Potential for the restoration of lowland and wet grassland upon ex-arable land

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POTENTIAL FOR THE RESTORATION OF LOWLAND WET GRASSLAND UPON EX-ARABLE LAND

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Abstract

Concerns about the impacts of intensive agriculture in the 20th century led to the introduction of policy initiatives intended to halt environmental deterioration and reverse biodiversity losses. In England, agri-environment schemes have enabled de-intensification of agricultural land management and active promotion of habitat types of conservation value within the farmed landscape. One such habitat, lowland wet grassland, is represented within several Environmentally Sensitive Areas (ESAs), including the Upper Thames Tributaries (UTT) ESA. Current UK agri-environment schemes provide the policy context for this study.

An investigation to determine whether soil seed banks of former, and extant, floodplain grasslands could contribute to the restoration of floristic diversity concluded that propagule availability was likely to be a major constraint on restoration and recreation of wet grassland as seed banks are too depauperate for restoration of all species.

At the beginning of the study, there was some doubt as to the efficacy of ESA prescriptions for reversion of arable land to wet grassland, which involved sowing a limited range of grass species only. Site-specific floristic targets for wet grassland re-creation at an ex-arable site in the UTT ESA were derived using a reference habitat. Several treatments, based on the re-introduction of species as seed, were formulated to test whether sowing a wider range of species would be more effective in generating the type of species-rich grassland aimed for under the ESA scheme.

The effectiveness of the seed treatments, including the ESA scheme's recommendation, at re-establishing species-rich wet grassland on ex-arable land was assessed in a field experiment which tested the site-specific targets developed and evaluation criteria. Results concurred with those of the seed bank investigation: restoration of diversity requires the introduction of increased numbers of species. The evaluation criteria developed enable progress towards the target to be quantified, but emphasise that reference conditions must be chosen with care.

Targets developed using a reference habitat were site-specific and unrealistic in the short-term. Objective, catchment-wide targets can be derived from the species distribution dataset for the study area using a number of approaches to enable identification of: (i) extant high quality lowland wet grassland – to be protected and to act as 'sources' of propagules for restoration; (ii) priority sites for restoration ('sink' fields), according to their potential to be restored to the target habitat; (iii) species that are constant in extant wet grasslands and that should form the basis of species-rich seed mixtures; and (iv) habitat-specific 'indicator species' to evaluate restoration success.

Re-creation of characteristic lowland wet grassland in the UTT ESA will be possible, although early ESA recommendations for reversion of arable land, based on simple management prescriptions and low intervention, will not achieve even the poorly-defined scheme objectives. The ESA scheme could make an increased contribution to the promotion of biodiversity within the UK by targeting high quality wet grasslands for protection, and sites for restoration based upon the ease with which species-rich grassland could be established. The 'value for money' of the scheme could be increased by careful selection of species for (re-) introduction and by monitoring the success of restoration using habitat measures based on the characteristics of the target habitat to identify why restoration may be failing and where further intervention may be required.

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Publications arising from the thesis

The following publications (bound at the back of the thesis) were prepared during the period of study:

Manchester, S.J., Treweek, J.R., Mountford, J.O. & Pywell, R.F. 1997. The success of nurse crops in the establishment of species-rich lowland wet grassland. In: *Grassland Management in Environmentally Sensitive Areas*, edited by R.D.Sheldrick, pp. 260-261. Reading: British Grassland Society. (British Grassland Society Occasional Symposium No.32).

Manchester, S.J. & Sparks, T.H. 1998. Determining the seed bank of former flood meadow fields. In: *Aspects of Applied Biology 51, Weed seedbanks: determination, dynamics and manipulation* (edited by G.T. Champion, A.C. Grundy, N.E. Jones, E.J.P. Marshall & R.J. Froud-Williams), pp. 15-22. Wellesbourne: Association of Applied Biologists.

Manchester, S.J., Treweek, J.R., Mountford, J.O., Pywell, R.F. & Sparks, T.H. 1998. Restoration of a target wet grassland community on ex-arable land. In: *European Wet Grasslands: Biodiversity, Management and Restoration.* (edited by C.B. Joyce and P.M. Wade) pp. 277-294. Wiley Publishers, Chichester.

Manchester, S.J., Mountford, J.O., Treweek, J.R. & Sparks, T.H. 1998. Experimental reconstruction and rehabilitation of floodplain grassland. In: *United Kingdom Floodplains* (edited by R.G. Bailey, P.V. José and B.R. Sherwood) chapter 25, pp. 379-394. Westbury Academic and Scientific Publishing.

Manchester, S.J., McNally, S, Treweek, J.R., Sparks, T.H. & Mountford, J.O. 1999. The cost and practicality of techniques for the reversion of arable land to lowland wet grassland - an experimental study and review. *Journal of Environmental Management,* **55**, 91-109.

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CHAPTER 1 INTRODUCTION TO THE STUDY

1.1 Aims of the study

Lowland wet grasslands have declined in the landscape, both in extent and quality, especially in the last 50 years. Whilst processes governing the maintenance and assembly of other grassland types have been studied, lowland wet grasslands have been relatively little researched. This study set out to investigate the potential of an ex-arable site to be successfully restored to lowland wet grassland. A number of specific issues associated with the restoration of wet grassland flora on ex-arable land were addressed within the context of current UK agri-environment schemes:

- The derivation of site-specific floristic targets for the re-creation of lowland wet grassland on ex-arable land;
- The formulation of a range of locally-appropriate seed mixtures to assess the level of propagule introduction necessary to restore species-rich wet grassland;
- The derivation of evaluation criteria suitable to assess the success of restoration;
- An investigation to determine whether the soil seed banks of former, and extant, floodplain grasslands have the potential to contribute to the restoration of floristic diversity;
- Assessment of the effectiveness of a number of different seed treatments (including an agri-environment scheme recommended seed mixture) at re-establishing species-rich wet grassland on ex-arable land. Effectiveness was assessed using the pre-determined evaluation criteria; and
- An investigation of the use of national and local scale species distribution data in the development of objective, appropriate local targets for restoration.

1.2 Lowland wet grassland communities

Wet grasslands occur in areas 'which are periodically flooded or waterlogged by fresh water and where management for agriculture (grazing, mowing or a combination of the two) promotes vegetation dominated by lower growing grasses, sedges and rushes' (RSPB *et al.*, 1997). Wet grasslands encompass a range of wetland types and include semi-natural floodplain grassland, which has developed where floodplains are subjected to a semi-natural hydrological regime (RSPB *et al.*, 1997).

Lowland wet grasslands normally occur in river valleys below 300m (ICOLE, 1994). They are neutral grasslands, i.e. 'semi-natural grasslands whose soil is not markedly alkaline nor very acid, mostly developed on the clays and loams' (Tansley, 1939). Wet grasslands of particular conservation interest are old, moist mesotrophic grasslands, which are not excessively drained or permanently waterlogged (Treweek and Sheail, 1991). Traditional agricultural management practices (associated with low-intensity livestock systems) were responsible for the development, and nature, of many grassland habitats, including lowland wet grasslands where the maintenance of hydrological features has also been important.

Lowland wet grassland is an ecologically valuable, semi-natural, species-rich habitat, supporting 16 of the rare and scarce vascular plants in Britain (Thomas *et al.*, 1995)). In some areas it also supports breeding waders including lapwing *Vanellus vanellus*, redshank *Tringa totanus* and snipe *Gallinago gallinago*, wintering wildfowl and passage birds. Approximately 32 Red Data Book (or candidate) species of birds are at least partly dependent on wet grasslands for breeding or wintering (Thomas *et al.*, 1995). The grasslands also support a wide variety of invertebrates whilst the ditches support approximately 130 of Britain's 170 species of freshwater and brackish water higher plants (Thomas *et al.*, 1995), and are important for aquatic invertebrates, especially dragonflies, water beetles and snails.

The floodplains that sustain wet meadows provide a wider range of benefits than just species conservation, including flood protection, nutrient cycling, reduced water erosion, ground water recharge, and recreational opportunities (Greeson *et al.*, 1979; Hammer, 1992; Hey and Philippi, 1995; Kadlec and Hey, 1994; Petts, 1998). Wet grasslands also perform an important function by reducing agricultural runoff and thus maintaining water quality (Muscutt *et al.*, 1993).

It is estimated that species-rich grassland has declined by over 97% since the middle of the 20th century (Fuller, 1987) and now accounts for less than 2% of the area of permanent lowland grassland in England and Wales (Blackstock *et al.*, 1999). Historically, the area of lowland wet grassland in England and Wales has been estimated at 1,200,000 ha (Thomas *et al.*, 1995). It is probable that only 220,000 ha now remain (Dargie, 1993), with possibly less than 20,000 ha of agriculturally unimproved wet grassland of high conservation value (Thomas *et al.*, 1995). Williams and Bowers (1987) estimated that, since 1930, 40% of the total area of wet grassland in Britain has been lost.

1.3 Reasons for decline

Following the Second World War the need to feed the people of Europe has resulted in the intensification of the agricultural industry. This intensification, brought about by Government support and technological advances, has been responsible for the loss of semi-natural habitats. In the post-war period, the Ministry of Agriculture actively encouraged the destruction of unimproved grasslands by introducing grants for the ploughing of grasslands to bring them into cultivation (Smith, 1969). Grant aid and mechanisation encouraged intensive forms of farming which destroyed extensive areas of semi-natural habitats and their associated species (Smith, 1969), particularly those that developed with traditional agricultural management. In addition, mixed farming has declined, with much of the lowlands now dominated by single-species arable cropping systems (NCC, 1990).

Habitats that developed with traditional agricultural management practices (i.e. almost the total land surface in the UK) were, and still are, particularly threatened by 'intensification'. Extensively managed meadows and pastures persisted on heavy clay soils with impeded drainage or on floodplain sites subject to periodic inundation, long after agricultural intensification had degraded other semi-natural communities in more workable situations (Manchester *et al.*, 1998). Eventually, improved drainage and mechanical cultivation made even wet sites amenable to intensification with the result that wet grassland rapidly declined within the farmed landscape. Large areas of grassland were converted to arable cropping regimes. Reseeding, and the increased use of inorganic fertilisers and herbicides, meant that grassland sites that were not converted to arable usage were nevertheless often intensified.

Lowland wet grasslands, and the species they support, are particularly vulnerable to changes in farming practices. For example, the addition of inorganic nutrients causes decreases in species-richness and can cause 'biological drying' (Rabotnov, 1977). Conversion from haymaking to silage production, a relatively recent innovation, results in a reduction in seed return to the sward because silage is cut earlier in the year than a traditional hay crop, before many grassland species have set seed. In addition, the immediate removal of the grass crop prevents those species that have set seed from returning propagules to the soil. Changes in grazing management can also affect the sward. Both under- and over-grazing are detrimental to sward structure and composition, as is the cessation of grazing altogether. Unsurprisingly, these grasslands are also particularly vulnerable to changes in the hydrological regime, e.g. lowered water tables or cessation of inundation.

1.4 Policy initiatives

Overproduction in the European Agricultural Community resulted in the introduction of policies to reduce agricultural output through devices such as set-aside. More importantly, following recognition of the adverse effects of intensive agriculture on the environment, funds have been made available to encourage more sympathetic land management through initiatives such as the Environmentally Sensitive Areas (ESA) (MAFF, 1989) and Countryside Stewardship (Countryside Commission, 1993) Schemes. In 1985, through Article 19 of EC Regulation 797, the UK Agricultural Departments introduced ESAs as part of a range of environmental land management schemes designed to protect and enhance the farmed landscape. Member States are required to adopt 'agricultural production methods compatible with the requirements of the protection of the environment and the maintenance of the countryside'. Through these schemes, farmers are eligible for positive incentives to manage land in an environmentally sensitive manner. Management agreements, designed to prevent further intensification and damage to landscape and wildlife, restrict agricultural inputs and outputs to levels that maximize environmental benefits. The use of environmental management payments and the reduced emphasis on price support enables integration of agricultural and environmental policy. The UK Biodiversity Action Plan Steering Group has identified agri-environment schemes as a key policy objective to achieving the targets set out in the Habitat Action Plans (Anon, 1998).

Although the ESAs were mainly designated due to the 'value' of existing wildlife and landscape, there are areas eligible for support within the ESAs that have a history of intensive use. These may be grasslands that have been drained, fertilised or reseeded, or land previously used for arable production. These areas are the focus of habitat restoration effort, since the scheme seeks not only to conserve existing 'valued countryside areas, features and resources', but also to enhance and restore them where possible.

The majority of existing habitats and communities of conservation value in Britain are semi-natural at best, developing as an integral part of the farmed landscape. The survival of such communities has depended, since their creation by agriculture, upon traditional agricultural management, and as farming practices have altered so the distribution and extent of such communities has altered also (Mountford, 1994). Contemporary agricultural policies, resulting from overproduction of agricultural commodities, now provide the opportunity to reinstate these communities within the landscape.

Lowland hay meadows of the National Vegetation Classification type *Alopecurus pratensis-Sanguisorba officinalis* grassland (MG4) have been included on Annex 1 of the EC Habitats and Species Directive (92/ 43 EEC) following designation as a habitat type of European Community interest (i.e. a habitat that (i) is in danger of disappearance in its natural range; or (ii) has a small natural range following its regression or by reason of its intrinsically restricted area; or (iii) presents an outstanding example of typical characteristics of one or more of the six following biogeographical regions: Alpine, Atlantic, Boreal, Continental, Macaronesian and Mediterranean). The other species-rich unimproved neutral lowland meadow communities, *Cynosurus cristatus-Centaurea nigra* grassland (MG5) and *Caltha palustris-Cynosurus cristatus* grassland (MG8), are not covered by legislation. However, Habitat Action Plans have been published for lowland meadows that outline objectives for the conservation and promotion of these three communities (Anon, 1998). Targets include arresting the depletion of unimproved lowland hay meadows throughout the UK and achieving favourable status within Sites of Special Scientific Interest (SSSI) and, in the longer term, securing favourable condition over as much of the total resource as is practicable.

The importance of the agri-environment schemes increased with EU regulation 2078/92, and recent reforms to the Common Agricultural Policy (CAP) allow up to 10% of all subsidies for agriculture to be paid for agri-environment schemes. There is now more concern that the schemes should be effective (Bunce *et al.*, 2001; Kleijn *et al.*, 2001).

1.5 Constraints on the restoration and re-creation of lowland wet grasslands

Diversification of improved grassland and re-creation of grassland on land released from arable use, are important means of achieving biodiversity policy objectives (Anon, 1998). However, previous studies suggest that many current prescriptions may be ineffective in achieving these aims (Wells *et al.*, 1994; Pywell *et al.*, 1997a; Kleijn *et al.*, 2001).

Areas of lowland wet grassland plant communities that have survived relatively intact tend to be small and isolated from one another. Such areas are often difficult to access with modern machinery or have been managed by landowners resistant to change (Carey *et al.*, 2001). The changes in soil properties and hydrological regime, the time under intensive management, together with ever-increasing distances from sources of propagules, make it unlikely that areas of intensified agricultural production will revert to 'semi-natural' species-rich wet grassland vegetation without intervention. Furthermore, intensive agricultural practices impose potential abiotic and biotic constraints on the restoration and re-creation of species-rich grassland (e.g. Mortimer *et al.*, 1998; Bakker & Berendse, 1999).

1.5.1 Biotic constraints

The main biotic constraint on the restoration of species-rich grassland vegetation is propagule supply, either *in situ* (the soil seed bank) or *ex situ* (the seed rain).

1.5.1.1 Soil seed banks

The soil seed bank consists of all viable seeds present in the soil, together with any seeds lying on the surface (Leck *et al.*, 1989). It comprises seed of species present in the above-ground vegetation, seeds of species of previous vegetation communities, and seed that has rained-in from further afield. Because the seed bank is, to some extent, an historical record of vegetation at a site, species within the seed bank are not necessarily represented in the vegetation, and presence in the vegetation does not guarantee presence in the underlying seed bank (Luken, 1990).

For many communities, there is little correspondence between the composition of the seed bank and the aboveground vegetation (Chippindale & Milton, 1934; Champness & Morris, 1948). Greatest similarity is generally observed between the seed bank and the vegetation in frequently disturbed habitats (Warr *et al.*, 1993), whilst, with increasing maturity of vegetation, this similarity declines. In addition, whilst the seed banks present in soils under climax communities do not correspond with the aboveground climax vegetation, seed of previous seral stages are to be found within such seed banks (Livingston & Allessio, 1968). This suggests that early successional species may be well represented in the seed bank, but that the rarer species indicative of unimproved grasslands may be under-represented.

Replacement of semi-natural vegetation with agricultural crops results in altered soil seed bank composition. Numbers of seeds of 'desirable' species decrease as seeds of earlier communities are buried more deeply by cultivation or germinate only to be removed before shedding seed. Concurrently, numbers of arable, annual weed seeds increase (Graham and Hutchings, 1988a, b; Leck *et al.*, 1989), resulting in an excess of undesirable weedy species in the seed bank (Hutchings & Booth, 1996; Pywell *et al.*, 1997b).

Species that persist well in disturbed habitats (e.g. arable land) generally have mechanisms for dispersal in time and space (Hodgson & Grime, 1990). Thus, species with long-lived seeds tend to be those associated with unpredictable habitats (Grime *et al.*, 1981; Roberts, 1986). Indeed, Chippindale & Milton (1934) found seed of species characteristic of arable land present in soils beneath pastures that had not been ploughed for 68 years.

The lack of similarity between seed banks and vegetation means that not all communities can be reestablished from seed banks. For example, both Graham & Hutchings (1988a,b) and Jefferson & Usher (1987) concluded that chalk grassland species were poorly represented in ex-arable seed banks, and that soil disturbance would only encourage undesirable species. Davies & Waite (1998) further demonstrated that few characteristic species of calcareous grassland form persistent seed banks, and that the seed bank is of limited value in restoring this grassland. Similarly, it appears that few species of flood meadows have long-term persistent seeds (McDonald, 1993) and thus the seed bank cannot be relied on as a source of the original vegetation (McDonald *et al.*, 1996).

However, where restoration sites are located close to extant vegetation there may be fresh inputs to the seed bank from the seed rain. Vegetation establishment is then not totally reliant upon *in situ* sources of propagules.

1.5.1.2 Local species pool

In addition to the impoverished nature of many seed banks, there is also often a lack of propagules of appropriate species locally (Treweek, 1991). Many potential restoration sites have been isolated from sources of suitable propagules by habitat fragmentation (Fisher *et al.*, 1996; Poschold *et al.*, 1998; Bakker & Berendse, 1999).

Prior to wide-scale agricultural intensification, the landscape of Britain was a mosaic of semi-natural habitat types interspersed with more extensively managed agricultural land. Traditional management practices, including rotation of land use, ensured the continuance of the full range of communities dependent upon historical agriculture. In addition, the area of land used for agricultural production fluctuated. Wells (1983) described how, historically, previously cultivated land was either left to recolonise naturally or was inoculated with hay-sweepings. Land taken into arable agriculture was thus able to revert to grassland after cessation of arable cultivation. On Porton Ranges, many of the species-rich grasslands existing today may in fact have been under arable cultivation within the past 200 years (Wells, 1983). Tansley (1939) cites the prevalence of ridge and furrow within grassland fields as evidence that much of the permanent neutral grassland in existence prior to the Second World War had in fact been arable at one time, before either being sown down or allowed to "tumble down" to grass.

This century however, the total area of land used for intensive agricultural production has continued to increase, to the detriment of semi-natural habitats. Land that is abandoned now does not have the same opportunity to revert as previously. Areas of semi-natural habitat have been reduced and fragmented to such as extent that propagules are often not available nearby to recolonise ex-arable land. Patches of semi-natural habitat are often remote from one another and much of the intervening land is hostile to the spread of many species.

Species composition and diversity of species-rich grassland are limited by recruitment (Tilman, 1997). The disruption of natural processes of dispersal by factors such as habitat fragmentation will result in a reduction in the numbers of propagules of 'desirable' species arriving at a site. The majority of seed in soil originates locally from the standing vegetation (Collins & Glenn, 1990) and the density of seed rain often declines exponentially with distance from the parent plant (Jefferson & Usher, 1989). It is the few seeds that randomly disperse greater distances that enable species to colonise similar, suitable habitats in other areas (Collins & Glenn, 1990).

If sites chosen for restoration are isolated from extant semi-natural vegetation, the communities that develop may never approach the target vegetation, as species with poor dispersal abilities are unlikely to arrive naturally. Fragmentation leaves isolated vegetation vulnerable to perturbations and, in the absence of immigration, recovery may not be possible.

1.5.2 Abiotic constraints

1.5.2.1 Soil nutrient status

Increases in soil nutrient availability associated with intensive agricultural management have implications for the rehabilitation of improved grassland, and also for the re-creation of species-rich grassland habitats on ex-arable land. The following section provides an outline of this drawing upon previous research.

The addition of nutrients to unimproved grassland generally results in increased biomass production, and reduced species diversity (Vermeer & Berendse, 1983; Berendse *et al.*, 1992; Marrs *et al.*, 1991). These changes may not be readily reversible, even after the cessation of agricultural practices (Berendse *et al.*, 1992; Mountford *et al.*, 1993, 1996a; van Duuren *et al.*, 1981). At low nutrient levels, a greater number of species are able to establish, and small-scale diversity increases. As levels increase, aboveground production increases also. Beyond a certain level of enrichment, species diversity decreases as one or a few species become dominant and intense competition for both light (above ground) and space (below ground) occurs (Vermeer & Berendse, 1983).

Berendse *et al.* (1992) investigated the premise that reducing both nutrient supply and annual biomass production could reverse the decline in grassland species richness. Although annual dry matter production was successfully reduced, species diversity did not increase significantly. They concluded that without a seed bank or other source of propagules, together with gaps in the vegetation for germination and establishment, successful restoration of species-rich meadows was unlikely.

Similarly, van Duuren *et al.* (1981) studied grasslands abandoned from agriculture. Although vegetative changes associated with the improvement of unfertilized grasslands were found to be irreversible, associated changes in management were thought to be partly responsible. For example, heavy grazing of productive grasslands suppresses seed production, which inevitably leads to impoverishment of the seed bank, altering the number and type of seedlings establishing.

An experimental investigation of the effects of differing levels of fertilization on species-rich wet grassland on peat moors in the Somerset Levels and Moors ESA (Mountford *et al.*, 1996a) found few species extinctions within the grassland, but significant changes in the abundance of species. Traditional management was continued during the experiment and, following cessation of fertilizer inputs, it was concluded that reversion to 'unfertilized' grassland would occur.

Fertilization alone then does not necessarily result in irreversible alterations to semi-natural swards. However, most fertilized grassland is also 'improved' in other ways, e.g. wetter land is often drained, whilst many grasslands are reseeded with more productive and palatable grass species. The increased productivity of improved grasslands allows increased stocking rates (suppressing seed production), increased numbers of hay cuts or a change to silage production, both of which will result in changes in floristic composition.

In particular, low levels of soil phosphorus appear to be required to sustain high levels of species co-existence in grasslands in the longer term (Janssens *et al.*, 1998). Similarly, Marrs *et al.* (1991) believe maintenance of 'semi-natural' grassland communities on arable soils is likely to be complicated by the high availability of phosphorus in arable soil. Extractable forms of phosphorus may be as much as four times greater in arable soil than in soils under semi-natural grasslands (Gough & Marrs, 1990). Fertilizer residue accounts for much of the phosphorus present, but without renewed inputs to maintain levels, much of this phosphorus becomes unavailable to plants. As grassland succession progresses on ex-arable land, residual extractable phosphorus does decline, although a reduction to levels similar to those found in semi-natural grassland soils may take up to 12 years (Marrs *et al.*, 1991). The rate of decline depends upon the amount of phosphorus remaining in the soil and the management applied, but levels of both nitrogen and phosphorus will fall in arable soils at a rate similar to the losses experienced by semi-natural grasslands once grassland has established.

Reduction of soil nutrients may not always be necessary, however. All species-rich grasslands are not inherently nutrient-poor systems, e.g. seasonally inundated wet grasslands traditionally receive nutrients with floodwaters. McDonald *et al.* (1996) reported that the *Alopecurus pratensis-Sanguisorba officinalis* association (MG4) is a low-fertility system. It may be so in terms of artificial inputs, but seasonal inundation of this community provides nutrients in the form of salts, alluvial silts and decaying organic matter (Rodwell, 1992b). Determining appropriate management practices to induce desired directional changes in grasslands, and ensuring adequate water supply, may be more important than soil fertility in these cases. Furthermore, many of the fields entered into agri-environment schemes tend to be marginal for agriculture and may thus not have been intensively improved in any case.

1.5.2.2 Hydrology

Wetlands, and wet grasslands, generally occur in the lower regions of landscapes, where inundation and waterlogging occur (Treweek & Sheail, 1991). Water depth, frequency, seasonality and duration of flooding, and water chemistry are the most important factors determining survival of wetland species (Hammer, 1992), with species demonstrating differing tolerances to depth and duration of flooding (e.g. Gowing *et al.*, 1998).

Land drainage is one of the major factors responsible for the decreasing area of lowland wet grassland. Alterations to the hydrological regime will ultimately result in a change in sward composition. However, too much water as well as too little will impact wet grassland composition. For example, Swetnam *et al.* (1998) found that continued existence of *Caltha palustris-Cynosurus cristatus* (MG8) grassland was threatened by an increase in the incidence of spring flooding. However, more wet grassland is under threat from a decrease, rather than an increase, in water supply. Species adapted to withstand inundation or high water tables may well be able to withstand dry periods, but are often outcompeted by species that, whilst intolerant of flooding, are better competitors under drier conditions.

When attempting to restore wet grassland, it may be necessary to reinstate the water regime that sustained the communities prior to the drainage. In many areas, such water management is possible. For example, in the Somerset Levels and Moors discrete blocks of land can be isolated, and the water regime manipulated independently of surrounding land. However, many areas that support wet grassland, including the parts of the Upper Thames Tributaries, do so through natural flooding events. Parcels of land are not discrete, and manipulation of water regimes may not be possible without impacting adjacent land. Additionally, water extraction and river 'improvements' have reduced and diverted water flows, and there may be insufficient water available to maintain wet grassland.

Where water manipulation can be achieved, infield drainage can be used to bring water from a watercourse to the field. Field drainage works by using pipes/ mole drains to collect water in the field and drain it back into surrounding ditches or rivers. The water level in the ditch or river is generally held below the level of the field drainage to allow gravity to move the water from the middle of the field. If the ditch water level is maintained above the level of the field drainage system, then water will drain back onto the land. A recent innovation is to use windmills (or other means) to pump water into fields; a curious reversal of the practice of centuries when windmills were used to drain agricultural land.

1.5.2.3 Agricultural land management

The management of grassland greatly influences botanical composition and often overrides other factors. Most grassland depends upon cutting, grazing, or both, for its persistence, and thus some form of management will be necessary to arrest succession and maintain grassland. Practices such as cutting and grazing may induce directional changes in grasslands through altered abundances of species, whilst fertilization may result in a sward dominated by one or a few grass species, with the extinction of other species (Duffey *et al.*, 1974). The particular management regime adopted will control the composition of the vegetation, and thus management suitable for the maintenance of the particular community must be reinstated.

Grassland of conservation interest largely developed under 'traditional' management practices. Lowland wet grassland generally receives a summer hay cut and relatively light aftermath grazing, with the meadows shut up during winter when the ground is too wet for grazing animals. The reinstatement of such management on improved grassland may result in reversion to a more 'semi-natural' sward. However, reinstatement of traditional management on ex-arable land would be unlikely to restore semi-natural vegetation in the short-term, if ever.

The length of time that a site has been managed intensively will also affect the potential for successful restoration, both for grassland and ex-arable sites. With increasing time under intensive management, site physical factors become increasingly dissimilar to those conditions necessary to support the species of the original community. Years of ploughing for arable cultivations, for example, will completely alter the soil characteristics. Thus with time the constraints on restoration become increasingly insurmountable. Chances of restoration success are generally lowest on sites with a long history of intensive management, especially those that are remote from extant habitat (sources of renewed species diversity).

1.6 Techniques for the restoration of species diversity

A new ecological discipline – restoration ecology – emerged as a direct result of conservationists' attempts to redress the balance in the countryside between intensive food production and the maintenance of wildlife habitats. However, restoration of habitats has been practiced in a piecemeal way, as opportunities arise. If habitat restoration is to achieve anything useful, then an objective, national strategy needs to be put in place to ensure the achievement of: i) a representative range of viable and sustainable wetland habitat types, taking account of regional variation, and ii) full representation of wetland species throughout their range, maintaining viable populations (Treweek and Sheail, 1991). The first of these objectives can be fulfilled by the conservation of existing habitat, the rehabilitation of degraded habitat and the creation of new habitat. The second requires that the decline of wetland species is arrested, and that rare and declining species are actively promoted.

The term 'habitat restoration' thus encompasses restoration, rehabilitation and recreation. Early attempts created visually attractive habitats, with little resemblance to any 'semi-natural' community, and were thus often of limited conservation interest. If habitat restoration is to be used to enable wide-scale recovery of declining communities, a greater understanding of communities is necessary and restoration attempts must be objective and rigorous (Box, 1996; Palmer *et al.*, 1997). Early attempts at restoration were often unsuccessful because the community was poorly understood, objectives were not clearly defined and species used were inappropriate for the environmental conditions. Another commonly encountered failing was the inability to evaluate the 'restored' habitat because of objectives that could not be quantified. Objectives for restoration vary, and may include criteria that are economic, educational, recreational or ecological. Ecological objectives considered to be important in restoration include the restoration of species and habitat diversity, species composition and ecological processes.

In practice, a successfully restored community should:

i) have a net productivity similar to that of the target community;

ii) be effective in nutrient retention, with fluxes similar to those of the target community;

iii) be functionally entire;

iv) be capable of perpetuating itself, or be dependent only upon traditional management practices under which the target community developed; and

v) be resistant to invasion by exotic species (Ewel, 1987).

On the majority of sites 'released' from arable cultivation, intensive management has altered both site-physical factors (through cultivation, drainage and inorganic additions such as fertilizer – see section 1.5.2) and biotic characteristics (through the increase in competitive weed species and corresponding decrease in desirable propagules – see section 1.5.1). These changes in soil structure, fertility, soil seed bank composition, and availability of propagules, brought about by intensive arable usage over prolonged periods of time, have rendered many ex-arable sites very different from the original, with the result that restoration of lost communities will not be a simple process.

For habitat restoration to be widely implemented, techniques must be straightforward, cost effective and relatively low cost as the agri-environment schemes are cash limited. The precise techniques adopted should depend on the physical characteristics of a site, e.g. its location relative to potential sources of colonizing species, and on the time and financial resources available. These techniques may be assessed according to their cost and technical feasibility, their relative ecological effectiveness and reliability, and the time taken, to achieve the desired ecological aims (Manchester *et al.*, 1999).

1.6.1 Natural regeneration (natural dispersal and colonisation)

To assess the potential for natural regeneration of plant communities, processes controlling the availability of propagules locally need to be considered. The set of species potentially capable of coexisting in a particular community has been termed the 'species pool' (Eriksson, 1993) and it is this that will determine the course of vegetation development at the local and community level. The community species pool is the set of species present in the target community (above- and below- ground), whilst the local species pool consists of species in the landscape type around the target community that are capable of co-existing in that community (Partel *et al.*, 1996). In order to fully restore any vegetation community, an assessment of the composition of the appropriate species pools, together with an evaluation of dispersal dynamics locally, are essential (Zobel *et al.*, 1998).

If appropriate propagules are available at or adjacent to the site, natural regeneration may be the cheapest method for restoring vegetation as it does not involve the acquisition of seeds and, given an appropriate soil seed bank, may only require light soil disturbance to encourage germination of species from the soil. It will also ensure that seed is of local provenance and of the correct ecotype for the region. However, on sites used for arable cropping for a number of years, seed banks are likely to be degraded, and thus seed dispersal from nearby areas of semi-natural vegetation will be vital to the natural establishment of desirable vegetation.

Information on the seed rain from individual plant species has been monitored, but the seed rain of all species in a community is difficult to assess with equal reliability (Rabinowitz & Rapp, 1980). In order to capture seed of all species present within a community, the variation in individual seeds in terms of shape, size, weight and height of release need to be taken into account (Jefferson & Usher, 1989). In addition, local wind conditions (Carey 1998) and movement by animals through vegetation (Carey and Watkinson, 1993) affect seed rain distribution.

