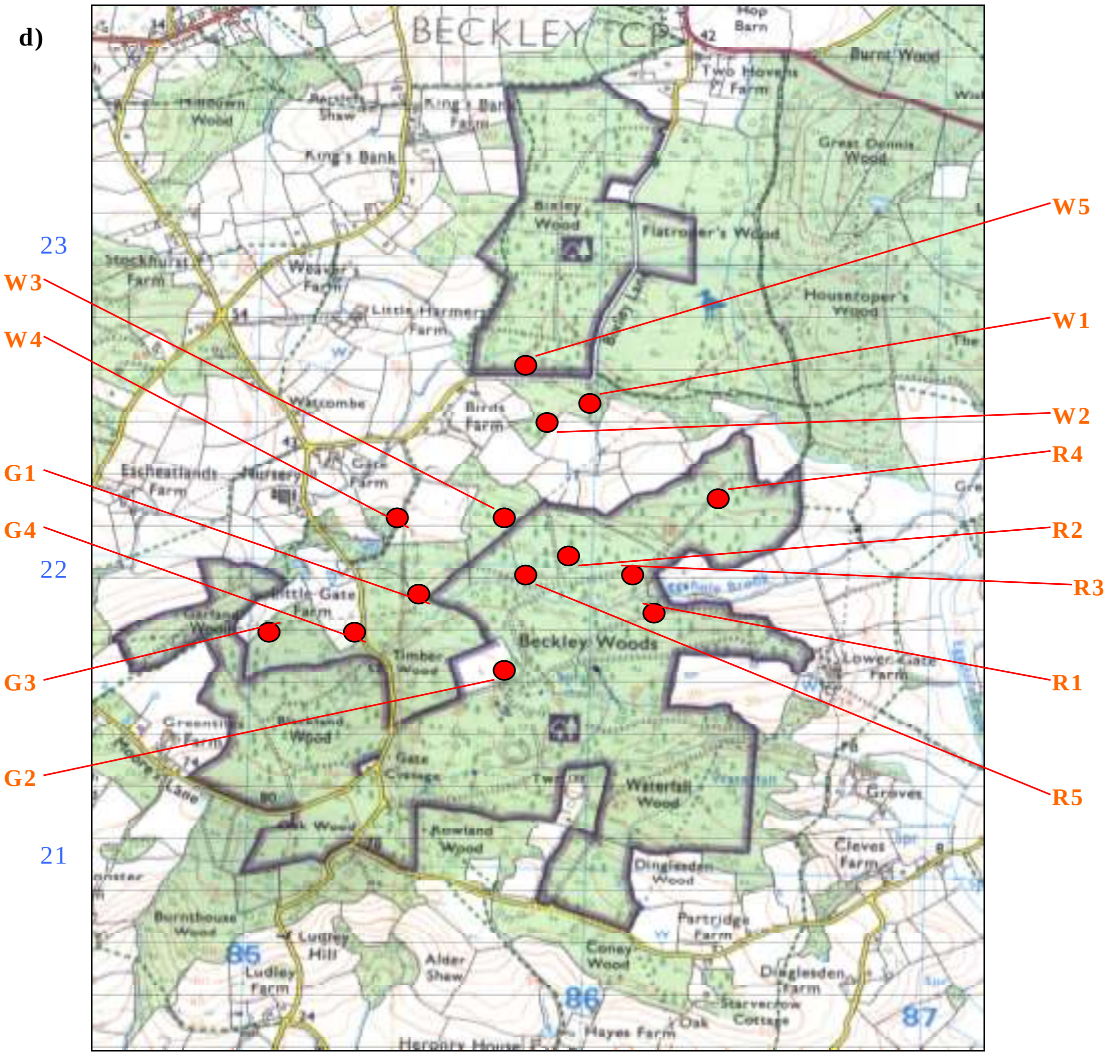

BECKLEY CP
23
22
21
W5
W3
W1
W4
W2
R4
G1
R2
G4
R3
G3
R1
G2
R5
Bixley Wood
Flatropers Wood
Housezoper's Wood
Great Dennis Wood
Burnt Wood
Two Hovels Farm
King's Bank
Stockhurst Farm
Weaver's Farm
Little Harmers Farm
Watcombe
Birds Farm
Escheatlands Farm
Gate Farm
Little Gate Farm
Garland Wood
Timber Wood
Beckley Woods
Blackland Wood
Greensite Farm
Gate Cottage
Rowland Wood
Waterfall Wood
Dinglesden Wood
Lower Gate Farm
Groves
Cleves Farm
Partridge Farm
Coney Wood
Burnthouse Wood
Ludley Hill
Ludley Farm
Alder Shaw
Dinglesden Farm
Starvecrow Cottage
Hayes Farm
85
86
87

**Fig 1.6 a-c:** Examples of different habitat types within the study area. **a)** = woodland ride site **R5**, **b)** = woodland site **W4**, **c)** = grassland site **G2**. Photos taken in 2002.

| Habitat | Sites | Grid Reference | Dominant Vegetation | Management | Closest NVC |
|---|---|---|---|---|---|
| **Woodland (W)** | | | | | |
| 1 | Sweet chestnut | 22.5: 85.9 | Dense bluebell and mixed ground flora | Coppicing | W10a |
| 2 | Sweet chestnut | 22.4: 85.8 | Dense bluebell and mixed ground flora | Coppicing | W10a |
| 3 | Beech Woodland | 22.2: 85.6 | Sparse bluebell population and mixed ground | Occasional thinning | W14 |
| 4 | Mixed deciduous | 22.2: 85.4 | Dense bluebell and mixed ground flora | Occasional thinning | W10a |
| 5 | Mixed deciduous | 22.6: 85.7 | Dense bluebell and mixed ground flora | Occasional thinning | W10a |
| **Grassland (G)** | | | | | |
| 1 | Woodland | 21.8: 85.5 | High diversity grassland plant community | Occasionally grazed by sheep | MG9b |
| 2 | Semi-wild | 21.6: 85.7 | High diversity grassland plant community | Never grazed, unmanaged | MG10 |
| 3 | Agricultural | 21.7: 85.0 | Low diversity grassland plant community | Frequently grazed by sheep | MG7b |
| 4 | Agricultural | 21.7: 85.3 | Low diversity grassland plant community | Frequently grazed by sheep | MG7b |
| **Woodland Rides (R)** | | | | | |
| 1 | Flanked by coniferous plantations | 21.8: 86.2 | High diversity grassland plant community | Edges cut back once a year | |
| 2 | Flanked by coniferous plantations | 22.1: 85.9 | High diversity grassland plant community | Edges cut back once a year | |
| 3 | Flanked by coniferous plantation and mixed deciduous woodland | 22.0: 86.1 | High diversity grassland plant community | Edges cut back once a year | |
| 4 | Flanked by coniferous plantation and mixed deciduous woodland | 22.3: 86.4 | High diversity grassland plant community | Edges cut back once a year | |
| 5 | Flanked by mixed deciduous saplings | 21.9: 85.7 | High diversity grassland plant community | Edges cut back once a year | |

**Table 1.1**. Summary of the fourteen sites used in this research. NVC classification derived from Rodwell (1991, 1992).

### *1.4.3 Importance of habitat types*

The habitats of the greatest importance to this research are those that are most affected by rooting and those that hold the greatest conservation value. These are various types of woodland, grassland and woodland rides (**Table 1.1**, **Fig 1.5 a-d**). With only relatively small fragments of lowland British woodland remaining in southern England today, the conservation and aesthetic value attached to this habitat type is high. Most ancient woodland was managed by coppicing during the last one thousand years (Peterken, 1993). By 1965, coppice management was largely abandoned over most of Britain and much reduced in the south-east; remaining practice today is predominantly carried out in south-east England (Peterken, 1993). Due to its management regime, coppiced woodland is home to a specific array of open-habitat plant species (Fuller and Warren, 1993; Gondard *et al.*, 2001; Mason and McDonald, 2002). Recently coppiced woodland tends to have a significantly greater diversity and abundance of ground flora than after the closure of the canopy in the years following this disturbance (Ash and Barkham, 1975; Ford and Newbould, 1977; Gondard *et al.*, 2001; Mason and McDonald, 2002) and when compared with mature woodland. The open, brighter conditions encourage germination of open-habitat species (Fuller and Whittington, 1993; Mason and McDonald, 2002), which would be far less abundant or non-existent in shadier deciduous woodland (except during early spring before canopy growth). A major disturbance such as coppicing, in part initiates secondary succession (Reade-Runkle, 1982; Sousa, 1984; Niemela, 1999; Mayer *et al.*, 2004; Rydgren *et al.*, 1998). Initially, the number of plant species (particularly annuals) greatly increases but only perennials tend to persist and most open-habitat species become excluded after ten years (Ash and Barkham, 1975; Gondard *et al.*, 2001, Mason and McDonald, 2002). Light levels and natural disturbance on the mature deciduous woodland floor are less than in actively coppiced woodland (Mason and McDonald, 2002). These low disturbance and light levels may support a lower ground floral diversity than coppice woodland. Growth of shade tolerant plants and those sensitive to disturbance may be encouraged under such conditions, and these support their own unique set of fauna (Peterken, 1993).

Also of conservation importance, are woodland rides (Warren and Fuller, 1993) (**Fig 1.6 a**). These habitats cover relatively small proportions of the overall woodland size but support relatively high levels of wildlife and tend to develop different flora and fauna to surrounding woodland (Warren and Fuller, 1993; Peterken, 1993; Sparks *et al.*, 1996). Rides provide semi-permanently open, light habitats; for many species, rides represent the only part of the woodland where they can exist (Peterken and Francis, 1999). Rides within conifer plantations are important as they contain the majority of deciduous trees and shrubs and associated fauna in the whole woodland, and thus increase ecosystem diversity (Sparks *et al.*, 1996). The plant communities within rides can represent important relics of unimproved pasture, 95% of which have disappeared from surrounding countryside over the previous fifty years (Warren and Fuller, 1993; Peterken and Francis, 1999). Plants, such as common bird's-foot trefoil (*Lotus corniculatus*) and many grasses, typical of unimproved pasture, cannot tolerate the shade of closed-canopy woodland and thrive in open sunny conditions of rides (Warren, 1985). Other plant species such as hedge woundwort (*Stachys sylvatica*), wood speedwell (*Veronica montana*) and common figwort (*Scrophularia nodosa*) are much more common on rides than any other habitat (Warren and Fuller, 1993). Diversity is one of the most frequently cited criteria for site selection by conservationists (Prendergast *et al.,* 1993) and rides are often the most diverse habitats within entire woodland ecosystems (Warren and Fuller, 1993; Peterken, 1993; Peterken and Francis 1999).

Most plant-feeding insects are host specific so the greatest numbers of such species can be found in ecosystems with the greatest plant diversity (Warren, 1985; Warren and Fuller, 1993) such as woodland rides (Warren and Fuller, 1993; Peterken, 1993; Peterken and Francis, 1999). The diversity of flowers in rides provides nectar and pollen to a great variety of adult insects (Warren and Fuller, 1993; Sparks *et al.,* 1996). Some insects, including the majority of butterflies, rely solely on open, sunny areas such as rides for their life cycle due to the provision of a warm microclimate (Warren, 1985; Warren and Fuller, 1993; Sparks *et al.*, 1996). A large proportion of moths breed on specific tree and shrub species, and thus shrubby margins of rides are important for moths (Warren and Fuller, 1993). A diverse plant-feeding invertebrate community is likely to support a greater range of bird species (Hunter, 1999).

Breeding migrant birds such as the common nightingale (*Luscinia megarhynchos*) are attracted to shrubby edges of rides, which are also a favoured nest site of the chiffchaff (*Phylloscopus collybitus*) (Fuller and Whittington, 1988). Other bird species use open rides for feeding such as the European robin (*Erithacus rubecula*), the song thrush (*Turdus philomelos*) and the common blackbird (*Turdus merula*) (Fuller and Whittington, 1988). Predatory bird species such as the sparrow hawk (*Accipiter nisus*) and the kestrel (*Falco tinnunculus*) use ride margins for hunting smaller birds and mammals (Fuller and Whittington, 1988). Rides can be beneficial to most small mammals such as some bat species (Chiroptera) which use rides for hunting grounds (Warren and Fuller, 1993).

Many semi-natural grassland ecosystems are valued for their relatively high biodiversity (Ingrouille, 1995). An increase in agricultural area and intensity, particularly over the past fifty years, has inevitably resulted in loss, through fragmentation and simplification, of semi-natural grassland communities throughout south-east England (Ingrouille, 1995). The subsequent and inevitable decline in grassland plant species has probably contributed to the overall reduction of biodiversity across grassland ecosystems (Ingrouille, 1995).

## 1.5 Aims

A continued presence of wild boar in southern England will unquestionably have ecological repercussions. Woodlands today are highly fragmented. Major disturbance regimes like rooting, although once a fundamental part of a natural management regime could now have far reaching consequences. With no current scientific literature focussing on effects of rooting on British woodland ecology, I aimed to fill this gap. The aim of my research was to assess the ecological impacts of rooting for up to three years on aspects of above and belowground population and community attributes and processes in semi-natural habitats in south-east England. The research tested the hypothesis that wild boar are important allogenic ecosystem engineers.

## 1.6 Hypotheses

- **Chapter 2**: Plant community attributes in rooted and non-rooted, exclosed and open plots were compared. It was hypothesised that, other than the initial effects of rooting, plant species richness, total plant cover and diversity across different woodland, grassland and woodland ride habitats would be significantly greater from rooted than non-rooted and fenced than unfenced treatments for up to two years. It was also hypothesised that rooting and fencing would differentially affect plant functional groups for up to two years in different habitats within rooted and non-rooted, exclosed and open plots, depending on the life history strategies that different functional groups and individual species exhibit. Site differences were hypothesised to significantly affect plants at the community and functional group level.

- **Chapter 3**: The study on bluebells (*Hyacinthoides non-scripta*) was carried out in woodland habitats over three growing seasons in exclosed and open plots. It was hypothesised that the number of *H. non-scripta* individuals and flowering stems would be significantly lower in rooted than non-rooted treatments. Further, it was hypothesised that the proportional change in *H. non-scripta* cover would be significantly greater from rooted than non-rooted treatments after one and two years. Fencing was hypothesised to aid the recovery of the *H. non-scripta* population after the affects of rooting. The number of *H. non-scripta* seeds were hypothesised to be unaffected by rooting, and the viability and weight of seeds were hypothesised to be greater from rooted than non-rooted treatments. Site differences were hypothesised to significantly affect all measured aspects of the *H. non-scripta* population.

- **Chapter 4**: It was hypothesised that species richness, number of individuals and diversity of emerging plant individuals would be significantly greater from rooted than non-rooted soil from the viable seed bank, from woodland and grassland habitats. It was also hypothesised that rooting would differentially affect various plant functional groups depending on different dormancy breaking and germination adaptations exhibited by seeds from different functional groups from woodland and grassland. Site differences were hypothesised to significantly affect emergence from the woodland and grassland seed bank at the community and functional group level.

- **Chapter 5**: It was hypothesised that decomposition rate of leaf litter and levels of nitrate and ammonium nitrogen would be significantly greater from rooted than non-rooted soil, and belowground live plant biomass would be significantly lower in rooted than non-rooted soil in woodland and grassland habitats. Site differences were hypothesised to significantly affect all measured belowground community attributes and processes.

# Chapter 2

## Impacts of wild boar rooting on plants at the community, functional group and individual species level

### 2.1 Introduction

**D**ue to the high intensity of soil disturbance created by rooting, it was expected that the initial effects (within two months of rooting) of any rooting event would decrease percentage cover (a surrogate measurement of biomass) and species richness of the plants present. However, single (or annual) rooting events were expected to lead to an increased plant cover and species richness in the short-term (between two months and two years) and medium-term (between two and three years), although diminish over the long-term (more than three years) as the plant assemblage recovers. Further, repeated rooting events (occurring at least twice per year) were expected to decrease the cover and species richness of plants in the short and medium-term compared to non-rooted environments. Long-term effects of frequently applied rooting are difficult to predict without long-term research, although based on Connell’s intermediate disturbance hypothesis, diversity of the community as a whole would likely to become low (Connell, 1978, 1979). Intermediate levels of disturbance in terms of frequency (such as annual rooting events) are known to maximise species richness and diversity, whereas very low (such as never rooted) or high levels of disturbance (such as repeated rooting events) over both relatively short and long temporal scales can cause species richness and diversity to be low (Connell, 1978, 1979).

Rooting causes a visually dramatic and widespread impact on the environment with apparently large consequences for the plant community. Exposure of the seed bank from lower to upper soil layers after rooting may affect re-establishment of the plant community through increased germination (**Chapter 4**). A typical rooting depth between 5-30cms will expose both the transient and long-term seed banks (Thompson and Grime, 1979; Thompson *et al.*, 1997) leading to germination and emergence of elements of the exposed seed bank (Kotanen, 1994), which could substantially alter the structure and size of the plant community. Other propagule sources that potentially contribute to plant community re-establishment after rooting include the seed rain and clonal fragments. The mixing of soil horizons through rooting modifies soil properties and processes (Lacki and Lancia, 1983; Singer *et al.*, 1984; Lacki and Lancia, 1986; Vtorov, 1993; Groot Bruinderink and Hazebroek, 1996; Bialy, 1996; Moody and Jones, 2000; Mohr *et al.*, 2005) (**Chapter 5**), which could significantly affect re-establishment of the plant community. Resources made available by ecological disturbances are likely to be exploited by colonists and regenerating survivors in the course of post disturbance succession (McIntyre *et al.*, 1995; Davis and Pelsor, 2001). For example, increased nitrates ($NO_3^-$) and light in rooted areas (Lacki and Lancia, 1983; Singer *et al.*, 1984) (**Table 5.1**, **Fig 5.6**) could promote competitive dominance by plants requiring higher resource levels resulting in an increased overall plant biomass and leading to reduced species richness (Davis and Pelsor, 2001).

Conversely, Kotanen (1994, 1995) in California, Welander (1995, 2000a) in Sweden, and Milton *et al.* (1997) in central Germany found an increased plant species richness in rooted areas across grassland habitats less than one year after rooting. Bowman and McDonough (1991), Onipchenko and Golikov (1996) and Arrington *et al.* (1999) found species richness had increased in areas that were rooted less than one year previously within a monsoon forest-wetland transition, northern Australia, alpine heath in the Teberda Reserve, Russia, and in marshland ecosystems in central Florida respectively. The increase in species richness could be explained by the intermediate disturbance hypothesis (Connell, 1978, 1979). Intermediate levels of rooting frequency in the short-term may have modified resources (such as nutrients, light and space) to levels that maximised the number of species.

Specific conditions produced through rooting are likely to favour some plants (such as disturbance-tolerant species) but not others (such as species sensitive to severe disturbance) and therefore some functional groups but not others (Aplet *et al.*, 1991). For example, Kotanen (1994, 1995) found annuals proliferated within one year after boar rooting in a Californian prairie. Welander (2000a) found that the large numbers of small sized, wind-dispersed seeds of annuals from the seed rain were ideal for the rapid establishment of recently rooted soil across several habitat types in Sweden, due to their relatively superior dispersal capacity.

The aim of this study was to investigate the short-term effects of rooting on plants at the community and functional group level in woodland, woodland rides and grassland in order to understand the impact of wild boar on plant communities.

### *2.1.1 Hypotheses*

#### *2.1.1.1 Community Level Effects*

Between two and seventeen months after rooting, plant species richness and diversity were hypothesised to be higher in rooted than non-rooted and in fenced than unfenced plots within the three habitat types, corresponding to Connell’s (1978, 1979) intermediate disturbance hypothesis. Due to the initial, relatively high levels of light and space in rooted plots, owing to the removal of plants, evenness of plant species abundance was hypothesised to be greater in rooted than non-rooted and in fenced than unfenced plots within the three habitat types within one year. After one year however, following the potential increase in competitive dominance, related increased biomass and the reduction of resources, it was hypothesised that there would be no significant difference between treatments in evenness, although species richness was predicted to be high in fenced rooted plots. Fencing was primarily to protect areas from further rooting disturbance such that the effects of single rooting events (from fenced rooted treatments) and potentially frequent rooting events (from unfenced rooted treatments) could be compared with the non-rooted controls. Boar tend to re-root the same areas every year (Falinski, 1986; Welander, 2000b; Goulding, 2003b; pers. obs.), hence rooted unfenced plots are particularly vulnerable to re-rooting. Although no quantitative assessment of the amount of re-rooting in rooted unfenced plots was carried out, I observed that the majority of these plots had been re-rooted to some extent at least once during this study. Species richness, diversity and evenness were predicted to be lower in unfenced than fenced plots due to the severity of the disturbance that potentially occurs from frequent rooting events.

I hypothesised that between two and seventeen months after rooting, total plant cover would be greater in rooted than non-rooted (Bowman and McDonough, 1991) and in fenced than unfenced plots within the three habitat types. Percentage cover in unfenced plots (due to their susceptibility to rooting) is predicted to be lower than in fenced plots in the short-term. Relatively higher levels of resources in rooted than non-rooted areas such as soil nitrate ($NO_3^-$) (Lacki and Lancia, 1983; Singer *et al.*, 1984) (**Table 5.1**, **Fig 5.6**) could promote competitive dominance in plants requiring higher $NO_3^-$ levels and thus lead to increased overall plant cover.

I hypothesised that there would be fewer numbers of ancient woodland indicator (AWI) species in rooted than non-rooted plots. AWI species have a low dispersal potential (Rackham, 1980; Hermy *et al.*, 1999; Bossuyt *et al.*, 2002; Wulf, 1997) therefore tend to be confined to permanent mature woodland sites and are thus indicative of them (Buckley *et al.*, 1997). Woodland sites used here are comparatively recent, thus relatively small numbers of AWI species were expected (**Appendix 1**). AWI species are characterised by a stress tolerant strategy (Hermy *et al.*, 1999; Wulf, 2003), therefore are not adapted for coping with severe disturbance such as boar rooting. Unfenced plots are more susceptible to greater disturbance than fenced plots and so were predicted to have fewer AWI species.

I hypothesised that site differences would significantly affect all measured elements of plants at the community level within each habitat type. Different forms of management imposed on the different sites across the three habitat types (**Table 1.1**) could potentially have large impacts on plants at the community level. For example in woodland, coppiced sites **W1** and **W2** (in comparison to those that are occasionally thinned (**W3** – **W5**)) are likely to have greater light levels penetrating the woodland floor, which could substantially affect plant diversity. The diversity of plant communities on the woodland floor with high light levels (such as recently coppiced woodland) is known to be greater than more mature, shadier woodland (Ash and Barkham, 1975; Ford and Newbould, 1977; Gondard *et al.*, 2001; Mason and McDonald, 2002). Further differences that could potentially affect the plant community exist due to the geographical distinction of each site within the three habitat types (**Table 1.1**). Different plant community or habitat types flank each site, hence the neighbouring plant communities, through differences in seed dispersal such as seed rain could affect the overall plant abundance and diversity of the neighbouring study sites. Further, potential inconsistencies between sites could exist in the soil environment, such as available nutrients to plants and, determined in part by differences in topography (**Table 1.1**); relative moisture levels.

#### 2.1.1.2 *Functional Group Level Effects*

Rooting is likely to favour plants that are successful in severely disturbed environments. Grime classifies such plants as ruderals: weedy species with short life cycles and high reproductive outputs that cope well in highly disturbed environments (like ploughed fields) such as many annual forbs (Grime, 1974; Grime, 1979; Grime *et al.*, 1989) (**Appendix 1**). Therefore, I predicted that rooting would favour the establishment of this plant functional group in the short-term (Kotanen, 1994, 1995; Milton *et al.*, 1997) in fenced and unfenced rooted plots within grassland and woodland rides.

I hypothesised that the cover of perennial graminoids would be less in rooted than non-rooted and in fenced than unfenced plots in the short-term within grassland and woodland rides. Graminoids include many adaptive strategies (Grime, 1974; Grime, 1979; Grime *et al.*, 1989) (**Appendix 1**) and therefore some are not specifically adapted for tolerating disturbance. It is therefore likely that other plant functional groups that are more disturbance-tolerant (particularly annual and perennial forbs) and thus with greater relative competitive ability in rooted areas, will establish in addition to graminoids in rooted plots and thus substantially reduce the monopoly of graminoids in grassland communities.

The frequency of occurrence of perennial forbs was hypothesised to be higher in rooted than non-rooted and in fenced than unfenced plots in the short-term within the three habitat types. Many perennial forbs have persistent seed banks (Fenner, 1992); germination from which was predicted to be relatively high due to the exposure of the seed bank to increased light, oxygen and temperature (Kotanen, 1994) in rooted patches. Many perennial forbs are ruderal in strategy (**Appendix 1**); therefore, these have a greater chance of colonising disturbed areas than other strategists. Establishment of such perennial forbs in rooted areas could therefore arise from both the exposed persistent seed bank and from the efficient dispersal of the smaller sized, wind-dispersed seeds in the seed rain (Welander, 2000a).

It was hypothesised that the coverage of moss would be greater in rooted than non-rooted and in fenced than unfenced plots in the short-term within woodland and woodland ride habitats. Wild boar have been reported to aid the dispersal of moss fragments to different rooted patches in their fur and hooves (Heinken *et al.*, 2001). Although the newly exposed rooted soil is likely to favour the establishment of mosses in the short-term since some are typically early successional colonisers (Rose, 1981), the potentially drier, brighter soil conditions of more frequently applied rooting (Brownlow, 1994; Kotanen, 1994) could reduce the growth of some mosses that require moist conditions for their life-cycles. Therefore, unfenced plots that are susceptible to frequent rooting were predicted to have a lesser cover of moss than fenced plots.

Woody species are characterised by persistent seedling banks (Grime, 1979; Grime *et al.*, 1989) through the production of small numbers of large seeds as and when recourses are available (Crawley, 1997) and when juvenile, are highly susceptible to disturbance (such as rooting and trampling by boar) and predation. Large seeds such as acorns are at higher risk of predation (Crawley, 1997) and are likely to be consumed by boar whilst rooting (Henry and Conley, 1972; Baber and Coblentz, 1987; Groot Bruinderink and Hazebroek, 1994; Massei *et* *al.*, 1996; Goulding *et al.*, 1998; Goulding, 2003b). Therefore, I hypothesised that the coverage of woody species would be lower in rooted than non-rooted and in unfenced than fenced plots in the short-term within the three habitat types.

I hypothesised that site differences would significantly affect the relative abundance of all plant functional groups within each habitat type. The different forms of management across sites create varying levels of disturbance that plant functional groups are differentially sensitive to in relation to germination and growth (Grime, 1979; Grime *et al.*, 1989). For example, in grassland, **G1**, and in particular **G2** are relatively undisturbed sites compared to **G3** and **G4**, which are frequently grazed and thus have relatively high levels of disturbance (**Table 1.1**). Annual and biennial forbs tend to be ruderal in strategy (Grime, 1979; Grime *et al.*, 1989) (**Appendix 1**) and thus proliferate in disturbed environments. Whereas, perennial graminoids, which represent a range of combinations of adaptive strategies (Grime, 1979; Grime *et al.*, 1989) (**Appendix 1**), are less likely to proliferate in the more disturbed environments. Additionally, the neighbouring plant communities to the study sites are likely to contribute to differences in the relative abundance of functional groups through dispersal from the seed rain. For example, seeds of annual forbs tend to be wind dispersed and so the relative abundance of annual forbs in neighbouring plant communities is likely to greatly affect their abundance in the neighbouring study sites.

## 2.2 Materials and Methods

### *2.2.1 Study Area*

This study was undertaken within the fourteen sites (**W1**-**W5**, **R1**-**R5**, **G1**-**G4**) across the three habitat types (**Table 1.1**) within the study area (**Fig 1.5 a-d**). The study sites were surveyed for recent rooting evidence. The rooted patches utilised in the study were at least 0.5m x 0.5m, and rooted no greater than three months previously to the start of the study in order that surveying could encompass the early changes that took place after rooting had occurred.

### *2.2.2 Experimental Design*

During April 2002, 50cm x 50cm fenced plots forming secure exclosures were set up around newly rooted and non-rooted patches. Each of the five woodland sites and four grassland sites were assigned twenty plots, divided into four treatments; fenced and un-fenced, rooted and non-rooted areas, five per treatment per site. Thus, there were twenty-five replicates of each treatment in woodland and twenty replicates in grassland. Each of the five woodland ride sites was assigned five rooted and five non-rooted unfenced plots. Plots were marked with fluorescent painted wooden stakes at each corner (**Fig 2.1 a-c**). Reinforced steel mesh cages (L80cm x W80cm x H50cm) made from 0.6cm reinforced steel mesh with 20cm x 10cm apertures (**Fig 2.1 a-c**) was used for fencing. The mesh size enabled all small mammals (including small badgers) to move freely through the cages. Small wild boar piglets could potentially move through the cages but they tend to stay with their mothers that cannot enter. No evidence of wild boar was found inside the fenced plots; I concluded that the caging was successful in excluding the boar. There was a 15cm buffer zone between the plot in the centre and the steel mesh surround. Fences were fixed in place with 50cm steel pegs.

All rooted treatments across the three habitats, were set up on soil that had been rooted within the previous three months. All rooted treatments were comparable in that no living plants were present when the plot areas were designated treatments, and almost no living plants were present at the beginning of the study. In contrast, all non-rooted treatments had a full coverage of plants, normal for the local area.

Although boar may be arbitrarily selecting areas to root, the selection process could be choice-based and dependent on information such as available food present hence the areas that become rooted could have been fundamentally different from the non-rooted areas at the beginning of the study. Therefore, the possibility exists that the non-rooted fenced and unfenced plots did not effectively control for the rooted fenced and unfenced plots.

**a)**

**b)**

**c)**

**Fig 2.1 a-c:** Examples of treatments used in this study within the study area. **a)** = rooted fenced treatment in grassland site **G3**, **b)** = non-rooted fenced treatment in grassland site **G2**, and **c)** = rooted unfenced treatment in woodland ride site **R5**. Photos taken 2002.

There were 230 plots in total within the three-habitat types: 45 rooted and 45 non-rooted fenced plots and 70 rooted and 70 non-rooted unfenced plots. There were 100 plots in woodland, 80 plots in grassland and 50 plots in woodland rides. The discrepancy in the number of plots per habitat type was because I was not allowed to use fenced exclosures in woodland rides for public safety reasons. In addition, I was unable to establish permission for use of more than four grassland sites.

Data collection was carried out during May and August/September 2002 and 2003. Plant species were identified using Phillips (1980), Rose (1981), Garrard and Streeter (1983) and Fitter (1987). The percentage cover of each plant species within each plot was assessed independently, ignoring overlaps, so that each species had a potential maximum of 100% cover, and was estimated by eye within each of the 0.5m x 0.5m plots.

Data were analysed in three ways:

1. **Community level** variables measured were:

- Total percentage cover per plot per treatment per habitat per season.

- Species richness (number of plant species) per plot per treatment per habitat per season.
- Frequency of AWI species per plot per treatment per habitat per season.

- Shannon Weiner Diversity Index ($\mathbf{H' = -\sum p_i \log p_i}$) and Shannon Evenness ($\mathbf{E = H' / \log S}$) per plot per treatment per habitat per season. ($\mathbf{p_i}$ = proportion of total sample belonging to the *i*th species. **H'** = index of species diversity, **S** = number of species, **E** = evenness), (Magurran, 2004).

2. **Functional group level** variables measured were:

- Percentage cover or frequency of occurrence of annual and perennial forbs, perennial graminoids, mosses and woody species per plot per treatment per habitat per season. Where possible mean percentage cover was used for analysis. Frequency of occurrence was used where percent covers were too small for analysis such as presence of several typically small species with tiny percentage covers.

## 2.3 Statistical Analysis

For data collected within woodland and grassland habitats, data were analysed with three-way ANOVA (Eddison, 2000; Dytham, 2003). For data collected within woodland rides, data were analysed with two-way ANOVA where data were normally distributed, and where data were not normally distributed and could not be transformed, the Scheirer Ray Hare test was used instead (two-way ANOVA equivalent for non-parametric data) (Eddison, 2000; Dytham, 2003). Data were log (Log10 x +1) or square root transformed to achieve normality where possible (specified **Table 2.1**, **2.2**, **2.3**, **2.4**, **2.5**, **2.6**). For data analysed from woodland and grassland, rooting and fencing were two fixed factors, site was the third and random factor and the interaction effect was analysed between rooting and fencing. For data analysed from woodland rides, rooting was the fixed factor and site was the second and random factor. All data were analysed using sequential sum of squares. Seven of the unfenced non-rooted plots became rooted before the spring 2003 data collection, hence the General Linear Model was used for analysis due to its ability to account for an unbalanced design (Eddison, 2000; Dytham, 2003).

## 2.4 Results

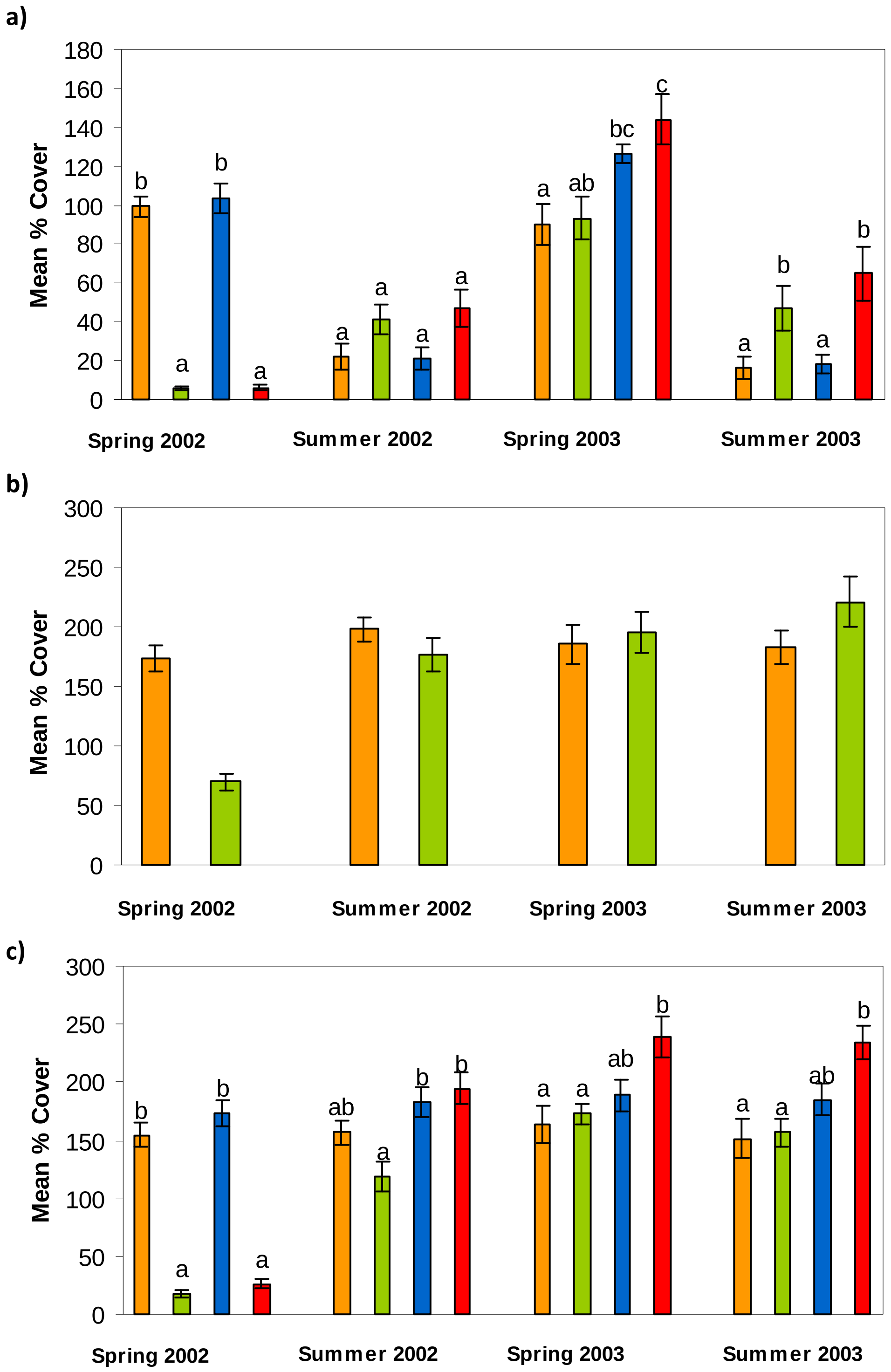
### 2.4.1 ***Community Level Effects***

There were consistent highly significant rooting effects for total plant cover in woodland, during spring 2002 and summer 2003 in woodland rides, and spring 2002 and spring and summer 2003 in grassland (**Table 2.1**, **2.2**, **2.3**, **Fig 2.2**). In woodland and grassland, one year after rooting (and after seventeen months in grassland) there was significantly greater plant cover in fenced rooted than both unfenced treatments, and after seventeen months in woodland there was significantly greater plant cover in both rooted than non-rooted treatments (**Appendices 2-4**, **Fig 2.2**). By summer 2003, plant cover was significantly greater in rooted than non-rooted treatments in woodland rides.

Highly significant fencing effects in spring 2002 and during 2003 in grassland shows that plant cover is the community level attribute most highly affected by fencing, and moreover, grassland is the only habitat in which fencing has an effect at this level (**Table 2.3**). Highly significant site effects during spring 2002 and summer 2003 in woodland, during spring and summer 2002 in woodland rides and consistently in grassland, show that site differences affect the abundance of plants independently of rooting.

**Fig 2.2.** Effects of rooting and fencing on total plant cover in **a)** woodland, **b)** woodland rides and **c)** grassland over spring and summer during 2002 and 2003. Differences between treatments (indicated by letters on the graphs) refer only to within each sampling period and were analysed using Tukey tests (significance level **$p<0.05$**). Error bars indicate standard error of the means. Woodland: n = 25, Woodland rides: n = 25, Grassland: n = 20. For ANOVA results see **Tables 2.1**, **2.2**, **2.3**.

🟧 Non-Rooted Unfenced 🟩 Rooted Unfenced 🟦 Non-Rooted Fenced 🟥 Rooted Fenced



Consistently significant and highly significant rooting effects in the three habitat types, show that rooting had a large impact on the species richness of plants (**Table 2.1**, **2.2**, **2.3**). Plant species richness was significantly greater in both rooted treatments in woodland, woodland rides and grassland. In grassland, in summer 2002 there was significantly greater plant species richness in fenced rooted than non-rooted treatments, and in 2003 there was greater species richness in both rooted than non-rooted treatments. There is a similar trend in woodland (**Fig 2.3**), although in spring 2003, multi-comparison tests show a significant difference between both rooted, and non-rooted unfenced treatments. A significant fencing effect in spring 2002 and summer 2003 indicates that fencing has some affect on species richness but only in grassland (**Table 2.3**). There are highly significant site effects during 2002 in the three habitat types, in spring 2003 in woodland, and spring and summer 2003 in grassland, which indicates that site differences across the three habitat types strongly affect plant species richness independently of rooting.

**Fig 2.3.** Effects of rooting and fencing on plant species richness in **a)** woodland, **b)** woodland rides and **c)** grassland over spring and summer during 2002 and 2003. Differences between treatments (indicated by letters on the graphs) refer only to within each sampling period and were analysed using Tukey tests (significance level **p<0.05**). Error bars indicate standard error of the means. Woodland: n = 25, Woodland rides: n = 25, Grassland: n = 20. For ANOVA results see **Tables 2.1**, **2.2**, **2.3**.

🟧 Non-Rooted Unfenced 🟩 Rooted Unfenced 🟦 Non-Rooted Fenced 🟥 Rooted Fenced

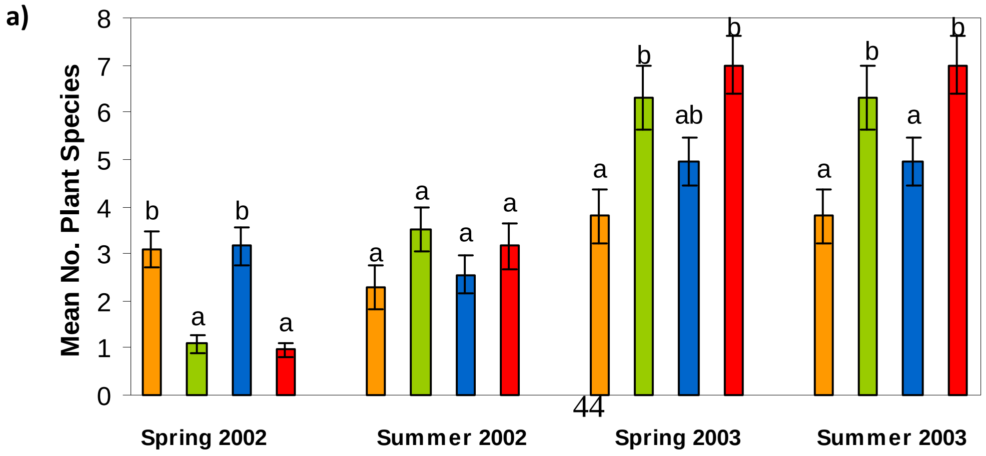
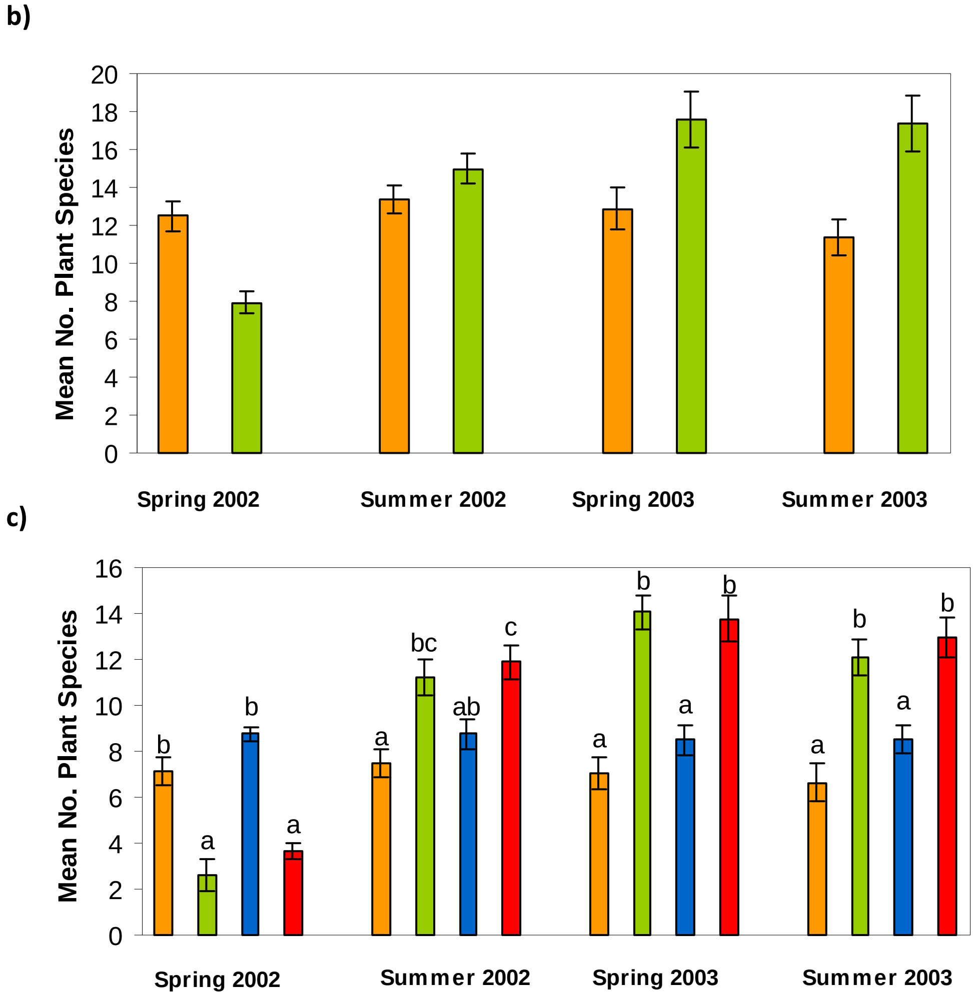


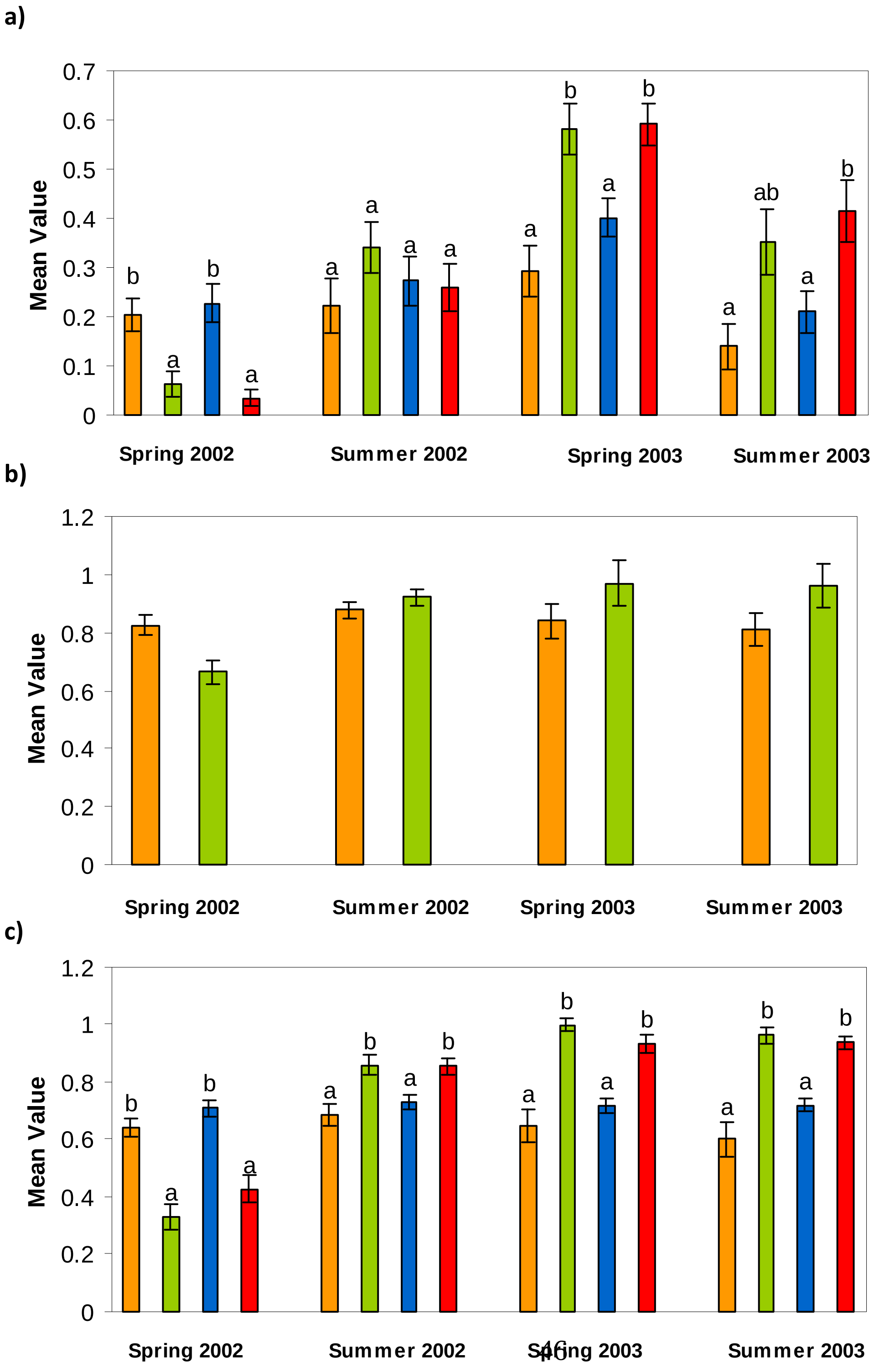
**Fig 2.3** continued



The Shannon diversity index shows consistently highly significant treatment effects in grassland, woodland rides and in 2002 and summer 2003 in woodland (**Table 2.1**, **2.2**, **2.3**). After the initial effects of rooting, Shannon diversity was significantly greater in rooted than non-rooted treatments in grassland and woodland rides, and after one year in woodland. During summer 2003 in woodland, plant diversity was significantly greater in fenced rooted than unfenced non-rooted treatments (**Fig 2.4**). Significant site effects in 2002 and summer 2003 in woodland, spring 2003 in woodland rides, and in 2003 in grassland show that site differences strongly affect the diversity of plants independently of rooting.

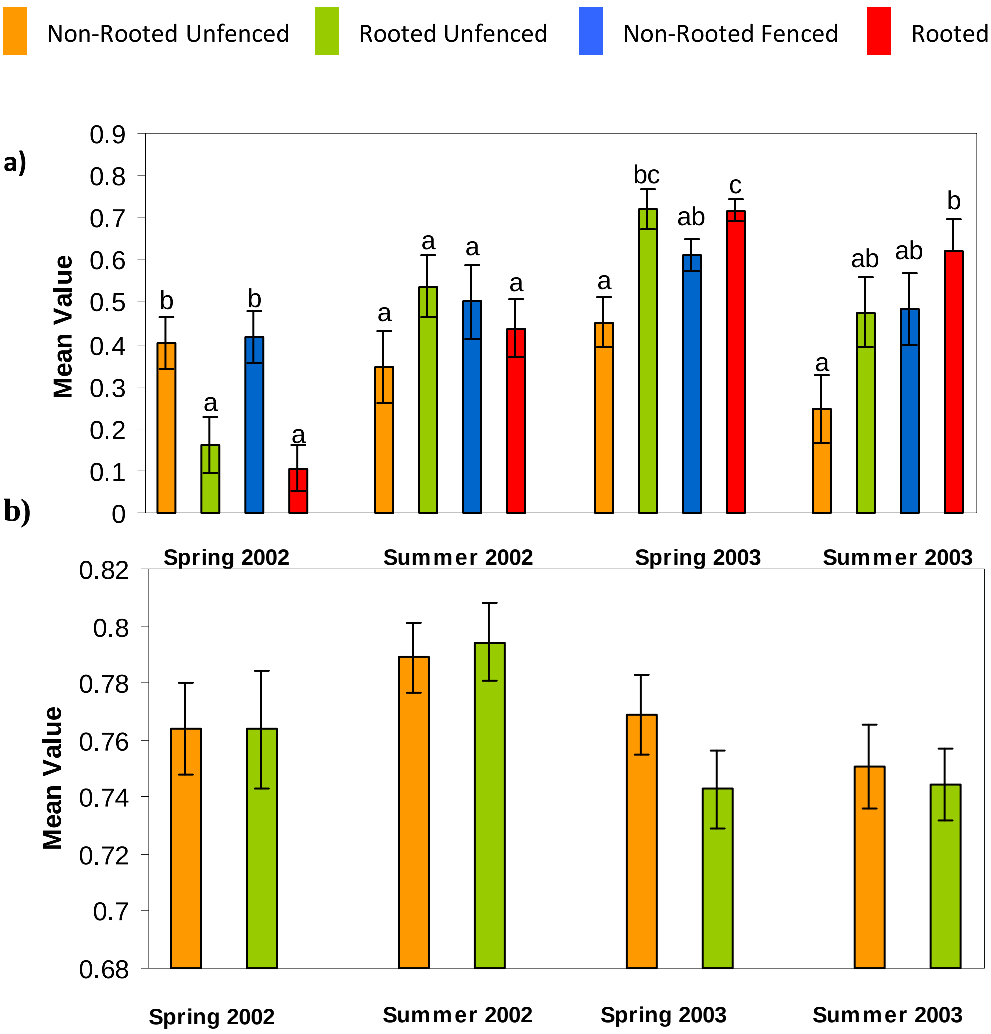
**Fig 2.4.** Effects of rooting and fencing on Shannon diversity in **a)** woodland, **b)** woodland rides and **c)** grassland over spring and summer during 2002 and 2003. Differences between treatments (indicated by letters on the graphs) refer only to within each sampling period and were analysed using Tukey tests (significance level **$p<0.05$**). Error bars indicate standard error of the means. Woodland: n = 25, Woodland rides: n = 25, Grassland: n = 20. For ANOVA results see **Tables 2.1**, **2.2**, **2.3**.

🟧 Non-Rooted Unfenced 🟩 Rooted Unfenced 🟦 Non-Rooted Fenced 🟥 Rooted Fenced

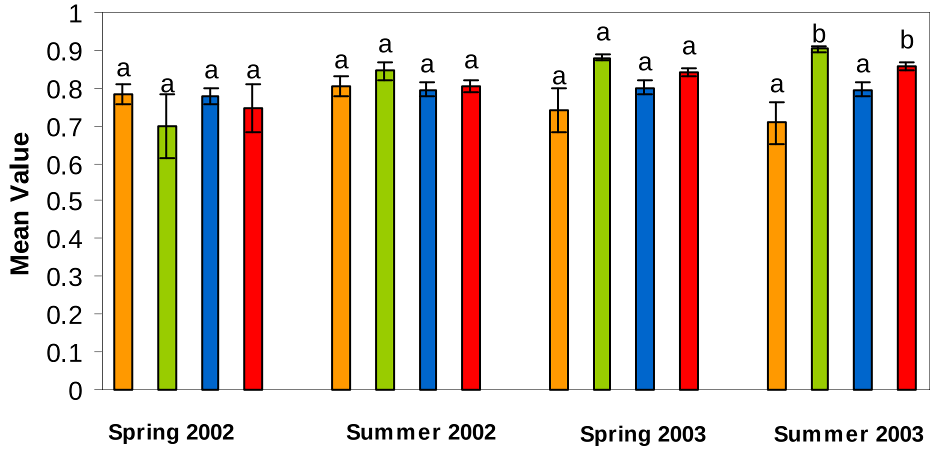


Evenness did not respond consistently to rooting. Significant rooting effects in 2003 in woodland rides and in spring 2002 in woodland shows that evenness was most notably unaffected by rooting in the short-term (**Table 2.1**, **2.2**, **2.3**). Significant site effects during 2003 in woodland and grassland, reveals that site differences affect evenness independently of rooting. Multi-comparison tests however show that in summer 2003 in grassland, evenness was higher in rooted than non-rooted treatments, and in woodland in 2003, evenness was significantly greater in fenced rooted than unfenced non-rooted treatments (**Fig 2.5**).

**Fig 2.5.** Effects of rooting and fencing on Shannon evenness in **a)** woodland, **b)** woodland rides and **c)** grassland over spring and summer during 2002 and 2003. Differences between treatments (indicated by letters on the graphs) refer only to within each sampling period and were analysed using Tukey tests (significance level **$p<0.05$**). Error bars indicate standard error of the means. Woodland: n = 25, Woodland rides: n = 25, Grassland: n = 20. For ANOVA results see **Tables 2.1**, **2.2**, **2.3**.



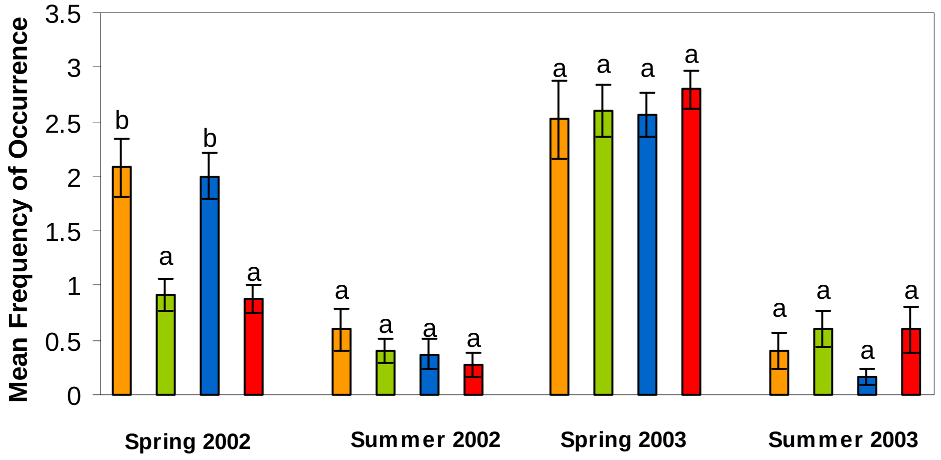
c)



Other than the initial effects of rooting, there was no difference in the frequency of occurrence of AWI species between treatments (Fig 2.6). Significant site effects in spring 2002 and during 2003 show that site differences had the greatest influence on the abundance of AWI species (Table 2.1, 2.2). A solitary interaction effect however in spring 2003 shows that rooting affects the abundance of AWI species, although this effect is determined by site.

**Fig 2.6.** Effects of rooting and fencing on AWI species in woodland over spring and summer during 2002 and 2003. Differences between treatments (indicated by letters on the graphs) refer only to within each sampling period and were analysed using Tukey tests (significance level **p<0.05**). Error bars indicate standard error of the means. Woodland: n = 25. For ANOVA results see **Table 2.1**.

🟧 Non-Rooted Unfenced 🟩 Rooted Unfenced 🟦 Non-Rooted Fenced 🟥 Rooted Fenced



## Effects of Rooting, Fencing and Site on Plants at the Community Level in Woodland

| Source | Transform | F value & df | P | Transform | F value & df | P | Transform | F value & df | P | Transform | F value & df | P |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Spring 2002 | | | Summer 2002 | | | Spring 2003 | | | Summer 2003 | | |
| ***Total Plant Cover*** (%) | $Log_{10}(x+1)$ | | | None | | | None | | | None | | |
| Site | | $F_{4,92} = 9.65$ | **< 0.001** | | $F_{4,92} = 1.36$ | 0.256 | | $F_{4,92} = 0.78$ | 0.539 | | $F_{4,92} = 3.84$ | **0.006** |
| Fencing | | $F_{1,92} = 0.03$ | 0.853 | | $F_{1,92} = 0.11$ | 0.741 | | $F_{1,92} = 0.99$ | 0.323 | | $F_{1,92} = 1.09$ | 0.299 |
| Rooting | | $F_{1,92} = 566.10$ | **< 0.001** | | $F_{1,92} = 8.65$ | **0.004** | | $F_{1,92} = 17.86$ | **< 0.001** | | $F_{1,92} = 16.90$ | **< 0.001** |
| Fencing*Rooting | | $F_{1,92} = 0.01$ | 0.909 | | $F_{1,92} = 0.20$ | 0.653 | | $F_{1,92} = 0.48$ | 0.489 | | $F_{1,92} = 0.69$ | 0.407 |
| ***Species Richness*** | $Log_{10}(x+1)$ | | | None | | | None | | | None | | |
| Site | | $F_{4,92} = 8.31$ | **< 0.001** | | $F_{4,92} = 3.81$ | **0.007** | | $F_{4,92} = 1.54$ | 0.196 | | $F_{4,92} = 5.86$ | **< 0.001** |
| Fencing | | $F_{1,92} = 0.04$ | 0.850 | | $F_{1,92} = 0.01$ | 0.928 | | $F_{1,92} = 2.47$ | 0.119 | | $F_{1,92} = 0.85$ | 0.358 |
| Rooting | | $F_{1,92} = 83.66$ | **< 0.001** | | $F_{1,92} = 4.37$ | **0.039** | | $F_{1,92} = 15.18$ | **< 0.001** | | $F_{1,92} = 21.35$ | **< 0.001** |
| Fencing*Rooting | | $F_{1,92} = 0.09$ | 0.769 | | $F_{1,92} = 0.53$ | 0.469 | | $F_{1,92} = 0.17$ | 0.683 | | $F_{1,92} = 0.00$ | 0.968 |
| ***Shannon Diversity*** | Square Root | | | None | | | None | | | None | | |
| Site | | $F_{4,92} = 2.97$ | **0.024** | | $F_{4,92} = 4.20$ | **0.004** | | $F_{4,92} = 0.177$ | 0.143 | | $F_{4,92} = 5.48$ | **0.001** |
| Fencing | | $F_{1,92} = 0.07$ | 0.792 | | $F_{1,92} = 0.11$ | 0.740 | | $F_{1,92} = 0.170$ | 0.195 | | $F_{1,92} = 1.74$ | 0.190 |
| Rooting | | $F_{1,92} = 48.77$ | **< 0.001** | | $F_{1,92} = 1.19$ | 0.278 | | $F_{1,92} = 27.42$ | **< 0.001** | | $F_{1,92} = 16.76$ | **< 0.001** |
| Fencing*Rooting | | $F_{1,92} = 0.33$ | 0.569 | | $F_{1,92} = 1.85$ | 0.177 | | $F_{1,92} = 1.15$ | 0.287 | | $F_{1,92} = 0.00$ | 0.958 |
| ***Shannon Evenness*** | None | | | None | | | None | | | None | | |
| Site | | $F_{4,92} = 4.02$ | **0.005** | | $F_{4,92} = 1.02$ | 0.400 | | $F_{4,92} = 4.53$ | **0.002** | | $F_{4,92} = 2.87$ | **0.027** |
| Fencing | | $F_{1,92} = 1.27$ | 0.262 | | $F_{1,92} = 0.22$ | 0.644 | | $F_{1,92} = 0.28$ | 0.601 | | $F_{1,92} = 0.22$ | 0.638 |
| Rooting | | $F_{1,92} = 7.39$ | **0.008** | | $F_{1,92} = 0.44$ | 0.511 | | $F_{1,92} = 0.35$ | 0.555 | | $F_{1,92} = 0.54$ | 0.463 |
| Fencing*Rooting | | $F_{1,92} = 0.10$ | 0.752 | | $F_{1,92} = 2.24$ | 0.138 | | $F_{1,92} = 0.51$ | 0.476 | | $F_{1,92} = 0.14$ | 0.713 |
| ***A.W.I Species*** (*Freq. Occ.*) | None | | | None | | | None | | | None | | |
| Site | | $F_{4,92} = 4.48$ | **0.002** | | $F_{4,92} = 1.32$ | 0.267 | | $F_{4,92} = 2.61$ | **0.040** | | $F_{4,92} = 2.74$ | **0.033** |
| Fencing | | $F_{1,92} = 0.01$ | 0.924 | | $F_{1,92} = 0.02$ | 0.888 | | $F_{1,92} = 0.45$ | 0.503 | | $F_{1,92} = 0.26$ | 0.609 |
| Rooting | | $F_{1,92} = 7.27$ | **0.001** | | $F_{1,92} = 0.02$ | 0.888 | | $F_{1,92} = 0.25$ | 0.615 | | $F_{1,92} = 2.38$ | 0.127 |
| Fencing*Rooting | | $F_{1,92} = 0.75$ | 0.390 | | $F_{1,92} = 3.39$ | 0.069 | | $F_{1,92} = 4.08$ | **0.046** | | $F_{1,92} = 0.307$ | 0.307 |

**Table 2.1**. Three-way ANOVA test results at significance levels; Non-Significant: $p > 0.05$, Significant: **$p < 0.05$, $p < 0.01$**, Highly Significant: **$p < 0.001$**. Data were analysed using sequential sum of squares for tests. **F** = test statistic. **P** = probability associated with F-test. Plant data collected in woodland during spring and summer 2002 and 2003. Significant **P** values shown in bold. Site = random factor. Fencing and Rooting = Fixed factors. Fencing*Rooting = Interaction. Freq. Occ = Frequency of Occurrence. Transform = method of normalising data.