Peart (1989) investigated the abundances of species in the standing vegetation, seed rain and resultant seedlings in grassland, and found that for some species of grass, the abundance in the seed rain was similar to the abundance in the local vegetative community. In addition, the local seed rain determined which seedlings appeared, with the dominant species appearing as seedlings.

The short-range dispersal of most seed has implications for habitat restoration, and in particular any scheme relying upon natural regeneration. It is believed that the chances of successfully restoring semi-natural communities will be improved by the proximity of a rich source of potential colonisers, i.e. adjacent species-rich meadows (Baines, 1989). However, in view of the limited dispersal capabilities of most species, whether in fact propagules do travel even as far as the next field in significant quantities is debatable and untested.

The term 'seed rain' is not limited solely to airborne propagules, although these may be the most abundant sources of seed. One process that could be important for habitats linked by water is that of hydrochory (dispersal by water). The literature pertaining to hydrochory is limited, but this method of dispersal is clearly relevant to wetland habitats and their related species. Successful dispersal by water will depend upon the seed or vegetative plant part remaining afloat, thus the potential duration of buoyancy and viability of seeds after submersion will be critical (Schneider & Sharitz, 1988).

Where restoration sites are in proximity to extant semi-natural vegetation, there is potential for seed input by desirable species. The probability of plants colonising new areas depends upon the frequency of reproduction, the reproductive effort, and the dispersability of the seeds (Macdonald & Smith, 1990). The majority of seed in soil originates locally from the standing vegetation (Collins & Glenn, 1990). The density of the seed rain falls off exponentially with distance from the parent plant (Jefferson & Usher, 1989), but the few seeds that disperse greater distances can enable species to colonise similar, suitable habitats in other areas (Collins & Glenn, 1990).

An important consequence of the dispersal abilities of plants, or rather lack of, and the fragmentation of remaining species-rich grassland, is that natural colonization will be a slow and uncertain process.

1.6.2 Deliberate reintroduction of plant material

As indicated above, for the majority of arable sites, natural regeneration will not be a viable option. Some degree of intervention will be necessary to encourage the vegetation to develop in the desired direction. In situations where naturally occurring sources of propagules are limiting, artificial reintroduction of appropriate species may be the only way to ensure their arrival and to accelerate the re-assembly of species-rich grassland communities on such sites (Wells *et al.*, 1981; Wells *et al.*, 1986).

Techniques for the reintroduction of species were considered in Manchester *et al.* (1999). Although most are not applicable to wide-scale habitat restoration and are not investigated in this study, all are outlined below. In most cases, propagules may be acquired from commercial sources or from extant habitat as seed or vegetative parts.

1.6.2.1 Seed mixtures

The most common technique for the restoration of habitats on degraded land is that of reseeding with suitable species to accelerate the establishment of 'desirable' vegetation (Countryside Commission, 1993). The use of seed is cheaper than introduction of species as transplants (Byrne, 1990), and has the advantage that it can be carried out using standard agricultural techniques. However, species may have specific requirements for germination that are not met in the field, species may be unavailable commercially or very expensive, and may be of the wrong ecotype or even from non-native sources.

There are general concerns about the suitability of plant material for reintroduction. It is often stated that seed of local provenance should be preferentially used to ensure that species introduced are pre-adapted to local conditions and also to avoid polluting local gene pools (e.g. Millar & Libby, 1989; Knapp & Rice, 1996; Akeroyd, 1994). One way of ensuring plant material is of local provenance is to use only seeds or plant parts acquired locally to the restoration site from extant vegetation. However, fragmented and isolated local populations may already have lost the genetic variation required for the establishment of new populations (Lesica & Allendorf, 1999). In such cases, the introduction of novel genetic variation to the local population may actually increase the adaptiveness of the local population. Moreover, where candidate sites for restoration have been heavily modified, locally adapted genotypes may not establish and novel genotypes may be necessary.

In the absence of locally produced seed, seed may need to be acquired from a commercial seed house. However, there are problems associated with the use of commercially produced seed, both for the producer and the user. Seed production in Britain is constrained by a variable and unpredictable climate, seed producers cannot select for improved seed production of wild flowers since the wild characteristics of these species must be retained, wild flowers produce seed over a period of time and often have mechanisms for the dispersal of seed once ripe (Brown, 1989). As a result, seed of certain species may be unavailable commercially, whilst seed of others may be prohibitively expensive.

1.6.2.2 Hay bales as a source of propagules

This technique uses seed harvested from nearby species-rich meadows and thus has the advantage that local provenance is ensured. Historically, this appears to have led to successful reversion to grassland, e.g. in the past, hay sweepings were sowed to revert arable land to grassland (Wells, 1983). Seed present in hay baled from a 'good' meadow may introduce species that would not be available commercially, and should ensure that all species introduced are 'desirable'. The use of hay, if available, is a relatively low cost option and is therefore important to those administering agri-environment schemes.

However, there are also disadvantages associated with this technique. If the hay is not threshed, but spread on the ground, the hay 'mat' can act in the same way as litter, which if thick or coarse may be a physical barrier preventing seed penetration to the soil (Chambers and Macmahon, 1994). As a result, germinating seedlings may not be able to emerge through the litter or their roots may not be able to reach the soil. The timing of the hay cut will determine the composition of the hay because not all species produce ripe seed at the same time; some will already have shed seed, while some may not have flowered. Some species will not be present in the hay at all. Following cutting, the seed of species in the sward enter a new phase as some fall to the ground while others continue to ripen on the cut vegetation. If the weather is wet seeds will soon begin to rot. The precise seed content of hay bales is thus difficult to predict, depending upon which species have ripe seed at the time of the hay cut, the length of time the hay lies on the ground, the prevailing weather conditions and the timing of the baling of the hay. There has been relatively little research on their use as a seed source, but Smith *et al.* (1996) suggest that the majority of seed in hay bales will be overwhelmingly of grass species.

1.6.2.3 Seed harvesting techniques

An alternative to using the 'normal' hay crop is to harvest a meadow for its seed. This can be achieved in a number of ways, ranging from hand-picking of seed to large-scale mechanical harvesting (Robinson, 2001).

It has been suggested that green-hay harvesting may ensure a wider range of species is introduced to the recipient site than through the use of hay baled in the traditional way (Robinson, 2001). Cut grass is baled and spread on the recipient site within a few hours of mowing. This will minimize the loss of seed from the bales as ripening takes place on the recipient site.

An increasingly popular technique is the use of a brush harvester to collect seed from existing meadows. A rotating brush is used to brush seed from the standing vegetation, which is then collected. The hay crop is not damaged and can subsequently be harvested. In theory the same meadow can be brush harvested several times during the summer to collect seed of species with different seed production dates.

Impacts on the botanical composition of swards harvested for seed have been identified (Porter, 1994), and will depend on the quantity of seed removed, the particular species involved, the timing of seed set relative to harvest date and whether there are species present that regenerate from seed on an annual basis. In order to maximize seed yields, meadows used for seed harvesting are left uncut until long after the hay would have traditionally been taken off, with the result that late-flowering species that would not normally shed seed may be given the opportunity to do so. If the hay cut is consistently set back, then the composition of the donor vegetation may eventually shift as a result of these altered seed inputs and the impact of an alteration to the timing and duration of grazing. Large-scale harvesting is also likely to be detrimental to invertebrates, and particularly those that use hay meadows for breeding. It is recommended that, to minimize the impacts of seed harvesting, fields used should not be harvested each year or that, on rotation, only a proportion of the site should be harvested in any one year.

1.7 Targeting restoration

At the local level, restoration effort should be targeted at sites that are most likely to provide the desired environmental benefits. In situations where money is a limiting factor, sites chosen should be those requiring the least intervention. Sites where target vegetation has only recently been eliminated, or those next to extant grassland, are generally thought to be most appropriate for restoration.

At the national level, an objective strategy is necessary to ensure that the country's full range of characteristic species and communities is maintained throughout their range in a viable state so that the requirements of the BAP are met (Anon, 1998). However, particular emphasis has been placed on certain habitats/ species assemblages (see section 1.4).

When restoring habitats, the natural range of the community must be taken into account, together with the particular environmental conditions known to sustain the community. When the species assemblage to be restored has been defined, the question of which species to reintroduce arises. A major difficulty for those attempting to restore grassland communities is that semi-natural communities developed over centuries, and the precise trajectories they followed to reach their current condition are unknown.

Egler (1954) first emphasized the consequence of the initial floristic composition for the subsequent composition and diversity of vegetation. Stockey and Hunt (1994) also found establishment within the first year of wetland mesocosms was likely to be a precondition for successful establishment in the long-term because of the difficulties associated with establishment within a closed turf. Weiher & Keddy (1995) addressed the question of what communities will assemble from a common species pool when varying environmental treatments, including fertility, water depth, soil texture and leaf litter, were applied. The resultant experimental communities differed, showing strong and consistent effects of fertility, water level and leaf litter on community composition. Thus, even with exactly the same starting species, differing environmental conditions were responsible for the establishment of species adapted to those particular conditions, and different communities resulted. Temporally or spatially varying disturbances may lead to recolonisation by different propagule sources. Where external sources of propagules are scarce (i.e. isolated sites), succession may proceed differently at different sites (Kotanen, 1996) and resultant communities may never converge. Where sources of propagules external to the site are abundant (i.e. the seed rain issuing from surrounding semi-natural habitat), succession should converge.

No two geographically distinct regions will have the same species pool or environmental conditions so in order to define targets for restoration, and to evaluate restoration success, regional differences need to be taken into account. One possibility to determine target communities is to use systems of vegetation classifications (see also section 1.8.2). However, vegetation classifications are based upon samples of vegetation, and are therefore limited by the number, type and locality of stands sampled. Regional differences between communities may be blurred or obscured in any national classification depending upon the accuracy of sampling.

1.8 National resources

1.8.1 Species distribution datasets

The Biological Records Centre (BRC) was created in 1964 to map the distribution of the flora and fauna of Britain. Now the Biological Databases Unit, it is Britain's national biodiversity centre, containing approximately 6 million records pertaining to the occurrence and distribution of more than 16000 species of plants and animals in the British Isles. The databases are geo-referenced using the Ordnance Survey OS) grid. Such spatially and temporally referenced data enables past and present species' ranges to be determined. Moreover, the temporal data allows the status (i.e. stable, declining, or increasing in frequency) of a species to be determined, and can thus aid in the establishment of priorities for conservation and restoration.

Whilst post-1980 data generally have a resolution of 100m, the data is more usually mapped using the 10km squares of the OS grid. This is a convenient scale for presentation and analysis, with 2860 sampling units in Britain. From the foundation of the BRC, data have been used to prepare maps summarizing the national distribution of species, for publication in atlases.

1.8.2 Classification of Plant communities

When restoring vegetation, there is a need to know not only where individual species occur, but also which species are commonly found growing together in recognisable communities, and then where and under what conditions these communities occur. Once distinct vegetation communities have been identified, national species distribution information can be used to map the co-occurrence of constituent species, thus providing a spatial representation of the potential distribution of the community type.

Tansley's (1939) account of the British flora recognised the tremendous variability within grassland swards, and only identified broad categories of acid grasslands, basic grasslands and neutral grasslands as being distinct from one another. The neutral grassland category was acknowledged to be rather vague, attributable to a lack of ecological investigation. Neutral grasslands have developed on the lowland clays and loams of the English midlands, southern England and valley alluvium in the north and west. Whilst they are much richer in nitrogen and minerals than acid or basic grasslands, some alluvial soils that support neutral grasslands are as alkaline as chalk or limestone soils. Indeed, Tansley stresses that the term "neutral" actually applies to the species that are characteristic of this habitat (neither markedly calcifuge or calcicole) rather than to the soil solution. The neutral grasslands were classified into meadows (regularly mown for hay) and pastures (grazed), because these differing treatments result in different swards. Meadows are characteristic of alluvial soils with a high water table and/or periodically flooded and are nutrient-rich habitats, the high initial fertility of the alluvial soil augmented by mineral salts deposited during flood events. A number of species were identified as the "kernel" of alluvial neutral grassland: *Anthoxanthum odoratum*, *Cynosurus cristatus*, *Deschampsia cespitosa*, *Festuca pratensis*, *Holcus lanatus*, *Lolium perenne*, *Poa pratensis*, *Poa trivialis*, *Cardamine pratensis*, *Cerastium vulgatum*, *Leontodon autumnalis*, *Lotus corniculatus*, *Plantago lanceolata*, *Ranunculus acris*, *Trifolium pratense* and *Trifolium repens*.

Ratcliffe (1977) classified neutral grasslands into 14 groups according to a combination of environmental and floristic characteristics. Within each group, the constituent species were distinguished into two separate groups: the constant and distinctive species, and those species, forming recognisable associations within the group, which could be separated further by more detailed classification. The wet grasslands of interest in this study occur on alluvial soils of wide river valleys, more common in the south. These are the water meadows and alluvial meadows. The water meadows were largely man-made to improve poorly-drained, non-productive alluvial meadows. Constant species of these grasslands include *Cardamine pratensis, Festuca arundinacea, F.pratensis, x Festulolium loliaceum, Lolium perenne* and *Senecio aquaticus*. Within the alluvial meadows, constant species are *Alopecurus pratensis, Briza media, Filipendula ulmaria, Ophioglossum vulgatum, Sanguisorba officinalis, Silaum silaus* and *Thalictrum flavum*. Co-dominants of this group are *Agrostis stolonifera, Anthoxanthum odoratum, Cynosurus cristatus, Festuca pratensis, Festuca rubra, Holcus lanatus, Lolium perenne* and *Poa trivialis* with *Centaurea nigra, Plantago lanceolata* and *Trifolium pratense.*

These early classifications of British vegetation, as developed by Tansley (1939) and Ratcliffe (1977), are not systematic and have since been largely superseded by the National Vegetation Classification (NVC; Rodwell, 1991 *et seq*).

The National Vegetation Classification (NVC) came about as a direct response to the large quantity of uncoordinated phytosociological accounts of British vegetation, and the perceived need for a national, systematic classification of vegetation (Rodwell, 1992). The aim was to produce a comprehensive, standardised vegetation classification of all natural, semi-natural and major artificial plant communities within Britain. Published descriptions began in 1991 with 'Woodlands and scrub' (Rodwell, 1991), and the volume relevant to this study ('Grasslands and montane communities') was published in 1992.

Samples for the grassland classification were located on the basis of floristic and structural homogeneity of vegetation, in order to produce a scheme that included both species-rich and more impoverished swards (Rodwell, 1992b). Much sampling was carried out in the highly improved agricultural landscape with only short-term leys being excluded. All vascular plants, bryophytes and macrolichens were recorded. The floristic data were supplemented with details of the vegetation structure, the context of the stand in the landscape, basic environmental data and information on management. Over 2000 samples were available for the analysis of grassland composition. Floristic records were used to characterise the vegetation types, whilst environmental data were used to interpret the results of the analysis.

The mesotrophic grasslands, as identified by Rodwell (1992b) can be split into:

- *Arrhenatherum elatius* grasslands (MG1) - A rank, species-poor grassland typical of road verges with ungrazed, coarse and tussocky swards;
- Well-drained permanent pastures and meadows (MG3, MG4, MG5, MG6) - closed swards of grasses and herbaceous dicotyledons. Most of the permanent agricultural grasslands fall within this category. The first three communities are generally unimproved grasslands, often managed traditionally as meadows, whilst MG6 represents an at least moderately improved permanent pasture.
- Long-term leys and related grasslands (MG7) - species-poor, grass-dominated swards, often sown.
- Ill-drained permanent pastures (MG8, MG9, MG10) - on more frequently waterlogged and less fertile soil profiles.
- Inundation grasslands (MG11, MG12, MG13) - characteristic of fine-textured mesotrophic soils alongside fluctuating sluggish or standing waters.

The NVC provides a means of defining vegetation assemblages, and can be used in conjunction with species distribution data at the national scale to determine geographic ranges of vegetation communities.

1.8.3 Mapping of community co-occurrence

The National Vegetation Classification (NVC; Rodwell, 1991 *et seq*) may be used at the national scale to indicate where the different community types occur. The potential distribution of a community can be determined by mapping the co-occurrence of constituent species derived from the published association tables. Such co-occurrence mapping results in a species-richness map for a community type, displaying the potential geographic range of the community and highlighting those areas with the greatest number of constituent species present. Mapping of recent records only would indicate which species (communities) might now occur, but we are interested not only in current but also historical distribution and thus no cut-off date should be used. The geographic ranges of many species, and hence communities, within Britain have contracted and fragmented and this can leave isolated populations and communities in ecologically marginal habitats (Lawton, 1993). Thus the current location of species and communities may not be optimal for survival and therefore it is important to ensure that communities and species are reinstated into areas with optimal conditions for survival, and not only into increasingly marginal areas. By including historic records, it should be possible to identify not only the marginal occurrences of species and communities, but also the former centres of their distributions also.

1.8.4 Limitations of vegetation classifications

1.8.4.1 A snapshot in time

Many accounts of vegetation only present a static view. Typically, vegetation is described using field methods, followed by classification and ordination to define plant communities and/or community gradients and associated environmental gradients. Classification and ordination are both techniques for floristic data reduction. These methods have been used to describe and recognise patterns in vegetation distribution, define plant communities and to examine plant and community distribution in relation to environmental factors and gradients (Kent and Ballard, 1988). Ordination is effective for showing relationships, placing similar entities in proximity, and producing an economical understanding of the data in terms of a few gradients in community composition (Gauch, 1982).

Plant communities are extremely variable (spatially and temporally) and dynamic in nature and therefore any static classification of vegetation is necessarily a 'snapshot in time'. Species freely and variously combine with one another in communities that intergrade with one another, so that communities can be seen as a continuum, with the change from one type to another occurring when the species composition and abundance is recognisably different between stands. Vegetation classifications that sample pristine communities (like the NVC) give a fixed view of composition, whilst in the real world all combinations of species composition and abundance that can occur do occur.

1.8.4.2 Small sample size

Whilst vegetation classifications may be flawed, they do provide an essential framework for the strategic planning of restoration and conservation at the national level. In heavily modified landscapes they may be the only way of inferring previous vegetation.

It should be recognised that classifications are only as good as the samples used to define the community types. Complete coverage of all vegetation assemblages is not feasible because of the constraints of time, and thus sampling effort will always be directed towards particular stands of vegetation (usually the more pristine, 'better' examples). For example, certain of the community descriptions contained within the NVC are based on low numbers of samples. Description of MG4 (*Alopecurus pratensis-Sanguisorba officinalis* grassland) was based on 22 samples, MG8 (*Cynosurus cristatus-Caltha palustris*) on 15 samples, whilst the *Lathyrus pratensis* sub-community of MG5 (*Cynosurus cristatus-Centaurea nigra*) on 137 samples.

1.8.4.3 Regional variation

Regional differences between stands of the same community type need to be taken into account when using vegetation classifications. When comparing existing stands of vegetation to published accounts of vegetation assemblages, a community is not necessarily of lower 'value' simply because it is missing certain species that may not occur locally. Thus, whilst the NVC will be utilized within this study, 'real' vegetation communities as observed within the study area should be used, where possible, as a 'yardstick' to measure success and identify targets.

Community co-occurrence maps are merely a description of which species have been recorded within the same 10km squares and they do not imply that the species are actually found growing together within fields in that square. Whilst it may be assumed that chances of successful restoration are higher in areas where all constituent floristic elements are present, it should be remembered that many species typical of wet grasslands are widespread within other plant communities. Squares seemingly suitable for the reinstatement of wet grasslands may never in fact have supported the habitat and may have no sites with suitable physical conditions.

National distribution data may be used simply, to determine which particular plant species and communities may be expected to occur within particular regions. The geographic range of a species is limited by both abiotic and biotic factors, but actually provides little information on the specific requirements of a species for survival. A more useful alternative to the geographic range (the extent of a species occurrence) is the area of occupancy of a species (Gaston 1991). A species will not occur uniformly across its geographic range: some areas will be unsuitable for survival and others, while suitable, may not be currently colonised. The area of occupancy of a species does not include such regions, and is thus a subset of the extent of occurrence. In general, it will not be possible to determine a species' area of occupancy remotely as data held at the national scale is of too coarse resolution. Data of finer resolution at the regional scale are necessary to determine inherent regional or local variation, and to identify which areas within the ranges of species and communities are presently occupied.

1.8.5 Reference habitats

Historical records and national vegetation classifications may be used to infer the likely vegetation composition at a location, but it may be more appropriate, where available, to use a local example of the target community to be restored as a 'template' for the restoration. Aronson *et al.* (1993) term this type of model for restoration an 'ecosystem of reference', which they define as 'some standard of comparison and evaluation, even if the choice made is somewhat arbitrary'. It is essential to have clearly defined objectives and some way of measuring the success of the project. Aronson *et al.* (1995) give anecdotal evidence for a large number of restoration projects where no evaluation of results was established, no baseline data were collected, and no reference system had been defined. Whilst there is no 'ideal' habitat to choose as a reference, some standard is needed in order to define objectives and evaluate success.

1.8.6 Decline statistics

Recent and historical data held by the BRC can be used to quantify changes in distribution and/or frequency of species. Firbank *et al.* (1994) generated such statistics to target conservation policy for set-aside land. More recently, Rich and Woodruff (1996) analysed changes across the British flora by comparing records collected for the Atlas of the British Flora (1952-1960) with results of the Botanical Society of the British Isles (BSBI) Monitoring Scheme (1987-1988). They found significant decreases in a number of species typical of wet grassland.

Mountford *et al.* (1997) derived statistics of change in the frequencies of species and communities at a regional scale. As with the national work of Rich and Woodruff (1996), changes in the frequency of constituent species were quantified by comparing historical data from the field survey for the *Atlas of the British Flora* (1952-1960) with Botanical Society of the British Isles (BSBI) Monitoring Scheme data (1987-1988). Of 62 plant species of wet grassland showing a marked national decline in frequency, 12 are important constants of the NVC wet grassland communities of conservation concern, including *Caltha palustris* (MG8), *Carex nigra*, *Carex panicea*, *Cirsium dissectum*, *Sanguisorba officinalis* (MG4), and *Succisa pratensis*. These species have not necessarily suffered declines in all regions of the country. Even regional information may not be sufficiently detailed to determine local trends in species declines. National statistics contain no information about the abundance or frequency of a species within a 10km square and may disguise smaller-scale changes. To take an extreme example, a previously ubiquitous and abundant species may have suffered huge population declines, but nationally derived statistics will not reflect this until the species is lost from its last site in a 10km square.

Whilst national decline statistics may indicate worrying trends for individual species, it would not be appropriate to select suites of species to restore to a particular site from lists of declining species. Many of the plant species typical of wet grasslands are not specialised wetland species, and occur widely within other grassland types. Many are not individually rare, rather it is the particular assemblages of species that come together within wet grasslands that are of interest and which are declining. A lack of evidence of individual species declines does not necessarily indicate that particular communities are not declining, especially when those species are not restricted to those communities (Fuller, 1987).

1.8.7 Species Indicator values

Ellenberg (1974, 1979) assigned 'indicator values' for moisture, light, temperature, nitrogen and acidity to approximately 2000 vascular plant species of western Central Europe. These indicator values can be used to calculate a 'mean indicator value' for the community under study (Ellenberg, 1988). This value is then an estimate of the value of any of these environmental factors at a site, derived by averaging the indicator values of all species present (Ter Braak and Gremmen, 1987). It is therefore possible to use the indicator values attached to different plant species to select species with particular preferences for particular conditions. For example, the reaction of plant species and communities to differing water regimes has been studied in the Netherlands (Mountford & Chapman, 1993). Whilst soil moisture is obviously important in the establishment and persistence of wetland species, it may not be the most important determinant. Ter Braak and Gremmen (1987) studied vegetation of woodland, grassland, marshes, ditches, heathland and bog. Their analysis showed nitrogen to be the environmental variable most responsible for floristic variation, with moisture the second most important variable. Indicator values based on those of Ellenberg for central Europe have been amended for the British flora (Hill *et al.*, 2000), making their use in the British context more valuable.

1.9 Restoration Potential

The potential of a site to be successfully restored to species-rich wet grassland depends on many interacting factors. Whilst the majority of 'good' wet grassland sites still in existence are likely to have remained under grass, there are some arable areas which, by virtue of their location and management history may have relatively high restoration potential. It is important to determine which sites offer the greatest potential for ecological restoration. The suitability of sites for restoration will be influenced by many factors including:

- current and historical species composition;
- type and intensity of management (past and present);
- scope for manipulation of water levels;
- proximity to existing 'good quality' wildlife habitat (sources of colonizing species);
- availability of livestock and management expertise for reinstatement of 'traditional' management; and
- technical feasibility factors, i.e. availability of water (Treweek *et al.*, 1998).

CHAPTER 2 INTRODUCTION TO THE STUDY AREA

2.1 The study site

The focal site for this study of grassland restoration is an ex-arable field (SA123) on the Oxfordshire-Buckinghamshire border (Grid Reference SP 65100 20290). The site occupies 4.19 ha of the floodplain of the River Ray (figure 2.1). The River Ray catchment (the study area) occupies approximately 35km^2 of low-lying land on the borders of the counties of Oxfordshire and Buckinghamshire in the Upper Thames Tributaries Environmentally Sensitive Area (ESA) (figure 2.2).

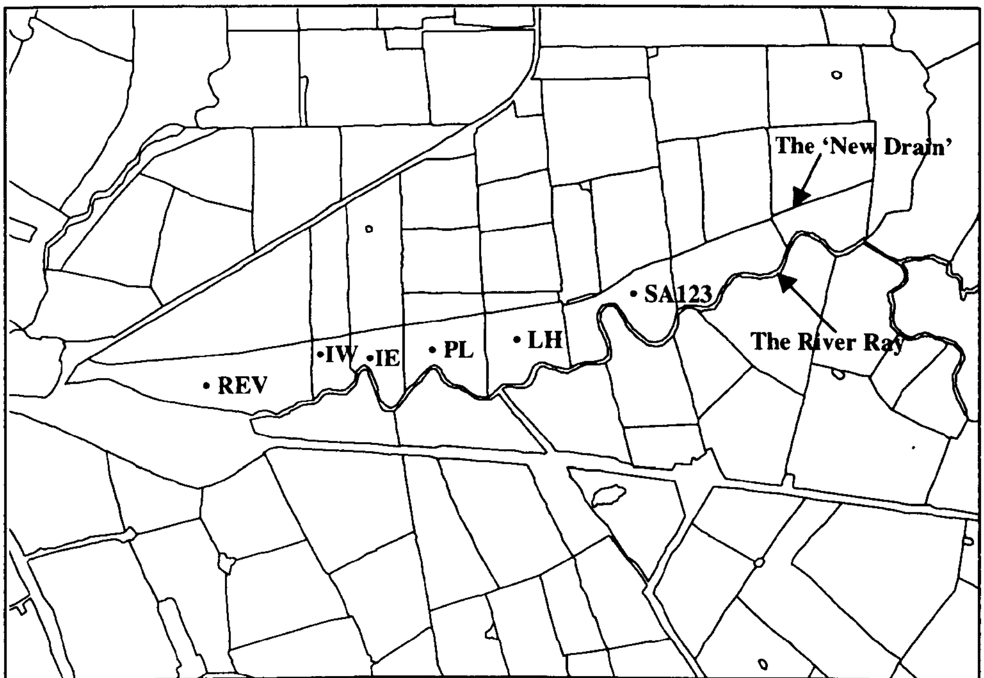
The study site (SA123) is bordered to the east by arable land. To the west lie a number of grassland fields, all now managed extensively (figure 2.1). Long Herdon Site of Special Scientific Interest (LH) is an unimproved species-rich wet meadow of 3.80 ha immediately adjacent to SA123. PL (3.50 ha) is similarly unimproved and species-rich, owned jointly by Plantlife and Timotei. Both IW (1.43 ha) and IE (2.07 ha) have been previously improved by underdrainage, whilst IE has also received fertiliser application. REV (7.22 ha) has previously been drained and reseeded, but is now being managed to allow reversion. The study site itself had been in arable usage for approximately 15 years until 1992 when it was entered into the Countryside Stewardship scheme, and became available in 1993 for reversion to wet grassland within the current investigation.

These fields are all bordered to the south by the River Ray and to the north by the 'New Drain', an artificial drainage ditch which flows into the River Ray upstream of the study fields (figure 2.1).

Data local to the restoration site were collected to complement national- and regional-level information and were used to refine national and regional information in the setting of site-specific targets for the restoration in terms of community and species co-occurrence and abundance.

Figure 2.1 Location of (i) the study sites; ii) the River Ray 'catchment' (study area).

(i)



(ii)

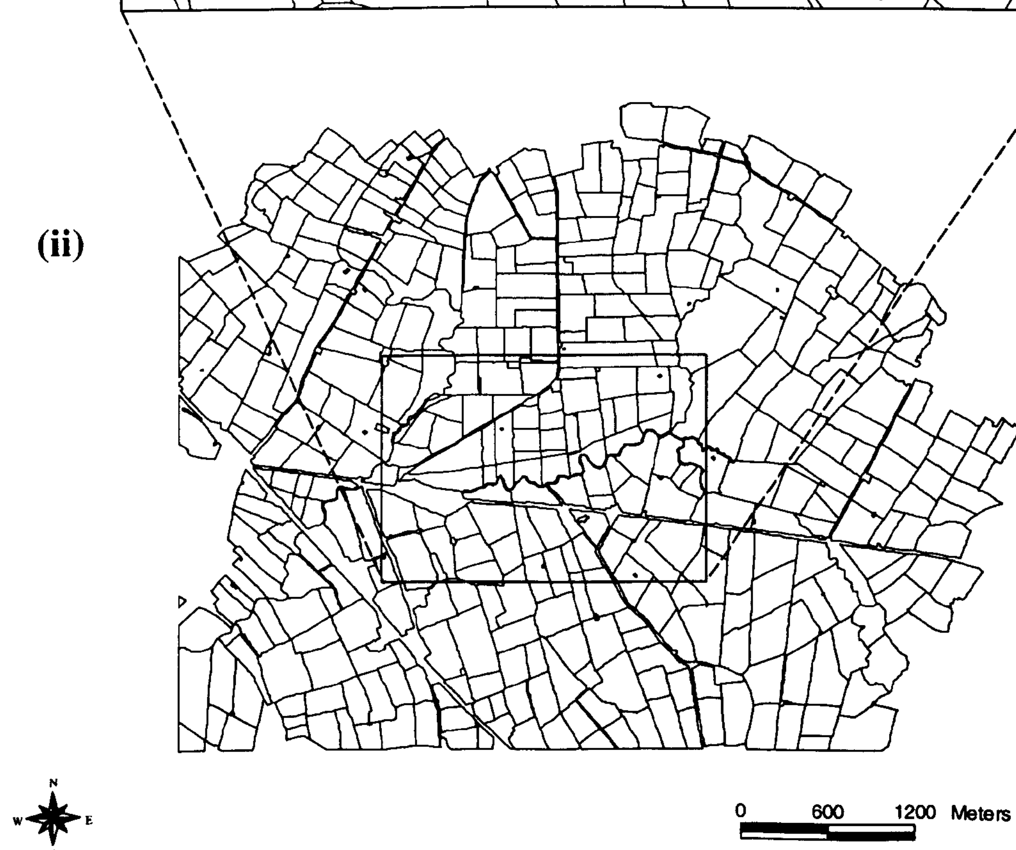
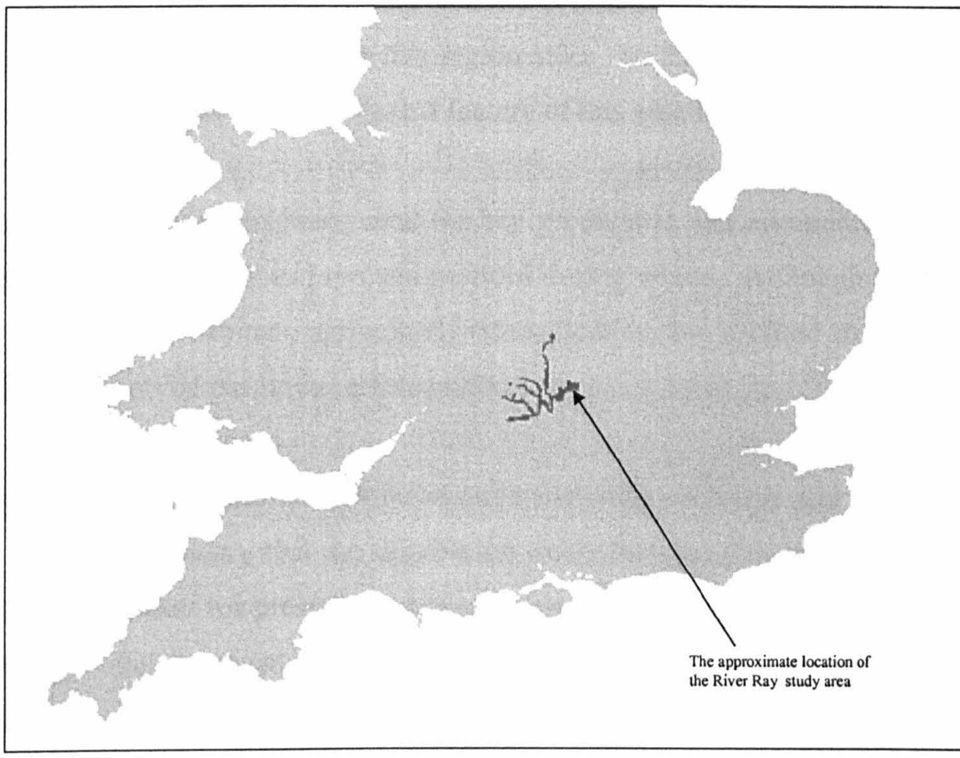


Figure 2.2 The location of the Upper Thames Tributaries Environmentally Sensitive Area.