## Effects of Rooting and Site on Plants at the Community Level in Woodland Rides

| Source | Transform | F value & df | P | Transform | F value & df | P | Transform | F value & df | P | Transform | F value & df | P |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Spring 2002 | | | Summer 2002 | | | Spring 2003 | | | Summer 2003 | | |
| ***Total Plant Cover*** (%) | None | | | None | | | None | | | None | | |
| Site | | $F_{4,44} = 5.97$ | **0.001** | | $F_{4,44} = 8.72$ | **< 0.001** | | $F_{4,44} = 2.15$ | 0.091 | | $F_{4,44} = 1.54$ | 0.207 |
| Rooting | | $F_{1,44} = 93.01$ | **< 0.001** | | $F_{1,44} = 2.33$ | 0.134 | | $F_{1,44} = 0.21$ | 0.647 | | $F_{1,44} = 2.38$ | **0.036** |
| | | | | | | | | | | | | |
| ***Species Richness*** | Square Root | | | None | | | None | | | None | | |
| Site | | $F_{4,44} = 4.56$ | **0.004** | | $F_{4,44} = 12.22$ | **< 0.001** | | $F_{4,44} = 1.86$ | 0.134 | | $F_{4,44} = 2.05$ | 0.104 |
| Rooting | | $F_{1,44} = 27.48$ | **< 0.001** | | $F_{1,44} = 4.30$ | **0.044** | | $F_{1,44} = 7.13$ | **0.011** | | $F_{1,44} = 12.38$ | **0.001** |
| | | | | | | | | | | | | |
| ***Shannon Diversity*** | None | | | None | | | None | | | None | | |
| Site | | $F_{4,44} = 2.13$ | 0.092 | | $F_{1,44} = 1.97$ | 0.167 | | $F_{4,44} = 3.51$ | **0.014** | | $F_{4,44} = 2.13$ | 0.092 |
| Rooting | | $F_{1,44} = 12.57$ | **0.001** | | $F_{4,44} = 7.76$ | **0.001** | | $F_{1,44} = 12.64$ | **0.001** | | $F_{1,44} = 18.57$ | **< 0.001** |
| | | | | | | | | | | | | |
| ***Shannon Evenness*** | None | | | None | | | None | | | None | | |
| Site | | $F_{4,44} = 1.29$ | 0.287 | | $F_{4,44} = 1.34$ | 0.270 | | $F_{4,44} = 0.47$ | 0.756 | | $F_{4,44} = 0.59$ | 0.672 |
| Rooting | | $F_{1,44} = 0.36$ | 0.550 | | $F_{1,44} = 0.98$ | 0.328 | | $F_{1,44} = 7.16$ | **0.010** | | $F_{1,44} = 6.48$ | **0.014** |

**Table 2.2**. Two-way ANOVA or Scheirer Ray Hare test results, at significance levels: Non-Significant: $p > 0.05$, Significant: $\mathbf{p < 0.05}$, $\mathbf{p < 0.01}$, Highly Significant: $\mathbf{p < 0.001}$. Data were analysed using sequential sum of squares for tests. Red data indicates Scheirer Ray hare test results. **F** = test statistic. **P** = probability associated with F-test. Plant data collected in rides during spring and summer 2002 and 2003. Significant **P** values shown in bold. Site = random factor. Rooting = Fixed factor. Transform = method of normalising data.

## Effects of Rooting, Fencing and Site on Plants at the Community Level in Grassland

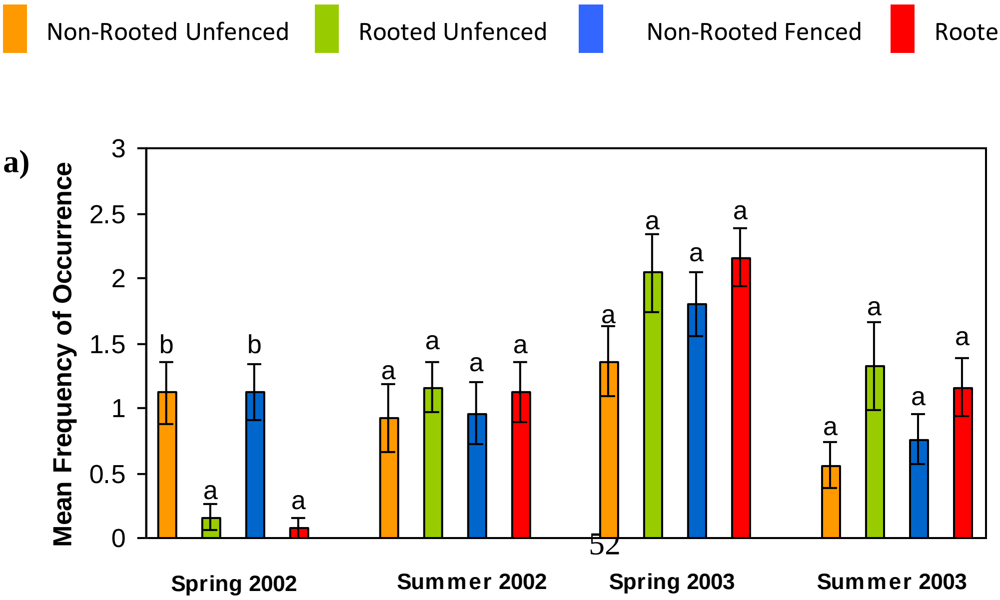
| Source | Transform | F value & df | P | Transform | F value & df | P | Transform | F value & df | P | Transform | F value & df | P |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Spring 2002 | | | Summer 2002 | | | Spring 2003 | | | Summer 2003 | | |
| ***Total Plant Cover** (%)* | None | | | None | | | None | | | None | | |
| Site | | $F_{3,73}$ = 7.42 | **< 0.001** | | $F_{3,73}$ = 10.19 | **< 0.001** | | $F_{3,73}$ = 10.74 | **< 0.001** | | $F_{3,73}$ = 12.99 | **< 0.001** |
| Fencing | | $F_{1,73}$ = 3.49 | 0.066 | | $F_{1,73}$ = 23.04 | **< 0.001** | | $F_{1,73}$ = 13.50 | **< 0.001** | | $F_{1,73}$ = 22.08 | **< 0.001** |
| Rooting | | $F_{1,73}$ = 380.24 | **< 0.001** | | $F_{1,73}$ = 1.53 | 0.220 | | $F_{1,73}$ = 5.63 | **0.020** | | $F_{1,73}$ = 5.28 | **0.024** |
| Fencing*Rooting | | $F_{1,73}$ = 0.50 | 0.482 | | $F_{1,73}$ = 5.38 | **0.023** | | $F_{1,73}$ = 2.68 | 0.106 | | $F_{1,73}$ = 3.40 | 0.069 |
| ***Species Richness*** | $\text{Log}_{10}$ (x+1) | | | Square Root | | | None | | | None | | |
| Site | | $F_{3,73}$ = 3.55 | **0.018** | | $F_{3,73}$ = 5.07 | **0.003** | | $F_{3,73}$ = 9.50 | **< 0.001** | | $F_{3,73}$ = 14.60 | **< 0.001** |
| Fencing | | $F_{1,73}$ = 8.14 | **0.006** | | $F_{1,73}$ = 2.80 | 0.098 | | $F_{1,73}$ = 0.72 | 0.400 | | $F_{1,73}$ = 4.86 | **0.031** |
| Rooting | | $F_{1,73}$ = 91.00 | **< 0.001** | | $F_{1,73}$ = 27.63 | **< 0.001** | | $F_{1,73}$ = 81. 30 | **< 0.001** | | $F_{1,73}$ = 62.39 | **< 0.001** |
| Fencing*Rooting | | $F_{1,73}$ = 0.30 | 0.584 | | $F_{1,73}$ = 0.30 | 0.584 | | $F_{1,73}$ = 1.66 | 0.202 | | $F_{1,73}$ = 0.71 | 0.403 |
| ***Shannon Diversity*** | None | | | None | | | $\text{Log}_{10}$ (x+1) | | | None | | |
| Site | | $F_{3,73}$ = 2.30 | 0.084 | | $F_{3,73}$ = 1.77 | 0.161 | | $F_{3,73}$ = 4.30 | **0.008** | | $F_{3,73}$ = 6.56 | **0.001** |
| Fencing | | $F_{1,73}$ = 4.59 | **0.035** | | $F_{1,73}$ = 0.39 | 0.535 | | $F_{1,73}$ = 0.19 | 0.663 | | $F_{1,73}$ = 2.07 | 0.154 |
| Rooting | | $F_{1,73}$ = 60.48 | **< 0.001** | | $F_{1,73}$ = 23.99 | **< 0.001** | | $F_{1,73}$ = 53.62 | **< 0.001** | | $F_{1,73}$ = 77.75 | **< 0.001** |
| Fencing*Rooting | | $F_{1,73}$ = 0.17 | 0.684 | | $F_{1,73}$ = 0.57 | 0.453 | | $F_{1,73}$ = 3.93 | 0.051 | | $F_{1,73}$ = 4.69 | **0.034** |
| ***Shannon Evenness*** | None | | | None | | | None | | | None | | |
| Site | | $F_{3,73}$ = 0.50 | 0.685 | | $F_{3,73}$ = 1.11 | 0.349 | | $F_{3,73}$ = 3.39 | **0.022** | | $F_{3,73}$ = 7.33 | **< 0.001** |
| Fencing | | $F_{1,73}$ = 0.122 | 0.122 | | $F_{1,73}$ = 0.01 | 0.941 | | $F_{1,73}$ = 0.41 | 0.522 | | $F_{1,73}$ = 0.44 | 0.511 |
| Rooting | | $F_{1,73}$ = 0.74 | 0.391 | | $F_{1,73}$ = 2,70 | 0.105 | | $F_{1,73}$ = 0.93 | 0.337 | | $F_{1,73}$ = 0.23 | 0.631 |
| Fencing*Rooting | | $F_{1,73}$ = 1.36 | 0.296 | | $F_{1,73}$ = 1.72 | 0.193 | | $F_{1,73}$ = 0.05 | 0.822 | | $F_{1,73}$ = 0.08 | 0.783 |

**Table 2.3.** Three-way ANOVA test results, at significance levels; Non-Significant: $p > 0.05$, Significant: **$p < 0.05$, $p < 0.01$**, Highly Significant: **$p < 0.001$**. Data were analysed using sequential sum of squares for tests. **F** = test statistic. **P** = probability associated with F-test. Plant data collected in grassland during spring and summer 2002 and 2003 Significant **P** values shown in bold. Site = random factor. Fencing and Rooting = Fixed factors. Fencing*Rooting = Interaction. Transform = method of normalising data.

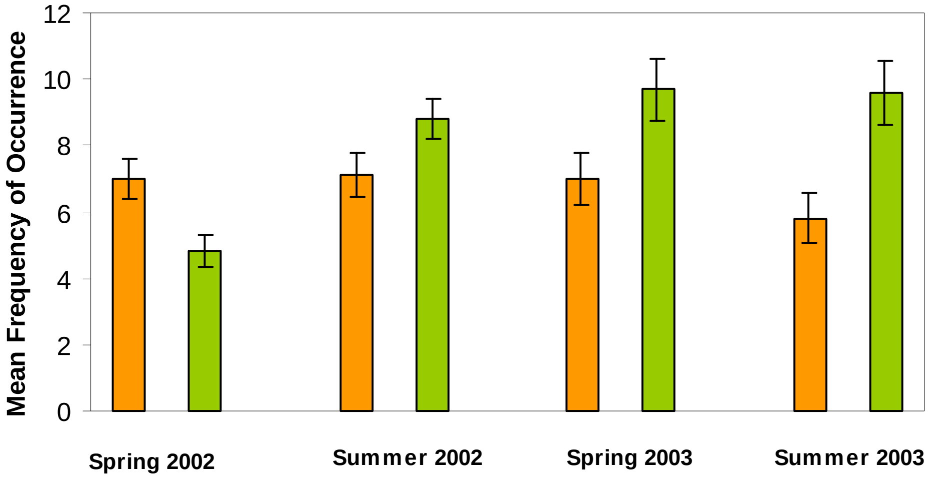
### *2.4.2 Functional Group Level Effects*

Consistent highly significant rooting and site effects in grassland and woodland rides, show that both rooting and site differences substantially affects the frequency of occurrence of perennial forbs in these habitats (**Table 2.4**, **2.5**, **2.6**). Other than the initial effects of rooting, there were a significantly greater number of perennial forbs in rooted than non-rooted treatments in grassland and woodland rides (**Fig 2.7**). There appears, from the graph and from significant rooting effects in spring 2002 and during 2003, to be a similar trend for woodland (**Fig 2.7**), although multi-comparison tests show there to be no significant difference between treatments. Significant site effects in spring 2002 and summer 2003 in woodland show that site differences also affect the numbers of perennial forbs, but not as significantly as rooting in this habitat type. Additionally, and similarly with community level effects, a highly significant fencing effect exists in summer 2002, but only in grassland (**Table 2.6**).

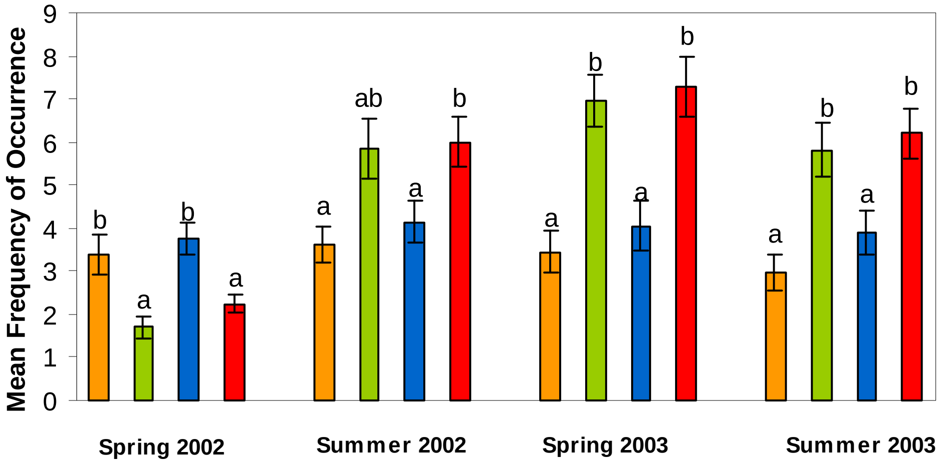
**Fig 2.7.** Effects of rooting and fencing on perennial forbs in **a)** woodland, **b)** woodland rides and **c)** grassland over spring and summer during 2002 and 2003. Differences between treatments (indicated by letters on the graphs) refer only to within each sampling period and were analysed using Tukey tests (significance level **p<0.05**). Error bars indicate standard error of the means. Woodland: n = 25, Woodland rides: n = 25, Grassland: n = 20. For ANOVA results see **Tables 2.4**, **2.5**, **2.6**.



**b)**

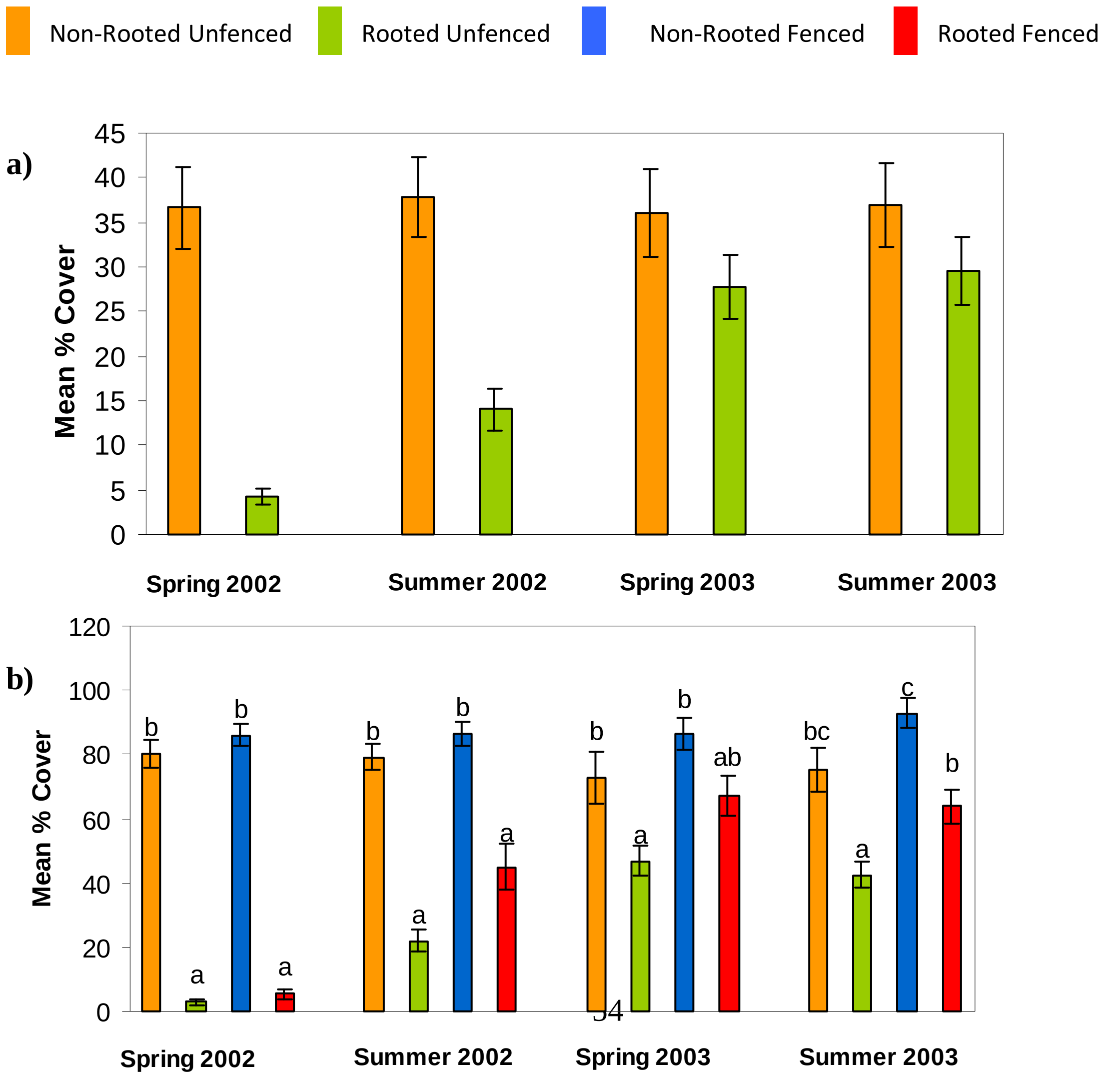


**c)**



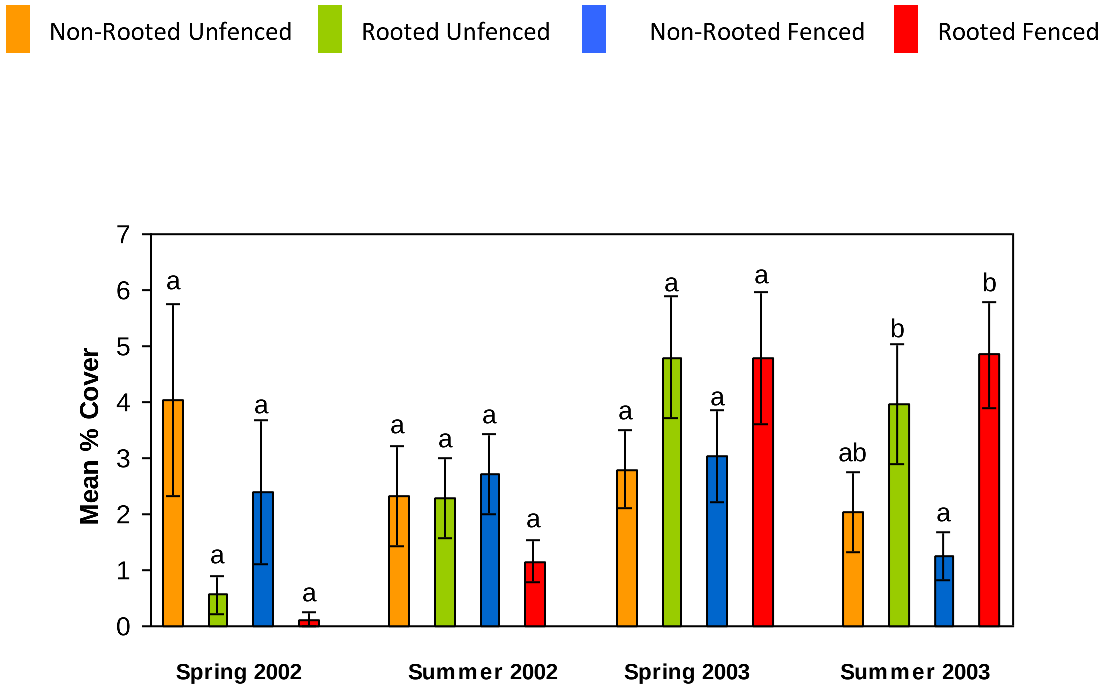
The opposite trend to perennial forbs can be seen for perennial graminoids, whereby consistent highly significant rooting effects in grassland, and during 2002 in woodland rides, show that there was a substantially lesser percentage cover of perennial graminoids within rooted than non-rooted treatments within these habitats (**Table 2.5**, **2.6**). The percentage cover of perennial graminoids in grassland was significantly lower in rooted than non-rooted treatments in 2002, and in unfenced rooted than non-rooted treatments in spring 2003, and unfenced rooted than all other treatments in summer 2003 (**Fig 2.8**). In woodland rides, there was a significantly smaller cover of perennial graminoids in rooted than non-rooted treatments during 2002, but not in 2003 although there was a similar trend (**Fig 2.8**). Consistent significant site effects in grassland show that site differences strongly affects the cover of perennial graminoids in this habitat type. A single significant site effect in spring 2003 in woodland rides indicates that site differences affect perennial graminoid distribution independently of rooting, but this effect is not consistent and suggests the influence of some transient aspect of site difference. Additionally, and similarly with community level effects, highly significant fencing effects exist in summer 2002 and during 2003, but only in grassland (**Table 2.6**).

**Fig 2.8.** Effects of rooting and fencing on perennial graminoids in **a)** woodland rides and **b)** grassland over spring and summer during 2002 and 2003. Differences between treatments (indicated by letters on the graphs) refer only to within each sampling period and were analysed using Tukey tests (significance level **p<0.05**). Error bars indicate standard error of the means. Woodland rides: n = 25, Grassland: n = 20. For ANOVA results see **Tables 2.5**, **2.6**.



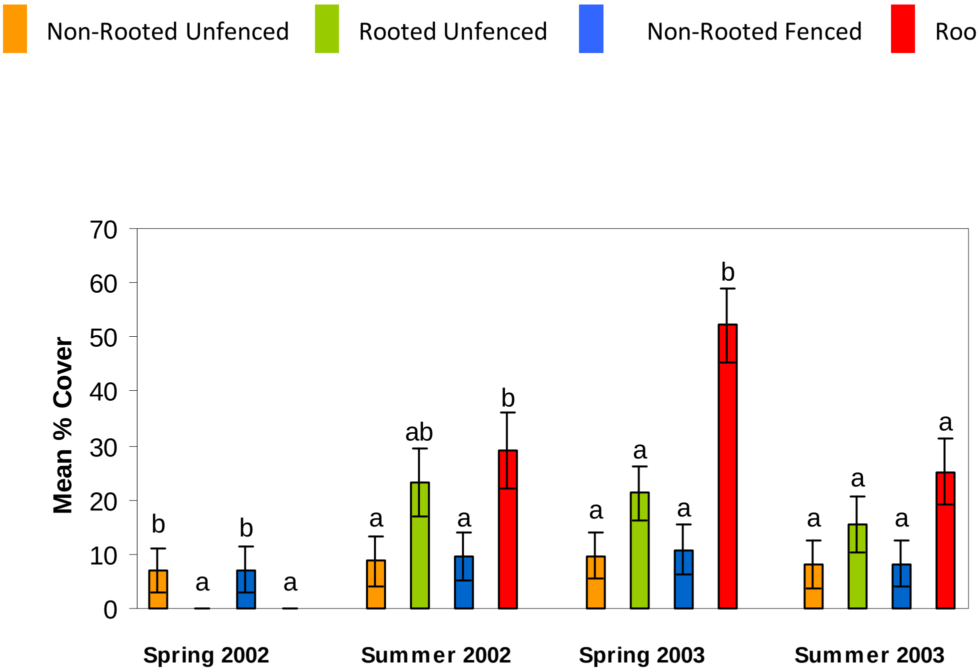
Although it appears from the graph (**Fig 2.9**) and from significant rooting effects in spring 2002 and 2003 and summer 2003 (**Table 2.4**) that the percentage cover of woody species in woodland was greatly affected by rooting, multi-comparison tests reveal little overall impact from rooting. There was no significant difference between treatments except for summer 2003, where there was significantly less cover of woody species in non-rooted fenced than all other treatments. The large standard errors (**Fig 2.9**) are indicative of substantial variation within the data; many woody species have a small percent cover (as seedlings), whilst a few have large percentage covers (as saplings). Significant site effects in spring 2002 and during 2003 in woodland show that site differences substantially affect the cover of woody species (**Table 2.4**).

**Fig 2.9.** Effects of rooting and fencing on woody species in woodland over spring and summer during 2002 and 2003. Differences between treatments (indicated by letters on the graphs) refer only to within each sampling period and were analysed using Tukey tests (significance level **p<0.05**). Error bars indicate standard error of the means. n = 25. For ANOVA results see **Table 2.4**.



Mosses were among the pioneer colonisers of bare soil after rooting in woodland. There were significant rooting effects in spring 2002 and summer 2003, and highly significant rooting effects during summer 2002 and spring 2003 for the percentage cover of moss in woodland (**Table 2.4**). However, although it appears that the graph reflects these results, multi-comparison tests reveal that the only remaining significance lies during summer 2002 and spring 2003, reflecting the highly significant results. The percentage cover of moss in woodland was significantly greater in rooted fenced than all other treatments during summer 2002 and spring 2003, but there was no significant difference between treatments in spring 2002 and summer 2003 (**Fig 2.10**). A solitary significant fencing effect exists in spring 2003, which is reflected in **Fig 2.10**. It is the only fencing effect to occur outside grassland.

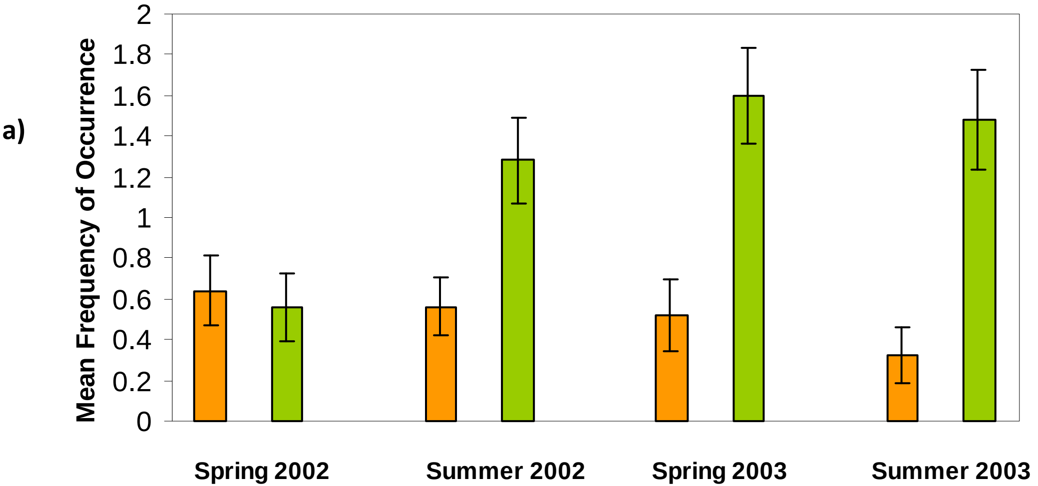
**Fig 2.10.** Effects of rooting and fencing on moss in woodland over spring and summer during 2002 and 2003. Differences between treatments (indicated by letters on the graphs) refer only to within each sampling period and were analysed using Tukey tests (significance level **p<0.05**). Error bars indicate standard error of the means. n = 25. For ANOVA results see **Table 2.4**.



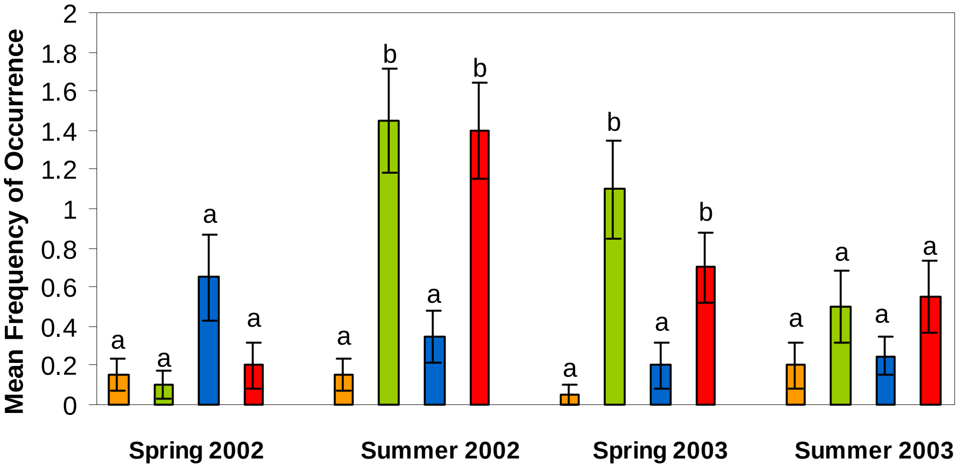
There were consistent significant and highly significant rooting effects in grassland, and highly significant rooting effects in summer 2002 and during 2003 in woodland rides (**Table 2.5**, **2.6**). Other than initial effects of rooting, it appears that the graphs reflect these results showing that the frequency of occurrence of annual forbs was significantly greater in rooted than non-rooted treatments within grassland and woodland ride habitats (**Fig 2.11**). However, in grassland, multi-comparison tests reveal that there was no significant difference between treatments during spring 2002 and most notably during summer 2003. Significant site effects in 2003 in grassland show that site differences predominantly affect the frequency of annual forbs compared to rooting during this time. Conversely, significant site effects in woodland rides during 2002 shows that the effects of site on annual forbs are more short-term in this habitat type. A solitary fencing effect in spring 2002 in grassland shows that fencing has only a small and immediate impact on the frequency of occurrence of annual forbs. There was not sufficient data within woodland habitats for analysis.

**Fig 2.11.** Effects of rooting and fencing on annual forbs in **a)** woodland rides and **b)** grassland over spring and summer during 2002 and 2003. Differences between treatments (indicated by letters on the graphs) refer only to within each sampling period and were analysed using Tukey tests (significance level **p<0.05**). Error bars indicate standard error of the means. Woodland rides: n = 25, Grassland: n = 20. For ANOVA results see **Tables 2.5**, **2.6**.

🟧 Non-Rooted Unfenced 🟩 Rooted Unfenced 🟦 Non-Rooted Fenced 🟥 Rooted Fenced



b)


Mean Frequency of Occurrence
2
1.8
1.6
1.4
1.2
1
0.8
0.6
0.4
0.2
0
a
a
a
a
a
b
a
b
a
b
a
b
a
a
a
a
Spring 2002
Summer 2002
Spring 2003
Summer 2003

## Effects of Rooting, Fencing and Site on Plants at the Functional Group Level in Woodland

| Source | Transform | F value & df | P | Transform | F value & df | P | Transform | F value & df | P | Transform | F value & df | P |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | Spring 2002 | | | Summer 2002 | | | Spring 2003 | | | Summer 2003 | |
| ***Perennial Forbs*** (*Freq. Occ.*) | Square Root | | | None | | | None | | | None | | |
| Site | | $F_{4,92} = 4.27$ | **0.003** | | $F_{4,92} = 2.27$ | 0.068 | | $F_{4,92} = 2.34$ | 0.061 | | $F_{4,92} = 3.28$ | **0.015** |
| Fencing | | $F_{1,92} = 0.03$ | 0.871 | | $F_{1,92} = 0.00$ | 1.000 | | $F_{1,92} = 1.21$ | 0.274 | | $F_{1,92} = 0.001$ | 0.931 |
| Rooting | | $F_{1,92} = 58.41$ | **< 0.001** | | $F_{1,92} = 0.77$ | 0.382 | | $F_{1,92} = 4.18$ | **0.044** | | $F_{1,92} = 6.34$ | **0.014** |
| Fencing*Rooting | | $F_{1,92} = 0.43$ | 0.513 | | $F_{1,92} = 0.03$ | 0.861 | | $F_{1,92} = 0.40$ | 0.531 | | $F_{1,92} = 0.61$ | 0.437 |
| | | | | | | | | | | | | |
| ***Moss*** (*% cover*) | None | | | None | | | None | | | None | | |
| Site | | $F_{4,92} = 1.79$ | 0.138 | | $F_{4,92} = 2.13$ | 0.083 | | $F_{4,92} = 0.46$ | 0.763 | | $F_{4,92} = 1.08$ | 0.372 |
| Fencing | | $F_{1,92} = 0.00$ | 0.961 | | $F_{1,92} = 0.37$ | 0.546 | | $F_{1,92} = 8.98$ | **0.004** | | $F_{1,92} = 0.96$ | 0.329 |
| Rooting | | $F_{1,92} = 5.85$ | **0.018** | | $F_{1,92} = 9.29$ | **0.003** | | $F_{1,92} = 24.86$ | **< 0.001** | | $F_{1,92} = 5.84$ | **0.018** |
| Fencing*Rooting | | $F_{1,92} = 0.00$ | 0.961 | | $F_{1,92} = 0.21$ | 0.645 | | $F_{1,92} = 7.84$ | **0.006** | | $F_{1,92} = 0.93$ | 0.337 |
| | | | | | | | | | | | | |
| ***Woody Species*** (*% cover*) | $Log_{10}(x+1)$ | | | None | | | None | | | None | | |
| Site | | $F_{4,92} = 3.93$ | **0.005** | | $F_{4,92} = 2.12$ | 0.084 | | $F_{4,92} = 9.64$ | **< 0.001** | | $F_{4,92} = 2.82$ | **0.029** |
| Fencing | | $F_{1,92} = 1.84$ | 0.179 | | $F_{1,92} = 0.28$ | 0.601 | | $F_{1,92} = 0.02$ | 0.886 | | $F_{1,92} = 0.00$ | 0.960 |
| Rooting | | $F_{1,92} = 9.73$ | **0.002** | | $F_{1,92} = 1.36$ | 0.246 | | $F_{1,92} = 5.07$ | **0.027** | | $F_{1,92} = 12.08$ | **0.001** |
| Fencing*Rooting | | $F_{1,92} = 0.11$ | 0.742 | | $F_{1,92} = 1.23$ | 0.271 | | $F_{1,92} = 0.02$ | 0.886 | | $F_{1,92} = 1.12$ | 0.293 |

**Table 2.4.** Three-way ANOVA test results at significance levels = Non-Significant: $p > 0.05$, Significant: **$p < 0.05$, $p < 0.01$,** Highly Significant: **$p < 0.001$**. Data were analysed using sequential sum of squares for tests. **F** = test statistic. **P** = probability associated with F-test. Plant data collected in woodland during spring and summer 2002 and 2003. Significant **P** values shown in bold. Site = random factor. Fencing and Rooting = Fixed factors. Fencing*Rooting = Interaction. Transform = method of normalising data.

## Effects of Rooting and Site on Plants at the Functional Group Level in Woodland Rides

| Source | Transform | F value & df | P | Transform | F value & df | P | Transform | F value & df | P | Transform | F value & df | P |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Spring 2002 | | | Summer 2002 | | | Spring 2003 | | | Summer 2003 | | |
| ***Perennial Graminoids*** (% cover) | $\text{Log}_{10}(x+1)$ | | | Square Root | | | None | | | None | | |
| Site | | $F_{4,44} = 1.19$ | 0.33 | | $F_{4,44} = 1.49$ | 0.222 | | $F_{4,44} = 3.41$ | **0.016** | | $F_{4,44} = 2.28$ | 0.075 |
| Rooting | | $F_{1,44} = 44.23$ | **< 0.001** | | $F_{1,44} = 20.40$ | **< 0.001** | | $F_{1,44} = 2.15$ | 0.149 | | $F_{1,44} = 1.66$ | 0.204 |
| | | | | | | | | | | | | |
| ***Perennial Forbs*** (*Freq. Occ.*) | Square Root | | | None | | | None | | | None | | |
| Site | | $F_{4,44} = 7.86$ | **< 0.001** | | $F_{4,44} = 21.03$ | **< 0.001** | | $F_{4,44} = 6.81$ | **< 0.001** | | $F_{4,44} = 6.04$ | **0.001** |
| Rooting | | $F_{1,44} = 11.71$ | **0.001** | | $F_{1,44} = 8.96$ | **0.005** | | $F_{1,44} = 7.16$ | **0.01** | | $F_{1,44} = 13.55$ | **0.001** |
| | | | | | | | | | | | | |
| ***Annual Forbs*** (*Freq. Occ*) | None | | | None | | | None | | | None | | |
| Site | | $F_{4,44} = 4.09$ | **0.007** | | $F_{4,44} = 4.44$ | **0.004** | | $F_{4,44} = 0.79$ | 0.535 | | $F_{4,44} = 2.42$ | 0.062 |
| Rooting | | $F_{1,44} = 0.14$ | 0.708 | | $F_{1,44} = 10.21$ | **0.003** | | $F_{1,44} = 13.17$ | **0.001** | | $F_{1,44} = 18.94$ | **< 0.001** |

**Table 2.5.** Two-way ANOVA test results at significance levels = Non-Significant: $p > 0.05$, Significant: $\mathbf{p < 0.05, p < 0.01,}$ Highly Significant: $\mathbf{p < 0.001}$. Data were analysed using sequential sum of squares for tests. **F** = test statistic. **P** = probability associated with F-test. Plant data collected in rides during spring and summer 2002 and 2003. Significant **P** values shown in bold. Site = random factor. Rooting = Fixed factor. Transform = method of normalising data.

## Effects of Rooting, Fencing and Site on Plants at the Functional Group Level in Grassland

| Source | Transform | F value & df | P | Transform | F value & df | P | Transform | F value & df | P | Transform | F value & df | P |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Spring 2002 | | | Summer 2002 | | | Spring 2003 | | | Summer 2003 | | |
| ***Perennial Graminoids*** (% cover) | None | | | None | | | None | | | None | | |
| Site | | $F_{3,73} = 3.13$ | **0.031** | | $F_{3,73} = 5.28$ | **0.002** | | $F_{3,73} = 6.98$ | **< 0.001** | | $F_{3,73} = 3.10$ | **0.032** |
| Fencing | | $F_{1,73} = 2.06$ | 0.155 | | $F_{1,73} = 11.72$ | **0.001** | | $F_{1,73} = 9.32$ | **0.003** | | $F_{1,73} = 14.17$ | **< 0.001** |
| Rooting | | $F_{1,73} = 785.05$ | **< 0.001** | | $F_{1,73} = 124.31$ | **< 0.001** | | $F_{1,73} = 16.52$ | **< 0.001** | | $F_{1,73} = 35.69$ | **< 0.001** |
| Fencing*Rooting | | $F_{1,73} = 0.03$ | 0.584 | | $F_{1,73} = 3.08$ | 0.084 | | $F_{1,73} = 0.35$ | 0.555 | | $F_{1,73} = 0.11$ | 0.736 |
| | | | | | | | | | | | | |
| ***Perennial Forbs*** (*Freq. Occ.*) | None | | | Square Root | | | Square Root | | | None | | |
| Site | | $F_{3,73} = 13.78$ | **< 0.001** | | $F_{3,73} = 21.22$ | **< 0.001** | | $F_{3,73} = 14.54$ | **< 0.001** | | $F_{3,73} = 17.89$ | **< 0.001** |
| Fencing | | $F_{1,73} = 2.59$ | 0.112 | | $F_{1,73} = 16.37$ | **< 0.001** | | $F_{1,73} = 0.88$ | 0.352 | | $F_{1,73} = 2.59$ | 0.112 |
| Rooting | | $F_{1,73} = 32.80$ | **< 0.001** | | $F_{1,73} = 21.37$ | **< 0.001** | | $F_{1,73} = 47.62$ | **< 0.001** | | $F_{1,73} = 37.74$ | **< 0.001** |
| Fencing*Rooting | | $F_{1,73} = 0.13$ | 0.721 | | $F_{1,73} = 0.00$ | 0.975 | | $F_{1,73} = 0.26$ | 0.615 | | $F_{1,73} = 0.43$ | 0.514 |
| | | | | | | | | | | | | |
| ***Annual Forbs*** (*Freq. Occ*) | None | | | None | | | Square Root | | | None | | |
| Site | | $F_{3,73} = 2.02$ | 0.119 | | $F_{3,73} = 0.27$ | 0.849 | | $F_{3,73} = 4.21$ | **0.008** | | $F_{3,73} = 4.12$ | **0.009** |
| Fencing | | $F_{1,73} = 5.06$ | **0.027** | | $F_{1,73} = 0.14$ | 0.708 | | $F_{1,73} = 0.18$ | 0.670 | | $F_{1,73} = 0.12$ | 0.728 |
| Rooting | | $F_{1,73} = 3.52$ | **0.045** | | $F_{1,73} = 34.63$ | **< 0.001** | | $F_{1,73} = 31.84$ | **< 0.001** | | $F_{1,73} = 4.40$ | **0.039** |
| Fencing*Rooting | | $F_{1,73} = 2.25$ | 0.138 | | $F_{1,73} = 0.39$ | 0.533 | | $F_{1,73} = 2.42$ | 0.124 | | $F_{1,73} = 0.00$ | 1.000 |

**Table 2.6**. Three-way ANOVA test results, at significance levels = Non-Significant: $p > 0.05$, Significant: **$p < 0.05$, $p < 0.01$,** Highly Significant: **$p < 0.001$**. Data were analysed using sequential sum of squares for tests. **F** = test statistic. **P** = probability associated with F-test. Plant data collected in grassland during spring and summer 2002 and 2003. Significant **P** values shown in bold. Site = random factor. Fencing and Rooting = Fixed factors. Fencing*Rooting = Interaction. Transform = method of normalising data.

## 2.5 Discussion

### *2.5.1 Community Level Effects*

As hypothesised, total plant cover was significantly greater in rooted than non-rooted treatments in woodland rides in 2003 and woodland, summer 2003. Further as hypothesised, plant cover was greater in fenced rooted than unfenced treatments in woodland spring 2003, and grassland spring and summer 2003 (**Appendices 2-4**, **Table 2.1**, **2.2**, **2.3**, **Fig 2.2**). Highly significant fencing effects for total plant cover during spring 2002 and during 2003 in grassland reflect this. The protection that fencing provides from further rooting and trampling by boar has an important positive impact on overall plant cover after rooting in grassland. Bowman and McDonough (1991) found plant cover to be greater in rooted than non-rooted treatments after approximately one year in a monsoon forest-wetland transition in northern Australia. However, Bratton (1974, 1975), Howe *et al.* (1981) and Singer *et al.* (1984) found that plant cover was reduced in rooted compared to non-rooted areas in deciduous woodland in the Great Smoky Mountains, U.S.A after approximately one year. Alexiou (1983), and Ralph and Maxwell (1984) also found plant cover to be lower in rooted than non-rooted areas across forest habitats in Smokers Gap, Australia and Hawaii respectively. Hone (1980) and Arrington *et al.* (1999) found reduced plant cover in rooted than non-rooted areas after one year in grasslands New South Wales, Australia and marshland ecosystems in Florida, U.S.A respectively.

Apart from Bowman and McDonough (1991), these studies are not consistent with my methodology or results. No fencing or method of protecting rooted and non-rooted treatments from rooting were utilised throughout the duration of the above studies, whereas both excluded and un-exclosed treatments in this study resulted in the relatively high level of plant cover after one year (**Fig 2.2**). Additionally, different climates of the countries where studies found that rooting decreased plant cover compared to the British climate, may have contributed to the different results. Further, differential temporal scales could also explain discrepancies in results. Studies that found rooting reduced plant cover were usually carried out within one year of rooting, whereas I believe that it is more helpful to investigate recovery from disturbances over a longer time-period in order to gain a wider perspective of changes that take place. In this study, after the initial effects of rooting, the most significant difference in plant cover between treatments occurred between twelve and seventeen months after rooting, showing that post-rooting changes in the plant community are modified over the short-term.

As hypothesised and corresponding to the intermediate disturbance hypothesis (Connell, 1978, 1979), after the initial effects of rooting, plant species richness was greater in rooted than non-rooted treatments to a highly significant extent in woodland rides, woodland in summer 2003, and in grassland 2003 (**Appendices 2-4**, **Table 2.1**, **2.2**, **2.3**, **Fig 2.3**). Corresponding with plant cover, species richness was highest in fenced rooted than unfenced non-rooted treatments within woodland and grassland in spring 2003. This further corresponds with studies by Kotanen (1994, 1995) in California, Welander (1995, 2000a) in Sweden and Milton *et al.* (1997) in central Germany, who found that species richness was greater in rooted than non-rooted plots in grassland habitats after approximately one year. Bowman and McDonough (1991), Onipchenko and Golikov (1996) and Arrington *et al.* (1999) also found that species richness was greatest in rooted than non-rooted plots within one year in a monsoon forest-wetland transition, northern Australia, alpine heath in the

Teberda Reserve, Russia, and in marshland ecosystems in central Florida respectively. Bratton's results (1974, 1975) however were atypical in that species richness was significantly lower in rooted than non-rooted areas in beech forest in the Great Smoky Mountains, U.S.A. As with plant cover, this result could be explained by the short temporal scale of the study, which was carried out less than one year after rooting, coupled with differences in methodology, whereby no fencing, or method of preventing re-rooting was utilised in the research.

Falinski (1986) observed patterns of re-rooting in the Bialowieza Forest in Poland. Seasonally based patterns of re-rooting were noted whereby corresponding areas were re-rooted at a similar intensity within the same seasons over seven years (Falinski, 1986). In Poland, intensive rooting begins in the phase of geophyte emergence in early spring and reaches its maximum level between mid June and mid July, coinciding with the substitution of the spring geophytes by summer hemicryptophytes. Geophytes are plants with subterranean organs such as tubers, rhizomes and bulbs. Examples of geophytes in my study area include the bulb bearing *Hyacinthoides non-scripta*, and *Anemone nemorosa*, which have rhizomes. Hemicryptophytes include the majority of vascular plants identified in my study area (**Appendix 1**) where the above ground portion dies back in adverse conditions and buds develop at ground level. The third peak in rooting intensity occurs between August and October but subsequently becomes considerably reduced in winter due to substantial snow cover and frozen ground (Falinski, 1986). This correlates with observed re-rooting patterns in East Sussex in that the same areas (but not necessarily the same patches) were re-rooted each year (Goulding, 2003b; pers. obs.). In contrast, however, in East Sussex, summer is the season with the lowest rooting intensity due to the hard, impenetrable soil at this time. Further, winter is a time of high rooting intensity along with autumn and spring, when the soil tends to be moist and easily penetrable. Distinct patches of plant biomass in the field layer demonstrate rooting patterns in Poland. The plant biomass as a whole was negatively correlated with the rooted surface area and frequency of rooting, which correlates with most research in other countries (Alexiou, 1983; Ralph and Maxwell, 1984; Bratton, 1974, 1975; Howe *et al.*, 1981; Singer *et al.*, 1984, Hone, 1980; Arrington *et al.*, 1999). However, although the above ground biomass was reduced within intensely rooted areas, the belowground portion of geophytes increased with rooting intensity. Falinski (1986) concluded that the segments of fragmented root systems of geophytes (such as the rhizome root system of *Anemone nemorosa* and the tuber root system of *Ranunculus ficaria*) gave rise to new individuals, stimulated their growth and enhanced vegetative reproduction. This corresponds with observed estimations of *A. nemorosa* cover across all the plots within the five woodland sites (**W1**- **W5**) in my study area (**Table 1.1**). Initially, the percentage cover of *A. nemorosa* was considerably reduced by rooting, but after both one and two years, cover had greatly exceeded that in non-rooted treatments (**Appendix 5**).

Studies researching the short-term effects of less extreme mechanical soil disturbance than rooting support this study in that plant species richness was significantly greater in areas with disturbance (removal of vegetation and litter) in grassland (Armesto and Pickett, 1985; Lavorel *et al.*, 1994) and mixed mountain forest (Mayer *et al.*, 2004). Although these disturbances are much less severe than rooting, this indicates that the effects of mechanical aspects of rooting are likely to largely contribute to the relatively high levels of species richness found across several studies. Further, the effects of rooting as a form of mechanical disturbance can be compared with pit and mounds in primeval forests. Uprooting, or tree-fall, has a major influence on the formation of areas of pit and mound topography, which is an important process in the creation of micro-topographic variation and like rooting, small-scale heterogeneity (Beatty and Stone, 1986; Peterson *et al*., 1990) and can play key roles in determining a regenerating forest community's composition and structure (Beatty, 1984). The influence on the immediate physical environment can, like rooting, be substantial, including the inversion of soil horizons (Beatty and Stone, 1986), exposure of buried seed (Putz, 1983) and the subsequent provision of important micro-sites for plant establishment (Sousa, 1984; Pickett and White, 1985). The presence of pits can increase localised litter accumulation, elevate nutrient concentration, increase soil moisture levels (Beatty, 1984; Beatty and Stone, 1986) and enhance plant species richness and total biomass (Peterson and Pickett, 1990). These effects can be directly related to similar elevations recorded within rooted compared to non-rooted patches Lacki and Lancia, 1983; Singer *et al.*, 1984; Bowman and McDonough, 1991; Kotanen, 1994; 1995; Welander, 1995, Onipchenko and Golikov, 1996; Milton *et al*., 1997; Arrington *et al*., 1999; Welander, 2000a; Mohr *et al.*, 2005). However, in contrast to the effects of rooting, pits, mounds and fallen trees can have an additional role in the protection of regenerating plant species against harsh microclimatological conditions and browsing where there are high densities of mammalian herbivores (Peterson and Pickett, 1995) due to their more extreme topographic variation compared with rooting. Illustrative of this, recorded in the southern Appalachians, an average pit depth, length and width is 1m, 1.5m and 3m respectively, and average mound thickness, height and width is 1m, 2m, and 3m respectively (Clinton and Baker, 2000), compared with the typical rooting depth of 5-15cm (Kotanen, 1995; Goulding, 2003b; Mohr *et al.*, 2005). However, during rooting, boar can uproot small trees with shallow roots, which can form similar topography to pit and mounds, although less extreme at depths of between approximately 20-30cms (**Fig 1.3a**) (Falinski, 1986).

A further useful comparison to rooting can be made with ground preparation for forestry. Soil handling is believed to be a critical activity in determining the success of land reclamation in forestry (Ramsay, 1986). Lack of soil aeration, and soil compaction are the main physical limiting factors that determine vegetation performance during land reclamation in woodland (Carey and Hendrick, 1986; Moffat and Bending, 2000). The main objective to ground preparation is to break up the soil surface in a way similar to ploughing (or replace the soil surface with loose soil) and to increase the nutrient levels, both of which also occurs through rooting (Lacki and Lancia, 1983; Singer *et al.*, 1984; Welander, 2000a; Mohr *et al.*, 2005). Vegetation re-establishment is substantially improved using ground preparation techniques in woodland (Ramsay, 1986; Carey and Hendrick, 1986; Moffat and Bending, 2000), which is therefore analogous to the effects of rooting (**Table 2.1**).

As hypothesised, other than the initial effects of rooting, plant diversity measured by the Shannon Diversity Index was significantly greater in all rooted than non-rooted treatments in grassland and woodland rides, and woodland in spring 2003. In woodland in summer 2003, diversity was greater in fenced rooted than unfenced plots (**Table 2.1**, **2.2**, **2.3**, **Fig 2.4**). The relatively high levels of diversity in rooted patches in the short-term could be explained by the intermediate disturbance hypothesis (Connell, 1978, 1979). Intermediate levels of rooting result in modifications in resource availability, levels of which may have promoted maximum levels of diversity. These high diversity patches may lead to an overall greater between-patch, medium-scale spatial heterogeneity in the environment (Ricklefs, 1977; Boeken *et al.*, 1995; Harrison, 1997), which could provide a greater diversity of niches and thus enhance the potential establishment of greater plant diversity at this scale (Stewart *et al.*, 2000; Wiens, 2000).

As hypothesised, site differences significantly affected all measurements of plants at the community level across the three habitat types (except for Shannon evenness in woodland rides) (**Table 2.1**, **2.2**, **2.3**). The most notable site effects occurred in grassland, with consistently highly significant site effects for total plant cover and species richness, and significant site effects for Shannon diversity and evenness during 2003 (**Table 2.3**). Differences across sites due to differential management regimes (**Table 1.1**) and neighbouring plant communities are probably the major determinants of the significant site affects on the plant community across the study sites. Different neighbouring plant communities to the study sites (particularly open habitats such as grassland) produce a potential source of seeds, that through dispersal could greatly contribute to altering community level attributes such as diversity, species richness and abundance across sites (Crawley, 1997), and hence largely contribute to the significant sites effects in grassland. Different management regimes (such as grazing (**G3** and **G4**) in contrast to no grazing (**G2**) in grassland) produce very different types of disturbance, which could significantly affect the plant community. For example, **G1** is managed through occasional sheep grazing; this intermediate level of disturbance could, at least in part account for it containing the highest level of species richness and diversity in comparison to the other three grassland sites. Additionally, potential unmeasured differences in the soil environment, such as nutrients available to plants and relative moisture levels, could lead to variation in the plant community across sites.

### *2.5.2 Functional Group Level Effects*

From the results of this study, it is clear that different functional groups respond differentially to rooting. Distinct functional groups have differential life strategies and cope differently with disturbance and physical changes in soil (Grime *et al.*, 1989). The study area is spatially heterogeneous, encompassing several distinct habitat and community types. Rooting could be diversifying the system further within sites by affecting the abundance of different functional groups.

As hypothesised, other than the initial effects of rooting, the frequency of occurrence of perennial forbs was consistently and significantly higher in rooted than non-rooted treatments within woodland rides and during 2003 in grassland. Further as hypothesised, in grassland in summer 2002, the frequency of perennial forbs was greater in fenced rooted than all other treatments (**Fig 2.7 b-c**). However, although there appears to be a similar trend from the graph (**Fig 2.7a**), there was no significant effect of rooting on perennial forbs in woodland, where they occurred at relatively low frequencies. Since many perennial forbs have persistent seed banks (Fenner, 1992), the exposure from lower to surface layers of the soil through rooting could aid germination, which may have contributed to this result. Further, since many perennial forbs are ruderal in strategy (**Appendix 1**), a proportion of this functional group is well adapted for post-disturbance colonisation from the seed rain. Consistent highly significant site effects in grassland and woodland rides, indicates that site differences also substantially affect the frequency of occurrence of this functional group. The fact that the highly significant site effects occurred in open habitats (in comparison to woodland), suggests that seed dispersal from neighbouring plant communities in the seed rain was a major determinant of the differences in frequency of this functional group across the study sites.

It seems probable that the patchiness of rooting creates a form of localised temporal heterogeneity. Patches of disturbed and undisturbed soil fluctuate temporally in terms of biotic and abiotic properties and seasonal variation, and combined, cause annual changes in the abundance of plant functional groups. For example, in frequently rooted areas in the woodland ride site **R5** (**Table 1.1**) during summer 2004, the highly ruderal perennial forbs *Chamaenerion angustifolium* (prolific in disturbed sites) and *Hypericum perforatum* were dominant in the vegetation (**Fig 2.12**). The previous year, *C. angustifolium* was dominant but *H. perforatum* was very sparse.

**Fig. 2.12:** A woodland ride site (**R5**) showing *Hypericum perforatum* and disturbance-tolerant *Chamaenerion angustifolium*. These two perennial forbs were growing on

As hypothesised, in grassland, the cover of perennial graminoids was significantly lowered in all rooted than non-rooted treatments in 2002, and in unfenced rooted than all other treatments in 2003. Significant fencing effects in 2003, in part, reflect these results whereby the protection that fencing provides from sheep grazing and other forms of disturbance (such as from rabbits) in grassland, could have largely determined the greater cover of perennial graminoids in fenced than unfenced non-rooted treatments. The cover of perennial graminoids was also significantly lower in rooted than non-rooted treatments in woodland rides during 2002 (Fig 2.8). Graminoids are not specifically adapted for coping with disturbance, thus other functional groups with a greater relative competitive ability in rooted soil will grow in addition to graminoids, reducing their monopoly in grassland habitats. Martinsen *et al.* (1990) researched the effects of pocket gopher (*Thomomys* spp.) soil disturbance on plant species diversity in a short grass prairie community. It was found that disturbance decreased the abundance of grasses and increased the abundance of dicotyledons. The result was attributed to the relative competitive abilities of the different functional groups in disturbed areas. In this study, consistent significant site effects in grassland, shows that site differences strongly influenced the cover of perennial graminoids in this open habitat. Since many perennial graminoids produce wind-dispersed seeds (Phillips, 1980), differential levels of intra and inter-habitat seed dispersal across sites seems a likely explanation for this result.

There was little overall impact from rooting on the cover of woody species in woodland, except for significantly less cover in non-rooted fenced than all other treatments in summer 2003 (**Fig 2.9**). However, in some circumstances rooting could be beneficial for the regeneration of woody species. For example, grasses are competitors for tree seedlings (Peterken, 1993). The cover of graminoids was significantly lower in rooted than non-rooted treatments (**Fig 2.8**, **Table 2.5**, **2.6**) illustrating that through the reduction of such competitors, rooting could potentially aid the regeneration of woody species in habitats where woody species and competitors co-occur. Andrezejewski and Jezierski (1978) found boar rooting enhanced the growth of pines (*Pinus spp*.) on nutrient-poor European soils. Further, Falinski (1986) frequently observed profuse tree stand regeneration in rooted areas. For example, in mixed forest environments in the Bialowieza Forest, spruce (*Picea* spp.) saplings were frequently seen to agglomerate on sites changed by the rooting activities of boar (Falinski, 1986). However, Groot Bruinderink and Hazebroek (1996) found rooting decreased regeneration of *Quercus robur*, *Q. petraea* and *Q. rubra* and *Fagus sylvatica* forests in the Netherlands. They suggested that this was due to an intense frequency of re-rooting the same patches. Peart *et al.* (1994) found boar rooting negatively affected woody-species abundance and regeneration beneath chaparral-oak woodland on Santa Cruz Island, California. Woody species were removed as seedlings via direct predation on seeds, trampling the fragile shoots and damage from over-turned soil from rooting. Lipscomb (1989) found rooting reduced long-leaf pine regeneration in coastal forest in Georgetown, South Carolina, U.S.A. Boar selected the larger seedlings to consume, and so surviving seedlings were less competitive and vigorous. In this study, significant site effects during spring 2002 and 2003, and summer 2003 indicate that site differences were the predominant influence on the cover of woody species across the study sites.

As hypothesised, other than initial effects of rooting, the frequency of occurrence of annual forbs was greater in rooted than non-rooted treatments to a highly significant extent throughout 2002 and 2003 across woodland rides and during summer 2002 and spring 2003 in grassland (**Fig 2.11**). Kotanen (1994, 1995) found annuals proliferated in rooted areas in the short-term in Californian prairie. Boeken *et al.* (1998) found annual plants increased in disturbed areas in a desert ecosystem. Annual forbs tend to be ruderal in strategy (**Appendix 1**) and typically tolerate disturbance better than any other functional group. The rooted gaps in vegetation are likely to be quickly re-colonised by the typical production of large numbers of small sized, wind-dispersed seeds from the seed rain due to their relatively superior dispersal capacity (Welander, 2000a). Furthermore, ruderal strategists are associated with large, persistent seed banks and their emergence is dependent on disturbance-created gaps (Grime, 1979; Grime *et al.*, 1989; Crawley, 1997) such as rooted patches. Germination is higher in disturbed (such as rooted) rather than undisturbed soil (Buckley *et al.*, 1997; Rydgren *et al.*, 1998; Hyatt and Casper, 2000; Jutila and Grace, 2002) thus the ruderal seed bank may also be contributing to this result. Additionally, the characteristic production of large numbers of wind-dispersed seeds of ruderals could be the major determinant of the significant site effects during 2002 in woodland rides and during 2003 in grassland. Different levels of seed dispersal from the different neighbouring plant communities could have significantly contributed to the different frequencies of annual forb occurrence across the study sites.

My results suggest that wild boar rooting is creating a mosaic environment in the short-term which is enhancing community attributes such as plant diversity, biomass and species richness in a patchy manner across woodland, grassland and woodland rides. A relatively stable wild boar population that produces an intermediate level of disturbance through rooting could help create and maintain a relatively high level of plant diversity, biomass and species richness across different habitats. Boar root in specific areas within habitats and often return to previously rooted areas, leaving many areas untouched (Welander, 2000b). The subsequent creation of environmental patchiness, and the diversification of niches on a local scale, favours the growth and success of some plant functional groups but not others. Further, the potential role of boar in the dispersal of plant propagules (Welader, 2000a; Heinken *et al*., 2001) could contribute to the differences in the plant community measured here. Welander (2000a) found a shift in the plant community towards species with small seeds (typically annual forbs) and concluded that diaspores adhered to the exterior surface of the animals and were dispersed when boar moved between feeding, resting and wallowing sites. In grassland and woodland rides, the frequency of annual and perennial forbs was significantly greater in rooted than non-rooted treatments, whereas perennial graminoids had a much smaller cover in rooted than non-rooted treatments in the short-term. Rooting had little effect on the abundance of functional groups within woodland habitats.

Fencing had less impact than expected on plant re-establishment after rooting at both the community and functional group level. However, as predicted, neither the abundance of any functional group or any community level measurement were significantly greater in unfenced rooted compared to fenced rooted treatments throughout the study within woodland or grassland. The most notable impact of fencing was in grassland, where the cover of perennial graminoids was significantly greater in fenced non-rooted than unfenced rooted treatments during 2003, and total plant cover was significantly greater in fenced rooted than unfenced treatments in summer 2002 and during 2003 (**Fig 2.2**, **2.8**). Relating to Connell's (1978) intermediate disturbance hypothesis, fencing rooted areas acts to mediate the intensity of the disturbance leading to increased species richness in the short-term. The greater relative impact of fencing in grassland than woodland and woodland rides could be due to the additional protection from grazing by sheep and deer.

# Chapter 3

## Impacts of wild boar rooting on Hyacinthoides non-scripta

### 3.1 Introduction

**T**he native bluebell (*Hyacinthoides non-scripta*) is a bulbous perennial geophyte that reproduces both vegetatively (via axillary daughter bulbs) and sexually (by seed) (Wilson, 1959; Knight, 1964; Thompson and Cox, 1978; Grabham and Packham, 1983; Merryweather and Fitter, 1995). It flowers during April and May and can dominate broadleaved deciduous and mixed woodland floors and hedge banks in Britain and parts of western Europe. *H. non-scripta* bulbs produce a single inflorescence, a vertical raceme 20-50cm high (Corbet, 1999). This plant is an ancient woodland indicator species (Rackham, 1980; Packham *et al*., 1992; Peterken, 1993) and has been legally protected since 1998 on Schedule 8 of the Wildlife and Countryside Act (1981) (Wigginton, 1999), which prohibits anyone to dig up the bulbs from the wild. *H. non-scripta* is much loved by the British public and has been described by Plantlife as being the UK's national wild flower emblem (Pilgrim and Hutchinson, 2004). *H. non-scripta* is currently under threat from several factors:

- **Hybridisation and Competition**. There are two different species of bluebell in Britain and a hybrid: the native *H. non-scripta*, the Spanish *H. hispanica* and the resulting readily produced fertile hybrid *H. hispanica* x *non-scripta* (*H. massartiana*). Both the non-indigenous bluebell and the hybrid are common in gardens across Britain and increasingly also in broadleaved woodland. A recent survey found 15% of bluebells in broadleaved woodland were *H. massartiana*, 1% were *H. hispanica* and 84% *H. non-scripta* (Pilgrim and Hutchinson, 2004). *H. hispanica* and *H. massartiana* compete with the native species. *H. non-scripta* grows predominantly in damp, wooded habitats where rotting leaf litter is found whereas the non-indigenous species thrives in a wide range of habitats.

- **Habitat loss and Fragmentation.** Britain was once covered with vast tracts of wild woodland (Rackham, 1980; Marren, 1990; Ingrouille, 1995; Rackham, 1997). Around six and a half thousand years ago, woodland clearance began, leading to the removal of all the wild wood by around 350AD, and the on-going development of a much reduced, highly fragmented and actively managed remaining woodland in Britain (Rackham, 1997; UK Agriculture, 2006). The decline in continuous broadleaved woodland has greatly reduced *H. non-scripta*'s natural habitat and has considerably reduced the number and size of *H. non-scripta* populations.

- **Rarity and Climatic Change**. Although locally common, *H. non-scripta* is globally rare with nearly half its total population in the UK (Thompson and Cox, 1978) (**Fig 3.1**), although it is reported that its numbers are in decline in southern England (Gow, 2002; Pilgrim and Hutchinson, 2004). It is also common in the north and west of France but more sparsely distributed in the east and south-east of France and in Holland, Belgium, north-west Germany and small areas of northern Spain (Blackman and Rutter, 1954; Thompson and Cox, 1978; Briggs 2004) (**Fig 3.1**). Its distribution is indicative of cool, damp, temperate environments; global warming could thus pose a threat to its future if climatic temperatures rise in Britain. The more vigorous *H. hispanica* is well adapted to both the cooler and warmer climate.

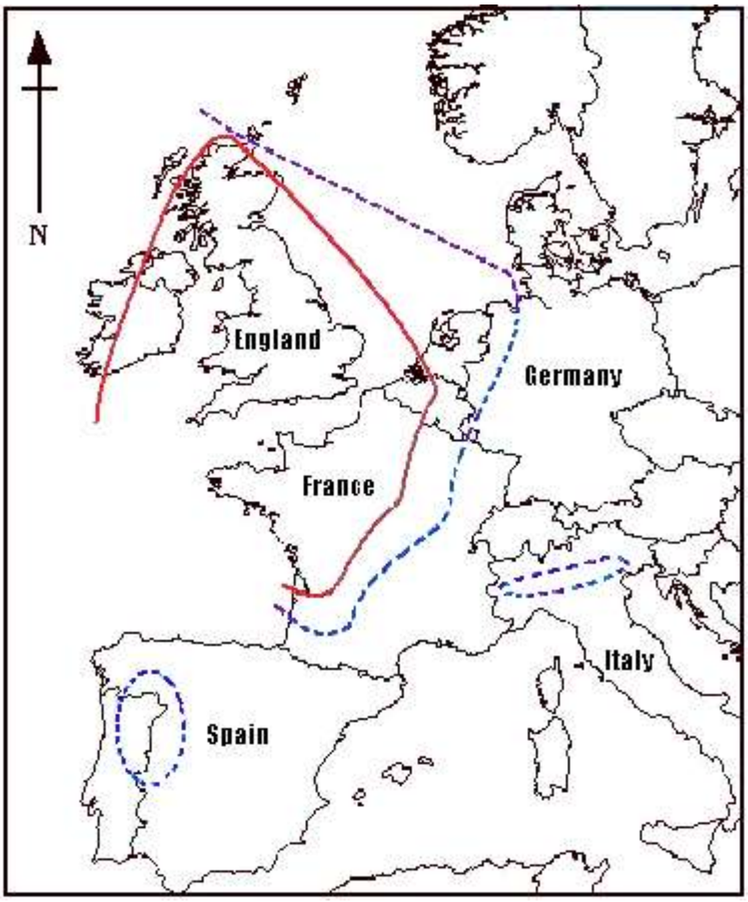


**Fig 3.1**. Map showing the approximate distribution of *Hyacinthoides non-scripta* in Western Europe. Area within which the species is abundant bounded by red line; areas within which it occurs erratically or in locally

- **Illegal bulb collection.** Although it is illegal to dig up *H. non-scripta* bulbs from the wild in the U.K for commercial purposes, this activity continues.

Wild boar consume a range of bulb-bearing plants (Howe and Bratton, 1976; Dardaillon, 1987; Kotanen, 1994) and are known to root intensively within *H. non-scripta* populations (**Fig 3.2**) and consume the bulbs (Goulding *et al.*, 1998; Goulding, 2003a, 2003b). Boar predominantly root for the bulbs in spring and summer, although this declines when other food sources become available (pers obs). Although no quantitative assessment was made on the numbers of *H. non-scripta* bulbs consumed by boar, where *H. non-scripta* populations were observed, relatively large amounts of rooting occurred, suggesting that *H. non-scripta* bulbs are an important food source of the boar. The wild boar's diet consists of between 90% (Henry and Conely, 1972; Genov, 1981) and 97% (Massei *et al.*, 1996) bulbs, tubers, roots, shoots, leaves, fruits and seeds (Falinski, 1986). The remaining proportion of their diet consists of insects, worms, larvae, eggs, nestlings, small mammals and carrion (Howe and Bratton, 1976; Wood and Roark, 1980; Genov, 1981; Falinski, 1986; Dardaillon, 1987; Goulding *et al.*, 1998). The extent that *H. non-scripta* bulbs contribute to the boar's diet is likely to vary from year to year depending on the relative availability of other food sources.

The overall intensity of rooting for *H. non- scripta* bulbs is also likely to vary due to the fluctuating size of the boar population, which results from hunting, poaching, dispersing individuals, fluctuating litter sizes and in particular, overall food supply (Henry and Conley, 1972; Baber and Coblentz, 1987; Groot Bruinderink and Hazebroek, 1994; Massei *et al.*, 1996). When available food is in short supply, litter sizes are reduced (Harman, pers comm).

**Fig 3.2 a-b**: Examples of heavily rooted patches within a dense *Hyacinthoides non-scripta* population in the sweet chestnut coppice site **W1** in the study area. This site had the most severe and widespread rooting of all the five woodland sites since at least 2001. Photos taken 2004.

There is considerable concern regarding the impact of rooting on the *H. non-scripta* population (Goulding, 2003a, 2003b; Wilson, pers comm). Wild boar have a large visible effect on *H. non-scripta* (**Fig 3.2**) but whether this effect is reflected in the overall size, structure and viability of the populations is unknown. It seems that rooting has a range of effects on different bulb-bearing geophytes. For example, in the Appalachians, south-east U.S.A, *Claytonia virginica* and *Erythronium americanum* were only weakly and transiently affected by rooting, although *Lilium superbum* suffered severe reductions in population size (Kotanen, 1994). Kotanen (1994) found that the native perennials *Dichelosremma capitatum* and *Brodiaea* spp. in Californian meadows remained abundant after more than a century of wild boar disturbance. It was suggested that the large numbers of tiny vegetative bulblets were difficult for the pigs to locate and pick up and the larger bulbs, which tend to be buried more deeply (>25cm) than the average rooting depth (5-15cm) tend not to be exposed and thus unconsumed. In comparison, observed estimations of percentage cover of the bulb bearing *Narcissus pseudonarcissus* across all the plots within woodland site **W2** in my study area (**Table 1.1, Figure 5.3a**), was either equal to or slightly greater in rooted compared to non-rooted treatments over three growing seasons (**Appendix 5**). This indicates that boar do not specifically target and consume the *N. pseudonarcissus* bulbs and have no notable impact on the population size.

*H. non-scripta* numbers have declined in southern England by 25-49% in the last twenty five years (Gow, 2002). Considering this decline, it is important to be able to predict whether boar will cause an increase, a further reduction or have no significant effect on *H. non-scripta* numbers. *H. non-scripta* may benefit from the increased nutrient turnover produced by rooting (Lacki and Lancia, 1983; Singer *et al.*, 1984) (**Chapter 5**). Blackman and Rutter (1947) found small responses in the seasonal dry weight gain of *H. non-scripta* through adding nitrogen, phosphorous and potassium, alone and in combination to fertile soils. However, adding nutrients, especially nitrogen to grassland can considerably reduce *H. non-scripta* growth due to competitive dominance of tall grasses, which out-shade *H. non-scripta* (Blackman and Rutter, 1950). *H. non-scripta* may benefit from other rooting effects on soil such as a refined crumb structure, improved drainage (Brownlow, 1994) or improved individual performance resulting from reduced density.

Rooting may lower the age structure of the *H. non-scripta* population. Growing from a bulb, *H. non-scripta* will flower first in about the fifth year (Woodhead, 1904) and thus the consumption of mature bulbs through rooting is likely to increase the number of immature bulbs and thus non-flowering individuals. Additionally, although seed is set every year, establishment of seedlings varies from year to year (pers obs) and is dependent on several biotic and abiotic factors (Blackman and Rutter, 1954). The physical and chemical modifications of recently rooted soil could favour germination and establishment of *H. non-scripta* seeds from the transient seed bank (**Chapter 4**). Further, wild boar could aid the dispersal of *H. non-scripta* seeds. During July and August, ripe *H. non-scripta* seeds are ejected from the dry capsules and fall close to the parent plant but potentially spread greater distances (such as 46-180cm) when disturbed by animal movements (Knight, 1964) such as large bulky wild boar when foraging for the bulbs.

Woodland plant communities are believed to have been more heterogeneous in the past than they are today (Ingrouille, 1995). Differential activities of large mammals such as wild boar would have helped shape the structure and composition of woodlands. Deer (Cervidae), for example, can cause major changes in the composition and structure of forest communities by browsing shrubs and tree seedlings and grazing under-story forbs (Falinski, 1986; Augustine and Jordan, 1998; Virtanen *et al*., 2002). In coppiced woodlands, deer can cause substantial damage to shoots that sprout after cutting, which can ultimately result in a shorter and more open canopy with repeated browsing (Kay, 1993). In contrast with boar rooting, reducing height, growth inhibition and decreased foliage density appears to be a typical impact of deer browsing in woodland at relatively high population densities (Falinski, 1986; Kay, 1993; Morecroft *et al*, 2001). Repeated browsing by red deer (*Cervus elaphus*) led to the regression and complete inhibition of regeneration of deciduous canopy species and the under-story layer, resulting in major changes in the structure and dynamics of the Bialowieza Forest (Falinski, 1986). Gill and Beardall (2001) reported that by characteristically reducing the height and growth of trees, shrubs and climbers through deer browsing in woodland, the biomass of under-story vegetation becomes typically reduced, leading to a more open and simplified vertical structure with decreased plant species richness and diversity. Augustine and Jordan (1998) also found a reduced plant species richness in small forest fragments, parks and preserves with relatively low densities of palatable plant species, where white-tailed deer (*Odocoileus virginianus*) densities were relatively high. On the Isle of Rum however, both tree regeneration and plant species richness (particularly prostrate herbs) were significantly lower in areas where *C. elaphus* were removed, compared to areas constantly occupied by deer (Virtanen *et al*., 2002).

This contrasts with the majority of studies on boar, whereby species richness was typically found to be significantly greater in rooted than non-rooted areas (Kotanen, 1994, 1995; Welander, 1995, 2000a; Bowman and McDonough, 1991; Onipchenko and Golikov, 1996; Arrington *et al.*, 1999), where boar population densities existed at managed, sustainable levels. Large mammal disturbance, such as boar rooting, can cause between-patch spatial heterogeneity (Milton *et al.*, 1997; Arrington *et al.*, 1999; Welander, 2000b). A more heterogeneous environment provides a wider range of niches and microenvironments (Onipchenko and Golikov, 1996; Arrington *et al.*, 1999), which could favour within-patch, plant species richness and diversity (**Chapter 2**) and for example create a more patchily distributed *H. non-scripta* population.

Blackman and Rutter (1946) found that variation in the density of *H. non-scripta* populations was highly correlated with variation in the degree of shading. 75% of total *H. non-scripta* density variation was attributed to light availability. It was found that *H. non-scripta* was intolerant of deep shade and sensitive to slight shading (Blackman and Rutter, 1946). With a sustained presence of boar over several years, it is likely that long-term structural changes would occur at the community level. For example, rooting on woodland floors could have a positive effect on the *H. non-scripta* population by keeping the structure of the woodland floor open. Reducing the number of potential shrub layer and canopy trees through their removal as seedlings, would increase light levels at the woodland floor and could help sustain suitable conditions for *H. non-scripta* populations.

No scientific literature exists on the ecological impacts of wild boar on *H. non-scripta* in the UK. However, as the potential for boar to affect *H. non-scripta* populations is large, and boar are likely to remain a feature of UK woodlands, this area deserves investigation. The overall aim of this study was to ascertain how rooting for *H. non-scripta* bulbs affects the *H. non-scripta* population in the short-term (between two months and two years).

### *3.1.1 Hypotheses*

Damage to *H. non-scripta* leaves due to trampling (for example by boar whilst rooting) leads to loss of vigour and successively smaller plants (Peace and Gilmour, 1949;

Blackman and Rutter, 1954; Cooke, 1997). With this consideration, coupled with the inevitable removal of *H. non-scripta* bulbs during rooting, I hypothesised that the numbers of *H. non-scripta* individuals and flowering stems would be significantly lower in rooted than non-rooted, fenced and unfenced plots within Spring 2003 and 2004. However, numbers of individuals and flowering stems within the rooted plots, and especially in fenced compared to unfenced plots (due to the protection that fencing provides from trampling and re-rooting) are expected to be higher in 2004 than in 2003 due to the potential re-establishment of the *H. non-scripta* population in the non-re-rooted plots.

Based on the preceding hypothesis (that rooting significantly reduces the number of *H. non-scripta* individuals), together with general observation, (**Fig 3.2 a-b**, **Fig 3.3 a** and **c**), I hypothesised that the proportional change in *H. non-scripta* cover after both one (between 2002 and 2003) and two (between 2002 and 2004) years would be significantly greater in rooted than non-rooted, fenced and unfenced plots. However, based on Connell's (1978) intermediate disturbance hypothesis, the proportional change in *H. non-scripta* cover over this time is expected to be greatest in rooted fenced plots after two years. This is due to the protection that fencing provides from trampling and re-rooting, coupled with the longer period for potential re-establishment of the *H. non-scripta* population in the non-re-rooted plots (Connell, 1978, 1979; Cooke, 1997).

**Fig 3.3 a-d:** Different woodland sites in the study area. **a)** Severe rooting in sweet chestnut coppice site **W1**; discarded *H. non-scripta* leaves and bulbs can be seen strewn over the disturbed soil surface. **b)** Collecting data within a dense *H. non-scripta* population in the mixed deciduous and sweet chestnut coppice site **W4**. **c)** A rooted fenced treatment within the mixed

Excessive leaf damage can lead to considerable reduction in flowering (Peace and Gilmour, 1949; Blackman and Rutter, 1954). However, even with a reduction of flowers the same number of seeds per plant may be produced as a compensatory mechanism against poor flower production (Grabham and Packham, 1983). Therefore, it was hypothesized that the number of seeds per plant and per capsule would be unaffected by rooting and fencing one year after rooting. Grabham and Packham (1983) demonstrated another possible compensatory mechanism whereby *H. non-scripta* seeds were heavier when produced by a smaller number of flowering individuals. Since recently rooted patches have significantly fewer plants (**Fig 3.2 a-b**), and based on Grabham and Packham's (1983) demonstration, I hypothesized that seed weight and viability would be greater from individuals in rooted than non-rooted, fenced and unfenced plots one year after rooting.

I hypothesised that site differences would significantly affect all measured aspects of the *H. non-scripta* population. The five woodland sites are managed in different ways (**Table 1.1**), which could potentially cause variation in physical, biological and chemical elements of above and belowground community attributes. For example, coppiced sites **W1** and **W2** (in comparison to those that are occasionally thinned (**W3** – **W5**)) are likely to have greater light levels penetrating the woodland floor, which could influence the *H. non-scripta* population. High light levels are known to increase seed production in *H. non-scripta*, due possibly to an observed increase in insect visitations, which increases their chance of fertilisation (Knight, 1964). Further, the diversity of woodland floor plant communities with high light levels (such as recently coppiced woodland) is known to be greater than shadier woodland (Ash and Barkham, 1975; Ford and Newbould, 1977; Gondard *et al.*, 2001; Mason and McDonald, 2002), which could strongly influence the relative abundance of the *H. non-scripta* population. Site differences were found to significantly affect overall species richness, Shannon diversity and evenness in woodland in my study area (**Table 2.1**). Additionally, potential unmeasured inconsistencies in the soil environment (known to modify *H. non-*

*scripta* growth (Grabham and Packham, 1983)) between sites could also strongly influence the *H. non-scripta* population, such as soil structure (litter/bulk ratio), drainage potential and available nutrients.

## 3.2 Materials and Methods

### *3.2.1 Study Area*

This study was undertaken in all five woodland sites (**W1-W5**, **Table 1.1**)) within the study area (**Fig 1.5 a-d**), (**section 2.2.1**).

### *3.2.2 Experimental Design*

This study utilised the same fenced and open plots set up in the five woodland sites in early spring 2002 for the investigation of the effects of rooting on plants at the community and functional group level (**Chapter 2**). Data were collected during April and early May 2002, 2003 and 2004 when *H. non-scripta* was flowering, with a single subsequent data collection in July 2003 of *H. non-scripta* seeds. Each of the spring data collections took approximately three weeks to complete. Sixteen out of the twenty plots that were initially set up per site were used in this study. There were four treatments and four replicates per site. Replicates were four rooted fenced and unfenced and four non-rooted fenced and unfenced treatments per site. Twenty plots per treatment over five woodland sites were used in this study.

To investigate the effects of rooting on abundance of *H. non-scripta,* the following measurements were made in each plot:

- Number of *H. non-scripta* individuals
- Number of flowering stems

- Percent cover of *H. non-scripta*
- Percent cover of bare soil
- Percent cover of other plant species

A 50cm x 50cm quadrat with a 10cm grid was used to count *H. non-scripta* individuals. Where juveniles clustered, it was sometimes difficult to distinguish between individual plants. In accordance with Knight (1964), Wilson (1959) and Grabham and Packham (1983) who consider vegetative reproduction as being relatively infrequent, where there was any doubt as to the individuality of each plant, they were counted as individual plants and therefore overall numbers of *H. non-scripta* may be over-estimated. Percentage cover measurements of *H. non-scripta* were taken over three growing seasons: spring 2002, 2003 and 2004 (**Fig 3.3**).

During July 2003, *H. non-scripta* inflorescences were collected for the investigation into rooting affects on fecundity, viability and variability of *H. non-scripta* seed size. The flowering stems had senesced, the brown withered capsules were open and a large proportion of the seeds had ripened at the time of collection. The same sixteen plots over the four treatments were also used for inflorescence collection at each of the five woodland sites. Complete inflorescences from two previously flowered individual *H. non-scripta* plants were collected from random locations within each of the sixteen plots over the five sites. All capsules from a single inflorescence were removed and stored in separate paper bags.

The collected material was stored in a cool, dry cupboard for four days. Seeds were then removed from each individual capsule and divided into three categories: ripe, unripe and aborted. Ripe seeds were identified as those with brown or black coloration and fully formed (relatively round with a hard surface). Unripe seeds were fully formed in shape but were white or pale in colour, and aborted seeds were unformed in shape (unspecific shape with a soft surface) and pale in colour. The seeds in each category from each capsule were counted and the ripe seeds were weighed. Fifty ripe seeds per treatment taken randomly across replicates were placed in groups of five into Petri dishes lined with two filter papers and moistened with de-ionised water. Each Petri dish was wrapped in Parafilm (**Fig 3.4a**). There were ten replicate Petri dishes (each containing five seeds) per treatment per site. Over the five sites, there were two hundred and fifty seeds per treatment (one thousand seeds in total).

The Petri dishes were wrapped in black plastic sacks to prevent light entry and stored in a cold room for three months at 5°C (July 2003-October 2003) for stratification of the seeds (Blackman and Rutter, 1954; Slade and Causton, 1979; Wood, pers comm). Petri dishes were regularly checked and de-ionised water added where appropriate to prevent the seeds desiccating.

After the chilling period, the Petri dishes were placed in a growth room for germination to occur at a temperature between 10-15°C. The Petri dishes were given a twelve-hour light twelve-hour dark lighting regime under metal halide lamps. After ten weeks, Giberellic acid (GA) dissolved in potassium hydroxide (KOH) and deionised water was added to speed up germination. It is currently unknown whether *H. non- scripta* responds to this or any other growth hormone. 14mg of 1M GA was dissolved in

**Fig 3.4 a-b**: Experiments on viability of *H. non-scripta* seeds 2003-2004. **a)** Checking *H. non-scripta* seeds for germination in Petri dishes that were kept under metal halide growing lamps. **b)** A proportion of the germinated *H. non-scripta* seeds.

600μg Potassium hydroxide (KOH) that was dissolved in 2 litres of de-ionised water. Petri dishes were inspected twice weekly throughout their duration in the growth room in order to replenish lost moisture and check for germination. After six months in the growth room (April 2004), with no germination occurrence (*H. non-scripta* seeds typically germinate in the dark under leaf litter (Knight, 1964)), seeds were wrapped in silver foil to prevent all light entry. Following this, at each inspection, germinated seeds were counted and removed from the Petri dishes (**Fig 3.4a-b**).

## 3.3 Statistical Analysis

Data were analysed with three-way ANOVA (Eddison, 2000; Dytham, 2003). Data were log (Log10 x +1) or square root transformed to achieve normality where possible (specified **Table 3.1**, **3.2**, **3.3**, **3.4**, **3.5**). Rooting and fencing were two fixed factors and site was the third and random factor. Data were analysed using the sequential sum of squares. Pair-wise comparisons were undertaken where appropriate to test for significance within factors using the Tukey test (Eddison, 2000; Dytham, 2003).

*H. non-scripta* percentage cover data (collected in spring 2002, 2003 and 2004) were converted into proportional change in cover over one, and two years using the following calculation: 2003 data - 2002 data / 2002 data, and, 2004 data – 2002 data / 2002 data respectively. The above statistical analysis was subsequently carried out on the converted data.

## 3.4 Results

### *3.4.1 Number of H. non-scripta individuals*

Highly significant rooting effects, both one and two years after rooting, illustrate that a significantly greater number of *H. non-scripta* individuals were recorded in non-rooted than rooted treatments in 2003 and 2004 (**Table 3.1**). A significant fencing effect in 2004 is illustrated by the increased number of *H. non-scripta* in non-rooted fenced than all other treatments, and between rooted and non-rooted unfenced treatments (**Fig 3.5 b**). Greater numbers of individuals exist in rooted fenced treatments in 2004 compared to 2003, showing that fencing increases recovery of *H. non-scripta* abundance in rooted treatments (**Appendix 5**, **Fig 3.5 a-b**). Significant site effects, both one and two years after rooting show that site differences strongly influence the numbers of *H. non-scripta* individuals independently of rooting.

### *3.4.2 Number of H. non-scripta flowering stems*

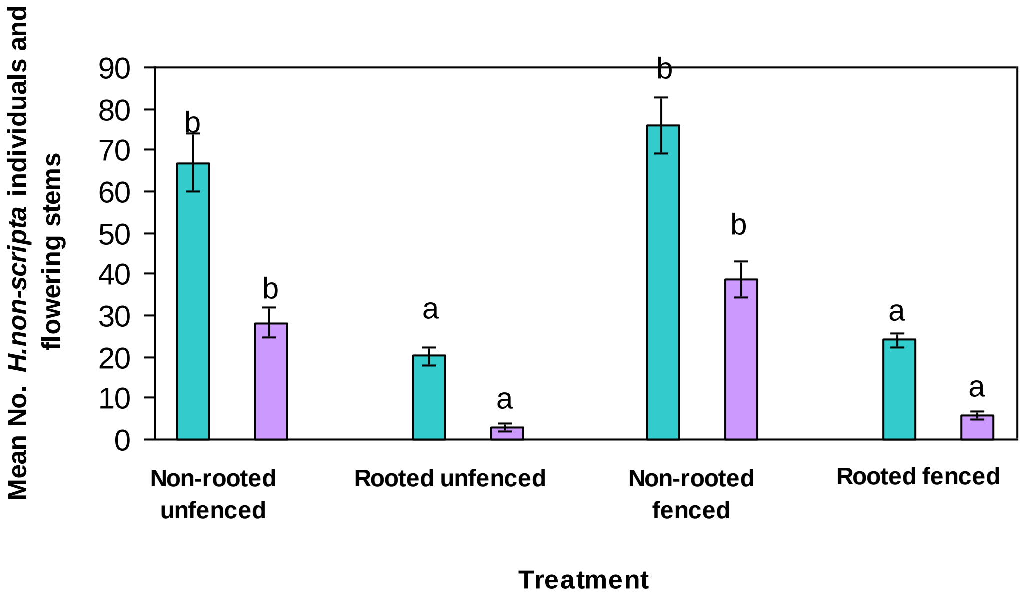
A significantly greater number of flowering stems were recorded in all non-rooted compared to rooted treatments one and two years after rooting (**Table 3.1**). However, in 2004, the number of flowering stems had increased compared to 2003 in both rooted treatments by five times, whereas there was a slight decline overall in non-rooted treatments (**Fig 3.5 a-b**). Highly significant fencing effects in 2003 and 2004 indicate that fencing increases the re-growth of *H. non-scripta* independently of rooting, although Tukey tests reveal that the only significant difference was a greater number of flowering stems in non-rooted unfenced compared to rooted unfenced treatments.

**Fig 3.5 a-b:** Effects of rooting and fencing on numbers of *H. non-scripta* individuals and numbers of flowering stems over two growing seasons from **a)** 2003 to **b)** 2004. Data were pooled across five woodland sites. Differences between treatments (indicated by different letters above bars) were analysed using the Tukey test (significance level **p<0.05**). Error bars indicate standard error of the means, n = 4. For ANOVA results see **Table 3.1**.

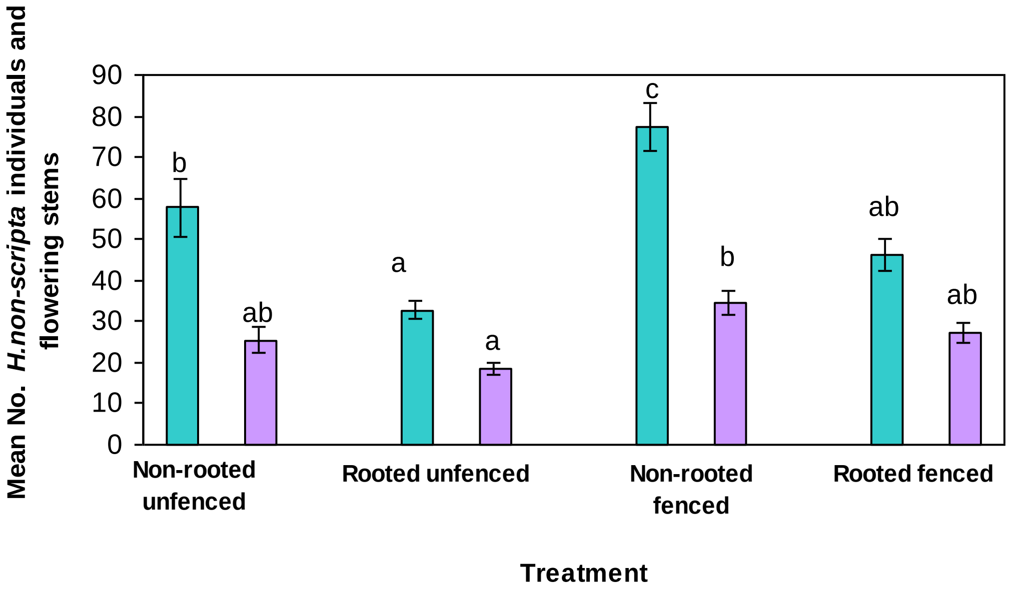
🟦 Number of *H. non-scripta* Individuals

🟪 Number of *H. non-scripta* Flowering Stems

**a) 2003**



**b) 2004**



# Effects of Rooting, Fencing and Site on demographic aspects of *H. non-scripta* populations

| Source | Transform | F value & df | P | Transform | F value & df | P |
|---|---|---|---|---|---|---|
| | | Spring 2003 | | | Spring 2004 | |
| **No. *H. non-scripta* individuals** | None | | | None | | |
| Site | | $F_{4, 72} = 4.36$ | **0.003** | | $F_{4, 72} = 3.89$ | **0.006** |
| Fencing | | $F_{1, 72} = 1.95$ | 0.167 | | $F_{1, 72} = 11.88$ | **0.001** |
| Rooting | | $F_{1, 72} = 113.49$ | **< 0.001** | | $F_{1, 72} = 34.00$ | **< 0.001** |
| Fencing*Rooting | | $F_{1, 72} = 0.32$ | 0.573 | | $F_{1, 72} = 0.42$ | 0.521 |
| **Percentage cover bare soil** | None | | | None | | |
| Site | | $F_{4, 72} = 2.46$ | 0.053 | | $F_{4, 72} = 2.26$ | 0.071 |
| Fencing | | $F_{1, 72} = 6.39$ | **0.014** | | $F_{1, 72} = 9.66$ | **0.003** |
| Rooting | | $F_{1, 72} = 94.25$ | **< 0.001** | | $F_{1, 72} = 1.67$ | 0.200 |
| Fencing*Rooting | | $F_{1, 72} = 0.40$ | 0.529 | | $F_{1, 72} = 1.17$ | 0.283 |
| **Percentage cover other plants** | None | | | None | | |
| Site | | $F_{4, 72} = 2.64$ | 0.051 | | $F_{4, 72} = 1.27$ | 0.291 |
| Fencing | | $F_{1, 72} = 0.24$ | 0.627 | | $F_{1, 72} = 0.10$ | 0.751 |
| Rooting | | $F_{1, 72} = 0.00$ | 0.980 | | $F_{1, 72} = 7.54$ | **0.008** |
| Fencing*Rooting | | $F_{1, 72} = 0.56$ | 0.458 | | $F_{1, 72} = 0.14$ | 0.708 |
| **No. flowering stems** | Square Root | | | None | | |
| Site | | $F_{4, 72} = 1.43$ | 0.233 | | $F_{4, 72} = 2.01$ | 0.102 |
| Fencing | | $F_{1, 72} = 9.64$ | **0.003** | | $F_{1, 72} = 12.46$ | **0.001** |
| Rooting | | $F_{1, 72} = 160.01$ | **< 0.001** | | $F_{1, 72} = 7.72$ | **0.007** |
| Fencing*Rooting | | $F_{1, 72} = 0.00$ | 0.978 | | $F_{1, 72} = 0.00$ | 0.945 |
| **No. flowers per stem** | None | | | None | | |
| Site | | $F_{4, 72} = 0.33$ | 0.860 | | $F_{4, 72} = 1.81$ | 0.135 |
| Fencing | | $F_{1, 72} = 10.97$ | **0.001** | | $F_{1, 72} = 2.35$ | 0.129 |
| Rooting | | $F_{1, 72} = 55.37$ | **< 0.001** | | $F_{1, 72} = 4.47$ | **0.038** |
| Fencing*Rooting | | $F_{1, 72} = 2.76$ | 2.760 | | $F_{1, 72} = 0.21$ | 0.645 |

**Table 3.1**. Three-way ANOVA test results at significance levels; Non-Significant: $p > 0.05$, Significant: **$p < 0.05$, $p < 0.01$**, Highly Significant: **$p < 0.001$**. Data were analysed using sequential sum of squares for tests. **F** = test statistic. **P** = probability associated with F-test. Significant **P** values shown in bold. Site = random factor. Fencing and Rooting = Fixed factors. Fencing*Rooting = Interaction. Transform = method of normalising data.

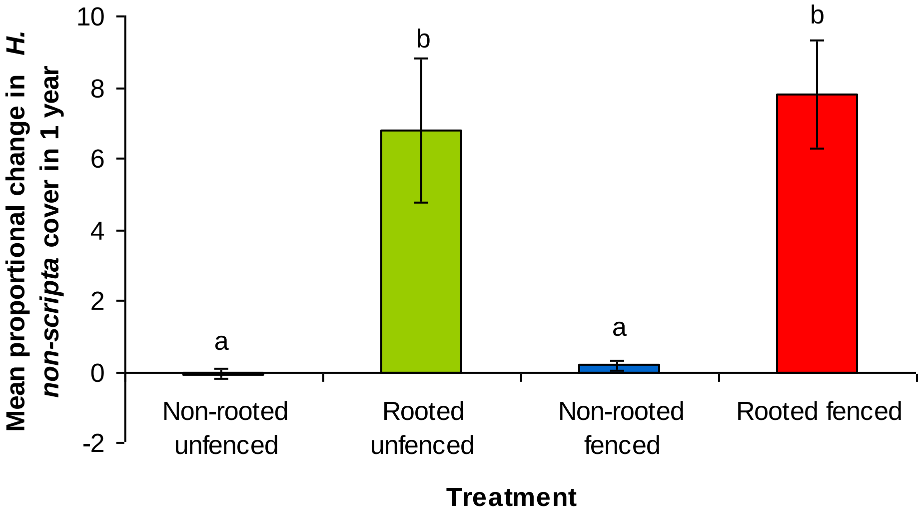
### *3.4.3 Proportional change in H. non-scripta cover*

Highly significant rooting effects show that the proportional change in *H. non-scripta* cover, after both one (between 2002 and 2003) and two (between 2002 and 2004) years was significantly greater in fenced and unfenced rooted than both non-rooted treatments (**Fig 3.6 a-b**, **Table 3.2**, **Appendix** 5).

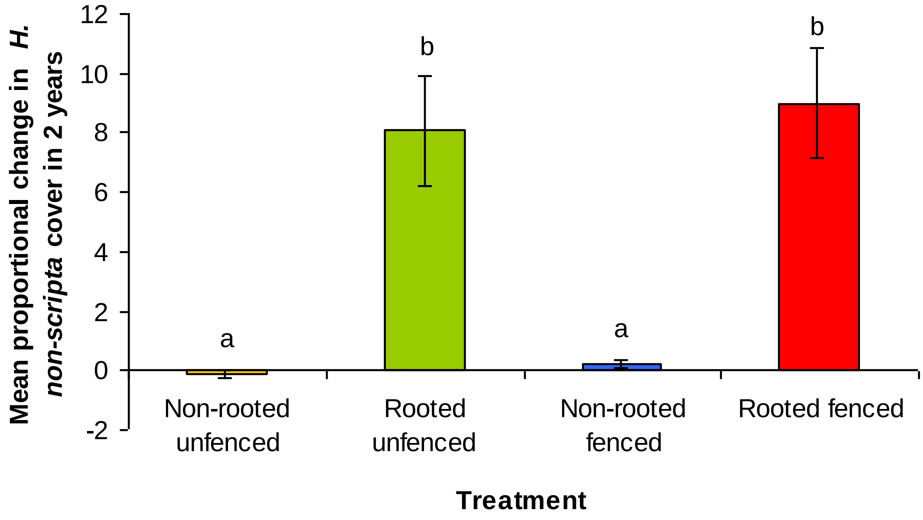
**Fig 3.6a-b:** Effects of rooting and fencing on the proportional change of *H. non-scripta* cover after one and two years. Data were pooled across five woodland sites between Spring 2002 and 2004. **a)** Mean proportional change in *H. non-scripta* cover over one year (2002 - 2003), **b)** Mean proportional change in *H. non-scripta* cover over two years (2002-2004). Differences between treatments (indicated by different letters above bars) were analysed using the Tukey test (significance level **p<0.05**). Error bars indicate standard error of the means, n = 4. For data conversion, and ANOVA results see **Section 3.3** and **Table 3.2** respectively.

🟧 Non-Rooted Unfenced 🟩 Rooted Unfenced 🟦 Non-Rooted Fenced 🟥 Rooted Fenced

**a) 2002 - 2003**



**b) 2002 - 2004**



**Effects of Rooting, Fencing and Site on the proportional change in *H. non-scripta* cover after one and two years**

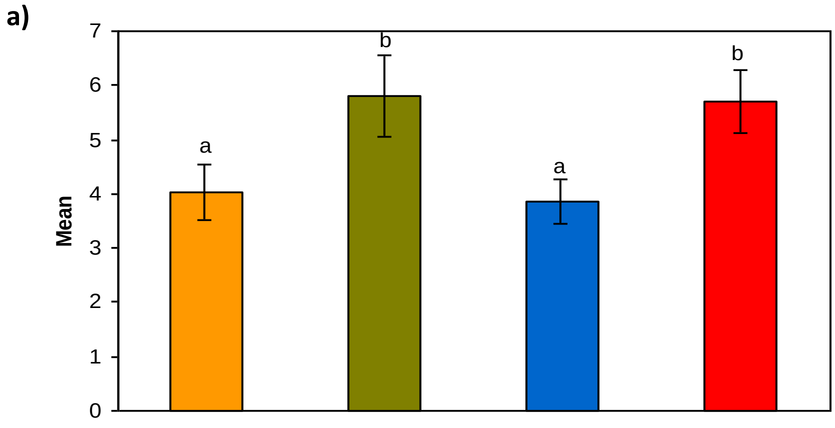
| **Source** | **F value & df** | **P** | **F value & df** | **P** |
|---|---|---|---|---|
| **Proportional Change in *H. non-scripta* Cover** | **After 1 Year (2002 - 2003)** | | **After 2 Years (2002 - 2004)** | |
| | | | | |
| Site | $F_{4, 72} = 0.72$ | 0.582 | $F_{4, 72} = 1.18$ | 0.325 |
| Fencing | $F_{1, 72} = 1.36$ | 0.248 | $F_{1, 72} = 2.27$ | 0.136 |
| Rooting | $F_{1, 72} = 158.88$ | **< 0.001** | $F_{1, 72} = 97.53$ | **< 0.001** |
| Fencing*Rooting | $F_{1, 72} = 0.06$ | 0.814 | $F_{1, 72} = 0.01$ | 0.932 |
| | | | | |

**Table 3.2**. The proportional change in *H. non-scripta* cover over one and two years. Statistical analysis was carried out on converted data (see **Section 3.3**). Three-way ANOVA test results at significance levels; Non-Significant: $p > 0.05$, Significant: **$p < 0.05$**, **$p < 0.01$**, Highly Significant: **$p < 0.001$**. Data were log10 (x +1) transformed. Data were analysed using sequential sum of squares for tests. **F** = test statistic. **P** = probability associated with F-test.

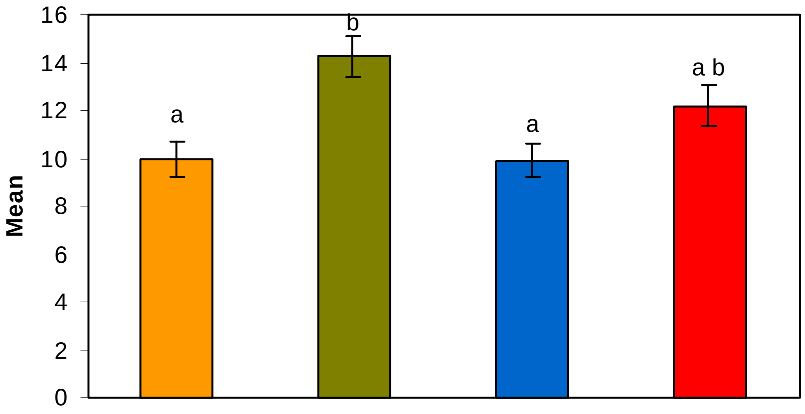
Rooting significantly altered the numbers of total seeds, ripe and unripe seeds, and ripe seeds produced per capsule and per flowering plant (Table 3.3). There were significantly more total seeds, ripe and unripe seeds, and ripe seeds produced per capsule and per plant in rooted than non-rooted treatments (Fig 3.7 a-b). There were also highly significant site effects for numbers of total seeds, and ripe and unripe seeds per capsule and per plant. There were no significant effects of treatment or site on the numbers of capsules per plant (Table 3.3, Fig 3.7c).

**Fig 3.7 a-c:** Effects of rooting and fencing on **a)** number of ripe seeds per capsule, **b)** total number of seeds per capsule and **c)** number of capsules per plant. Data were pooled across five woodland sites. Seeds were collected in a single season (2003). Differences between treatments (indicated by different letters above bars) were analysed using the Tukey test (significance level **p<0.05**). Error bars indicate standard error of the means, n = 4. For ANOVA results see **Table 3.3**.

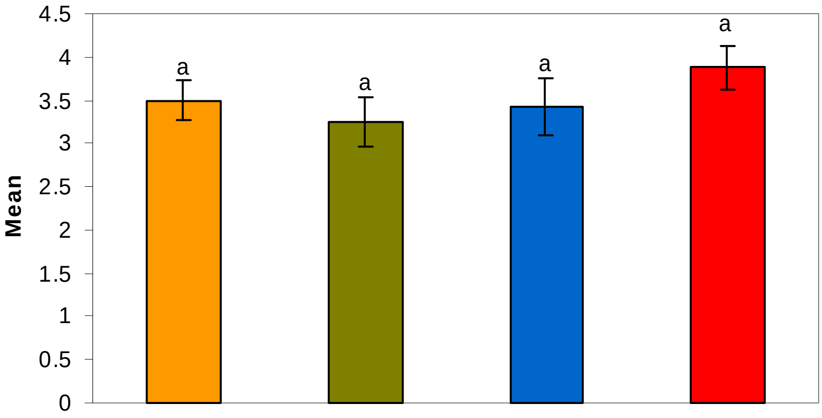
🟧 Non-Rooted Unfenced 🟩 Rooted Unfenced 🟦 Non-Rooted Fenced 🟥 Rooted Fenced



**b)**



**c)**



## Effects of Rooting, Fencing and Site on numbers of ripe, unripe and aborted *H. non-scripta* seeds in 2003.

| Source | F value & df | P | | Source | F value & df | P |
|---|---|---|---|---|---|---|
| ***Total no. seeds per plot*** | | | | ***No. ripe seeds per capsule*** | | |
| Site | $F_{4, 72}$ = 6.08 | **< 0.001** | | Site | $F_{4, 72}$ = 2.32 | 0.065 |
| Fencing | $F_{1, 72}$ = 0.06 | 0.800 | | Fencing | $F_{1, 72}$ = 0.09 | 0.763 |
| Rooting | $F_{1, 72}$ = 0.83 | 0.365 | | Rooting | $F_{1, 72}$ = 5.81 | **0.019** |
| Fencing*Rooting | $F_{1, 72}$ = 0.15 | 0.702 | | Fencing*Rooting | $F_{1, 72}$ = 0.16 | 0.687 |
| | | | | | | |
| ***Total no. seeds per plant*** | | | | ***No. unripe Seeds Per Plant*** | | |
| Site | $F_{4, 72}$ = 10.27 | **< 0.001** | | Site | $F_{4, 72}$ = 10.61 | **< 0.001** |
| Fencing | $F_{1, 72}$ = 1.29 | 0.260 | | Fencing | $F_{1, 72}$ = 0.82 | 0.367 |
| Rooting | $F_{1, 72}$ = 15.48 | **< 0.001** | | Rooting | $F_{1, 72}$ = 6.82 | **0.011** |
| Fencing*Rooting | $F_{1, 72}$ = 0.03 | 0.866 | | Fencing*Rooting | $F_{1, 72}$ = 0.12 | 0.728 |
| | | | | | | |
| ***Total no. seeds per capsule*** | | | | ***No. unripe seeds per capsule*** | | |
| Site | $F_{4, 72}$ = 10.27 | **< 0.001** | | Site | $F_{4, 72}$ = 10.61 | **< 0.001** |
| Fencing | $F_{1, 72}$ = 1.29 | 0.260 | | Fencing | $F_{1, 72}$ = 0.82 | 0.367 |
| Rooting | $F_{1, 72}$ = 15.48 | **< 0.001** | | Rooting | $F_{1, 72}$ = 6.82 | **0.011** |
| Fencing*Rooting | $F_{1, 72}$ = 0.03 | 0.866 | | Fencing*Rooting | $F_{1, 72}$ = 0.12 | 0.728 |
| | | | | | | |
| ***No. ripe and unripe seeds per plant*** | | | | ***No. aborted seeds per plant*** | | |
| Site | $F_{4, 72}$ = 4.54 | **0.002** | | Site | $F_{4, 72}$ = 3.69 | **0.009** |
| Fencing | $F_{1, 72}$ = 0.62 | 0.435 | | Fencing | $F_{1, 72}$ = 0.96 | 0.331 |
| Rooting | $F_{1, 72}$ = 20.50 | **< 0.001** | | Rooting | $F_{1, 72}$ = 2.36 | 0.129 |
| Fencing*Rooting | $F_{1, 72}$ = 0.47 | 0.494 | | Fencing*Rooting | $F_{1, 72}$ = 0.00 | 0.945 |
| | | | | | | |
| ***No. ripe and unripe seeds per capsule*** | | | | ***No. aborted seeds per capsule*** | | |
| Site | $F_{4, 72}$ = 4.54 | **0.002** | | Site | $F_{4, 72}$ = 3.69 | **0.009** |
| Fencing | $F_{1, 72}$ = 0.62 | 0.435 | | Fencing | $F_{1, 72}$ = 0.96 | 0.331 |
| Rooting | $F_{1, 72}$ = 20.50 | **< 0.001** | | Rooting | $F_{1, 72}$ = 2.36 | 0.129 |
| Fencing*Rooting | $F_{1, 72}$ = 0.47 | 0.494 | | Fencing*Rooting | $F_{1, 72}$ = 0.00 | 0.945 |
| | | | | | | |
| ***No. ripe seeds per plant*** | | | | ***No. capsules per plant*** | | |
| Site | $F_{4, 72}$ = 2.32 | 0.065 | | Site | $F_{4, 72}$ = 2.36 | 0.061 |
| Fencing | $F_{1, 72}$ = 0.09 | 0.763 | | Fencing | $F_{1, 72}$ = 3.43 | 0.068 |
| Rooting | $F_{1, 72}$ = 5.81 | **0.019** | | Rooting | $F_{1, 72}$ = 0.56 | 0.459 |
| Fencing*Rooting | $F_{1, 72}$ = 0.16 | 0.687 | | Fencing*Rooting | $F_{1, 72}$ = 0.10 | 0.752 |

**Table 3.3.** Three-way ANOVA results at significance levels; Non-significant: $p > 0.05$, Significant: $\mathbf{p < 0.05}$, $\mathbf{p < 0.01}$, Highly Significant: $\mathbf{p < 0.001}$. Data were analysed using sequential sum of squares for tests. All data were square root transformed. Significant **P** values shown in bold. **F** = test statistic. **P** = probability associated with F-test. Site = random factor. Fencing and Rooting = Fixed factors.

### *3.4.5 Seed weight*

There were no significant effects of rooting or fencing on any of the three measurements of seed weight (Table 3.4). However, significant site effects for ripe seed weight per plot, ripe seed weight per capsule and ripe seed weight per capsule per flowering plant indicate that site differences independently of rooting affect seed weight.

**Effects of Rooting, Fencing and Site on *H. non-scripta* ripe seed weights in 2003.**

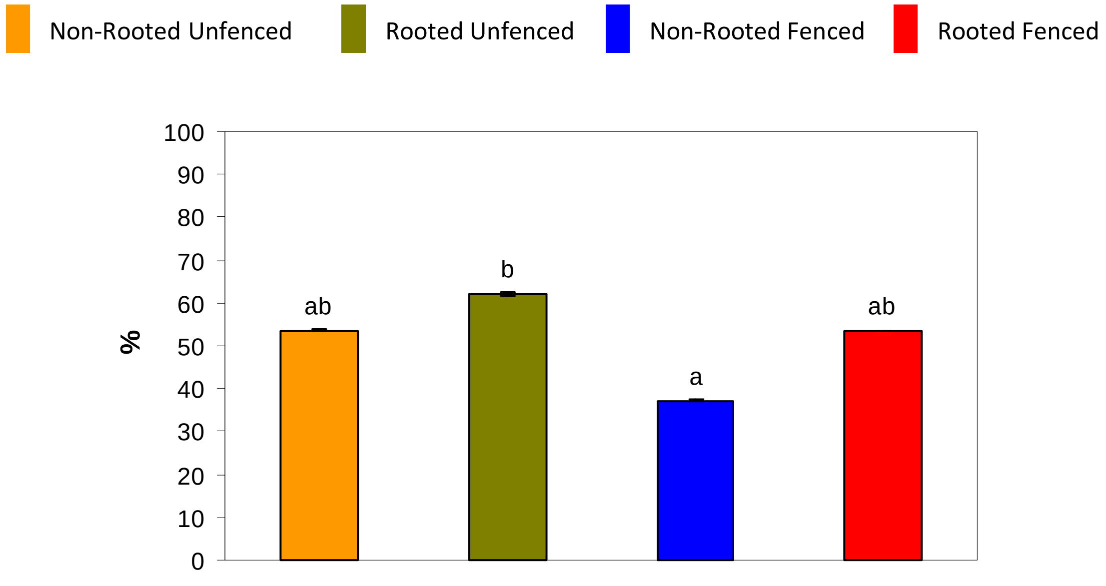
| **Source** | **F value & df** | **P** |
|---|---|---|
| | | |
| ***Ripe seed weight per plot*** | | |
| Site | $F_{4, 72} = 3.53$ | **0.011** |
| Fencing | $F_{1, 72} = 1.51$ | 0.223 |
| Rooting | $F_{1, 72} = 0.25$ | 0.618 |
| Fencing*Rooting | $F_{1, 72} = 0.05$ | 0.829 |
| | | |
| ***Ripe seed weight per capsule*** | | |
| Site | $F_{4, 72} = 2.88$ | **0.029** |
| Fencing | $F_{1, 72} = 0.76$ | 0.387 |
| Rooting | $F_{1, 72} = 3.90$ | 0.052 |
| Fencing*Rooting | $F_{1, 72} = 0.00$ | 0.995 |
| | | |
| ***Ripe seed weight per capsule per plant*** | | |
| Site | $F_{4, 72} = 2.88$ | **0.029** |
| Fencing | $F_{1, 72} = 0.76$ | 0.387 |
| Rooting | $F_{1, 72} = 3.90$ | 0.052 |
| Fencing*Rooting | $F_{1, 72} = 0.00$ | 0.995 |

### *3.4.6 Seed viability*

51.6 % (516 seeds) of the total 1000 *H. non-scripta* seeds germinated. Of the seeds that germinated, 26% were from unfenced non-rooted treatments, 18% from fenced non-rooted, 30% from unfenced rooted and 26% from fenced rooted treatments. For percentage germination from each treatment of 250 seeds, see **Fig 3.8** below. A highly significant rooting effect indicates that rooting strongly influenced the viability of *H. non-scripta* seeds germinating under lab conditions. More *H. non-scripta* seeds from unfenced rooted treatments germinated than from any other treatment (**Table 3.5**). A highly significant site effect shows that site differences affect the viability of *H. non-scripta* seeds independently of rooting. Pair-wise comparisons reveal a significant difference between rooted unfenced and non-rooted fenced treatments (**Fig 3.8**).

**Fig 3.8:** Effects of rooting and fencing on the number of germinating *H. non-scripta* seeds. Data were pooled across fi n treatments (indicate **<0.05**). Error bars indi

**Table 3.4**. Three-way ANOVA results at significance levels; Non-significant: p > 0.05, Significant: **p < 0.05**, **p < 0.01**, Highly Significant: **p < 0.001**. Data were analysed using sequential sum of squares for tests. All data were square root transformed. Significant **P** values shown in bold. **F** = test statistic. **P** =



## Effects of Rooting, Fencing and Site on the viability of *H. non-scripta* seeds

| **Source** | **F value & df** | **P** |
|---|---|---|
| | | |
| ***Total no. germinated* H. non-scripta *seeds*** | | |
| Site | $F_{4, 72} = 26.08$ | **< 0.001** |
| Fencing | $F_{1, 72} = 0.47$ | 0.495 |
| Rooting | $F_{1, 72} = 14.16$ | **< 0.001** |
| Fencing*Rooting | $F_{1, 72} = 0.39$ | 0.531 |

**Table 3.5**. Three-way ANOVA results at significance levels; Non-significant: $p > 0.05$, Significant: $\mathbf{p < 0.05}$, $\mathbf{p < 0.01}$, Highly Significant: $\mathbf{p < 0.001}$. Data were analysed using sequential sum of squares for tests. Data were $Log_{10}$ (X +1) transformed. Significant **P** values shown in bold. **F** = test statistic. **P** = probability associated with F-test. Site = random factor. Fencing and Rooting

## 3.5 Discussion

As hypothesised, rooting had a significant impact on the *H. non-scripta* population. The numbers of *H. non-scripta* in each plot were significantly reduced immediately after rooting and after a further one and two years (**Fig 3.5**, **Fig 3.6 a-b**, **Table 3.1**, **Appendix 5**). Further, the proportional change in *H. non-scripta* cover after both one and two years was significantly greater in both rooted treatments. Boar consume *H. non-scripta* bulbs when they root amongst them (Goulding *et al.*, 1998; Goulding, 2003a, 2003b), thus it is not surprising that *H. non-scripta* abundance is reduced in rooted areas. However, re-establishment of *H. non-scripta* back into rooted treatments over time appears to be rapid in both fenced and unfenced treatments. The ability of *H. non-scripta* to re-establish within rooted areas, coupled with its competitive ability over other plant species that grow in these rooted gaps is probably crucial to its continued existence in an environment occupied by wild boar.

As hypothesised, site differences had a significant impact on the number of *H. non-scripta* individuals present after one and two years (**Table 3.1**). Different management regimes across the five sites (**Table 1.1**) probably largely contributed to the significant site effects through potentially creating variation in aspects of physical (such as light), biological (such as population densities) and chemical (such as nutrients) elements of the above and belowground community. This is illustrated by inconsistencies in *H. non-scripta* population densities across sites for example; **W3** is considerably less densely populated by *H. non-scripta* than the other four sites (**Table 1.1**).

Although rooting is a form of disturbance that significantly reduces the abundance of *H. non-scripta*, typical characteristics of rooting allows the facilitation of subsequent recovery. Due to the typically small (~1m$^2$) patches of rooting (**Fig 3.2**), coupled with the relatively poor dispersal ability of *H. non-scripta* (Blackman and Rutter, 1954), the close proximity of many surrounding adult individuals to disturbed patches provides advantageous conditions for the re-establishment of this plant species by seed. Furthermore, *H. non-scripta* bulb growth tends to respond poorly to impeded drainage, which can particularly occur in heavy soils (Grabham and Packham, 1983). A light soil with a fine crumb structure allows the bulb to migrate deeper into the soil, which can provide a measure of protection from, for example, boar rooting (Grabham and Packham, 1983). Not all the exposed *H. non-scripta* bulbs are consumed during a rooting episode (**Fig 3.3a**), and a proportion of those discarded may survive. Rootlets on bulbs have the ability to pull the bulbs back down into well-drained soil during the growing season (Knight, 1964) and re-establish. This is especially likely to occur in fenced rooted areas due to the prevention of further rooting and trampling that fencing provides. Soil structure is therefore of particular importance to bulbous perennials whose storage of assimilates are belowground in a bulb that is renewed annually. Rooting aerates soil and can ‘improve’ crumb structure (Brownlow, 1994), which could have contributed to the recovery of *H. non-scripta* after rooting in the predominantly clay soils in the study area.

As hypothesised, the numbers of flowering *H. non-scripta* stems were significantly lower in rooted treatments, at both one and two years after rooting (**Table 3.1**, **Fig 3.5**). However, the numbers of flowering stems in fenced rooted treatments equalled that in non-rooted treatments after two years, illustrating a rapid recovery of *H. non-scripta* in the short-term when the disturbance is suspended. Peace and Gilmour (1949) found repeated damage to *H. non-scripta* leaves due to trampling (as might happen for example whilst rooting) produced successively smaller plants. Leaf damage would adversely affect photosynthesis.

This could in turn affect fruit formation, reproduction, and the size, vigour and number of successive plants (Peace and Gilmour, 1949). Cooke (1997) looked at the effects of grazing *H. non-scripta* by muntjac deer (*Muntiacus reevesi*) and found reductions of leaf length for unfenced plants to be 43% in April and 33% in May. It was determined that damaged leaves were not replaced and severe damage led to successively smaller plants in the following years. Cooke (1997) demonstrated a correlation between leaf damage and the number of inflorescences surviving to flower. In this study, although not quantified, the observed leaf damage caused by trampling whilst rooting may have produced the significant reduction of flowering stems in rooted treatments after one and two years. Moreover, the prevention of further trampling in fenced rooted treatments probably aided the significant recovery of numbers of flowering stems in fenced rooted, than all other treatments. Partial recovery of leaf length in populations within fenced areas in Cooke's study (1997) reveals that the production of damage-induced smaller plants may be reversible if the disturbance is suspended.

Damage early in the growing season has the greatest effect on *H. non-scripta* (Cooke, 1997; Blackman and Rutter, 1954). The seasonal timing of leaf damage through rooting could therefore be an important factor affecting the severity of damage to the *H. non-scripta* population. For example, rooting that occurs before *H. non-scripta* reproduction could reduce the number of flowering stems more than rooting that occurs after reproduction. Over successive years, this could lead to a considerably reduced abundance of *H. non-scripta.*

Contrary to predictions, the total numbers of all seeds and ripe seeds per capsule were higher in rooted than non-rooted treatments (**Fig 3.7 a-b**), although treatment had no affect on numbers of capsules per plant (**Fig 3.7 c**). Fencing did not affect seed production (**Table 3.3**).

High numbers of aborted seeds present in the capsules at collection exemplify a premature collection. A later seed collection could potentially have increased the proportion of ripe seeds, and thus the value of these data. Seed production in *H. non-scripta* varies from 10-100 viable seeds per plant (Blackman and Rutter, 1947; Grabham and Packham, 1983). Mean numbers of *H. non-scripta* seeds per plant in combined rooted treatments were 47 and in combined non-rooted treatments were 33. Rooting did not affect *H. non-scripta* seed weight (**Table 3.4**). Grabham and Packham (1983) however found *H. non-scripta* seeds were heavier when produced by a smaller number of flowering individuals although numbers of seeds produced were the same. This may be a compensatory mechanism of *H. non-scripta* in order to produce heavier seeds that are more viable in times of stress.