2.2 The Upper Thames Tributaries

The Upper Thames Tributaries (UTT) Environmentally Sensitive Area (ESA) was designated in 1994 because of its river valley grassland and other valuable wetland habitats, with the main ecological interest of the area lying in the unimproved and other extensively managed wet meadows and pastures. The ESA itself is based around the flood plains of the Rivers Ray, Cherwell, Glyme, Evenlode and Windrush, together with the Upper Thames from Oxford to Kelmscott (MAFF, 1992).

Lambrick & Robinson (1988) reconstructed the development of floodplain grasslands in the Upper Thames valley. Until the Iron Age the water table was relatively low, with flooding and alluviation minimal. After this period a rise in the permanent water table occurred, resulting in flooding and impeded surface drainage. During the Roman period a substantial thickness of clayey alluvium accumulated, and flooding increased while surface drainage became progressively worse. Grassland has been the predominant vegetation type of the Upper Thames floodplains since the clearance of woodland, and composition would have been considerably influenced by the altered hydrological regime. The best known botanical community of the region is the *Alopecurus pratensis-Sanguisorba officinalis* grassland (MG4). Favourable conditions for this community include seasonal flooding and a deep alluvial profile with free drainage in summer, conditions which have occurred in this region since the Roman period, indicating that MG4 communities have probably been a feature of this area for at least 2000 years.

Traditionally, the meadows were used for hay production and aftermath grazing, with extensive flooding of the meadows and pastures during winter. Although the UTT ESA is still rich in these habitats, agricultural intensification has resulted in conversion of grassland to improved pasture or arable cultivation.

Agri-environment schemes are not necessarily mutually exclusive and there are fields within the ESA boundary that are eligible for entry into the Countryside Stewardship Scheme (CSS), either for prescriptions that are not available under the ESA scheme or because these fields were entered into the CSS before the ESA was designated.

2.3 Characterizing the study area

2.3.1 Soils and hydrology

The majority of the study area (the catchment of the River Ray) is underlain by Oxford Clay. The soils of the area are dominated by the Fladbury series and the Denchworth soil associations (Clayden & Hollis, 1984). These clay soils are strongly gleyed and highly impermeable and, together with seasonal inundation, are largely responsible for maintaining the wetness of the area. Soils of the Denchworth series (undisturbed alluvial clay) have a high clay content, and in the absence of drainage there is very little downward movement of water and so the dominant hydrological process is runoff. The Fladbury soil series (same material reworked by alluvial processes) has a slightly less clayey texture, but similar properties. Both soils are inherently wet, and are thus frequently improved by underdrainage. Indeed, both mole drainage and deep subsoiling have been employed on the floodplain of the River Ray (Armstrong *et al.*, 1996).

The hydrology of three of the study fields (REV, PL, SA123) was monitored between 1993 and 1996 by ADAS. Dipwells were aligned in transects perpendicular to both the New Drain and the River Ray itself, in order to characterise the watertable between field centres and ditches. Autographic watertable meters were installed in these fields to continuously record the depth to the watertable in the field centres. Autographic ditch water level recorders in the River Ray and New Drain provided data on the relationship between ditch water levels and in-field watertables. In addition, hydraulic conductivity tests were carried out in SA123 to determine the ability of the soil to transmit water (Rose & Armstrong, 1994).

The soil within SA123 is unable to transmit water rapidly to depth and a perched water table is quickly created during wet conditions, leading to frequent surface flooding caused by saturation of the topsoil (Rose and Armstrong, 1994). Despite the adjacency of the River Ray and the New Drain, the soil also does not rapidly transmit large volumes of water laterally. Soil surface saturation and surface flooding largely influence the position of the water table. SA123 had been underdrained, and the residual effects are still apparent: during periods when the water levels in the water courses are low, there is substantial drawdown of the water table profile from the field centre towards the River and New Drain. An undrained meadow along the same stretch of river (PL) generally had a water table 20-30cm nearer to the soil surface than the ex-arable field during periods when the surface was not inundated. During winter and spring, the undrained meadow was constantly in a state of shallow flooding or near to soil saturation, conditions typical of undrained wet meadowland on clay soils. The drained field (SA123) experiences a wider range of depths to the water table over time, indicating that the underdrainage still has some effect in drawing the water table down. However, the level of water table control by the underdrainage is less efficient than would be expected if it were regularly maintained.

The channel of the River Ray has been both deepened and straightened in the past to prevent flooding and improve drainage of adjacent land and also downstream areas. As a result water levels in the river can be very low, particularly during the summer months. However, relatively small rainfall events can cause the water level in the River Ray to rise quickly as runoff from the surrounding land reaches the channel. Flow constrictions further downstream cause delayed drainage and increased flooding.

In addition, microtopography within fields affects hydrology. The ground level rises from the New Drain towards the River Ray, indicating that material removed from the River during past clearance operations has been spread along the riverbanks. This rise in ground level means that after flood events, when the river level drops, water left on the undrained fields can only return to the river via the New Drain.

Microtopography and surface flooding largely control the soil hydrology. Field underdrainage systems have not been efficiently maintained and are of lesser importance. Control of the watertable for the maintenance of wetland vegetation will only be achieved by retaining flood water on the soil surface during the spring. Within this period, the natural wetness of the impermeable soils should be sufficient to maintain wetland vegetation. Retaining a watertable within 50cm of the soil surface over the summer will prove extremely difficult (Rose and Armstrong, 1994).

2.3.2 Communities and species

In 1993, several fields of different management history[1] (including the study fields) were surveyed in detail. Marked differences in species richness (m^{-2}) existed between swards of differing management histories, with unimproved grassland (19.94 ± 0.46 species) richer in species than improved swards (14.27 ± 0.52), which were themselves richer than swards reverting from sown grassland (9.54 ± 0.27). Poorest of all were the set-aside fields (<2 years duration) with an average of 6.96 ± 0.37 species m^{-2}.

The unimproved grasslands included more species typical of damp sites than the improved fields. Indeed, the increased species richness of the unimproved swards is due largely to a better representation of *Carex* species, *Juncus* species and obligate wetland forbs. Additionally, a trend was observed from grasslands composed of species typical of average (or somewhat lower) nitrogen availability in the unimproved fields to one composed of species indicative of more nitrogen-rich sites in the set-aside fields (Table 2.1).

Table 2.1 Mean Ellenberg moisture (mF) and nitrogen (mN) indicator values

Management history	mF	mN
Unimproved	6.48	4.66
Improved	5.93	5.36
Reverting	5.95	6.46
Set-aside	6.12	6.85

This reflects the fact that unimproved fields tend to contain more species typical of above average soil moisture conditions and below average nitrogen availability than any other management type.

The results of the vegetation survey were entered into TABLEFIT (Hill, 1991) to assign the observed vegetation to NVC community types. Unimproved grassland swards on the floodplain of the River Ray was found to be a mosaic of three main communities:
MG4 *Alopecurus pratensis-Sanguisorba officinalis* grassland (more elevated areas);
MG8 *Cynosurus cristatus-Caltha palustris* grassland (lower-lying areas liable to flooding); and
MG9 *Holcus lanatus-Deschampsia cespitosa* (damp hollows)

In addition, elements of MG5 *Centaurea nigra-Cynosurus cristatus* grassland occur in higher areas such as the ridges of ridge-and-furrow fields.

The application of high rates of nitrogenous fertiliser to previously unimproved grassland generally results in a decrease in species-richness. Indeed, where wet grasslands have received high rates of nitrogen application, more species-poor, mesotrophic swards generally replace the characteristic vegetation. Fields 'improved' for agriculture in the area thus tend to develop vegetation communities that reflect both improved drainage and higher levels of fertiliser application:
MG6 *Lolium perenne-Cynosurus cristatus* grassland; and
MG7 *Lolium perenne* leys and related grassland

[1] Unimproved – undrained old grassland, no evidence of fertiliser use; improved – old grassland that has been drained and/or fertilised but not reseeded; reverting – reseeded grasslands and leys, now managed extensively; set-aside – sites where intensive arable production has ceased

2.4 The Geographical Information System

Available information as described above (sections 2.3.1 and 2.3.2) was supplemented with the results of intensive botanical survey of the majority of grass fields within the catchment in 1994 and 1995. The information was entered into a Geographical Information System (GIS) to allow spatial visualisation and analysis of pattern from spatially referenced data for the River Ray catchment. Aerial photography, commissioned by the ITE from Geonex Ltd. of Leicester, was digitised and the line-work imported directly into the GIS. Detailed land use information was interpreted directly from the aerial photographs. The GIS was developed using ARC/INFO software on a SUN SPARC 20 workstation, but has been imported into ARCVIEW, a PC-based GIS package, for the purposes of this study.

2.4.1 Data available for the study area

Land use, agriculture and management. The original base-map and interpreted land-use classification was compared with an independent classification of agricultural land-use on a field-by-field basis. Management information for the majority of grassland fields includes the presence of cutting, grazing, cattle, sheep (together with grazing intensity in livestock units), underdrainage, herbicide application, use of farmyard manure, and application rates for inorganic fertiliser. Additional information classifies the fields in terms of designation, i.e. Sites of Special Scientific Interest (SSSI), Countryside Stewardship scheme (CSS) fields, or fields entered into the Environmentally Sensitive Areas (ESA) scheme itself.

Ecology: plant species and communities. For each grassland field in the study area (>220 in total), a species list is available (qualitative), compiled during a systematic walk-through. In addition, between 3 and 6 m^2 quadrats were recorded within each field and so quantitative information is also available. These datasets were derived from the results of intensive field surveys during 1994 and 1995 when all grassland fields in the study area were surveyed, with the exception of a small number where access had been denied. The corridor of fields bordering the River Ray (Figure 2.1) are the main focus of this study and these had been surveyed during 1993, with between 20 and 60 quadrats recorded per field. In total, the quantitative dataset, formed from the 1993, 1994 and

1995 surveys, contains over 1200 individual quadrat records. Each of these quadrats was classified according to NVC community type using Tablefit (Hill, 1991). Nomenclature follows Stace (1997) for higher plants and Smith (1978) for bryophytes.

Hydrological information for the River Ray falls into three main categories, namely: monitored dipwell data from selected fields, modelled data for each field and subjective information on flooding frequency. ADAS Land Research Centre provided the monitored and modelled data.

2.5 Assessment of site suitability for restoration

With the information available at the start of the study, the focal site (SA123) was considered to be suitable for restoration to lowland wet grassland due to a number of factors:

- despite hydrological 'improvements', the River Ray (bordering the site to the south) does flood, resulting in seasonal inundation of the study site (and area). Even though previously underdrained, SA123 experiences a 'natural' hydrological regime (which it shares with adjacent existing wet grassland) and thus would not require engineering work to reinstate appropriate hydrological conditions. In addition, the existing drainage channel (the New Drain) directly to the north of the field offered the potential to manipulate water levels if required;
- SA123 is immediately adjacent to an unimproved species-rich wet meadow (LH). It is generally acknowledged that chances of successful restoration of vegetation may be improved by proximity to a source of propagules;
- restoring the arable field adjacent to the nature reserve will add to the existing area of semi-natural and rehabilitating grasslands, and should function to buffer the nature reserve to some degree from the effects of intensive agriculture; and
- such a frequently inundated site on heavy clay soil is probably marginal for agriculture (see Chapter 8 for discussion of nutrient status).

CHAPTER 3 DERIVING SITE-SPECIFIC TARGETS FOR ARABLE REVERSION TO WET GRASSLAND

3.1 Introduction

The study site, an ex-arable field (SA123) within the Upper Thames Tributaries ESA, was actually entered into the Countryside Stewardship Scheme just prior to the designation of the ESA. Since the ESA designation, however, all land now entering agri-environment schemes within that designated area is entered into the ESA scheme and so this thesis considers the restoration in the context of the ESA.

Within the boundary of the Upper Thames Tributaries ESA, arable land is eligible for reversion to extensive permanent grassland for the benefit of wildlife and the landscape (Tier 3A) or to wet grassland for increased benefit to wildlife (Tier 3B) (MAFF 1992). Both tier 3A and 3B are targeted at arable land on the floodplain, tier 3B particularly where it adjoins existing areas of wet grassland. Reversion to wet grassland (or extensive permanent grassland) is expected to 'encourage a gradual recolonisation of the characteristic wildlife of river valley grassland' and 'enhance the river valley grassland landscape' (MAFF, 1992). The study site, adjacent to a SSSI, is thus suitable for entry into Tier 3B for reversion to wet grassland.

The ability to evaluate effectiveness of restoration depends on clearly defined objectives. A rigorous and repeatable approach to deriving criteria and evaluating success is desirable. The aims of both the ESA Tier 3B (reversion of arable land to wet grassland) and of Countryside Stewardship 'restoration of waterside landscapes' (regeneration of semi-natural vegetation on cultivated land) are too broad for precise definition of targets or for evaluating the success of restoration at the local level.

When this study was begun, reversion Tier 3B required a permanent grass sward to be established using at least five species chosen from an approved list of grass species (table 3.1). Seed of wild flower species typical of wet grassland could be included in the seed mixture in addition to the specified grass species if desired. Within the Countryside Stewardship Scheme the prescription for the regeneration of semi-natural vegetation on cultivated land was similar, although there was no provision for including wildflowers and a minimum of four grass species (rather than five) should be chosen, appropriate to locality and soils, from the approved list (table 3.1). The agri-environment recommendations are based on the premise that a grass 'matrix' will allow for subsequent community development following spontaneous colonisation by 'target' herbaceous species and do not therefore recommend or support a diverse seed mixture.

Table 3.1 Approved list of grass species for UTT ESA Tier 3B and Countryside Stewardship Scheme (CSS) restoration of waterside landscapes.

Species	UTT ESA		CSS
	Tier 3A	Tier 3B	
Agrostis capillaris	•	•	•
Alopecurus pratensis	•	•	•
Anthoxanthum odoratum	•	•	•
Briza media	•		
Bromus commutatus			•
Cynosurus cristatus	•	•	•
Dactylis glomerata			•
Deschampsia cespitosa			•
Festuca arundinacea	•	•	•
Festuca ovina			•
Festuca pratensis	•	•	•
Festuca rubra	•	•	•
Holcus lanatus	•	•	
Hordeum secalinum			•
Phleum pratense	•	•	
Phleum pratense bertolonii			•
Poa pratensis	•		•
Trisetum flavescens	•		•

3.2 National targets for the restoration of lowland wet grassland

Nationally, particular concern and conservation effort is focused upon the unimproved, species-rich community types, i.e. MG5, MG8, and notably the MG4 flood-meadows (see Chapter 1, Section 1.3). National and regional vegetation targets for lowland wet grassland were established within the framework of the National Vegetation Classification (NVC) for the Upper Thames Tributaries Environmentally Sensitive Area

(ESA) in Chapter 2 and Manchester *et al.* (1999). However, these targets need refining for the local area before implementation in habitat restoration.

3.3 Local restoration targets

Within the study area, none of the target NVC community types occurs in isolation (Chapter 2) and so it would be inappropriate to attempt to restore a whole field to one community type only. Another consideration when using the NVC is the need to maintain characteristic natural species distributions: it would be inappropriate to introduce constituent species of community types that do not occur naturally within the region.

In this study, locally applicable targets were defined broadly in terms of NVC types, but specifically in terms of local community composition (Chapter 2).

3.3.1 Lowland wet grassland targets for the study area

It is important to ensure that the area chosen for restoration currently supports, or has in the past supported, the target communities. A botanical survey of river valley grasslands within the study area (Treweek *et al.*, 1993) found the older swards to be a mosaic of MG4, MG8 and MG9, with MG5 limited to areas of higher elevation and freely draining soils.

Whilst MG4 is the only relevant community type to be explicitly mentioned in a European context, with the publication of Biodiversity Action Plans, any of the less 'improved' communities could be considered of importance and in need of protection and restoration. Moreover, they are all important in the context of the UTT ESA (Lambrick and Robinson, 1988). Whilst maintenance of the existing communities may ensure that all relevant species groups benefit, habitat restoration on the floodplain should focus principally upon the reinstatement of species-rich wet grassland.

3.3.2 Site-specific targets for the restoration experiment

Once broad targets have been identified at the national scale, and modified regionally, site-specific targets can be formulated. For the purposes of this study, a local 'reference habitat' ('target' community) was used to define the targets for the reversion experiment, thereby ensuring consistency with both the local environment and national species distributions. Long Herdon (LH) Site of Special Scientific Interest (SSSI), adjacent to the ex-arable reversion site, was chosen as the 'template' for the restoration. The reference habitat currently supports species-rich wet grassland and has not been improved or cultivated for at least 60 years (Lambourne, pers.comm.).

3.3.2.1 Target NVC community types

The NVC (Rodwell, 1992b) was used to characterise the vegetation of the reference habitat, in order to permit subsequent extrapolation to other sites. The results of the intensive 1993 vegetation survey were analysed using a computer program written specifically for the NVC, TABLEFIT (Hill, 1991), to classify the vegetation. The program measures goodness-of-fit between data on observed vegetation and descriptions of standard types in association tables and assigns the observed vegetation to the closest community type (provided that the vegetation corresponds to a type included within the NVC). The NVC is rapidly becoming the standard classification of British vegetation as used by ecologists, and TABLEFIT is a means of simplifying the process of assigning vegetation to a classification. In addition, the use of a package such as TABLEFIT removes the subjectivity associated with the matching of vegetation to written keys and community descriptions by the individual.

The vegetation of the reference habitat did not give an obvious 'fit' to any one NVC community type. Instead, it approximated to a mixture of Mesotrophic Grassland (MG) 4 (*Alopecurus pratensis-Sanguisorba officinalis* grassland), MG8 (*Cynosurus cristatus-Caltha palustris* grassland), MG9 (*Holcus lanatus-Deschampsia cespitosa* grassland) and MG5a (*Lathyrus pratensis* subcommunity of *Cynosurus cristatus-Centaurea nigra* grassland). With the exception of MG9 (unimproved but species-poor), these communities are characterised by a species-rich, variable sward of grasses and herbaceous dicotyledons, with no single, constantly dominant species (Rodwell, 1992b).

There is much variation within fields, as well as between fields. Reasonably small changes in topography, hydrology and soils are responsible for transitions between community types. It was deemed inappropriate to isolate any particular community type as an absolute target, particularly since the target vegetation comprises elements of all communities of old grassland within the region.

3.3.2.2 Target species

The survey of the reference habitat also led to the formulation of specific floristic targets for the experiment. Officially scheduled sites (such as the SSSI reference habitat) may be assumed to be 'valuable' and hence all grassland species recorded were considered suitable for the restoration and were designated as Class II target species (Appendix 3.1). The appearance of any of these within the restored vegetation would be considered desirable since they are constituents of old, unimproved wet grassland in the area. Deliberate reintroduction of *all* species present within the reference habitat would not be feasible; not only were many of the species commercially unavailable, and perhaps difficult to hand-collect in sufficient quantity, but the inclusion of such a diverse range of species would make any seed mixture prohibitively expensive to all but the richest landowners or committed conservation organisations. The proximity of a species-rich source of suitable propagules should also ensure that at least some of the species might colonise the restoration site naturally, making deliberate reintroduction unnecessary. The seed bank of the ex-arable field could potentially contain seeds of desirable species, either that were present prior to arable cultivation or that have dispersed from nearby grassland and been incorporated into the soil seed bank during the period of cultivation.

Accordingly, a subset of the Class II species (Class I species; Appendix 3.2) were defined on the basis of species' requirements for available soil moisture and nitrogen (Ellenberg, 1988). The species list for the SSSI was classified according to their 'F' (moisture) and 'N' (fertility) indicator values as assigned by Ellenberg (1988). 'F' values are based on the correlation between species distributions in relation to site wetness, representing a continuum from dry rocky slopes with shallow soils, to marshy ground, and then shallow and deep, open water. An 'F' value of less than 5 indicates a preference for drier sites, and of greater than 5 for wetter sites. Similarly, 'N' values represent the distribution of species in relation to available soil nitrogen. An 'N' value of less than 5 indicates a preference for sites with below average available nitrogen, while a value greater than 5 indicates a preference for sites with above average nitrogen availability. Selection of species for inclusion in experimental seed mixtures was based on the use of these two indicator values, those species characteristic of wetter and less fertile conditions being preferentially selected over those characteristic of more well-drained and nitrogen-rich conditions. Once species considered broadly suitable in terms of soil moisture and nitrogen preferences had been selected, the list was further restricted by exclusion of species regarded as ruderal or weedy (Treweek and Mountford, pers.comm.). The list of species remaining (Class I target species; Appendix 3.3) would form the 'core' of desirable species for inclusion within wet grassland from which experimental seed mixtures were derived.

3.3.2.3 Derivation of seed mixtures

The mean ground cover of all Class I species in the reference habitat was calculated and species were ranked according to this value. The ranks were then used as a guide to the proportions, by weight, of each species to be included in the seed mixtures (at a ratio of 80:20 grasses: herbs to reflect the fact that semi-natural grasslands are largely grass-dominated).

Seed Mix 1 (appendix 3.3) was representative of the species mixtures recommended by the agri-environment schemes (table 3.1). The species chosen were all of fairly frequent occurrence in the SSSI. Species likely to do particularly well on improved soil or in disturbed conditions, and thus become dominant at the expense of more desirable species, were excluded. Due to its fairly ubiquitous presence within the communities that are characteristic of lowland wet grassland, *Alopecurus pratensis* was selected for inclusion within the mix despite an apparent preference for nitrogen-rich conditions. *Anthoxanthum odoratum* was not available commercially, and sufficient quantities of seed of this species could not be readily hand-picked. Thus another species (*Phleum bertolonii*) was chosen from the Countryside Stewardship recommended list as a replacement.

Seed Mix 2 included a limited number of herb species with the grasses of seed mix 1 plus two others. The herb species were commercially available and relatively inexpensive. The aim of this 'intermediate' mix, that included a wider range of species than Seed Mix 1, but omitted the rarer, more expensive seed of Seed Mix 3, was to increase floristic diversity at relatively low extra cost.

Seed Mix 3 was a much more diverse mix based on the species composition of the adjacent SSSI. Although including 23 species of grass and forb, it was originally planned to include 34 of the 77 plant species recorded in the SSSI during 1993. However, commercial availability of species meant that not all the required species could be acquired. This seed mix would be an expensive alternative with the aim of achieving the desired floristic diversity in a short time.

3.4 Deriving evaluation criteria

If the experimental treatments had been based specifically on NVC community composition, a simple measure of success might have been the similarity of vegetation to the NVC community type. However, since grassland vegetation in proximity to the restoration site comprised a number of community types, the NVC was not an appropriate 'yardstick' against which to assess restoration. A more appropriate target for restoration was the vegetation present within the reference habitat (in this case the SSSI). Species were only designated as 'targets' if present in the reference habitat and their contribution to seed mixtures was determined from their relative abundance. Restoration success could be judged by comparing the recorded composition and abundance of species in the new habitat with those of the reference habitat. This comparison is the ultimate measure of success, but vegetation is dynamic and individual species are unlikely to achieve the same frequency and abundance within different fields especially over a short time period of 3-5 years.

A better way of determining the success of restoration would therefore be to assess the progress of the vegetation towards the target. In order to do this, evaluation criteria must first be established.

The target communities are all species-rich, and thus the presence of high numbers of species is a desirable criterion. However, the sward with the highest *total number of species* present will not necessarily be the best, as these species may be inappropriate for the target community (e.g. arable weeds). Therefore, some measure of the presence of 'habitat-specific' or 'desirable' species is a second criterion. The vegetation may be assessed in terms of the *number and ground cover of both Class II and Class I target species*. The target vegetation (the reference habitat and the NVC target communities; see Manchester *et al.*, 1999) is also species-rich at a small-scale so a third criterion is *small-scale species-richness* (species m^{-2}).

National Biodiversity Action Plan targets are based on the NVC. To assess the contribution of restored sites towards meeting these targets, some measure of the *similarity of restored swards to the NVC target vegetation communities* is needed and becomes the fourth criterion.

Lastly, the effectiveness of the individual seed mixtures used must be considered, and so the success of *establishment of sown species* is the fifth criterion.

In summary, the overall measure of success is the degree to which the restored habitat compares with the reference community. Six criteria were adopted as measures of progress towards the re-establishment of species-rich lowland wet grassland:

I total number of species present

II numbers and ground cover of Class II and Class I species

III small-scale species-richness (m^{-2})

IV similarity to identified NVC target vegetation communities

V success of establishment of sown species

VI the performance of individual species

3.5 Assessing the use of a reference habitat

The reference habitat, Long Herdon SSSI, was re-surveyed in 1996 to establish whether the vegetation had been stable between 1993 and 1996. In 1993, 73 species had been recorded and in 1996, 76 species were recorded. 67 species were present in both surveys, with 6 of the species recorded in 1993 being missing from the 1996 survey, whilst the flora gained 9 species not recorded during 1993. The absolute number of species was thus not dissimilar between years and all species identified as 'targets' in 1993 were again present in 1996. However, there had been considerable changes in the abundance of individual species (Appendix 3.4 and Table 3.2), and of groups of species (Table 3.3).

Table 3.2 Species varying significantly in abundance within the reference habitat between 1993 and 1996

Species	Significance (p)	Change
Agrostis canina	<0.01	+
Carex nigra	<0.05	-
Carex panicea	<0.05	-
Centaurea nigra	<0.01	+
Festuca pratensis	<0.01	+
Filipendula ulmaria	<0.10	+
Holcus lanatus	<0.01	-
Hordeum secalinum	<0.10	-
Juncus conglomeratus	<0.001	-
Vicia hirsuta	<0.10	+

Table 3.3 Ground cover of species groups within the reference habitat (Long Herdon SSSI).

Sample date	Grass	Forb	Sedge/rush
1993	56.31	23.67	20.02
1996	59.79	34.23	5.98

Although no Target Species were lost from the reference habitat, changes in the abundance of species between years do have implications. If the reference habitat is being used to formulate seed mixtures for the reintroduction of species to restoration sites, the altered abundance of individual species would have implications not only for the composition of any seed mixture, but also for the overall cost of the restoration project (Appendix 3.6) depending in which year the reference habitat was surveyed. Changes in the reference habitat over time will make it difficult to assess the success of restoration in terms of the similarity of the restored vegetation to its target. If the target was the reference habitat in year t, then should the comparison of the restored habitat in year t + x be with the reference habitat in year t or year t + x?

3.6 Discussion and conclusions

The agri-environment schemes (ESA, CSS) did not provide the clear guidance necessary to adequately plan and execute ecological restoration. This chapter has investigated the derivation of targets and evaluation criteria for the re-creation of wet grassland upon ex-arable land. Without clearly defined objectives (i.e. endpoints) at the outset, it will be difficult to assess the success of restoration or to tailor agricultural management to achieve the desired results. Where a reference habitat is to be used to guide restoration, the aim is simply to restore vegetation that approximates to this target. By characterising the target vegetation, it is possible to derive measurable criteria so that progress towards the endpoint can be objectively determined. If the restored vegetation does not 'progress' towards the target, this will be identified by the evaluation criteria and, if appropriate management intervention is required, the criteria should also help focus which aspect of the restoration is failing.

A further benefit of using measurable criteria to monitor the success of restoration arises from the temporal variability of vegetation. The reference community selected for this study has not remained static over time but, as is the case for the majority of vegetation, has been dynamic with species replacing one another and large changes in the abundance of individual species and groups of species also. Although these fluctuations will make it difficult to use the reference community as a 'yardstick' for the restoration, criteria based on the characteristics of the reference habitat should remain appropriate.

CHAPTER 4 CHARACTERISING THE SEED BANKS OF FLOODPLAIN (AND FORMER) GRASSLANDS

4.1 Introduction

Following soil disturbance, and in the absence of artificial introduction, the species that (re-) colonise are determined by naturally occurring sources of propagules. Since the majority of seeds disperse short distances only, the contribution of *ex situ* sources of propagules (the seed rain) is likely to be negligible. Seeds already *in situ* (the seed bank) will be largely responsible for shaping early vegetation development.

If seeds of desired species are present within the soil, the seed bank can act as a source of recruiting species during habitat restoration. However, the literature suggests (section 1.5.1.1) that the seed banks of intensively farmed land are generally unsuitable for the restoration of species-rich vegetation (e.g. Graham & Hutchings, 1988a,b; Jefferson & Usher, 1987), whilst sites where native vegetation has only recently been eliminated make the best candidates for restoration (Leck *et al.*, 1989).

Whilst seed banks of many types of grassland have been studied (see Chapter 1), when this investigation began, studies of floodplain grasslands were scarce (except see McDonald 1993; McDonald *et al.*, 1996).

This study aimed primarily to determine the potential for the soil seed bank of the ex-arable reversion site to contribute to the restoration of lowland wet grassland. Previous studies found that with increasing time under arable cultivation seeds of 'desirable' species decrease whilst those of annual arable weeds increase. The seed bank of the study site may have been of little use in the restoration of species-rich vegetation. However, this site may not be typical of all arable fields as at the time of writing it was in close proximity to existing species-rich grassland and potentially received propagules of grassland species via the airborne seed rain. The site was also still subject to seasonal inundation, and so there was the potential for seed dispersal by water. Characterisation of the seed bank of the study site was considered necessary to determine whether the existing soil seed bank was an adequate source of appropriate colonising species in sufficient quantities to revert the site to grassland.

The literature also highlights the lack of correspondence between the seed banks of established vegetation and the associated aboveground flora. If this is the case, then even sites where target vegetation has only recently been eliminated will not necessarily be suitable for restoration. In addition to the ex-arable reversion site (SA123), a number of adjacent grasslands at various stages of improvement were also selected for investigation: LH, PL (unimproved grassland); IE, IW (improved grassland); REV (re-seeded grassland) (see Chapter 2 for description and figure 2.1 for location).

If increasing levels of intensification do indeed result in decreasing suitability of the seed bank for restoration of species-rich grassland, there should be a 'gradient' in the seed banks of the fields studied. However, as the fields studied were adjacent to one another, and were all subject to inundation, seed transfer between fields may have resulted in increased homogeneity of seed bank composition.

Characterisation of seed bank composition will inform habitat restoration, i.e. whether the existing ex-arable soil seed bank is an appropriate source of colonising species or whether species need to be artificially reintroduced. Moreover, the wider investigation should determine which suitable species (if any) form persistent seed banks (in grassland or arable soils) at densities suitable for habitat restoration. In addition, characterisation of the seed banks of existing wet grasslands at various stages of improvement provides a baseline against which to assess the arable seed bank.

The determination of the composition and size of the ex-arable seed bank also complements the field experiment (Chapter 5) that investigates techniques for the re-creation of lowland wet grassland. The results of the present study will aid interpretation of the results of the field experiment, by making it possible to attribute the presence of establishing species to a particular source. Ideally the seed bank would have been characterised prior to establishment of the field trial. However, time constraints meant that this was not possible and both studies ran concurrently.

4.2 Method

4.2.1 Sampling

The majority of seed within soil is present within the surface 5cm (Thompson and Grime, 1979; Roberts, 1981). The objective behind sampling was primarily to investigate the total seed content of the soil, and not to differentiate between the different soil profiles, samples were taken to a depth of 5cm using a soil corer of diameter 4cm (soil volume of $62.83cm^3$). Aboveground vegetation and litter were removed from soil samples.

4.2.1.1 Ex-arable reversion field (SA123)

Soil samples were removed following seedbed preparation, prior to the introduction of seed during September 1993. Five cores were removed within each of 30 experimental plots (chapter 5, figure 5.1), positioned, as are the spots on the five-face of a die. The five cores from each plot were then pooled, giving a total of 30 samples, each of volume $314cm^3$.

4.2.1.2 Grassland fields

Samples were also taken from REV, IE, IW, LH and PL. These grasslands were surveyed in 1993 using quadrats located along north-south transects (from the 'New Drain' towards the River Ray). The first quadrat was located 5m south of the ditchbank, with subsequent quadrats recorded at 10m intervals along these north-south transects at 90° to the ditch. The transects were relocated and soil samples taken at 7m south of the ditchbank and every 10m thereafter. At each sampling point two soil cores (4cm diameter) were removed to a depth of 5cm from within 25cm of one another.

4.2.2 Investigation

Soil samples were air dried in a glasshouse during September 1993. Samples were thoroughly mixed and divided into two equal parts by volume: one half to be grown on immediately, whilst the other was stored until the following spring. Accordingly, one half of each sample was sprinkled onto sterile compost in a plant tray (20cm x 15cm) lined to prevent seed loss and covered by a thin layer of compost, to be grown on over winter, whilst the other half was placed in a cold room (at 4°C).

Samples were watered regularly, and germinating seedlings identified before removal, with numbers of emerging seedlings of each species recorded per sample on a monthly basis. Following removal of seedlings, the soil was stirred to encourage further germination. The chilled samples were removed from cold storage in the spring of 1994 and the experiment repeated. The experiment was monitored for two years, ending in the autumn of 1995 for the non-chilled samples, and in the spring of 1996 for the chilled samples.

Seedlings were identified at the first true leaf stage if possible, and if not were transplanted and grown on until identifiable. Most species were identified, but certain caused problems, for example, seedlings of *Brassica, Bromus, Polygonaceae* and *Rumex* species were not identified past this level.

4.2.3 Analysis

Seed bank data do not generally lend themselves to parametric statistical methods. Data on the distribution and density of seeds in the soil are often difficult to analyse. Seed bank data are often highly skewed and standard transformations such as log and square roots are insufficient to normalise the data (Warr *et al.*, 1993). Statistical tests are only suitable for analysing the distributions of species that occur abundantly in the seed bank. The species of greatest ecological interest are often present at low frequency and abundance. Moreover, the majority of species occur at low frequencies and low abundance, whilst only a few species occur widely and abundantly.

Much of the interpretation of seed bank results is necessarily qualitative. However, within this study a variety of numerical and statistical techniques were applied to the data, in addition to presentation of the data in descriptive form.

For each species, mean vegetative (above-ground) cover and mean number of seedlings m^{-2} (seed bank) were calculated for each field.

Differences between fields were investigated using analysis of variance and Tukey's HSD:

1. The mean species richness of grassland field seed banks (m^{-2}) was compared, in terms of seedlings of all species, seedlings of grassland and weed species. Numbers of weed seedlings were log transformed prior to analysis. Seedlings emerging from the ex-arable seed bank were not included in this analysis because of the unequal volume of soil sampled.
2. Differences in mean Ellenberg moisture (F) and nitrogen (N) values between fields.
3. Differences in the abundance of individual species within the seed banks of different fields. Only the more abundant species could be analysed and thus species occurring at seed densities greater than 20 seeds per field were included. Data were logarithmically transformed prior to analysis.

The relationship between the relative abundance of individual species in the seed bank and the vegetation was tested by rank correlation (null hypothesis: there is no relationship between abundance in the seed bank and in the vegetation).

For the purposes of ecological restoration of plant species, the possession of a long-term persistent seed bank is a desirable characteristic that enables vegetation recovery following perturbation. The seed bank data was also investigated according to seed bank types I-IV (*sensu* Thompson & Grime, 1979), a classification that describes the longevity of seeds in the soil. Type I and II species have transient seed banks: seeds of type I species germinate in the autumn following seed shed, type II species germinate in the spring of the following year. Seeds of type III species either germinate soon after seed shed or become incorporated into the long-term persistent seed bank, while seeds of type IV species are truly persistent and enter the long-term seed bank.

Similarity between the composition of seed banks of different fields and also between seed banks and the associated aboveground vegetation were assessed primarily by ordination techniques. Seed bank data were standardised as a percentage of the total number of seedlings emerging within each field. Aboveground vegetation data (1993 and 1996 surveys) were standardised as a percentage cover value following removal of the contribution made by bryophyte cover, litter and bare ground. Ordination was performed by Detrended Correspondence Analysis (DECORANA) (Hill, 1979, 1994). Default values were used and the resulting species and site scores were plotted. The mean seed bank composition of the ex-arable field was included within the ordination despite the lack of comparable aboveground vegetation data in order to compare the position of the arable seed bank with those of the grasslands in ordination space.

In addition to ordination techniques, similarity between seed banks and above-ground vegetation was quantified. Euclidean distances and Sorensen Community Coefficients were calculated between samples. The Sorensen Community Coefficient is calculated using presence absence data and thus all species are given the same emphasis. Euclidean distance is calculated using abundance data, and emphasizes larger abundances (dominant species are given greater weight).

Lastly, the homogeneity of the study field was investigated. It is possible that seeds will not have been accumulated uniformly through the field due to the changes in microtopography within the fields adjacent to the river (section 2.5.1), and also due to the differing land uses in adjacent fields. Because the sampling locations within the study field were based on the experimental layout for the subsequent field experiment, the effects of differing positions within the field could be examined. Differences in mean Ellenberg indicator values, and mean numbers of seeds and species, between blocks (adjacent to the river, middle of the field, adjacent to the drainage channel) were analysed. Numbers of seedlings emerging were also summarised at the plot level and input into DECORANA to look for patterns in the distribution of species between plots.

4.3 Results

4.3.1 Summary: by field

The largest number of species germinated from the seed bank of the ex-arable field (Table 4.1), although the majority of these species were arable weeds (Appendices 4.1 and 4.2). 60% of all seedlings germinated from the chilled samples, and this proportion of autumn versus spring germination was similar for the less improved grasslands. The two most improved fields (SA123, REV) were the exceptions, with a greater number of seedlings germinating from the non-chilled samples. The differences between chilled and non-chilled samples were not as marked in terms of the numbers of species emerging, in total or for any individual field.

Table 4.1. Numbers of seedlings (m^{-2}) and species germinated. Data are also presented on the numbers (and percentage of the total) of seedlings and species germinated from chilled and non-chilled samples.

	Germinating seedlings				Species		
Field	**Total**	**$m^{-2} \pm$ S.E.**	**Autumn (%)**	**Spring (%)**	**Total**	**Autumn (%)**	**Spring (%)**
SA123	3644	19852 ± 1654	2330 (64)	1314 (36)	51	41 (80)	41 (80)
LH	5837	50497 ± 3977	1501 (26)	4336 (74)	47	35 (74)	37 (79)
IE	751	15978 ± 2922	333 (44)	418 (56)	27	20 (74)	19 (70)
IW	829	13194 ± 1454	304 (37)	525 (63)	36	29 (81)	27 (75)
PL	3483	35074 ± 3066	1162 (33)	2321 (67)	45	36 (80)	35 (78)
REV	1388	10027 ± 703	742 (53)	646 (47)	44	37 (84)	30 (68)
Total			6372 (40)	9560 (60)	121	71 (59)	67 (55)

In all fields, a relatively small proportion of species accounted for the majority of seedlings. Many species were recorded from the seed banks of all fields at very low seed densities. For example, 11% of species recorded from the seed bank of LH were represented by a single seedling only, compared to 41% from IE, 30% from REV, 27%, 22% and 22% from PL, IW and SA123 respectively.

There is poor correspondence between seed banks and vegetation in terms of the species complement. In general, a higher proportion of species from the seed bank occurs in the vegetation than vice versa. The unimproved grasslands (most species-rich) show the most marked differences: 68% of species from LH seed bank occurring in the vegetation, but only 44% of species present above-ground occur in the seed bank.

Similarly, 58% of species in the PL seed bank occur in the vegetation, with only 41% of aboveground species contributing to the seed bank. The number of species in common increases within the more improved fields. For example, 36% of REV seed bank species occur aboveground, with 40% of vegetative species occurring as seed.

For all fields, with the exception of REV, a greater number of species were recorded in the vegetation than in the associated seed bank.

4.3.2 Seed bank species richness

Table 4.2. Mean numbers of species emerging per sampling point (soil surface area 25.13cm^2 for all grasslands; 62.83cm^2 for ex-arable field).

Field	Total	Chilled	Non-chilled
LH	10.93 ± 0.40	8.92 ± 0.38	6.49 ± 0.17
PL	10.02 ± 0.26	7.36 ± 0.11	6.99 ± 0.57
IE	5.53 ± 0.65	3.80 ± 0.07	3.42 ± 0.67
IW	8.46 ± 0.46	5.86 ± 0.47	4.80 ± 0.11
REV	5.64 ± 0.39	3.06 ± 0.22	3.93 ± 0.25
P value	<0.001	<0.001	<0.001
F statistic	36.73	84.25	16.49
SA123	*14.03 ± 0.61*	*8.57 ± 0.51*	*10.70 ± 0.49*

4.3.2.1 Differences between fields

There were significant differences between grassland seed banks in terms of mean number of species per sampling point, with LH, PL and IW seed banks containing a greater number of species than the IE or REV fields (Table 4.2).

4.3.2.2 Differences between chilled, non-chilled and pooled samples

There were significant differences ($P<0.01$; F 7.07; df 2) between the mean numbers of species emerging, with the mean total number of species emerging from each seed bank generally being greater than the means recorded for either half of the sample (chilled or non-chilled) (Table 4.2). However, for LH, the total mean number of species and the chilled mean are greater than the number emerged from the non-chilled samples. For both REV and SA123, the mean number emerging from the non-chilled samples is higher than that emerging from the chilled samples.

4.3.2.3 Mean numbers of seeds of grassland and 'weed' species m^{-2}.

There were significant differences in the numbers of total seedlings (and grassland species seedlings) germinated from the seed banks, with a greater number recorded from the PL and LH meadows than from any other field (Table 4.3) and LH was also richer than the PL seed bank in terms of total seedlings and grassland species seedlings. There were significant differences between fields in terms of weed seeds m^{-2}, with the SA123 seed bank containing a higher number of weed seeds per unit area than any of the grassland fields.

Table 4.3. Mean number of seeds germinated: total, grassland and 'weed' species (m^{-2})

Field	mean seeds	Grassland species	Weed species
IW	13194 ± 1454	11937 ± 1413	1257 ± 396.3
IE	15978 ± 2922	14366 ± 2613	1612 ± 880
PL	35074 ± 3066	34417 ± 3074	656.5 ± 102.3
REV	10027 ± 703	8826 ± 722.9	1201 ± 145.1
LH	50497 ± 3977	50177 ± 3990	320 ± 75.86
SA123	19852 ± 1654	7894 ± 955.4	11958 ± 1307
Significance	P<0.001	P<0.001	P<0.001
F statistic	39.24	48.03	53.89

4.3.2.4 Proportion of seed bank contributed by grasses, forbs, sedges/rushes

Table 4.4. The absolute proportion of the seed bank and the vegetation accounted for by grass, forb, and other (sedge/rush) species (presented as numbers of seedlings as a percentage of total germination, and percentage ground cover with bryophyte, litter and bare ground removed).

Field	seedlings m^{-2} (%)			vegetation (%)		
	grass	forb	sedge/rush	grass	forb	sedge/rush
IE	88.47	8.65	2.88	86.65	13.35	-
IW	69.52	22.29	8.19	75.71	25.29	-
REV	80.9	13.48	5.62	73.8	26.28	-
PL	82.43	15.14	2.43	75.76	18.36	5.88
LH	66.58	11.99	21.43	56.31	23.67	20.02
SA123	80.33	17.34	2.32	26.79	71.43	1.79

Although grass seedlings contributed 88% of the total seed bank of IE (the highest of any field), the seed banks of PL and LH actually contained higher numbers of grass seeds m^{-2} than any other field ($P<0.001$; F 25.84; df 5) (Table 4.4). Similarly, PL and

LH contained higher numbers of forb seeds than IE, IW or REV ($P<0.001$; F 13.90), and LH contained a greater number than SA123 seed bank also. The LH seed bank contained significantly greater numbers of seeds of sedges and rushes than any other field ($P<0.001$; F 26.96).

4.3.3 Individual species results – differences between fields

Table 4.5. Individual species: differences in mean numbers of seeds m^{-2} between fields

Species	P	F	Tukey's HSD
Agrostis sp.	<0.001	8.17	LH>REV; PL,LH>SA123
Alopecurus pratensis	<0.001	13.21	PL>IW, REV; all>SA123
Anthoxanthum odoratum	<0.001	22.22	LH>IE, IW, PL, EX-A; all>SA123
Cardamine pratensis	≤0.001	8.11	PL, LH > IW
Holcus lanatus	<0.001	50.24	LH>all; PL>IE,IW,REV,SA123; IE,IW>SA123
Juncus conglomeratus	<0.001	29.92	LH>all
Lolium sp.	<0.001	11.29	REV>IW; SA123>all
Poa trivialis	≤0.001	4.28	PL>IW,REV,SA123
Ranunculus sp.	≤0.001	4.50	PL>SA123
All forb seeds	<0.001	13.90	PL,LH>IE,IW,REV; LH>SA123
All grass seeds	<0.001	25.84	PL,LH>IE,IW,REV,SA123
All sedge/rush seeds	<0.001	26.96	LH>all; PL>SA123

There were relatively few significant results for individual species between fields (Table 4.5). With the exception of *Lolium* species, abundance tended to be higher for individual species (and groups of species) within the unimproved fields.

4.3.4 Differences in Ellenberg mean indicator values between fields.

4.3.4.1 Seed bank

The REV seed bank had a higher mean nitrogen value than LH (Table 4.6). The ex-arable field had a significantly higher mean nitrogen value than any other field. There were no significant differences between field seed banks in terms of their mean moisture values.

4.3.4.2 Vegetation

The vegetation of SA123 was not surveyed prior to soil cultivation (see Chapter 4), although a species list was compiled (Appendix 4.3). Thus the mean Ellenberg values derived were not included in the analysis, but are presented for comparison.

Mean moisture: There were significant differences between fields, with LH vegetation being indicative of significantly 'wetter' conditions than that of IE or REV, and PL similarly had a significantly higher mean moisture value than the vegetation of IE.

Mean nitrogen: There were significant differences between fields with REV having a higher mean nitrogen value than LH or PL.

Table 4.6. Mean Ellenberg indicator values (+ S.E.): vegetation and soil seed banks

Sample / **Field**	**mF (moisture)**	**mN (nitrogen)**
Seedbank		
LH	5.89 ± 0.04	5.15 ± 0.12
PL	5.71 ± 0.05	5.48 ± 0.06
IE	5.65 ± 0.03	5.39 ± 0.01
IW	5.94 ± 0.07	5.42 ± 0.08
REV	5.92 ± 0.08	5.67 ± 0.11
SA123	5.90 ± 0.06	6.33 ± 0.08
Significance	*ns*	*P<0.001*
		F 3.96
Vegetation – 1993		
LH	6.36 ± 0.09	4.63 ± 0.09
PL	6.10 ± 0.09	4.89 ± 0.02
IE	5.38 ± 0.05	5.36 ± 0.05
IW	5.85 ± 0.12	5.16 ± 0.07
REV	5.73 ± 0.10	5.81 ± 0.25
[SA123	5.82	6.47]
Significance	*P≤0.001*	*P<0.01*
	F 13.23	*F 7.83*

4.3.5 Spatial effects within SA123 (reversion site)

4.3.5.1 Block effects: mean Ellenberg indicator values and numbers of seeds.

There were no differences between blocks (chapter 5, figure 5.1) in terms of their mean moisture value, although there did appear to be an increase in value towards the less elevated block 1 (Table 4.7). There was however a significant difference between the mean nitrogen values of the different blocks.

In terms of the mean numbers of species, block 1 was richer than block 3, although there is no significant difference between the blocks in terms of the mean number of seedlings emerged.

Table 4.7. Mean Ellenberg moisture (mF) and nitrogen (mN) values, mean number of species and mean number of seeds for the SA123 soil seed bank (± standard error).

Block	**mF**	**mN**	**species**	**seeds**
1	6.03 ± 0.12	6.02 ± 0.12	15.8 ± 0.61	136.9 ± 22.64
2	5.90 ± 0.09	6.49 ± 0.13	14.9 ± 1.06	131.5 ± 14.84
3	5.75 ± 0.09	6.48 ± 0.13	11.8 ± 1.11	105.8 ± 15.83
significance (df 2)	ns	P<0.05 F 4.47	P<0.05 F 4.83	ns

4.3.5.2 DECORANA: Differences between individual plots

There is no clear pattern within the soil of the ex-arable field (Figures 4.1 and 4.2). The lengths of the first two axes are relatively short, and thus plots at either end are not dissimilar. There is no clustering of adjacent plots.

4.3.6 Comparison with above-ground vegetation

For the majority of species there is no rank correlation between abundance in the seed bank and abundance in the vegetation or indeed in the vegetation between years. For those species where there is a rank correlation (Table 4.8), the strongest relationship appears to be between abundance in the vegetation between years. Both persistent species (*A.odoratum*, *C.pratensis*) and transients (*L.perenne*, *R.acetosa*) show a relationship between their contribution to the seed bank and aboveground flora. Where seed bank and vegetation correlate, abundance in the vegetation between years is largely correlated also.

Figure 4.1 Sample ordination. DCA of experimental plots by species

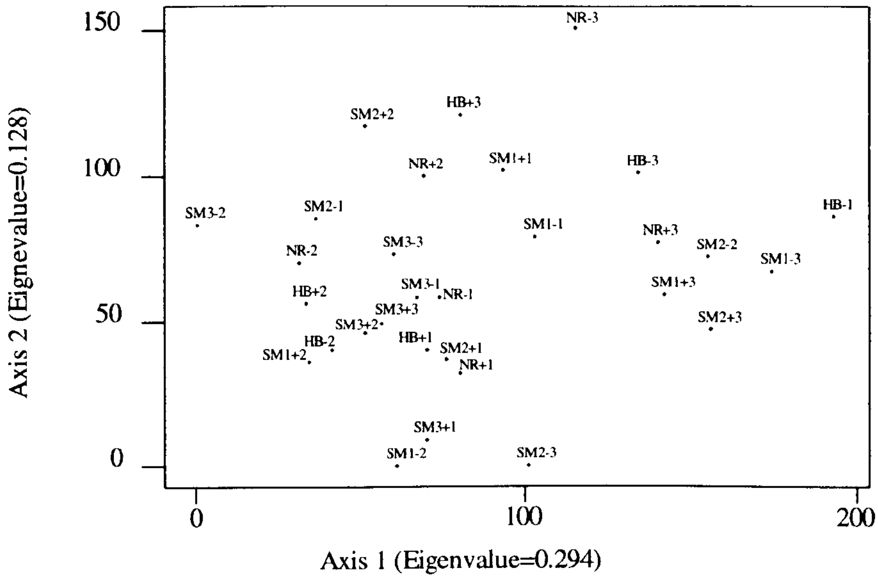
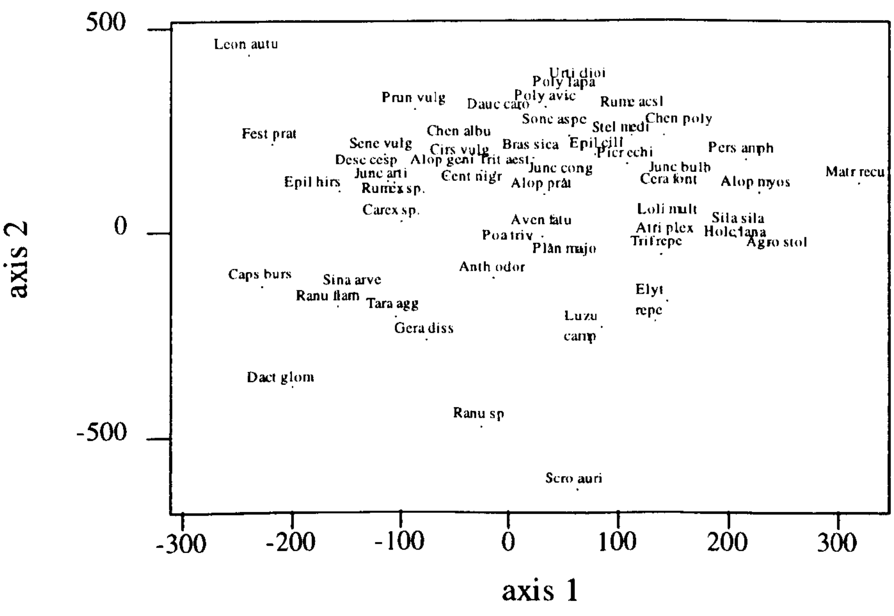


Figure 4.2 Species ordination. DCA of seed bank species by experimental plots



4.3.6.1 Rank correlations.

Table 4.8. Significant rank correlations between abundance in the seed bank and vegetation in 1993 (sb-93), seed bank and vegetation in 1996 (sb-96) and between years in the vegetation (93-96).

	Values of r_s			P values		
Species	**sb-93**	**sb-96**	**93-96**	**sb-93**	**sb-96**	**93-96**
Anthoxanthum odoratum	0.937	0.627	0.667	0.05	1	1
Bromus hordeaceus	0.805	-	0.755	0.05	ns	0.5
Cardamine pratensis	0.834	0.845	0.904	0.05	0.05	0.05
Carex panicea	-	-	0.933	ns	ns	0.05
Carex riparia	0.746	0.814	0.962	0.5	0.05	0.05
Centaurea nigra	0.574	0.665	0.844	ns	1	0.05
Dactylis glomerata	-	-	0.845	ns	ns	0.05
Deschampsia cespitosa	-	-	0.871	ns	ns	0.05
Festuca rubra	-	-	0.833	ns	ns	0.05
Holcus lanatus	-	0.867	0.652	ns	0.05	1
Lolium perenne	0.912	0.859	0.890	0.05	0.05	0.05
Rumex acetosa	0.875	0.746	0.705	0.05	0.5	0.5
Taraxacum agg.	-	-	0.911	ns	ns	0.05
Trifolium repens	-	-	0.803	ns	ns	0.05

4.3.6.2 Sample ordination: River Ray corridor grasslands (vegetation and seed bank)

The ordination clearly shows that the ex-arable seed bank is separated on the basis of the suite of arable weeds contained, and also that the grassland seed banks are somewhat weedier than the corresponding above-ground vegetation (figures 4.3 to 4.5). None of the seed banks are similar to the above-ground vegetation, although the above-ground vegetation between years is also variable.

The first and second axes explain the majority of the variation (respective Eigenvalues of 0.555 and 0.367). The first axis appears to be related to 'weediness', with the ex-arable seed bank (SA123sb) at the right-hand side, and the unimproved vegetation on the left-hand side (i.e. LH96, PL96). The length of the first axis is approximately 4 standard deviations, indicating that the unimproved fields and the ex-arable seed bank are unlikely to have many species in common. Indeed, the species scores show a trend from arable weed species to species more typical of improved grassland to unimproved grassland species (e.g. *Succisa pratensis*, *Cirsium dissectum*, *Lysimachia nummularia*).

Figure 4.3 Sample ordination – River Ray corridor grasslands: vegetation and seed bank. IE: Improved East; IW: Improved West; PL: Plantlife Meadow; LH: Long Herdon SSSI; REV: Reverting Field; SA123: ex-arable study field. Suffix '93' and '96' refer to year of botanical survey. Suffix 'sb' refers to seed bank samples.

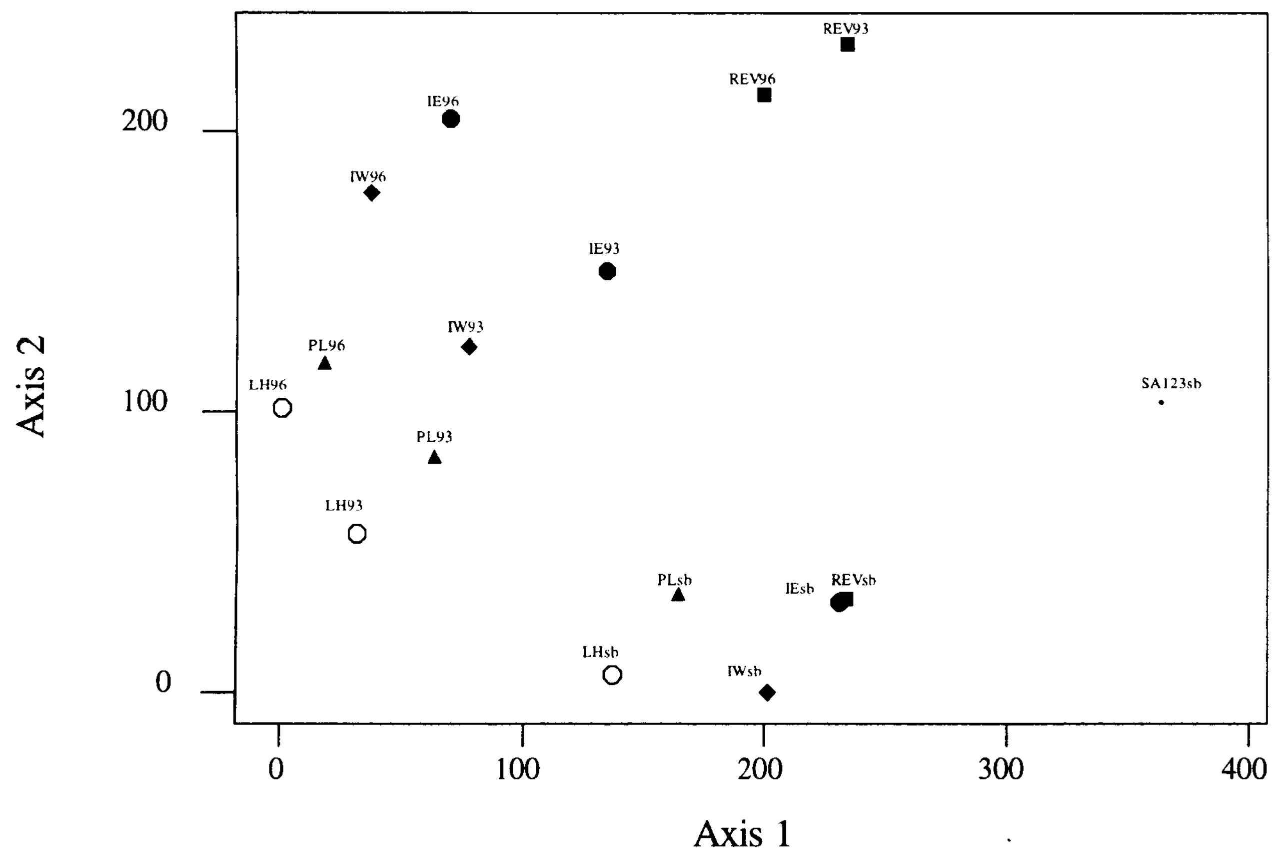


Figure 4.4 Species ordination: River Ray corridor grasslands: comparison of vegetation and seed bank

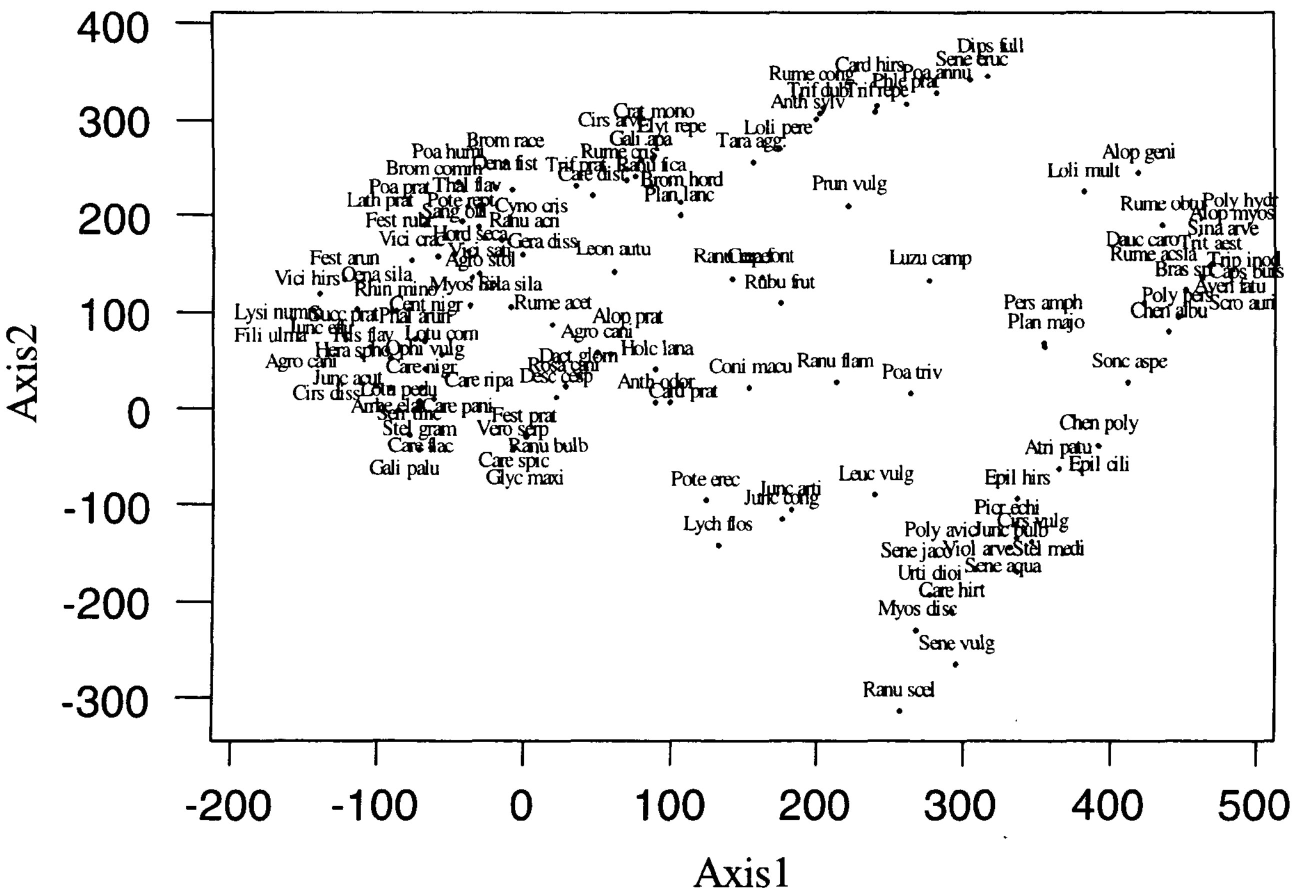
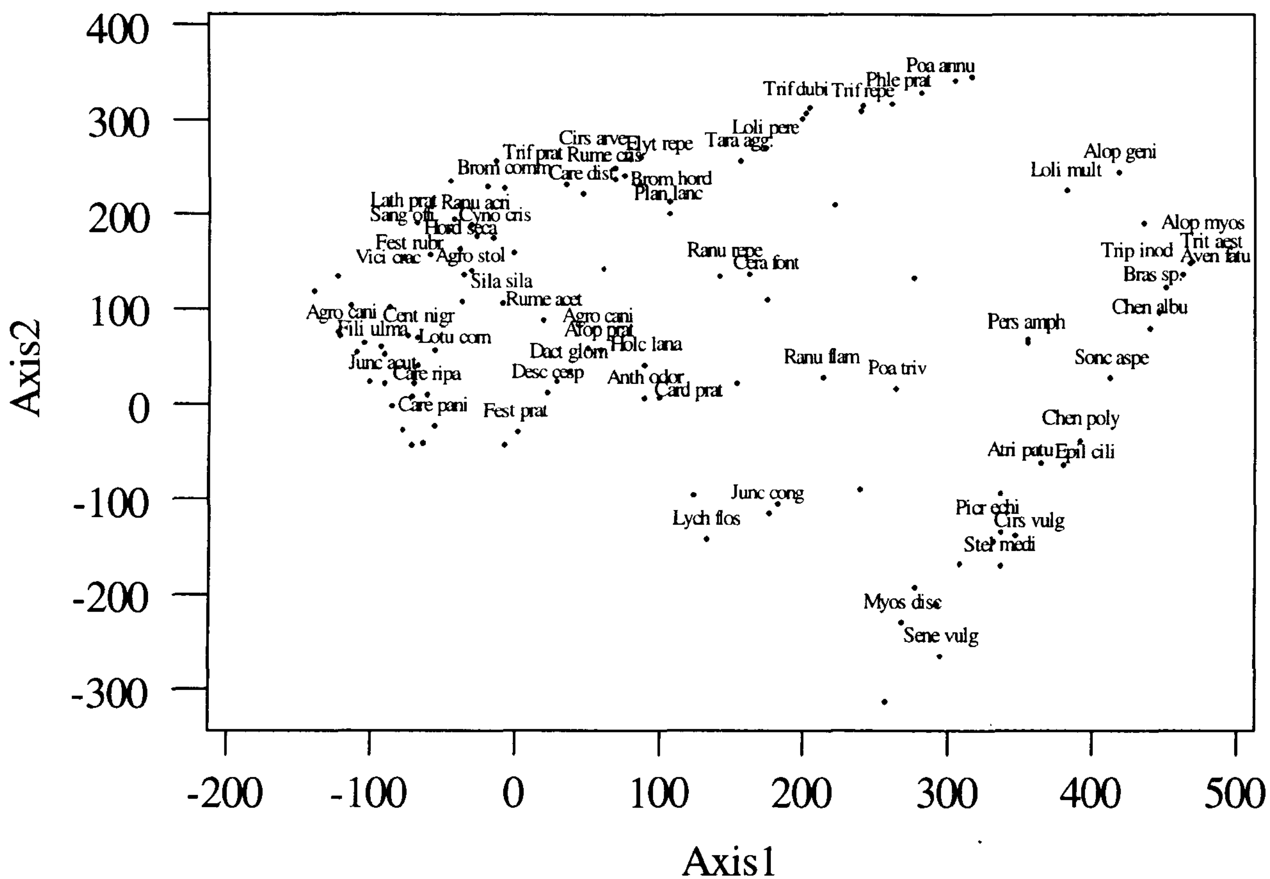


Figure 4.5 Species ordination. River Ray corridor grasslands: comparison of vegetation and seed bank. Species are named only if they occur in greater than 20% of samples within any one field



4.3.6.3 Numerical assessment of similarity

Sorensen Community Coefficients (table 4.9) reveal that seed banks are indeed more similar to one another than to the associated aboveground vegetation. The seed bank of SA123 is most similar to the seed bank of REV, and is as similar to the vegetation of REV as is the seed bank of that field. Calculated Euclidean Distances (table 4.10) suggest that the seed banks of PL and IW are most similar, followed by REV and IE. The least similar seed banks are LH-IE, LH-REV, then LH-SA123.