Allocation of resources to seeds during development is partitioned between number and size, and size is the least plastic of the components of reproductive output (Harper *et al.*, 1970). Seed number and seed size represent alternative strategies in the disposition of reproductive resources (Harper *et al.*, 1970). If seed weight is constant (as in this study), it is more likely that resources will be allocated to seed number during development (Harper *et al.*, 1970) as found here. The number of seeds borne by a plant is largely determined by the size of the "annual assimilated income", and the proportion of this devoted to seed number and size (Harper *et al.*, 1970). Reduced intra-specific competition resulting from increased space in rooted patches and more sparsely populated sites (such as **W3**) increases resource availability for plants (Kotanen, 1994). The higher levels of nutrients that accumulate in rooted soil (Lacki and Lancia, 1983; Singer *et al.*, 1984; Lacki and Lancia, 1986; Groot Bruinderink and Hazebroek, 1996; Bialy, 1996; Moody and Jones, 2000) (**Chapter 5**) could increase the annual assimilated income per individual *H. non-scripta* plant and lead to increased seed production. Additionally, potentially different nutrient levels across sites could also lead to inconsistencies in the annual assimilated income and thus contribute to the significant site effects relating to seed weight and numbers. Higher light levels were found to increase seed production in *H. non-scripta* by up to ten times than individuals growing in adjacent, more shaded areas (Knight, 1964). Knight (1964) suggested that an observed increase in insect visitations to plants growing in sunnier areas, increasing their chance of fertilisation, could explain the greater seed out-put. Potential differences in light levels across sites determined by different management regimes (such as coppicing in **W1** and **W2** in contrast with occasional thinning in **W3** – **W5**) could therefore have contributed to a variation in seed out-put across the five woodland sites.

516 out of 1000 (51%) seeds germinated under lab conditions. This low success rate is similar to that found by others (Slade and Causton, 1979). Slade and Causton (1979) however used scarification as a prerequisite for germination and found that this process significantly increased germination, but only with non-stratified seeds. Knight (1964) stated darkness provided by leaf litter is an important prerequisite for germination of *H. non-scripta* seeds. It appeared that the dark treatment given to seeds in this study catalysed the initiation of germination seven months after stratification.

As hypothesised, of the seeds that germinated, those from unfenced rooted plots had the highest germination rate (**Fig 3.8**, **Table 3.5**). Rooting increased the viability of *H. non-scripta* seeds. This could be because rooted soil provides more accessible nutrients (Lacki and Lancia, 1983; Singer *et al.*, 1984; Lacki and Lancia, 1986; Groot Bruinderink and Hazebroek, 1996; Bialy, 1996; Moody and Jones, 2000) (**Chapter 5**), which could lead to an increased annual assimilated income during plant growth, ultimately resulting in greater seed viability, whether measured in the field or in the lab. The germination responses, (including soil-moisture, light intensities, and, most importantly according to Thompson and Cox

(1978), air and soil temperature fluctuations) appear to provide a crucially balanced mechanism giving maximum chance of success on established sites (Thompson and Cox, 1978). However, lab conditions for germination in this experiment provide homogenous germination responses for all treatments, so any observed differences in germination are due to differences in seed viability, which was determined during growth. Similarly with seed weight and numbers, site differences significantly affected seed viability. Potential variation across sites in nutrient and light levels leading to possible differences in the annual assimilated income during growth, could have largely contributed to differences in the viability of *H. non-scripta* seeds across the five sites.

As hypothesised, rooting had a significant negative impact on the size of the *H. non-scripta* population. However, recovery of *H. non-scripta* in rooted treatments was very successful within one and two years after rooting. Rooting is patchy and fluctuates in distribution both temporally and spatially allowing recovery of *H. non-scripta* populations. Rooting had a dramatic and highly significant localised effect on all measured aspects of *H. non-scripta* ecology in the short-term, but thereafter, recovery quickly appears to take place. Research into the recovery process (extending into at least a fourth or fifth growing season) could be very illuminating and important work relating to long-term effects of rooting on *H. non-scripta*. Further, relating to the recovery of *H. non-scripta*, leaf length could be an important element of future research. Knowledge of the relative effects of single and repeated rooting events on leaf length could be used as an indicator of *H. non-scripta* recovery patterns in fenced and unfenced areas.

Fencing had less effect than hypothesised, but affected some demographic measurements of *H. non-scripta* populations, most notably numbers of individuals and flowering stems one year after rooting. There are no effects of fencing on the proportional change in *H. non-scripta* cover, or the numbers or weights of *H. non-scripta* seeds. Fencing reduced the disturbance frequency and thus the overall intensity of rooting. Suspension of further rooting through fencing on recently rooted areas for two years, aids the re-establishment of *H. non-scripta*, and could be a valuable management and conservation tool in the protection of *H. non-scripta* populations in relatively small, severely rooted woodland areas.

Site also had less impact than hypothesised, although greatly affected numbers of *H. non-scripta* individuals after both one and two years. Additionally, site differences substantially influenced seed weight, number and viability. Probable reasons stem from differences in management across sites leading to variation in light and nutrient levels and the *H. non-scripta* population densities.

# Chapter 4

## Effects of rooting on the viable seed bank at the community and functional group level

### 4.1 Introduction

Soils of most woodland and grassland habitats contain viable, ungerminated seeds produced in current or previous reproductive years. Seeds form transient (< 1 year), short term persistent (1-5 years) and long term persistent (> 5 years) seed banks (Thompson *et al.,* 1997), which are integral components of plant communities and fundamental for the population dynamics of the participating plant species (Hyatt and Casper, 2000). Seeds from transient seed banks occur in upper soil layers and are adapted to exploit gaps created by 'seasonally predictable' damage such as plant mortality (Thompson and Grime, 1979). Persistent seed banks exist predominantly in lower soil layers. Dormancy is broken when soil disturbance is unpredictable both temporally and/or spatially, providing suitable conditions for regeneration (Thompson and Grime, 1979).

Persistent seed banks have several ecological functions. For example, by acting as a 'propagule reservoir', chances of plant population extinctions are reduced (Venable and Brown, 1988). Persistent seed banks may alter population genetic structure in that their allelic frequencies may be different to those in the above ground section of the population (Levin, 1990). Consequently, the persistent seed bank represents a store of evolutionary memory (Harper, 1977). Species with a seed bank may co-exist in temporally variable environments (Pake and Venable, 1995) and thus contribute to community diversity (Thompson and Grime, 1979; Lunt, 1997; Diaz-Villa *et al.*, 2003). An important role of the persistent seed bank in relation to my research is the potential for establishment of the plant community following environmental changes. For example tree falls, ecological restoration and biopedturbation, all of which trigger germination of dormant seed (Putz, 1983; Martinsen *et al.*, 1990; Bekker *et al.*, 1997; Brown, 1998; Hyatt, 1999; Hyatt and Casper, 2000; McAlpine and Drake, 2002).

Disturbances modify spatial and temporal availability of resources and thus play a central role in structuring plant and animal communities (Picket and White, 1985). Disturbance of vegetation and soil enhances germination and seedling emergence via breaking seed dormancy mechanisms in many plant species (Sauer and Struik, 1964; Moore and Wein, 1977; Putz, 1983; Buckley *et al.*, 1997; Pugnaire and Lozano, 1997; Rydgren *et al.*, 1998; Hyatt and Casper, 2000; Jutila and Grace, 2002). Seeds of different plant species have different dormancy breaking and germination requirements, thus respond differently to environmental conditions (Bradbeer, 1988; Fenner, 1985, 1992). Therefore, the soil environment coupled with particular traits of different seeds controls the ability of dormant seed to germinate (Baskin and Baskin, 1998).

Perturbations that cause terrain movement (such as boar rooting) aid seed establishment by bringing seeds to the upper soil layer whereby exposure to physical agents trigger dormancy breaking mechanisms and germination. Rooting increases aeration (thus oxygen levels), temperature fluctuation and light levels (Brownlow, 1994; Kotanen, 1994; Arrington *et al.*, 1999), which are intrinsic physical prerequisites for dormancy breaking mechanisms and germination in the majority of plant species (Sauer and Struik, 1964; Thompson *et al.*, 1977; Grime *et al.*, 1981; Thompson and Grime, 1983; Fenner, 1992; Jutila and Grace, 2002). Rooting is therefore likely to have a substantial impact on emergence from the seed bank. Establishment from the seed bank following disturbance (such as rooting, which is typically patchy) could increase spatial heterogeneity and plant diversity (Thompson and Grime, 1979; Lunt, 1997; McAlpine and Drake, 2002; Diaz-Villa *et al.*, 2003).

Several field studies have shown rooting to enhance plant species richness over different habitat types (Bowman and McDonough, 1991; Kotanen, 1994, 1995; Welander, 1995; Onipchenko and Golikov, 1996; Milton *et al.*, 1997; Arrington *et al.*, 1999; Welander, 2000a) (**Chapter 2**, **Appendices 2-4**). However, information on relative contributions from the seed bank, seed rain or re-establishment from remaining clonal fragments is lacking. Newly exposed viable seeds in recently rooted environments to increased space, light, water and nutrients (due to removal of competition from above ground vegetation) (Kotanen, 1994) have greater opportunities for establishment. Field investigation of post-rooting seed bank emergence cannot eliminate propagules coming from other sources. Therefore, it was important to carry out a laboratory-based enquiry into the sole effects of rooting on emergence of seedlings from the viable seed bank as an exclusive source of novel plant growth.

The chemical environment is an important trigger for germination in the majority of plants (Bradbeer, 1988; Fenner, 1985, 1992). Nitrate ($NO_3^-$) can stimulate seed germination in plants across functional groups, but in particular, ruderal strategists (Saini *et al.*, 1985a; Saini *et al.*, 1985b; Fenner 1985, 1992; Crawley, 1997; Bradbeer, 1988; Goudey *et al.*, 1988). $NO_3^-$ interacts with biotic and abiotic factors such as water, temperature, light, responsiveness of seeds and other chemical soil constituents to stimulate germination in a complex system (Fenner, 1992).

Rooting increases soil $NO_3^-$ levels in the Great Smoky Mountains and Californian grasslands (Lacki and Lancia, 1983; Singer *et al.*, 1984; Kotanen, 1994) and in my study area (**Table 5.1**, **Fig 5.6**). Significant enhancement of $NO_3^-$ availability to the seed bank (coupled with increased light, oxygen and temperature) (Brownlow, 1994; Kotanen, 1994; Arrington *et al.*, 1999) is likely to have a large impact on seed germination in rooted environments. Since rooting is typically patchy (**Fig 1.1 a-c**), it could effectively form $NO_3^-$-rich 'resource islands' with increased temperature, light and oxygen levels, which could promote seed germination in a patchy manner and thus medium-scale (between-patch) spatial heterogeneity (Welander, 2000b) and species richness (Bowman and McDonough, 1991; Kotanen, 1995; Welander, 1995; Milton *et al.*, 1997) (**Table 2.1**, **2.2**, **2.3**).

Understanding emergence from the seed bank and thus the potential role of seed dormancy and germination for the regeneration of plants at the community and functional group level within disturbed, semi-natural woodland and grassland habitats, could be important for conservation (Thompson and Grime, 1979; Hill and Stevens, 1981; Dougall and Dodd, 1997). The effects of wild boar rooting (as a potentially severe form of disturbance) on emergence from the seed bank is, although patchy, likely to be extensive. Therefore, a continued presence of this former native animal in Britain substantiates the importance of understanding the consequences of rooting, for both the seed bank and recovery of the plant community.

Due to the severity of the disturbance to both soil and vegetation, rooting is likely to substantially affect emergence from the seed bank. The aim of this study was to assess short-term (> two months, < two years) impacts of rooting on emergence from the seed bank at the community and functional group level.

### *4.1.1* ***Hypotheses***

#### 4.1.1.1 *Community level*

I hypothesised that there would be a greater number of plant individuals, diversity, evenness and plant species richness from seedling emergence in rooted than non-rooted soil from grassland and woodland habitats. This hypothesis was based on the following reasoning:

1. Under laboratory conditions, seeds will be exposed to experimentally controlled homogenous physical conditions. Variation remaining between soil samples is due to soil treatment before collection. Rooted soil contains greater levels of $NO_3^-$ than non-rooted soil (Lacki and Lancia, 1983; Singer *et al.*, 1984; Kotanen, 1994) (**Table 5.1**, **Fig 5.6**), and $NO_3^-$ stimulates seed germination (Saini *et al.*, 1985a; Saini *et al.*, 1985b; Fenner, 1985, 1992; Crawley, 1997; Bradbeer, 1988; Goudey *et al.*, 1988).

2. Seed scarification is a prerequisite for dormancy breaking mechanisms in some species (Baskin and Baskin, 1998). Trampling by hoofed animals (such as boar whilst rooting) could scarify some seeds (Bradbeer, 1988) in rooted soil before sample collection and effectively increase germination and emergence of seeds under lab conditions.

3. Trampling by boar and re-rooting may increase the incorporation of seeds that have recently landed from the seed rain onto the soil surface into the seed bank. Furthermore, mixing soil layers through rooting is likely to bring dormant, viable seed from the persistent seed bank to the upper soil layers, within the typical rooting zone (5-15cm). Thus, the sampled seed bank from rooted soil probably contains transient and persistent seeds and a greater overall number of seeds in the upper layers.

I hypothesised that fewer ancient woodland indicator (AWI) species (Marren, 1990; Rose, 1999) would emerge as seedlings in rooted than in non-rooted soil from woodland habitats. This hypothesis was based on the following reasoning:

- AWI species are characterised by a stress tolerant strategy (Hermy *et al.*, 1999; Wulf, 2003), thus they are not adapted for coping with severe disturbance such as boar rooting. It is therefore likely that the above ground vegetation in previously regularly rooted areas would not contain a high richness or abundance of AWI species. Having a characteristically poor dispersal capacity (Rackham, 1980; Hermy *et al.*, 1999; Bossuyt *et al.*, 2002; Wulf, 1997), the incorporation of seeds of AWI species into the seed bank is reliant on close proximity to populations of AWI species. Furthermore, AWI species are not associated with persistent seed banks (Grime, 1979) as the seeds tend to be short lived. The AWI species seed bank is therefore reliant on annual replenishment from currently occurring, nearby populations of AWI species and so is likely to be smaller in rooted compared to non-rooted areas.

I hypothesised that site differences would significantly affect the number of plant individuals, diversity, evenness, plant species richness and number of AWI species emerging as seedlings from the grassland and woodland seed bank. Different management regimes imposed on the different sites (**Table 1.1**) could have strongly influenced the above ground portion of the plant assemblage at the community level (**Table 2.1**, **2.2**, **2.3**). Subsequently, incorporation of seeds into the seed bank could be affected, altering community level elements of the seed bank such as overall size and diversity. Further, inconsistencies of physical disturbance (other than rooting) through the different forms of management across sites is likely to differentially induce germination and thus emergence from the seed bank across sites, additionally influencing community level elements of the seed bank such as overall size. Furthermore, the geographical distinction of the sites in relation to the different habitats and plant communities flanking each study site, would probably influence the seed bank composition due to differences in dispersal from the surrounding land, and contribute to altering the size and diversity of the seed bank across sites.

#### 4.1.1.2 *Functional group level*

I hypothesised that seeds of different functional groups would respond differentially to rooting (Aplet *et al.*, 1991) due to specific dormancy breaking and germination requirements of different plant groups (Jonsson and Essen, 1998). I specifically hypothesised that rooting would increase emergence of annual, biennial and perennial forbs, and decrease emergence of perennial graminoids and woody species in woodland and grassland. These hypotheses were based on the following ratiocination:

- **Annual and biennial forbs**: A greater number of seeds from species with ruderal strategies (majority of annual and many biennial forbs, **Appendix 1**) are likely to be incorporated into, and emerge from, the seed bank from rooted soil. Ruderals characteristically produce large numbers of small, wind dispersed seeds that are less likely to be consumed by granivores than larger, more palatable seeds (Crawley, 1997). Trampling by boar whilst rooting may increase incorporation of these seeds (due to their greater numbers and small size) from the soil surface into the seed bank. Ruderal strategists are associated with large, persistent seed banks and their emergence is dependent on disturbance-created gaps (Grime, 1979; Grime *et al.*, 1989; Crawley, 1997), such as rooted patches. The majority of ruderal seed loss from soil is attributable to germination (Crawley, 1997), and thus is greatest in disturbed (such as rooted) rather than undisturbed soil (Buckley *et al.*, 1997; Rydgren *et al.*, 1998; Hyatt and Casper, 2000; Jutila and Grace, 2002). Therefore, above ground population densities of seed producing ruderals and thus ruderal seed banks are probably greater in annually rooted than non-rooted environments. However, more frequently applied, severe rooting that prevents seed production is likely to deplete the seed bank and thus reduce the above ground vegetation. Furthermore, $NO_3^-$ is known to promote germination in dormant seed, and particularly from a variety of weed species (Saini *et al.*, 1985a; Saini *et al.*, 1985b; Goudey *et al.*, 1988; Fenner, 1992). Since rooted soil samples contain higher levels of $NO_3^-$ than non-rooted soil (Lacki and Lancia, 1983; Singer *et al.*, 1984; Kotanen, 1994) (**Table 5.1**,

**Fig 5.6**), emergence from seeds of ruderal strategists are predicted to be greater in rooted soil.

- **Perennial forbs**: Perennial forbs typically have persistent seed banks (Fenner, 1992) and many species within this functional group are ruderal in strategy (**Appendix 1**). Rooted soil is likely to contain more seeds from the persistent seed bank in the upper layers (top 10cm) than non-rooted soil due to the transfer of seeds from lower to upper layers during the mixing of soil horizons whilst rooting. Germination and emergence of these seeds is likely to be greater from rooted than non-rooted soil, due to a greater number of seeds present in the seed bank coupled with increased exposure to $NO_3^-$ (Lacki and Lancia, 1983; Singer *et al.*, 1984; Kotanen, 1994) (**Table 5.1**, **Fig 5.6**).

- **Perennial graminoids**: Although many perennial graminoids are associated with persistent seed banks (Thompson *et al.*, 1997), they represent a range of combinations of adaptive strategies within Grime's classification (Grime, 1974; Grime, 1979; Grime *et al.*, 1989) (**Appendix 1**), and therefore are not specifically adapted for coping with disturbance compared to other functional groups. Emergence of this functional group is therefore likely to be relatively lower than other functional groups in rooted than non-rooted soil in woodland and grassland.

- **Woody species**: Stress tolerant strategists (represented by many woody species (**Appendix 1**), are adapted for survival in continuously

> unproductive, non-highly disturbed environments (Grime, 1979; Grime *et al.*, 1989). They are not characterised by persistent seed banks, but rather with persistent seedling banks (Grime, 1979; Grime *et al.*, 1989) through the production of small numbers of large seeds as and when resources are available (Crawley, 1997). Seed production therefore does not necessarily occur annually and thus small numbers of seeds may be present in soil used in this study. Large seeds such as acorns are at higher risk of predation (Crawley, 1997) and thus are more likely to be palatable and consumed by boar whilst rooting (Henry and Conley, 1972; Baber and Coblentz, 1987; Groot Bruinderink and Hazebroek, 1994; Massei *et al.*, 1996; Goulding *et al.*, 1998; Goulding, 2003b). It is therefore expected that fewer seeds from woody species would be present in highly disturbed rooted soil collections and hence less emergence from rooted than non-rooted soil.

I hypothesised that site differences would significantly affect the number of individuals and species from each functional group emerging from the grassland and woodland seed bank. Different forms of management across different sites (**Table 1.1**) could have strongly influenced the relative abundance of the above ground portion of the plant assemblage at the functional group level (**Table 2.4**, **2.5**, **2.6**). Subsequently, it is likely that the belowground portion of the plant community would in part reflect this. Seeds from different functional groups have different dormancy breaking and germination requirements (Grime, 1979; Grime *et al.*, 1989; Saini *et al.*, 1985a; Goudey *et al.*, 1988; Fenner, 1992; Crawley, 1997; Buckley *et al.*, 1997; Rydgren *et al.*, 1998; Hyatt and Casper, 2000; Jutila and Grace, 2002). Different disturbance regimes imposed by different forms of management across sites could differentially affect the soil environment and thus emergence from the seed bank at this level. Subsequently, the seed bank size of each functional group would be likely to vary across sites. Furthermore, the geographical distinction of the sites in relation to the different habitats and plant communities flanking each study site, would probably influence the seed bank composition due to differences in dispersal from the surrounding land, and contribute to altering the relative abundance of functional groups across sites.

## 4.2 Materials and Methods

### *4.2.1* ***Study Area***

This study was undertaken within the five woodland (**W1-W5**) and the four grassland sites (**G1-G4**) in the study area (**Fig 1.5 a-d**, **Table 1.1**).

### *4.2.2 Experimental Design*

One hundred and twenty soil cores were randomly collected equally between rooted and non-rooted treatments within the five woodland and the four grassland sites (thirty soil cores per treatment per habitat type) using a 7cm diameter (38.5cm$^2$) x 10cm depth (385cm$^3$)

soil auger (**Fig 5.1**) during December 2003. The majority of viable seeds occur in and germinate from the upper layers of soil (Fenner, 1985; Moore and Wein, 1977). The sample depth corresponds with that adopted by Brown and Oosterhuis (1981), Hill and Stevens (1981), Milberg (1995), Rydgren and Hestmark (1997) and Jalili *et al.* (2003) and is within the typical rooting zone (5-15cm). At the soil surface, any loose litter was removed before soil cores were collected from both rooted and non-rooted sample sites. The time of collection allowed for supplementation of the transient seed bank from the current year's seed rain (Thompson *et al.*, 1997). The recent colder winter temperatures may have broken dormancy for those species that require cold stratification (Baskin and Baskin, 1998). Rooted treatments were rooted patches of varying size and were rooted between 6 and 12 months previously.

Soil samples were air-dried (Graham and Hutchings, 1988; Dutoit and Alard, 1995) on plastic trays and then carefully sieved through a 4mm mesh sieve to remove stones, dead plant biomass and other debris. Large seeds remaining in the sieve were easily visible and returned to the soil. The prepared soil/seed mixture was spread in a layer ~ 1 cm deep, over a ~ 2cm layer of washed coarse sand (for moisture retention) in 35cm x 21cm plastic trays (735cm$^2$) (Thompson and Grime, 1979; Graham and Hutchings, 1988; Dutoit and Alard, 1995; Thompson *et al.*, 1997; Onaindia and Amezaga, 1997, 2000). The trays were layered with paper towelling for further moisture retention. Two soil cores (770cm$^3$) from the same treatment, site and habitat were placed in each tray. There were 30 trays in total; 15 per treatment per habitat. The trays were placed in a growth room incorporating two shelves under a 16-hour light/ 8-hour dark regime (Thompson and Grime, 1979; Onaindia and Amezaga, 1997, 2000) (**Fig 4.1 a-b**, **4.2 a-c**). The trays were evenly rotated weekly into different positions on the shelves to avoid positional bias relating to light or temperature differences. Lighting utilised was twenty Phillips 100w 8ft white-light fluorescent tubes. Light intensity of the tubes on the lower shelf was 240 micromols per metre squared per second, and 104 micromols per metre squared per second on the upper shelf. Temperatures fluctuated by approximately 8°C around a mean of 20°C, which is close to the ideal of a fluctuation of 10°C around a mean of 20°C (Baskin and Baskin, 1998).

The trays were watered approximately three times weekly and, to prevent panning of the soil and burial of seeds and seedlings, watering was misting from above. Seedlings were identified to species (Phillips, 1980; Rose, 1981; Garrard and Streeter, 1983; Fitter, 1987) (**Appendix 1**) and counted twice weekly as seedlings emerged and grew (**Fig 4.3 a-b**). Identified seedlings were subsequently removed to prevent re-counting. Seedlings not readily distinguishable were potted and grown on until identification was possible. After all the species were identified and counted, the soil/

**a)**

**b)**

**Fig 4.1 a-b:** A large proportion of the seed-trays under growing lamps showing the second flush of seedling emergence across the two treatments and habitat types. Photo

**a)**

**b)**

**c)**

**Fig 4.2 a-c:** A selection of the seed trays showing the second flush of seedling emergence of predominantly perennial graminoids, across mainly rooted treatments in woodland. Photo taken 2004.

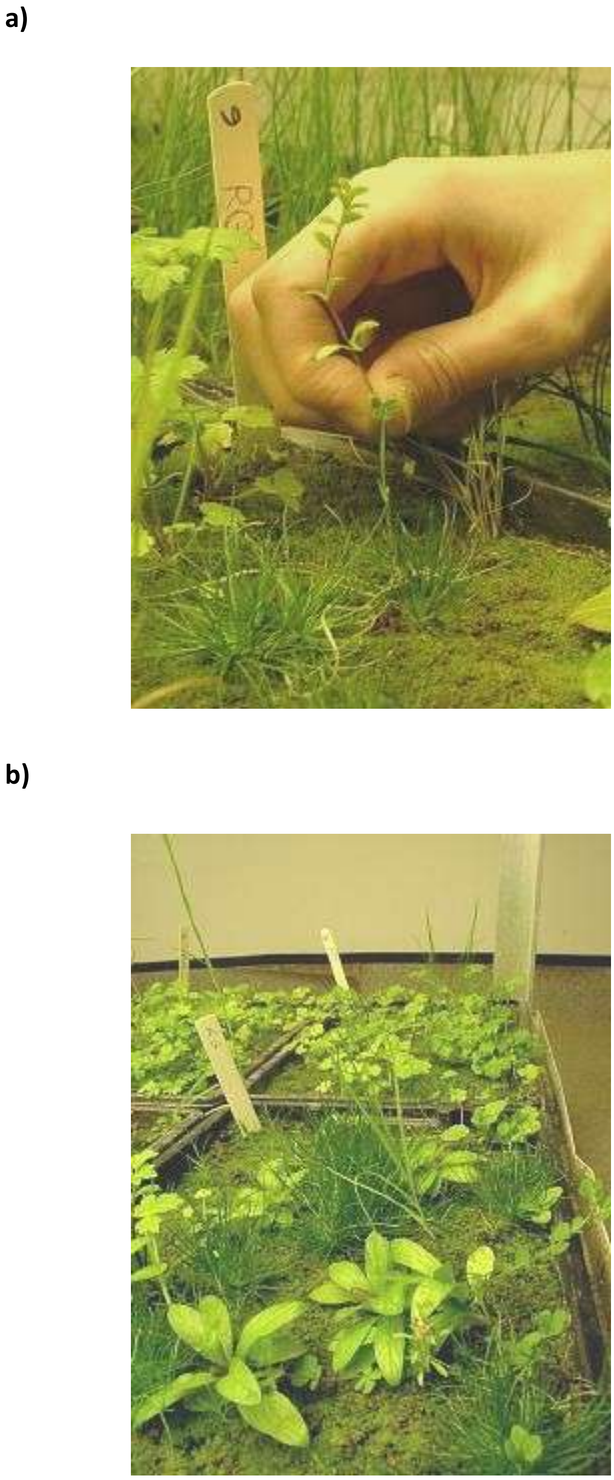


**Fig 4.3 a-b:** Close-up of a selection of seed trays from rooted treatments from grassland. **a)** Identifying species **b)** An initial flush of seedling emergence of predominantly *Ranunculus repens.* Photo taken 2004.

seed layer was turned to promote germination in any remaining viable ungerminated seeds. Identification continued until all germination and emergence of seedlings had ceased (6 months, December 2003- June 2004). Data were analysed in two ways:

1. **Community level** variables measured were:

   - Total number of individual seedlings per tray per treatment per habitat.

   - Total number of seedling species per tray per treatment per habitat.

   - Frequency of AWI species and individuals were counted per tray per treatment from woodland soil.

   - Shannon Weiner Diversity Index ($\mathbf{H' = -\sum p_i \log p_i}$) and Shannon Evenness ($\mathbf{E = H' / \log S}$) per tray per treatment per habitat. (**pi** = proportion of total sample belonging to the *i*th species. **H’** = index of species diversity, **S** = number of species, **E** = evenness) (Magurran, 2004).

2. **Functional group level** variables measured were:

   - Number of species and individuals of annual, biennial and perennial forbs, perennial graminoids, bulbous and woody species were counted per tray per treatment per habitat.

## 4.3 Statistical Analysis

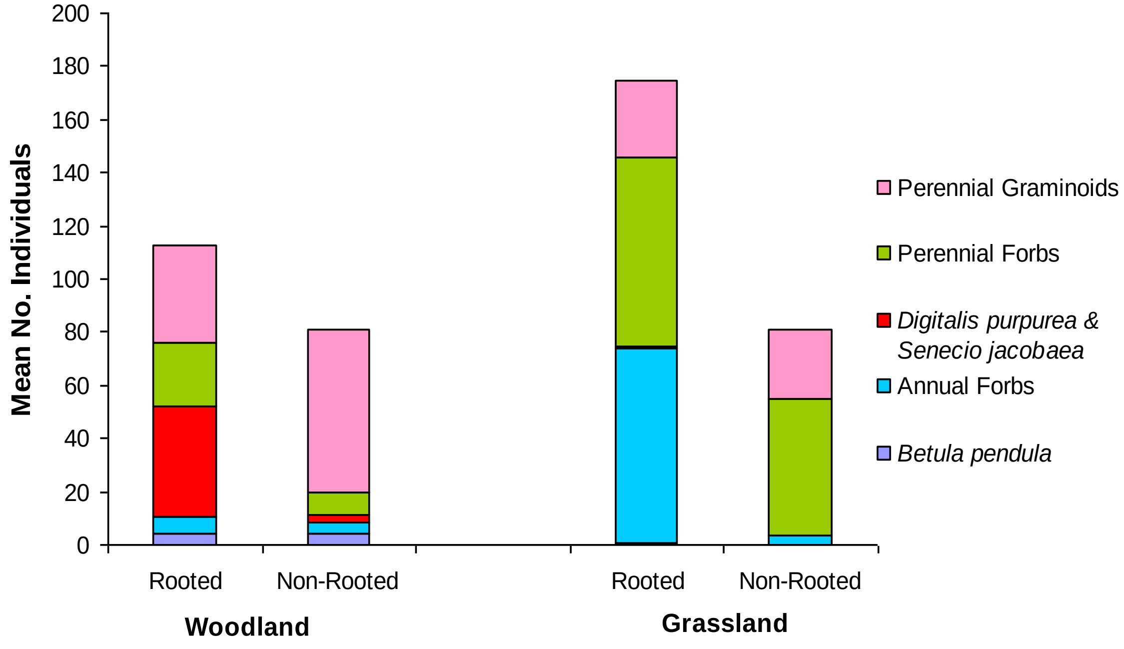
Data were analysed with two-way ANOVA where data were normally distributed (Eddison, 2000; Dytham, 2003). Data were log (Log10 x +1) or square root transformed to achieve normality where possible (specified **Table 4.1**, **4.2**, **4.3**). Where data were not normally distributed and could not be transformed, the Scheirer Ray Hare test was used instead (Eddison, 2000; Dytham, 2003). In both cases, rooting was a fixed factor and site was the second and random factor. Data were analysed independently within different habitat types and using the sequential sum of squares.

## 4.4 Results

### *4.4.1 Community level*

Highly significant rooting effects (**Table 4.1**) show that a greater number of plant individuals emerged from rooted than non-rooted treatments from the grassland seed bank (**Fig 4.4 a-b**). There was no effect of rooting on the number of individuals emerging from the woodland seed bank. Highly significant rooting effects (**Table 4.1**) show that a greater number of plant species emerged from rooted than non-rooted treatments from the grassland and woodland seed banks (**Fig 4.5 a-b**). There was no significant effect of site on the number of plant individuals and species emerging from the grassland and woodland seed bank.

**a)**



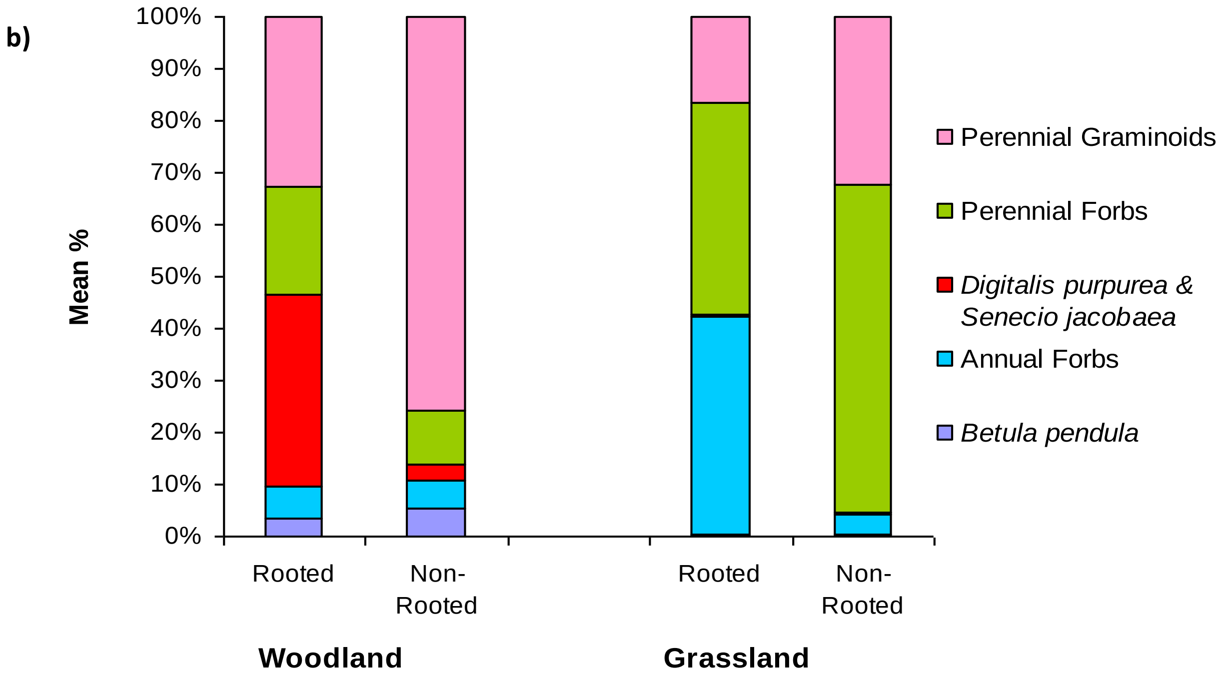
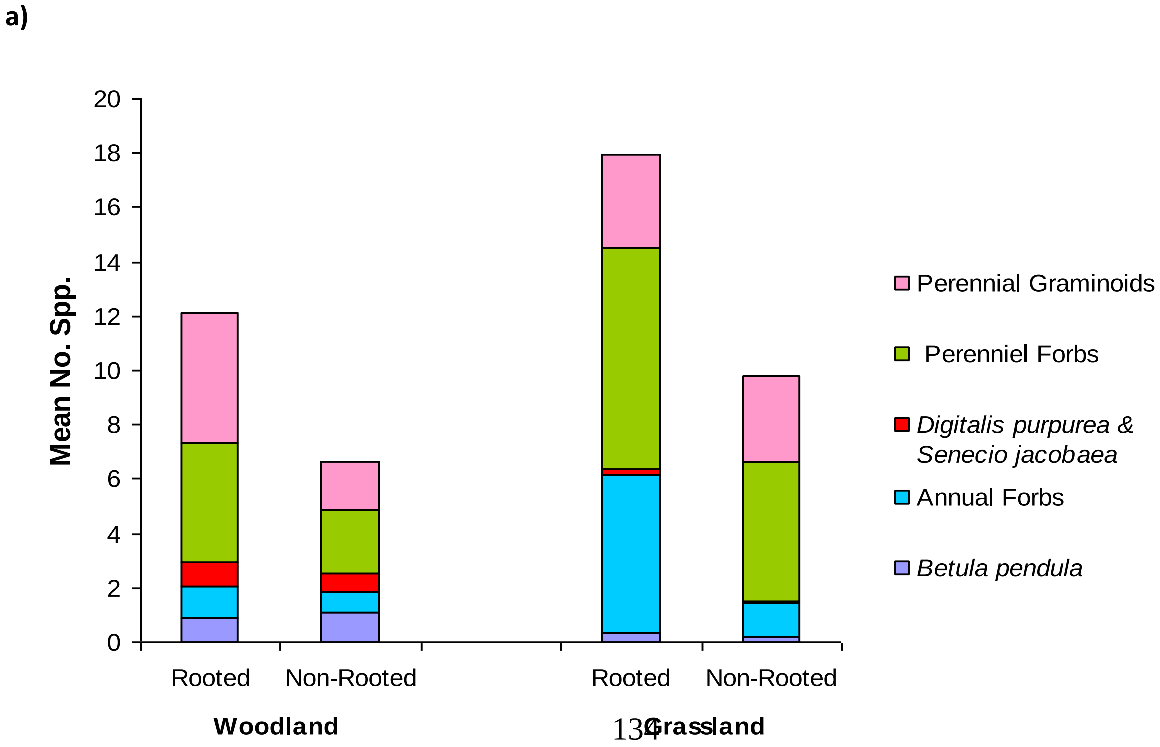


**Fig 4.4 a-b:** Effects of rooting on emergence from the viable seed bank across grassland and woodland habitats. Data represents a) number of different plant individuals and b) proportion of different plant individuals emerging as seedlings during 2004 across different functional groups from soil samples collected 2003. n = 15. For ANOVA results see Tables 4.1, 4.2, 4.3.



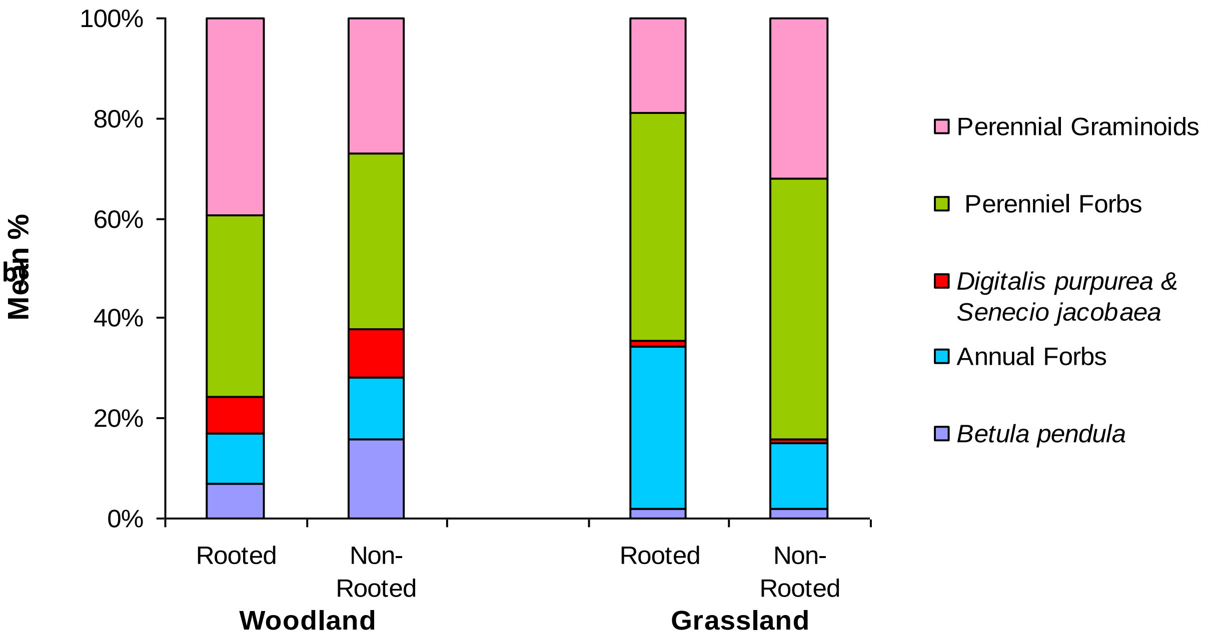


**Fig 4.5 a-b:** Effects of rooting on emergence from the viable seed bank across grassland and woodland habitats. Data represents a) number of different plant species and b) proportion of different plant species emerging as seedlings (during 2004) across different functional groups from soil samples collected 2003. n = 15. For ANOVA results see Tables 4.1, 4.2, 4.3.

Highly significant rooting effects show that the Shannon diversity index for seedling emergence was greater in rooted than non-rooted treatments within grassland and woodland seed banks. Shannon evenness however was significantly greater in rooted than non-rooted treatments from the woodland seed bank but there was no significant difference between treatments from the grassland seed bank (**Table 4.1**, **Fig 4.6 a-b**). Site had no significant effect on the Shannon diversity index or evenness of emergence from the woodland and grassland seed bank.

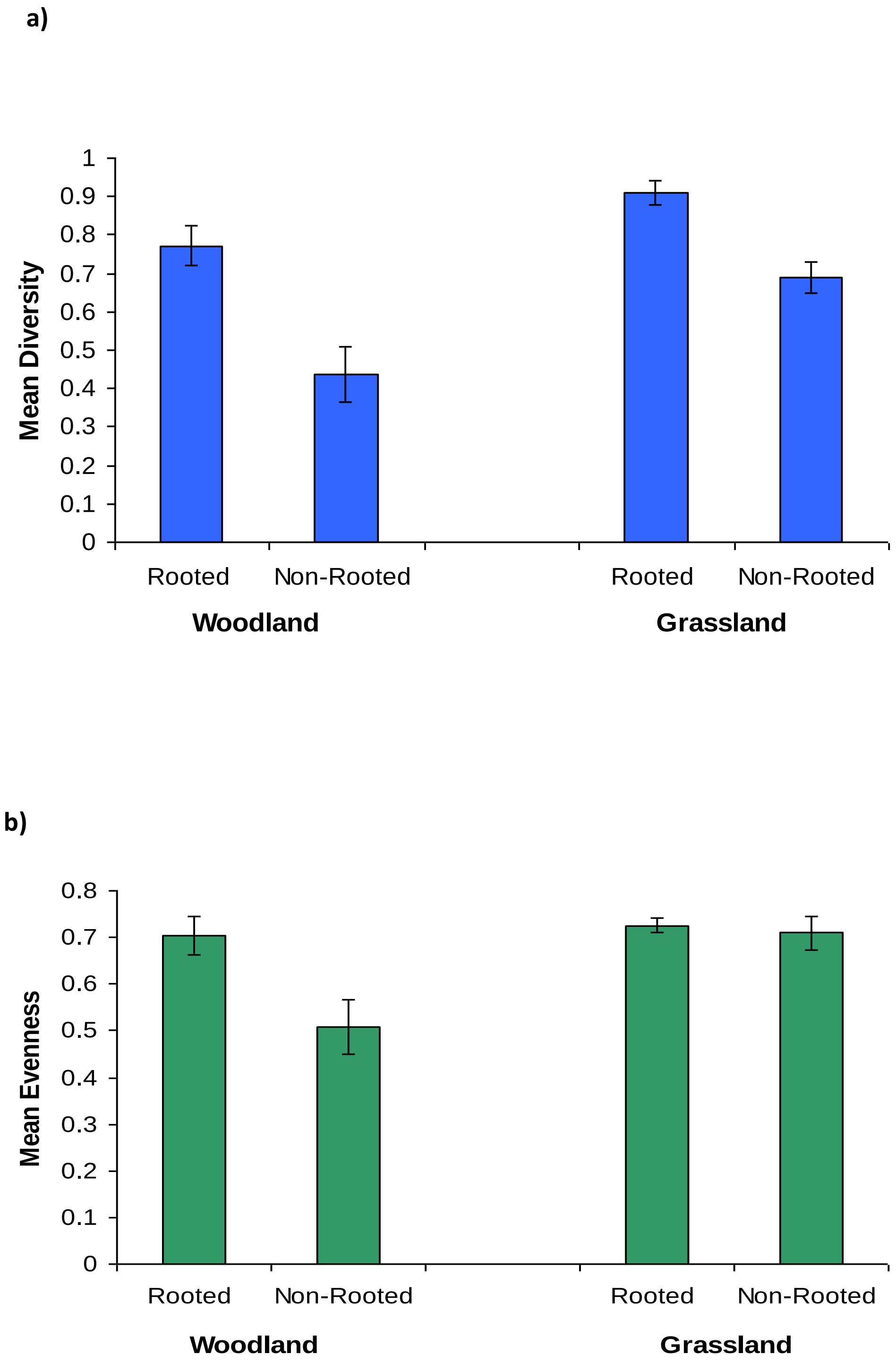


**Fig 4.6 a-b:** Effects of rooting on emergence from the viable seed bank across grassland and woodland habitats. Data represents **a)** Shannon diversity, and **b)** Shannon evenness of seedling emergence (during 2004) from soil samples collected 2003. Error bars indicate standard error of the means, n = 15. For ANOVA results see **Table 4.1.**

Highly significant rooting effects show that there were a greater number of AWI individuals and species from rooted than non-rooted treatments (**Table 4.1**, **Fig 4.7**). Non-significant site effects indicate that site differences did not affect the emergence of AWI individuals and species from the woodland seed bank.

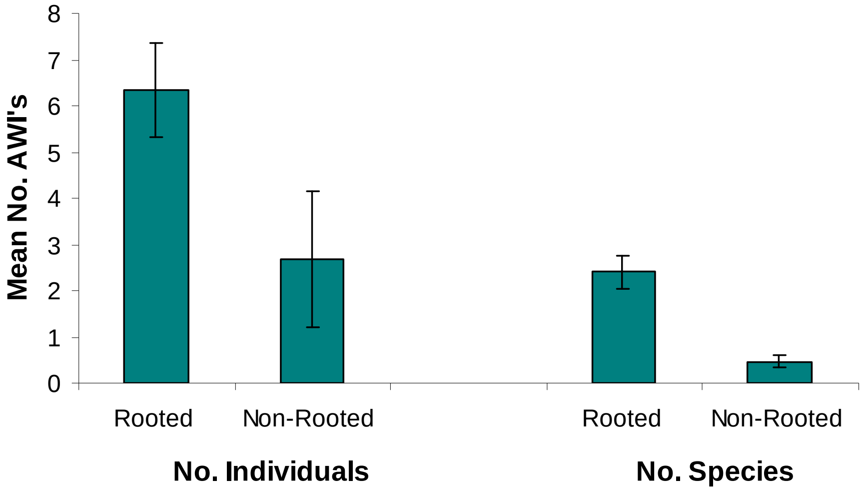


**Fig 4.7** Effects of rooting on emergence from the viable seed bank across woodland habitats. Data represents number of ancient woodland indicator (AWI) species and individuals emerging as seedlings (during 2004) from soil samples collected 2003. Error bars indicate standard error of the means, n =

**Effects of Rooting and Site on Seedling Emergence from the Seed Bank at the Community Level**

| Source | Woodland | | | | Grassland | | | |
|---|---|---|---|---|---|---|---|---|
| | Test | Transform | F value & df | P | Test | Transform | F value & df | P |
| ***Total Number Individuals*** | 2-way ANOVA | $Log_{10}$ (X+1) | | | 2-way ANOVA | None | | |
| Site | | | $F_{4, 24} = 0.56$ | 0.694 | | | $F_{3, 25} = 2.31$ | 0.101 |
| Rooting | | | $F_{1, 24} = 3.17$ | 0.088 | | | $F_{1, 25} = 21.57$ | **< 0.001** |
| ***Species Richness*** | 2-way ANOVA | None | | | 2-way ANOVA | None | | |
| Site | | | $F_{4, 24} = 1.04$ | 0.409 | | | $F_{3, 25} = 1.25$ | 0.313 |
| Rooting | | | $F_{1, 24} = 34.63$ | **< 0.001** | | | $F_{1, 25} = 37.40$ | **< 0.001** |
| ***Shannon Diversity*** | 2-way ANOVA | None | | | 2-way ANOVA | None | | |
| Site | | | $F_{4, 24} = 0.31$ | 0.869 | | | $F_{3, 25} = 2.23$ | 0.110 |
| Rooting | | | $F_{1, 24} = 12.85$ | **0.001** | | | $F_{1, 25} = 22.19$ | **< 0.001** |
| ***Shannon Evenness*** | 2-way ANOVA | None | | | 2-way ANOVA | None | | |
| Site | | | $F_{4, 24} = 0.36$ | 0.837 | | | $F_{3, 25} = 1.59$ | 0.216 |
| Rooting | | | $F_{1, 24} = 6.68$ | **0.016** | | | $F_{1, 25} = 0.39$ | 0.540 |
| ***Ancient Wood. Indicator.*** *(No. Individuals)* | Scheirer Ray Hare | Square root | | | | | | |
| Site | | | $F_{4, 24} = 0.70$ | 0.600 | | | | |
| Rooting | | | $F_{1, 24} = 18.42$ | **< 0.001** | | | | |
| ***Ancient Wood. Indicator.*** *(No. Species)* | Scheirer Ray Hare | Square root | | | | | | |
| Site | | | $F_{4, 24} = 1.52$ | 0.228 | | | | |
| Rooting | | | $F_{1, 24} = 29.36$ | **< 0.001** | | | | |

**Table 4.1**. Two-way ANOVA or Scheirer Ray Hare test results at significance levels; Non-Significant: $p > 0.05$, Significant: $\mathbf{p < 0.05}$, $\mathbf{p < 0.01}$, Highly Significant: $\mathbf{p < 0.001}$. Data were analysed using sequential sum of squares for tests. Significant **P** values shown in bold. **F** = test statistic. **P** = probability associated with F-test. Site = random factor. Rooting = fixed factor. Transform = method of normalising data.

### *4.4.2 Number of individuals across functional groups*

A significant rooting effect for perennial graminoids in woodland (**Table 4.2**) shows that the number of emerged perennial graminoid individuals was greater in non-rooted than rooted treatments from the woodland, but not the grassland seed bank (**Fig 4.4 a-b**). Conversely, a highly significant rooting effect for perennial forbs in woodland shows that the number of emerged perennial forb individuals was greater in rooted than non-rooted treatments from woodland, but not the grassland seed bank. Non-significant site effects for both perennial graminoids and forbs show that site differences do not affect the number of individuals of this functional group emerging from woodland and grassland seed banks.

Although there was a significantly greater number of individual annual forbs that emerged from rooted than non-rooted treatments from the woodland and grassland seed banks, emergence was much greater from grassland (**Table 4.2**, **Fig 4.4 a-b**). Additionally, there was a significant site effect in grassland showing that site differences strongly influenced the emergence of annual forbs in this habitat. The number of biennial forb seedlings was significantly greater in rooted than non-rooted treatments in the woodland seed bank but such data from grassland were insufficient for analysis. A highly significant site effect reveals that site differences greatly affected the numbers of biennial forbs that emerged from the woodland seed bank (**Table 4.2**, **Fig 4.4 a-b**). There was no significant difference between treatments for numbers of woody seedlings emerging from the woodland and grassland seed bank. However, a significant site effect in woodland shows that site differences strongly affected the emergence of woody species in this habitat type.

**Effects of Rooting and Site on Number of Individuals Emerging from the Seed Bank at the Functional Group Level**

| Source | Woodland |  |  |  | Grassland |  |  |  |
|---|---|---|---|---|---|---|---|---|
|  | Test | Transform | F value & df | P | Test | Transform | F value & df | P |
| ***Perennial Graminoids*** | 2-way ANOVA | $\text{Log}_{10}$ (X+1) |  |  | 2-way ANOVA | Square root |  |  |
| Site |  |  | $F_{4,24} = 1.36$ | 0.278 |  |  | $F_{3,25} = 1.93$ | 0.151 |
| Rooting |  |  | $F_{1,24} = 4.84$ | **0.038** |  |  | $F_{1,25} = 0.14$ | 0.715 |
| ***Perennial Forbs*** | 2-way ANOVA | Square root |  |  | 2-way ANOVA | None |  |  |
| Site |  |  | $F_{4,24} = 2.71$ | 0.054 |  |  | $F_{3,25} = 1.12$ | 0.358 |
| Rooting |  |  | $F_{1,24} = 11.44$ | **0.002** |  |  | $F_{1,25} = 1.51$ | 0.231 |
| ***Annual Forbs*** | Scheirer Ray Hare | Square root |  |  | 2-way ANOVA | $\text{Log}_{10}$ (X+1) |  |  |
| Site |  |  | $F_{4,24} = 1.17$ | 0.452 |  |  | $F_{3,25} = 4.92$ | **0.008** |
| Rooting |  |  | $F_{1,24} = 8.32$ | **0.011** |  |  | $F_{1,25} = 174.24$ | **< 0.001** |
| ***D. purpurea & S. jacobaea*** | Scheirer Ray Hare | None |  |  |  |  |  |  |
| Site |  |  | $F_{4,24} = 9.00$ | **< 0.001** |  |  |  |  |
| Rooting |  |  | $F_{1,24} = 29.60$ | **0.001** |  |  |  |  |
| ***B. pendula*** | 2-way ANOVA | Square root |  |  | 2-way ANOVA | None |  |  |
| Site |  |  | $F_{4,24} = 4.63$ | **0.007** |  |  | $F_{3,25} = 0.31$ | 0.821 |
| Rooting |  |  | $F_{1,24} = 1.37$ | 0.254 |  |  | $F_{1,25} = 0.59$ | 0.451 |

**Table 4.2.** Two-way ANOVA or Scheirer Ray Hare test results at significance levels; Non-Significant: $p > 0.05$, Significant: $\mathbf{p < 0.05}$, $\mathbf{p < 0.01}$, Highly Significant: $\mathbf{p < 0.001}$. Data were analysed using sequential sum of squares for tests. Significant **P** values shown in bold. **F** = test statistic. **P** = probability associated with F-test. Site = random factor. Rooting = fixed factor. Transform = method of normalising data.

### 4.4.3 *Number of species across functional groups*

The number of emerging perennial graminoid species from the woodland seed bank was significantly greater in rooted than non-rooted treatments, but not from the grassland seed bank (**Table 4.3**, **Fig 4.5 a-b**). Significant rooting effects reveal that the number of perennial forb species from both the woodland and grassland seed banks were much greater in rooted than non-rooted treatments. Non-significant site effects show that site differences did not affect the emergence of perennial forb species from both habitat types.

The number of emerging annual forb species from the grassland seed bank was significantly greater in rooted than non-rooted treatments, but not from the woodland seed bank (**Table 4.3**). *Betula pendula* was the only woody species to emerge, and *Digitalis purpurea* and *Senecio jacobaea* were the only two biennial forb species to emerge. Rooting had no effect on the number of *B. pendula*, *D. purpurea* and *S. jacobaea* emerging from the woodland and grassland seed bank. However, a significant site effect for the two biennial forb species in woodland shows that differences across sites strongly affects the number of *D. purpurea* and *S. jacobaea* emerging from this habitat type (**Table 4.3**, **Fig 4.5 a-b**).

**Effects of Rooting and Site on Number of Species Emerging from the Seed Bank at the Functional Group Level**

| Source | Woodland | | | | Grassland | | | |
|---|---|---|---|---|---|---|---|---|
| | Test | Transform | F value & df | P | Test | Transform | F value & df | P |
| ***Perennial Graminoids*** | 2-way ANOVA | $Log_{10}$ (X+1) | | | 2-way ANOVA | None | | |
| Site | | | $F_{4, 24} = 1.01$ | 0.054 | | | $F_{3, 25} = 0.81$ | 0.501 |
| Rooting | | | $F_{1, 24} = 57.76$ | **< 0.001** | | | $F_{1, 25} = 0.37$ | 0.548 |
| ***Perennial Forbs*** | 2-way ANOVA | None | | | 2-way ANOVA | None | | |
| Site | | | $F_{4, 24} = 0.98$ | 0.436 | | | $F_{3, 25} = 0.73$ | 0.546 |
| Rooting | | | $F_{1, 24} = 13.78$ | **0.001** | | | $F_{1, 25} = 9.48$ | **0.005** |
| ***Annual Forbs*** | Scheirer Ray Hare | None | | | 2-way ANOVA | None | | |
| Site | | | $F_{4, 24} = 0.51$ | 0.723 | | | $F_{3, 25} = 1.11$ | 0.362 |
| Rooting | | | $F_{1, 24} = 2.44$ | 0.115 | | | $F_{1, 25} = 99.40$ | **< 0.001** |
| ***D. purpurea & S. jacobaea*** | Scheirer Ray Hare | None | | | | | | |
| Site | | | $F_{4, 24} = 6.00$ | **0.008** | | | | |
| Rooting | | | $F_{1, 24} = 2.84$ | 0.203 | | | | |
| ***B. pendula*** | 2-way ANOVA | None | | | 2-way ANOVA | None | | |
| Site | | | $F_{4, 24} = 2.50$ | 0.070 | | | $F_{3, 25} = 0.45$ | 0.718 |
| Rooting | | | $F_{1, 24} = 0.96$ | 0.338 | | | $F_{1, 25} = 0.58$ | 0.452 |

**Table 4.3.** Two-way ANOVA or Scheirer Ray Hare test results; = Non-Significant: $p > 0.05$, Significant: **$p < 0.05$, $p < 0.01$**, Highly Significant: **$p < 0.001$**. Data were analysed using sequential sum of squares for tests. Significant **P** values shown in bold. **F** = test statisitic. **P** = probability associated with F-test. Site = random factor. Rooting = fixed factor. Transform = method of normalising data.

## 4.5 Discussion

### *4.5.1 Community Level*

As predicted, significantly greater numbers of individuals emerged from rooted than non-rooted soil from grassland (**Appendix 6**, **Table 4.1**, **Fig 4.4**), but there was no significant difference between treatments in woodland despite a similar trend. Seed banks within many grassland habitats are typically more persistent and may consequently be larger than in woodland since woodland plant species tend to contribute only a transient seed bank, or no seed bank at all (Brown and Oosterhuis, 1981; Fenner, 1992; Onaindia and Amezaga, 2000; Bossuyt *et al.*, 2002; Diaz-Villa *et al.*, 2003). Two processes may be operating to generate the effects of rooting on grassland seed banks. Soil disturbance created by boar trampling and rooting may increase incorporation of seeds into the seed bank from the seed rain or the soil surface, leading to greater seed bank density. Increased soil $NO_3^-$ concentration in rooted areas (Lacki and Lancia, 1983; Singer *et al.*, 1984; Kotanen, 1994) (**Table 5.1**, **Fig 5.6**) would increase dormancy breaking mechanisms and germination in a potentially greater number of seeds. Greater initial seed bank density coupled with increased soil $NO_3^-$ concentration could therefore increase emergence of numbers of individuals from rooted than non-rooted soil.

Species richness and Shannon diversity were greater in rooted than non-rooted soil from grassland and woodland habitats (**Table 4.1**, **Fig 4.5**, **Fig 4.6**). Although species richness in rooted samples was greater from grassland (mean of eighteen) than woodland (mean of twelve) (**Fig 4.5**), evenness of species abundance was greater from woodland than grassland. A larger overall seed bank size coupled with increased $NO_3^-$ concentration could largely contribute to an increased overall emergence leading to higher species richness and diversity from rooted treatments. However, the homogenised physical conditions produced through sieved soil in the lab, may have contributed to the increased evenness in rooted soil from woodland habitats.

Rooting increased the mean number of AWI species and individuals across woodland habitats (**Table 4.1**, **Fig 4.7**), although the numbers that emerged were relatively small: the mean number of species per tray was three and the mean number of individuals was six (**Fig 4.7**). However, some AWI species (such as *Anemone nemorosa,* which is abundant in my study area) mainly reproduce vegetatively, and thus are absent from the seed bank (**Appendix 1**, **Appendix 5**).

The AWI species that emerged in rooted treatments in this study were *Stellaria neglecta, Primula vulgaris, Lysimachia nemorum, Luzula pilosa, Carex pendula, Poa nemoralis and Milium effusum*, compared to *L. pilosa, C. pendula and P. nemoralis* that emerged from non-rooted treatments (**Appendix 1** and **6**). Many AWI species have a transient seed bank (Bossuyt *et al.*, 2002), therefore their seeds tend to be relatively short lived; only existing in the seed bank if the species have established seed forming populations in the above ground vegetation (Rackham, 1980; Hermy *et al.*, 1999; Bossuyt *et al.*, 2002). This is demonstrated by *S. neglecta* and *M. effusum*, which were the only two species emerging from seed that were untypical of the above ground vegetation in the study area (**Appendix 1**). However, eleven species existed in the above ground vegetation (**Appendix 1**) that did not emerge from the woodland seed bank (**Appendix 6**), although five were woody species (**Appendix 1** and **3**); one of the least abundant functional groups emerging from the seed bank (**Fig 4.5 a-b**) probably due to low representation in the soil samples.

Unlike AWI species, seed banks typically contain plant species that are absent from the above ground vegetation (Thompson and Grime, 1979; Hill and Stevens, 1981; Brown and Oosterhuist, 1981; Lunt, 1997; Jutila and Heli, 1998; Onaindia and Amezaga, 2000; Bossuyt *et al.*, 2002; Miller and Cummins, 2003). For example, *S. neglecta, Cerastium arvense, Carex sylvatica, Poa annua,* (from rooted and non-rooted treatments) and *Sagina apetala, Chenopodium rubrum, Chenopodium botryodes, Coronopus squamatus, Luzula campestris, Dactylis glomerata, Deschampsia caepitosa* and *M. effusum* (from rooted treatments) emerged from the seed bank from woodland and grassland, but were absent from the above ground vegetation in the study area (**Chapter 2**, **Appendices 1-6**). This illustrates the potential for the seed bank to contribute to the maintenance of species richness and diversity of plant communities following rooting. Intermediate levels of rooting (rooted once or twice annually) could therefore have conservation implications relating to restoration of plant communities via the seed bank. For example, emergence from the seed bank as a source of plant re-establishment following rooting, is probably important in many studies that found rooting enhanced plant cover or species richness (Bowman and McDonough, 1991; Kotanen, 1994, 1995; Welander, 1995; Onipchenko and Golikov, 1996; Milton *et al.*, 1997; Arrington *et al.*, 1999; Welander, 2000a).

In later successional habitats, especially where there are low levels of disturbance, there are typically substantial differences between the composition of the viable seed bank and the above ground vegetation. Furthermore, these differences tend to increase with successional age (Thompson and Grime, 1979; Fenner, 1992; Loony and Gibson, 1995; Bakker *et al.*, 1996; Bakker and Bekker, 1997; Bekker *et al.*, 1997; Baskin and Baskin, 1998). However, in some cases the composition of the viable seed bank is represented more highly in the growing vegetation. For example, Granstrom (1982) found that most seedlings that emerged from five boreal forest stands in northern Sweden, aged 16-169 years, belonged to plant species present in the growing vegetation. It is suggested that this is most likely to occur when plant species have effective dispersal means and persistent seed banks, coupled with high levels of disturbance in the community (Thompson and Grime, 1979; Loony and Gibson, 1995; Granstrom, 1982). Rooting that occurs in later successional habitats (such as in my study sites, **Table 1.1**), through over-turning horizons and exposing persistent seeds could enhance the number of species both absent and present in the growing vegetation, as results here demonstrate (**Appendix 2**, **Appendix 3**, **Appendix 6**). Several studies on the regeneration of plants after rooting indicate that rooting aids the re-colonisation of species that were previously present (Challies, 1975; Baron, 1982; Kotanen, 1994; Groot Bruinderink and Hazebroek, 1996; Milton *et al.*, 1997), although no studies specify the source of re-generation.