Table 4.9 Sorensen's Community Coefficient comparing the floristic composition of seed banks and plant communities of each field.

Field	*PL*	*IE*	*IW*	*REV*	*LH*
Vegetation93/vegetation93					
PL	-	66.67	73.58	46.60	82.35
IE	-	-	78.48	60.53	81.01
IW	-	-	-	57.83	67.24
REV	-	-	-	-	51.35
Seed bank with each plant community					
	46.30	47.62	20.25	33.33	53.33
Vegetation96/vegetation96					
PL	-	76.92	77.78	54.55	80.60
IE	-	-	83.33	64.37	65.57
IW	-	-	-	68.13	68.25
REV	-	-	-	-	56.41
Seed bank with each plant community					
	46.60	52.83	46.51	37.65	53.66
Seed bank/seed bank					
SA123	58.33	53.85	57.47	69.47	59.18
PL	-	61.11	74.07	67.43	78.26
IE	-	-	63.49	53.52	54.05
IW	-	-	-	65.00	69.88
REV	-	-	-	-	61.54

Sorensen's Community Coefficient: Vegetation 1993 and 1996

1993	*PL*	*IE*	*IW*	*REV*	*LH*
1996					
PL	85.95	70.21	73.27	51.02	77.86
IE	71.56	85.37	80.90	58.14	65.55
IW	74.34	76.74	88.17	64.44	68.29
REV	55.77	70.13	64.29	66.67	52.63
SSSI	79.14	60.71	63.87	55.17	89.93

Sorensen Community Coefficent: Ex-arable seed bank and vegetation samples

	PL	*IE*	*IW*	*REV*	*LH*
1993	29.82	34.48	29.79	39.56	35.48
1996	31.19	30.93	33.66	34.78	33.07

Table 4.10 Euclidean distance, comparing the floristic composition of seed banks and plant communities of each field.

Field	*PL*	*IE*	*IW*	*REV*	*LH*
Vegetation93/vegetation93					
PL	-	21.97	13.18	41.84	14.64
IE	-	-	15.76	34.26	26.31
IW	-	-	-	39.19	16.86
REV	-	-	-	-	44.77
Seed bank with each plant community					
	30.86	54.11	38.57	50.53	28.49
Vegetation96/vegetation96					
PL	-	37.34	22.56	47.54	15.24
IE	-	-	30.58	42.82	36.47
IW	-	-	-	38.48	19.83
REV	-	-	-	-	46.67
Seed bank with each plant community					
	44.53	68.93	46.66	46.72	38.24
Seed bank/seed bank					
SA123	40.46	46.64	37.22	43.64	49.52
PL	-	31.31	16.52	30.11	32.81
IE	-	-	24.08	19.78	58.69
IW	-	-	-	23.11	38.28
REV	-	-	-	-	55.83

Euclidean distance: Vegetation 1993 and 1996

1993	*PL*	*IE*	*IW*	*REV*	*LH*
1996					
PL	17.86	32.98	23.88	46.79	23.58
IE	36.67	33.59	34.58	43.55	37.97
IW	24.15	27.67	21.03	38.78	25.20
REV	42.12	33.95	39.37	20.62	46.25
SSSI	16.25	29.50	19.53	45.37	15.06

Euclidean distance: Ex-arable seed bank and vegetation samples

	PL	*IE*	*IW*	*REV*	*LH*
1993	39.69	38.38	41.69	44.82	42.94
1996	46.24	51.11	45.04	48.71	42.94

4.4 Discussion

This experiment has shown that seed banks are variable between different fields and habitats. This variation should always be considered when assessing fields for suitability for restoration. The following sections discuss the difficulties associated with the measurement of seed banks, the variation between seed banks, why this variation occurs, and the consequences this has for the restoration of lowland wet grassland.

4.4.1 Methodology

4.4.1.1 Timing of sampling

Many authors (e.g. Donelan & Thompson 1980; Kirkham & Kent 1997) have stated that in order to determine the persistent seed bank as opposed to the persistent and transient seed bank, soil sampling should occur early in spring and certainly prior to seed shed in the current year. However, in this investigation one objective behind sampling of the ex-arable field was to be able to attribute a likely source to species recorded in the developing vegetation and so the determination of the transient seed bank was considered unimportant. For this study, the appropriate time for sampling was immediately after seed bed preparation prior to the re-introduction of seed when all seeds that may become part of the vegetation would be present. The grasslands were sampled at the same time (September) despite the fact that the contribution of certain species to the seed bank may be overestimated, i.e. transient species. Therefore, the time of sampling in this study may have resulted in an overestimation of the size of the persistent seed bank of a number of species, particularly the grasses. Williams (1984) sampled the seed bank beneath long-term pasture at intervals, and found that when the vegetation was not cut until September, the size of the seed bank of species other than *Juncus* was more than doubled by October. In particular, the number of *Holcus lanatus* seeds was considerably higher in October where maximum seeding was permitted in September. A similar trend was observed for *Poa trivialis*.

The vegetation within all grasslands in this study was cut for hay in the first week of July in 1993, with soil samples taken in early September, and so it is unlikely that maximum seeding occurred. Moreover, much grass seed is retained within hay bales and subsequently ripens within the bale; the majority of seed that drops from the hay before incorporation into bales is from forb species (Smith *et al.*, 1996). Although some freshly shed seed will have germinated in the autumn of 1993, it is difficult to quantify how much the size of the persistent seed bank will have been overestimated by fresh input of seed.

4.4.1.2 Size of samples taken

Due to the large spatial variation in seed banks, it is recommended that a larger number of small samples be taken rather than a small number of large samples (Roberts 1981). Numata *et al.* (1964) and Hayashi and Numata (1971) recommended a minimum soil volume of 400-600cm^3 to accurately estimate the seed banks of arable fields or grasslands. In this study, a total soil volume of 9425 cm^3 was removed from 150 sampling points within the ex-arable field (SA123), 5780 cm^3 from 46 sampling points within the SSSI (LH), 4901cm^3 from 39 points within the Plantlife (PL) meadow, 3142 cm^3 from 25 points in Improved West (IW), 2388 cm^3 from 19 points in Improved East (IE), 6911 cm^3 from 55 points within the Reverting field (REV). These would appear to be adequate soil volumes to characterise the seed banks of these fields. Different volumes of soil were taken from each field, as the sampling locations were those used during the earlier botanical survey. The number of quadrats recorded per field varied according to the size of the field.

4.4.1.3 Duration of study

Warr *et al.* (1993) state that, if the aim is to estimate *all* viable seeds within the soil, emergence monitoring should continue for at least two years. The emergence of seedlings from the collected samples was therefore followed for an extended period of two years, although many studies reported in the literature have monitored seedling emergence for shorter durations. Thompson and Grime (1979) reported negligible germination after 36 days. McDonald *et al.* (1996), sampling similar habitats to those investigated within the current study, monitored samples for 8 weeks until no more seedlings emerged; Pakeman and Hay (1996) maintained heathland seed bank samples for 6 months; and Froud-Williams *et al.* (1983) found that the majority of seeds emerged during the first year of their two-year study.

In this study, emergence of 95% of species and seedlings from the non-chilled samples occurred within 17 months (although 95% of seedlings had emerged within 10 months). In contrast, 95% of species and 95% of all seedlings appeared within 8 months in the pre-chilled samples.

If this study had been carried out over the shorter time-scales of other studies (e.g. Thompson & Grime, 1979; McDonald et al., 1996; Pakeman & Hay, 1996) then the seed bank would have been underestimated.

One criticism of seed bank investigations is the large amount of glasshouse space and time required for the experiment. Many studies may be constrained by the resources required. Certainly, effort has been targeted at reducing the time required to achieve (and improving) emergence (Ter Heerdt *et al.*, 1999).

4.4.1.4 Germination method (Chilling versus non-chilling)

In this study, numbers of species recorded were similar between chilled and non-chilled samples. Gross (1990) reported that germination with cold-stratification revealed a greater number of species than either direct germination or washing using an elutriation system, whilst Dickie *et al.* (1988) found a slight, but significant, reduction in total seedling emergence due to stratification. In terms of the number of species emerging per sampling point from chilled and non-chilled samples, it was only in LH that significantly more species emerge following chilling. Only REV and SA123 seed banks had more species emerging from the non-chilled samples. These differences appear to be largely random however. Information on germination requirements is not available for all species, but current knowledge does not suggest that LH contains a higher proportion of species with a requirement for chilling than the more improved fields.

McDonald *et al.* (1996) simulated the stratification of seeds during the winter period by chilling half their samples at 5°C for 15 days, and then assessed the seed bank using the seedling emergence method. They did not report whether or not a significantly greater number of seedlings emerged before or after stratification. However, they did find that more seeds of individual species tended to germinate after stratification and that certain species only germinated following stratification (e.g. *Cardamine pratensis*, *Plantago major*, *Rhinanthus minor*, *Rorippa palustris* and *Silaum silaus*) and so discounted the results from the samples that were not subject to stratification. However, in this study these findings were not verified: *C.pratensis* germinated from both the non-chilled and chilled samples, the majority of seedlings of *P.major* germinated from the non-chilled samples, and seedlings of *S.silaus* also emerged from both samples, although the majority did germinate following stratification. One possible explanation for the differences between the present study and that of McDonald *et al.* (1996) could be that the individuals that emerged from the non-chilled samples in this study had already been chilled in previous years whilst they were in the field and therefore did not require further chilling to break dormancy.

Those species that only occurred in one set of samples (chilled or not chilled) tend to occur at low seed densities and so in terms of total seed density these species add little information to the study, but in terms of species composition the results of both chilled and non-chilled samples must be considered.

It does appear that cold stratification increases the numbers of seedlings emerging in total by vernalising seeds, but perhaps more importantly it acts to encourage earlier germination.

4.4.2 Number of seeds

Total seed density: The greatest densities of seeds are generally found in soils subject to disturbance, i.e. arable fields, where species tend to have short generation times, rely on regeneration from seed and produce large numbers of seeds (Hodgson & Grime, 1990). Thompson (1978) noted that as succession progressed towards maturity of vegetation, the size of the seed bank decreased and thus we might expect low seed bank densities from all grassland fields. The results from this study (table 4.1) suggest otherwise, with the greatest number of seeds present in fields subject to least disturbance, i.e. LH and PL meadows. An explanation of this result is that application of nitrogenous fertiliser to grasslands is known to stimulate seed germination (Williams, 1983; Pons, 1989), effectively depleting reserves of viable seeds in the soil (Kirkham and Kent, 1997). This would certainly help explain why the two most intensively improved grasslands fields (Improved East and Reverting) have the smallest seed banks.

This study could have has underestimated the size of the ex-arable field seed bank. Whilst it is generally accepted that the majority of seed within (less disturbed) soil will be present within the surface 5cm (Thompson and Grime, 1979; Roberts, 1981), this is not the case for arable soils where a sampling depth of up to 25cm may be more appropriate. As soil cultivation buries seeds more deeply, it is possible that germinable seeds exist at greater depths in arable soil and could contribute to the developing above-ground vegetation. It was assumed that the seed within the top 5cm was a representative sample of the total buried seed population rather than being a sample that represented 95% of seed.

McDonald *et al.* (1996) recovered 11 333 seeds m^{-2} from cattle-grazed Somerford Mead and 3 376 seedsm^{-2} in 0-10cm soil layer from cut and grazed Oxey Mead (both found beside the River Thames, north of Oxford, UK). This is low in comparison to the results of this, and other, studies (e.g. Jensen 1998), particularly when you consider that McDonald *et al.* based their calculations on only the results of samples that were subject to cold stratification, and that they (and other workers) have found the number of seedlings emerging to be greatly increased by chilling. Moreover, McDonald *et al.* sampled the seed bank during November and thus their results will have been influenced by fresh input of seed from the summer just passed. If they had sampled during the late spring, then seeds recovered might have been only those that are truly persistent. The number of seeds recovered should have been augmented by the transient seed bank during autumn sampling (when seed numbers would presumably be at their maximum), but instead the total number of seeds apparently recovered was low. This could be due to the relatively short period spent monitoring seedling emergence during their study (8 weeks); the numbers of seeds recovered during the present study would probably have been considerably lower had emergence been monitored for 8 weeks only (this is certainly true of SA123: Manchester and Sparks, 1998). An alternative explanation for the low numbers of seeds recovered by McDonald *et al.* (1996) could be that large reserves of seed had not accumulated in the soils sampled, possibly because the management regime had either prevented species from setting seed or seed from falling to the ground or being incorporated into the soil.

4.4.3 Species-richness

The seed banks of the two less improved grasslands (PL, LH) were more species-rich per unit area than the two more improved grasslands (IE, IW). In part this may be due to the relative species-poor nature of the vegetation within the improved fields (above-ground total of 35 and 40 species respectively). Whilst the Improved East (IE) seed bank contained fewer species in total than any other, the Reverting (REV) seed bank contained a similar number of species in total to that of Plantlife (PL) and the SSSI (LH), and more than either improved field. This suggests that certainly within the Reverting field seed bank, species distributions must be such that many species are present at low densities and are infrequent.

Emerging seedlings can be coarsely classified as either grassland or weed species. Using this classification, the two least improved fields (LH, PL) contained a significantly greater number of seeds of grassland species than the other fields. The ex-arable (SA123) seed bank was nearly as rich in seeds of grassland species as the Reverting field (REV) and was not significantly dissimilar from those of the improved fields (IE, IW). However, the number of weed seeds was significantly higher in the ex-arable seed bank than in the grassland field seed banks. This suggests that the ex-arable seed bank has a similar potential to contribute to grassland establishment as do the seed banks of the more improved fields following widespread soil disturbance but that the large numbers of weed seeds in the ex-arable seed bank could detrimentally affect grassland establishment. If the establishing weed species could be prevented from returning seed to the soil by topping, whilst the slower-growing perennials establish, then once the developing sward has closed, ruderals will be outcompeted and unable to establish (Fenner and Spellerberg, 1988).

4.4.4 Composition

The ex-arable (SA123) seed bank was dominated by *Poa trivialis*, *Lolium multiflorum* and *Alopecurus myosuroides*, contributing 67% of recorded seedlings. Of the species present, 28 were weedy and contributed 11931 seeds m^{-2}; 23 could be considered characteristic of wet grasslands, contributing 7401 seeds m^{-2}. This suggests that initially high flushes of ruderals are to be expected in the field, but that provided propagules of perennials are available to colonise the site, these weed species should not persist long-term. The presence of such species within the seed bank of grassland should not necessarily be cause for concern

The seed bank of the reseeded Reverting (REV) field is more similar to that of the ex-arable field (SA123) than any of the other fields. Indeed, the ex-arable field contains a similar quantity of seeds of grassland species to the seed banks of the more improved grasslands, which suggests that arable seed banks are a limited source of propagules. However, the majority of fields do not contain the species necessary to recreate species-rich grassland within their seed banks.

4.4.5 Arable weed seeds in the SSSI

Species that persist in disturbed habitats (e.g. arable land) generally have mechanisms for dispersal in time and space (Hodgson & Grime, 1990). Species with long-lived seeds tend to be those associated with unpredictable habitats (Grime et al., 1981; Roberts, 1986). Indeed, Chippindale & Milton (1934) found seed of species characteristic of arable land present in soils beneath pastures that had not been ploughed for 68 years. Some of this seed may not have persisted in the soil, instead being dispersed from adjacent arable land and then incorporated into the seed bank, but this seems unlikely for large numbers of seeds considering the decrease in numbers deposited with distance from the parent plant (Jefferson & Usher, 1989; Collins & Glenn, 1990). Within this study, a number of arable weed seedlings (*Atriplex patula, Avena fatua, Cirsium vulgare, Epilobium hirsutum, Plantago major, Sonchus asper, Urtica dioica*) germinated at low densities from the seed bank of the SSSI. Whilst many of these seeds can persist for decades, the continued presence of arable fields in proximity to the nature reserve makes it possible that seeds have dispersed more recently and been incorporated into the seed bank. Although ruderal species will not persist in established grassland with a closed canopy, and hence few gaps for germination and establishment, (Fenner, 1978), the presence of a large number of weed seeds could play a more significant role if the above-ground vegetation of the nature reserve were damaged. Thus even if there was no clear benefit to be derived from selecting this particular arable field rather than another in the catchment for restoration, the creation of grassland on the adjacent ex-arable site would be desirable to buffer the SSSI from further ingress of arable weed seeds.

4.4.6 The contribution of grass species to seed banks

Within this study, high numbers of seeds of grass species were found within the seed banks of grasslands (table 4.5). Roberts (1981) commented that seeds of grasses often do form an appreciable percentage of the total number of seeds in soil, but other workers have not found this to be true. Chippendale and Milton (1934) found very low numbers of grasses in the seed bank and attributed this to dependence in most grass species upon vegetative reproduction, with the exceptions of *Holcus lanatus*, *Poa trivialis*, *P.annua* and *Agrostis* sp.

Roberts (1986) reported that seeds of many species of grassland remain on, or near to, the surface after dispersal and germinate as soon as their requirements for moisture and temperature are met. Such species frequently form only a transient seed bank (Thompson and Grime 1979) with no carry over from one year to the next. However, certain species with high initial levels of germination following seed shed (e.g. *Agrostis capillaris*, *Deschampsia cespitosa*, *Holcus lanatus*, *Poa annua*, *Poa trivialis*) have a small proportion of seeds that do not germinate immediately and may remain alive in the soil for extended periods (Grime *et al.*, 1981). Indeed, a high number of seeds of *Poa trivialis* were recovered from the uppermost 5cm of the ex-arable field soil, even following soil cultivation. Since soil cultivation acts to bury seeds, reserves of *Poa* seeds must be reasonably high through the soil profile of this field.

4.4.7 Correspondence between seed bank and aboveground vegetation within grassland fields

For many communities there is very little correspondence between the composition of the soil seed bank and the aboveground vegetation (Chippindale & Milton, 1934; Champness & Morris, 1948). Greater similarity is generally observed between the seed bank and the vegetation in frequently disturbed habitats (Warr *et al.*, 1993), whilst, with increasing maturity of vegetation, this similarity declines.

The results of the current study support a lack of similarity between vegetation and seed bank. In addition to reasons already suggested in the literature for this lack of correspondence, there are a number of other possible explanations. Seeds that require light for germination may not have germinated in the field, but germinated in the glasshouse when soil samples were spread thinly. Similarly, some species may germinate in the field from greater depths than were sampled. Such species would thus not have been identified within the current study.

Ellenberg indicator values

As expected, values for mean Nitrogen availability were significantly higher within the ex-arable (SA123) reversion site than for any other field. This reflects the larger contribution of competitive arable weed species. However, mean Ellenberg nitrogen values were generally higher for the seedlings emerging from the seed bank than in the aboveground flora even for the grasslands studied. Bekker *et al.* (1997) found that seed banks were generally dominated by species indicative of mid-range nutrient conditions. Kirkham and Kent (1997) found that after 5 years of fertiliser use on hay meadows, the balance of species in the seed bank changed in favour of those that were both more competitive under fertile conditions and less persistent in the soil. These changes could prolong dominance of these species in the vegetation after fertilizer use has been discontinued.

Behaviour of species germinating from the seed banks

<u>Type 1 seed banks</u>

Seeds of these species tend to germinate mostly in the autumn following seed shed and do not form persistent seed banks. Within this study, several Type 1 species (*Cynosurus cristatus, Festuca pratensis, Lolium perenne, Phleum pratense, Cirsium vulgare*) did conform to this behaviour, with the majority of seed germinating from the samples that were grown on immediately in the autumn following collection. A lesser number of seedlings of these species did germinate from the spring-germinating samples, suggesting that even within these species there is the capacity for some over-wintering. A number of other Type 1 species behaved differently. For example, more seeds of *Alopecurus pratensis* and *Taraxacum* agg. emerged from the spring-germinating samples, whilst roughly equal numbers of seed of *Rumex acetosa* germinated from both samples.

Type II seed banks

Type II seed banks are also transient, and thus species in this category would be expected to behave similarly to species with a Type I seed bank. Grime *et al.* (1988) suggest that both *Alopecurus geniculatus* and *Atriplex patula* have either a Type II or Type IV seed bank. All germination of seeds of these species occurred within the autumn-germinating samples, suggesting that in fact Type II seed banks are more likely. All seed of *Conium maculatum* germinated in autumn, but there were only two seedlings.

Type III seed banks

These are persistent seed banks, intermediate between Type II and Type IV. Some seed germinates immediately following shedding, but much incorporated into a persistent seed bank. Some of the most widespread, successful species (e.g. *Holcus lanatus* and *Poa trivialis*) have Type III seed banks, enabling rapid population expansion and persistence (Grime *et al.*, 1988).

In general, species with Type III and Type IV seed banks are much more common in seed banks than those with transient seed banks.

Type IV seed banks

These seed banks are truly persistent, enabling regeneration of species from seed in situations where disturbance of vegetation is temporally unpredictable. Many arable weed species recorded within this study have Type IV seed banks, e.g. *Capsella bursa-pastoris*, *Chenopodium album*, *Plantago major*, *Polygonum* sp., *Rumex* species, *Sinapis arvensis*, *Stellaria media* and *Urtica dioica*. Seeds of these species are adapted to germinate whenever they find themselves in conditions suitable for germination.

In general, germination patterns did reflect what would be expected by categorising according to seedbank type. However, the relatively artificial conditions of the glasshouse may mean that patterns observed are not necessarily those that would be observed in the field.

Observations on individual species

<u>Species only recorded within the seed bank:</u>

All grassland fields: *Atriplex patula*, *Cirsium vulgare*, *Myosotis discolor*, *Stellaria media*, *Urtica dioica*.

Two species (*Avena fatua* and *Potentilla erecta*) occur within the seed bank of the SSSI but in no other grassland (although *A.fatua* occurs within the ex-arable seed bank also). It seems likely that seed of *A.fatua* have dispersed from nearby arable fields and been incorporated into the seed bank of the SSSI. *P.erecta* was only recorded in two fields in the study area and at low densities.

The following species were not recorded in the above-ground vegetation of any of the grasslands, but occur in the seed banks of two or three of the fields: *Alopecurus myosuroides*, *Chenopodium polyspermum*, *Leucanthemum vulgare*, *Conium maculatum*, *Epilobium ciliatum*, *E.hirsutum*, *Senecio vulgaris*, *Brassica* sp.

<u>Species only present within the vegetation:</u>

This is a long list of species, notable for the number of species 'characteristic' of unimproved fields, including *Achillea ptarmica*, *Agrostis canina*, *Cirsium dissectum*, *Lathyrus pratensis*, *Lysimachia nummularia*, *Oenanthe fistulosa* and *O.silaifolia*, *Rhinanthus minor*, *Sanguisorba officinalis*, *Serratula tinctoria*, *Succisa pratensis*, *Thalictrum flavum* and *Trisetum flavescens*. There are a number of potential reasons why these species do not occur within the seed bank. Late successional species do not accumulate large persistent seed banks, and it may be that the seeds of these species do not persist in the soil. However, many of these species are not widespread or abundant within the vegetation (*A.ptarmica*, *L.nummularia*, *O.fistulosa*, *O.silaifolia*, *R.minor*, *S.tinctoria*, *S.pratensis*, *T.flavum*, *T.flavescens*). These species may have restricted dispersal and thus any reserves of seeds within the soil are likely to be highly localised, increasing the chances of being missed by sampling.

<u>Species common to both the seed bank and the vegetation:</u>

A number of species occurred widely within the grasslands, contributing both to the above-ground vegetation and to the seed bank: *Alopecurus pratensis*, *Anthoxanthum* *odoratum*, *Carex riparia*, *Holcus lanatus*, *Lolium perenne*, *Poa trivialis*, *Ranunculus flammula*, *Ranunculus repens* and *Trifolium repens*. These are all relatively widespread and abundant species within the vegetation of these fields.

In addition, there were a number of species common to the seed bank and vegetation for most fields, but not present in one or other for particular fields.

Cynosurus cristatus was present in the vegetation in all fields and seeds were present in all field seed banks with the exception of the Reverting field (REV). This species was considerably less frequent and also less abundant within the Reverting field than within the other fields. *Cynosurus* does not possess a persistent seed bank. Seed recovered is likely to have been only shallowly buried, originating from the current year's vegetation. Again it is possible that there were low densities of seeds localised within the Reverting field that were not recovered by the sampling. *Cardamine pratensis* was present in the vegetation of all fields except Improved East (IE). Seeds were recovered from all seed banks except the IE and REV fields.
Rumex acetosa was present within the vegetation of the SSSI, PL, IW and REV fields in 1993, and was recorded from the seed banks of the SSSI, PL and IW fields also. Although present within the vegetation in the REV field in 1993, no seedlings emerged from the seed bank, and by 1996 this species had disappeared from the vegetation. *R.acetosa* does not possess a persistent seed bank, and thus once the plant has disappeared, regeneration from seed is dependent upon dispersal.

Trifolium pratense was present in the vegetation of all fields, and was also recorded within the seed bank, but no seedlings were recorded from the REV field or from IE.

Two species, *Juncus articulatus* and *J.conglomeratus* occurred quite widely within the seed bank, but were missing from the vegetation within the more improved fields. With the exception of the unimproved fields, sedges and rushes generally appear to be under-represented in the above-ground vegetation of fields sampled. This is a commonly reported phenomenom for species of *Juncus* (although most frequently for *J.bufonius*); large differences between abundance in the seed bank and in the vegetation are indicative of persistent seed banks (Roberts, 1981; Grime *et al.*, 1988). Kirkham and Kent (1997) reported large discrepancies between the abundance of Juncaceae in the seed bank and in the vegetation of hay meadows on a Somerset peat moor. On unfertilised plots, Juncaceae species contributed only 1% towards vegetation cover, but 40% of the seed bank. Such extreme differences between the seed bank and vegetation were not observed during this study. However, *Juncus conglomeratus* was consistently more abundant in the seed bank than in the vegetation and was not necessarily represented in the vegetation despite the presence of viable seeds in the soil.

A further three species, *Picris echioides*, *Plantago major* and *Sonchus asper* tended to occur widely within the seed banks, but only contributed to the vegetation of the Reverting field and then only in 1993 (not 1996).

4.4.8 Utility to habitat restoration

McDonald *et al.* (1996) sampled the seed bank of similar grasslands to those in the current study, and concluded that very few species of the *Alopecurus pratensis-Sanguisorba officinalis* association have long-term persistent seeds. This conclusion was based on samples taken from a flood-meadow that had been fertilized for 20 years. This study suggests that few species of this association form any sort of persistent seed bank. It is not only species characteristic of this association that are missing from the seed bank, but also species characteristic of any of the more species-rich unimproved neutral grassland types.

Both Graham & Hutchings (1988a,b) and Jefferson & Usher (1987) concluded that chalk grassland species were poorly represented in ex-arable seed banks, and that soil disturbance would only encourage undesirable species. The findings of the current study do suggest that this is also true for lowland wet grassland. However, the quantity of seed of grassland species in the seed bank of the ex-arable site was not dissimilar to that found within the more improved grasslands. Thus it is not necessarily only arable seed banks that are lacking in these typical grassland species, but improved grasslands also. The most marked difference lay instead in the huge contribution made to the arable seed bank by ‘weed’ species which may outcompete the desirable component of any vegetation developing.

Bekker *et al.* (1997) studied the seed banks of a number of species-rich grasslands within Europe, and found evidence that the communities studied did not contain many species with a long-term persistent seed bank. They conclude, as do many other authors, that the restoration of species-rich grasslands cannot be achieved from the soil seed bank. Therefore, maintenance of existing species-rich grasslands should be given higher priority.

4.5 Conclusions

The seed bank of the ex-arable reversion site does not contain sufficient seed of suitable species to enable restoration of species-rich wet grassland vegetation solely from *in situ* propagule sources. It is clear that even seed reserves under 'pristine' grasslands are limited and would themselves be unsuitable for restoration of the whole community.

The results of this study stress again the importance of preserving the remaining species-rich grassland resource. Not only are seed banks of ex-arable fields unsuitable for the restoration of a whole community, seed banks of grasslands are generally too dissimilar from the above-ground vegetation, and too species-poor, to be of use in restoration (e.g. Treweek, 1990; Bekker *et al.*, 1997). A number of the species indicative of species-rich swards and unimproved conditions do not appear to form even transient seed banks. Thus if these species are to form part of restored vegetation communities they will need to be (re-) introduced from elsewhere, either through natural dispersal from nearby vegetation or deliberately through intervention. Any diversification of improved grasslands through the extensification offered by agri-environment schemes is likely to be slow and uncertain if it relies on the dispersal and establishment of ex-situ propagules.

CHAPTER 5 RE-ESTABLISHMENT OF LOWLAND WET GRASSLAND ON EX-ARABLE LAND

5.1 Introduction

The investigation of the seed banks of the ex-arable reversion site and the adjacent grasslands (Chapter 4) revealed that seed reserves in the soil sources are too depauperate to result in the restoration of species-rich wet grassland. To facilitate recolonisation of the study site by grassland species, deliberate introduction of propagules is likely to be necessary. A number of methods are available to restore species to a site, utilising either commercial sources or propagules from extant local vegetation. For the purposes of this study, a number of different treatments based on the reintroduction of seed were chosen for comparison.

Natural regeneration/ recolonisation provides a field-based assessment of naturally occurring propagules within the ex-arable field. Whilst the assessment of seed bank composition (chapter 4) suggested that seed of undesirable species dominate within the ex-arable soil, with the cessation of soil cultivation any seeds raining in from adjacent vegetation have the chance to germinate and establish. On the floodplain this rain may be in the form of wind- or bird- dispersed seed, propagules dispersed by floodwater or seed transported by machinery or livestock.

The adjacent species-rich grassland enabled the use of seed of local provenance. Permission for large-scale seed harvesting was not given, but hay baled within the year of experimental establishment was available. The use of *hay bales* from a species-rich meadow was selected as a second experimental treatment.

The remaining experimental treatments focused on the reintroduction of species in the form of *seed mixtures* (see chapter 3, section 3.3.2.3).

5.2 Experimental design and objectives

5.2.1 Experimental site

The study area and the study site (SA123) were described in Chapter 2. The site was in arable use for at least 15 years prior to being set-aside in 1992. The reasons the site was considered suitable for restoration are outlined in Chapter 2 and Manchester *et al.* (1999).

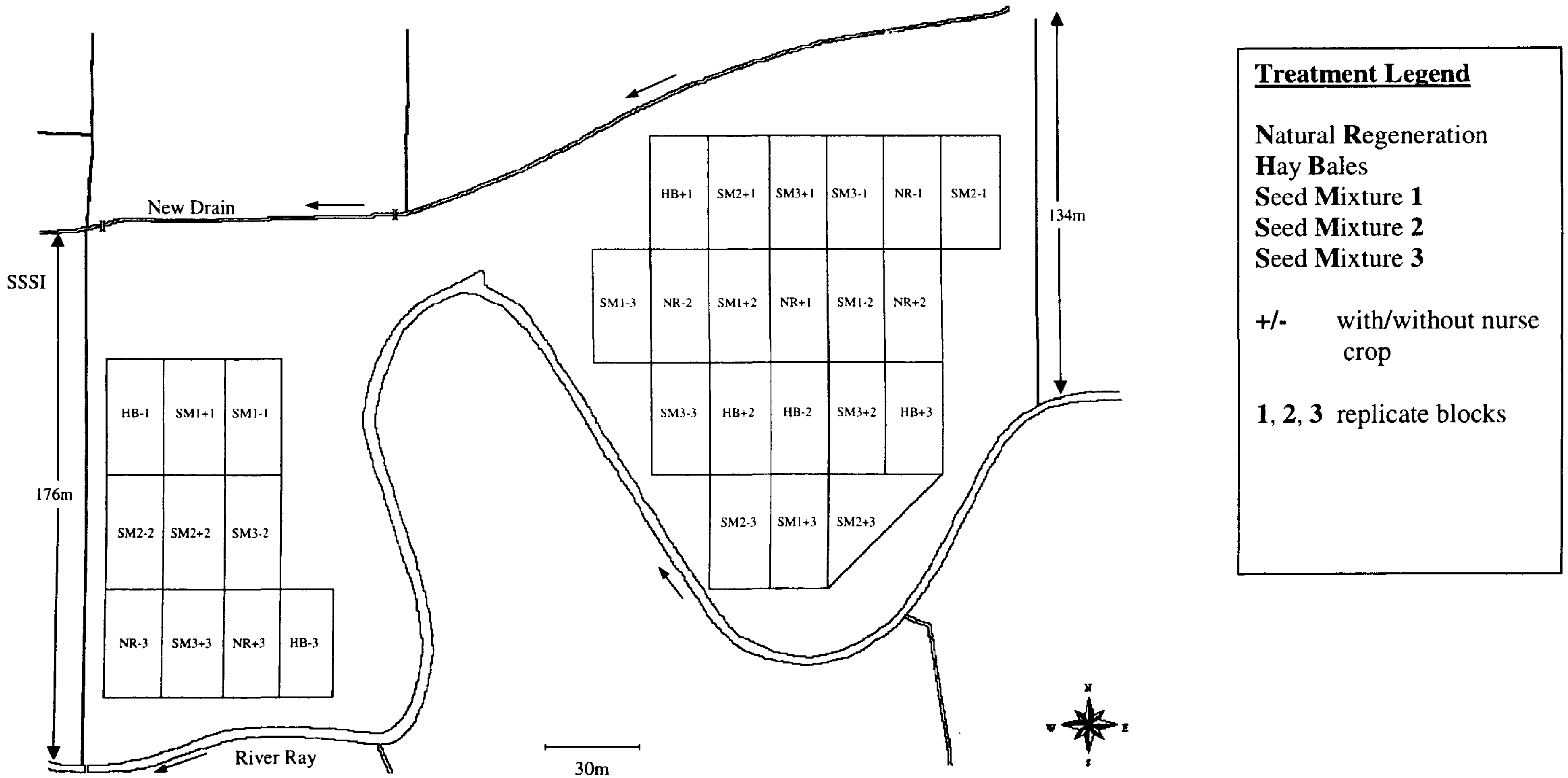
5.2.2 Experimental treatments

The experiment layout consisted of a randomised block design with three replicate blocks, each consisting of ten 18 x 38m plots, to which the 2 x 5 factorial combination of treatments were allocated at random (Figure 5.1). Plot sizes were determined by the size and shape of the field. Five basic treatments were used, each repeated with and without a 'nurse' crop of *Lolium multiflorum*. The use of a 'nurse' or 'cover' crop was tested since traditionally a nurse crop was often used to aid establishment of a grass sward. Wells *et al.* (1989) listed the benefits of using a nurse crop, such as an annual Westerwolds Rye Grass: i) quick germination and establishment of green vegetation; ii) suppression of excessive annual weed growth, and iii) amelioration of harsh conditions and provision of shelter for the slower germinating forbs.
There were thus ten experimental treatments in total:

NR (Natural regeneration)	no seed added;
HB (Hay Bales)	seed derived from hay harvested from the SSSI;
SM1 (Seed Mixture 1)	'Basic' mixture of four species of grass;
SM2 (Seed Mixture 2)	'Intermediate' mixture of six grass and five forb species;
SM3 (Seed Mixture 3)	'Comprehensive' mixture of eight grass and fifteen forb species.

These treatments were chosen to achieve representation of reference/ target communities (as discussed in chapter 3) and to address the practical issues of cost and availability of seed. The composition of Seed Mixtures 1-3 is presented in Appendix 3.3.

Figure 5.1 River Ray arable restoration site: plot layout and experimental treatments



5.2.3 seed acquisition and cost

Commercial availability of seed was a limiting factor in the selection of species (Appendix 3.3). Seed of *Oenanthe fistulosa*, *Thalictrum flavum*, and some *Sanguisorba officinalis* and *Filipendula ulmaria* was picked by hand. The first two were not available commercially, and the latter two grow in abundance in the locality of the site.

5.2.4 Calculation of quantities of seed required

Recommended sowing rates for grassland vary between 20 and 40kg ha^{-1} (Wells *et al.*, 1989) and so seed mixtures were sown at 40kg ha^{-1} (2.736kg per 18m x 38m plot). The cover crop, Westerwolds annual ryegrass 'Karamba' was sown at a low rate of 10kg ha^{-1} 0.684kg $plot^{-1}$) so that individuals would establish but not form dense cover (Wells, pers.comm.).

5.2.5 Seedbed preparation

The seedbed of the study site was prepared during late August 1993. The 'set-aside' vegetation that had developed was removed and the soil harrowed. Ideally, a fine tilth would have been achieved but this was not possible due to the impermeable, unworkable nature of the soil following recent rainfall.

5.2.6 Acquisition of bales

Hay harvested in the summer immediately before the establishment of the experiment (1993) was available. Wells *et al.* (1986) had previously found hay from Cricklade meadow contained approximately 1kg of seed per bale. With a sowing rate of 2.736kg per plot, and assuming the same quantity of hayseed, three bales per plot were acquired. Hay bales were shaken vigorously over the appropriate plot to loosen the seed, and spread on the ground.

5.2.7 Sowing of seed mixtures

Seed was mixed separately for each plot to ease operations in the field and ensure that equal quantities of seed were applied. Following soil seed bank sampling, the seed was sown by hand.

5.2.8 Seed assessment

Two extra hay bales from the target community, from the same harvest as was used in the Hay Bale treatment, were acquired to assess the likely seed composition of the bales used. Each bales were threshed separately over a tarpaulin and then sieved repeatedly to remove the seed from the hay and chaff. Once seed had been extracted from the bales, subsamples were taken in order to identify those species likely to be introduced into the field. Subsampling and seed identification were performed by the National Institute of Agricultural Botany (NIAB).

The viability of the seed that was sown in the field experiment was assessed in the laboratory in order that field observations could be better interpreted, i.e. failure to establish in the field could be attributed to low seed viability or unsuitable field conditions. Germination tests were carried out using plastic petri dishes containing a water-agar substrate. For each species, five replicates each of 20 seeds were established, by placing the seed on the agar in a regular grid pattern. Dishes were placed in a cabinet maintained at 4°C and examined at regular intervals for germinated seeds, which were then removed. Germination was deemed to have occurred when the radicle and/or cotyledon had emerged through the testa. The germination trial was continued for a period of 20 weeks.

5.3 Data collection and analysis

5.3.1 Assessment of seed bank

The sampling methodology and results are presented in Chapter 4. The contribution of the seed bank to establishing vegetation is assessed within the present chapter.

5.3.2 Field botanical survey

During June 1994-1997 and again in 1999, five 1m^2 quadrats were recorded per plot, located as are the spots on the five-face of a die. Within each, all species of vascular plant and bryophyte were recorded together with percentage cover of each species (estimated by eye).

5.3.3 *Within and between individual years*

Results from each year's botanical survey were analysed using analysis of variance (ANOVA), following an angular transformation of the data, to identify significant differences between treatments. Total species richness does not provide information about the composition of the vegetation, thus the data were also investigated in terms of the numbers of target species present. Differences between years were also analysed using ANOVA to assess changes between years.

5.3.4 *Comparison to target vegetation*

Ordination techniques were employed to arrange the botanical survey data in ordination space. Sites/treatments were ordinated using detrended correspondence analysis (DECORANA; Hill, 1979). The default settings were used in the analysis.

5.4 Results

5.4.1 Numbers of species: by treatment

5.4.1.1 Total species richness

Seed Mix 3 (SM3) has consistently contained a higher total number of species (including target species) than the other treatments (Table 5.1). Seed Mix 2 (SM2) performs similarly to Natural Regeneration (NR), Hay Bales (HB) and Seed Mix 1 (SM1) in terms of total number, and number of Class II species, but does contain a greater number of Class I species than these treatments.

Table 5.1 Absolute numbers of species recorded, numbers of Class II and Class I target species within all plots.

Treatment	NR	HB	SM1	SM2	SM3
All species, including bryophytes					
1994	36	40	44	47	55
1995	20	27	27	28	32
1996	28	36	28	37	45
1997	31	41	37	37	48
1999	40	40	43	40	51
Class II species					
1994	21	22	24	30	36
1995	18	17	17	19	24
1996	21	25	19	27	34
1997	21	23	23	26	36
1999	28	30	27	29	38
Class I species					
1994	8	9	11	14	20
1995	5	5	7	8	13
1996	5	9	8	12	20
1997	8	7	9	12	20
1999	10	14	12	16	21

There were significant differences between treatments in terms of the mean number of species per plot (Table 5.2): in 1996 ($p<0.01$) and in all other years ($p<0.001$). SM3 has consistently contained greater numbers of species than NR, HB and SM1, and was richer than SM2 in 1994, 1995 and 1999. SM2 itself has been significantly richer than NR in most years, than HB and SM1 in 1995 and HB again in 1999.

Table 5.2 Mean number of species per plot: all species

Treatment	1994	1995	1996	1997	1999
NR	18.17 ± 0.95	10.50 ± 0.99	15.50 ± 1.02	14.33 ± 0.76	19.0 ± 1.39
HB	20.83 ± 0.83	12.17 ± 1.45	16.83 ± 1.14	17.17 ± 1.54	19.33 ± 0.49
SM1	21.33 ± 1.20	14.83 ± 0.75	17.67 ± 0.88	16.50 ± 1.65	20.33 ± 0.67
SM2	24.33 ± 2.04	20.67 ± 0.92	19.50 ± 2.09	22.00 ± 1.44	23.50 ± 0.85
SM3	32.17 ± 0.79	29.67 ± 0.56	24.67 ± 2.78	27.33 ± 2.84	29.33 ± 1.33
Significance	P<0.001	P<0.001	P<0.01	P<0.001	P<0.001
F value	17.49	59.51	4.86	9.63	24.76

The mean number of species decreased in all treatments between 1994 and 1995 ($p<0.05$; F 4.21; df 4), with larger decreases in NR, HB and SM1 than in SM3. The reduction in NR and HB treatments was significantly greater than the decrease in SM2. Between 1995 and 1996, the mean number of species had increased in NR, HB and SM1, but decreased in SM2 and SM3 ($p<0.01$; F 8.82). Between 1996 and 1997, numbers of species decreased in NR and SM1, but increased in HB, SM2 and SM3. However, differences between 1996 and 1997 were not significant. Between 1997 and 1999, mean numbers of species increased in all treatments.

5.4.1.2 Class I target species

Table 5.3 Mean number of species per plot: Class I target species

Treatment	1994	1995	1996	1997	1999
NR	2.50 ± 0.56	2.00 ± 0.52	2.00 ± 0.26	2.50 ± 0.62	3.67 ± 0.56
HB	3.67 ± 0.42	1.83 ± 0.54	3.00 ± 0.86	3.00 ± 0.45	5.17 ± 0.48
SM1	3.67 ± 0.88	4.33 ± 3.00	4.33 ± 0.33	4.50 ± 0.50	6.33 ± 0.49
SM2	6.33 ± 1.12	11.17 ± 0.17	6.67 ± 1.02	7.83 ± 0.91	9.33 ± 0.42
SM3	12.0 ± 1.48	19.33 ± 0.21	10.17 ± 2.64	12.00 ± 2.05	13.83 ± 1.25
Significance	P<0.001	P<0.001	P≤0.001	P<0.001	P<0.001
F value	21.75	361.12	7.72	12.72	40.21

The number of Class I target species differed between treatments in all years (table 5.3), with SM3 consistently richer than NR, HB, SM1 and richer than SM2 in 1994 and 1999. SM2 was significantly richer than NR and HB in all years except 1996, and SM1 in 1995 and 1999.

Between 1994 and 1995, the mean number of Class I species decreased in NR and HB, but increased in SM1, SM2 and SM3 ($p<0.001$; F 28.55). Between 1995 and 1996, there was a decrease in species within SM3 and SM2, an increase in HB, and no change in NR or SM1 ($p<0.001$; F 16.09). There were no significant differences between 1996 and 1997, although increases in the number of Class I species were recorded for NR, SM1, SM2 and SM3. Again, there were no significant changes between 1997 and 1999, although numbers increased in all treatments.

Ground cover of Class I species

Significant differences between treatments were also observed in the abundance of target species. During 1994, ground cover of Class I species was higher ($p<0.05$; F 4.02) in SM3 (6.61% ± 1.54) than in NR (1.89% ± 0.99) or HB (1.28% ± 0.49). In 1995, abundance of Class I species was again affected by the treatment ($p<0.05$; F 3.78), with higher cover in SM3 (3.01% ± 0.87) than NR (0.86% ± 0.43). In 1997, abundance was higher ($P<0.001$; F 7.17) in SM3 (17.42% ± 3.56) and SM2 (15.89% ± 3.56) than NR (4.27% ± 1.22) or HB (6.43% ± 0.82). By 1999, ground cover was higher in SM3 (46.67% ± 3.82) than in NR (28.31% ± 4.92) or SM1 (28.28% ± 5.14) ($P<0.01$; F 5.74).

5.4.1.3 Class II target species

Table 5.4 Mean number of species per plot: Class II target species

Treatment	1994	1995	1996	1997	1999
NR	10.00 ± 0.58	9.33 ± 0.92	10.83 ± 0.95	10.17 ± 0.91	15.17 ± 0.98
HB	11.83 ± 0.48	9.17 ± 0.87	12.67 ± 0.76	11.33 ± 0.80	15.50 ± 0.76
SM1	12.33 ± 0.99	10.83 ± 1.19	11.83 ± 0.95	11.83 ± 1.45	15.83 ± 1.19
SM2	15.00 ± 1.59	18.00 ± 0.89	15.50 ± 1.80	17.00 ± 1.91	19.33 ± 0.49
SM3	22.33 ± 1.48	26.83 ± 0.40	20.17 ± 2.86	22.17 ± 2.82	25.50 ± 1.48
Significance	P<0.001	P<0.001	P<0.01	P≤0.001	P<0.001
F value	26.12	61.60	5.97	7.75	22.27

There were significant differences in the mean number of Class II target species between treatments in all years. In 1994 more Class II species were present in SM3 than in any other treatment. SM2 plots also contained significantly more Class II species than NR. In 1995 SM2 and SM3 plots were richer in Class II species than NR, HB or SM1. In 1996 and 1997, SM3 was richer than NR, HB, or SM1. In 1999, SM2 and SM3 were richer than NR, but SM3 was also richer than HB, SM1 or SM2.

Between 1994 and 1995, decreases in the mean number of Class II species were recorded in NR, HB, SM1, with significant increases in SM2 and SM3 ($p<0.001$; F 12.32). Between 1995 and 1996, numbers decreased in SM2 and SM3, but increased in NR, HB and SM1 ($p<0.001$; F 10.52). Between 1996 and 1997, numbers declined in NR and HB, but increased in SM2 and SM3. Numbers of Class II species increased in all treatments between 1997 and 1999.

Class II species ground cover

In 1999, the ground cover of Class II species was significantly higher in SM3 (75.53% ± 2.35) than in SM1 (49.74% ± 7.20) ($p < 0.05$; F 4.02).

5.4.1.4 Small-scale species richness

Table 5.5 Mean number of species m^{-2}

Treatment	1994	1995	1996	1997	1999
NR	10.50 ± 0.66	6.07 ± 0.50	8.17 ± 0.81	7.93 ± 0.60	10.37 ± 0.56
HB	10.20 ± 0.40	6.40 ± 0.53	8.17 ± 0.67	8.23 ± 0.64	10.30 ± 0.37
SM1	11.73 ± 0.74	6.70 ± 0.61	9.43 ± 0.91	9.57 ± 0.77	12.67 ± 0.40
SM2	14.13 ± 1.21	8.50 ± 1.09	11.20 ± 1.64	12.77 ± 1.21	13.97 ± 1.07
SM3	17.23 ± 0.88	9.37 ± 0.97	13.27 ± 1.74	14.30 ± 2.00	16.43 ± 1.01
Significance	P<0.001	P<0.05	P<0.05	P<0.01	P<0.001
F value	11.18	3.24	3.02	5.65	15.88

Mean species m^{-2} were higher in SM3 than in NR and HB in 1994, 1997 and 1999, and higher than SM1 in 1994 and 1999. SM2 was richer than NR and HB in 1999. Changes over time were not significant at the 5% level.

5.4.2 *Establishment of sown species*

5.4.2.1 Number of sown species established

During 1994, the establishment of sown species (table 5.6) was affected by the nurse crop with higher percentage establishment in plots without the nurse crop ($P<0.05$; F 7.36; df 1); and the position in the field with establishment being highest in plots furthest from the river ($P<0.05$; F 5.39; df 2). In 1995 establishment was higher in SM1 than SM2 or SM3, and higher in SM2 than SM3 ($P<0.01$; F 11.34; df 2). All plots with a nurse crop continued to have fewer species established ($P<0.05$; F 9.42). In both 1996 ($p < 0.01$; F 12/07) and 1997 ($p \leq 0.001$; F 25.26), establishment was higher in SM1 than SM2 or SM3. By 1999, numbers established were higher in both SM1 and SM2 than in SM3 ($p < 0.01$; F 11.24).

Table 5.6 Mean number of sown species established per plot

Treatment	SM1	SM2	SM3
1994	1.33 ± 0.21	5.50 ± 0.96	11.00 ± 1.26
1995	2.50 ± 0.34	5.00 ± 0.37	6.17 ± 1.49
1996	3.67 ± 0.21	6.17 ± 0.91	9.50 ± 2.20
1997	3.67 ± 0.21	7.33 ± 0.67	11.50 ± 1.86
1999	3.17 ± 0.17	8.17 ± 0.17	12.83 ± 1.05

In terms of the total number of sown species established, all four species of SM1 were established by 1995 and have been recorded each year. The number of species established within SM2 has varied between 7 and 10. *Filipendula ulmaria* was recorded in 1994, but has not been recorded since, and is the missing 11th species. Numbers of species established in SM3 were at a minimum in 1995 (12 species), but each subsequent year 18 of the 23 sown species have been recorded. Species absent, or that have failed to persist in SM3, are *Filipendula ulmaria*, *Oenanthe fistulosa*, *Rhinanthus minor*, *Sanguisorba officinalis* and *Thalictrum flavum*.

5.4.2.2 Abundance of sown species

Table 5.7 Mean percentage ground cover of sown species

Year	Seed Mix 1	Seed Mix 2	Seed Mix 3
1994	2.25 ± 0.72	3.96 ± 1.60	6.26 ± 1.61
1995	0.98 ± 0.47	3.64 ± 1.72	2.97 ± 0.88
1996	8.65 ± 2.35	11.05 ± 3.24	11.19 ± 3.07
1997	14.84 ± 3.70	15.82 ± 3.54	17.23 ± 3.58
1999	14.42 ± 4.90	37.79 ± 2.87	46.07 ± 3.72

In 1995 ($p < 0.05$; F 8.26) and 1997 ($p < 0.05$; F 9.17), differences in the ground cover of sown species were attributable to the nurse crop. In both 1995 (4.02% ± 1.14 cf. 1.04% ± 0.32) and 1997 (21.31% ± 2.12 cf. 10.62% ± 2.21), covers were highest in the absence of the nurse crop. By 1999, cover of sown species was higher in SM2 and SM3 than SM1 ($p < 0.001$; F 43.38) (Table 5.7).

The abundance of sown species declined in all treatments between 1994 and 1995, with the largest decline witnessed in SM3, but increased in all treatments between 1995 and 1997. Between 1997 and 1999, there was a small decline in abundance in SM1 but large increases in both SM2 and SM3 ($p \leq 0.001$; F 49.84).

5.4.3 Ellenberg indicator values

5.4.3.1 Mean moisture (mF) values

There were significant differences at the 5% level in one year only. In 1995, the mF value for NR was higher than that for SM2 ($p < 0.05$; F 3.41; df 4) (table 5.8). There were no significant differences at the 5% level between years.

Table 5.8 Mean Ellenberg moisture (mF) values (± Standard Error): treatments

Year	NR	HB	SM1	SM2	SM3
1994	5.47 ± 0.07	5.47 ± 0.09	5.53 ± 0.07	5.46 ± 0.11	5.47 ± 0.07
1995	6.01 ± 0.10	5.98 ± 0.11	5.77 ± 0.17	5.51 ± 0.10	5.89 ± 0.09
1996	5.68 ± 0.11	5.96 ± 0.10	5.60 ± 0.07	5.70 ± 0.11	5.84 ± 0.09
1997	5.54 ± 0.05	5.65 ± 0.12	5.69 ± 0.06	5.61 ± 0.07	5.75 ± 0.06
1999	5.93 ± 0.13	5.86 ± 0.12	5.83 ± 0.12	5.74 ± 0.07	5.80 ± 0.06

5.4.3.2 Mean nitrogen (mN) values

In 1994 all treatments had significantly higher mN values (table 5.9) than SM3 ($p \leq 0.001$; F 7.46; df 4), but by 1995 only the value for the Hay Bale treatment was significantly higher than those of Seed Mixes 2 or 3 ($p<0.01$; F 4.89). In both 1997 ($p \leq 0.001$; F 7.19) and 1999 ($p<0.001$; F 10.74), SM2 and SM3 had significantly lower mN values than either NR or HB. In 1999 SM3 also had a significantly lower mN value than SM1.

Table 5.9 Mean Ellenberg nitrogen (mN) values (± Standard Error): treatments

Year	NR	HB	SM1	SM2	SM3
1994	6.28 ± 0.11	6.36 ± 0.11	6.04 ± 0.08	6.10 ± 0.15	5.62 ± 0.11
1995	6.33 ± 0.13	6.64 ± 0.11	6.27 ± 0.05	6.02 ± 0.06	5.99 ± 0.19
1996	6.08 ± 0.12	6.18 ± 0.12	6.12 ± 0.08	5.89 ± 0.16	5.81 ± 0.11
1997	6.46 ± 0.05	6.36 ± 0.08	6.22 ± 0.09	5.90 ± 0.17	5.79 ± 0.15
1999	6.05 ± 0.09	5.95 ± 0.09	5.89 ± 0.09	5.62 ± 0.06	5.56 ± 0.04

5.4.4 Similarity to target vegetation

5.4.4.1 Ordination

Ordination 1 Experimental treatments and seed bank samples (Figures 5.2a, b).

The first axis of the ordination explains a large part of the variation within the dataset (eigenvalue=0.502). The ordination of sites (figure 5.2a) reveals that samples are clearly separated in ordination space on the basis of the year of sampling and are also distinct from the seed bank samples. The length of the first ordination axis (3 S.D.) indicates that the seed bank, whilst dissimilar, is not so dissimilar from the vegetation, even by 1999, that it does not have species in common with the developing sward. Examination of the species ordination (figure 5.2b) suggests that the first axis is related to early successional changes in vegetation. The (ex-) arable seed bank (to the right of the ordination) is characterised by species adapted to survive the frequent soil disturbance associated with arable cultivation and is dominated by ruderal and/or competitive species that form long-term persistent seed banks. The composition of the restored vegetation in its first year (1994) was more similar to the seed bank composition than the vegetation of later years. Changes over time in the vegetation developed from the different seed treatments reflect early successional processes as the initially dominant annual arable weed species are lost from the vegetation, replaced by perennial grassland species

It is also interesting to note that, in the earliest years (1994 and 1995) the differing treatments were extremely similar to one another, but as time progresses the vegetation appears to diverge, so that by 1997 and 1999 the difference between the vegetation developed from the various species introductions appears much greater than in the earlier years.

Figure 5.2a. Sample ordination: Experimental seed treatments and seed banks. NR: Natural Regeneration; HB: Hay Bales; SM1-SM3: Seed mixtures 1-3. Suffix (94-99) corresponds to year of sampling; sb: seed bank samples.

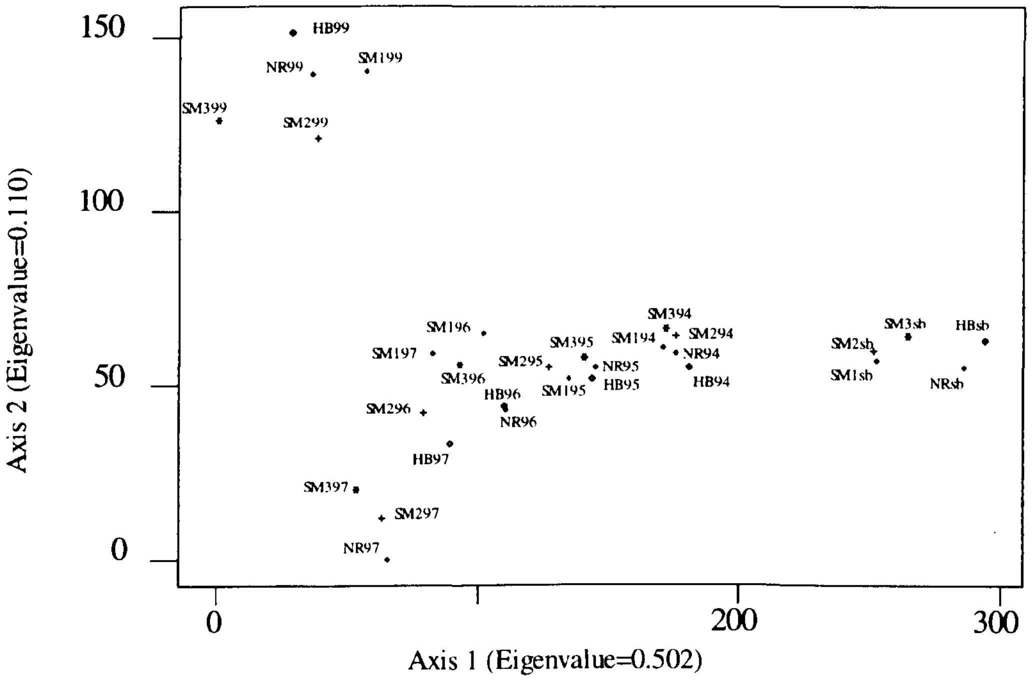
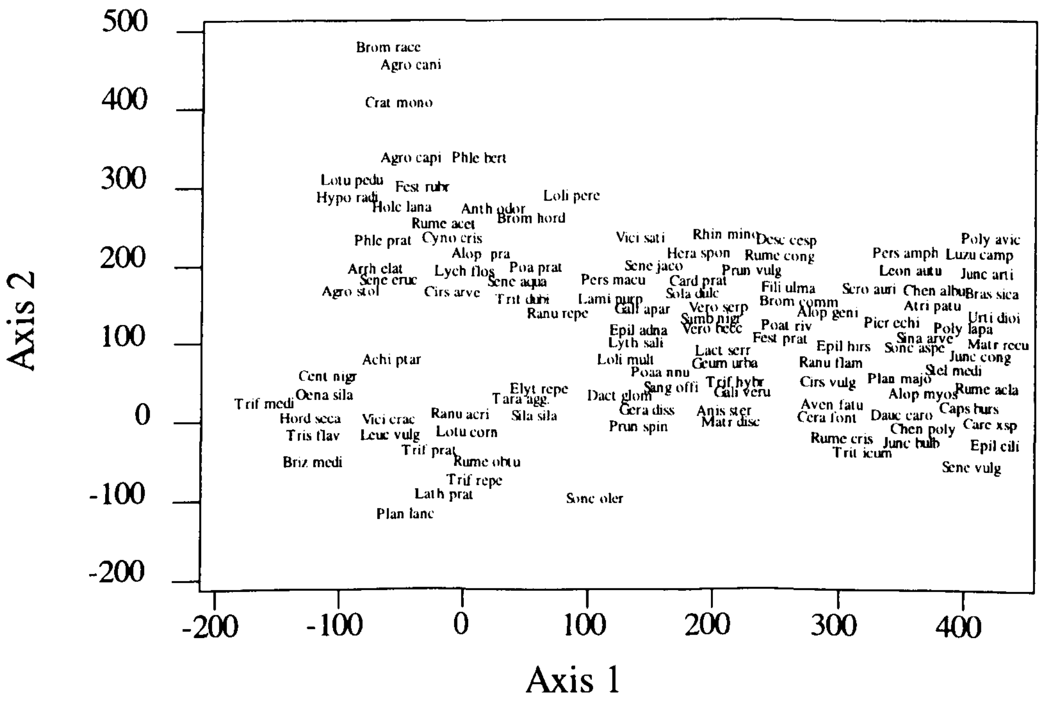


Figure 5.2b Species Ordination: Experimental treatments and seed bank



Ordination 2 Experimental treatments & target vegetation transects (Figures 5.3a, b).
In order to assess the similarity of the restoration treatments to the target vegetation of Long Herdon SSSI, the second ordination included the different seed treatments relative to samples from the target vegetation. The first axis explains the majority of variation in the dataset (eigenvalue= 0.676), clearly representing the differences between the SSSI vegetation and the newly established grassland (figure 5.3a). The SSSI is characterised by perennial species (figure 5.3b), some of which are stress tolerators, whilst the new vegetation is characterised by competitive ruderal annual species. There also appears to be differences between the SSSI and the treatment vegetation in terms of species preferences for moisture and soil nitrogen, with the SSSI samples characterised by species with a preference for moist to wet soil conditions and/or low available nitrogen (classified according to Ellenberg, 1988). Species abundant in the restored vegetation appear more preferential for conditions of high nitrogen availability and average soil wetness.

The ordination of sites (figure 5.3a) clearly shows that the restored vegetation is dissimilar to the reference habitat. The pattern first suggested by ordination 1, of different treatments being more similar to one another within years than to themselves between years, is confirmed. This grouping of treatments on the basis of a common year highlights the similarity of the underlying vegetation before the addition of seed treatments and suggests that changes between years are attributable more to the process of natural regeneration and early succession following the cessation of arable cultivation than to the different treatments. In the later years (1997, 1999), the treatments appear to be diverging which perhaps suggests that the different seed inputs will enable the vegetation to follow different successional pathways. Development of the treatment swards over time showed movement only along axis one for the first four years, as annual arable weeds are lost from the vegetation. Between 1997 and 1999, however, the treatments displayed movement along the second axis towards the target vegetation. This trend appears to be away from arable weeds and annual species and towards more perennial grassland species.

Figure 5.3a. Sample ordination: Experimental seed treatments and target vegetation. NR, HB, SM1-3: experimental treatments. 94-99: year of survey. SS93v1-3: SSSI target vegetation samples (3) recorded in 1993; SS96v1-3: SSSI target vegetation samples (3) recorded in 1996.