Disturbance of soil is known to enhance germination (and therefore seedling emergence) by breaking seed dormancy mechanisms in many plant species through altered physical conditions (Sauer and Struik, 1964; Moore and Wein, 1977; Putz, 1983; Buckley *et al.*, 1997; Pugnaire and Lozano, 1997; Rydgren *et al.*, 1998; Hyatt and Casper, 2000; Jutila and Grace, 2002). Since one cannot consider the physical modifications that rooting imposes on soil in this study (since seedlings all emerged from sieved soil in the lab), enhanced $NO_3^-$ concentration, scarification inducement and a larger seed bank in rooted soil prior to sample collection, are the most likely factors acting to increase germination and emergence from rooted treatments (Lacki and Lancia, 1983; Singer *et al.*, 1984; Kotanen, 1994) (**Table 5.1**, **Fig 5.6**).

### *4.5.2 Functional group level*

The numbers of individuals and species of annual forbs was significantly greater in rooted than non-rooted treatments from grassland, and the number of individuals was significantly greater in rooted treatments from woodland (**Table 4.2**, **4.3**, **Fig 4.4**, **4.5**). Recorded as numbers of emerging individuals, annual forbs were the most predominant functional group from rooted soil from grassland (**Fig 4.4**). Welander (1995) also found annuals were the predominant functional group that re-established after intense rooting across several habitat types in Sweden. Under lab conditions in this study however, a combination of increased $NO_3^-$ and a larger ruderal seed bank potentially explain this result.

The number of individuals of annual forbs emerging from the grassland seed bank was also significantly affected by site differences. The different types of grassland and forms of management imposed on the sites (**Table 1.1**) coupled with dispersal from neighbouring plant communities would be likely to influence the relative abundance of functional groups contained in the seed bank. Different disturbance regimes imposed by the different forms of management across sites could differentially affect the soils physical, biological and chemical environment, and hence the emergence across sites within functional groups is likely to vary and become reflected in the seed bank size of each functional group. Considering the disturbance-tolerant nature of annual forbs (Grime, 1979; Grime *et al.*, 1989; Crawley, 1997), it is not surprising that the different levels of disturbance across sites would significantly affect emergence and hence the size of the annual forb seed bank. Furthermore, seeds of annual forbs, characterised by a ruderal strategy (Grime, 1979; Grime *et al.*, 1989), tend to be wind dispersed, indicating that levels of dispersal from the seed rain from neighbouring plant communities was an important contributor towards differences in the size of the annual forb seed bank across sites. In the lab, differences in the size of the annual forb seed bank could have contributed to significant differences in emergences across sites.

Continuity of boar presence could, through rooting, increase both incorporation into and emergence from ruderal seed banks in semi-natural British habitats, which could contribute to community diversity (Thompson and Grime, 1979; Lunt, 1997; Diaz-Villa *et al.*, 2003). Sixteen annual forb species in total emerged from the seed bank. Five were exclusive to the seed bank (*S. neglecta, S. apetala, C. rubrum, C. botryodes and C. squamatus*) and eleven were additionally present in the above ground vegetation (*Stellaria media, L. nemorum, Veronica arvensis, Veronica persica, Pedicularis sylvatica, Stachys arvensis, Gnaphalium uliginosum, Centaurium erythraea, Centaurium pulchellum, Chenopodium polyspermum, Juncus bufonius*) (**Appendix 1** and **6**).

The number of individuals of *D. purpurea* and *S. jacobaea* was significantly greater in rooted treatments from woodland habitats, although contrary to predictions there was no significant difference between treatments in the number of species. The relatively large number of biennial forb individuals (**Fig 4.4 a-b**) represented by only two biennial forb species; *D. purpurea* and *S. jacobaea* (**Fig 4.5 a-b**) from woodland and grassland, explains this result (**Appendix 1** and **6**). Site differences strongly affected the numbers of individuals and species of biennial forbs emerging from the woodland and grassland seed bank. Corresponding with annual forbs, large potential differences in the soil environment across sites (largely determined by differential disturbance regimes (**Table 1.1**) could lead to a large variation in emergence and thus seed bank size of biennial forbs. In the lab, potential differences in the size of the biennial forb seed bank could have contributed to significant differences in emergences across sites.

*Betula pendula* was the only woody species that emerged (**Appendix 1** and **6**) indicating the probability that few seeds of woody species were present in soil samples in both rooted and non-rooted treatments in woodland and grassland. This may be due to low seed production or high predation rates during 2003. However, it is possible that more seeds than emerged were present, but the highly disturbed environment of the sieved soil under lab conditions suppressed germination due to the high disturbance susceptibility of this functional group. Site differences also influenced the emergence of woody individuals from the woodland seed bank. Potential differences in both predation rates and levels of emergence across sites (related to different levels of disturbance) is likely to affect the number of woody seeds contained in soil samples used in this study and hence the number of individuals that emerged in the lab.

Surprisingly, rooting led to an increased emergence of numbers of individuals and species of perennial graminoids in woodland, but as hypothesised, not from grassland (**Table 4.2**, **4.3**, **Fig 4.4**, **4.5**). This was interesting since the high numbers of perennial graminoids emerging from the woodland seed bank (**Fig 4.2 a-c**) were not represented in the above ground vegetation. More perennial graminoids emerged exclusively from the woodland seed bank than any other functional group; they were *L. campestris*, *D. glomerata*, *D. caepitosa, P. annua* (typical of grassland), and *M. effusum and C. sylvatica* (typical of woodland and grassland) (Phillips, 1980) (**Appendix 1** and **6**). These species were not present in the above ground plant communities within any treatment in grassland, woodland or woodland rides, discussed in **chapter 2**. This illustrates the differences that can exist between the seed bank and above ground plant communities (Thompson and Grime, 1979; Hill and Stevens, 1981; Brown and Oosterhuist, 1981; Lunt, 1997; Jutila and Heli, 1998; Onaindia and Amezaga, 2000; Bossuyt *et al.*, 2002; Miller and Cummins, 2003) and the potential importance of seed banks for plant diversity in disturbed habitats. Since many perennial graminoids produce wind-dispersed seeds (such as those that emerged from the woodland seed bank) (Phillips, 1980), the close proximity of grassland to woodland sites in the study area (**Table 1.1**) could potentially increase inter-habitat dispersal of these species, which could then enter the persistent seed bank until conditions favour germination. The two actively managed coppiced sites in the study area (**Table 1.1**) cyclically exist with no canopy, thus they are intermittently an open habitat. The removal of canopy, coupled with the close proximity of grassland, may lead to an increase of immigration of seeds from grassland into open woodland. There, the dispersed seeds are probably more easily incorporated into the seed bank of rooted than non-rooted soil due the mechanical disturbance imposed by boar trampling whilst rooting, thus increasing the density of graminoids in seed banks in rooted areas.

Rooting clearly has a large impact on short-term seed bank dynamics at the community and functional group level from woodland and grassland habitats. It seems probable that through disturbing the soil's physical environment, rooting significantly increased the seed bank density and other soil properties (such as $NO_3^-$ levels) prior to collection, such that emergence was greater from rooted than non-rooted treatments. Results clearly show greater overall numbers of individuals from rooted grassland sites and greater species richness and diversity from rooted woodland and grassland sites emerging from the seed bank. Amongst functional groups, results indicate that seeds of ruderal strategists (some perennial, biennial, and particularly annual forbs) are best adapted for short-term post rooting colonisation, such that emergence of ruderals was greater from rooted than non-rooted soil.

Surprisingly, site differences had a much smaller effect than predicted, having no impact on emergence from the seed bank at the community level, and a relatively small impact on emergence at the functional group level. The ruderal-type strategists (annual and biennial forbs) were the most affected by site, possibly mainly due to differences in disturbance levels across sites causing a large variation in emergence and hence seed bank size of these functional groups. Neighbouring plant communities to the different study sites could also have contributed to compositional variation of the seed bank due to differential levels of dispersal, particularly the seed rain, since ruderals seeds tend to be wind-dispersed (Grime, 1979; Grime *et al.*, 1989).

The seed bank in the field is likely to be affected differentially by various rooting intensities. Following my results, emergence within intermediately disturbed areas (such as annual rooting) would result in plant assemblages dominated by ruderal strategists, with overall increased species richness, abundance and diversity. In less frequently rooted areas (rooted biennially) $NO_3^-$ would probably begin to decrease, returning to levels similar to that which existed prior to rooting following greater plant uptake by increased above ground biomass. The seed bank turnover would probably also decrease, returning to a similar size to that which existed prior to rooting following the decline in emergence as above ground biomass increases, and possibly reduced incorporation with decreased trampling in the absence of rooting. Emergence of functional groups would be likely to display a shift in dominance toward more competitive and stress tolerant strategists such as many perennial forbs, woody species and perennial graminoids (**Appendix 1**) following the recovery of vegetation and subsequent physical and chemical modifications in the soil environment. Species richness, abundance and diversity however are likely to remain relatively high within this temporal scale, before competitive dominance predominates as resources become depleted and biomass increases. Frequent rooting (rooted more than twice annually, and over successive years) may lead to the seed bank becoming depleted due to increased emergence after each rooting event coupled with inadequate recovery periods, which result in decreased seed production.

# Chapter 5

## Impacts of wild boar rooting on belowground community attributes and processes

### 5.1 Introduction

**S**oil properties and processes are fundamental to the structure and functioning of the terrestrial community. Soil contains a variety of substrates including a complex array of inorganic nutrients and organic matter, which are intrinsic components of net primary productivity (NPP) (Hunter, 1999). The process of decomposition; an integral component of NPP, may involve leaching from detritus by water flow (Mason, 1977), fragmentation of detritus via invertebrate detritivores, and mineralization of organic particles into inorganic, mineral nutrients available for immobilization (Chapman and Reiss, 1999; Smith and Smith, 2001).

Disturbance by fossorial mammals is known to increase soil mineral nitrogen (ammonia ($NH_3$), ammonium nitrogen ($NH_4^+$), nitrite nitrogen ($NO_2^-$) and nitrate nitrogen ($NO_3^-$)), which may accumulate in the soil due to the lack of plant $NO_3^-$ uptake where live plants are removed (Canals *et al.*, 2003). Soil heterotrophic micro-organisms utilize carbon released from root exudates to grow and assimilate nitrogen from the soil (van Veen *et al.*, 1989). The removal of live plants by fossorial mammals may reduce root enhanced microbial immobilization of $NO_3^-$ and thus lead to increased soil mineral nitrogen (Canals *et al.*, 2003). Fossorial mammal digging disrupts the physical structure of the soil, altering properties such as temperature, aeration (including oxygen levels) and moisture (Huntly and Inouye, 1988; Huntly and Reichman, 1994). Such physical changes in soil can affect biological and chemical processes such as decomposition, inorganic nitrogen production and nitrogen cycling (Meentemeyer, 1978; Kochy and Wilson, 1997; Cortez, 1998; Guo and Sims, 1999). Litter decomposition is the principal means by which nutrients enter forest soils (Aber and Melillo, 1979). Since nitrogen availability is believed to be one of the main factors controlling NPP (Burke *et al.*, 1997), and available nitrogen for plants is governed by its mineralization, biopedturbations by fossorial mammals could affect decomposition, nitrogen cycling and ultimately NPP which governs plant growth and ecosystem dynamics.

Wild boar inflict severe soil disturbance through rooting. This ‘rototiller-like’ foraging activity (Ray, 1988; Moody and Jones, 2000) is a form of soil disturbance that fluctuates in area, depth and intensity. Consequently, rooting alters the structural complexity of the soil surface as it exposes a variety of substrates, such as humus, mineral soil, belowground plant biomass, rocks and stones in a patchy manner (Milton *et al.*, 1997; Welander, 2000a, 2000b). Rooting mixes soil horizons, over-turns leaf litter (reducing surface build up through incorporation into soil) and removes or redistributes vegetation in a patchy manner (Spatz and Mueller-Dombois, 1975; Bratton, 1975; Singer *et al.*, 1984). Therefore, the amount of dead organic plant matter such as leaf litter, uprooted plants and fragmented dead belowground plant parts, is likely to increase greatly within soil in rooted patches. Rooting is known to modify biological, physical and chemical properties of soil including populations of soil microarthropods (Vtorov, 1993; Mohr *et al.*, 2005), aeration and oxygen levels, drainage, temperature, crumb structure, the seed bank (Brownlow, 1994;

Kotanen, 1994) and nutrient cycling (Lacki and Lancia, 1983; Singer *et al.*, 1984; Mohr *et al.*, 2005). Increased levels of dead organic matter in soil, coupled with an altered physical and chemical soil environment could accelerate decomposition of organic matter and thus alter nitrogen transformation processes.

There is little experimental data on the effects of rooting on soil properties and processes (Singer *et al.*, 1984) and that which exists is largely contradictory. For example, Moody and Jones (2000) found no effect of rooting on $NO_3^-$ and total inorganic nitrogen in a *Quercus agrifolia* forest on Santa Cruz Island, California. Groot Bruinderink and Hazebroek (1996) found no effect of rooting on mineral nitrogen levels in soil, decomposition rate, and levels of organic matter in deciduous and coniferous forest in the Veluwe, Netherlands. Mohr *et al.* (2005) found no effect of rooting on soil nitrogen levels in sloping oak forests in Germany. Singer *et al.* (1984) however found significantly increased $NO_3^-$ levels in rooted than non-rooted soil within the Great Smoky Mountains National Park, which was attributed to enhanced decomposition in rooted areas. Lacki and Lancia (1983) detected enhanced litter decomposition and soil nutrient mobilization with increased length of disturbance by feral pigs in beech gaps of Great Smoky Mountains National Park. Kotanen (1994) found an increase in mineral nitrogen in rooted than non-rooted areas in Californian grasslands. Lastly, Falinski (1986) frequently observed a speeding up of the soil mineralisation process in the accumulation-humus layer in rooted environments within the Bialowieza Forest. Poland. Such discrepancies across different research studies may reflect differences in boar population densities; soil processes may respond differentially to varying rooting intensities. In addition, variations in climate, geology, ecosystem and community structure (relating to physical, biological and chemical variation in soil) could lead to different responses of soil properties and processes to rooting.

The aim of this study was to investigate the effects of rooting on belowground properties and processes, specifically belowground live plant biomass, decomposition of leaf litter and inorganic nitrogen available to plants as $NO_3^-$ and $NH_4^+$.

### *5.1.1 Hypotheses*

I hypothesised that rooting would significantly reduce belowground live plant biomass in both woodland and grassland habitats. Since boar root primarily to locate and consume belowground plant parts such as roots, bulbs, rhizomes, tubers and corms (Wood and Roark, 1980; Focardi *et al.*, 2000), it is likely that rooted patches would contain less living plant biomass than non-rooted patches. I hypothesised that site differences in woodland and grassland habitats would significantly contribute to differences in belowground live plant biomass. The Sites across both habitat types used in this study (**W1** and **W5**, **G2** and **G3**) are subject to very different forms of management (**Table 1.1**), which is reflected by the subsequent variation in the species complement (**Table 1.1**) and overall abundance of the above (**Fig 2.2 – 2.11**) and therefore predicted belowground portions of the plant assemblage present.

I hypothesised that rooting would significantly increase leaf decomposition rate in both woodland and grassland habitats. Rooting increases soil aeration and temperature (Kotanen, 1994); these physical conditions are known to increase organic breakdown of leaf litter (Stanford and Smith, 1972; Meentemeyer, 1978; Kochy and Wilson, 1997; Cortez, 1998; Chapman and Reiss, 1999). Rooting also breaks up soil and mixes horizons, which incorporates higher levels of organic matter into the upper soil layers (Lacki and Lancia, 1983; Singer *et al.*, 1984; Bialy, 1996). Where there are high levels of soil organic matter coupled with a modified physical environment, decomposition is likely to be increased. Further, I hypothesised that the site differences across woodland and grassland would significantly affect decomposition rate. The different plant assemblage across the sites used in this study (largely determined by different management regimes) (**Table 1.1**, **Fig 2.2 – 2.11**), could contribute to a more diverse soil environment, altering biological, chemical and physical properties of the soil and thus lead to altered decomposition rates across sites. Management differences across the sites used in this study (**W1** and **W2**, **G1** and **G2**) are particularly prevalent in grassland (**Table 1.1**), therefore I predicted that site effects would be stronger in grassland than woodland.

I hypothesised that rooting would significantly increase the levels of soil $NO_3^-$ and $NH_4^+$ in woodland and grassland habitats. The presence of $NO_3^-$ in soil is controlled by fragmentation, organic nitrogen mineralization and inorganic nitrogen immobilization. If decomposition rate is increased in rooted patches (Lacki and Lancia, 1983) it is probable there will be an increase in nitrogen mineralization resulting in the formation of $NH_3$, $NO_2^-$ and ultimately $NH_4^+$ and $NO_3^-$. Plants absorb nitrogen in the form of $NO_3^-$ (Salisbury and Ross, 1992; Mix *et al.*, 1992); once in the plants' tissue, $NO_3^-$ is converted to $NH_4^+$, which is then incorporated into new tissue cells. Plants are also able to assimilate $NH_4^+$ directly from soil (Salisbury and Ross, 1992; Taiz and Zeiger, 1998). The removal of plants through rooting would eliminate plant uptake of $NO_3^-$ and $NH_4^+$ and reduce root enhanced microbial immobilization of $NO_3^-$ (Canals *et al.*, 2003; Booth *et al.*, 2005), which could lead to higher levels of $NO_3^-$ and $NH_4^+$ in rooted soil. I hypothesised that site differences (largely determined by inconsistent management regimes) would significantly affect the levels of soil $NO_3^-$ and $NH_4^+$ in woodland and grassland. For example in woodland, active coppicing in **W1** and **W2** removes wood through harvest regimes, which can increase nutrient losses such as nitrogen (Hunter, 1999). In comparison, **W3** and **W5** have less management-related disturbance (**Table 1.1**) and may suffer less from nutrient losses. Further, differences in the plant assemblage across all sites in terms of species and abundance (**Fig 2.2 – 2.11**), is likely to result in different levels of plant $NO_3^-$ and $NH_4^+$ uptake and thus largely dictate their concentration in soil.

## 5.2 Materials and Methods

### *5.2.1 Study Area*

This study was undertaken within four out of the five woodland sites and the four grassland sites within the study area (**Fig 1.5 a-d**). Sites utilised in this study were woodland sites **W1**, **W2**, **W3** and **W5** (utilised in **Chapters 2-4**), and grassland sites **G1**-**G4** (utilised in **Chapters 2** and **4**), (**Table 1.1**).

### *5.2.2 Experimental Design*

#### *5.2.2.1 Belowground live plant biomass*

Four sites were used within this study, two woodland sites (**W1** and **W5)** and two grassland sites (**G2** and **G3**) (**Table 1.1**). Forty soil cores were randomly collected equally between rooted and non-rooted treatments over the four sites (five replicates per treatment per site) using a 7cm diameter (38.5cm$^2$) x 10cm depth (385cm$^3$) soil auger (**Fig 5.1**) during early October 2003. Rooted treatments were recently rooted patches (< 3 months) and the depth at which each soil core was collected was within the typical rooting zone (5-15cm). Live plant biomass (predominantly roots, bulbs and rhizomes) were separated from dead plant material (predominantly humus and dead wood) and soil. The contents of the soil cores were separated into the categories using a fine-meshed sieve and by running water over the samples at low pressure. Soil and dead organic matter were discarded and the live plant material was dried within absorbent paper bags at approximately 45ºC in an oven for two days and for a further five days at room temperature with drying crystals for the absorption of any remaining moisture. Once dried, the plant material was weighed. Weights across treatments and habitats were analysed.

**Fig 5.1:** 7cm diameter x 10cm depth soil auger (on left) used for collection of soil core samples for the investigation into the effects of rooting on soil nitrogen levels and belowground live plant biomass.

#### 5.2.2.2 *Decomposition rate of leaf litter*

Leaves from different tree species decompose at various rates depending on the relative amounts of lignin within their structure (Aber and Melillo, 1982; Melillo *et al.*, 1982; Berg and McClaugherty, 1989; Cornelissen, 1996). Leaves relatively high in lignin have slow decomposition rates (Meentemeyer, 1978). Palatable leaves, relatively lower in lignin support higher populations of decomposers and decompose quicker (Smith and Smith, 2001). *C. sativa* leaves were specifically chosen for this experiment because they have a relatively fast decay rate (Mason, 1977; Cortez, 1998; Cortez and Bouche, 1998) and *C. sativa* trees were abundant in woodland sites **W1** and **W2**. It was therefore expected that there would be adequate densities of suitable decomposer organisms in the soil (Olofsson and Oksanen, 2002).

*C. sativa* leaves that had senesced and fallen from the trees were collected during November and December 2002 from woodland sites **W1** and **W2** (**Table 1.1**) within the study area. All leaves at the time of collection were intact, non-photosynthetic, had brown colouration and no visible signs of decomposition. Collected *C. sativa* leaves were carefully washed with cold water and air dried at room temperature for four days before being sterilised in an oven at 80ºC for three hours. The leaves remained in the oven for a further three days to cool with drying crystals for absorption of any remaining moisture. Once dried, *C. sativa* leaves were weighed into 220 batches each weighing approximately 4g. Each 4g batch was placed into a nylon 15cm x 20cm litterbag with a 2.5mm x 2.5mm mesh size (**Fig 5.2a**) and secured together by sewing with nylon fishing thread and labelled with two nylon-sewn laminated reference number tags. Lousier and Parkinson (1976), Melillo *et al.* (1982), Berg and McClaugherty (1989), Cornelissen (1996) and Middleton and McKee (2001) used similar techniques (utilising litterbags with varying area and mesh sizes) for investigations involving litter decay rates. The mesh size that was used in this study was

**Fig 5.2 a-b: a)** A nylon 15cm x 20cm litterbag with a 2.5mm x 2.5mm mesh size used for the experiment on decomposition rate. **b)** One of the litterbags just dug up in **G1** to examine decomposition of *C. sativa* leaves. Photo taken April 2003.

fine enough for the retention of small leaf particles and moisture whilst allowing the passage of all appropriate microflora, microfauna, mesofauna, but exclusion of some macrofauna (Lousier and Parkinson, 1976; Cornelissen, 1996; Melillo *et al.*, 1982).

Two woodland sites (**W1** and **W2**) and two grassland sites (**G1** and **G2**) (**Table 1.1**) were used for this experiment (**Fig 5.3**). Litterbags were randomly assigned locations (at a minimum distance apart of 3m) divided equally between recently rooted (< 3 months) and non-rooted treatments and unequally across the four sites (27 replicates in **G1** and **W1**, 28 replicates in **G2** and **W2**) and marked with fluorescent painted wooden stakes (**Fig 5.3**). Litterbags were buried (fifty five per treatment per habitat type) between 10cm–15cm (within the typical rooting zone) during February 2003. Ten weeks after burial a small random sample of litterbags were briefly dug up to examine how far decomposition had progressed (**Fig 5.2b**). In July 2003, all the litterbags were dug up and the remaining *C. sativa* leaves were removed and carefully washed in cold water to remove soil traces over a fine-meshed sieve to collect all fragments of leaf litter. The partially decomposed leaves were air-dried at room temperature for four days before being completely dried and sterilised within absorbent paper bags in an oven at 80°C for three hours. The leaves were left to dry and cool in the oven for a further three days with drying crystals. The dry leaf remains were weighed; weight differences across treatments and habitats were analysed. Dry weight loss over time represents the result of decomposition.

**a)**

**b)**

**Fig 5.3 a-b:** Two habitats containing fluorescent painted wooden stakes used for marking the position of buried litterbags for the investigation into the effects of rooting on decomposition rate of leaf litter. **a)** Sweet chestnut coppice site **W1** **b)** Semi-wild grassland site **G2** (**Table 1.1**). Photos taken February 2003.

### *5.2.2.3 Soil nitrogen levels*

Four woodland sites (**W1**, **W2**, **W3**, and **W5**) and the four grassland sites (**G1**-**G4**) were used in this study (**Table 1.1**). Eighty soil cores were randomly collected equally between rooted and non-rooted treatments over the eight sites (five replicates per treatment per site) using the 7cm diameter x 10cm depth soil auger during early October 2003. Rooted treatments were recently rooted patches (< 3 months) and the depth at which each soil core was collected was within the typical rooting zone (5-15cm). The soil cores were sent to the Natural Resource Management Ltd. in Bracknell, Berkshire for analysis of total nitrogen available to plants ($NO_3^-$ and $NH_4^+$).

## 5.3 Statistical Analysis

Data were analysed with two-way ANOVA where data were normally distributed (Eddison, 2000; Dytham, 2003). Data were log (Log10 x +1) or square root transformed to achieve normality where possible (specified **Table 5.1**). Where data were not normally distributed and could not be transformed, the Scheirer Ray Hare test was used instead (Eddison, 2000; Dytham, 2003). In both cases, rooting was a fixed factor and site was the second and random factor. Data were analysed independently for two of the habitat types and using the sequential sum of squares. For the analysis of decomposition rate, an unbalanced design was used due to there being an unequal distribution of replicates between sites. All other data were analysed incorporating a balanced design.

## 5.4 Results

### *5.4.1 Belowground live plant biomass*

Highly significant rooting effects (**Table 5.1**) show that rooting greatly decreased the weight of belowground live plant biomass (measured in grams) in both woodland and grassland habitats (**Fig 5.4**). Non-significant site effects show that site differences across woodland and grassland did not affect belowground live plant biomass.

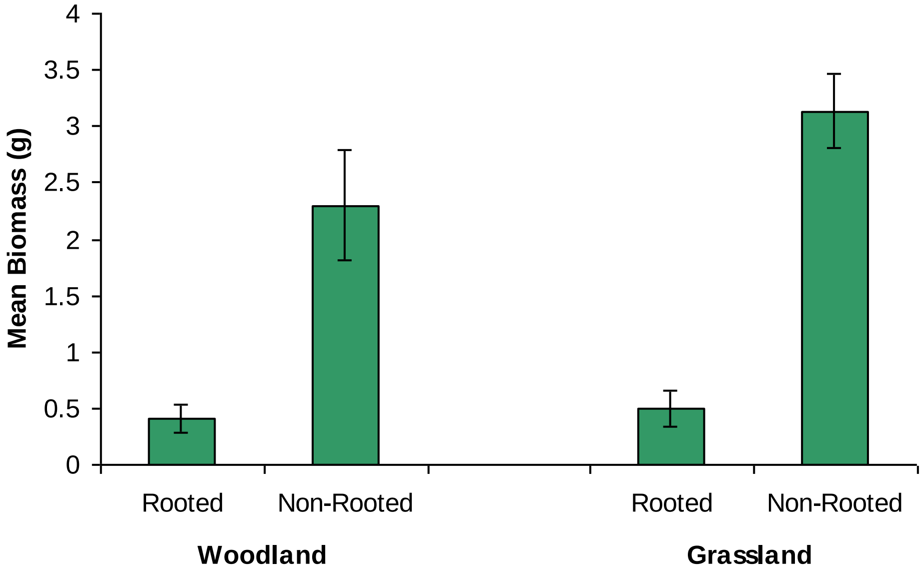


**Fig 5.4:** Effects of rooting on belowground live plant biomass in woodland and grassland during early October 2003. Data from different sites were pooled. Two-way ANOVA results shown in **Table 5.1**. Error bars indicate standard error of the means, n = 10.

### *5.4.2 Decomposition rate of leaf litter*

A highly significant rooting effect in woodland (**Table 5.1**) shows that rooting increased the decomposition rate of *C. sativa* litter in woodland but not in grassland habitats (**Fig 5.5**). A significant site effect in grassland however, shows that site differences influence decomposition rate independently of rooting in grassland but not woodland habitats.

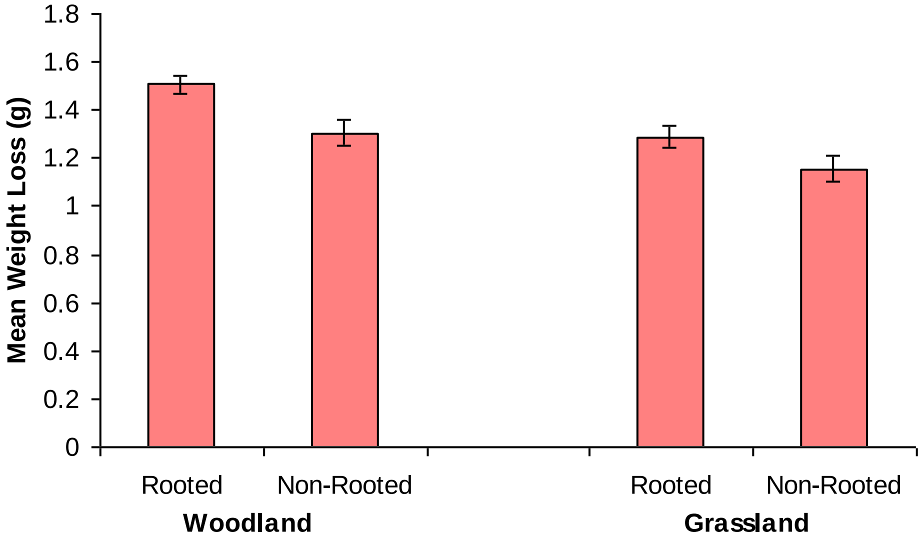


**Fig 5.5:** Effects of rooting on decomposition rate of leaf litter in woodland and grassland, from February to July 2003. Data from different sites were pooled. Two-way ANOVA results shown in **Table 5.1**. Error bars indicate standard error of the means, n = 55.

### *5.4.3 Soil NO3- levels*

Highly significant rooting effects (**Table 5.1**) show that rooting significantly increased the levels of soil $NO_3^-$ in woodland and grassland habitats (**Fig 5.6**). The absence of significant site effects, indicate that rooting greatly increased soil $NO_3^-$ concentration independently of habitat.

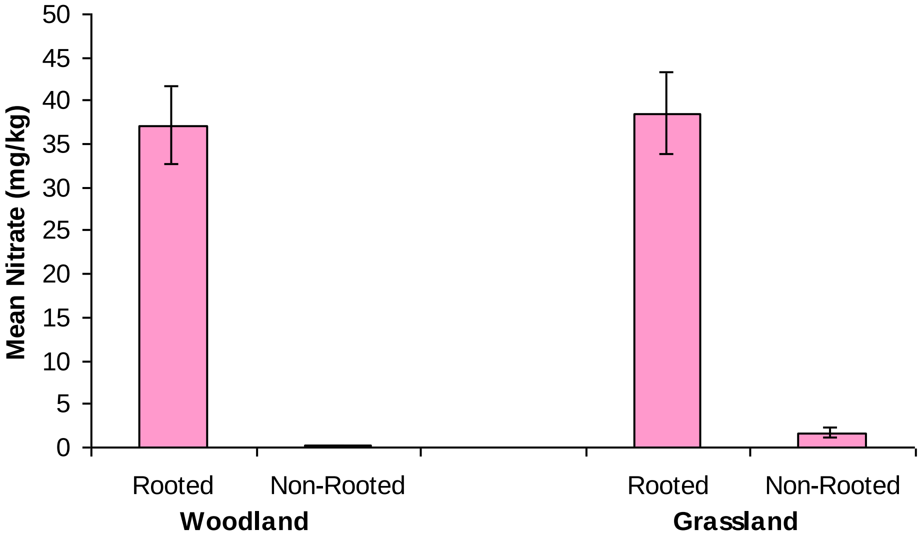


**Fig 5.6:** Effects of rooting on soil nitrate-nitrogen levels in woodland and grassland during early October 2003. Data from different sites were pooled. Scheirer Ray Hare test results shown in **Table 5.1**. Error bars indicate standard error of the means, n = 20.

### *5.4.4 Soil NH4+ levels*

A significant rooting effect in grassland (**Table 5.1**), shows that rooting increased soil $NH_4^+$ levels in grassland but not woodland habitats (**Fig 5.7**). Non-significant site effects in both habitat types show that site differences are not affecting the concentration of soil $NH_4^+$.

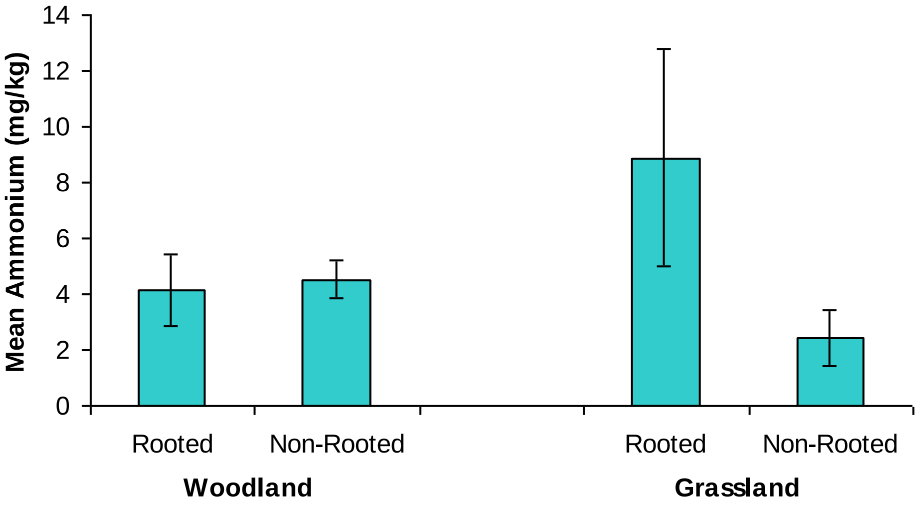


**Fig 5.7**: Effect of rooting on soil ammonium-nitrogen levels in woodland and grassland during early October 2003. Data from different sites were pooled. Scheirer Ray Hare test results shown in **Table 5.1**. Error bars indicate standard error of the means, n = 20.

## Effects of Rooting and Site on Belowground Properties and Processes

| | Woodland | | | | Grassland | | | |
|---|---|---|---|---|---|---|---|---|
| **Source** | **Test** | **Transform** | **F value & df** | **P** | **Test** | **Transform** | **F value & df** | **P** |
| ***Ammonium nitrogen levels*** | Scheirer-Ray-Hare | None | | | Scheirer-Ray-Hare | None | | |
| Site | | | $F_{3,35} = 1.56$ | 0.234 | | | $F_{3,35} = 1.27$ | 0.345 |
| Rooting | | | $F_{1,35} = 3.19$ | 0.088 | | | $F_{1,35} = 6.09$ | **0.021** |
| ***Nitrate nitrogen levels*** | Scheirer-Ray-Hare | None | | | Scheirer-Ray-Hare | None | | |
| Site | | | $F_{3,35} = 1.11$ | 0.843 | | | $F_{3,35} = 2.22$ | 0.661 |
| Rooting | | | $F_{1,35} = 119.57$ | **< 0.001** | | | $F_{1,35} = 121.38$ | **< 0.001** |
| ***Belowground live plant biomass*** | 2-way ANOVA | Square root | | | 2-way ANOVA | Square root | | |
| Site | | | $F_{1,17} = 0.00$ | 0.965 | | | $F_{1,17} = 2.46$ | 0.135 |
| Rooting | | | $F_{1,17} = 22.12$ | **< 0.001** | | | $F_{1,17} = 67.12$ | **< 0.001** |
| ***Decomposition rate of leaf litter*** | 2-way ANOVA | None | | | 2-way ANOVA | None | | |
| Site | | | $F_{1,107} = 0.01$ | 0.921 | | | $F_{1,107} = 4.40$ | **0.038** |
| Rooting | | | $F_{1,107} = 9.45$ | **0.003** | | | $F_{1,107} = 3.54$ | 0.063 |

**Table 5.1**: Two-way ANOVA or Scheirer Ray Hare test results at significance levels; Non-Significant: $p > 0.05$, Significant: $\mathbf{p < 0.05}$, $\mathbf{p < 0.01}$, Highly Significant: $\mathbf{p < 0.001}$. Data were analysed using sequential sum of squares for tests. Significant **P** values shown in bold. **F** = test statistic. **P** = probability associated with F-test. Site = random factor. Rooting = fixed factor. Transform = method of normalising data.

## 5.5 Discussion

Rooting had a significant and direct impact on belowground community properties and processes. The potential for indirect, knock–on affects for the plant community generated from these direct effects of rooting is likely to be widespread.

As hypothesised, there was significantly greater belowground live plant biomass in non-rooted than rooted treatments in woodland and grassland habitats (**Table 5.1**, **Fig 5.4**). Removal of belowground live plant biomass through rooting inevitably removes above ground live plant biomass from the community, which affects the abundance and species richness of plant populations (**Chapter 2**). Specific plant populations of conservation concern where boar root for the belowground plant parts include *Anemone nemorosa* and *Hyacinthoides non-scripta.* Boar root for, and consume rhizomes of *A. nemorosa* (**Appendix 5**) and bulbs of *H. non-scripta* (**Chapter 3**, **Appendix 5**). Bialy (1996) suggested rooting for the edible *A. nemorosa* rhizomes in an oak-hornbeam stand in the Bialowieza primeval forest in Poland reduced their population growth. This contradicts an earlier finding of Falinski (1986) who suggested that the population growth of *A. nemorosa* increased resulting from the regeneration of rhizomes that were fragmented by rooting, which ultimately led to an enhancement of vegetative reproduction. Falinski (1986) concluded that the removal of geophytes and enhancement of vegetative reproduction through rooting and was probably an ancient and co-evolved interrelationship that substantially contributed to the regulation of the structure of the field layer.

Removal of belowground live biomass may decrease food sources for other organisms such as soil dwelling micro-invertebrates and above ground herbivorous animals. Widespread rooting (**Fig 1.2a-c**, **Fig 1.3b**) in my study area could have important implications for organisms that normally live in or feed on the removed above or belowground vegetation.

The removal of belowground live plant biomass through rooting increases belowground dead plant biomass through fragmentation and disruption of root and storage organ systems, discarded uprooted whole plants, and the over-turning of the litter layer and its incorporation into the soil. Singer *et al.* (1984) detected a decrease in the bulk density (biomass per unit volume) of rooted soil in deciduous forest within the Great Smoky Mountains National Park, due to an increase in soil organic matter. In this study however, biomass was measured per unit weight; the highly significant decrease in weight found here suggests that although unmeasured, the volume of biomass would probably also have decreased in rooted than non-rooted treatments. Unconsumed dead plant material increases soil organic matter, which is likely to enhance decomposition (Lacki and Lancia, 1983), leading to enhanced levels of inorganic nutrients within soil (Guo and Sims, 1999), which can become available for uptake by plants. Therefore, the potential indirect impacts on nutrient cycling generated from rooting of belowground live plant biomass could be significant and lead to increased NPP (Lacki and Lancia, 1983, 1986).

As hypothesised, decomposition rate of *C. sativa* leaves was significantly greater in rooted than non-rooted treatments in woodland (**Table 5.1**, **Fig 5.5**). However, contrary to the hypothesis, there was no significant difference in decomposition rate between treatments in grassland habitats although there was a similar trend. A possible explanation for these results could be the differences in the physical environment between rooted and non-rooted soil. Enhanced levels of organic matter in rooted soil (Lacki and Lancia, 1983; Singer *et al.*, 1984), coupled with the increased aeration and temperatures generated by rooting (Kotanen,

1994) is probably accelerating decomposition (Smith and Smith, 2001). Ray (1988) suggests that decomposition is greatly accelerated by the mechanical breakdown of organic matter during the rooting process. Further, the woodland floor tends to have a substantially larger litter layer than grassland due to shedding of the canopy during autumn and winter. The mechanical action of fragmenting and mixing large amounts of litter into deeper soil horizons in woodland through rooting (Singer *et al.*, 1984), coupled with the altered physical environment that it generates, (such as increased aeration, oxygen levels and temperature, (Brownlow, 1994; Kotanen, 1994)), could lead to an increased decay rate in rooted areas within woodland compared to grassland habitats. Furthermore, the greater abundance of leaf litter on woodland floors could support a more abundant and diverse decomposer community than grassland (Zimmer, 2002). This combined with rooting could also contribute to a greater decomposition rate in woodland than grassland habitats (Zimmer, 2002).

As hypothesised, site differences within grassland significantly affected decomposition rate. The two grassland sites used in this study are subject to different forms of management (**Table 1.1**). Although both sites contain high diversity grassland communities, the plant community in **G1** comprises a shorter sword (**Fig 1.1b**) than in **G2** (**Fig 5.8**), indicating greater above ground plant biomass and productivity in **G2**. This could result in chemical, physical and biological inconsistencies in the soils environment between the two sites. For example, variation in physical aspects of the soil such as inconsistent levels of aeration, oxygen, drainage, temperature and crumb structure could have contributed to significant differences in decomposition rate. Further, there could be significant differences in biological aspects of the soil environment between sites such as the micro-invertebrate and decomposer community. Such properties of the soil environment are integral to the decomposition process (Meentemeyer, 1978; Kochy and Wilson, 1997; Cortez, 1998; Guo and Sims, 1999) and variation in this between sites is hence likely to lead to inconsistencies in decomposition rate.

**Fig 5.8** A population of *Cirsium vulgare* in grassland site **G2** in the study area. Photos taken 2004.

Increased decomposition rate can enhance nutrient cycling (Guo and Sims, 1999) and therefore nutrients available to plants (Melillo *et al.*, 1982). Lacki and Lancia (1983), Singer *et al.* (1984), Kotanen (1994) and Moody and Jones (2000) found that rooting significantly altered soil nutrient levels. In most forest sites, nitrogen is a major limiting factor for plant growth (Hunter, 1999). In coppiced woodland (such as in **W1** and **W2**), the removal of wood through harvest regimes increases nutrient losses such as nitrogen, which causes a decline in woodland productivity (Hunter, 1999). Lacki and Lancia (1983), Singer *et al.* (1984) and Kotanen (1994) found higher levels of mineral nitrogen in rooted patches in deciduous forest and grassland habitats. The higher levels of mineral nitrogen available for uptake by plants could create a more productive environment and could increase NPP in coppiced woodland.

However, nutrient enrichment can lead to eutrophication in semi-natural terrestrial systems. The conservation objective of many semi-natural systems is to reduce eutrophication in order to increase or maintain plant species richness and diversity. Therefore, nutrient enrichment from enhanced decomposition rate in rooted patches could lead to decreased plant species richness and diversity due to competitive dominance by a small number of plant species.

As hypothesised, there are significantly greater levels of $NO_3^-$ and $NH_4^+$ in rooted than non-rooted soil in grassland and $NO_3^-$ in woodland (**Table 5.1**, **Fig 5.6**, **5.7**). Plants absorb nitrogen in the form of $NO_3^-$ and $NH_4^+$. The removal of plants through rooting would eliminate plant uptake of $NO_3^-$ and $NH_4^+$ and reduce root enhanced microbial immobilization of $NO_3^-$ (Canals *et al.* 2003; Booth *et al.*, 2005) leading to the increased soil $NO_3^-$ concentration. Significant enhancement of the availability of $NO_3^-$ and $NH_4^+$ to plants is likely to increase the NPP of communities that are re-establishing in rooted environments due to the increased uptake by plant roots and the stimulation of germination from the seed bank (Fenner, 1992) (**Chapter 4**). Most inorganic nitrogen, along with other major nutrient stores from decomposition, is found in the upper horizons of soil (Lacki and Lancia, 1983). Since rooting mixes soil horizons (Spatz and Mueller-Dombois, 1975; Bratton, 1975; Singer *et al.*, 1984) it would seem likely that rooting could incorporate inorganic forms of nitrogen into the deeper layers of the soil, available for uptake by a greater range of plants relative to the depth of their root systems. For example, boar are believed to enhance the growth of pines on nutrient-poor European soils by increasing levels of nutrient cycling in European forests (Andrezejewski and Jezierski, 1978). Since rooting is typically heterogeneous (Welander, 2000b), it could effectively form $NO_3^-$ and $NH_4^+$ rich ‘resource islands’, which could promote productivity in a patchy manner. However, greater levels of $NO_3^-$ and $NH_4^+$ in rooted patches could lead to decreased species richness and diversity due to competitive dominance in nitrogen tolerant plants.

In contrast, oxides of nitrogen, such as $NO_3^-$ are soluble and easily leached by surface run-off or ground water transport. Along with losses during wood harvesting, this is a major reason why nitrogen oxides can become important limiting macronutrients in terrestrial ecosystems (Mix *et al.*, 1992). Rooting, which is known to increase drainage (Brownlow, 1994) via the breaking up of soil horizons, could increase soil $NO_3^-$ losses during periods of heavy rain.

Contrary to hypotheses, rooting had no affect on soil $NH_4^+$ levels in woodland (**Table 5.1**, **Fig 5.7**). In many soils, $NH_4^+$ is readily oxidised to $NO_3^-$ by nitrifying bacteria, particularly in aerated soil where oxygen levels are high (Salisbury and Ross, 1992). Nitrification is inhibited by low oxygen levels due to the low tolerance of nitrifying bacteria for anaerobic conditions (Stanford and Smith, 1972; Paul and Clark, 1989). Further, nitrification can be inhibited in soils of climax grassland communities either by low soil pH or by tannins or phenolic compounds (Salisbury and Ross, 1992). Rooting increases soil aeration (Brownlow, 1994; Kotanen, 1994) and thus probably oxygen levels, which could accelerate oxidation of $NH_4^+$ into $NO_2^-$ and $NO_3^-$, and thus enhance nitrification but not ammonification. This could explain the high soil $NO_3^-$ concentrations in rooted treatments, but the relatively small increase of $NH_4^+$ in rooted than non-rooted soil in grassland and no difference between treatments of $NH_4^+$ in woodland.

Boar rooting significantly decreased belowground live plant biomass in woodland and grassland, increased decomposition rate of leaf litter in woodland, increased soil $NH_4^+$ and

$NO_3^-$ levels in grassland and $NO_3^-$ levels in woodland. These belowground attributes are determinants of community dynamics, thus rooting could enhance the performance of ecosystems with generally low levels of nutrient availability and low overall productivity by enhancing NPP. Surprisingly, site differences had a much smaller effect than predicted, having no impact on belowground biomass and $NO_3^-$ and $NH_4^+$ concentrations in woodland and grassland and decomposition rate in woodland. Only decomposition rate in grassland was significantly affected by site, probably due in part to differences in the soil environment determined largely by substantial differences in the plant community.

# Chapter 6

## General Discussion

### 6.1 Summary of findings

**A**s hypothesised, wild boar rooting markedly affects above and belowground community and population attributes in woodland, grassland and woodland ride habitats in southern England. This is the first in-depth scientific enquiry into the interactions between wild boar rooting and ecosystem dynamics in Britain. It has been shown that plant cover (a surrogate measurement of biomass), diversity and species richness were significantly greater in rooted than non-rooted areas seventeen months after rooting in the three habitat types. This demonstrates that wild boar substantially modify the structure and composition of the plant community in the short-term within different semi-natural habitats in Britain.

Findings of greater species richness in rooted than non-rooted areas confirm those of other studies that found similar results in a range of ecosystems in other countries (Kotanen, 1994, 1995; Welander, 1995, 2000a; Bowman and McDonough, 1991; Onipchenko and Golikov, 1996; Arrington *et al.*, 1999) (**Chapter 2**). However, unlike my findings, the majority of work has found that rooting reduced plant cover in a wide range of ecosystems in other countries (Alexiou, 1983; Ralph and Maxwell, 1984; Bratton, 1974, 1975; Howe *et al.*, 1981; Singer *et al.*, 1984, Hone, 1980; Arrington *et al.*, 1999). Reasons for the discrepancies between research results on impacts of rooting on plant cover could be two-fold. The methodology of the studies that found plant cover was reduced by rooting incorporated only two treatments; rooted and non-rooted plots or transects. No fencing or method of protecting rooted and non-rooted treatments from rooting throughout the duration of the studies were utilised. In my study however, the greatest plant cover occurred in the rooted, fenced treatments. Protection from re-rooting and trampling by boar had a large impact on plant re-establishment, such that plant cover in rooted fenced treatments exceeded the cover of unprotected plots between four and seventeen months in grassland. Additionally, different temporal scales could be responsible for the discrepancies in research results. Studies that found rooting reduced plant cover were carried out less than one year after rooting. Since rooting involves the removal or redistribution of vegetation, plant cover will inevitably recover over time. In my study, the greatest plant cover was at seventeen months after rooting in rooted fenced treatments in grassland and after one year in rooted fenced treatments in woodland.

This is the first study to show that different plant functional groups respond differentially to rooting in Britain. The frequency of annual and perennial forbs was significantly greater in rooted than non-rooted treatments, and the cover of perennial graminoids was significantly lower in rooted than non-rooted treatments in grassland and woodland rides up to seventeen months after rooting (**Chapter 2**). These results are similar to those of Kotanen (1994, 1995) who found that annuals proliferated in the year following boar rooting in a Californian prairie, and Welander (2000a), who found that plant re-colonisation after rooting was dominated by plants with small, wind-dispersed seeds.

This work investigating the effects of rooting on the viable seed bank is the first demonstration of the potential source of re-establishment of plants after boar rooting. This research demonstrates that rooting substantially modifies emergence from the seed bank, at the community and functional group level from different semi-natural British habitats, in the lab. The emergence of seedlings was significantly greater from rooted than non-rooted treatments at the community level, leading to greater species richness, number of individuals and diversity from woodland and grassland habitats. Seeds of ruderal strategists (particularly annual forbs) were best adapted for short-term (six months–one year) post rooting colonisation and predominantly contribute to the greater species richness and diversity of plants that emerged from rooted than non-rooted soil (**Chapter 4**).

I have shown that belowground live plant biomass was significantly lower in rooted than non-rooted areas in woodland and grassland, and decomposition rate of leaf litter was significantly higher in rooted than non-rooted treatments in woodland. Further, soil $NO_3^-$ concentrations were significantly greater from rooted than no-rooted soil in woodland and grassland, and $NH_4^+$ concentrations were significantly greater from rooted than no-rooted soil in grassland. These belowground community attributes are determinants of productivity and dynamics of the community. This is the first demonstration in Britain that wild boar substantially modify belowground community properties (**Chapter 5**).

Some other studies have found that soil mineral nitrogen in grassland was greater from rooted than non-rooted areas (Lacki and Lancia, 1983; Singer *et al.*, 1984; Kotanen, 1994), while others found that rooting had no effect on soil mineral-nitrogen levels in forest ecosystems (Groot Bruinderink and Hazebroek, 1996; Moody and Jones, 2000; Mohr *et al.*, 2005). Further, Groot Bruinderink and Hazebroek (1996) found no effect of boar rooting on decomposition rate and levels of organic matter in forest ecosystems, whereas Lacki and Lancia (1983) detected enhanced litter decomposition with increased length of disturbance by feral pigs in woodland.

The abundance of *H. non-scripta* was significantly lower in rooted than non-rooted treatments in different semi-natural woodland habitats over three spring growing seasons. However, the proportional change in *H. non-scripta* cover significantly increased over that time in rooted than non-rooted treatments. Re-establishment was substantial, and full recovery in fenced rooted plots was expected by a fourth growing season (**Chapter 3**). This is the first demonstration of the effects of rooting on *H. non-scripta* populations. It was shown that by preventing further rooting, *H. non-scripta* populations began to recover substantially after one and two years (**Fig 3.6 a-b**). *H. non-scripta* bulbs are an important and abundant food source of wild boar (Goulding *et al.*, 1998) available at the time in the year when other food sources are particularly scarce (Goulding, 2003b). This is of conservation importance considering that *H. non-scripta* is globally rare with nearly half its total population in the UK (Thompson and Cox, 1978) and its numbers declining in southern England (Gow, 2002; Pilgrim and Hutchinson, 2004).

## 6.2 Implications of findings

When addressing any aspect of the community affected by rooting, one must first understand the heterogeneous and fluctuating nature of rooting in terms of time, space, frequency and severity (depth of rooting). This contributes to the overall intensity of the disturbance, which is of primary importance for understanding both the short and long-term effects of rooting. With this understanding, one can make predictions related to recovery of rooted patches, based on the intensity of the disturbance. Rooting intensity fluctuates spatially and temporally and in its severity, and variations in this intensity profoundly affect the consequences for the plant community. For example, rooting for *H. non-scripta* bulbs tends to be deeper (15-30cm) than the typical rooting depth (15cm) and occurs in well-defined patches (~1m$^2$) during a defined period in the year (~ May-August). At other times of the year, these patches may be completely ignored by boar and thus given the opportunity to recover. The following year, boar searching for *H. non-scripta* bulbs will probably target the same site, but could root both previously non-rooted and previously rooted patches, effectively creating a mosaic of both undisturbed and rooted patches of various ages and thus differential intensities of disturbance. The intensity of rooting within one year and over successive years will vary greatly and thus differentially influence *H. non-scripta* re-establishment in those patches.

There appears to be a continuum of rooting intensity, with small (< 0.5m$^2$), relatively shallow (1-5cm), once rooted patches at one end, and large (> 1ha) patches that are deeply (> 15cm) and frequently re-rooted throughout the current and successive years, at the other. At any point in the continuum, it may be possible to predict the likely impact of rooting on ecosystem dynamics. For example, short and long-term effects of frequently applied severe rooting are likely to result in low plant diversity. Impacts of rooting on above and belowground community attributes of small, shallow, singularly rooted patches are likely to be insignificant, with rapid recovery of the plant assemblage. Conversely, subsequent recovery of large, previously repeatedly rooted areas would lead to a significantly modified plant community in the short-term, occupied predominantly by ruderal strategists (Chapter 2) due to their relatively high disturbance tolerance. Seed bank turnover would probably increase, with greater incorporation of seeds from pioneer species due to the physical disturbance from boar rooting and trampling, and greater emergence through an altered physical and chemical soil environment that stimulate dormancy breaking mechanisms and germination (Chapter 4). Nutrient turnover is likely to increase through nutrient release from enhanced decomposition (Lacki and Lancia, 1983) and decreased nutrient uptake, in particular $NO_3^-$ (Lacki and Lancia, 1983; Singer *et al.*, 1984; Kotanen, 1994) (Chapter 5) through vegetation removal. Nutrient enrichment could result in eutrophication leading to low plant species richness due to competitive dominance in some plant species. Conversely, higher levels of nutrients in the soil could enhance NPP (particularly in ecosystems with generally low levels of nutrient availability and low overall productivity). This could lead to greater plant diversity, species richness (Bowman and McDonough, 1991; Kotanen, 1994, 1995; Welander, 1995, 2000a; Onipchenko and Golikov, 1996; Arrington *et al.*, 1999) (Chapter 2) and biomass (Bowman and McDonough, 1991) (Chapter 2), in a resource-rich environment (Hunter, 1999). Potentially greater levels of localised light and space in rooted areas would probably lead to greater evenness of plant species abundance in the short-term.

Over the longer-term (> three years), where intensive re-rooting is prevented, the structure and composition of the plant community is likely to return to a similar equilibrium state to that of adjacent non-rooted areas. As vegetation recovers, it is probable that functional groups would display a shift in dominance from ruderals towards competitive and stress tolerant strategists such as perennial forbs and woody species, following changes within the soil. For example, nutrient levels would probably lower, due to a higher nutrient uptake from a greater plant biomass, and lower nutrient release from reduced decomposition. Additionally, localised light and space would probably decrease resulting from a greater plant biomass, and subsequent competition for all resources would increase, leading to a reduction in evenness of plant species abundance, following competitive dominance of some plant species. As competition increases, species richness may begin to lower until an equilibrium state is reached.

Disturbance-caused patchiness (such as rooting, which is typically patchy) is important for the evolution and maintenance of heterogeneity in ecosystems (White and Pickett, 1985; Whitford and Kays, 1999). Mammals create important forms of biopedturbation, and are substantial contributors to the development of heterogeneity within communities (Whitford and Kays, 1999). Wild boar rooting is an intensive and patchy form of biopedturbation. Intermediate intensity levels of rooting are likely to create a more heterogeneous (medium-scale) (Stewart *et al.*, 2000; Wiens, 2000) and mosaic woodland environment. A more heterogeneous woodland environment would create a greater number of niches and therefore, likely to promote biodiversity and species richness (Hunter, 1999).

It is clear from my research that wild boar fundamentally modify the structure of communities within semi-natural woodland, grassland and woodland ride habitats. In light of this, wild boar can be described as allogenic ecosystem engineers because they mechanically change the environment by transforming biotic and abiotic materials from one physical state to another (Jones *et al.*, 1994, 1997). This is the first consideration of wild boar as ecosystem engineers, and their rooting, as allogenic engineering. Boar engineer their environment by substantially altering the energy flow through different ecosystems by modifying the availability of elemental resources which underlie food webs within the community. Intermediate levels of their engineering in semi-natural woodland and woodland ride habitats could become a valued natural form of management, sustaining potentially enhanced levels of NPP, plant species richness, diversity and small-scale heterogeneity.

## 6.3 Boar past and present

It is probable that before wild boar became extinct in Britain, their engineering acted as a natural management regime that helped shape the structure and composition of forest ecosystems over thousands of years. Their return into Britain brings with them their prominent and very specific form of ecosystem engineering. However, it is a very different world they have returned to, with remaining semi-natural ecosystems small and fragmented, plant communities changed in composition, distribution, diversity and species richness.

Nothing of the wildwood remains today, and the surviving patches of woodland have been subject to thousands of years of use, modification and fragmentation. From pollen record, the classical view point was that the wildwood of Britain was dense and dark, dominated by the deciduous *Tilia cordata* or *Ulmus glabra* with *Corylus avellana,* each of which casts substantial shade (Rackham, 1980; Ingrouille, 1995; Rackham, 1997). Trees would have grown tall and closely spaced with a suppressed under-flora, so that only shade tolerant species could grow (Rackham, 1980; Ingrouille, 1995). Gaps in the forest would have been created by tree-falls, forming pit and mound topography, which would have led to patches of new growth creating subsequent heterogeneity and patches of elevated species richness (Beatty, 1984; Beatty and Stone, 1986; Peterson *et al*., 1990; Peterson and Pickett, 1990). However, this view has recently been challenged by Frans Vera (2000), who proposed that large herbivores were present at high enough densities to maintain a much more open landscape and hence prevented the dominance of closed canopy forest. Mitchell (2005) however disputes Vera's view, and through pollen record, comparisons of areas where large herbivores were abundant and where they were mostly absent found no significant difference in the relative abundance of major deciduous species. Mitchell (2005) concluded that there was no evidence to support Vera’s view that high herbivore densities created frequent areas of open vegetation and no tree canopy.

My study area lies deep inside the Weald of Kent and Sussex in southern Britain, and is thought to be one the last and largest areas of wild wood to be cleared (Rackham, 1997). Over thousands of years of active management, the wild wood within the Weald was developed into a vast wood pasture, managed with grazing livestock, and pollarding trees to produce repeated crops of wood and sometimes leaves, and occasionally coppicing (Rackham, 1997). The pannaging of pigs, a famous use of wood pasture, would also have been carried out in the Weald; the consumption of acorns and beech mast would have contributed to wood pasture management by reducing tree seedling growth and hence the encroachment of secondary woodland (Brownlow, 1994; Rackhan, 1997). Today, the Weald is one of the most wooded parts of Britain, with around 70% of the woodland still classified as ancient (since it has been continually wooded since at least 1600AD) and hence is of great importance to wildlife (Kaye-Smith, 1973; Brandon, 2003).

Ancient woodlands are believed to have been more heterogeneous in the past than they are today, largely determined by the activities of large mammals such as wild boar. Since rooting tends to occur in patches (Kotanen, 1995; Welander, 2000b) and helps to create spatial heterogeneity (Milton *et al.*, 1997; Arrington *et al.*, 1999; Welander, 2000b), *H. non-scripta*, for example, may have once been more patchily distributed than the continuous bluebell ‘carpets’ that can be seen in British woodlands today. It seems probable, that wild boar in modern Britain, may contribute to changing the ecology of woodlands, in particular those that support low diversity or monoculture populations such as *H. non-scripta*, to a more patchy and mosaic environment, similar to that which may have existed hundreds of years ago, before wild boar became extinct.

The fundamental question is whether the former native wild boar can live in balance with our changed, modern and fragmented Britain? Connell's (1978) intermediate disturbance hypothesis suggests that an intermediate intensity of rooting disturbance (such as annual or biennial rooting events) will alter ecosystem dynamics, although modification will be moderate and heterogeneous. With knowledge of current rooting intensities within different habitat types, subsequent management may be necessary to either provide protection for areas that are vulnerable to intense rooting, or contain the population to within a sustainable size. If boar remain a permanent feature of British semi-natural habitats, suspension of rooting through a rotational fencing management system over one to three year periods, could prove an important conservation tool in the re-establishment of plant cover in severely and repeatedly rooted areas. My study suggests that after rooting, the re-establishment of the plant community in protected areas over three years would be substantial.

Permanent fencing could totally prevent any inappropriate and unwanted 'damage' in specific areas. However, standard stock fencing is inadequate for excluding wild boar, and the costs for the construction of boar-proof fencing are likely to be very expensive (Wilson, 2005). Further, permanent fencing is likely to block public rights of way resulting in the further necessity of incorporating styles, which is a further expense. Permanent boar-proof fencing therefore could be an appropriate method for preventing rooting within vulnerable and relatively small, conservation-rich or agricultural land, where afforded. Over larger areas, and in general, permanent fencing would likely to be an impractical and expensive method of control. Boar tend to re-root the same sites every year (Falinski, 1986; Welander, 2000b; Goulding, 2003b; pers. obs.), hence periodically fencing these sites with temporary fencing would rotationally transfer the disturbance to other, formerly non-rooted sites. This would effectively spread the disturbance spatially, but moderate the intensity of the disturbance to a more sustainable, intermediate level. Temporary electric fencing is a potential cheaper option, although currently it has not been tested in the UK for the successful exclusion of boar (Wilson, 2005).