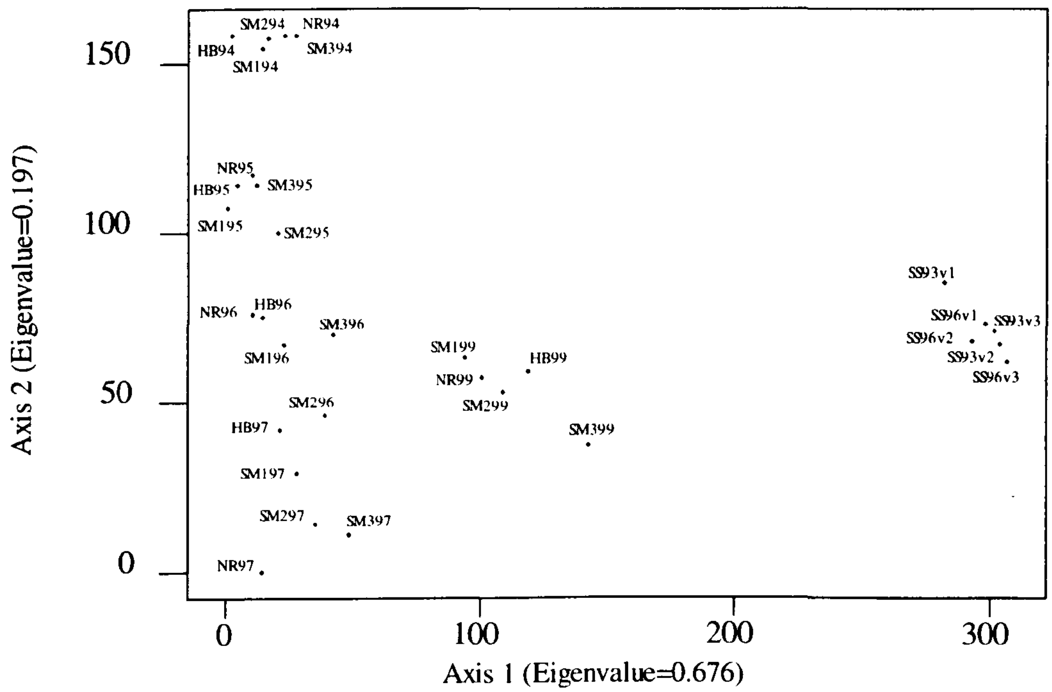
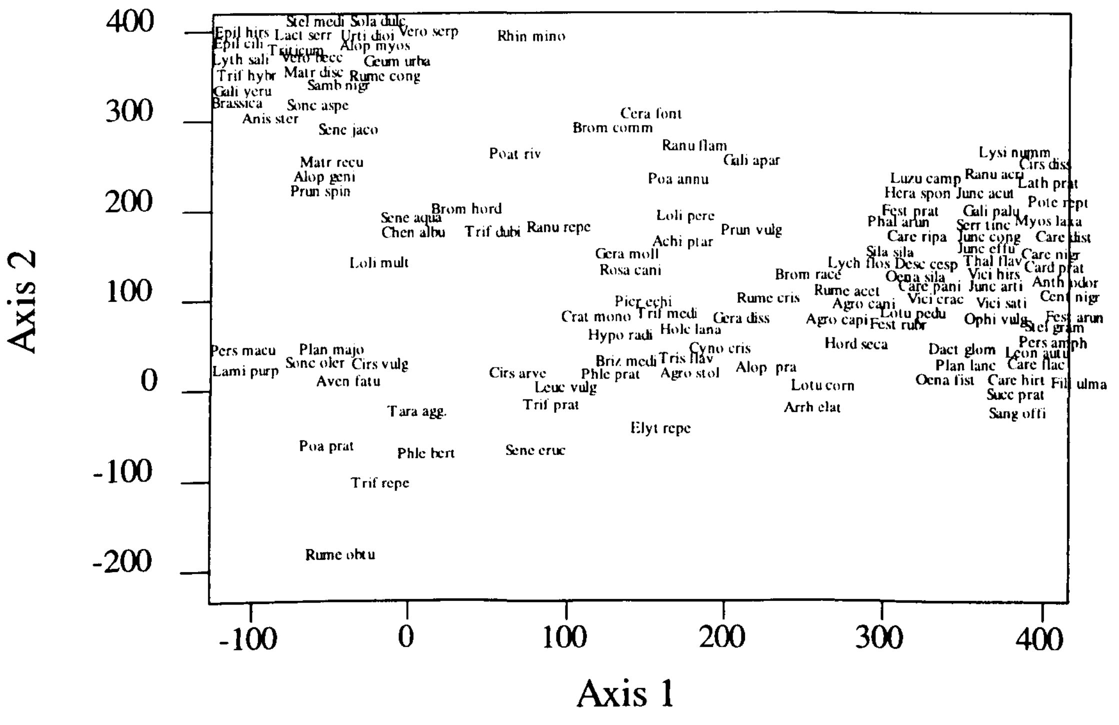


Figure 5.3b Species ordination: experimental treatments and target vegetation



Ordination 3 Treatments with corridor grasslands (Figures 5.4a, b)

It is unrealistic to expect the vegetation of a newly created grassland to approximate to that of a SSSI and so the third, and final, ordination was of data that comprised the restoration treatments, the target vegetation and the other adjacent grasslands of varying management history. Once again, the first axis explains the larger part of variation in the dataset (eigenvalue=0.620), and represents a gradient from perennial species typical of above average soil moisture and below average nitrogen availability on the right hand side of the ordination (figure 5.4b) towards competitive and ruderal species typical of average soil moisture and high nitrogen availability on the left hand side. The Plantlife meadow (PL) is closest to the SSSI in ordination space (figure 5.4a), reflecting the similarity of the vegetation within these meadows. The separation of the experimental treatments and the Reverting (REV) field from the remaining grasslands may be due to the history of cultivation of these two fields, i.e. not only have both been 'improved' by fertilisation but also reseeded (following soil cultivation). The Reverting field also differs from the restoration treatments, however, and some of this variation appears to be attributable to differing species preferences for available nitrogen with an abundance of species in the Reverting field with a preference for high nitrogen availability. The species at the extremes of the second axis (figure 5.4b) suggest that there is more influence of ruderal and annual species in the Reverting field than in the restored vegetation of the experimental field.

Figure 5.4a. Sample ordination: Experimental seed treatments and grasslands of differing management history. NR, HB, SM1-3 94-99: experimental treatments and year of sample. IE: Improved East; IW: Improved West; PL: Plantlife meadow; SS: Long Herdon SSSI; REV: Reverting field. 93, 96: year of sample. v1-v4: vegetation samples.

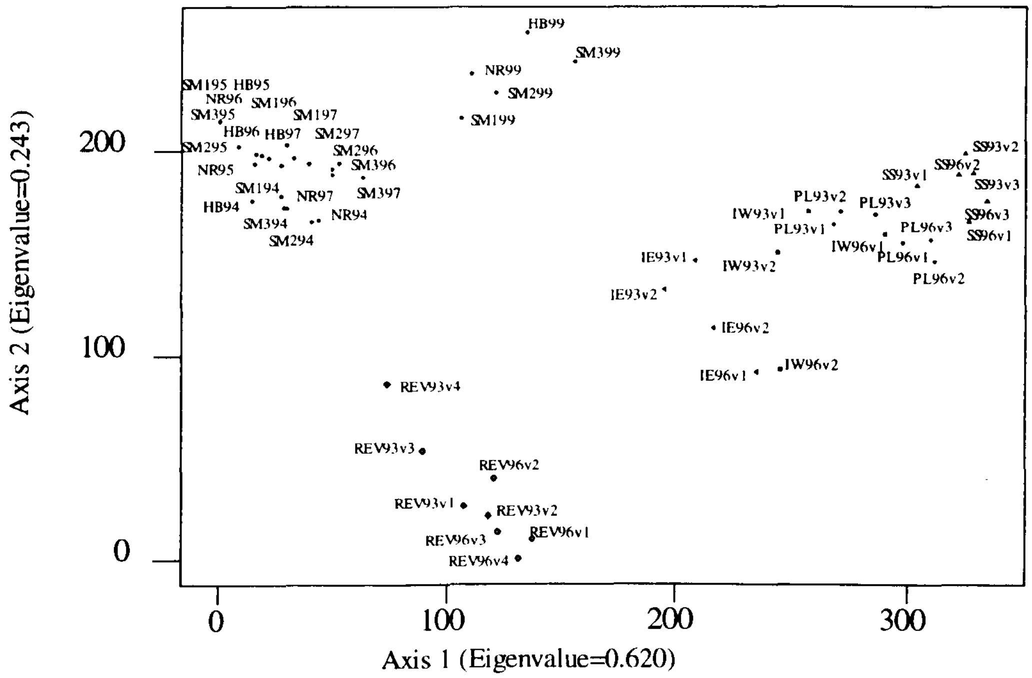
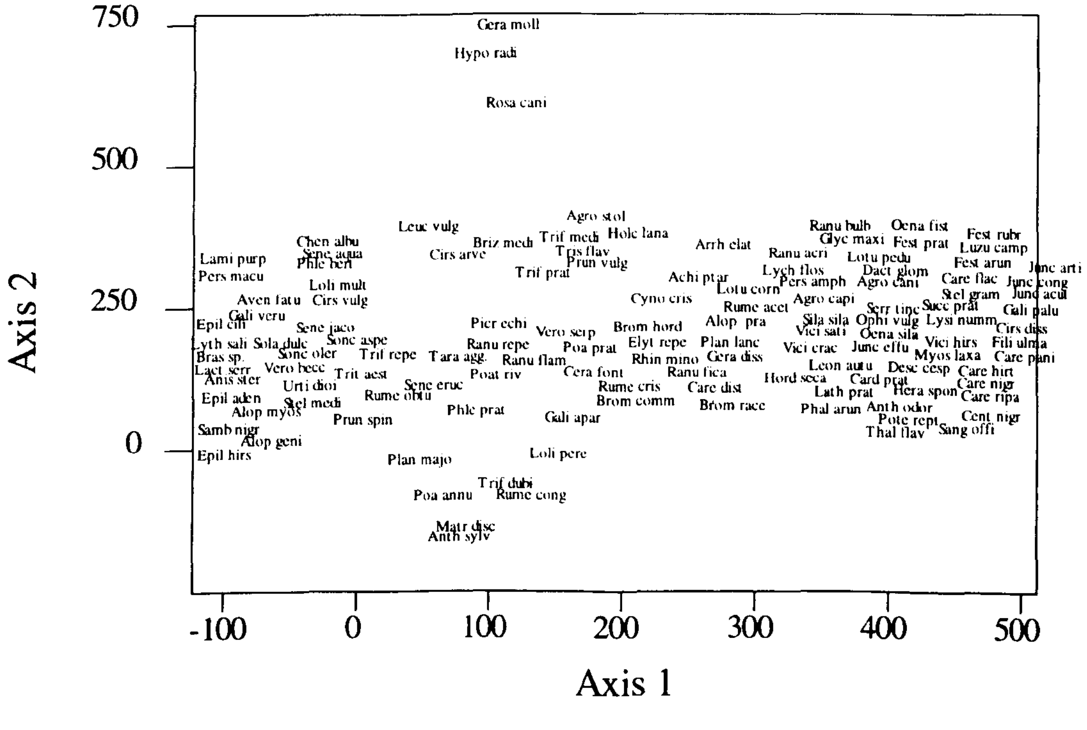


Figure 5.4b Species ordination: experimental treatments and adjacent grasslands



5.4.4.2 National Vegetation Classification (NVC) communities

The target grassland vegetation types within the catchment were identified as *Alopecurus pratensis-Sanguisorba officinalis* grassland (MG4), the *Lathyrus pratensis* subcommunity of *Centaurea nigra-Cynosurus cristatus* grassland (MG5a), and *Cynosurus cristatus-Caltha palustris* grassland (MG8) (Rodwell 1992b). In order to further assess the development of the vegetation arising from the different treatments, TABLEFIT (Hill, 1991) was employed to assign probable community types to each treatment based on the species composition and cover. When individual quadrats and pooled treatment results were analysed, a common core of mesotrophic grassland types was identified. Thus only summary results for treatments with and without the nurse crop are presented (table 5.10). These results further demonstrate the influence of the nurse crop on treatments. For example, the sward within SM3 without the nurse crop has generally been assigned to mesotrophic communities of reduced conservation value (MG6a, MG9a). However, in the absence of the nurse crop stands of SM3 appear more similar to the target communities, and in particular MG4 and MG8.

Table 5.10 National Vegetation Classification (NVC) community types present.

Plot means (with and without the nurse crop) assigned to NVC community types by Tablefit (Hill, 1991).

Natural Regeneration

	With nurse Best Fit	MG4	MG5a	MG8	Without nurse Best Fit	MG4	MG5a	MG8
1994	MG9a (30)			18	MG9a (32)			
1995	MG9a (41)				MG9a (42)			
1996	MG9a (37)				MG9a (36)			
1997	MG6a (36)			30	MG6a (31)			28
1999	MG9a (36)			29	MG9a (37)			

Hay Bales

	With nurse Best Fit	MG4	MG5a	MG8	Without nurse Best Fit	MG4	MG5a	MG8
1994	MG7c, MG9a (23)				MG7b (25)			22
1995	MG7b (32)				MG9a (30)			
1996	MG9a (25)		20		MG7b (29)			26
1997	MG9a (25)			23	MG6a (36)			31
1999	MG9a (43)			31	MG9a (48)			34

Seed Mixture 1

	With nurse Best Fit	MG4	MG5a	MG8	Without nurse Best Fit	MG4	MG5a	MG8
1994	MG9a (29)				MG6a (24)	21		
1995	MG6a (33)				MG6a (31)			
1996	MG6a (39)			30	MG6a (34)			
1997	MG6a (42)	28	26	41	MG6a (35)			
1999	MG9a (34)	29		31	MG9a (41)	34		34

Seed Mixture 2

	With nurse Best Fit	MG4	MG5a	MG8	Without nurse Best Fit	MG4	MG5a	MG8
1994	MG9a (31)	24	24	24	MG8 (27)	25	22	27
1995	MG9a (33)		25		MG9a (32)		30	27
1996	MG6a (34)		31	31	MG6a (44)		38	41
1997	MG6a (37)		29	34	MG8 (39)	34	35	39
1999	MG9a (41)	37	38	39	MG8 (40)	37	39	40

Seed Mixture 3

	With nurse Best Fit	MG4	MG5a	MG8	Without nurse Best Fit	MG4	MG5a	MG8
1994	MG4 (28)	28	21	25	MG4 (35)	35	25	28
1995	MG6a (33)		23		MG9a (32)	29		28
1996	MG6a (25)	22		23	MG4, MG8 (33)	33	31	33
1997	MG6a (40)	37		37	MG4 (42)	42	35	38
1999	MG9a (40)	38	34	34	MG4 (39)	39	37	35

5.4.5 *The effects of the nurse crop*

The nurse crop, *Lolium multiflorum*, achieved maximum abundance one year after sowing (1994) ranging from 45.30% (SM2) to 57.10% (SM1), and has been largely declining since then. Changes were significant between 1995 and 1996 ($p<0.05$; F 4.63; df 2), when *Lolium* increased in block 2 (+8.15% ± 2.80) but decreased in block 1 (-4.96% ± 2.39). Between 1996 and 1997, *Lolium* declined ($p<0.05$; F 5.64) in block 2 (-9.15% ± 3.05) relative to block 3 (+1.49% ± 3.41), and also declined in block 1 (-4.70% ± 2.76). There were also significant changes within treatments between 1996 and 1997 ($p<0.01$; F 4.62): a decrease in abundance in NR (-14.48% ± 2.31) and slight increases in HB (+2.65% ± 2.66) and SM2 (+3.58% ± 4.44). By 1999, abundance was higher in SM1 (33.21% ± 4.72) and SM2 (31.33 ± 4.11) than in SM3 (15.11% ± 2.31) ($p < 0.05$).

5.4.5.1 Effects on species richness

5.4.5.2

Table 5.11a Effects of the nurse crop: Total mean number of species

Year	Nurse	No nurse
1994	22.80 (1.41)	23.93 (1.55)
1995	16.93 (1.94)	18.20 (1.94)
1996	17.73 (1.0)	19.93 (1.57)
1997	18.27 (1.11)	20.67 (1.98)
1999	21.07 (1.08)	23.53 (1.22)

Table 5.11b Effects of the nurse crop: Class II species: mean number and mean abundance

Year	Nurse		No nurse	
	Number	Ground cover	Number	Ground cover
1994	13.80 (1.21)	31.86 (5.19)	14.80 (1.44)	40.95 (4.30)
1995	14.33 (1.93)	9.06 (2.26)	15.33 (1.86)	20.87 (4.00)
1996	13.27 (0.90)	24.16 (5.39)	15.13 (1.63)	34.56 (5.68)
1997	13.67 (1.22)	38.82 (6.73)	15.33 (1.86)	45.82 (5.75)
1999	17.07 (1.08)	56.20 (3.34)	19.47 (1.26)	66.33 (4.05)

Table 5.11c Effects of the nurse crop: Class I species: number and abundance

Year	Nurse		No nurse		Cover	
	Number	Ground cover	Number	Ground cover	Signif.	F value
1994	5.0 (1.01)	2.47 (0.53)	6.27 (1.12)	4.57 (1.04)	P<0.05	4.89
1995	7.53 (1.86)	0.78 (0.22)	7.93 (1.76)	2.92 (0.76)	P<0.01	11.05
1996	4.47 (0.67)	4.75 (1.10)	6.00 (1.40)	8.76 (2.00)	P<0.05	4.61
1997	5.53 (0.90)	7.85 (1.41)	6.40 (1.34)	13.50 (2.19)	P<0.05	6.31
1999	7.07 (0.91)	31.69 (2.65)	8.27 (1.15)	39.91 (3.31)	P<0.05	7.29

The nurse crop has consistently depressed total numbers of species (Table 5.11a), and in 1999 this was significant ($p<0.01$; F 10.12). Class I target species have been similarly affected (Table 5.11c), and in 1999 this difference was significant ($p\leq0.05$; F 4.47). A more marked difference was observed in the abundance of Class I species, being higher in the absence of the nurse crop in all years ($p < 0.05$).

Similarly, numbers of Class II target species have been depressed in all years by the presence of the nurse crop (Table 5.11b), and again in 1999 this difference was significant ($p\leq0.01$; F 8.37). Ground cover of Class II species was similarly depressed by the nurse crop ($p < 0.05$; F 5.50) in 1999.

The presence of the nurse crop also affected small-scale species richness. In 1999, the mean number of species m^{-2} differed significantly ($p<0.05$; F 7.10; df 1) between plots that received the nurse crop (11.97 ± 0.75) and those plots that did not (13.52 ± 0.72).

5.4.5.2 Effects on individual species

The presence of the nurse crop was responsible for significantly depressing the abundance of a number of species during the monitored period (Table 5.12). The only species to respond positively to the presence of the nurse crop was *Lolium multiflorum* itself.

In addition, there were a number of significant treatment nurse interactions. In the majority of cases these were for species within the treatment where they were sown.

Species significantly more abundant in plots of SM3 without the nurse crop:

1995: *Lathyrus pratensis* ($p<0.05$; F 3.68),

1996: *Trisetum flavescens* ($p<0.05$; F 3.70), *Silaum silaus* ($p<0.05$; F 3.73), *Vicia cracca* ($p<0.01$; F 5.43).

1997: *T.flavescens* ($p<0.05$; F 3.33), *L.pratensis* ($p<0.001$; F 85.27), *Hordeum secalinum* ($p<0.001$; F 8.84)

In SM2 without the nurse crop:

1999: *Lotus corniculatus* ($p<0.05$; F 8.18), *Briza media* ($p<0.05$; F 4.00)

In HB without the nurse crop:

1994: *Anthoxanthum odoratum* ($p<0.05$; F 3.19).

Table 5.12 Species showing significant differences in abundance with the nurse crop

Species	Year	Signif.	F	nurse (±S.E.)	no nurse
Alopecurus myosuroides	1994	P≤0.05	6.19	2.12 ± 0.61	7.58 ± 2.06
Lolium multiflorum	1994	P≤0.05	4.96	61.05 ± 4.37	41.55 ± 6.42
Ranunculus acris	1994	P≤0.05	4.93	0.03 ± 0.01	0.16 ± 0.06
Agrostis capillaris	1995	P≤0.05	4.66	0.003 ± 0.002	0.34 ± 0.25
Cynosurus cristatus	1995	P≤0.05	4.53	0.06 ± 0.03	0.34 ± 0.15
Poa trivialis	1995	P≤0.01	11.45	4.03 ± 0.82	8.65 ± 1.34
Hordeum secalinum	1996	P≤0.01	6.21	0.001 ± 0.001	0.19 ± 0.11
Alopecurus pratensis	1997	P≤0.05	4.73	0.90 ± 0.31	2.07 ± 0.59
Elytrigia repens	1997	P≤0.05	5.77	0.01 ± 0.01	0.93 ± 0.58
Festuca rubra	1997	P≤0.05	7.44	0.13 ± 0.09	0.57 ± 0.19
Lathyrus pratensis	1997	P≤0.001	85.27	0.00 ± 0.0	0.07 ± 0.04
Leucanthemum vulgare	1997	P≤0.05	7.09	0.31 ± 0.17	1.32 ± 0.55
Bromus racemosus	1999	P≤0.05	4.41	0.41 ± 0.17	1.05 ± 0.34
Lotus corniculatus	1999	P≤0.01	11.86	0.003 ± 0.003	0.20 ± 0.13

5.4.6 *Summary of the contribution of the seed bank*

Fifty-one species germinated from the seed bank of SA123 (the experimental field) (Chapter 4): 22 Class II species and 7 Class I species (*Alopecurus pratensis, Anthoxanthum odoratum, Carex spp., Festuca pratensis, Holcus lanatus, Ranunculus flammula* and *Silaum silaus*). Target species contributed very little to the total number of seedlings, however, with 90% of seedlings emerged from the seed bank produced by just 11 species:

- six grasses (*Poa trivialis, Lolium multiflorum, Alopecurus myosuroides, A.geniculatus, Triticum aestivum*);
- four forb species (*Matricaria recutita, Sonchus asper, Stellaria media, Epilobium ciliatum*); and
- one rush (*Juncus conglomeratus*).

Of these, only three occur within the target vegetation (*P.trivialis, A.geniculatus, J.conglomeratus*).

The species that contribute the majority of seeds within the seed bank are broad-leaved arable weeds, volunteer crop species and grass arable weeds (appendix 4.1).

The species that dominate in the seed bank have also made a marked contribution to the establishing above-ground vegetation, particularly in the early years. The composition of the seed bank (sampled within plots of the Natural Regeneration treatment) was compared with the sward developed from the NR treatment (the only treatment that did not receive propagules). In the first year of establishment (1994), nine of the 11 dominant seed bank species were recorded in the vegetation, making up 25% of species present and accounting for 87% of the ground cover. Subsequent to 1994, however, the number of these dominant seed bank species present in the vegetation declined, as did their contribution to the ground cover so that by 1999, only three of these species were present in the vegetation (7.5% of the number of species) and their contribution to ground cover had declined to 33%.

Not all species present within the seed bank have germinated in the field. Indeed, 12 seed bank species, many of them arable weeds, were not recorded within the developing vegetation of the ex-arable field at any time during the monitored period.

5.4.7 Individual species results: by treatment

There were several species whose mean percentage cover differed significantly between treatments (table 5.13; treatment means for individual species abundance are presented in Appendices 5.21-5.25). The majority of these species were sown within the seed treatments, and largely tended to occur more abundantly where they were sown. There were exceptions, however, for example *Holcus lanatus* occurred more abundantly in the HB treatment, despite being sown only in SM2 and SM3.

Table 5.13 Significant results for individual species. Treatments listed are those where species was more abundant. Where the treatment acronym is in bold, this indicates that the species was sown within the treatment and was also more abundant there also. '**x**': species was sown within treatment, but not present at higher abundance.

Species	**Year**	**NR**	**HB**	**SM1**	**SM2**	**SM3**	**Significance**	**F**
Cynosurus cristatus	1994			**SM1**	**SM2**	**SM3**	p<0.05	15.32
Filipendula ulmaria	1994				**x**	**SM3**	p≤0.001	6.95
Rhinanthus minor	1994					**SM3**	p<0.001	18.84
Rumex acetosa	1994					**SM3**	p<0.001	14.83
Silaum silaus	1994					**SM3**	p<0.001	23.27
Trifolium pratense	1994				**x**	**SM3**	p<0.01	6.49
Cynosurus cristatus	1995			**SM1**	**SM2**	**SM3**	p<0.05	4.20
Lathyrus pratensis	1995					**SM3**	p<0.05	3.66
Leucanthemum vulg.	1995				**SM2**	**x**	p≤0.001	7.48
Sanguisorba officin.	1995					**SM3**	p<0.05	4.00
Silaum silaus	1995					**SM3**	p<0.01	4.76
Trifolium pratense	1995				**SM2**	**x**	p<0.01	5.32
Phleum bertolonii	1995			**SM1**			p<0.001	8.81
Anthoxanthum odor.	1996		HB				p<0.01	5.00
Cynosurus cristatus	1996			**SM1**	**SM2**	**SM3**	p<0.001	62.67
Hordeum secalinum	1996					**SM3**	p<0.01	6.66
Leucanthemum vulg.	1996				**SM2**	**SM3**	p<0.05	4.40
Ranunculus acris	1996				**SM2**	**x**	p<0.05	3.84
Silaum silaus	1996					**SM3**	p<0.05	4.30
Trifolium pratense	1996				**SM2**	**x**	p≤0.001	7.17
Trisetum flavescens	1996					**SM3**	p<0.05	3.70
Vicia cracca	1996					**SM3**	p<0.01	4.12
Lolium perenne	1996			SM1			p<0.01	5.04
Phleum pratense	1996				SM2	SM3	p<0.05	3.86
Phleum bertolonii	1996			**SM1**			p<0.001	14.13
Alopecurus pratensis	1997			**SM1**	**SM2**	**SM3**	p≤0.001	16.33
Centaurea nigra	1997					**SM3**	p<0.05	4.49
Cynosurus cristatus	1997			**SM1**	**SM2**	**SM3**	p<0.001	23.39
Festuca rubra	1997			**SM1**	**x**	**SM3**	p<0.01	5.29
Holcus lanatus	1997		HB		**x**	**x**	p<0.05	3.81
Hordeum secalinum	1997					**SM3**	p<0.001	17.24
Lathyrus pratensis	1997					**SM3**	p<0001	85.27
Leucanthemum vulg.	1997				**SM2**	**SM3**	p<0.001	12.52
Lotus corniculatus	1997				**SM2**		p<0.001	8.78
Trifolium pratense	1997				**SM2**	**SM3**	p≤0.001	7.62
Trisetum flavescens	1997					**SM3**	p<0.001	9.75
Phleum pratense	1997					SM3	p<0.05	3.25
Phleum bertolonii	1997			**SM1**			p<0.001	22.50

Table 5.13 Significant results for individual species (continued).

Species	Year	NR	HB	SM1	SM2	SM3	Significance	F
Alopecurus pratensis	1999			**SM1**	**SM2**	**SM3**	p<0.001	14.42
Anthoxanthum odor.	1999		HB				p<0.05	4.11
Briza media	1999					**SM3**	p<0.05	4.00
Centaurea nigra	1999					**SM3**	p<0.05	3.59
Cynosurus cristatus	1999			**SM1**	**SM2**	**SM3**	p<0.001	33.47
Festuca rubra	1999			x	**SM2**	x	p<0.05	3.76
Holcus lanatus	1999		HB		x	x	p<0.05	4.03
Hordeum secalinum	1999					**SM3**	p<0.001	26.20
Lathyrus pratensis	1999					**SM3**	p<0.05	4.49
Leucanthemum vulg.	1999				**SM2**	**SM3**	p<0.001	13.29
Lotus corniculatus	1999				**SM2**		p<0.001	10.90
Ranunculus acris	1999				**SM2**	**SM3**	p<0.001	25.62
Silaum silaus	1999					**SM3**	p<0.01	6.66
Trisetum flavescens	1999					**SM3**	p<0.001	11.83
Vicia cracca	1999					**SM3**	p<0.01	4.82
Agrostis canina	1999			SM1			p<0.05	3.20
Lolium multiflorum	1999	x	x	**SM1**	**SM2**	x	p<0.05	4.18

5.4.8 Viability of sown seeds

Species varied widely in their germination rates in the laboratory, with grass species tending to achieve higher germination rates than the wild flowers (Appendix 5.3). Success or failure to germinate on agar did not predict performance in the field, however. For example, 43% of seeds of *Oenanthe fistulosa* germinated in the laboratory but this species was not recorded in the experimental vegetation at any time, whilst seed of *Silaum silaus* failed to germinate on agar but by 1999 this species was present in 67% of field plots.

5.4.9 Composition of hay bales

The two extra bales sampled to determine the likely species composition of hay varied in terms of total numbers of species present (10, 19 species) (table 5.14). The most species-poor bale appeared to be dominated by grass species (7). Two species recorded were not species of grassland (*Betula* spp. and *Rubus* spp.).

Table 5.14. Species present as seed within hay baled from Long Herdon SSSI

Species	Bale I	Bale 2
Agrostis spp.	x	x
Anthoxanthum odoratum	x	x
Betula spp.		x
Carex spp.	x	x
Centaurea nigra	x	
Cirsium spp.	x	
Cynosurus cristatus	x	x
Deschampsia cespitosa	x	
Festuca pratensis	x	
Festuca spp.	x	x
Holcus lanatus	x	x
Juncus spp.	x	x
Lychnis flos-cuculi	x	
Oenanthe spp.	x	
Poa annua		x
Poa trivialis	x	x
Prunella vulgaris	x	
Ranunculus flammula	x	
Ranunculus repens	x	
Rubus spp.	x	
Trifolium pratense	x	

5.5 Discussion

This experiment aimed to determine the likely success of grassland re-creation upon ex-arable land from different levels of propagule introduction based on defined criteria for success. At the time of sowing, guidelines for the reversion of arable land were similar within both agri-environment schemes in the area (UTT ESA and Countryside Stewardship). They recommended the introduction of only a small number of grass species as seed to ex-arable sites to establish, and act almost as a nurse crop, providing a matrix within which desirable species could establish. The success of this simple sowing is dependent upon natural processes of dispersal and colonisation to ensure that propagules do indeed reach the site, germinate and establish.

Prior to the experiment, literature reviews already suggested that this prescription might not have the desired effect. The frequent soil cultivation associated with arable agriculture favours species typical of disturbed habitats (Hodgson & Grime, 1990) and so, with time, arable seed banks become dominated by seed of short-lived species such as arable weeds, which are adapted to the disturbance regime (Hutchings & Booth, 1996; Pywell *et al.*, 1996). Concurrent to this, species more typical of grassland habitats decline in the seed bank. Habitat fragmentation interrupts natural processes of dispersal and colonisation and many potential restoration sites are now isolated from sources of suitable propagules. The sowing of a simple grass mixture is most likely to succeed on sites where native grassland vegetation has only recently been eliminated (where the seed bank still contains suitable propagules) and that are adjacent to species-rich sources of propagules (for natural dispersal).

The agri-environment schemes did not provide detailed guidance to aid in setting targets or determining the success of arable reversion, which is a major failing. An aim here was to set targets on a site-specific basis for this experiment. The use of a Site of Special Scientific Interest to provide a target vegetation type set extremely high standards for habitat restoration on this site because semi-natural grasslands developed with traditional agricultural practices over many years whereas recently developed grassland cannot be expected to be a perfect facsimile of such habitats. The majority of land in the wider countryside (defined as land outside of ESAs, Haines-Young *et al.*, 2000) is not of SSSI quality and, in the short term, such rigorous goals for arable reversion are unrealistic. An added complication is that the vegetation within the Nature Reserve is dynamic (Chapter 3, section 3.5) and the evaluation of the reversion must allow for this by measuring success against the target habitat both at the beginning and at a later date. This would allow any changes caused by natural phenomena such as drought or flooding to be taken into account. The evaluation criteria selected for this project are somewhat simplistic, but are achievable and practical and allow for an assessment of progress towards the desired endpoint. By assessing each treatment against the evaluation criteria, the direction of succession and the success so far may be determined.

Criterion I   total numbers of species present

In all experimental treatments (and plots), mean numbers of species have fluctuated between years (tables 5.1-5.2). After six years of monitoring, the vegetation was not stable and was still developing. Judged against criterion I, the reversion has not been successful. However, on its own, the total number of species present is a poor indicator of success.