Wild boar populations on Auckland Island, New Zealand (Challies, 1975) and Horn Island, Mississippi (Baron, 1982) are examples where a balance has existed, and was predicted to continue, between boar and their engineered environment. In both cases, reports were of little overall vegetation disruption over the long-term due to rapid recovery of plants within between six months and one year after rooting. Challies (1975) concluded that by leaving the boar population undisturbed, the animals and their modified environment find their own equilibrium. In this country, public opinion on the desirability and appropriateness of a new boar-mediated ecological equilibrium has not been quantified but is likely to be controversial. The main objections of people to a continued boar presence seem to be associated with potential financial losses relating mainly to disease and ‘damage’ to agricultural land (Goulding *et al.*, 1998; Goulding, 2003b; pers. comm.). Fears associated with ecological issues appear mainly to relate to specific plant species such as *H. non-scripta*, which are specifically targeted by boar. With this work, and further research on the ecological impacts of rooting coupled with a managed and sustainable population size, I believe that the new ecological equilibrium could be accepted by the majority of people and acknowledged as an important contributor of plant species richness and diversity, particularly in woodland.

## 6.4 Future work

One of the main problems with determining the intensity of rooting (in order to predict and manage its effects) is the fluctuating size of the wild boar population. Their numbers vary considerably from year to year (Goulding, 2003b) in relation to a food supply that fluctuates annually, in particular acorns and beech mast (Henry and Conley, 1972; Baber and Coblentz, 1987; Groot Bruinderink and Hazebroek, 1994; Massei *et al.*, 1996). Future work into the determination of an accurate wild boar population density (including mean annual fluctuation), could provide valuable information which could be directly related to accurate estimates of a sustainable population size, relating to acceptable and sustainable rooting intensities within a given habitat. The ability to accurately estimate potential rooting intensities from known boar population densities, and how this is likely to fluctuate over time, could be used to determine potential management regimes in relation to the ecological impacts that different rooting intensities have in various wildlife-rich semi-natural habitats in southern England.

Boar tend to re-root the same sites every year (Falinski, 1986; Welander, 2000b; Goulding, 2003b; pers. obs.). Since other sites are available to the boar and remain non-rooted, it is possible, though unquantified, that boar are at least partially choosing these sites in preference to others. Following this possibility, in this study the rooted treatments may have been significantly different from the non-rooted plots at the beginning of the study, which implies that the non-rooted treatments did not effectively control for the rooted treatments, and thus the comparisons between treatments may not be wholly reliable. In light of this, future work into potential rooting preferences could provide valuable information relating to the protection of specific sites that are likely to receive the greatest rooting pressure. If rooting preferences exist, such knowledge could contribute towards the formation of legislation by DEFRA, which could potentially involve subsides or advice for fencing programmes in boar-occupied conservation or wildlife-rich areas, and badly affected agricultural land, particularly in smallholdings where a potentially greater proportion of land could be affected by rooting. Further, legislation is necessary to protect wild boar, domestic animals and the public from the cruelty and dangers of poaching, in order that safe and humane methods, if required, may be utilised to contain the boar population to within a sustainable size.

It is known that a greater number of individuals and species emerge from the viable seed bank in rooted than non-rooted soil in the lab (Chapter 4), but it is not known if this occurs in the field, and how important the seed rain is to the re-establishment of plants in rooted areas. Further, the potential role of boar in the dispersal of plant propagules could contribute to the re-establishment of plants in rooted areas. Future work on the relative contributions to plant re-establishment from the viable seed bank, the seed rain and dispersal into sites via adhering to the exterior of boar, would provide information on the likely composition, nature and speed of plant re-colonisation in previously rooted areas of varying intensities, across different communities and habitats. Such information could aid predictions of plant recovery after rooting and be utilised for conservation related management in rooted areas.

The higher levels of resources available to plants in rooted areas such as nutrients, for example $NO_3^-$, (Lacki and Lancia, 1983; Singer *et al.*, 1984) (Chapter 5) probably substantially contribute to the significant community and functional group level changes that occur within the plant community in rooted environments (Chapter 2). Plants with the greatest competitive ability may be out-competing others where resources are high, leading to increased biomass and altered species richness. Future work on inter-specific and intra-specific competition across areas of different rooting intensities within different habitats could provide important information relating to the re-establishment of plants at the population and community level. Such information could aid predictions of plant recovery after rooting and greatly contribute to forming management regimes for conservation in rooted areas.

Spatial heterogeneity has been found to be an important impact of rooting across different habitat types (Milton *et al.*, 1997; Arrington *et al.*, 1999; Welander, 2000b). Heterogeneity is important because it contributes to plant community diversity and species richness (Hunter, 1999). The impact of rooting on different spatial scales has not been investigated. There is considerable scope for future work into the effects of rooting on spatial and temporal heterogeneity at different scales across different habitat types.

A greater understanding of the effects of rooting in various habitats and seasons on soil moisture retention is important relating to seed dormancy breaking mechanisms and germination and plant growth and diversity. Where leaf litter is abundant on the soil surface, rooting will increase incorporation of litter into the soil (Singer *et al.*, 1984) and may increase the moisture retention potential. This may be beneficial for seed germination, plant growth and soil fauna in areas with predominantly clay soils (which condenses and cracks in times of drought, such as in my study area) by increasing moisture retention and rendering the soil more penetrable for roots and soil dwelling and fossorial animals. However, soil compaction, leading to water logging, drainage problems and leaching of nutrients, is potentially increased in rooted soils (particularly with low levels of dead organic matter). Contrary to this, particularly in soils low in organic matter, rooting could reduce soil moisture through breaking up and exposing soil layers, which could potentially reduce seed dormancy breaking mechanisms, prevent seed germination (Fenner, 1985; Bradbeer, 1988; Fenner, 1992) and reduce plant growth.

Further chemical analysis such as measuring phosphate and potassium levels of soils of various rooting intensities may be beneficial in predicting rooting effects on plant re-establishment. Phosphate and potassium are important nutrients for plant growth (Salisbury and Ross, 1992; Taiz and Zeiger, 1998); rooting may increase or reduce levels in the soil, which could vary according to rooting intensity and habitat type. Soil pH measurements at various rooting intensities and different habitat types could be valuable information relating to seed dormancy mechanisms and germination (Fenner, 1985; Bradbeer, 1988; Fenner, 1992), and therefore could significantly affect the plant community. Soil temperature, important for decomposition and seed dormancy breaking mechanisms and germination, could also be measured across various rooting intensities. Although Kotanen (1994) reported increased soil temperatures in rooted areas in Californian prairie, no other investigation into this exists. Measuring different light intensities within a range of rooting ages and intensities on the woodland floor across different woodland types could be important information contributing to an understanding of post-rooting plant re-establishment, seed germination and overall structure of the community.

Rooting is likely to have a substantial impact on the belowground invertebrate community, which plays a vital role in the breakdown of organic matter during the process of decomposition, and thus is a determinant of NPP. Belowground invertebrates are also important in the life cycles of plants and valuable food sources for a number of different organisms across taxa. Apart from Vtorov (1993) and Mohr *et al.* (2005) who found that rooting decreased populations of soil microarthropods, no other work exists in this field. Future work on how rooting affects the species richness and abundance of belowground invertebrates could be important ecological research that could contribute towards the overall understanding of rooting and the formation of related management regimes.

## 6.5 Conclusions

Since humans are responsible for the dramatic decline in woodland and the former extinction of wild boar in Britain, it seems appropriate that we embrace the re-establishment of this fascinating animal into Britain and respond to the situation generously and with integrity. The reaction of many people, mainly those directly and commercially affected by the presence of these animals, is to want a partial or total eradication of boar. This research into the ecological effects of boar rooting within semi-natural habitats in Britain has thrown considerable light onto the ecological effects of rooting, which is important information that could largely contribute to the formation of greatly needed, appropriate forms of management and legislation. With this in action, aided by future research, I believe wild boar can live in balance with our changed, modern and fragmented Britain.

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Appendix 1**. Complete species list incorporating all plant species recorded in the study area across woodland, grassland and woodland ride ecosystems** (Chapter 2)**, and all species that germinated and grew from the seedbank** (Chapter 4) **(Phillips, 1980; Rose, 1981; Garrard and Streeter, 1983; Fitter, 1987). Species in Black = species only recorded on site in the study area** (Chapter 2)**.** Species in Red = Species only recorded in growth room from seed bank **(**Chapter 4**).** Species in Blue = Species present in both study area and grown from seed bank**. Grime's classification of functional groups taken from Grime et al (1989): C = Competitive, S = Stress Tolerant, R = Ruderal. Key for general functional group classification: 1 = moss, 2 = woody species, 3 = annual forb, 4 = biennial forb, 5 = perennial forb, 6 = bulbous species, 7 = annual graminoid, 8 = perennial graminoid. Ancient woodland indicator species determined from Stewart et al (1994), Wigginton (1999) and Rose (1999).**

# Kingdom Plantae

## Division Bryophyta

### Class Musci

| | | Ancient Woodland Indicator Species | Functional Group (Grime's Classification) | General Functional Group Classification |
|---|---|---|---|---|
| *Atrichum undulatum* | Common Smoothcap | | S-R | 1 |
| *Brachythecium rutabulum* | Rough Stalked Feather Moss | | S-R | 1 |
| *Calliergon cuspidatum* | Pointed Spear Moss | | S-R | 1 |
| *Dicranella heteromalla* | Silky Forklet Moss | | S-R | 1 |
| *Dicranum scoparium* | Broom Fork Moss | | S-R | 1 |
| *Eurhynchium praelongum* | Common Feather Moss | | S-R | 1 |
| *Eurhynchium swartzii* | Swartz's Feather Moss | | S-R | 1 |
| *Fissidens taxifolius* | Common Pocket Moss | | S-R | 1 |
| *Hypnum jutlandicum* | Heath Plait Moss | | S-R | 1 |
| *Mnium hornum* | Swan's Neck Thyme Moss | | S-R | 1 |
| *Plagiomnium undulatum* | Hart's Tongue Thyme Moss | | S-R | 1 |
| *Polytrichum formosum* | Bank Haircap | | S-R | 1 |
| *Pottia trunchator* | Common Pottia | | S-R | 1 |
| *Pseudoscleropodium purum* | Neat Feather Moss | | S-R | 1 |
| *Rhytidiadelphus squarrosus* | Springy Tough Moss | | S-R | 1 |
| *Thuidium tamariscinum* | Common Tamarisk Moss | | S-R | 1 |

## Division Filicinophyta

### Family Polypodiaceae

| | | | | |
|---|---|---|---|---|
| *Dryopteris austriaca* | Broad Buckler Fern | | S-C/ C-S-R | 5 |
| *Dryopteris filix-mas* | Common Male Fern | | S-C | 5 |
| *Pteridium aquilinum* | Bracken | | C | 5 |

## Division Angiospermophyta

### Class Dicotyledoneae
### Order Ranales
### Family Ranunculaceae

| | | | | |
|---|---|---|---|---|
| *Anemone nemorosa* | Wood Anemone | yes | S/ SR | 5 |
| *Ranunculus ficaria* | Lesser Celendine | | R/ S-R | 5 |
| *Ranunculus flammula* | Lesser Spearwort | | C-R/ C-S-R | 5 |
| *Ranunculus parviflorus* | Small-flowered Buttercup | | R | 3 |
| *Ranunculus repens* | Creeping Buttercup | | C-R | 5 |

## Order Plantaginales
## Family Plantaginaceae

| | | | | |
|---|---|---|---|---|
| *Plantago major* | Greater Plantain | | R | 5 |

## Order Violales
## Family violaceae

| | | | | |
|---|---|---|---|---|
| *Viola lactea* | Pale Dog Violet | | S | 5 |
| *Viola riviniana* | Common Dog Violet | | S | 5 |

## Order Histiflora
## Family Hypericaceae

| | | | | |
|---|---|---|---|---|
| *Hypericum hirsutum* | Hairy St. Johns-wort | | S/ C-S-R | 5 |
| *Hypericum humafusum* | Trailing St. Johns-wort | | S/ S-R | 5 |
| *Hypericum maculatum* | Imperforate St. Johns-wort | | C-R/ C-S-R | 5 |
| *Hypericum perforatum* | Perforate St. Johns-wort | | C-R/ C-S-R | 5 |

## Order Centrospermae
## Family Caryophyllaceae

| | | | | |
|---|---|---|---|---|
| *Cerastium arvense* | Field Mouse-ear | | R/ C-S-R | 5 |
| *Cerastium holosteoides* | Common Mouse-ear | | R/ C-S-R | 5 |
| *Lychnis flos-cuculi* | Ragged Robin | | C-S-R | 5 |
| *Moehringia trinervia* | Three Nerved Sandwort | yes | S-R | 3 |
| *Sagina apetala* | Common Annual Pearlwort | | R/ C-S-R | 3 |
| *Sagina procumbens* | Procumbent Pearlwort | | R/ C-S-R | 5 |
| *Silene dioica* | Red Campion | | C-S-R | 5 |
| *Stellaria graminea* | Lesser Stitchwort | | C-S-R | 5 |
| *Stellaria media* | Common Chickweed | | R | 3 |
| *Stellaria neglecta* | Greater Chickweed | yes | R | 3 |

## Family Oxalidaceae

| | | | | |
|---|---|---|---|---|
| *Oxalis acetosella* | Wood Sorrel | yes | C-S-R/ S | 5 |

## Order Malvales
## Family Oleaceae

| | | | | |
|---|---|---|---|---|
| *Fraxinus excelsior* | Ash Seedling | | C | 2 |

## Order Sapindales
## Family Aceraceae

| | | | | |
|---|---|---|---|---|
| *Acer campestre* | Field Maple Seedling | yes | S-C | 2 |

## Order Celastrales
## Family Aquifoliaceae

| | | | | |
|---|---|---|---|---|
| *Ilex aquifolium* | Holly Seedling | yes | S-C | 2 |

## Order Rosales
## Family Rosaceae

| | | | | |
|---|---|---|---|---|
| *Crataegus monogyna* | Hawthorn Seedling | yes | S-C | 2 |
| *Potentilla anserina* | Silver Weed | | S/ C-S-R | 5 |
| *Potentilla erecta* | Tormentil | | S/ C-S-R | 5 |
| *Potentilla reptans* | Creeping Cinquefoil | | C-R/ C-S-R | 5 |
| *Potentilla sterilis* | Barren Strawberry | yes | S | 5 |
| *Prunus avium* | Wild Cherry Seedling | yes | S-C | 2 |
| *Prunus spinosa* | Sloe Seedling / Blackthorn | | S-C | 2 |
| *Rosa canina* | Common Dog Rose | | S-C | 2 |
| *Rubus fruticosus* | Bramble | | S-C | 2 |

## Family Leguminosae

| | | | | |
|---|---|---|---|---|
| *Lotus corniculatus* | Common Bird's-Foot Trefoil | | S/ C-S-R | 5 |
| *Medicago lupulina* | Black Medick | | S-R/ R | 3 |
| *Sarothamnus scoparius* | Broom | | S-C | 2 |
| *Trifolium dubium* | Lesser Trefoil | | R/ S-R | 3 |
| *Trifolium micranthum* | Slender Trefoil | | S-R | 3 |
| *Trifolium repens* | White Clover | | C-S-R/ C-R | 5 |
| *Vicia sativa* | Common Vetch | | C/ C-S-R | 3 |

## Order Myrtales
## Family Onagracese

| | | | | |
|---|---|---|---|---|
| *Chamaenerion angustifolium* | Rosebay Willow Herb | | C | 5 |
| *Circaea lutetiana* | Enchanter’s Nightshade | | C-R | 5 |
| *Epilobium hirsutum* | Great Willow Herb | | C | 5 |
| *Epilobium montanum* | Broad Leaved Willow Herb | | C-S-R | 5 |
| *Epilobium obscurum* | Short Fruited Willow Herb | | C-S-R | 5 |
| *Epilobium parviflorum* | Hoary Willow Herb | | R/ C-S-R | 5 |
| *Epilobium tetragonum* | Square Stalked Willow Herb | | C-S-R | 5 |

## Order Umbellales
## Family Araliaceae

| | | | | |
|---|---|---|---|---|
| *Hedera helix* | Ivy | | S-C | 2 |

## Family Umbelliferae

| | | | | |
|---|---|---|---|---|
| *Conopodium majus* | Pignut | yes | S/ C-S-R | 5 |
| *Heracleum sphondylium* | Hog Weed | | C-R | 5 |

## Order Euphorbiales
## Family Euphorbiaceae

| | | | | |
|---|---|---|---|---|
| *Mercurialis perennis* | Dog Mercury | | S-C | 5 |

**Order Polygonales**
**Family Polygonaceae**

| | | | | |
|---|---|---|---|---|
| *Rumex acetosa* | Common Sorrel | | C-S-R | 5 |
| *Rumex acetosella* | Sheep Sorrel | | S-R/ C-S-R | 3 |
| *Rumex obtusifolius* | Broad leaved Dock | | C-R | 5 |
| *Rumex sanguineus* | Wood Dock | | C-R | 5 |

**Order Urticales**
**Family Urticaceae**

| | | | | |
|---|---|---|---|---|
| *Urtica dioica* | Stinging Nettle | | C | 5 |

**Order Fagales**
**Family Betulaceae**

| | | | | |
|---|---|---|---|---|
| *Betula pendula* | Silver Birch Seedling | | C/ S-C | 2 |

**Family Corylaceae**

| | | | | |
|---|---|---|---|---|
| *Carpinus betulus* | Hornbeam Seedling | yes | S-C | 2 |
| *Corylus avellana* | Hazel Seedling | | S-C | 2 |

**Family Fagaceae**

| | | | | |
|---|---|---|---|---|
| *Castanea sativa* | Sweet Chestnut Seedling | | S-C | 2 |
| *Fagus sylvatica* | Beech Seedling | | S-C | 2 |
| *Quercus robur* | Pedunculate Oak Seedling | | S-C | 2 |

**Order Salicales**
**Family Salicaceae**

| | | | | |
|---|---|---|---|---|
| *Salix caprea* | Goat Willow Sapling | | C | 2 |

**Order Ericales**
**Family Ericaceae**

| | | | | |
|---|---|---|---|---|
| *Calluna vulgaris* | Ling | | S-C | 5 |

**Order Primulales**
**Family Primulaceae**

| | | | | |
|---|---|---|---|---|
| *Anagallis arvensis* | Scarlet Pimpernel | | R/ S-R | 5 |
| *Lysimachia nemorum* | Yellow Pimpernel | yes | S/ C-S-R | 3 |
| *Primula vulgaris* | Primrose | yes | S | 5 |

## Order Tubiflorae
## Family Scrophulariaceae

| | | | | |
|---|---|---|---|---|
| *Digitalis purpurea* | Foxglove | | C-R/ C-S-R | 4 |
| *Euphrasia nemorosa* | Eyebright | | S-R | 3 |
| *Kickxia elatine* | Sharp leaved Fluellen | | S-R | 3 |
| *Pedicularis sylvatica* | Lousewort | | S-R | 3 |
| *Veronica arvensis* | Wall Speedwell | | S-R | 3 |
| *Veronica montana* | Wood Speedwell | yes | S/ C-S-R | 5 |
| *Veronica officinalis* | Heath Speedwell | | S/ C-S-R | 5 |
| *Veronica persica* | Common Field Speedwell | | R | 3 |
| *Veronica serpyllifolia* | Thyme Leaved Speedwell | | R/ C-S-R | 5 |

## Family Labiatae

| | | | | |
|---|---|---|---|---|
| *Ajuga reptans* | Bugle | | C-S-R | 5 |
| *Betonica officinalis* | Betony | | C-S-R | 5 |
| *Glechoma hederacea* | Ground Ivy | | C-S-R | 5 |
| *Lamiastrum galeobdolon* | Yellow Archangel | yes | S/ S-C | 5 |
| *Mentha arvensis* | Corn Mint | | C-R | 5 |
| *Prunella vulgaris* | Selfheal | | C-S-R | 5 |
| *Stachys arvensis* | Field Woundwort | | R | 3 |
| *Stachys sylvatica* | Hedge Woundwort | | C/ C-R | 5 |
| *Teucrium scorodonia* | Wood Sage | | C-S-R | 5 |

## Order Rubiales
## Family Rubiaceae

| | | | |
|---|---|---|---|
| *Galium aparine* | Cleavers, Goosegrass | C-R | 3 |
| *Galium palustre* | Common Marsh Bedstraw | C-R/ C-S-R | 5 |

## Family Caprifoliaceae

| | | | |
|---|---|---|---|
| *Lonicera periclymenum* | Honeysuckle | *S-C* | 2 |
| *Sambucus nigra* | Elder Seedling | C | 2 |

## Order Asterales
## Family Compositae

| | | | |
|---|---|---|---|
| *Arctium lappa* | Greater Burdock | C-R | 5 |
| *Bellis perennis.* | Common Daisy | R/ C-S-R | 5 |
| *Centaurea nigra* | Common knapweed | S/ C-S-R | 5 |
| *Cirsium arvense* | Creeping Thistle | C | 5 |
| *Cirsium palustre* | Marsh Thistle | C-S-R | 5 |
| *Cirsium vulgare* | Spear Thistle | C-R | 5 |
| *Gnaphalium uliginosum* | Marsh Cudweed | R | 3 |
| *Pilosella officinarum* | Mouse-ear Hawkweed | R/ C-S-R | 5 |
| *Pulicaria dysenterica* | Common Fleabane | S-C | 5 |
| *Senecio jacobaea* | Common Ragwort | R/ C-R | 4 |
| *Sonchus oleraceus* | Smooth Sow Thistle | R/ C-R | 3 |
| *Taraxacum officinale* | Dandelion | R/ C-S-R | 5 |
| *Tussilago farfara* | Coltsfoot | C-R | 8 |

## Order Gentianales
## Family Gentianaceae

| | | | | |
|---|---|---|---|---|
| *Centaurium erythraea* | Common Centaury | | S-R | 3 |
| *Centaurium pulchellum* | Lesser Centaury | | S-R | 3 |

## Order Genraniales
## Family Polygalaceae

| | | | | |
|---|---|---|---|---|
| *Polygala serpyllifolia* | Heath Milkwort | | S | 5 |
| *Polygala vulgaris* | Common Milkwort | | S | 5 |

## Order Chenopodiales
## Family Portulacaceae

| | | | | |
|---|---|---|---|---|
| *Montia ontana* | Blinks | | R/ S-R | 3 |
| *Montia perfoliata* | Spring Beauty | | R/ S-R | 3 |

## Family Chenopodiaceae

| | | | | |
|---|---|---|---|---|
| *Chenopodium botryodes* | Small Red Goosefoot | | R/ C-R | 3 |
| *Chenopodium polyspermum* | Many-Seeded Goosefoot | | R/ C-R | 3 |
| *Chenopodium rubrum* | Red Goosefoot | | R/ C-R | 3 |

## Order Rhoeadales
## Family Cruciferae

| | | | | |
|---|---|---|---|---|
| *Cardamine pratensis* | Cuckooflower | | R/ C-S-R | 5 |
| *Coronopus squamatus* | Swinecress | | R/ C-S-R | 3 |

## Order Ericales
## Family Ericaceae

| | | | | |
|---|---|---|---|---|
| *Erica cinerea* | Bell-Heather | | S | 2 |
| *Myosotis arvensis* | Field Forget-me-not | | R/ S-R | 3 |

## Class Monocotyledoneae

## Order Liliaflorae
## Family Lilaceae

| | | | | |
|---|---|---|---|---|
| *Hyacinthoides non-scripta* | Bluebell | yes | S-R/ C-S-R | 6 |
| *Narcissus pseudonarcissus* | Wild Daffodil | | S-R/ C-S-R | 6 |

## Family Juncaceae

| | | | | |
|---|---|---|---|---|
| *Juncus bufonius* | Toad Rush | | R | 3 |
| *Juncus bulbosis* | Bulbous Rush | | S-R/ C-S-R | 8 |
| *Juncus effusus* | Soft Rush | | C/ S-C | 8 |
| *Luzula campestris* | Field Woodrush | | S | 8 |
| *Luzula multiflora* | Heath Woodrush | | S | 8 |
| *Luzula pilosa* | Hairy Woodrush | yes | S | 8 |

### Family Orchidaceae

| | | | | |
|---|---|---|---|---|
| *Dactylorhiza fuchsii* | Common Spotted Orchid | | S/ C-S-R | 5 |

### Order Cyperales
### Family Cyperaceae

| | | | | |
|---|---|---|---|---|
| *Carex pendula* | Pendulous Sedge | yes | S/ S-C | 8 |
| *Carex sylvatica* | Wood Sedge | | S/ S-C | 8 |

### Order Glumiflora
### Family Gramineae

| | | | | |
|---|---|---|---|---|
| *Agrostis canina* | Brown Bent Grass | | C-S-R | 8 |
| *Agrostis capillaris* | Common Bent Grass | | C-S-R | 8 |
| *Agrostis stolonifera* | Creeping Bent Grass | | C-R | 8 |
| *Anthoxanthum odoratum* | Sweet Vernal Grass | | S-R/ C-S-R | 8 |
| *Arrhenatherum elatius* | False Oat Grass | | C | 8 |
| *Brachypodium sylvatica* | Wood -False Brome | | S/ S-C | 8 |
| *Bromus ramosus* | Wood Brome Grass | | C-S-R | 8 |
| *Cynosurus cristatus* | Crested Dogs Tail | | C-S-R | 8 |
| *Dactylis glomerata* | Cocksfoot | | C-S-R | 8 |
| *Deschampsia caepitosa* | Tufted Hair Grass | | C-S-R | 8 |
| *Festuca rubra* | Red Fescue | | C-S-R | 8 |
| *Holcus lanatus* | Yorkshire Fog | | C-S-R | 8 |
| *Hordeum secalinum* | Meadow Barley | | C-S-R | 7 |
| *Lolium perenne* | Common Rye Grass | | C-R/ C-S-R | 8 |
| *Milium effusum* | Wood Millet | yes | C-S-R | 8 |
| *Phleum pratense* | Timothy Grass (Catstail) | | C-S-R/ C-R | 8 |
| *Poa annua* | Annual Meadow Grass | | C-S-R | 8 |
| *Poa nemoralis* | Wood Meadow Grass | yes | S/ C-S-R | 8 |
| *Poa pratensis* | Smooth Stalked Meadow Grass | | C-S-R | 8 |
| *Poa trivialis* | Rough Stalked Meadow Grass | | C-S-R/ C-R | 8 |
| *Vulpia bromoides* | Squirrel-Tail Fescue | | S-R | 7 |

**Appendix 2:** Complete species list incorporating **m**ean % covers for each species present per treatment, per season within grassland habitats during data collections i
**2003**, (**Chapter 2**). **RU** = Rooted Unfenced, **N-R U** = Non-Rooted Unfenced, **RF** = Rooted Fenced, **N-R F** = Non-Rooted Fenced. * = species not present.

| | | **Habitat** | **Grassland** | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | **Season/Year** | **Spring 2002** | | | | **Summer 2002** | | | | **Spring 2003** | | | | **Summer 2003** | | | |
| | | **Treatment** | **R U** | **N-R U** | **R F** | **N-R F** | **R U** | **N-R U** | **R F** | **N-R F** | **R U** | **N-R U** | **R F** | **N-R F** | **R U** | **N-R U** | **R F** | **N-R F** |
| **1** | *Acer campestre* | Field Maple Seedling | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **2** | *Agrostis canina* | Brown Bent Grass | * | 4.25 | 0.25 | 9.50 | 4.65 | 2.75 | 3.50 | 8.50 | 5.25 | 3.50 | 5.55 | 7.00 | 9.25 | 2.50 | 11.65 | 11.40 |
| **3** | *Agrostis capillaris* | Common Bent Grass | * | * | * | * | * | 0.25 | 0.50 | * | 1.25 | 0.50 | 0.75 | * | 0.75 | 0.40 | 0.50 | * |
| **4** | *Agrostis stolonifera* | Creeping Bent Grass | * | 7.25 | * | 3.25 | * | 7.25 | * | 3.25 | * | 7.50 | 0.50 | 3.25 | * | 6.50 | 1.00 | 3.50 |
| **5** | *Ajuga reptans* | Bugle | 0.50 | 0.50 | 0.50 | 0.10 | 2.50 | 0.80 | 2.45 | * | 2.40 | 1.40 | 2.50 | 0.10 | 1.40 | 1.00 | 2.00 | 0.15 |
| **6** | *Anagallis arvensis* | Scarlet Pimpernel | * | * | 1.50 | * | 4.35 | * | 8.35 | * | 2.00 | * | 1.15 | * | 0.75 | * | 0.10 | * |
| **7** | *Anemone nemorosa* | Wood Anemone | * | * | * | * | * | * | * | * | * | 0.15 | 0.60 | * | * | * | * | * |
| **8** | *Anthoxanthum odoratum* | Sweet Vernal Grass | 0.50 | 13.75 | 1.25 | 14.25 | 1.00 | 12.75 | 2.00 | 14.25 | 4.85 | 11.00 | 5.50 | 9.75 | 1.90 | 11.65 | 3.00 | 14.25 |
| **9** | *Arctium Lappa L.* | Greater Burdock | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **10** | *Arrhenatherum Elatius* | False Oat Grass | * | * | * | 0.75 | 0.75 | * | 3.50 | 1.00 | 2.65 | * | 5.25 | 1.75 | 1.00 | 0.20 | 5.00 | 2.15 |
| **11** | *Atrichum undulatum* | Common Smooth Cap | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **12** | *Bellis perennis L.* | Common Daisy | * | 0.05 | * | 0.65 | * | * | * | 0.30 | * | * | * | * | * | * | * | * |
| **13** | *Betonica officinalis* | Betony | * | * | * | * | * | * | 0.15 | * | 0.20 | * | 0.75 | * | * | * | * | * |
| **14** | *Betula pendula* | Silver Birch Seedling | * | * | * | * | * | * | 0.35 | 0.15 | 0.75 | 0.15 | 0.65 | * | 0.80 | 0.10 | 0.70 | 0.20 |
| **15** | *Betula pendula* | Silver Birch Sapling | * | * | * | * | * | * | * | * | * | * | * | * | 0.20 | * | 1.40 | * |
| **16** | *Brachypodium sylvatica* | Wood False Brome | * | 2.50 | * | 3.90 | * | 2.50 | 1.00 | 4.00 | * | 2.00 | 0.50 | 4.00 | 0.25 | 2.50 | 0.75 | 4.25 |
| **17** | *Brachythecium* | Rough Stalked Feather | 1.7 | 24.50 | 2.5 | 22.6 | 7.2 | 26.90 | 13.15 | 22.7 | 13. | 22.90 | 21. | 24.0 | 10. | 22.65 | 17. | 23.5 |

|  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
|  | *rutabulum* | Moss | 5 |  | 0 | 5 |  | 5 |  |  | 5 |  | 50 |  | 65 | 0 |  | 15 |  | 65 | 0 |
| **18** | *Bromus ramosus* | Wood Brome Grass | * | * | 0.50 | * |  | * | * | * | * |  | * | * | * | * |  | * | * | * | * |
| **19** | *Calliergon cuspidatum* | Pointed Spear Moss | * | 1.25 | 0.25 | 2.25 |  | 0.75 | 1.25 | 2.75 | 2.75 |  | 2.00 | 1.25 | 5.25 | 3.45 |  | 1.75 | 1.25 | 5.00 | 3.00 |
| **20** | *Calluna Vulgaris* | Ling | * | * | * | * |  | * | 0.20 | * | * |  | * | * | * | * |  | * | * | * | * |
| **21** | *Cardamine pratensis* | Cuckooflower | * | * | * | * |  | 0.20 | * | 0.20 | * |  | * | * | * | * |  | * | * | * | * |
| **22** | *Carex pendula* | Pendulous Sedge | * | * | * | * |  | * | * | * | * |  | * | * | * | * |  | * | * | * | * |
| **23** | *Carex sylvatica* | Wood Sedge | * | * | * | * |  | * | * | * | * |  | * | * | * | * |  | * | * | * | * |
| **24** | *Carpinus betulus* | Hornbeam Seedling | * | 0.10 | 0.20 | 0.65 |  | 0.40 | 0.15 | 0.85 | 0.45 |  | 3.05 | 0.20 | 1.70 | 0.85 |  | 1.80 | * | 0.80 | 0.75 |
| **25** | *Carpinus betulus* | Hornbeam Sapling | * | * | * | * |  | * | * | * | * |  | * | * | * | * |  | * | * | 0.65 | 0.20 |
| **26** | *Castanea Sativa* | Sweet Chestnut Seedling | * | * | * | * |  | * | * | * | * |  | 0.10 | 0.10 | * | * |  | * | * | * | * |
| **27** | *Centaurea nigra* | Common knapweed | * | * | * | * |  | * | * | * | * |  | * | * | * | * |  | * | * | * | * |
| **28** | *Centaurium erythraea* | Common Centaury | 0.35 | * | 0.15 | 0.25 |  | * | * | 0.25 | 0.15 |  | 0.50 | * | 0.35 | * |  | 2.00 | * | 0.90 | 0.10 |
| **29** | *Centaurium pulchellum* | Lesser Centaury | * | * | * | * |  | 0.55 | * | * | * |  | * | * | * | * |  | * | * | * | * |
| **30** | *Cerastium arvense* | Field Mouse-ear | * | * | * | * |  | * | * | * | * |  | * | * | * | * |  | * | * | * | * |
| **31** | *Cerastium holosteoides* | Common Mouse Ear | * | 1.65 | 0.35 | 2.05 |  | 1.75 | 1.30 | 2.80 | 0.70 |  | 3.70 | 0.45 | 4.45 | 0.75 |  | 2.65 | 0.10 | 4.85 | 1.45 |
| **32** | *Chamaenerion angustifolium* | Rosebay Willowherb | * | * | * | * |  | * | * | 0.50 | * |  | * | * | 0.25 | * |  | * | * | 0.40 | * |
| **33** | *Chenopodium botryodes* | Small Red Goosefoot | * | * | * | * |  | * | * | * | * |  | * | * | * | * |  | * | * | * | * |
| **34** | *Chenopodium polyspermum* | Many-Seeded Goosefoot | * | * | * | * |  | 0.90 | * | 2.85 | * |  | 0.20 | * | 0.15 | * |  | * | * | * | * |
| **35** | *Chenopodium rubrum* | Red Goosefoot | * | * | * | * |  | * | * | * | * |  | * | * | * | * |  | * | * | * | * |
| **36** | *Circaea lutetiana* | Enchanter's Nightshade | * | * | * | * |  | * | * | * | * |  | 0.15 | * | * | * |  | * | * | * | * |
| **37** | *Cirsium arvense* | Creeping Thistle | 7.20 | 3.55 | 5.80 | 1.10 |  | 22.90 | 4.65 | 27.50 | 5.55 |  | 18.90 | 8.85 | 19.25 | 9.00 |  | 20.90 | 12.60 | 25.25 | 13.30 |
| **38** | *Cirsium palustre* | Marsh Thistle | 0.25 | 0.35 | 0.15 | 0.10 |  | 5.60 | 1.90 | 6.10 | 1.40 |  | 2.60 | 2.15 | 3.25 | 2.75 |  | 5.90 | 1.05 | 5.90 | 1.00 |
| **39** | *Cirsium Vulgare* | Spear Thistle | * | * | * | 0.75 |  | 0.80 | * | 1.45 | 0.75 |  | 0.35 | * | 3.80 | 0.85 |  | 0.90 | * | 2.90 | 0.90 |
| **40** | *Conopodium majus* | Pignut | * | * | * | * |  | * | * | * | * |  | * | * | * | * |  | * | * | * | * |
| **41** | *Coronopus squamatus* | Swinecress | * | * | * | * |  | * | * | * | * |  | * | * | * | * |  | * | * | * | * |
| **42** | *Corylus avellana* | Hazel Seedling | * | * | * | * |  | * | * | * | * |  | * | * | * | * |  | * | * | * | * |
| **43** | *Corylus avellana* | Hazel Sapling | * | * | * | * |  | * | * | * | * |  | * | * | * | * |  | * | * | * | * |

| | | Habitat | Grassland | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | Season/Year | Spring 2002 | | | | Summer 2002 | | | | Spring 2003 | | | | Summer 2003 | | | |
| | | Treatment | R U | N-R U | R F | N-R F | R U | N-R U | R F | N-R F | R U | N-R U | R F | N-R F | R U | N-R U | R F | N-R F |
| **44** | *Crataegus monogyna* | Hawthorn Seedling | * | * | * | * | * | * | * | * | 0.20 | * | * | * | * | * | * | * |
| **45** | *Cynosurus cristatus* | Crested Dogs Tail | 0.50 | * | 0.65 | 3.00 | 1.00 | * | 2.00 | 3.00 | 2.25 | * | 1.75 | 1.75 | 1.50 | * | 1.50 | 1.50 |
| **46** | *Dactylis glomerata* | Cocksfoot | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **47** | *Dactylorhiza fuchsii* | Common Spotted Orchid | * | * | * | 0.15 | * | * | * | * | * | * | * | 1.00 | * | * | * | * |
| **48** | *Deschampsia caespitosa* | Tufted Hair-grass | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **49** | *Dicranum Scoparium* | Broom Fork Moss | * | * | * | * | * | * | * | * | 0.50 | * | * | * | * | * | * | * |
| **50** | *Dicranella heteromalla* | Silky Forklet Moss | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **51** | *Digitalis purpurea* | Foxglove | * | * | * | * | 0.50 | * | 1.50 | * | 1.00 | * | 4.75 | * | 1.25 | * | 4.75 | * |
| **52** | *Dryopteris austriaca* | Broad Buckler Fern | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **53** | *Dryopteris filix-mas* | Common Male Fern | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **54** | *Epilobium hirsutum* | Great Willow Herb | * | * | 0.25 | 0.10 | * | * | * | * | * | * | * | * | * | * | * | * |
| **55** | *Epilobium montanum* | Broad-Leaved Willowherb | * | * | * | * | 0.35 | * | 0.25 | * | 1.25 | * | 0.55 | * | 0.75 | * | 0.75 | * |
| **56** | *Epilobium obscurum* | Short Fruited Willowherb | * | * | * | * | * | * | * | * | 0.10 | * | * | * | * | * | 0.25 | * |
| **57** | *Epilobium parviflorum* | Hoary Willowherb | * | * | * | * | 1.45 | * | 1.20 | 0.10 | 1.80 | * | 1.25 | * | 2.60 | * | 3.15 | 0.50 |
| **58** | *Epilobium tetragonum* | Square Stalked Willow Herb | 0.55 | * | 0.15 | * | 0.75 | * | 1.65 | * | 0.75 | 0.10 | 1.45 | * | 0.20 | * | 0.25 | * |
| **59** | *Erica cinerea L.* | Bell-Heather | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **60** | *Euphrasia nemorosa* | Eyebright | * | * | * | 0.10 | 0.15 | 0.15 | 0.20 | 2.00 | 0.55 | * | * | * | * | 0.50 | 0.75 | 0.75 |
| **61** | *Eurhynchium Praelongum* | Common Feather Moss | * | * | * | * | 0.50 | * | * | * | 1.50 | * | 1.00 | * | 1.00 | * | * | * |
| **62** | *Eurhynchium swartzii* | Swartz's Feather Moss | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **63** | *Fagus sylvatica* | Beech Seedling | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **64** | *Festuca Rubra* | Red Fescue | * | 3.00 | * | * | 0.25 | 3.00 | * | * | * | 2.75 | * | * | 1.00 | 3.50 | 0.50 | * |
| **65** | *Fissidens taxifolius* | Common Pocket Moss | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **66** | *Fraxinus excelsior* | Ash Seedling | 0.10 | 0.10 | 0.50 | * | 0.65 | 0.20 | 0.50 | * | 1.00 | * | 0.10 | 0.10 | 0.45 | * | 0.25 | * |
| **67** | *Fraxinus excelsior* | Ash Sapling | * | * | * | * | * | * | * | * | * | * | * | * | 0.2 | * | * | * |

| | | | | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | | | | | | | | | | 5 | | | |
| **68** | *Galium aparine* | Cleavers, Goosegrass | * | * | * | * | 0.20 | * | * | * | * | * | * | * | * | * | * | * |
| **69** | *Galium Palustre* | Common Marsh Bedstraw | * | * | * | * | * | * | * | * | 0.65 | * | 0.20 | * | 0.25 | * | * | * |
| **70** | *Glechoma hederacea* | Ground Ivy | 0.15 | 1.10 | 1.00 | 0.50 | 1.20 | 0.55 | 6.50 | 1.95 | 2.25 | 1.50 | 4.90 | 2.00 | 3.90 | 1.75 | 5.75 | 2.05 |
| **71** | *Gnaphalium uliginosum* | Marsh Cudweed | * | * | * | * | 0.80 | * | 0.40 | * | * | * | * | * | 0.75 | * | 0.25 | * |
| **72** | *Hedera helix* | Ivy | * | * | * | * | * | * | * | * | * | * | * | * | 0.15 | * | * | * |
| **73** | *Heracleum Sphondylium* | Hog Weed | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **74** | *Holcus lanatus* | Yorkshire Fog | 0.25 | 21.40 | 1.40 | 20.25 | 7.40 | 21.00 | 17.20 | 19.75 | 11.55 | 19.25 | 23.35 | 26.20 | 12.30 | 22.05 | 21.80 | 21.50 |
| **75** | *Hordeum Secalinum* | Meadow Barley | * | * | * | 0.50 | * | * | * | 0.50 | 1.25 | 0.25 | 0.25 | * | * | * | * | * |
| **76** | *Hyacinthoides non-scripta* | Bluebell | * | * | * | * | * | * | * | * | 0.35 | * | 0.50 | * | * | * | * | * |
| **77** | *Hypericum hirsutum* | Hairy St. Johns-wort | * | * | * | * | 0.50 | * | 0.30 | * | 0.85 | 0.15 | 0.80 | 0.15 | * | * | 0.25 | * |
| **78** | *Hypericum humafusum* | Trailing St. Johns-wort | * | * | * | * | * | * | * | * | * | * | * | * | 1.00 | * | 0.70 | * |
| **79** | *Hypericum maculatum* | Imperforate St. Johns-Wort | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **80** | *Hypericum perforatum* | Perforate St. Johns-wort | * | * | * | * | 0.20 | * | 0.90 | 0.25 | 1.65 | * | 3.10 | 0.25 | 1.15 | * | 1.60 | * |
| **81** | *Hypnum jutlandicum* | Heath Plait Moss | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **82** | *Ilex aquifolium* | Holly Seedling | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **83** | *Juncus bufonius* | Toad Rush | * | * | * | * | 0.50 | * | 1.55 | * | 2.05 | * | 0.50 | * | * | * | * | * |
| **84** | *Juncus bulbosis* | Bulbous Rush | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **85** | *Juncus effusus* | Soft Rush | * | * | * | 1.50 | * | * | 0.40 | 2.40 | 0.75 | * | 0.50 | 2.50 | * | * | 0.50 | 3.50 |
| **86** | *Kickxia elatine* | Sharp leaved Fluellen | * | * | * | * | 2.25 | * | * | * | * | * | * | * | 1.05 | * | 0.40 | * |

| | | **Habitat** | **Grassland** | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | **Season/Year** | **Spring 2002** | | | | **Summer 2002** | | | | **Spring 2003** | | | | **Summer 2003** | | | |
| | | **Treatment** | **R U** | **N-R U** | **R F** | **N-R F** | **R U** | **N-R U** | **R F** | **N-R F** | **R U** | **N-R U** | **R F** | **N-R F** | **R U** | **N-R U** | **R F** | **N-R F** |
| **87** | *Lamiastrum galeobdolon* | Yellow Archangel | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **88** | *Lolium perenne* | Common Rye Grass | * | 0.75 | * | 1.75 | 0.50 | 1.75 | 2.00 | 1.75 | 0.50 | 0.90 | 1.50 | 1.75 | 1.00 | 0.90 | 2.25 | 1.40 |

| | | | | | | | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **89** | *Lonicera periclymenum* | Honeysuckle | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **90** | *Lotus corniculatus* | Common Bird's-Foot Trefoil | * | 3.00 | 0.25 | 5.30 | | 1.30 | 4.75 | 0.70 | 5.35 | | 0.15 | 2.75 | 0.25 | 3.40 | | 1.35 | 1.05 | 0.50 | 1.40 |
| **91** | *Luzula campestris* | Field Woodrush | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **92** | *Luzula multiflora* | Heath Woodrush | * | * | * | 0.50 | | * | * | * | 0.50 | | 0.10 | 0.75 | 0.75 | * | | * | * | * | * |
| **93** | *Luzula pilosa* | Hairy Wood-rush | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **94** | *Lychnis flos-cuculi* | Ragged Robin | * | * | * | * | | * | * | * | * | | 0.75 | * | * | * | | * | * | * | * |
| **95** | *Lysimachia nemorum* | Yellow Pimpernel | * | * | * | * | | * | * | * | * | | 0.25 | * | * | * | | * | * | * | * |
| **96** | *Medicago lupulina* | Black Medick | * | 3.00 | * | 1.10 | | 0.25 | 3.15 | * | 1.25 | | 0.50 | 1.50 | 0.10 | 0.25 | | * | 2.50 | * | 0.50 |
| **97** | *Mentha arvensis* | Corn Mint | * | 0.50 | * | 1.00 | | 0.90 | 1.00 | 0.40 | 2.50 | | 0.85 | 1.00 | 1.40 | 2.25 | | 2.15 | 3.75 | 2.95 | 2.50 |
| **98** | | Unidentified Moss | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **99** | *Mercurialis perennis* | Dog Mercury | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | 0.25 | * |
| **100** | *Milium effusum* | Wood Millet | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **101** | *Mnium Hornum* | Swan's Neck Thyme Moss | * | * | * | * | | * | * | * | * | | 0.45 | * | * | * | | * | * | * | * |
| **102** | *Moehringia trinervia* | Three Nerved Sandwort | * | * | * | * | | * | * | * | * | | 0.40 | * | * | * | | * | * | * | * |
| **103** | *Montia fontana* | Blinks | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **104** | *Montia perfoliata* | Spring Beauty | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **105** | *Myosotis arvensis* | Field Forget-me-not | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **106** | *Oxalis acetosella* | Wood Sorrel | * | * | * | * | | 0.15 | * | * | * | | * | * | * | * | | * | * | 0.20 | * |
| **107** | *Pedicularis sylvatica* | Lousewort | * | * | * | * | | 0.20 | * | * | * | | * | * | * | * | | * | * | * | * |
| **108** | *Phleum pratense* | Timothy Grass (Catstail) | * | * | * | 0.25 | | * | * | 0.50 | 0.25 | | * | * | 0.40 | 0.25 | | 0.25 | 1.00 | 0.25 | 0.25 |
| **109** | *Pilosella officinarum* | Mouse-ear Hawkweed | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **110** | *Plagiomnium undulatum* | Hart's Tongue Thyme Moss | * | * | * | * | | * | * | 1.00 | * | | * | * | 1.00 | * | | * | * | 0.75 | * |
| **111** | *Plantago major* | Greater Plantain | * | * | * | * | | 1.30 | * | 0.90 | * | | 1.60 | * | 1.15 | 0.15 | | 1.45 | * | 1.40 | * |
| **112** | *Poa annua* | Annual Meadow Grass | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **113** | *Poa Nemoralis* | Wood Meadow Grass | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **114** | *Poa pratensis* | Smooth Stalked Meadow Grass | * | * | 0.75 | * | | * | * | 2.75 | * | | 1.25 | * | 4.00 | * | | * | * | 3.00 | 0.25 |
| **115** | *Poa trivialis* | Rough Stalked Meadow Grass | 1.65 | 27.50 | 0.60 | 27.10 | | 6.00 | 28.00 | 9.65 | 28.00 | | 15.50 | 24.50 | 16.75 | 28.05 | | 12.55 | 24.00 | 12.05 | 29.00 |
| **116** | *Polygala serpyllifolia* | Heath Milkwort | * | * | * | * | | * | 0.25 | * | 0.40 | | * | * | * | 0.10 | | * | * | * | * |
| **117** | *Polygala Vulgaris* | Common Milkwort | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |

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|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **118** | *Polytrichum formosum* | Bank Haircap Moss | * | 0.75 | * | * | 0.40 | 0.40 | 0.50 | * | 0.90 | * | 0.75 | * | 0.50 | 0.50 | 0.75 | * |
| **119** | *Potentilla anserina* | Silver Weed | * | * | 0.25 | * | * | 1.00 | * | * | * | * | 2.00 | * | * | * | 1.75 | * |
| **120** | *Potentilla erecta* | Tormentil | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **121** | *Potentilla reptans* | Creeping Cinquefoil | * | * | * | * | 0.15 | * | 0.15 | * | * | * | * | 0.25 | 0.25 | 0.75 | 0.65 | 0.25 |
| **122** | *Potentilla sterilis* | Barren Strawberry | * | 0.50 | * | 1.00 | * | 0.50 | 0.40 | 1.15 | 0.25 | 3.15 | 0.95 | 3.50 | * | 1.05 | 1.40 | 1.00 |
| **123** | *Pottia trunchator* | Common Pottia Moss | * | * | * | * | 5.75 | * | 13.50 | * | 1.75 | * | 6.75 | * | 7.50 | * | 15.25 | * |
| **124** | *Primula vulgaris* | Primrose | * | 0.10 | * | 0.25 | * | * | * | 0.25 | * | 0.25 | 0.20 | * | * | * | * | 0.40 |
| **125** | *Prunella vulgaris* | Selfheal | 0.70 | 0.50 | 0.85 | 2.00 | 3.20 | 1.15 | 4.00 | 1.90 | 6.00 | 1.05 | 10.05 | 1.20 | 3.80 | 0.60 | 4.35 | 0.60 |
| **126** | *Prunus avium* | Wild Cherry Seedling | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **127** | *Prunus spinosa* | Sloe Seedling / Blackthorn | * | * | * | * | 0.25 | * | 0.65 | * | * | * | * | * | * | * | 0.25 | * |
| **128** | *Pseudoscleropodium purum* | Neat Feather Moss | 0.25 | 7.75 | 1.25 | 5.25 | 1.00 | 3.90 | 5.00 | 5.25 | 3.25 | 7.50 | 5.25 | 5.25 | 2.00 | 8.00 | 6.50 | 5.25 |
| **129** | *Pteridium aquilinum* | Bracken | * | * | * | 3.00 | * | * | * | 5.75 | * | * | * | 4.50 | * | 0.15 | * | 6.50 |

| | | **Habitat** | **Grassland** | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | **Season/Year** | **Spring 2002** | | | | **Summer 2002** | | | | **Spring 2003** | | | | **Summer 2003** | | | |
| | | **Treatment** | **R U** | **N-R U** | **R F** | **N-R F** | **R U** | **N-R U** | **R F** | **N-R F** | **R U** | **N-R U** | **R F** | **N-R F** | **R U** | **N-R U** | **R F** | **N-R F** |
| **130** | *Pulicaria dysenterica* | Common Fleabane | * | * | * | * | * | * | 0.25 | 0.15 | 0.25 | * | 0.10 | * | 0.50 | * | 0.50 | 0.75 |
| **131** | *Quercus robur* | Oak Sapling | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **132** | *Quercus robur* | Pedunculate Oak Seedling | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **133** | *Ranunculus parviflorus* | Small-flowered Buttercup | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **134** | *Ranunculus ficaria* | Lesser Celendine | * | * | * | * | * | * | * | * | 0.60 | * | * | * | * | * | * | * |
| **135** | *Ranunculus flammula* | Lesser Spearwort | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **136** | *Ranunculus repens* | Creeping Buttercup | 2.30 | 11.50 | 4.05 | 14.25 | 7.05 | 11.30 | 18.65 | 14.55 | 19.30 | 13.10 | 30.00 | 19.60 | 12.45 | 8.65 | 24.35 | 16.00 |
| **137** | *Rhytidiadelphus squarrosus* | Springy Tough Moss | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **138** | *Rosa canina* | Common Dog Rose | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **139** | *Rubus fruticosus* | Bramble | * | * | * | 0.25 | 1.05 | * | 0.50 | 0.35 | 1.20 | * | 1.45 | * | 2.40 | 0.10 | 3.75 | 0.50 |

| | | | | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **140** | *Rumex acetosa* | Common Sorrel | * | * | * | * | * | 0.25 | * | * | * | * | * | * | * | * | * | * |
| **141** | *Rumex acetosella* | Sheep Sorrel | * | * | * | * | 0.75 | * | * | * | * | * | * | * | 0.45 | * | * | * |
| **142** | *Rumex obtusifolius* | Broad leaved Dock | * | * | * | * | * | * | * | * | * | * | * | * | 0.15 | * | 0.15 | * |
| **143** | *Rumex sanguineus* | Wood Dock | * | * | * | * | 0.10 | * | 0.75 | * | * | * | 0.45 | * | * | * | 0.50 | * |
| **144** | *Sagina apetala* | Common Annual Pearlwort | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **145** | *Sagina procumbens* | Procumbent Pearlwort | * | * | * | * | 6.30 | * | 5.25 | * | 3.75 | 0.25 | 4.55 | * | * | * | * | * |
| **146** | *Salix caprea* | Goat Willow Sapling | * | * | * | * | * | * | * | * | * | * | * | * | * | * | 0.50 | * |
| **147** | *Sambucus nigra* | Elder Seedling | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **148** | *Sarothamnus scoparius* | Broom | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **149** | *Senecio jacobaea* | Common Ragwort | * | * | * | * | 1.90 | 0.50 | 2.75 | 0.15 | 2.00 | 0.75 | 5.25 | 0.35 | 2.85 | 0.75 | 5.00 | * |
| **150** | *Silene dioica* | Red Campion | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **151** | *Sonchus oleraceus* | Smooth Sow Thistle | * | * | * | * | * | * | 0.50 | * | * | * | * | * | * | * | * | * |
| **152** | *Stachys arvensis* | Field Woundwort | * | * | * | * | * | * | * | * | * | * | 0.20 | * | * | * | 0.15 | * |
| **153** | *Stachys sylvatica* | Hedge Woundwort | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | 0.20 |
| **154** | *Stellaria graminea* | Lesser Stitchwort | * | 2.00 | * | 0.85 | 0.15 | 1.00 | * | 1.65 | 0.15 | 1.15 | * | 1.70 | 0.40 | 0.15 | * | 2.25 |
| **155** | *Stellaria media* | Common Chickweed | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **156** | *Stellaria neglecta* | Greater Chickweed | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **157** | *Taraxacum officinale* | Dandelion | * | * | * | 0.25 | * | * | * | * | 0.20 | 0.20 | * | 0.20 | * | * | * | 0.20 |
| **158** | *Teucrium scorodonia* | Wood Sage | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **159** | *Thuidium tamariscinum* | Common Tamarisk Moss | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **160** | *Trifolium dubium* | Lesser Trefoil | * | * | * | 0.60 | * | * | * | 0.15 | 0.20 | * | * | 0.15 | * | 0.40 | 0.40 | * |
| **161** | *Trifolium micranthum* | Slender Trefoil | * | * | 0.05 | 1.75 | * | * | * | 0.50 | 0.15 | * | 0.15 | 0.25 | * | 0.50 | * | 0.35 |
| **162** | *Trifolium repens* | White Clover | 0.25 | 8.75 | 0.50 | 15.60 | 1.10 | 7.85 | 3.25 | 12.85 | 4.20 | 15.15 | 4.80 | 10.30 | 2.50 | 3.00 | 2.45 | 3.55 |
| **163** | *Tussilago farfara* | Coltsfoot | * | * | * | * | 0.50 | * | * | * | 1.00 | * | * | * | 0.75 | * | * | * |
| **164** | *Urtica dioica* | Stinging Nettle | 0.35 | * | 0.25 | 0.35 | 2.00 | * | 3.40 | 0.40 | 2.40 | * | 3.90 | 0.65 | 2.75 | * | 2.90 | 0.55 |
| **165** | *Veronica arvensis* | Wall Speedwell | * | * | * | 0.90 | * | * | * | * | 0.75 | * | 0.25 | * | * | * | * | * |
| **166** | *Veronica montana* | Wood Speedwell | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **167** | *Veronica officinalis* | Heath Speedwell | * | * | * | * | 0.10 | * | * | * | * | * | * | * | 2.65 | * | 1.75 | * |

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|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **168** | *Veronica persica* | Common Field Speedwell | * | * | * | * | | * | * | * | * | | 1.15 | * | 1.70 | * | | * | * | * | * |
| **169** | *Veronica serpyllifolia* | Thyme Leaved Speedwell | * | 0.75 | * | * | | * | * | * | * | | 0.50 | * | 0.50 | * | | * | * | * | * |
| **170** | *Vicia sativa* | Common Vetch | * | 0.25 | * | 0.60 | | * | * | * | * | | * | * | * | 0.25 | | * | * | * | * |
| **171** | *Viola lactea* | Pale Dog Violet | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **172** | *Viola riviniana* | Common Dog Violet | 0.25 | 0.55 | 0.15 | 0.75 | | 2.15 | 0.90 | 1.35 | 2.00 | | 0.70 | 2.05 | 2.25 | 2.75 | | 1.50 | 1.80 | 2.25 | 1.50 |
| **173** | *Vulpia bromoides* | Squirrel-Tail Fescue | * | 1.25 | * | 0.50 | | 0.95 | 1.25 | 1.00 | * | | 2.65 | 1.50 | 1.25 | 0.40 | | 2.50 | 1.25 | 1.25 | * |

**Appendix 2:** Mean % covers for each species present per treatment, per season within grassland habitats during data collections in **2002** and **2003**, (**Chapter 2**). **RU**
Unfenced, **N-R U** = Non-Rooted Unfenced, **RF** = Rooted Fenced, **N-R F** = Non-Rooted Fenced. * = species not present.

| | | **Habitat** | **Grassland** | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | **Season/Year** | **Spring 2002** | | | | **Summer 2002** | | | | **Spring 2003** | | | | **Summer 2003** | | | |
| | | **Treatment** | **R U** | **N-R U** | **R F** | **N-R F** | **R U** | **N-R U** | **R F** | **N-R F** | **R U** | **N-R U** | **R F** | **N-R F** | **R U** | **N-R U** | **R F** | **N-R F** |
| **1** | *Acer campestre* | Field Maple Seedling | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **2** | *Agrostis canina* | Brown Bent Grass | * | 4.25 | 0.25 | 9.50 | 4.65 | 2.75 | 3.50 | 8.50 | 5.25 | 3.50 | 5.55 | 7.00 | 9.25 | 2.50 | 11.65 | 11.40 |
| **3** | *Agrostis capillaris* | Common Bent Grass | * | * | * | * | * | 0.25 | 0.50 | * | 1.25 | 0.50 | 0.75 | * | 0.75 | 0.40 | 0.50 | * |
| **4** | *Agrostis stolonifera* | Creeping Bent Grass | * | 7.25 | * | 3.25 | * | 7.25 | * | 3.25 | * | 7.50 | 0.50 | 3.25 | * | 6.50 | 1.00 | 3.50 |
| **5** | *Ajuga reptans* | Bugle | 0.50 | 0.50 | 0.50 | 0.10 | 2.50 | 0.80 | 2.45 | * | 2.40 | 1.40 | 2.50 | 0.10 | 1.40 | 1.00 | 2.00 | 0.15 |
| **6** | *Anagallis arvensis* | Scarlet Pimpernel | * | * | 1.50 | * | 4.35 | * | 8.35 | * | 2.00 | * | 1.15 | * | 0.75 | * | 0.10 | * |
| **7** | *Anemone nemorosa* | Wood Anemone | * | * | * | * | * | * | * | * | * | 0.15 | 0.60 | * | * | * | * | * |
| **8** | *Anthoxanthum odoratum* | Sweet Vernal Grass | 0.50 | 13.75 | 1.25 | 14.25 | 1.00 | 12.75 | 2.00 | 14.25 | 4.85 | 11.00 | 5.50 | 9.75 | 1.90 | 11.65 | 3.00 | 14.25 |
| **9** | *Arctium Lappa L.* | Greater Burdock | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **10** | *Arrhenatherum Elatius* | False Oat Grass | * | * | * | 0.75 | 0.75 | * | 3.50 | 1.00 | 2.65 | * | 5.25 | 1.75 | 1.00 | 0.20 | 5.00 | 2.15 |
| **11** | *Atrichum undulatum* | Common Smooth Cap | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **12** | *Bellis perennis L.* | Common Daisy | * | 0.05 | * | 0.65 | * | * | * | 0.30 | * | * | * | * | * | * | * | * |
| **13** | *Betonica officinalis* | Betony | * | * | * | * | * | * | 0.15 | * | 0.20 | * | 0.75 | * | * | * | * | * |
| **14** | *Betula pendula* | Silver Birch Seedling | * | * | * | * | * | * | 0.35 | 0.15 | 0.75 | 0.15 | 0.65 | * | 0.80 | 0.10 | 0.70 | 0.20 |
| **15** | *Betula pendula* | Silver Birch Sapling | * | * | * | * | * | * | * | * | * | * | * | * | 0.20 | * | 1.40 | * |
| **16** | *Brachypodium sylvatica* | Wood False Brome | * | 2.50 | * | 3.90 | * | 2.50 | 1.00 | 4.00 | * | 2.00 | 0.50 | 4.00 | 0.25 | 2.50 | 0.75 | 4.25 |
| **17** | *Brachythecium rutabulum* | Rough Stalked Feather Moss | 1.75 | 24.50 | 2.50 | 22.65 | 7.25 | 26.90 | 13.15 | 22.75 | 13.50 | 22.90 | 21.65 | 24.00 | 10.15 | 22.65 | 17.65 | 23.50 |
| **18** | *Bromus ramosus* | Wood Brome Grass | * | * | 0.50 | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **19** | *Calliergon cuspidatum* | Pointed Spear Moss | * | 1.25 | 0.25 | 2.25 | 0.75 | 1.25 | 2.75 | 2.75 | 2.00 | 1.25 | 5.25 | 3.45 | 1.75 | 1.25 | 5.00 | 3.00 |
| **20** | *Calluna Vulgaris* | Ling | * | * | * | * | * | 0.20 | * | * | * | * | * | * | * | * | * | * |
| **21** | *Cardamine pratensis* | Cuckooflower | * | * | * | * | 0.20 | * | 0.20 | * | * | * | * | * | * | * | * | * |
| **22** | *Carex pendula* | Pendulous Sedge | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |

| | | | | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **23** | *Carex sylvatica* | Wood Sedge | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **24** | *Carpinus betulus* | Hornbeam Seedling | * | 0.10 | 0.20 | 0.65 | 0.40 | 0.15 | 0.85 | 0.45 | 3.05 | 0.20 | 1.70 | 0.85 | 1.80 | * | 0.80 | 0.75 |
| **25** | *Carpinus betulus* | Hornbeam Sapling | * | * | * | * | * | * | * | * | * | * | * | * | * | * | 0.65 | 0.20 |
| **26** | *Castanea Sativa* | Sweet Chestnut Seedling | * | * | * | * | * | * | * | * | 0.10 | 0.10 | * | * | * | * | * | * |
| **27** | *Centaurea nigra* | Common knapweed | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **28** | *Centaurium erythraea* | Common Centaury | 0.35 | * | 0.15 | 0.25 | * | * | 0.25 | 0.15 | 0.50 | * | 0.35 | * | 2.00 | * | 0.90 | 0.10 |
| **29** | *Centaurium pulchellum* | Lesser Centaury | * | * | * | * | 0.55 | * | * | * | * | * | * | * | * | * | * | * |
| **30** | *Cerastium arvense* | Field Mouse-ear | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **31** | *Cerastium holosteoides* | Common Mouse Ear | * | 1.65 | 0.35 | 2.05 | 1.75 | 1.30 | 2.80 | 0.70 | 3.70 | 0.45 | 4.45 | 0.75 | 2.65 | 0.10 | 4.85 | 1.45 |
| **32** | *Chamaenerion angustifolium* | Rosebay Willowherb | * | * | * | * | * | * | 0.50 | * | * | * | 0.25 | * | * | * | 0.40 | * |
| **33** | *Chenopodium botryodes* | Small Red Goosefoot | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **34** | *Chenopodium polyspermum* | Many-Seeded Goosefoot | * | * | * | * | 0.90 | * | 2.85 | * | 0.20 | * | 0.15 | * | * | * | * | * |
| **35** | *Chenopodium rubrum* | Red Goosefoot | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **36** | *Circaea lutetiana* | Enchanter's Nightshade | * | * | * | * | * | * | * | * | 0.15 | * | * | * | * | * | * | * |
| **37** | *Cirsium arvense* | Creeping Thistle | 7.20 | 3.55 | 5.80 | 1.10 | 22.90 | 4.65 | 27.50 | 5.55 | 18.90 | 8.85 | 19.25 | 9.00 | 20.90 | 12.60 | 25.25 | 13.30 |
| **38** | *Cirsium palustre* | Marsh Thistle | 0.25 | 0.35 | 0.15 | 0.10 | 5.60 | 1.90 | 6.10 | 1.40 | 2.60 | 2.15 | 3.25 | 2.75 | 5.90 | 1.05 | 5.90 | 1.00 |
| **39** | *Cirsium Vulgare* | Spear Thistle | * | * | * | 0.75 | 0.80 | * | 1.45 | 0.75 | 0.35 | * | 3.80 | 0.85 | 0.90 | * | 2.90 | 0.90 |
| **40** | *Conopodium majus* | Pignut | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **41** | *Coronopus squamatus* | Swinecress | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **42** | *Corylus avellana* | Hazel Seedling | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **43** | *Corylus avellana* | Hazel Sapling | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |

**Appendix 2**: Mean % covers for each species present per treatment, per season within grassland habitats during data collections in **2002** and **2003**, (**Chapter 2**), **RU** Unfenced, **N-R U** = Non-Rooted Unfenced, **RF** = Rooted Fenced, **N-R F** = Non-Rooted Fenced. * = species not present.

| | | **Habitat** | | | | | | | | | **Grassland** | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | **Season/Year** | **Spring 2002** | | | | **Summer 2002** | | | | **Spring 2003** | | | | **Summer 2003** | | | |
| | | **Treatment** | **R U** | **N-R U** | **R F** | **N-R F** | **R U** | **N-R U** | **R F** | **N-R F** | **R U** | **N-R U** | **R F** | **N-R F** | **R U** | **N-R U** | **R F** | **N-R F** |
| **44** | *Crataegus monogyna* | Hawthorn Seedling | * | * | * | * | * | * | * | * | 0.20 | * | * | * | * | * | * | * |
| **45** | *Cynosurus cristatus* | Crested Dogs Tail | 0.5 | * | 0.6 | 3.00 | 1.00 | * | 2.00 | 3.00 | 2.25 | * | 1.75 | 1.75 | 1.50 | * | 1.50 | 1.50 |

|  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
|  |  |  | 0 |  | 5 |  |  |  |  |  |  |  |  |  |  |  |  |  |
| **46** | *Dactylis glomerata* | Cocksfoot | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **47** | *Dactylorhiza fuchsii* | Common Spotted Orchid | * | * | * | 0.15 | * | * | * | * | * | * | * | 1.00 | * | * | * | * |
| **48** | *Deschampsia caespitosa* | Tufted Hair-grass | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **49** | *Dicranum Scoparium* | Broom Fork Moss | * | * | * | * | * | * | * | * | 0.50 | * | * | * | * | * | * | * |
| **50** | *Dicranella heteromalla* | Silky Forklet Moss | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **51** | *Digitalis purpurea* | Foxglove | * | * | * | * | 0.50 | * | 1.50 | * | 1.00 | * | 4.75 | * | 1.25 | * | 4.75 | * |
| **52** | *Dryopteris austriaca* | Broad Buckler Fern | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **53** | *Dryopteris filix-mas* | Common Male Fern | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **54** | *Epilobium hirsutum* | Great Willow Herb | * | * | 0.25 | 0.10 | * | * | * | * | * | * | * | * | * | * | * | * |
| **55** | *Epilobium montanum* | Broad-Leaved Willowherb | * | * | * | * | 0.35 | * | 0.25 | * | 1.25 | * | 0.55 | * | 0.75 | * | 0.75 | * |
| **56** | *Epilobium obscurum* | Short Fruited Willowherb | * | * | * | * | * | * | * | * | 0.10 | * | * | * | * | * | 0.25 | * |
| **57** | *Epilobium parviflorum* | Hoary Willowherb | * | * | * | * | 1.45 | * | 1.20 | 0.10 | 1.80 | * | 1.25 | * | 2.60 | * | 3.15 | 0.50 |
| **58** | *Epilobium tetragonum* | Square Stalked Willow Herb | 0.55 | * | 0.15 | * | 0.75 | * | 1.65 | * | 0.75 | 0.10 | 1.45 | * | 0.20 | * | 0.25 | * |
| **59** | *Erica cinerea L.* | Bell-Heather | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **60** | *Euphrasia nemorosa* | Eyebright | * | * | * | 0.10 | 0.15 | 0.15 | 0.20 | 2.00 | 0.55 | * | * | * | * | 0.50 | 0.75 | 0.75 |
| **61** | *Eurhynchium Praelongum* | Common Feather Moss | * | * | * | * | 0.50 | * | * | * | 1.50 | * | 1.00 | * | 1.00 | * | * | * |
| **62** | *Eurhynchium swartzii* | Swartz's Feather Moss | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **63** | *Fagus sylvatica* | Beech Seedling | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **64** | *Festuca Rubra* | Red Fescue | * | 3.00 | * | * | 0.25 | 3.00 | * | * | * | 2.75 | * | * | 1.00 | 3.50 | 0.50 | * |
| **65** | *Fissidens taxifolius* | Common Pocket Moss | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **66** | *Fraxinus excelsior* | Ash Seedling | 0.10 | 0.10 | 0.50 | * | 0.65 | 0.20 | 0.50 | * | 1.00 | * | 0.10 | 0.10 | 0.45 | * | 0.25 | * |
| **67** | *Fraxinus excelsior* | Ash Sapling | * | * | * | * | * | * | * | * | * | * | * | * | 0.25 | * | * | * |
| **68** | *Galium aparine* | Cleavers, Goosegrass | * | * | * | * | 0.20 | * | * | * | * | * | * | * | * | * | * | * |
| **69** | *Galium Palustre* | Common Marsh Bedstraw | * | * | * | * | * | * | * | * | 0.65 | * | 0.20 | * | 0.25 | * | * | * |
| **70** | *Glechoma hederacea* | Ground Ivy | 0.15 | 1.10 | 1.00 | 0.50 | 1.20 | 0.55 | 6.50 | 1.95 | 2.25 | 1.50 | 4.90 | 2.00 | 3.90 | 1.75 | 5.75 | 2.05 |
| **71** | *Gnaphalium uliginosum* | Marsh Cudweed | * | * | * | * | 0.80 | * | 0.40 | * | * | * | * | * | 0.75 | * | 0.25 | * |
| **72** | *Hedera helix* | Ivy | * | * | * | * | * | * | * | * | * | * | * | * | 0.15 | * | * | * |
| **73** | *Heracleum Sphondylium* | Hog Weed | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **74** | *Holcus lanatus* | Yorkshire Fog | 0.25 | 21.40 | 1.40 | 20.25 | 7.40 | 21.00 | 17.20 | 19.75 | 11.55 | 19.25 | 23.35 | 26.20 | 12.30 | 22.05 | 21.80 | 21.50 |
| **75** | *Hordeum Secalinum* | Meadow Barley | * | * | * | 0.50 | * | * | * | 0.50 | 1.25 | 0.25 | 0.25 | * | * | * | * | * |
| **76** | *Hyacinthoides non-scripta* | Bluebell | * | * | * | * | * | * | * | * | 0.35 | * | 0.50 | * | * | * | * | * |
| **77** | *Hypericum hirsutum* | Hairy St. Johns-wort | * | * | * | * | 0.50 | * | 0.30 | * | 0.85 | 0.15 | 0.80 | 0.15 | * | * | 0.25 | * |

| | | | | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **78** | *Hypericum humafusum* | Trailing St. Johns-wort | * | * | * | * | * | * | * | * | * | * | * | * | 1.00 | * | 0.70 | * |
| **79** | *Hypericum maculatum* | Imperforate St. Johns-Wort | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **80** | *Hypericum perforatum* | Perforate St. Johns-wort | * | * | * | * | 0.20 | * | 0.90 | 0.25 | 1.65 | * | 3.10 | 0.25 | 1.15 | * | 1.60 | * |
| **81** | *Hypnum jutlandicum* | Heath Plait Moss | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **82** | *Ilex aquifolium* | Holly Seedling | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **83** | *Juncus bufonius* | Toad Rush | * | * | * | * | 0.50 | * | 1.55 | * | 2.05 | * | 0.50 | * | * | * | * | * |
| **84** | *Juncus bulbosis* | Bulbous Rush | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **85** | *Juncus effusus* | Soft Rush | * | * | * | 1.50 | * | * | 0.40 | 2.40 | 0.75 | * | 0.50 | 2.50 | * | * | 0.50 | 3.50 |
| **86** | *Kickxia elatine* | Sharp leaved Fluellen | * | * | * | * | 2.25 | * | * | * | * | * | * | * | 1.05 | * | 0.40 | * |

**Appendix 2**: Mean % covers for each species present per treatment, per season within grassland habitats during data collections in **2002** and **2003**, (**Chapter 2**). **RU**
Unfenced, **N-R U** = Non-Rooted Unfenced, **RF** = Rooted Fenced, **N-R F** = Non-Rooted Fenced. * = species not present.

| | | **Habitat** | | | | | | | | | **Grassland** | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | **Season/Year** | **Spring 2002** | | | | **Summer 2002** | | | | **Spring 2003** | | | | **Summer 2003** | | | |
| | | **Treatment** | **R U** | **N-R U** | **R F** | **N-R F** | **R U** | **N-R U** | **R F** | **N-R F** | **R U** | **N-R U** | **R F** | **N-R F** | **R U** | **N-R U** | **R F** | **N-R F** |
| **87** | *Lamiastrum galeobdolon* | Yellow Archangel | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **88** | *Lolium perenne* | Common Rye Grass | * | 0.75 | * | 1.75 | 0.50 | 1.75 | 2.00 | 1.75 | 0.50 | 0.90 | 1.50 | 1.75 | 1.00 | 0.90 | 2.25 | 1.40 |
| **89** | *Lonicera periclymenum* | Honeysuckle | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **90** | *Lotus corniculatus* | Common Bird's-Foot Trefoil | * | 3.00 | 0.25 | 5.30 | 1.30 | 4.75 | 0.70 | 5.35 | 0.15 | 2.75 | 0.25 | 3.40 | 1.35 | 1.05 | 0.50 | 1.40 |
| **91** | *Luzula campestris* | Field Woodrush | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **92** | *Luzula multiflora* | Heath Woodrush | * | * | * | 0.50 | * | * | * | 0.50 | 0.10 | 0.75 | 0.75 | * | * | * | * | * |
| **93** | *Luzula pilosa* | Hairy Wood-rush | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **94** | *Lychnis flos-cuculi* | Ragged Robin | * | * | * | * | * | * | * | * | 0.75 | * | * | * | * | * | * | * |
| **95** | *Lysimachia nemorum* | Yellow Pimpernel | * | * | * | * | * | * | * | * | 0.25 | * | * | * | * | * | * | * |
| **96** | *Medicago lupulina* | Black Medick | * | 3.00 | * | 1.10 | 0.25 | 3.15 | * | 1.25 | 0.50 | 1.50 | 0.10 | 0.25 | * | 2.50 | * | 0.50 |
| **97** | *Mentha arvensis* | Corn Mint | * | 0.50 | * | 1.00 | 0.90 | 1.00 | 0.40 | 2.50 | 0.85 | 1.00 | 1.40 | 2.25 | 2.15 | 3.75 | 2.95 | 2.50 |
| **98** | | Unidentified Moss | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **99** | *Mercurialis perennis* | Dog Mercury | * | * | * | * | * | * | * | * | * | * | * | * | * | * | 0.25 | * |
| **100** | *Milium effusum* | Wood Millet | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **101** | *Mnium Hornum* | Swan's Neck Thyme Moss | * | * | * | * | * | * | * | * | 0.45 | * | * | * | * | * | * | * |
| **102** | *Moehringia trinervia* | Three Nerved Sandwort | * | * | * | * | * | * | * | * | 0.40 | * | * | * | * | * | * | * |

| | | | | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **103** | *Montia fontana* | Blinks | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **104** | *Montia perfoliata* | Spring Beauty | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **105** | *Myosotis arvensis* | Field Forget-me-not | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **106** | *Oxalis acetosella* | Wood Sorrel | * | * | * | * | 0.15 | * | * | * | * | * | * | * | * | * | 0.20 | * |
| **107** | *Pedicularis sylvatica* | Lousewort | * | * | * | * | 0.20 | * | * | * | * | * | * | * | * | * | * | * |
| **108** | *Phleum pratense* | Timothy Grass (Catstail) | * | * | * | 0.25 | * | * | 0.50 | 0.25 | * | * | 0.40 | 0.25 | 0.25 | 1.00 | 0.25 | 0.25 |
| **109** | *Pilosella officinarum* | Mouse-ear Hawkweed | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **110** | *Plagiomnium undulatum* | Hart's Tongue Thyme Moss | * | * | * | * | * | * | 1.00 | * | * | * | 1.00 | * | * | * | 0.75 | * |
| **111** | *Plantago major* | Greater Plantain | * | * | * | * | 1.30 | * | 0.90 | * | 1.60 | * | 1.15 | 0.15 | 1.45 | * | 1.40 | * |
| **112** | *Poa annua* | Annual Meadow Grass | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **113** | *Poa Nemoralis* | Wood Meadow Grass | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **114** | *Poa pratensis* | Smooth Stalked Meadow Grass | * | * | 0.75 | * | * | * | 2.75 | * | 1.25 | * | 4.00 | * | * | * | 3.00 | 0.25 |
| **115** | *Poa trivialis* | Rough Stalked Meadow Grass | 1.65 | 27.50 | 0.60 | 27.10 | 6.00 | 28.00 | 9.65 | 28.00 | 15.50 | 24.50 | 16.75 | 28.05 | 12.55 | 24.00 | 12.05 | 29.00 |
| **116** | *Polygala serpyllifolia* | Heath Milkwort | * | * | * | * | * | 0.25 | * | 0.40 | * | * | * | 0.10 | * | * | * | * |
| **117** | *Polygala Vulgaris* | Common Milkwort | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **118** | *Polytrichum formosum* | Bank Haircap Moss | * | 0.75 | * | * | 0.40 | 0.40 | 0.50 | * | 0.90 | * | 0.75 | * | 0.50 | 0.50 | 0.75 | * |
| **119** | *Potentilla anserina* | Silver Weed | * | * | 0.25 | * | * | 1.00 | * | * | * | * | 2.00 | * | * | * | 1.75 | * |
| **120** | *Potentilla erecta* | Tormentil | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **121** | *Potentilla reptans* | Creeping Cinquefoil | * | * | * | * | 0.15 | * | 0.15 | * | * | * | * | 0.25 | 0.25 | 0.75 | 0.65 | 0.25 |
| **122** | *Potentilla sterilis* | Barren Strawberry | * | 0.50 | * | 1.00 | * | 0.50 | 0.40 | 1.15 | 0.25 | 3.15 | 0.95 | 3.50 | * | 1.05 | 1.40 | 1.00 |
| **123** | *Pottia trunchator* | Common Pottia Moss | * | * | * | * | 5.75 | * | 13.50 | * | 1.75 | * | 6.75 | * | 7.50 | * | 15.25 | * |
| **124** | *Primula vulgaris* | Primrose | * | 0.10 | * | 0.25 | * | * | * | 0.25 | * | 0.25 | 0.20 | * | * | * | * | 0.40 |
| **125** | *Prunella vulgaris* | Selfheal | 0.70 | 0.50 | 0.85 | 2.00 | 3.20 | 1.15 | 4.00 | 1.90 | 6.00 | 1.05 | 10.05 | 1.20 | 3.80 | 0.60 | 4.35 | 0.60 |
| **126** | *Prunus avium* | Wild Cherry Seedling | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **127** | *Prunus spinosa* | Sloe Seedling / Blackthorn | * | * | * | * | 0.25 | * | 0.65 | * | * | * | * | * | * | * | 0.25 | * |
| **128** | *Pseudoscleropodium purum* | Neat Feather Moss | 0.25 | 7.75 | 1.25 | 5.25 | 1.00 | 3.90 | 5.00 | 5.25 | 3.25 | 7.50 | 5.25 | 5.25 | 2.00 | 8.00 | 6.50 | 5.25 |
| **129** | *Pteridium aquilinum* | Bracken | * | * | * | 3.00 | * | * | * | 5.75 | * | * | * | 4.50 | * | 0.15 | * | 6.50 |

**Appendix 2**: Mean % covers for each species present per treatment, per season within grassland habitats during data collections in **2002** and **2003**, (**Chapter 2**). **RU**
Unfenced, **N-R U** = Non-Rooted Unfenced, **RF** = Rooted Fenced, **N-R F** = Non-Rooted Fenced.  * = species not present.

**Habitat** **Grassland**

| | | Season/Year | Spring 2002 | | | | Summer 2002 | | | | Spring 2003 | | | | Summer 2003 | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | Treatment | R U | N-R U | R F | N-R F | R U | N-R U | R F | N-R F | R U | N-R U | R F | N-R F | R U | N-R U | R F | N-R F |
| **130** | *Pulicaria dysenterica* | Common Fleabane | * | * | * | * | * | * | 0.25 | 0.15 | 0.25 | * | 0.10 | * | 0.50 | * | 0.50 | 0.75 |
| **131** | *Quercus robur* | Oak Sapling | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **132** | *Quercus robur* | Pedunculate Oak Seedling | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **133** | *Ranunculus parviflorus* | Small-flowered Buttercup | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **134** | *Ranunculus ficaria* | Lesser Celendine | * | * | * | * | * | * | * | * | 0.60 | * | * | * | * | * | * | * |
| **135** | *Ranunculus flammula* | Lesser Spearwort | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **136** | *Ranunculus repens* | Creeping Buttercup | 2.30 | 11.50 | 4.05 | 14.25 | 7.05 | 11.30 | 18.65 | 14.55 | 19.30 | 13.10 | 30.00 | 19.60 | 12.45 | 8.65 | 24.35 | 16.00 |
| **137** | *Rhytidiadelphus squarrosus* | Springy Tough Moss | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **138** | *Rosa canina* | Common Dog Rose | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **139** | *Rubus fruticosus* | Bramble | * | * | * | 0.25 | 1.05 | * | 0.50 | 0.35 | 1.20 | * | 1.45 | * | 2.40 | 0.10 | 3.75 | 0.50 |
| **140** | *Rumex acetosa* | Common Sorrel | * | * | * | * | * | 0.25 | * | * | * | * | * | * | * | * | * | * |
| **141** | *Rumex acetosella* | Sheep Sorrel | * | * | * | * | 0.75 | * | * | * | * | * | * | * | 0.45 | * | * | * |
| **142** | *Rumex obtusifolius* | Broad leaved Dock | * | * | * | * | * | * | * | * | * | * | * | * | 0.15 | * | 0.15 | * |
| **143** | *Rumex sanguineus* | Wood Dock | * | * | * | * | 0.10 | * | 0.75 | * | * | * | 0.45 | * | * | * | 0.50 | * |
| **144** | *Sagina apetala* | Common Annual Pearlwort | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **145** | *Sagina procumbens* | Procumbent Pearlwort | * | * | * | * | 6.30 | * | 5.25 | * | 3.75 | 0.25 | 4.55 | * | * | * | * | * |
| **146** | *Salix caprea* | Goat Willow Sapling | * | * | * | * | * | * | * | * | * | * | * | * | * | * | 0.50 | * |
| **147** | *Sambucus nigra* | Elder Seedling | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **148** | *Sarothamnus scoparius* | Broom | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **149** | *Senecio jacobaea* | Common Ragwort | * | * | * | * | 1.90 | 0.50 | 2.75 | 0.15 | 2.00 | 0.75 | 5.25 | 0.35 | 2.85 | 0.75 | 5.00 | * |
| **150** | *Silene dioica* | Red Campion | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **151** | *Sonchus oleraceus* | Smooth Sow Thistle | * | * | * | * | * | * | 0.50 | * | * | * | * | * | * | * | * | * |
| **152** | *Stachys arvensis* | Field Woundwort | * | * | * | * | * | * | * | * | * | * | 0.20 | * | * | * | 0.15 | * |
| **153** | *Stachys sylvatica* | Hedge Woundwort | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | 0.20 |
| **154** | *Stellaria graminea* | Lesser Stitchwort | * | 2.00 | * | 0.85 | 0.15 | 1.00 | * | 1.65 | 0.15 | 1.15 | * | 1.70 | 0.40 | 0.15 | * | 2.25 |
| **155** | *Stellaria media* | Common Chickweed | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **156** | *Stellaria neglecta* | Greater Chickweed | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **157** | *Taraxacum officinale* | Dandelion | * | * | * | 0.25 | * | * | * | * | 0.20 | 0.20 | * | 0.20 | * | * | * | 0.20 |
| **158** | *Teucrium scorodonia* | Wood Sage | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **159** | *Thuidium tamariscinum* | Common Tamarisk Moss | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **160** | *Trifolium dubium* | Lesser Trefoil | * | * | * | 0.60 | * | * | * | 0.15 | 0.20 | * | * | 0.15 | * | 0.40 | 0.40 | * |
| **161** | *Trifolium micranthum* | Slender Trefoil | * | * | 0.05 | 1.75 | * | * | * | 0.50 | 0.15 | * | 0.15 | 0.25 | * | 0.50 | * | 0.35 |

| | | | | | | | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **162** | *Trifolium repens* | White Clover | 0.25 | 8.75 | 0.50 | 15.60 | | 1.10 | 7.85 | 3.25 | 12.85 | | 4.20 | 15.15 | 4.80 | 10.30 | | 2.50 | 3.00 | 2.45 | 3.55 |
| **163** | *Tussilago farfara* | Coltsfoot | * | * | * | * | | 0.50 | * | * | * | | 1.00 | * | * | * | | 0.75 | * | * | * |
| **164** | *Urtica dioica* | Stinging Nettle | 0.35 | * | 0.25 | 0.35 | | 2.00 | * | 3.40 | 0.40 | | 2.40 | * | 3.90 | 0.65 | | 2.75 | * | 2.90 | 0.55 |
| **165** | *Veronica arvensis* | Wall Speedwell | * | * | * | 0.90 | | * | * | * | * | | 0.75 | * | 0.25 | * | | * | * | * | * |
| **166** | *Veronica montana* | Wood Speedwell | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **167** | *Veronica officinalis* | Heath Speedwell | * | * | * | * | | 0.10 | * | * | * | | * | * | * | * | | 2.65 | * | 1.75 | * |
| **168** | *Veronica persica* | Common Field Speedwell | * | * | * | * | | * | * | * | * | | 1.15 | * | 1.70 | * | | * | * | * | * |
| **169** | *Veronica serpyllifolia* | Thyme Leaved Speedwell | * | 0.75 | * | * | | * | * | * | * | | 0.50 | * | 0.50 | * | | * | * | * | * |
| **170** | *Vicia sativa* | Common Vetch | * | 0.25 | * | 0.60 | | * | * | * | * | | * | * | * | 0.25 | | * | * | * | * |
| **171** | *Viola lactea* | Pale Dog Violet | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **172** | *Viola riviniana* | Common Dog Violet | 0.25 | 0.55 | 0.15 | 0.75 | | 2.15 | 0.90 | 1.35 | 2.00 | | 0.70 | 2.05 | 2.25 | 2.75 | | 1.50 | 1.80 | 2.25 | 1.50 |
| **173** | *Vulpia bromoides* | Squirrel-Tail Fescue | * | 1.25 | * | 0.50 | | 0.95 | 1.25 | 1.00 | * | | 2.65 | 1.50 | 1.25 | 0.40 | | 2.50 | 1.25 | 1.25 | * |

**Appendix 3:** Complete species list incorporating mean % covers for each species present per treatment, per season within woodland habitats during data collections in **2003** (**Chapter 2**). **RU** = Rooted Unfenced, **N-R U** = Non-Rooted Unfenced, **RF** = Rooted Fenced, **N-R F** = Non-Rooted Fenced. * = species not present.

| | | **Habitat** | **Woodland** | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | **Season/Year** | **Spring 2002** | | | | **Summer 2002** | | | | **Spring 2003** | | | | **Summer 2003** | | | |
| | | **Treatment** | **R U** | **N-R U** | **R F** | **N-R F** | **R U** | **N-R U** | **R F** | **N-R F** | **R U** | **N-R U** | **R F** | **N-R F** | **R U** | **N-R U** | **R F** | **N-R F** |
| **1** | *Acer campestre* | Field Maple Seedling | * | * | * | 0.08 | 0.08 | 0.04 | * | 0.16 | 0.20 | 0.08 | 0.52 | * | 0.08 | * | 0.44 | * |
| **2** | *Agrostis canina* | Brown Bent Grass | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **3** | *Agrostis capillaris* | Common Bent Grass | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **4** | *Agrostis stolonifera* | Creeping Bent Grass | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **5** | *Ajuga reptans* | Bugle | * | * | * | * | * | * | * | * | * | * | * | * | * | * | 0.20 | * |
| **6** | *Anagallis arvensis* | Scarlet Pimpernel | * | * | * | * | * | * | * | * | * | * | * | * | * | * | 0.20 | * |
| **7** | *Anemone nemorosa* | Wood Anemone | 0.20 | 3.96 | 0.04 | 6.04 | * | * | * | * | 8.08 | 10.00 | 9.60 | 13.76 | 0.12 | * | * | * |
| **8** | *Anthoxanthum odoratum* | Sweet Vernal Grass | * | * | * | * | * | * | * | * | 2.60 | * | 2.60 | 1.00 | 1.20 | * | 2.20 | 0.40 |
| **9** | *Arctium Lappa L.* | Greater Burdock | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **10** | *Arrhenatherum Elatius* | False Oat Grass | * | 0.80 | 0.32 | 0.40 | 0.80 | 0.16 | 1.28 | 1.00 | 1.12 | 0.16 | 2.04 | 1.72 | 1.40 | * | 1.44 | 1.20 |
| **11** | *Atrichum undulatum* | Common Smooth Cap | * | * | * | 0.20 | * | * | * | 0.20 | * | * | * | 0.20 | * | * | * | * |
| **12** | *Bellis perennis L.* | Common Daisy | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **13** | *Betonica officinalis* | Betony | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **14** | *Betula pendula* | Silver Birch Seedling | * | * | * | * | 0.40 | * | 0.08 | 0.04 | 0.20 | * | 0.40 | 0.08 | 0.36 | * | 0.52 | * |
| **15** | *Betula pendula* | Silver Birch Sapling | * | * | * | * | * | * | * | * | * | * | * | * | 0.20 | * | 0.52 | * |
| **16** | *Brachypodium sylvatica* | Wood False Brome | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **17** | *Brachythecium rutabulum* | Rough Stalked Feather Moss | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **18** | *Bromus ramosus* | Wood Brome Grass | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **19** | *Calliergon cuspidatum* | Pointed Spear Moss | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **20** | *Calluna Vulgaris* | Ling | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **21** | *Cardamine pratensis* | Cuckooflower | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |

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|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **22** | *Carex pendula* | Pendulous Sedge | * | * | * | * | * | * | * | * | * | * | * | * | 0.20 | * | * | * |
| **23** | *Carex sylvatica* | Wood Sedge | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **24** | *Carpinus betulus* | Hornbeam Seedling | 0.04 | 0.60 | 0.12 | 0.48 | 0.48 | 0.88 | 0.28 | 0.44 | 1.52 | 1.72 | 1.80 | 0.96 | 1.68 | 0.96 | 1.08 | 0.24 |
| **25** | *Carpinus betulus* | Hornbeam Sapling | * | * | * | * | * | * | * | * | * | * | * | * | * | * | 0.20 | * |
| **26** | *Castanea Sativa* | Sweet Chestnut Seedling | * | * | * | * | * | * | 0.04 | * | 1.04 | 0.16 | 0.92 | * | * | 0.04 | * | * |
| **27** | *Centaurea nigra* | Common knapweed | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **28** | *Centaurium erythraea* | Common Centaury | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **29** | *Centaurium pulchellum* | Lesser Centaury | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **30** | *Cerastium arvense* | Field Mouse-ear | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **31** | *Cerastium holosteoides* | Common Mouse Ear | * | * | * | * | * | * | * | * | 0.20 | * | * | * | * | * | * | * |
| **32** | *Chamaenerion angustifolium* | Rosebay Willowherb | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **33** | *Chenopodium botryodes* | Small Red Goosefoot | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **34** | *Chenopodium polyspermum* | Many-Seeded Goosefoot | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **35** | *Chenopodium rubrum* | Red Goosefoot | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **36** | *Circaea lutetiana* | Enchanter's Nightshade | * | * | * | 0.20 | * | * | * | 0.08 | 0.76 | * | 0.08 | * | * | * | * | 0.16 |
| **37** | *Cirsium arvense* | Creeping Thistle | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **38** | *Cirsium palustre* | Marsh Thistle | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **39** | *Cirsium Vulgare* | Spear Thistle | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **40** | *Conopodium majus* | Pignut | * | 0.40 | * | 0.48 | * | * | * | * | 0.40 | 0.60 | * | 0.80 | * | * | * | * |
| **41** | *Coronopus squamatus* | Swinecress | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **42** | *Corylus avellana* | Hazel Seedling | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **43** | *Corylus avellana* | Hazel Sapling | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| | | **Habitat** | **Woodland** | | | | | | | | | | | | | | | |
| | | **Season/Year** | **Spring 2002** | | | | **Summer 2002** | | | | **Spring 2003** | | | | **Summer 2003** | | | |
| | | **Treatment** | **R U** | **N-R U** | **R F** | **N-R F** | **R U** | **N-R U** | **R F** | **N-R F** | **R U** | **N-R U** | **R F** | **N-R F** | **R U** | **N-R U** | **R F** | **N-R F** |
| **44** | *Crataegus monogyna* | Hawthorn Seedling | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **45** | *Cynosurus cristatus* | Crested Dogs Tail | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **46** | *Dactylis glomerata* | Cocksfoot | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |

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| **47** | *Dactylorhiza fuchsii* | Common Spotted Orchid | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **48** | *Deschampsia caespitosa* | Tufted Hair-grass | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **49** | *Dicranum Scoparium* | Broom Fork Moss | * | 0.04 | * | * | | 3.92 | 0.40 | 0.80 | 0.16 | | 0.40 | 0.20 | 3.40 | 0.44 | | 1.00 | 0.16 | 1.00 | 0.20 |
| **50** | *Dicranella heteromalla* | Silky Forklet Moss | * | * | * | * | | 11.72 | * | 14.00 | 0.36 | | 8.80 | 0.80 | 19.60 | 1.32 | | 9.12 | * | 14.20 | 0.40 |
| **51** | *Digitalis purpurea* | Foxglove | * | * | * | * | | 5.20 | 0.44 | 5.28 | 0.36 | | 12.16 | * | 13.92 | 0.72 | | 4.92 | * | 11.80 | 1.36 |
| **52** | *Dryopteris austriaca* | Broad Buckler Fern | * | 1.00 | * | * | | * | 1.20 | * | * | | * | * | * | * | | * | 0.40 | * | * |
| **53** | *Dryopteris filix-mas* | Common Male Fern | * | * | * | 3.00 | | * | 1.00 | * | * | | * | * | * | 0.80 | | * | * | * | 1.00 |
| **54** | *Epilobium hirsutum* | Great Willow Herb | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **55** | *Epilobium montanum* | Broad-Leaved Willowherb | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **56** | *Epilobium obscurum* | Short Fruited Willowherb | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **57** | *Epilobium parviflorum* | Hoary Willowherb | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **58** | *Epilobium tetragonum* | Square Stalked Willow Herb | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **59** | *Erica cinerea L.* | Bell-Heather | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **60** | *Euphrasia nemorosa* | Eyebright | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **61** | *Eurhynchium Praelongum* | Common Feather Moss | * | 0.40 | * | 3.12 | | 2.00 | 1.48 | 1.72 | 3.44 | | 0.20 | 2.88 | 5.00 | 4.76 | | 1.72 | 1.64 | 5.40 | 3.76 |
| **62** | *Eurhynchium swartzii* | Swartz's Feather Moss | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **63** | *Fagus sylvatica* | Beech Seedling | * | * | * | * | | 0.08 | * | 0.08 | * | | 0.48 | * | 0.20 | * | | * | * | * | * |
| **64** | *Festuca Rubra* | Red Fescue | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **65** | *Fissidens taxifolius* | Common Pocket Moss | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **66** | *Fraxinus excelsior* | Ash Seedling | * | * | * | * | | * | 0.12 | 0.08 | 0.44 | | 0.52 | 0.04 | 0.32 | 0.20 | | * | * | * | 0.12 |
| **67** | *Fraxinus excelsior* | Ash Sapling | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **68** | *Galium aparine* | Cleavers, Goosegrass | * | 0.36 | * | 0.96 | | * | * | * | * | | * | 0.32 | * | 1.08 | | * | * | * | * |
| **69** | *Galium Palustre* | Common Marsh Bedstraw | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **70** | *Glechoma hederacea* | Ground Ivy | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **71** | *Gnaphalium uliginosum* | Marsh Cudweed | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **72** | *Hedera helix* | Ivy | * | * | * | * | | * | * | * | * | | * | * | * | * | | 0.20 | * | 0.12 | * |
| **73** | *Heracleum Sphondylium* | Hog Weed | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **74** | *Holcus lanatus* | Yorkshire Fog | 0.12 | * | * | * | | 0.68 | * | 0.76 | * | | 1.00 | * | 1.60 | 0.36 | | 1.12 | * | 2.08 | * |
| **75** | *Hordeum Secalinum* | Meadow Barley | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |

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| **76** | *Hyacinthoides non-scripta* | Bluebell | 4.04 | 78.60 | 4.16 | 78.40 | * | * | * | * | 27.20 | 61.20 | 45.32 | 83.80 | * | * | * | * |
| **77** | *Hypericum hirsutum* | Hairy St. Johns-wort | * | * | * | * | 0.12 | * | * | * | 0.52 | 0.12 | 0.28 | * | * | * | * | * |
| **78** | *Hypericum humafusum* | Trailing St. Johns-wort | * | * | * | * | * | * | * | * | * | * | * | * | 0.56 | * | 0.32 | * |
| **79** | *Hypericum maculatum* | Imperforate St. Johns-Wort | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **80** | *Hypericum perforatum* | Perforate St. Johns-wort | * | * | * | * | * | * | * | * | 0.92 | * | 0.20 | * | 0.40 | * | * | 0.08 |
| **81** | *Hypnum jutlandicum* | Heath Plait Moss | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **82** | *Ilex aquifolium* | Holly Seedling | * | * | * | * | * | 0.04 | * | * | * | 0.04 | * | * | 0.16 | 0.08 | 0.08 | * |
| **83** | *Juncus bufonius* | Toad Rush | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **84** | *Juncus bulbosis* | Bulbous Rush | * | * | * | * | 0.40 | * | * | * | 0.40 | * | * | * | 1.00 | * | * | * |
| **85** | *Juncus effusus* | Soft Rush | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **86** | *Kickxia elatine* | Sharp leaved Fluellen | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |

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| | | **Habitat** | **Woodland** | | | | | | | | | | | | | | | |
| | | **Season/Year** | **Spring 2002** | | | | **Summer 2002** | | | | **Spring 2003** | | | | **Summer 2003** | | | |
| | | **Treatment** | **R U** | **N-R U** | **R F** | **N-R F** | **R U** | **N-R U** | **R F** | **N-R F** | **R U** | **N-R U** | **R F** | **N-R F** | **R U** | **N-R U** | **R F** | **N-R F** |
| **87** | *Lamiastrum galeobdolon* | Yellow Archangel | * | 0.04 | * | 0.08 | * | * | * | * | * | 0.08 | * | * | * | * | * | * |
| **88** | *Lolium perenne* | Common Rye Grass | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **89** | *Lonicera periclymenum* | Honeysuckle | * | * | * | * | * | * | * | * | * | * | * | 0.12 | * | * | * | * |
| **90** | *Lotus corniculatus* | Common Bird's-Foot Trefoil | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **91** | *Luzula campestris* | Field Woodrush | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **92** | *Luzula multiflora* | Heath Woodrush | * | * | * | * | * | * | * | * | 0.60 | * | 0.40 | * | * | * | * | * |
| **93** | *Luzula pilosa* | Hairy Wood-rush | * | * | * | * | * | * | 0.12 | * | 0.20 | * | * | * | * | * | * | * |
| **94** | *Lychnis flos-cuculi* | Ragged Robin | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **95** | *Lysimachia nemorum* | Yellow Pimpernel | * | * | * | * | * | * | * | * | 0.20 | * | * | 0.80 | 2.12 | * | * | * |
| **96** | *Medicago lupulina* | Black Medick | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **97** | *Mentha arvensis* | Corn Mint | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| **98** | | Unidentified Moss | * | * | * | * | 3.200 | * | 10.000 | * | 9 | 1.920 | 22.600 | * | 0.800 | * | * | * |
| **99** | *Mercurialis perennis* | Dog Mercury | * | * | 1.20 | * | 0.24 | 0.28 | 0.08 | 1.28 | * | * | * | 0.80 | * | * | * | 0.40 |

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|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **100** | *Milium effusum* | Wood Millet | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **101** | *Mnium Hornum* | Swan's Neck Thyme Moss | * | 6.40 | * | 3.80 | | 2.16 | 6.84 | 2.20 | 5.32 | | 2.60 | 3.80 | * | 3.80 | | 2.40 | 6.36 | 2.20 | 3.88 |
| **102** | *Moehringia trinervia* | Three Nerved Sandwort | * | * | * | 0.20 | | 0.72 | 0.08 | * | 0.12 | | 0.68 | * | 0.12 | * | | * | * | 0.32 | * |
| **103** | *Montia fontana* | Blinks | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **104** | *Montia perfoliata* | Spring Beauty | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **105** | *Myosotis arvensis* | Field Forget-me-not | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **106** | *Oxalis acetosella* | Wood Sorrel | * | 0.12 | * | 0.20 | | * | 0.40 | 0.12 | 0.16 | | * | 1.36 | 0.16 | 0.20 | | * | 0.32 | 0.16 | 0.32 |
| **107** | *Pedicularis sylvatica* | Lousewort | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **108** | *Phleum pratense* | Timothy Grass (Catstail) | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **109** | *Pilosella officinarum* | Mouse-ear Hawkweed | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **110** | *Plagiomnium undulatum* | Hart's Tongue Thyme Moss | * | * | * | * | | * | * | 0.20 | * | | * | * | 0.40 | * | | * | * | 0.40 | * |
| **111** | *Plantago major* | Greater Plantain | * | * | * | * | | * | * | * | * | | * | * | * | * | | 0.28 | * | 0.16 | * |
| **112** | *Poa annua* | Annual Meadow Grass | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **113** | *Poa Nemoralis* | Wood Meadow Grass | * | 1.72 | 0.12 | 0.40 | | 0.16 | 0.20 | 0.16 | 0.12 | | 1.92 | 1.92 | 1.16 | 1.96 | | 0.60 | * | 0.48 | * |
| **114** | *Poa pratensis* | Smooth Stalked Meadow Grass | * | * | * | * | | * | * | * | * | | * | * | * | * | | 0.08 | * | * | * |
| **115** | *Poa trivialis* | Rough Stalked Meadow Grass | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | 1.00 | * |
| **116** | *Polygala serpyllifolia* | Heath Milkwort | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **117** | *Polygala Vulgaris* | Common Milkwort | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **118** | *Polytrichum formosum* | Bank Haircap Moss | * | * | * | * | | 0.08 | * | * | 0.04 | | 0.32 | 0.08 | 0.40 | 0.20 | | 0.28 | * | * | * |
| **119** | *Potentilla anserina* | Silver Weed | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **120** | *Potentilla erecta* | Tormentil | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **121** | *Potentilla reptans* | Creeping Cinquefoil | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **122** | *Potentilla sterilis* | Barren Strawberry | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **123** | *Pottia trunchator* | Common Pottia Moss | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |

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|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 124 | *Primula vulgaris* | Primrose | * | 0.12 | * | * | * | 0.12 | * | * | * | 0.20 | * | * | * | 0.20 | * | * |
| 125 | *Prunella vulgaris* | Selfheal | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| 126 | *Prunus avium* | Wild Cherry Seedling | 0.12 | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| 127 | *Prunus spinosa* | Sloe Seedling / Blackthorn | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| 128 | *Pseudoscleropodium purum* | Neat Feather Moss | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| 129 | *Pteridium aquilinum* | Bracken | 0.80 | 0.20 | * | 2.00 | 0.60 | 4.40 | 0.52 | 2.32 | 0.60 | * | 1.40 | 0.84 | 1.60 | 4.00 | 3.00 | 1.80 |

| | | **Habitat** | **Woodland** | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | **Season/Year** | **Spring 2002** | | | | **Summer 2002** | | | | **Spring 2003** | | | | **Summer 2003** | | | |
| | | **Treatment** | **R U** | **N-R U** | **R F** | **N-R F** | **R U** | **N-R U** | **R F** | **N-R F** | **R U** | **N-R U** | **R F** | **N-R F** | **R U** | **N-R U** | **R F** | **N-R F** |
| 130 | *Pulicaria dysenterica* | Common Fleabane | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| 131 | *Quercus robur* | Oak Sapling | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| 132 | *Quercus robur* | Pedunculate Oak Seedling | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| 133 | *Ranunculus parviflorus* | Small-flowered Buttercup | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| 134 | *Ranunculus ficaria* | Lesser Celendine | * | 0.12 | * | * | * | * | * | * | 0.56 | 1.08 | 0.40 | 0.72 | * | * | * | * |
| 135 | *Ranunculus flammula* | Lesser Spearwort | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| 136 | *Ranunculus repens* | Creeping Buttercup | * | * | * | * | * | * | 0.12 | * | * | * | * | * | * | * | * | * |
| 137 | *Rhytidiadelphus squarrosus* | Springy Tough Moss | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| 138 | *Rosa canina* | Common Dog Rose | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| 139 | *Rubus fruticosus* | Bramble | 0.40 | 1.64 | * | 0.44 | 1.08 | 1.24 | 0.52 | 1.60 | 0.84 | 0.76 | 0.64 | 1.52 | 1.12 | 0.96 | 1.60 | 0.72 |
| 140 | *Rumex acetosa* | Common Sorrel | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| 141 | *Rumex acetosella* | Sheep Sorrel | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| 142 | *Rumex obtusifolius* | Broad leaved Dock | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * | * |
| 143 | *Rumex sanguineus* | Wood Dock | * | * | * | 0.20 | * | 1.20 | * | 0.20 | * | * | * | 0.64 | * | * | * | * |

|  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **144** | *Sagina apetala* | Common Annual Pearlwort | * | * | * | * |  | * | * | * | * |  | * | * | * | * |  | * | * | * | * |
| **145** | *Sagina procumbens* | Procumbent Pearlwort | * | * | * | * |  | * | * | * | * |  | * | * | * | * |  | * | * | * | * |
| **146** | *Salix caprea* | Goat Willow Sapling | * | * | * | * |  | * | * | * | * |  | * | * | * | * |  | * | * | * | * |
| **147** | *Sambucus nigra* | Elder Seedling | * | 1.80 | * | 1.40 |  | 0.16 | * | 0.08 | 0.04 |  | * | * | * | 0.16 |  | 0.16 | * | 0.08 | 0.16 |
| **148** | *Sarothamnus scoparius* | Broom | * | * | * | * |  | * | * | * | * |  | * | * | * | * |  | * | * | 0.20 | * |
| **149** | *Senecio jacobaea* | Common Ragwort | * | * | * | * |  | * | * | * | * |  | * | * | * | * |  | * | * | * | * |
| **150** | *Silene dioica* | Red Campion | * | 0.40 | * | 0.16 |  | 3.36 | 0.36 | 3.92 | 0.80 |  | 4.40 | 0.12 | 4.04 | 1.00 |  | 5.80 | 0.20 | 5.20 | 0.40 |
| **151** | *Sonchus oleraceus* | Smooth Sow Thistle | * | * | * | * |  | * | * | * | * |  | * | * | * | * |  | * | * | * | * |
| **152** | *Stachys arvensis* | Field Woundwort | * | * | * | * |  | * | * | * | * |  | * | * | * | * |  | * | * | * | * |
| **153** | *Stachys sylvatica* | Hedge Woundwort | * | 0.52 | * | 0.80 |  | 2.08 | 0.88 | 3.60 | 1.92 |  | 2.60 | 0.52 | 3.28 | 1.76 |  | 0.52 | 0.48 | 2.44 | 0.40 |
| **154** | *Stellaria graminea* | Lesser Stitchwort | * | * | * | * |  | * | * | * | * |  | * | * | * | * |  | * | * | * | * |
| **155** | *Stellaria media* | Common Chickweed | * | * | * | 0.16 |  | * | * | * | * |  | * | * | * | * |  | * | * | * | * |
| **156** | *Stellaria neglecta* | Greater Chickweed | * | * | * | * |  | * | * | * | * |  | * | * | * | * |  | * | * | * | * |
| **157** | *Taraxacum officinale* | Dandelion | * | * | * | * |  | * | * | * | * |  | * | * | * | * |  | * | * | * | * |
| **158** | *Teucrium scorodonia* | Wood Sage | * | * | * | 0.20 |  | * | * | * | * |  | * | * | * | * |  | 4.40 | * | 2.12 | 0.88 |
| **159** | *Thuidium tamariscinum* | Common Tamarisk Moss | * | * | * | * |  | 0.08 | * | 0.20 | * |  | * | * | * | * |  | 0.08 | * | 0.72 | * |
| **160** | *Trifolium dubium* | Lesser Trefoil | * | * | * | * |  | * | * | * | * |  | * | * | * | * |  | * | * | * | * |
| **161** | *Trifolium micranthum* | Slender Trefoil | * | * | * | * |  | * | * | * | * |  | * | * | * | * |  | * | * | * | * |
| **162** | *Trifolium repens* | White Clover | * | * | * | * |  | * | * | * | * |  | * | * | * | * |  | * | * | * | * |
| **163** | *Tussilago farfara* | Coltsfoot | * | * | * | * |  | * | * | * | * |  | * | * | * | * |  | * | * | * | * |
| **164** | *Urtica dioica* | Stinging Nettle | * | * | * | * |  | 0.80 | * | 0.52 | 0.44 |  | * | * | 0.32 | * |  | 0.76 | 0.16 | 1.24 | 0.32 |
| **165** | *Veronica arvensis* | Wall Speedwell | * | * | * | * |  | * | * | * | * |  | * | * | * | * |  | * | * | * | * |
| **166** | *Veronica montana* | Wood Speedwell | * | * | * | * |  | * | * | * | * |  | * | * | * | * |  | 0.20 | * | * | * |
| **167** | *Veronica officinalis* | Heath Speedwell | * | * | * | * |  | 0.20 | * | * | * |  | * | * | * | * |  | * | * | * | * |

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|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **168** | *Veronica persica* | Common Field Speedwell | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **169** | *Veronica serpyllifolia* | Thyme Leaved Speedwell | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **170** | *Vicia sativa* | Common Vetch | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **171** | *Viola lactea* | Pale Dog Violet | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |
| **172** | *Viola riviniana* | Common Dog Violet | * | 0.12 | * | * | | 0.16 | 0.24 | 0.16 | 0.04 | | * | * | * | * | | 0.36 | 0.24 | 0.32 | * |
| **173** | *Vulpia bromoides* | Squirrel-Tail Fescue | * | * | * | * | | * | * | * | * | | * | * | * | * | | * | * | * | * |

**Appendix 4:** Mean % covers for each species present per treatment, per season within woodland ride habitats during data collections in **2002** and **2003** (**Chapter 2**). **RU** = Rooted Unfenced, **N-R U** = Non-Rooted Unfenced, **RF** = Rooted Fenced, **N-R F** = Non-Rooted Fenced. * = species not present.

| | | **Habitat** | **Woodland Rides** | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | **Season/Year** | **Spring 2002** | | | **Summer 2002** | | | **Spring 2003** | | | **Summer 2003** | |
| | | **Treatment** | **R U** | **N-R U** | | **R U** | **N-R U** | | **R U** | **N-R U** | | **R U** | **N-R U** |
| **1** | *Acer campestre* | Field Maple Seedling | * | * | | * | * | | * | * | | * | * |
| **2** | *Agrostis canina* | Brown Bent Grass | 0.92 | 16.80 | | 2.52 | 15.40 | | 8.44 | 12.20 | | 10.72 | 15.60 |
| **3** | *Agrostis capillaris* | Common Bent Grass | 0.60 | 6.12 | | 4.84 | 8.68 | | 6.52 | 9.44 | | 6.76 | 10.52 |
| **4** | *Agrostis stolonifera* | Creeping Bent Grass | * | 0.80 | | * | * | | * | * | | * | * |
| **5** | *Ajuga reptans* | Bugle | 1.72 | 2.88 | | 4.24 | 3.60 | | 4.52 | 1.84 | | 5.76 | 1.96 |
| **6** | *Anagallis arvensis* | Scarlet Pimpernel | 0.48 | 0.08 | | 0.56 | 0.12 | | 2.20 | 0.16 | | 0.12 | * |
| **7** | *Anemone nemorosa* | Wood Anemone | 0.12 | 0.24 | | * | * | | * | 0.52 | | * | * |
| **8** | *Anthoxanthum odoratum* | Sweet Vernal Grass | 0.32 | 4.16 | | 1.20 | 4.20 | | 3.64 | 7.00 | | 1.60 | 3.80 |
| **9** | *Arctium Lappa L.* | Greater Burdock | * | * | | * | 0.16 | | * | * | | 0.20 | * |
| **10** | *Arrhenatherum Elatius* | False Oat Grass | * | * | | * | * | | 0.92 | 0.40 | | 0.52 | * |
| **11** | *Atrichum undulatum* | Common Smooth Cap | * | * | | * | * | | * | * | | * | * |
| **12** | *Bellis perennis L.* | Common Daisy | * | * | | * | * | | * | * | | * | * |
| **13** | *Betonica officinalis* | Betony | * | * | | 0.28 | * | | 0.48 | * | | 0.12 | * |
| **14** | *Betula pendula* | Silver Birch Seedling | 0.44 | 0.68 | | 2.88 | 1.48 | | 1.00 | 0.20 | | 1.56 | 0.16 |
| **15** | *Betula pendula* | Silver Birch Sapling | * | * | | * | * | | 3.60 | 2.60 | | 4.32 | 3.60 |
| **16** | *Brachypodium sylvatica* | Wood False Brome | * | * | | * | * | | * | * | | * | * |
| **17** | *Brachythecium rutabulum* | Rough Stalked Feather Moss | * | * | | * | * | | * | * | | * | * |
| **18** | *Bromus ramosus* | Wood Brome Grass | * | 0.40 | | * | * | | * | * | | * | * |
| **19** | *Calliergon cuspidatum* | Pointed Spear Moss | 1.00 | 3.00 | | 0.80 | 2.80 | | 0.40 | 0.32 | | 2.00 | 3.40 |
| **20** | *Calluna Vulgaris* | Ling | * | 0.44 | | * | 0.60 | | * | 0.92 | | 0.20 | 1.32 |
| **21** | *Cardamine pratensis* | Cuckooflower | * | * | | * | * | | * | * | | * | * |
| **22** | *Carex pendula* | Pendulous Sedge | * | * | | 0.28 | 0.52 | | 0.48 | 0.40 | | 0.84 | 0.20 |
| **23** | *Carex sylvatica* | Wood Sedge | * | * | | * | * | | * | * | | * | * |
| **24** | *Carpinus betulus* | Hornbeam Seedling | 0.16 | 0.32 | | 0.08 | 0.48 | | 0.88 | 0.36 | | 0.52 | 0.36 |
| **25** | *Carpinus betulus* | Hornbeam Sapling | * | * | | * | * | | * | * | | * | * |

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|---|---|---|---|---|---|---|---|---|---|---|
| **26** | *Castanea Sativa* | Sweet Chestnut Seedling | * | * | 0.08 | * | * | * | * | * |
| **27** | *Centaurea nigra* | Common knapweed | * | 0.40 | * | 0.40 | * | 0.20 | * | * |
| **28** | *Centaurium erythraea* | Common Centaury | * | * | * | 0.08 | 0.32 | 0.20 | 0.80 | 0.16 |
| **29** | *Centaurium pulchellum* | Lesser Centaury | * | * | 0.16 | 0.20 | * | * | * | * |
| **30** | *Cerastium arvense* | Field Mouse-ear | * | * | * | * | * | * | * | * |
| **31** | *Cerastium holosteoides* | Common Mouse Ear | * | 0.60 | 0.44 | 0.76 | 0.80 | 0.32 | 0.12 | 0.20 |

| | | **Habitat** | **Woodland Rides** | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | **Season/Year** | **Spring 2002** | | **Summer 2002** | | **Spring 2003** | | **Summer 2003** | |
| | | **Treatment** | **R U** | **N-R U** | **R U** | **N-R U** | **R U** | **N-R U** | **R U** | **N-R U** |
| **32** | *Chamaenerion angustifolium* | Rosebay Willowherb | 8.48 | 1.20 | 19.16 | 6.00 | 9.32 | 4.40 | 9.92 | 3.12 |
| **33** | *Chenopodium botryodes* | Small Red Goosefoot | * | * | * | * | * | * | * | * |
| **34** | *Chenopodium polyspermum* | Many-Seeded Goosefoot | * | * | * | * | 0.12 | 0.20 | * | * |
| **35** | *Chenopodium rubrum* | Red Goosefoot | * | * | * | * | * | * | * | * |
| **36** | *Circaea lutetiana* | Enchanter's Nightshade | * | 0.28 | 0.32 | 0.36 | 0.68 | 0.48 | 0.56 | 0.48 |
| **37** | *Cirsium arvense* | Creeping Thistle | 0.40 | 0.24 | 3.80 | 3.80 | 3.20 | 0.80 | 2.80 | 1.00 |
| **38** | *Cirsium palustre* | Marsh Thistle | 0.20 | 0.28 | 0.60 | 0.60 | 0.56 | 0.40 | 0.32 | 0.36 |
| **39** | *Cirsium Vulgare* | Spear Thistle | 0.40 | 0.20 | 0.60 | 0.32 | * | * | 1.40 | * |
| **40** | *Conopodium majus* | Pignut | * | * | * | * | * | * | * | * |
| **41** | *Coronopus squamatus* | Swinecress | * | * | * | * | * | * | * | * |
| **42** | *Corylus avellana* | Hazel Seedling | * | * | 0.20 | * | * | * | * | * |
| **43** | *Corylus avellana* | Hazel Sapling | * | * | * | * | 0.40 | * | 0.80 | * |
| **44** | *Crataegus monogyna* | Hawthorn Seedling | * | * | * | * | * | * | * | * |
| **45** | *Cynosurus cristatus* | Crested Dogs Tail | * | * | * | * | * | * | * | * |
| **46** | *Dactylis glomerata* | Cocksfoot | * | * | * | * | * | * | * | * |
| **47** | *Dactylorhiza fuchsii* | Common Spotted Orchid | * | * | * | * | * | * | * | * |
| **48** | *Deschampsia caespitosa* | Tufted Hair-grass | * | * | * | * | * | * | * | * |
| **49** | *Dicranum Scoparium* | Broom Fork Moss | 0.16 | 0.40 | 0.36 | 3.40 | 0.64 | 3.20 | 0.52 | 3.40 |
| **50** | *Dicranella heteromalla* | Silky Forklet Moss | * | * | * | * | * | * | * | * |
| **51** | *Digitalis purpurea* | Foxglove | * | 0.08 | 0.64 | 0.12 | 1.24 | * | 1.64 | 0.80 |
| **52** | *Dryopteris austriaca* | Broad Buckler Fern | * | * | * | * | * | * | * | * |
| **53** | *Dryopteris filix-mas* | Common Male Fern | * | * | * | * | * | * | * | * |
| **54** | *Epilobium hirsutum* | Great Willow Herb | * | * | * | * | * | * | 0.20 | * |

| | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| **55** | *Epilobium montanum* | Broad-Leaved Willowherb | * | 0.60 | 0.12 | * | * | * | * | * |
| **56** | *Epilobium obscurum* | Short Fruited Willowherb | * | * | 0.32 | 0.20 | 0.92 | 0.24 | 1.28 | 0.40 |
| **57** | *Epilobium parviflorum* | Hoary Willowherb | * | * | * | * | * | * | * | * |
| **58** | *Epilobium tetragonum* | Square Stalked Willow Herb | 0.12 | 0.12 | * | * | 0.20 | 0.08 | * | * |
| **59** | *Erica cinerea L.* | Bell-Heather | * | * | * | 0.16 | * | * | * | 0.16 |
| **60** | *Euphrasia nemorosa* | Eyebright | * | 0.12 | 0.60 | 0.52 | 1.00 | 0.24 | 2.88 | 0.28 |
| **61** | *Eurhynchium Praelongum* | Common Feather Moss | * | * | * | 0.40 | * | * | * | * |
| **62** | *Eurhynchium swartzii* | Swartz's Feather Moss | 0.32 | 5.60 | 1.24 | 1.20 | 1.80 | 1.20 | 2.12 | 1.40 |

| | | **Habitat** | **Woodland Rides** | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | **Season/Year** | **Spring 2002** | | **Summer 2002** | | **Spring 2003** | | **Summer 2003** | |
| | | **Treatment** | **R U** | **N-R U** | **R U** | **N-R U** | **R U** | **N-R U** | **R U** | **N-R U** |
| **63** | *Fagus sylvatica* | Beech Seedling | * | * | * | * | * | * | * | * |
| **64** | *Festuca Rubra* | Red Fescue | * | * | * | * | * | * | * | * |
| **65** | *Fissidens taxifolius* | Common Pocket Moss | 1.80 | 0.20 | 0.80 | * | 0.80 | 2.20 | * | * |
| **66** | *Fraxinus excelsior* | Ash Seedling | * | * | * | * | * | 0.08 | * | * |
| **67** | *Fraxinus excelsior* | Ash Sapling | * | * | * | * | * | * | * | * |
| **68** | *Galium aparine* | Cleavers, Goosegrass | 0.28 | 0.72 | 0.68 | 0.56 | 0.88 | * | 0.44 | * |
| **69** | *Galium Palustre* | Common Marsh Bedstraw | 0.12 | 1.32 | 3.80 | 4.24 | 3.64 | 3.16 | 5.44 | 2.76 |
| **70** | *Glechoma hederacea* | Ground Ivy | 0.72 | 2.16 | 6.80 | 4.12 | 2.72 | 3.80 | 3.80 | 1.72 |
| **71** | *Gnaphalium uliginosum* | Marsh Cudweed | 0.16 | * | 0.72 | * | 0.68 | * | 0.72 | 0.08 |
| **72** | *Hedera helix* | Ivy | * | * | * | * | * | * | * | * |
| **73** | *Heracleum Sphondylium* | Hog Weed | * | 0.08 | * | 0.12 | 0.20 | 0.40 | * | 0.16 |
| **74** | *Holcus lanatus* | Yorkshire Fog | 0.52 | 3.68 | 2.64 | 4.08 | 3.72 | 3.60 | 5.32 | 4.08 |
| **75** | *Hordeum Secalinum* | Meadow Barley | * | * | * | * | * | * | * | * |
| **76** | *Hyacinthoides non-scripta* | Bluebell | * | 0.24 | * | * | * | 0.12 | * | * |
| **77** | *Hypericum hirsutum* | Hairy St. Johns-wort | * | * | * | * | 0.24 | 0.04 | * | * |
| **78** | *Hypericum humafusum* | Trailing St. Johns-wort | 0.12 | 0.72 | 1.36 | 0.12 | 1.44 | * | 1.32 | * |
| **79** | *Hypericum maculatum* | Imperforate St. Johns-Wort | * | * | 0.36 | * | 0.28 | * | 0.48 | 0.52 |
| **80** | *Hypericum perforatum* | Perforate St. Johns-wort | * | * | 0.80 | 0.60 | 2.04 | 0.20 | 1.92 | * |
| **81** | *Hypnum jutlandicum* | Heath Plait Moss | 0.20 | * | * | * | * | * | * | * |
| **82** | *Ilex aquifolium* | Holly Seedling | * | * | * | * | * | * | * | * |
| **83** | *Juncus bufonius* | Toad Rush | 0.36 | 3.40 | 1.68 | 3.20 | 3.84 | 2.12 | 4.52 | 2.00 |

|  |  |  |  |  |  |  |  |  |  |  |
|---|---|---|---|---|---|---|---|---|---|---|
| **84** | *Juncus bulbosis* | Bulbous Rush | * | * | 0.20 | * | 0.20 | * | 0.60 | * |
| **85** | *Juncus effusus* | Soft Rush | 0.64 | 0.52 | 0.60 | 0.60 | 2.12 | 0.84 | 1.28 | 0.12 |
| **86** | *Kickxia elatine* | Sharp leaved Fluellen | * | * | * | * | * | * | * | * |
| **87** | *Lamiastrum galeobdolon* | Yellow Archangel | * | * | * | * | * | * | * | * |
| **88** | *Lolium perenne* | Common Rye Grass | * | * | * | * | * | * | * | 0.20 |
| **89** | *Lonicera periclymenum* | Honeysuckle | * | 0.36 | * | 0.20 | * | * | 0.32 | * |
| **90** | *Lotus corniculatus* | Common Bird's-Foot Trefoil | 0.52 | 2.04 | 4.08 | 2.08 | 2.32 | 1.36 | 3.08 | 1.68 |
| **91** | *Luzula campestris* | Field Woodrush | * | * | * | * | * | * | * | * |
| **92** | *Luzula multiflora* | Heath Woodrush | 0.60 | 1.76 | 0.60 | 1.76 | 1.20 | 0.36 | 0.72 | 0.40 |
| **93** | *Luzula pilosa* | Hairy Wood-rush | * | * | * | * | * | * | * | * |

|  |  | **Habitat** | **Woodland Rides** |  |  |  |  |  |  |  |
|---|---|---|---|---|---|---|---|---|---|---|
|  |  | **Season/Year** | **Spring 2002** |  | **Summer 2002** |  | **Spring 2003** |  | **Summer 2003** |  |
|  |  | **Treatment** | **R U** | **N-R U** | **R U** | **N-R U** | **R U** | **N-R U** | **R U** | **N-R U** |
| **94** | *Lychnis flos-cuculi* | Ragged Robin | * | * | * | 0.40 | * | 0.32 | * | 0.60 |
| **95** | *Lysimachia nemorum* | Yellow Pimpernel | * | * | 0.16 | 0.12 | 0.52 | * | 0.12 | * |
| **96** | *Medicago lupulina* | Black Medick | * | * | * | * | * | * | * | * |
| **97** | *Mentha arvensis* | Corn Mint | 0.80 | 0.16 | 3.44 | 0.88 | 0.76 | 1.12 | 1.44 | 2.20 |
| **98** |  | Unidentified Moss | * | * | * | * | * | * | * | * |
| **99** | *Mercurialis perennis* | Dog Mercury | 0.60 | 0.36 | 0.72 | 0.20 | 1.00 | 0.80 | 0.88 | 0.60 |
| **100** | *Milium effusum* | Wood Millet | * | * | * | * | * | * | * | * |
| **101** | *Mnium Hornum* | Swan's Neck Thyme Moss | * | * | * | * | * | * | * | * |
| **102** | *Moehringia trinervia* | Three Nerved Sandwort | * | * | * | * | * | * | * | * |
| **103** | *Montia fontana* | Blinks | 0.04 | * | * | * | * | * | * | * |
| **104** | *Montia perfoliata* | Spring Beauty | 0.04 | * | 1.12 | * | * | * | 0.20 | 0.40 |
| **105** | *Myosotis arvensis* | Field Forget-me-not | * | * | * | * | 0.80 | 0.48 | * | * |
| **106** | *Oxalis acetosella* | Wood Sorrel | * | 0.16 | 0.16 | * | 0.32 | * | 0.08 | * |
| **107** | *Pedicularis sylvatica* | Lousewort | 0.24 | 0.28 | 0.20 | 0.08 | * | * | * | * |
| **108** | *Phleum pratense* | Timothy Grass (Catstail) | * | * | * | * | * | * | * | * |
| **109** | *Pilosella officinarum* | Mouse-ear Hawkweed | * | * | * | 0.32 | 0.32 | 0.32 | 0.12 | 0.20 |
| **110** | *Plagiomnium undulatum* | Hart's Tongue Thyme Moss | * | * | 0.12 | * | 0.52 | 0.12 | 0.32 | * |
| **111** | *Plantago major* | Greater Plantain | * | 0.28 | 0.72 | 0.44 | 0.80 | 0.40 | 1.08 | 0.16 |
| **112** | *Poa annua* | Annual Meadow Grass | * | * | * | * | * | * | * | * |

|  |  |  |  |  |  |  |  |  |  |  |
|---|---|---|---|---|---|---|---|---|---|---|
| **113** | *Poa Nemoralis* | Wood Meadow Grass | * | * | 0.16 | 0.20 | 0.20 | 0.20 | * | 0.32 |
| **114** | *Poa pratensis* | Smooth Stalked Meadow Grass | * | * | * | * | * | * | * | * |
| **115** | *Poa trivialis* | Rough Stalked Meadow Grass | 0.60 | 2.40 | 1.00 | 2.40 | 0.32 | 1.60 | 1.20 | 1.72 |
| **116** | *Polygala serpyllifolia* | Heath Milkwort | 0.12 | 0.04 | * | 0.12 | 0.04 | * | * | * |
| **117** | *Polygala Vulgaris* | Common Milkwort | * | * | * | 0.08 | * | 0.12 | * | * |
| **118** | *Polytrichum formosum* | Bank Haircap Moss | * | 4.92 | 0.40 | 5.24 | 1.00 | 4.96 | 0.60 | 5.60 |
| **119** | *Potentilla anserina* | Silver Weed | 2.00 | 0.12 | 3.52 | 0.28 | 0.52 | 0.20 | 0.72 | 0.20 |
| **120** | *Potentilla erecta* | Tormentil | 1.40 | 0.60 | * | * | * | * | * | * |
| **121** | *Potentilla reptans* | Creeping Cinquefoil | 18.40 | 16.60 | 33.48 | 20.80 | 30.04 | 21.40 | 32.20 | 23.16 |
| **122** | *Potentilla sterilis* | Barren Strawberry | * | 0.16 | 0.84 | 0.36 | 0.56 | * | 1.44 | 0.20 |
| **123** | *Pottia trunchator* | Common Pottia Moss | * | * | * | * | * | * | * | * |
| **124** | *Primula vulgaris* | Primrose | 0.32 | 1.20 | 0.28 | 0.80 | * | 0.80 | 0.20 | 0.32 |

|  |  | **Habitat** | **Woodland Rides** |  |  |  |  |  |  |  |
|---|---|---|---|---|---|---|---|---|---|---|
|  |  | **Season/Year** | **Spring 2002** |  | **Summer 2002** |  | **Spring 2003** |  | **Summer 2003** |  |
|  |  | **Treatment** | **R U** | **N-R U** | **R U** | **N-R U** | **R U** | **N-R U** | **R U** | **N-R U** |
| **125** | *Prunella vulgaris* | Selfheal | 2.24 | 2.60 | 12.28 | 4.36 | 12.60 | 3.80 | 11.40 | 2.72 |
| **126** | *Prunus avium* | Wild Cherry Seedling | * | * | * | * | * | * | * | * |
| **127** | *Prunus spinosa* | Sloe Seedling / Blackthorn | * | * | 0.24 | * | * | * | 0.20 | * |
| **128** | *Pseudoscleropodium purum* | Neat Feather Moss | 3.28 | 19.60 | 4.36 | 21.20 | 4.88 | 17.00 | 6.44 | 18.32 |
| **129** | *Pteridium aquilinum* | Bracken | 0.12 | 6.80 | * | 8.60 | 0.20 | 2.20 | * | 4.60 |
| **130** | *Pulicaria dysenterica* | Common Fleabane | * | * | * | 0.12 | * | 0.08 | * | * |
| **131** | *Quercus robur* | Oak Sapling | * | * | * | * | * | * | 0.20 | * |
| **132** | *Quercus robur* | Pedunculate Oak Seedling | * | * | * | * | 0.20 | * | 0.36 | * |
| **133** | *Ranunculus parviflorus* | Small-flowered Buttercup | * | * | 0.60 | 0.08 | 0.72 | 0.64 | 0.48 | * |
| **134** | *Ranunculus ficaria* | Lesser Celendine | * | * | * | * | * | * | * | * |
| **135** | *Ranunculus flammula* | Lesser Spearwort | * | 3.60 | 0.24 | 4.12 | 0.44 | 3.12 | 1.08 | 2.80 |
| **136** | *Ranunculus repens* | Creeping Buttercup | 2.04 | 3.52 | 2.24 | 2.88 | 5.36 | 2.68 | 3.44 | 1.60 |
| **137** | *Rhytidiadelphus squarrosus* | Springy Tough Moss | 9.92 | 16.40 | 18.60 | 17.40 | 21.92 | 20.00 | 24.80 | 18.60 |
| **138** | *Rosa canina* | Common Dog Rose | * | * | * | 0.12 | * | 0.32 | * | 0.40 |
| **139** | *Rubus fruticosus* | Bramble | 3.76 | 6.96 | 11.16 | 9.80 | 13.52 | 11.72 | 25.36 | 16.44 |
| **140** | *Rumex acetosa* | Common  Sorrel | * | * | * | * | * | * | * | * |
| **141** | *Rumex acetosella* | Sheep Sorrel | 0.12 | 0.12 | 0.88 | 0.04 | 0.52 | 0.40 | 0.76 | 0.24 |

|  |  |  |  |  |  |  |  |  |  |  |
|---|---|---|---|---|---|---|---|---|---|---|
| **142** | *Rumex obtusifolius* | Broad leaved Dock | * | * | * | * | * | * | * | * |
| **143** | *Rumex sanguineus* | Wood Dock | * | * | * | * | 0.20 | * | * | * |
| **144** | *Sagina apetala* | Common Annual Pearlwort | * | * | * | * | * | * | * | * |
| **145** | *Sagina procumbens* | Procumbent Pearlwort | 0.16 | 0.40 | * | 0.40 | 0.64 | 0.40 | 2.44 | 0.40 |
| **146** | *Salix caprea* | Goat Willow Sapling | * | * | * | * | * | * | 0.60 | * |
| **147** | *Sambucus nigra* | Elder Seedling | * | * | * | * | * | * | * | * |
| **148** | *Sarothamnus scoparius* | Broom | * | * | * | * | * | * | * | * |
| **149** | *Senecio jacobaea* | Common Ragwort | * | 0.52 | 2.16 | 0.96 | 1.04 | 1.40 | 1.16 | 0.92 |
| **150** | *Silene dioica* | Red Campion | * | * | * | * | * | * | * | * |
| **151** | *Sonchus oleraceus* | Smooth Sow Thistle | * | * | * | * | * | * | * | * |
| **152** | *Stachys arvensis* | Field Woundwort | * | * | * | * | * | * | * | * |
| **153** | *Stachys sylvatica* | Hedge Woundwort | 0.12 | 0.24 | 0.68 | 0.24 | 1.04 | 0.64 | 0.72 | 0.08 |
| **154** | *Stellaria graminea* | Lesser Stitchwort | * | 2.24 | * | * | * | 2.60 | * | * |
| **155** | *Stellaria media* | Common Chickweed | * | * | * | * | * | * | * | * |

|  |  | **Habitat** | **Woodland Rides** |  |  |  |  |  |  |  |
|---|---|---|---|---|---|---|---|---|---|---|
|  |  | **Season/Year** | **Spring 2002** |  | **Summer 2002** |  | **Spring 2003** |  | **Summer 2003** |  |
|  |  | **Treatment** | **R U** | **N-R U** | **R U** | **N-R U** | **R U** | **N-R U** | **R U** | **N-R U** |
| **156** | *Stellaria neglecta* | Greater Chickweed | * | * | * | * | * | * | * | * |
| **157** | *Taraxacum officinale* | Dandelion | * | * | * | * | * | * | * | * |
| **158** | *Teucrium scorodonia* | Wood Sage | 0.44 | 4.20 | 1.60 | 4.72 | 2.44 | 5.92 | 2.76 | 5.20 |
| **159** | *Thuidium tamariscinum* | Common Tamarisk Moss | * | 4.00 | * | 3.60 | * | 0.48 | * | 1.20 |
| **160** | *Trifolium dubium* | Lesser Trefoil | * | * | * | * | * | * | * | * |
| **161** | *Trifolium micranthum* | Slender Trefoil | * | 1.60 | * | * | * | * | * | * |
| **162** | *Trifolium repens* | White Clover | * | 0.88 | * | 0.20 | 0.12 | 0.80 | * | 0.68 |
| **163** | *Tussilago farfara* | Coltsfoot | * | * | * | * | * | * | * | * |

| | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **164** | *Urtica dioica* | Stinging Nettle | 0.08 | 0.12 | | 0.20 | 0.16 | | 0.16 | 0.32 | | 0.32 | 0.12 |
| **165** | *Veronica arvensis* | Wall Speedwell | 0.08 | 0.36 | | * | * | | * | * | | * | * |
| **166** | *Veronica montana* | Wood Speedwell | * | * | | * | * | | * | * | | * | * |
| **167** | *Veronica officinalis* | Heath Speedwell | 0.04 | 1.68 | | 0.36 | 0.36 | | 1.60 | 0.80 | | 1.12 | 0.92 |
| **168** | *Veronica persica* | Common Field Speedwell | * | * | | * | * | | * | * | | * | * |
| **169** | *Veronica serpyllifolia* | Thyme Leaved Speedwell | * | * | | * | * | | 0.72 | 0.16 | | 0.08 | 0.12 |
| **170** | *Vicia sativa* | Common Vetch | * | * | | * | * | | 0.32 | * | | * | * |
| **171** | *Viola lactea* | Pale Dog Violet | 0.16 | 0.40 | | * | * | | 1.68 | 2.80 | | * | * |
| **172** | *Viola riviniana* | Common Dog Violet | 1.08 | 6.44 | | 2.96 | 7.56 | | 5.20 | 7.08 | | 5.76 | 5.40 |
| **173** | *Vulpia bromoides* | Squirrel-Tail Fescue | 0.20 | 0.40 | | 0.40 | * | | 1.52 | 0.60 | | 0.40 | * |

**Appendix 4:** Mean % covers for each species present per treatment, per season within woodland ride habitats during data collections in **2002** and **2003** (**Chapter 2**). **RU** = Rooted Unfenced, **N-R U** = Non-Rooted Unfenced, **RF** = Rooted Fenced, **N-R F** = Non-Rooted Fenced. * = species not present.

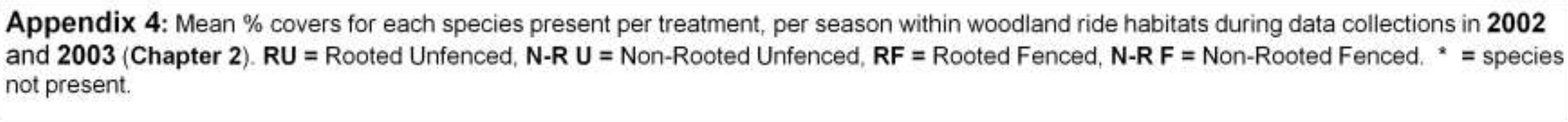
| | | **Habitat** | **Woodland Rides** | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | **Season/Year** | **Spring 2002** | | **Summer 2002** | | **Spring 2003** | | **Summer 2003** | |
| | | **Treatment** | **R U** | **N-R U** | **R U** | **N-R U** | **R U** | **N-R U** | **R U** | **N-R U** |
| **1** | *Acer campestre* | Field Maple Seedling | * | * | * | * | * | * | * | * |
| **2** | *Agrostis canina* | Brown Bent Grass | 0.92 | 16.80 | 2.52 | 15.40 | 8.44 | 12.20 | 10.72 | 15.60 |
| **3** | *Agrostis capillaris* | Common Bent Grass | 0.60 | 6.12 | 4.84 | 8.68 | 6.52 | 9.44 | 6.76 | 10.52 |
| **4** | *Agrostis stolonifera* | Creeping Bent Grass | * | 0.80 | * | * | * | * | * | * |
| **5** | *Ajuga reptans* | Bugle | 1.72 | 2.88 | 4.24 | 3.60 | 4.52 | 1.84 | 5.76 | 1.96 |
| **6** | *Anagallis arvensis* | Scarlet Pimpernel | 0.48 | 0.08 | 0.56 | 0.12 | 2.20 | 0.16 | 0.12 | * |
| **7** | *Anemone nemorosa* | Wood Anemone | 0.12 | 0.24 | * | * | * | 0.52 | * | * |
| **8** | *Anthoxanthum odoratum* | Sweet Vernal Grass | 0.32 | 4.16 | 1.20 | 4.20 | 3.64 | 7.00 | 1.60 | 3.80 |
| **9** | *Arctium Lappa L.* | Greater Burdock | * | * | * | 0.16 | * | * | 0.20 | * |
| **10** | *Arrhenatherum Elatius* | False Oat Grass | * | * | * | * | 0.92 | 0.40 | 0.52 | * |
| **11** | *Atrichum undulatum* | Common Smooth Cap | * | * | * | * | * | * | * | * |
| **12** | *Bellis perennis L.* | Common Daisy | * | * | * | * | * | * | * | * |
| **13** | *Betonica officinalis* | Betony | * | * | 0.28 | * | 0.48 | * | 0.12 | * |
| **14** | *Betula pendula* | Silver Birch Seedling | 0.44 | 0.68 | 2.88 | 1.48 | 1.00 | 0.20 | 1.56 | 0.16 |
| **15** | *Betula pendula* | Silver Birch Sapling | * | * | * | * | 3.60 | 2.60 | 4.32 | 3.60 |
| **16** | *Brachypodium sylvatica* | Wood False Brome | * | * | * | * | * | * | * | * |
| **17** | *Brachythecium rutabulum* | Rough Stalked Feather Moss | * | * | * | * | * | * | * | * |
| **18** | *Bromus ramosus* | Wood Brome Grass | * | 0.40 | * | * | * | * | * | * |
| **19** | *Calliergon cuspidatum* | Pointed Spear Moss | 1.00 | 3.00 | 0.80 | 2.80 | 0.40 | 0.32 | 2.00 | 3.40 |
| **20** | *Calluna Vulgaris* | Ling | * | 0.44 | * | 0.60 | * | 0.92 | 0.20 | 1.32 |
| **21** | *Cardamine pratensis* | Cuckooflower | * | * | * | * | * | * | * | * |
| **22** | *Carex pendula* | Pendulous Sedge | * | * | 0.28 | 0.52 | 0.48 | 0.40 | 0.84 | 0.20 |
| **23** | *Carex sylvatica* | Wood Sedge | * | * | * | * | * | * | * | * |
| **24** | *Carpinus betulus* | Hornbeam Seedling | 0.16 | 0.32 | 0.08 | 0.48 | 0.88 | 0.36 | 0.52 | 0.36 |
| **25** | *Carpinus betulus* | Hornbeam Sapling | * | * | * | * | * | * | * | * |

|  |  |  |  |  |  |  |  |  |  |  |
|---|---|---|---|---|---|---|---|---|---|---|
| **26** | *Castanea Sativa* | Sweet Chestnut Seedling | * | * | 0.08 | * | * | * | * | * |
| **27** | *Centaurea nigra* | Common knapweed | * | 0.40 | * | 0.40 | * | 0.20 | * | * |
| **28** | *Centaurium erythraea* | Common Centaury | * | * | * | 0.08 | 0.32 | 0.20 | 0.80 | 0.16 |
| **29** | *Centaurium pulchellum* | Lesser Centaury | * | * | 0.16 | 0.20 | * | * | * | * |
| **30** | *Cerastium arvense* | Field Mouse-ear | * | * | * | * | * | * | * | * |
| **31** | *Cerastium holosteoides* | Common Mouse Ear | * | 0.60 | 0.44 | 0.76 | 0.80 | 0.32 | 0.12 | 0.20 |

**Appendix 4:** Mean % covers for each species present per treatment, per season within woodland ride habitats during data collections in **2002** and **2003** (**Chapter 2**). **RU** = Rooted Unfenced, **N-R U** = Non-Rooted Unfenced, **RF** = Rooted Fenced, **N-R F** = Non-Rooted Fenced. * = species not present.

|  |  | **Habitat** | **Woodland Rides** |  |  |  |  |  |  |  |
|---|---|---|---|---|---|---|---|---|---|---|
|  |  | **Season/Year** | **Spring 2002** |  | **Summer 2002** |  | **Spring 2003** |  | **Summer 2003** |  |
|  |  | **Treatment** | **R U** | **N-R U** | **R U** | **N-R U** | **R U** | **N-R U** | **R U** | **N-R U** |
| **32** | *Chamaenerion angustifolium* | Rosebay Willowherb | 8.48 | 1.20 | 19.16 | 6.00 | 9.32 | 4.40 | 9.92 | 3.12 |
| **33** | *Chenopodium botryodes* | Small Red Goosefoot | * | * | * | * | * | * | * | * |
| **34** | *Chenopodium polyspermum* | Many-Seeded Goosefoot | * | * | * | * | 0.12 | 0.20 | * | * |
| **35** | *Chenopodium rubrum* | Red Goosefoot | * | * | * | * | * | * | * | * |
| **36** | *Circaea lutetiana* | Enchanter's Nightshade | * | 0.28 | 0.32 | 0.36 | 0.68 | 0.48 | 0.56 | 0.48 |
| **37** | *Cirsium arvense* | Creeping Thistle | 0.40 | 0.24 | 3.80 | 3.80 | 3.20 | 0.80 | 2.80 | 1.00 |
| **38** | *Cirsium palustre* | Marsh Thistle | 0.20 | 0.28 | 0.60 | 0.60 | 0.56 | 0.40 | 0.32 | 0.36 |
| **39** | *Cirsium Vulgare* | Spear Thistle | 0.40 | 0.20 | 0.60 | 0.32 | * | * | 1.40 | * |
| **40** | *Conopodium majus* | Pignut | * | * | * | * | * | * | * | * |
| **41** | *Coronopus squamatus* | Swinecress | * | * | * | * | * | * | * | * |
| **42** | *Corylus avellana* | Hazel Seedling | * | * | 0.20 | * | * | * | * | * |
| **43** | *Corylus avellana* | Hazel Sapling | * | * | * | * | 0.40 | * | 0.80 | * |
| **44** | *Crataegus monogyna* | Hawthorn Seedling | * | * | * | * | * | * | * | * |
| **45** | *Cynosurus cristatus* | Crested Dogs Tail | * | * | * | * | * | * | * | * |
| **46** | *Dactylis glomerata* | Cocksfoot | * | * | * | * | * | * | * | * |
| **47** | *Dactylorhiza fuchsii* | Common Spotted Orchid | * | * | * | * | * | * | * | * |
| **48** | *Deschampsia caespitosa* | Tufted Hair-grass | * | * | * | * | * | * | * | * |
| **49** | *Dicranum Scoparium* | Broom Fork Moss | 0.16 | 0.40 | 0.36 | 3.40 | 0.64 | 3.20 | 0.52 | 3.40 |
| **50** | *Dicranella heteromalla* | Silky Forklet Moss | * | * | * | * | * | * | * | * |
| **51** | *Digitalis purpurea* | Foxglove | * | 0.08 | 0.64 | 0.12 | 1.24 | * | 1.64 | 0.80 |

| | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **52** | *Dryopteris austriaca* | Broad Buckler Fern | * | * | | * | * | | * | * | | * | * |
| **53** | *Dryopteris filix-mas* | Common Male Fern | * | * | | * | * | | * | * | | * | * |
| **54** | *Epilobium hirsutum* | Great Willow Herb | * | * | | * | * | | * | * | | 0.20 | * |
| **55** | *Epilobium montanum* | Broad-Leaved Willowherb | * | 0.60 | | 0.12 | * | | * | * | | * | * |
| **56** | *Epilobium obscurum* | Short Fruited Willowherb | * | * | | 0.32 | 0.20 | | 0.92 | 0.24 | | 1.28 | 0.40 |
| **57** | *Epilobium parviflorum* | Hoary Willowherb | * | * | | * | * | | * | * | | * | * |
| **58** | *Epilobium tetragonum* | Square Stalked Willow Herb | 0.12 | 0.12 | | * | * | | 0.20 | 0.08 | | * | * |
| **59** | *Erica cinerea L.* | Bell-Heather | * | * | | * | 0.16 | | * | * | | * | 0.16 |
| **60** | *Euphrasia nemorosa* | Eyebright | * | 0.12 | | 0.60 | 0.52 | | 1.00 | 0.24 | | 2.88 | 0.28 |
| **61** | *Eurhynchium Praelongum* | Common Feather Moss | * | * | | * | 0.40 | | * | * | | * | * |
| **62** | *Eurhynchium swartzii* | Swartz's Feather Moss | 0.32 | 5.60 | | 1.24 | 1.20 | | 1.80 | 1.20 | | 2.12 | 1.40 |



**Appendix 4:** Mean % covers for each species present per treatment, per season within woodland ride habitats during data collections in **2002** and **2003** (**Chapter 2**). **RU** = Rooted Unfenced, **N-R U** = Non-Rooted Unfenced, **RF** = Rooted Fenced, **N-R F** = Non-Rooted Fenced. * = species not present.

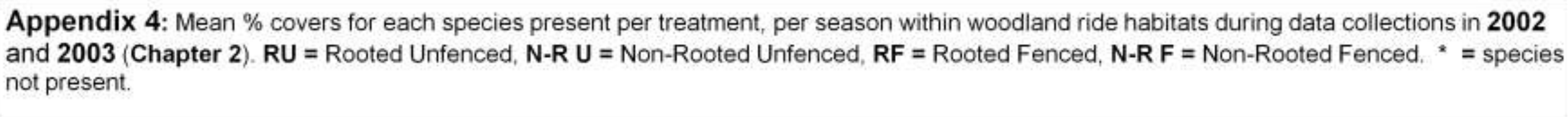
| | | **Habitat** | **Woodland Rides** | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | **Season/Year** | **Spring 2002** | | | **Summer 2002** | | | **Spring 2003** | | | **Summer 2003** | |
| | | **Treatment** | **R U** | **N-R U** | | **R U** | **N-R U** | | **R U** | **N-R U** | | **R U** | **N-R U** |
| **63** | *Fagus sylvatica* | Beech Seedling | * | * | | * | * | | * | * | | * | * |
| **64** | *Festuca Rubra* | Red Fescue | * | * | | * | * | | * | * | | * | * |
| **65** | *Fissidens taxifolius* | Common Pocket Moss | 1.80 | 0.20 | | 0.80 | * | | 0.80 | 2.20 | | * | * |
| **66** | *Fraxinus excelsior* | Ash Seedling | * | * | | * | * | | * | 0.08 | | * | * |
| **67** | *Fraxinus excelsior* | Ash Sapling | * | * | | * | * | | * | * | | * | * |
| **68** | *Galium aparine* | Cleavers, Goosegrass | 0.28 | 0.72 | | 0.68 | 0.56 | | 0.88 | * | | 0.44 | * |
| **69** | *Galium Palustre* | Common Marsh Bedstraw | 0.12 | 1.32 | | 3.80 | 4.24 | | 3.64 | 3.16 | | 5.44 | 2.76 |
| **70** | *Glechoma hederacea* | Ground Ivy | 0.72 | 2.16 | | 6.80 | 4.12 | | 2.72 | 3.80 | | 3.80 | 1.72 |
| **71** | *Gnaphalium uliginosum* | Marsh Cudweed | 0.16 | * | | 0.72 | * | | 0.68 | * | | 0.72 | 0.08 |
| **72** | *Hedera helix* | Ivy | * | * | | * | * | | * | * | | * | * |
| **73** | *Heracleum Sphondylium* | Hog Weed | * | 0.08 | | * | 0.12 | | 0.20 | 0.40 | | * | 0.16 |
| **74** | *Holcus lanatus* | Yorkshire Fog | 0.52 | 3.68 | | 2.64 | 4.08 | | 3.72 | 3.60 | | 5.32 | 4.08 |
| **75** | *Hordeum Secalinum* | Meadow Barley | * | * | | * | * | | * | * | | * | * |
| **76** | *Hyacinthoides non-scripta* | Bluebell | * | 0.24 | | * | * | | * | 0.12 | | * | * |
| **77** | *Hypericum hirsutum* | Hairy St. Johns-wort | * | * | | * | * | | 0.24 | 0.04 | | * | * |

| | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| **78** | *Hypericum humafusum* | Trailing St. Johns-wort | 0.12 | 0.72 | 1.36 | 0.12 | 1.44 | * | 1.32 | * |
| **79** | *Hypericum maculatum* | Imperforate St. Johns-Wort | * | * | 0.36 | * | 0.28 | * | 0.48 | 0.52 |
| **80** | *Hypericum perforatum* | Perforate St. Johns-wort | * | * | 0.80 | 0.60 | 2.04 | 0.20 | 1.92 | * |
| **81** | *Hypnum jutlandicum* | Heath Plait Moss | 0.20 | * | * | * | * | * | * | * |
| **82** | *Ilex aquifolium* | Holly Seedling | * | * | * | * | * | * | * | * |
| **83** | *Juncus bufonius* | Toad Rush | 0.36 | 3.40 | 1.68 | 3.20 | 3.84 | 2.12 | 4.52 | 2.00 |
| **84** | *Juncus bulbosis* | Bulbous Rush | * | * | 0.20 | * | 0.20 | * | 0.60 | * |
| **85** | *Juncus effusus* | Soft Rush | 0.64 | 0.52 | 0.60 | 0.60 | 2.12 | 0.84 | 1.28 | 0.12 |
| **86** | *Kickxia elatine* | Sharp leaved Fluellen | * | * | * | * | * | * | * | * |
| **87** | *Lamiastrum galeobdolon* | Yellow Archangel | * | * | * | * | * | * | * | * |
| **88** | *Lolium perenne* | Common Rye Grass | * | * | * | * | * | * | * | 0.20 |
| **89** | *Lonicera periclymenum* | Honeysuckle | * | 0.36 | * | 0.20 | * | * | 0.32 | * |
| **90** | *Lotus corniculatus* | Common Bird's-Foot Trefoil | 0.52 | 2.04 | 4.08 | 2.08 | 2.32 | 1.36 | 3.08 | 1.68 |
| **91** | *Luzula campestris* | Field Woodrush | * | * | * | * | * | * | * | * |
| **92** | *Luzula multiflora* | Heath Woodrush | 0.60 | 1.76 | 0.60 | 1.76 | 1.20 | 0.36 | 0.72 | 0.40 |
| **93** | *Luzula pilosa* | Hairy Wood-rush | * | * | * | * | * | * | * | * |

**Appendix 4:** Mean % covers for each species present per treatment, per season within woodland ride habitats during data collections in **2002** and **2003** (**Chapter 2**). **RU =** Rooted Unfenced, **N-R U =** Non-Rooted Unfenced, **RF =** Rooted Fenced, **N-R F =** Non-Rooted Fenced. * = species not present.

| | | **Habitat** | **Woodland Rides** | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | **Season/Year** | **Spring 2002** | | **Summer 2002** | | **Spring 2003** | | **Summer 2003** | |
| | | **Treatment** | **R U** | **N-R U** | **R U** | **N-R U** | **R U** | **N-R U** | **R U** | **N-R U** |
| **94** | *Lychnis flos-cuculi* | Ragged Robin | * | * | * | 0.40 | * | 0.32 | * | 0.60 |
| **95** | *Lysimachia nemorum* | Yellow Pimpernel | * | * | 0.16 | 0.12 | 0.52 | * | 0.12 | * |
| **96** | *Medicago lupulina* | Black Medick | * | * | * | * | * | * | * | * |
| **97** | *Mentha arvensis* | Corn Mint | 0.80 | 0.16 | 3.44 | 0.88 | 0.76 | 1.12 | 1.44 | 2.20 |
| **98** | | Unidentified Moss | * | * | * | * | * | * | * | * |
| **99** | *Mercurialis perennis* | Dog Mercury | 0.60 | 0.36 | 0.72 | 0.20 | 1.00 | 0.80 | 0.88 | 0.60 |
| **100** | *Milium effusum* | Wood Millet | * | * | * | * | * | * | * | * |
| **101** | *Mnium Hornum* | Swan's Neck Thyme Moss | * | * | * | * | * | * | * | * |
| **102** | *Moehringia trinervia* | Three Nerved Sandwort | * | * | * | * | * | * | * | * |
| **103** | *Montia fontana* | Blinks | 0.04 | * | * | * | * | * | * | * |

| | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| **104** | *Montia perfoliata* | Spring Beauty | 0.04 | * | 1.12 | * | * | * | 0.20 | 0.40 |
| **105** | *Myosotis arvensis* | Field Forget-me-not | * | * | * | * | 0.80 | 0.48 | * | * |
| **106** | *Oxalis acetosella* | Wood Sorrel | * | 0.16 | 0.16 | * | 0.32 | * | 0.08 | * |
| **107** | *Pedicularis sylvatica* | Lousewort | 0.24 | 0.28 | 0.20 | 0.08 | * | * | * | * |
| **108** | *Phleum pratense* | Timothy Grass (Catstail) | * | * | * | * | * | * | * | * |
| **109** | *Pilosella officinarum* | Mouse-ear Hawkweed | * | * | * | 0.32 | 0.32 | 0.32 | 0.12 | 0.20 |
| **110** | *Plagiomnium undulatum* | Hart's Tongue Thyme Moss | * | * | 0.12 | * | 0.52 | 0.12 | 0.32 | * |
| **111** | *Plantago major* | Greater Plantain | * | 0.28 | 0.72 | 0.44 | 0.80 | 0.40 | 1.08 | 0.16 |
| **112** | *Poa annua* | Annual Meadow Grass | * | * | * | * | * | * | * | * |
| **113** | *Poa Nemoralis* | Wood Meadow Grass | * | * | 0.16 | 0.20 | 0.20 | 0.20 | * | 0.32 |
| **114** | *Poa pratensis* | Smooth Stalked Meadow Grass | * | * | * | * | * | * | * | * |
| **115** | *Poa trivialis* | Rough Stalked Meadow Grass | 0.60 | 2.40 | 1.00 | 2.40 | 0.32 | 1.60 | 1.20 | 1.72 |
| **116** | *Polygala serpyllifolia* | Heath Milkwort | 0.12 | 0.04 | * | 0.12 | 0.04 | * | * | * |
| **117** | *Polygala Vulgaris* | Common Milkwort | * | * | * | 0.08 | * | 0.12 | * | * |
| **118** | *Polytrichum formosum* | Bank Haircap Moss | * | 4.92 | 0.40 | 5.24 | 1.00 | 4.96 | 0.60 | 5.60 |
| **119** | *Potentilla anserina* | Silver Weed | 2.00 | 0.12 | 3.52 | 0.28 | 0.52 | 0.20 | 0.72 | 0.20 |
| **120** | *Potentilla erecta* | Tormentil | 1.40 | 0.60 | * | * | * | * | * | * |
| **121** | *Potentilla reptans* | Creeping Cinquefoil | 18.40 | 16.60 | 33.48 | 20.80 | 30.04 | 21.40 | 32.20 | 23.16 |
| **122** | *Potentilla sterilis* | Barren Strawberry | * | 0.16 | 0.84 | 0.36 | 0.56 | * | 1.44 | 0.20 |
| **123** | *Pottia trunchator* | Common Pottia Moss | * | * | * | * | * | * | * | * |
| **124** | *Primula vulgaris* | Primrose | 0.32 | 1.20 | 0.28 | 0.80 | * | 0.80 | 0.20 | 0.32 |



**Appendix 4:** Mean % covers for each species present per treatment, per season within woodland ride habitats during data collections in **2002** and **2003** (**Chapter 2**). **RU** = Rooted Unfenced, **N-R U** = Non-Rooted Unfenced, **RF** = Rooted Fenced, **N-R F** = Non-Rooted Fenced. * = species not present.

| | | **Habitat** | **Woodland Rides** | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | **Season/Year** | **Spring 2002** | | **Summer 2002** | | **Spring 2003** | | **Summer 2003** | |
| | | **Treatment** | **R U** | **N-R U** | **R U** | **N-R U** | **R U** | **N-R U** | **R U** | **N-R U** |
| **125** | *Prunella vulgaris* | Selfheal | 2.24 | 2.60 | 12.28 | 4.36 | 12.60 | 3.80 | 11.40 | 2.72 |
| **126** | *Prunus avium* | Wild Cherry Seedling | * | * | * | * | * | * | * | * |
| **127** | *Prunus spinosa* | Sloe Seedling / Blackthorn | * | * | 0.24 | * | * | * | 0.20 | * |
| **128** | *Pseudoscleropodium purum* | Neat Feather Moss | 3.28 | 19.60 | 4.36 | 21.20 | 4.88 | 17.00 | 6.44 | 18.32 |
| **129** | *Pteridium aquilinum* | Bracken | 0.12 | 6.80 | * | 8.60 | 0.20 | 2.20 | * | 4.60 |

| | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| **130** | *Pulicaria dysenterica* | Common Fleabane | * | * | * | 0.12 | * | 0.08 | * | * |
| **131** | *Quercus robur* | Oak Sapling | * | * | * | * | * | * | 0.20 | * |
| **132** | *Quercus robur* | Pedunculate Oak Seedling | * | * | * | * | 0.20 | * | 0.36 | * |
| **133** | *Ranunculus parviflorus* | Small-flowered Buttercup | * | * | 0.60 | 0.08 | 0.72 | 0.64 | 0.48 | * |
| **134** | *Ranunculus ficaria* | Lesser Celendine | * | * | * | * | * | * | * | * |
| **135** | *Ranunculus flammula* | Lesser Spearwort | * | 3.60 | 0.24 | 4.12 | 0.44 | 3.12 | 1.08 | 2.80 |
| **136** | *Ranunculus repens* | Creeping Buttercup | 2.04 | 3.52 | 2.24 | 2.88 | 5.36 | 2.68 | 3.44 | 1.60 |
| **137** | *Rhytidiadelphus squarrosus* | Springy Tough Moss | 9.92 | 16.40 | 18.60 | 17.40 | 21.92 | 20.00 | 24.80 | 18.60 |
| **138** | *Rosa canina* | Common Dog Rose | * | * | * | 0.12 | * | 0.32 | * | 0.40 |
| **139** | *Rubus fruticosus* | Bramble | 3.76 | 6.96 | 11.16 | 9.80 | 13.52 | 11.72 | 25.36 | 16.44 |
| **140** | *Rumex acetosa* | Common Sorrel | * | * | * | * | * | * | * | * |
| **141** | *Rumex acetosella* | Sheep Sorrel | 0.12 | 0.12 | 0.88 | 0.04 | 0.52 | 0.40 | 0.76 | 0.24 |
| **142** | *Rumex obtusifolius* | Broad leaved Dock | * | * | * | * | * | * | * | * |
| **143** | *Rumex sanguineus* | Wood Dock | * | * | * | * | 0.20 | * | * | * |
| **144** | *Sagina apetala* | Common Annual Pearlwort | * | * | * | * | * | * | * | * |
| **145** | *Sagina procumbens* | Procumbent Pearlwort | 0.16 | 0.40 | * | 0.40 | 0.64 | 0.40 | 2.44 | 0.40 |
| **146** | *Salix caprea* | Goat Willow Sapling | * | * | * | * | * | * | 0.60 | * |
| **147** | *Sambucus nigra* | Elder Seedling | * | * | * | * | * | * | * | * |
| **148** | *Sarothamnus scoparius* | Broom | * | * | * | * | * | * | * | * |
| **149** | *Senecio jacobaea* | Common Ragwort | * | 0.52 | 2.16 | 0.96 | 1.04 | 1.40 | 1.16 | 0.92 |
| **150** | *Silene dioica* | Red Campion | * | * | * | * | * | * | * | * |
| **151** | *Sonchus oleraceus* | Smooth Sow Thistle | * | * | * | * | * | * | * | * |
| **152** | *Stachys arvensis* | Field Woundwort | * | * | * | * | * | * | * | * |
| **153** | *Stachys sylvatica* | Hedge Woundwort | 0.12 | 0.24 | 0.68 | 0.24 | 1.04 | 0.64 | 0.72 | 0.08 |
| **154** | *Stellaria graminea* | Lesser Stitchwort | * | 2.24 | * | * | * | 2.60 | * | * |
| **155** | *Stellaria media* | Common Chickweed | * | * | * | * | * | * | * | * |

**Appendix 4:** Mean % covers for each species present per treatment, per season within woodland ride habitats during data collections in **2002** and **2003** (**Chapter 2**). **RU =** Rooted Unfenced, **N-R U =** Non-Rooted Unfenced, **RF =** Rooted Fenced, **N-R F =** Non-Rooted Fenced. * = species not present.

| **Habitat** | **Woodland Rides** | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| **Season/Year** | **Spring 2002** | | **Summer 2002** | | **Spring 2003** | | **Summer 2003** | |
| **Treatment** | **R U** | **N-R U** | **R U** | **N-R U** | **R U** | **N-R U** | **R U** | **N-R U** |

| | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **156** | *Stellaria neglecta* | Greater Chickweed | * | * | | * | * | | * | * | | * | * |
| **157** | *Taraxacum officinale* | Dandelion | * | * | | * | * | | * | * | | * | * |
| **158** | *Teucrium scorodonia* | Wood Sage | 0.44 | 4.20 | | 1.60 | 4.72 | | 2.44 | 5.92 | | 2.76 | 5.20 |
| **159** | *Thuidium tamariscinum* | Common Tamarisk Moss | * | 4.00 | | * | 3.60 | | * | 0.48 | | * | 1.20 |
| **160** | *Trifolium dubium* | Lesser Trefoil | * | * | | * | * | | * | * | | * | * |
| **161** | *Trifolium micranthum* | Slender Trefoil | * | 1.60 | | * | * | | * | * | | * | * |
| **162** | *Trifolium repens* | White Clover | * | 0.88 | | * | 0.20 | | 0.12 | 0.80 | | * | 0.68 |
| **163** | *Tussilago farfara* | Coltsfoot | * | * | | * | * | | * | * | | * | * |
| **164** | *Urtica dioica* | Stinging Nettle | 0.08 | 0.12 | | 0.20 | 0.16 | | 0.16 | 0.32 | | 0.32 | 0.12 |
| **165** | *Veronica arvensis* | Wall Speedwell | 0.08 | 0.36 | | * | * | | * | * | | * | * |
| **166** | *Veronica montana* | Wood Speedwell | * | * | | * | * | | * | * | | * | * |
| **167** | *Veronica officinalis* | Heath Speedwell | 0.04 | 1.68 | | 0.36 | 0.36 | | 1.60 | 0.80 | | 1.12 | 0.92 |
| **168** | *Veronica persica* | Common Field Speedwell | * | * | | * | * | | * | * | | * | * |
| **169** | *Veronica serpyllifolia* | Thyme Leaved Speedwell | * | * | | * | * | | 0.72 | 0.16 | | 0.08 | 0.12 |
| **170** | *Vicia sativa* | Common Vetch | * | * | | * | * | | 0.32 | * | | * | * |
| **171** | *Viola lactea* | Pale Dog Violet | 0.16 | 0.40 | | * | * | | 1.68 | 2.80 | | * | * |
| **172** | *Viola riviniana* | Common Dog Violet | 1.08 | 6.44 | | 2.96 | 7.56 | | 5.20 | 7.08 | | 5.76 | 5.40 |
| **173** | *Vulpia bromoides* | Squirrel-Tail Fescue | 0.20 | 0.40 | | 0.40 | * | | 1.52 | 0.60 | | 0.40 | * |

**Appendix 5:** Mean % covers for each species present per treatment, per season within woodland habitats during data collections in **2002, 2003** and **2004** (**Chapter 2** and **3**). **RU =** Rooted Unfenced, **N-R U =** Non-Rooted Unfenced, **RF =** Rooted Fenced, **N-R F =** Non-Rooted Fenced.
*Anemone nemorosa* and *Hyacinthoides non-scripta* data were collected during May. *Narcissus pseudonarcissus* data were collected during March.

| | | **Habitat** | | | | | | | | **Woodland** | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | **Season/Year** | **Spring 2002** | | | | | **Spring 2003** | | | | | **Spring 2004** | | | | |
| | | **Treatment** | **R U** | **N-R U** | **R F** | **N-R F** | | **R U** | **N-R U** | **R F** | **N-R F** | | **R U** | **N-R U** | **R F** | **N-R F** | |
| **a** | *Narcissus pseudonarcissus* | Wild Daffodil | 38.00 | 36.33 | 34.00 | 32.00 | | 42.00 | 37.00 | 36.38 | 36.34 | | 36.00 | 29.00 | 32.00 | 32.26 | |
| **b** | *Anemone nemorosa* | Wood Anemone | 0.20 | 13.96 | 0.04 | 16.04 | | 8.08 | 10.00 | 9.60 | 13.76 | | 15.40 | 18.80 | 11.72 | 19.20 | |
| **c** | *Hyacnthoides non-scripta* | Bluebell | 4.04 | 78.60 | 4.16 | 78.40 | | 19.75 | 71.50 | 26.25 | 78.75 | | 44.25 | 65.75 | 58.50 | 85.75 | |

**Appendix 5:** Mean % covers for each species present per treatment, per season within woodland habitats during data collections in **2002, 2003** and **2004** (**Chapter 2** and **3**). **RU =** Rooted Unfenced, **N-R U =** Non-Rooted Unfenced, **RF =** Rooted Fenced, **N-R F =** Non-Rooted Fenced. *Anemone nemorosa* and *Hyacinthoides non-scripta* data were collected during May. *Narcissus pseudonarcissus* data were collected during March.

| | | **Habitat** | | | | | | **Woodland** | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | **Season/Year** | **Spring 2002** | | | | | **Spring 2003** | | | | | **Spring 2004** | | | | |
| | | **Treatment** | **R U** | **N-R U** | **R F** | **N-R F** | | **R U** | **N-R U** | **R F** | **N-R F** | | **R U** | **N-R U** | **R F** | **N-R F** |
| **a** | *Narcissus pseudonarcissus* | Wild Daffodil | 38.00 | 36.33 | 34.00 | 32.00 | | 42.00 | 37.00 | 36.38 | 36.34 | | 36.00 | 29.00 | 32.00 | 32.26 |
| **b** | *Anemone nemorosa* | Wood Anemone | 0.20 | 13.96 | 0.04 | 16.04 | | 8.08 | 10.00 | 9.60 | 13.76 | | 15.40 | 18.80 | 11.72 | 19.20 |
| **c** | *Hyacnthoides non-scripta* | Bluebell | 4.04 | 78.60 | 4.16 | 78.40 | | 19.75 | 71.50 | 26.25 | 78.75 | | 44.25 | 65.75 | 58.50 | 85.75 |

j

**Appendix 6**: Complete species list incorporating mean numbers of individuals of plant species emerging as seedlings from the seed bank per treatment, per habitat from soil samples under lab conditions (**Chapter 4**). * = species not present.

| | Habitat | Grassland | | Woodland | |
|---|---|---|---|---|---|
| | Treatment | Rooted | Non-Rooted | Rooted | Non-Rooted |
| *Acer campestre* | Field Maple Seedling | * | * | * | * |
| *Agrostis canina* | Brown Bent Grass | * | * | * | * |
| *Agrostis capillaris* | Common Bent Grass | 8.067 | 11.600 | 2.000 | * |
| *Agrostis stolonifera* | Creeping Bent Grass | 1.400 | 10.200 | 1.533 | 0.667 |
| *Ajuga reptans* | Bugle | 0.200 | * | * | * |
| *Anagallis arvensis* | Scarlet Pimpernel | 3.933 | * | * | * |
| *Anemone nemorosa* | Wood Anemone | * | * | * | * |
| *Anthoxanthum odoratum* | Sweet Vernal Grass | * | * | * | * |
| *Arctium Lappa L.* | Greater Burdock | * | * | * | * |
| *Arrhenatherum Elatius* | False Oat Grass | * | * | * | * |
| *Atrichum undulatum* | Common Smooth Cap | * | * | * | * |
| *Bellis perennis L.* | Common Daisy | * | * | * | * |
| *Betonica officinalis* | Betony | * | * | * | * |
| *Betula pendula* | Silver Birch Seedling | * | * | 2.800 | 0.133 |
| *Betula pendula* | Silver Birch Sapling | * | * | * | * |
| *Brachypodium sylvatica* | Wood False Brome | * | * | * | * |
| *Brachythecium rutabulum* | Rough Stalked Feather Moss | * | * | * | * |
| *Bromus ramosus* | Wood Brome Grass | * | * | * | * |
| *Calliergon cuspidatum* | Pointed Spear Moss | * | * | * | * |
| *Calluna Vulgaris* | Ling | * | * | * | * |
| *Cardamine pratensis* | Cuckooflower | 2.067 | 0.733 | * | * |
| *Carex pendula* | Pendulous Sedge | * | * | 1.867 | * |
| *Carex sylvatica* | Wood Sedge | 0.133 | * | 0.867 | 2.000 |
| *Carpinus betulus* | Hornbeam Seedling | * | * | * | * |
| *Carpinus betulus* | Hornbeam Sapling | * | * | * | * |
| *Castanea Sativa* | Sweet Chestnut Seedling | * | * | * | * |
| *Centaurea nigra* | Common knapweed | * | * | * | * |
| *Centaurium erythraea* | Common Centaury | 0.400 | * | * | * |
| *Centaurium pulchellum* | Lesser Centaury | 0.067 | * | * | * |
| *Cerastium arvense* | Field Mouse-ear | 0.267 | 0.133 | * | * |
| *Cerastium holosteoides* | Common Mouse Ear | 6.667 | 2.933 | * | * |
| *Chamaenerion angustifolium* | Rosebay Willowherb | 0.467 | * | 0.133 | 0.267 |
| *Chenopodium botryodes* | Small Red Goosefoot | 0.733 | * | * | * |
| *Chenopodium polyspermum* | Many-Seeded Goosefoot | 10.133 | 0.600 | 0.133 | 0.067 |
| *Chenopodium rubrum* | Red Goosefoot | 0.200 | * | 0.067 | * |
| *Circaea lutetiana* | Enchanter's Nightshade | * | * | * | 0.067 |
| *Cirsium arvense* | Creeping Thistle | 0.067 | * | 0.067 | * |
| *Cirsium palustre* | Marsh Thistle | 0.067 | * | 0.267 | * |
| *Cirsium Vulgare* | Spear Thistle | 0.133 | 0.400 | * | 0.267 |
| *Conopodium majus* | Pignut | * | * | * | * |
| *Coronopus squamatus* | Swinecress | 3.333 | * | * | * |
| *Corylus avellana* | Hazel Seedling | * | * | * | * |
| *Corylus avellana* | Hazel Sapling | * | * | * | * |
| *Crataegus monogyna* | Hawthorn Seedling | * | * | * | * |
| *Cynosurus cristatus* | Crested Dogs Tail | * | * | * | * |
| *Dactylis glomerata* | Cocksfoot | 0.333 | * | * | * |
| *Dactylorhiza fuchsii* | Common Spotted Orchid | * | * | * | * |

| | Habitat | Grassland | | Woodland | |
|---|---|---|---|---|---|
| | Treatment | Rooted | Non-Rooted | Rooted | Non-Rooted |
| *Deschampsia caespitosa* | Tufted Hair-grass | * | * | 0.200 | * |
| *Dicranum Scoparium* | Broom Fork Moss | * | * | * | * |
| *Dicranella heteromalla* | Silky Forklet Moss | * | * | * | * |
| *Digitalis purpurea* | Foxglove | * | * | 41.600 | 2.600 |
| *Dryopteris austriaca* | Broad Buckler Fern | * | * | * | * |
| *Dryopteris filix-mas* | Common Male Fern | * | * | * | * |
| *Epilobium hirsutum* | Great Willow Herb | * | * | * | * |
| *Epilobium montanum* | Broad-Leaved Willowherb | * | * | * | * |
| *Epilobium obscurum* | Short Fruited Willowherb | 0.733 | * | 0.133 | * |
| *Epilobium parviflorum* | Hoary Willowherb | 0.267 | * | 0.133 | * |
| *Epilobium tetragonum* | Square Stalked Willow Herb | 0.467 | * | * | * |
| *Erica cinerea L.* | Bell-Heather | * | * | * | * |
| *Euphrasia nemorosa* | Eyebright | * | * | * | * |
| *Eurhynchium Praelongum* | Common Feather Moss | * | * | * | * |
| *Eurhynchium swartzii* | Swartz's Feather Moss | * | * | * | * |
| *Fagus sylvatica* | Beech Seedling | * | * | * | * |
| *Festuca Rubra* | Red Fescue | * | * | * | * |
| *Fissidens taxifolius* | Common Pocket Moss | * | * | * | * |
| *Fraxinus excelsior* | Ash Seedling | * | * | * | * |
| *Fraxinus excelsior* | Ash Sapling | * | * | * | * |
| *Galium aparine* | Cleavers, Goosegrass | * | * | * | * |
| *Galium Palustre* | Common Marsh Bedstraw | * | * | * | * |
| *Glechoma hederacea* | Ground Ivy | 0.133 | * | 0.067 | 0.333 |
| *Gnaphalium uliginosum* | Marsh Cudweed | 3.333 | 0.267 | 0.133 | 0.733 |
| *Hedera helix* | Ivy | * | * | * | * |
| *Heracleum Sphondylium* | Hog Weed | * | * | * | * |
| *Holcus lanatus* | Yorkshire Fog | 4.467 | 2.933 | 1.000 | * |
| *Hordeum Secalinum* | Meadow Barley | * | * | * | * |
| *Hyacinthoides non-scripta* | Bluebell | * | * | 1.133 | * |
| *Hypericum hirsutum* | Hairy St. Johns-wort | 0.333 | * | 0.200 | * |
| *Hypericum humafusum* | Trailing St. Johns-wort | 0.133 | * | 0.267 | * |
| *Hypericum maculatum* | Imperforate St. Johns-Wort | * | * | * | * |
| *Hypericum perforatum* | Perforate St. Johns-wort | 2.000 | * | 5.667 | 0.333 |
| *Hypnum jutlandicum* | Heath Plait Moss | * | * | * | * |
| *Ilex aquifolium* | Holly Seedling | * | * | * | * |
| *Juncus bufonius* | Toad Rush | 45.733 | 1.933 | 6.067 | 2.533 |
| *Juncus bulbosis* | Bulbous Rush | 13.067 | 1.267 | 22.800 | 57.933 |
| *Juncus effusus* | Soft Rush | 0.267 | * | 4.400 | * |
| *Kickxia elatine* | Sharp leaved Fluellen | * | * | * | * |
| *Lamiastrum galeobdolon* | Yellow Archangel | * | * | * | * |
| *Lolium perenne* | Common Rye Grass | * | * | * | * |
| *Lonicera periclymenum* | Honeysuckle | * | * | * | * |
| *Lotus corniculatus* | Common Bird's-Foot Trefoil | 0.200 | 0.133 | * | * |
| *Luzula campestris* | Field Woodrush | 0.933 | * | * | * |
| *Luzula multiflora* | Heath Woodrush | * | * | * | * |
| *Luzula pilosa* | Hairy Wood-rush | 0.267 | * | 0.533 | 0.133 |
| *Lychnis flos-cuculi* | Ragged Robin | * | * | * | * |
| *Lysimachia nemorum* | Yellow Pimpernel | 0.200 | * | 0.333 | * |
| *Medicago lupulina* | Black Medick | * | * | * | * |
| *Mentha arvensis* | Corn Mint | * | * | * | * |
| | Unidentified Moss | * | * | * | * |
| *Mercurialis perennis* | Dog Mercury | * | * | * | 0.067 |
| *Milium effusum* | Wood Millet | * | * | 0.400 | * |
| *Mnium Hornum* | Swan's Neck Thyme Moss | * | * | * | * |
| *Moehringia trinervia* | Three Nerved Sandwort | * | * | * | * |

| | | | | | |
|---|---|---|---|---|---|
| *Montia fontana* | Blinks | * | * | * | * |
| *Montia perfoliata* | Spring Beauty | * | * | * | * |
| *Myosotis arvensis* | Field Forget-me-not | * | * | * | * |
| *Oxalis acetosella* | Wood Sorrel | * | * | * | * |
| *Pedicularis sylvatica* | Lousewort | 0.133 | * | * | * |
| *Phleum pratense* | Timothy Grass (Catstail) | * | * | * | * |
| *Pilosella officinarum* | Mouse-ear Hawkweed | * | * | * | * |
| *Plagiomnium undulatum* | Hart's Tongue Thyme Moss | * | * | * | * |
| *Plantago major* | Greater Plantain | 2.867 | 2.000 | 0.667 | 0.133 |
| *Poa annua* | Annual Meadow Grass | 0.867 | 0.867 | * | * |
| *Poa Nemoralis* | Wood Meadow Grass | * | * | 1.200 | 0.533 |
| *Poa pratensis* | Smooth Stalked Meadow Grass | * | * | * | * |
| *Poa trivialis* | Rough Stalked Meadow Grass | * | * | * | * |
| *Polygala serpyllifolia* | Heath Milkwort | * | * | * | * |
| *Polygala Vulgaris* | Common Milkwort | * | * | * | * |
| *Polytrichum formosum* | Bank Haircap Moss | * | * | * | * |
| *Potentilla anserina* | Silver Weed | 0.200 | * | * | * |
| *Potentilla erecta* | Tormentil | 0.533 | 0.200 | * | * |
| *Potentilla reptans* | Creeping Cinquefoil | 2.067 | * | * | * |
| *Potentilla sterilis* | Barren Strawberry | * | * | * | * |
| *Pottia trunchator* | Common Pottia Moss | * | * | * | * |
| *Primula vulgaris* | Primrose | 0.133 | * | * | * |
| *Prunella vulgaris* | Selfheal | * | * | * | * |
| *Prunus avium* | Wild Cherry Seedling | * | * | * | * |
| *Prunus spinosa* | Sloe Seedling / Blackthorn | * | * | * | * |
| *Pseudoscleropodium purum* | Neat Feather Moss | * | * | * | * |
| *Pteridium aquilinum* | Bracken | * | * | * | * |
| *Pulicaria dysenterica* | Common Fleabane | * | * | * | * |
| *Quercus robur* | Oak Sapling | * | * | * | * |
| *Quercus robur* | Pedunculate Oak Seedling | * | * | * | * |
| *Ranunculus parviflorus* | Small-flowered Buttercup | * | * | * | * |
| *Ranunculus ficaria* | Lesser Celendine | * | * | 0.133 | * |
| *Ranunculus flammula* | Lesser Spearwort | * | * | * | * |
| *Ranunculus repens* | Creeping Buttercup | 31.933 | 6.933 | * | 0.200 |
| *Rhytidiadelphus squarrosus* | Springy Tough Moss | * | * | * | * |
| *Rosa canina* | Common Dog Rose | * | * | * | * |
| *Rubus fruticosus* | Bramble | 0.467 | 0.267 | 1.133 | 4.133 |
| *Rumex acetosa* | Common Sorrel | * | * | * | * |
| *Rumex acetosella* | Sheep Sorrel | * | * | * | * |
| *Rumex obtusifolius* | Broad leaved Dock | 1.867 | 1.733 | 6.867 | 1.333 |
| *Rumex sanguineus* | Wood Dock | * | * | 0.533 | 0.200 |
| *Sagina apetala* | Common Annual Pearlwort | 1.733 | * | * | * |
| *Sagina procumbens* | Procumbent Pearlwort | 14.000 | 32.533 | 0.200 | 0.333 |
| *Salix caprea* | Goat Willow Sapling | * | * | * | * |
| *Sambucus nigra* | Elder Seedling | * | * | * | * |
| *Sarothamnus scoparius* | Broom | * | * | * | * |

| | **Habitat** | **Grassland** | | **Woodland** | |
|---|---|---|---|---|---|
| | **Treatment** | **Rooted** | **Non-Rooted** | **Rooted** | **Non-Rooted** |
| *Senecio jacobaea* | Common Ragwort | 1.000 | 0.067 | * | * |
| *Silene dioica* | Red Campion | * | * | 1.533 | 1.200 |
| *Sonchus oleraceus* | Smooth Sow Thistle | * | * | * | * |
| *Stachys arvensis* | Field Woundwort | * | * | * | * |
| *Stachys sylvatica* | Hedge Woundwort | * | * | 3.667 | 3.667 |
| *Stellaria graminea* | Lesser Stitchwort | * | * | * | * |
| *Stellaria media* | Common Chickweed | 2.333 | 0.133 | * | * |
| *Stellaria neglecta* | Greater Chickweed | * | * | 0.467 | 1.000 |

| | | | | | |
|---|---|---|---|---|---|
| *Taraxacum officinale* | Dandelion | * | * | * | * |
| *Teucrium scorodonia* | Wood Sage | * | * | 2.733 | 0.133 |
| *Thuidium tamariscinum* | Common Tamarisk Moss | * | * | * | * |
| *Trifolium dubium* | Lesser Trefoil | * | * | * | * |
| *Trifolium micranthum* | Slender Trefoil | * | * | * | * |
| *Trifolium repens* | White Clover | * | 0.533 | * | * |
| *Tussilago farfara* | Coltsfoot | * | * | * | * |
| *Urtica dioica* | Stinging Nettle | 0.200 | 2.200 | * | * |
| *Veronica arvensis* | Wall Speedwell | 0.533 | 0.333 | * | * |
| *Veronica montana* | Wood Speedwell | * | * | * | * |
| *Veronica officinalis* | Heath Speedwell | * | * | * | * |
| *Veronica persica* | Common Field Speedwell | 0.600 | * | * | * |
| *Veronica serpyllifolia* | Thyme Leaved Speedwell | 2.667 | 0.733 | * | * |
| *Vicia sativa* | Common Vetch | * | * | * | * |
| *Viola lactea* | Pale Dog Violet | * | * | * | * |
| *Viola riviniana* | Common Dog Violet | * | * | * | * |
| *Vulpia bromoides* | Squirrel-Tail Fescue | * | * | * | * |