Criterion II   numbers and ground cover of Class II and Class I species

If criterion II is used, SM3 is the most successful treatment as it has consistently contained higher numbers of Class II species than any other treatment. Interestingly, although total numbers of species dropped in all treatments in 1995, numbers of Class II species actually increased in SM2 and SM3. The reduction in total numbers of species, despite an increase in 'habitat specific' (grassland) species, was due to the absence of a number of arable weeds that have been transient in the vegetation such as *Lactuca serriola*, *Matricaria* spp., *Sonchus* spp. and *Stellaria media*. As with total species richness, numbers of Class II species have fluctuated between all years, although by 1999 numbers had generally reached their highest in all treatments since sowing. In terms of the core of target species, Class I species, SM3 has consistently contained more than any other treatment, although not always significantly more than SM2, which itself has been richer than NR, HB or SM1. Not only have target species been better represented in numbers in plots that received the most diverse seed mixture, but ground cover of target species has also been highest in SM3. By using criterion II, the arable reversion has shown some success, especially in SM3.

Criterion III   small-scale species-richness

Criterion III was the restoration of small-scale diversity (i.e. numbers of species m^{-2}) since the target communities of MG4, MG5a and MG8 (as well as that of the reference habitat) are species-rich. Unimproved grasslands adjacent to the reversion site contained an average of 19.9 species m^{-2}. In addition, analysis of grasslands within the wider catchment resulted in the calculation of mean numbers of species m^{-2} for each of the NVC community types present (Manchester *et al.*, 1999). The target communities were generally the richest, with both MG4 and MG5 containing over 20 species m^{-2}, with MG8 vegetation slightly poorer (19.3).

Whilst none of the experimental treatments has yet achieved the small-scale species-richness (table 5.5) observed in these unimproved grasslands, SM3 most closely approaches this with 16 species m^{-2}. Numbers have been increasing in all plots since 1995, when NR, HB and SM1 were similar in species-richness to set-aside fields in the study area. This does suggest that there is immigration of propagules into the experimental field, either from the seed bank (*in situ*) or the seed rain (*ex situ*). In addition, although the absolute number of species in SM3 has only increased by three between 1997 and 1999 (for example), the mean number of species per quadrat has increased by two during the same time period. This is not because the three 'new' species established in a large number of quadrat locations. Many species initially established at low frequency and abundance, but with time representation over all plots of the treatment, as opposed to maintaining localised distributions, has been increasing.

The continuing establishment of species is possible because the sward has remained relatively open, with bare ground present in all years. Egler (1954) first emphasized the consequence of the initial floristic composition for the subsequent composition and diversity of vegetation in secondary succession. Stockey and Hunt (1994) found establishment within the first year of wetland mesocosms was likely to be a precondition for successful establishment in the long term because of the difficulties associated with establishment within a closed turf. If species-rich vegetation is the aim, then the availability of niches for regeneration in the early years of restoration will be essential, despite the likelihood of colonisation by undesirable species. Of course, if dispersal is a limiting factor, bare ground will only be colonised by species already present within the sward or represented in the seed bank and thus total richness may not increase although small-scale diversity may.

The maintenance of bare ground within the experimental field is attributable to a number of factors, including poor initial establishment of sown species, a sub-optimal hydrological regime during the early years (see Chapter 8 for a discussion of hydrology), overgrazing during 1995, and the decline of *Lolium multiflorum*.
The arable reversion appears to have had limited success using criterion III, although SM3 is the most successful treatment.

Criterion IV similarity to NVC target communities
The swards developed from the different seed treatments do not closely resemble any NVC community type, with goodness of fit values ranging from 18-44 (Appendix 5.1). The swards are most commonly assigned to MG6 and MG9, but both the more diverse seed treatments (SM2 and SM3) also have affinities to the target communities. For four out of five survey years, the sward developing in SM3 without the nurse crop has been most similar to MG4, whilst for three of the survey years, SM2 without the nurse crop has approximated to MG8. The goodness of fit values (presented in appendix 5.1) suggest that increasing additions of seed have resulted in more diverse swards that (at this relatively early stage of sward development) display weak similarities to a greater number of the target communities. The poor 'fit' of the restored swards to NVC community types should not necessarily be cause for concern at this stage; the established wet meadows tend to approximate to a number of community types with relatively low goodness of fit values. It should be borne in mind that the classic expression of each NVC community type, as recorded in constancy tables, is itself a distillation of many samples, a considerable number of which will have a composition markedly different to this summary ideal. The poor goodness of fit in the current study may still be within the envelope of MG4, as it is understood (Mountford, pers.comm.). Using this criterion, SM3 and SM2 have been most successful at restoring approximations to NVC target communities. SM3 must be considered more successful, however, since this treatment approximates to the highest priority community of MG4.

Criterion V success of establishment of sown species

Establishment of sown species, both in frequency and cover, was disappointing considering the relatively high seeding rate adopted but has undoubtedly contributed towards the continued development of the vegetation. Stevenson *et al.* (1995) found that higher seeding rates (i.e. 4g m^{-2}) will inhibit weeds more successfully, but will also establish a closed turf sooner than lower sowing rates (e.g. 0.1-0.4g m^{-2}), which can themselves be used to successfully establish chalk grassland vegetation. On this site, it is questionable whether a reduced sowing rate would have been anymore successful.

One concern often voiced about habitat restoration on arable land is the influence of large reserves of undesirable species within the seed bank (Leck *et al.*, 1989; Hutchings & Booth, 1996). Results of the seed bank study (Chapter 4) suggest that the experimental field contains somewhere of the order of 7 000 seeds m^{-2} of Class II species and 300 of Class I species. For comparison, a seeding rate of 40 kg ha^{-1} (4g m^{-2}), with an average of between 2 000 and 6 000 seeds g^{-1}, would result in between 8 000 and 24 000 seeds being sown m^{-2}. The number of seeds of Class II species in the seed bank may therefore be somewhat low, and seedlings might be out competed by the faster growing weedy species present in greater numbers.

Lolium multiflorum dominated the vegetation within the early years, but has generally been declining over time. One of the main benefits of nurse crops is considered to be that as they die out they release sites for establishment of other species (Wells *et al.*, 1989). The present findings appear to concur with this, although early in the experiment the nurse crop did appear generally detrimental, consistently depressing total numbers of species, Class II and Class I species, and small-scale species-richness. With the exception of *Lolium multiflorum* itself, no other species appear to have been benefited from the presence of the nurse crop. The large contribution made to the seed bank by *Lolium multiflorum* negated the need for a nurse crop to be sown at this site. The presence of large numbers of *L.multiflorum* in the seed bank is curious and was unexpected because the rationale behind using this species is that it produces little viable seed and dies out quickly. It appears that, in the experimental site, this species is producing viable (although not persistent) seed. However, if *Lolium* species (from both the nurse crop and seed bank) continues to decline in abundance and release microsites for the establishment of other species, then this may have been a valuable contribution to the restoration. Whilst initial establishment has very much shaped the vegetation that has developed, the continued development of unimproved grassland of conservation value now depends upon propagules of desirable species not already present in the vegetation (or that have been transient in the vegetation) reaching the site and establishing in gaps.

The success of vegetation restoration from seed will be determined partly by the performance of the species sown, i.e. percentage germination and establishment. The lowest mean percentage establishment of sown species was recorded in SM3. This is perhaps to be expected as this mixture deliberately included a wider range of species than either SM1 or SM2. The species that failed to germinate, establish or persist in SM3 were *Filipendula ulmaria*, *Oenanthe fistulosa*, *Rhinanthus minor*, *Sanguisorba officinalis* and *Thalictrum flavum*. *Filipendula ulmaria* also failed to establish in SM2. Two sown species have never been recorded in the vegetation (*T.flavum* and *O.fistulosa*). Seed of both these species was of local provenance, collected within the study area. *O.fistulosa* germinated poorly in the laboratory, whilst *T.flavum* did not germinate. Two important species of the *Alopecurus pratensis- Sanguisorba officinalis* grassland also performed poorly: *S.officinalis* itself and *Filipendula ulmaria*. *S.officinalis* was also collected locally. Both species achieved low percentage germination in the laboratory, and although both have been recorded in the vegetation, individuals have not reached maturity and neither species has persisted. Stevenson *et al.* (1995) found that seed collected from wild populations of chalk grassland species had lower viability than commercial seed, which may at least partly explain the failure of hand-picked species in the present experiment. If viable seed production in these species is consistently low in most years then they are unlikely ever to reappear in a restoration site. Experimental investigation of the responses of key wet grassland plants to differing water levels suggested that obligate wet grassland species may fail to establish from seed even on suitable sites (Mountford *et al.*, 1996c), and concluded that introduction as plug plants was likely to be more effective for species such as *Sanguisorba officinalis*. It was expected that material of local provenance might be pre-adapted to the local environmental conditions and thus be more suitable for use in the restoration than other ecotypes. Indeed, the term 'local provenance' has become a 'buzzword' in ecology, with a plethora of papers in the literature voicing concerns over the use of non-local (or native) genotypes in conservation (e.g. Akeroyd, 1994; Millar &

Libby, 1989; Knapp & Rice, 1996). Whilst the present research cannot resolve the question of the importance of local genetic material in restoration, results suggest that locally adapted populations, i.e. adapted to existing semi-natural wet grasslands, are not adapted to conditions in ex-arable sites. As conditions within the study site 'deteriorate' from arable improvement towards an extensively managed undrained unimproved grassland, conditions may become more suitable for the establishment of locally adapted genotypes and local material may prevail.

Despite the failure of several species to establish, the sowing of increased numbers of species in both SM2 and SM3 has resulted in higher total ground cover of sown species than in SM1, despite the fact that an equal quantity of seed was sown. Following fairly poor initial establishment, the abundance of sown species increased in both SM2 and SM3 between 1997 and 1999 compared with SM1. With a wider range of species then, there will be species that can differentially utilise resources (and regeneration niches), and these are now increasing, whilst the small number of grass species sown in SM1 appear not to be able to benefit from the bare ground present/opportunities for increase.

While SM3 is more species-rich than NR, HB and SM1, the magnitude of difference is relatively small, with plots of SM3 containing on average 10 more Class I species than NR in 1999, despite SM3 receiving seed of 23 species, 18 of which have established. Similarly, SM2 is only on average 6 Class I species richer than NR, although 10 of 11 species sown have established. This further illustrates the relatively poor spatial establishment of sown species and, taking into account the cost of the seed, the restoration has been of limited success using criterion V.

Criterion VI the performance of individual species

Examination of trends in individual species reveals that the majority of significant differences between treatments are for sown species. From these, a number of different groups can be determined:

a Species, mainly grasses, that established reasonably well from seed and by 1999 were increasing, but were generally more abundant where sown. Examples are: *Alopecurus pratensis, Festuca rubra, Cynosurus cristatus, Trisetum flavescens, Hordeum secalinum, Centaurea nigra, Leucanthemum vulgare, Ranunculus acris, Trifolium pratense* and *Vicia cracca*. These species appear to establish relatively easily, increase in frequency and abundance with time, and once established can spread by seed and establish new individuals. They are thus likely to be valuable in restoring grassland to ex-arable land.

b Species establishing from seed but remaining localised (i.e. where sown), at low abundance and frequency, such as *Briza media, Lathyrus pratensis, Lotus corniculatus* and *Silaum silaus.* If seed of these had not been sown, their presence would be unlikely.

c Species sown but not observed in the vegetation, and thus assumed to have not germinated, i.e. *Thalictrum flavum* and *Oenanthe fistulosa.*

d Species present in the early years, which did not persist, e.g. *Sanguisorba officinalis, Filipendula ulmaria* and *Rhinanthus minor. Sanguisorba officinalis* was recorded from 1994 to 1997, although at low (and decreasing) frequencies and abundance where sown.

e Species sown, but probably unnecessarily included in seed mixtures. For example, *Holcus lanatus* has been fairly frequent in all treatments, not only where sown. The high cover in HB in 1999, relative even to treatments where it was sown, suggests that this species was introduced in the hay bales. Presence in other treatments suggests that this species is also effective at dispersal and establishment. The presence of *Lolium multiflorum* in the seed bank negated the need for this species to be sown as a nurse crop. *Phleum bertolonii* replaced *Anthoxanthum odoratum* in SM1, where it has occurred at higher abundance, but this species is not normally a constituent of wet grasslands.

SUMMARY OF TREATMENTS as assessed by criteria I-VI

Natural Regeneration

The sward developed by natural regeneration is relatively species-poor. The total number of species present has consistently been lower in this treatment than elsewhere, although by 1999 (after 6 years) numbers were similar to those in HB and SM1 (criterion I). This treatment resulted in the lowest numbers of Class II and Class I species, although again by 1999 the mean number of Class II species was virtually identical to that recorded for HB and SM1 (criterion II). It also resulted in the lowest abundance of target species. Small-scale species-richness has also been lowest in this treatment, but again by 1999 a similar species-richness was recorded to that in HB (criterion III). The sward resulting from this treatment has most in common with the NVC communities MG6a and MG9a (criterion IV). MG6 is the *Lolium perenne-Cynosurus cristatus* grassland, the widespread 'improved dairying and fattening pasture' on moist, freely draining land, whilst MG9 is *Holcus lanatus-Deschampsia cespitosa* grassland, floristically poor and also of low agricultural value. As no species were sown within this treatment, criterion V does not apply. No individual species performed better in this treatment, and no species was recorded in this treatment only (criterion VI).

Hay Bales

This treatment received seed input from hay bales harvested from the adjacent SSSI. In terms of the mean number of species present (criterion I), and the mean number of Class II species (criterion II), this treatment performed similarly to NR and SM1 in the later years. However, mean numbers of Class I species were higher than in NR (criterion II). Small-scale richness was also similar to that in NR by 1999 (criterion III). This sward has affinities to MG6a, MG9a and MG7 (criterion IV). MG7 grasslands are the *Lolium perenne* leys. These are improved swards, often sown, in lowland river valleys (Rodwell, 1992b), and tend to be characterised by a high representation of grass species. This accords well with the findings of Smith *et al.* (1996) who found that seed of grass species was retained in hay bales and continued to ripen, whilst seeds of forb species were often lost from the hay. Similarly, Wells (1983) found hay bales from Cricklade NNR were dominated by seed of grass species, particularly *Dactylis glomerata, Festuca rubra, Holcus lanatus, Lolium perenne, Poa trivialis* and *Trisetum flavescens.* In addition, bales were extremely variable in terms of their composition depending on the area of origin. In the present study, there was variation between the hay bales sampled in terms of species composition and total numbers of species (Appendix 5.4). Such variability makes it difficult to say with any certainty exactly which species, and in what proportions were introduced (criterion V). However the results do suggest that more species of grass than of forb may have been introduced, e.g. *Holcus lanatus, Hordeum secalinum, Anthoxanthum odoratum* and *Agrostis canina.* In the early years of restoration, plots of this treatment looked more promising than either natural regeneration or the simple grass seed mix. However, with time plots under all three treatments are becoming more similar. Jones *et al.* (1989, 1993) found that using freshly cut hay, rather than dried hay, resulted in the transfer and subsequent establishment of a greater quantity of seed of a larger range of species. It seems likely that the sward derived from either fresh or dried hay would be grass-dominated because grasses tend to be the major seed producers on the donor sites at the time of year of harvest. However, the use of freshly cut hay should prevent the large seed loss that occurs during the drying process and would introduce a greater number of seeds of wild flowers. It is unlikely, however, that the removal of freshly cut hay from nature reserves would be acceptable on a large scale for a number of reasons. Immediate removal of the hay crop reduces the amount of seed likely to be shed onto the donor site. If this happened repeatedly, the vegetation composition of the donor site would inevitably change. Many species characteristic of wet grassland do not possess persistent seed banks and rely on annual recharge of seed for their persistence.

Anthoxanthum odoratum and *Holcus lanatus* both performed well in this treatment. Five species were unique to this treatment (appendix 5.22), although only *Galium verum* could be considered as a grassland species (criterion VI).

Seed Mix 1

This treatment performs similarly to NR and HB in terms of the numbers of species present (criteria I and II). However, by 1999, the sward is closer to that of SM2 in terms of small-scale richness (criterion III). This sward again has affinities to MG6a and MG9a grasslands (criterion IV). All four sown species established in the field (criterion V). Species that have performed well (criterion VI) within this treatment include the sown species of *Alopecurus pratensis*, *Cynosurus cristatus* and *Phleum bertolonii*, *Lolium multiflorum* and the unsown *Agrostis canina* and *Lolium perenne*. Seven species were unique to this treatment (criterion VI (appendix 5.23), although none were target species. It would thus appear that the prescription that this treatment is based upon will be ineffective in achieving the reinstatement of species-rich lowland wet grassland. It might be that there are agronomic benefits of sowing a small number of grass species on isolated restoration sites in terms of production of grass for stock grazing, but in the present experiment abundance of palatable grass species is not significantly higher within this treatment.

Seed Mix 2

This treatment performs better than NR, HB or SM1 in terms of mean total numbers (criterion I), Class II and Class I species (criterion II). Small-scale species richness is also higher in this treatment than in NR, HB or SM1 (criterion III). In terms of the target communities, the best fits are generally to MG6a and MG9a, but this sward does also approximate to MG5a and MG8 particularly (criterion IV). Indeed in 1997 and 1999, the sward developed without the nurse crop was assigned to MG8, with goodness of fit values for MG4 and 5a not dissimilar. Abundance of sown species was higher here than in SM1 in 1999 (criterion V). Species that performed well in this treatment (criterion VI) were generally those that were sown, e.g. *Cynosurus cristatus, Leucanthemum vulgare, Lotus corniculatus* and *Trifolium pratense*. Four species recorded during the monitored period were unique to this treatment (*Juncus conglomeratus, Geum urbanum, Hypochaeris radicata* and *Urtica dioica*).

Seed Mix 3

This treatment has consistently outperformed all other treatments, having higher absolute numbers of species and significantly higher mean numbers of all species (criterion I), including target species (criterion II). Abundance of Class I target species has also been highest in this treatment (criterion II). The sowing of a larger number of species has also resulted in the highest small-scale richness (criterion III). Where the nurse crop was sown, the sward has affinities to MG6a and MG9a; the sward without the nurse crop has been assigned to MG4 each year (except 1995) (criterion IV). This sowing also resulted in higher abundance of sown species than in SM1 in 1999 (criterion V). With hindsight, certain species sown in this treatment would have been better omitted. The more discerning species of wet grassland failed to establish, and would undoubtedly be better introduced at a later date following the establishment of a reasonably diverse grass sward. A large number of species (appendix 5.25) performed best in this treatment, the majority of them sown (criterion VI) and 12 species recorded within the field were unique to this treatment (5 sown) including *Oenanthe silaifolia, Heracleum sphondylium* and *Senecio aquaticus*.

5.6 Conclusions

The sowing of an increased number of species from the target community has been most successful in restoring lowland wet grassland to the study site. The continued increase in numbers of species in all treatments suggests that natural dispersal is contributing propagules to this site. The maintenance of bare ground over the monitored period has undoubtedly played its part in the continued provision of niches for regeneration, as has the decline of *Lolium multiflorum*.

There is now very little difference between the swards developed from NR, HB, SM1 or SM2 in terms of total numbers of species present. Only the most expensive, most diverse seed mixture has resulted in a sward differing from the remaining treatments.

If the only measure of success used were similarity to the target vegetation, then it would be difficult to separate the different treatments, as they are all dissimilar. Evaluation of the restored vegetation according to pre-defined criteria (Chapter 3) allows for an objective assessment of the progress of the vegetation towards the endpoint of the target vegetation. SM3 outperforms all other seed treatments when success is measured using a range of such criteria.

CHAPTER 6 TARGETING RESTORATION IN THE STUDY AREA

6.1 Introduction and aims of this chapter

The setting of targets (objectives, goals) is a necessary part of conservation and ecological restoration (e.g. Treweek and Sheail, 1991; Anderson and Dugger, 1998). Without targets, there is no defined endpoint and so the success of restoration cannot be assessed (Gilbert and Anderson, 1998). This thesis has so far been largely concerned with geographically restricted experimental approaches carried out in an ex-arable field (SA123), but this chapter considers the possibility that there may be other, more suitable sites for restoration, as there may be more suitable reference habitats than that selected for the restoration experiment (Chapter 3). Identification and characterisation of the wet grassland resource is central not only to the definition of floristic targets, but also to the selection of sites to be restored.

The Upper Thames Tributaries ESA was designated because of its wet grassland and wetland habitats, with particular emphasis on the unimproved and extensively managed wet meadows and pastures. The targets for conservation and ecological restoration are thus the characteristic communities of formal lowland floodplain, the wet mesotrophic grasslands. However, when the current study was begun, ESA scheme 'objectives' were too vague to be of use in restoration planning, simply stating that 'the main ecological interest of the area lies in the unimproved and other extensively managed wet meadows and pastures...' (MAFF, 1992). Similarly, the reversion Tier objectives, which were the same for both reversion to wet grassland (Tier 3B) and for reversion to extensive permanent grassland (3A), could not be used to guide restoration: reversion of arable land to wet grassland, by establishing a permanent grass sward within 7 months of the start of the agreement, will 'encourage a gradual recolonisation of the characteristic wildlife of river valley grassland' and 'enhance the river valley grassland landscape' (MAFF, 1992).

Targets for the field experiment (Chapter 3) were developed without clear guidance in terms of local vegetation composition. Whilst the grassland vegetation adjacent to the study site contained elements of the species-rich communities of mesotrophic grassland (MG4, 5 and 8), the sward of the reference habitat did not fit well with any single community description and thus targets were not based upon these National Vegetation Classification (NVC) community types.

The NVC has, however, been widely adopted within Britain as a framework for use in conservation and restoration of vegetation. MG4 grassland has been listed on Annex 1 of the Habitats Directive and MG4, MG5 and MG8 are the focal communities of the Biodiversity Action Plan for lowland meadows (Anon 1998). Since the inception of the field study, the UTT ESA has been recognised as one of the remaining strongholds for MG4 (c. 150 ha) and also as containing important concentrations of MG5 (c. 500 ha) (ADAS, 1998a). This represents approximately 10% of the national resource of MG4 and somewhere of the order of 5-10% of the MG5 resource in England and Wales (Anon, 1998). Therefore these communities ought to be the focus of conservation and restoration in the UTT ESA.

It could therefore be argued that, despite the limitations of the NVC (Chapter 1, 1.7.4), it does enable observed vegetation in any location to be placed in the context of a national classification. Co-occurrence mapping of the constituent species of a community (Chapter 1, 1.7.3) enables determination of the potential geographic range of that community. For example, co-occurrence maps of constituent species of MG4 and MG5a indicate that many species of these communities do occur within the 10km squares containing the UTT ESA and the study area (Figures 6.1-6.2).

The co-occurrence of high numbers of constituent species within a 10km square is no assurance that the community itself is present. Species that apparently co-occur within a 10km square will not necessarily be found together at any site within that square. Species' co-occurrence at the 10km scale merely indicates that the community *could* occur if environmental conditions were suitable. Many species associated with wet grassland also occur in other grassland communities and are thus more widespread than the wet grassland resource itself.

Figure 6.1. Co-occurrence map of NVC community MG4 *Alopecurus pratensis-Sanguisorba officinalis* grassland species.

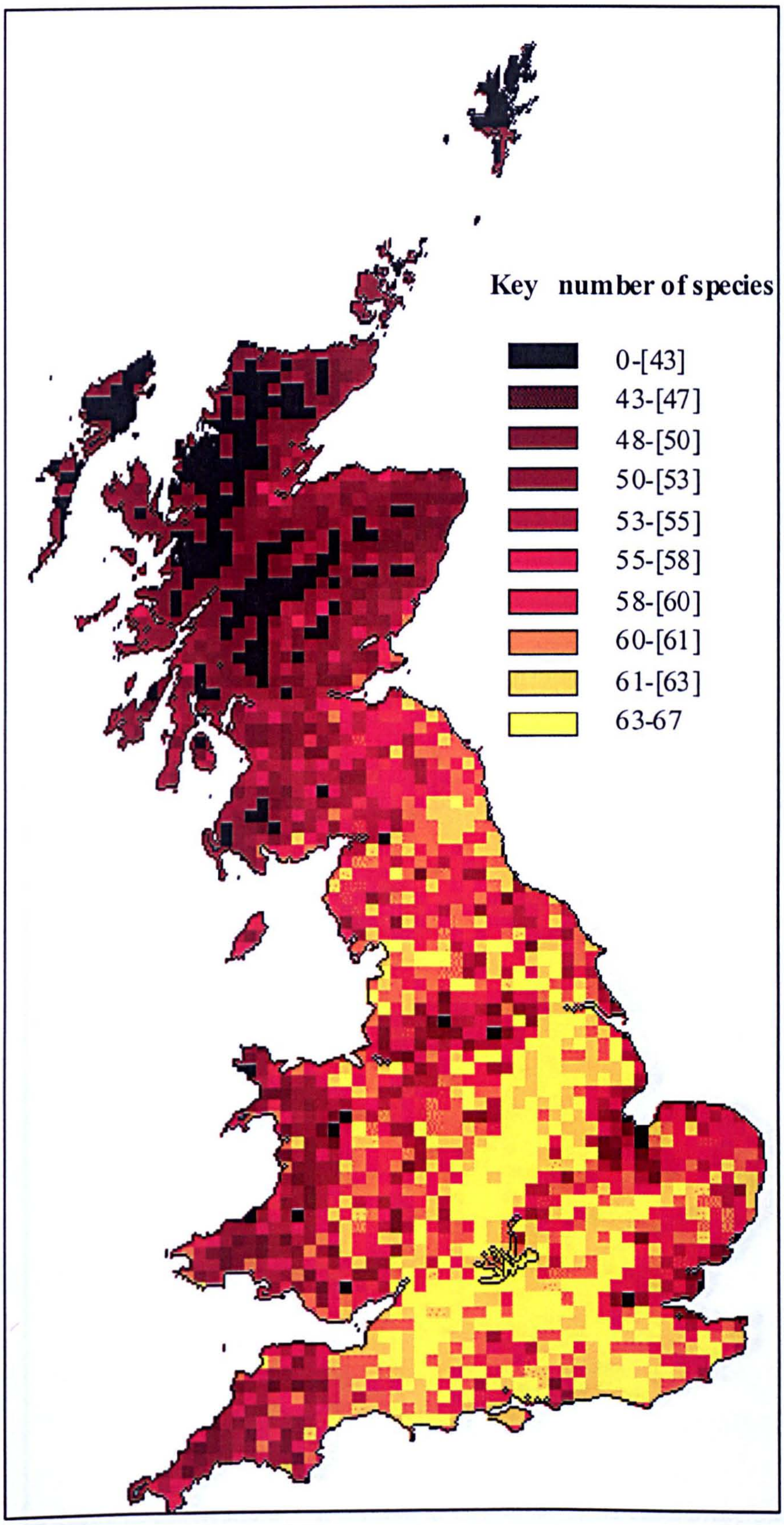
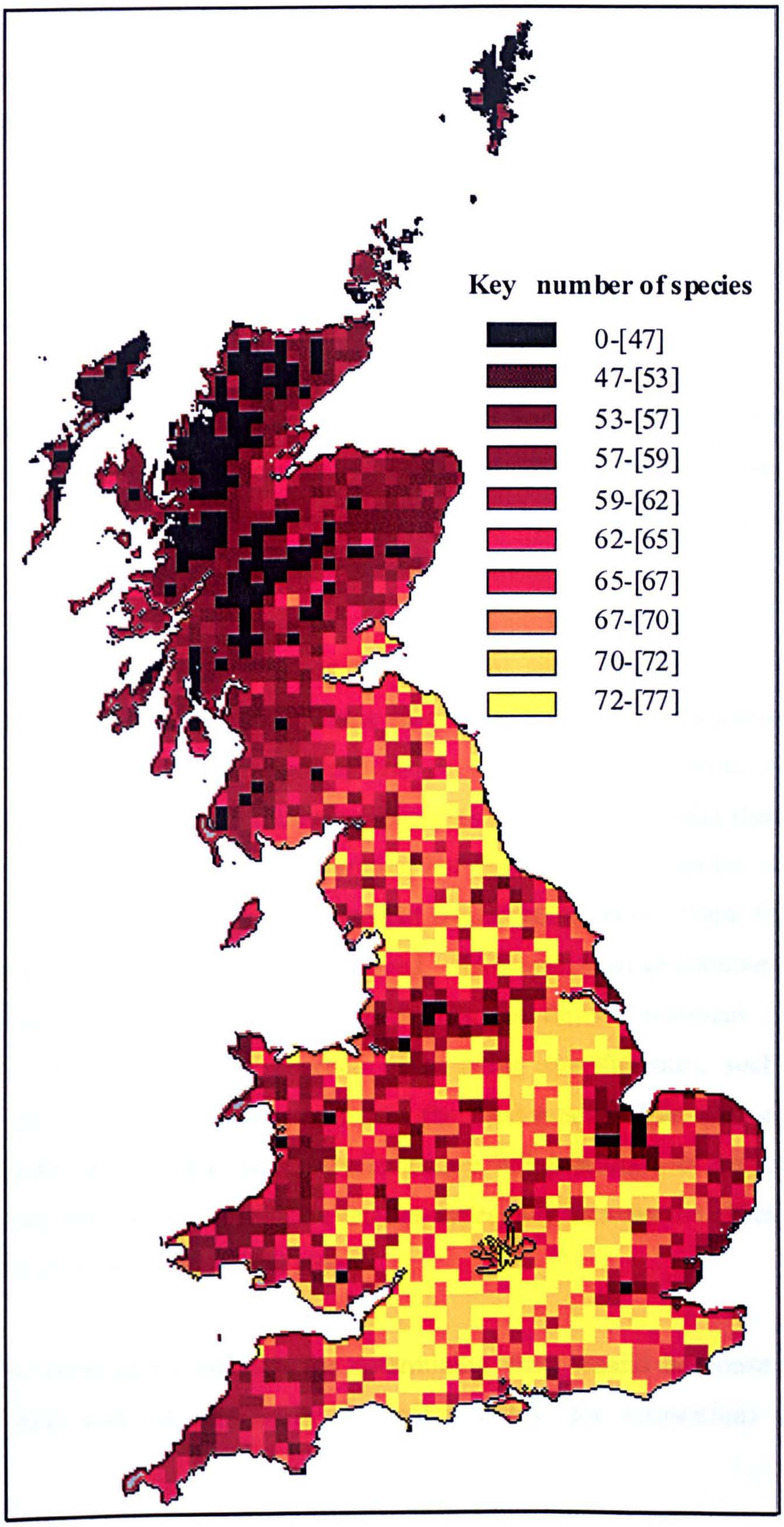


Figure 6.2. Co-occurrence map of NVC community MG5a *Cynosurus cristatus*-*Centaurea nigra* grassland species.



Status statistics (Chapter 1, 1.8.6) can also be used to inform the decision-making process. Co-occurrence mapping in combination with status statistics, using national species distribution data, can be used to indicate where communities have declined. Mountford *et al.* (1997) derived such statistics for NVC communities that are characteristic of lowland wet grassland. Their results suggest that the species complement of MG4 has declined by 1.1% nationally, by 1.5% in southeast England and by only 0.01% in southwest England between 1952-1960 and 1987-1988. These (apparently) small declines are due to the fact that the associated species of this community are more widespread than the community itself. The community has declined but constituent species are still present within many 10km squares surveyed so that the community appears largely unchanged in range.

Mapping community occurrence using 10km data delineates a community's extent of occurrence, but not its area of occupancy. The survey data collected for the study area afford an opportunity to investigate the area of occupancy of the target communities and to derive objective targets appropriate to the whole catchment.

It is generally agreed that restoration will be most successful on sites adjacent to species-rich sources of propagules. These 'species-rich sources' are areas of land that support populations of target species (and communities), and from which new populations may be established following dispersal. As such, it is likely that these sites will not only be source fields for restoration but also the best examples of the target communities and thus the core of grassland of conservation value. These fields should be the absolute minimum that is conserved. However, agri-environment schemes largely promote conservation on an ad-hoc basis, i.e. farmers nominate fields to be entered into schemes. In the absence of targeting specific sites, such voluntary initiatives may either fail to fully represent the biodiversity of an area or require a greater number of sites than necessary to ensure full protection of valued features. Targeting sites that support specific vegetation types may increase the efficiency of the ESA scheme in achieving its stated aims.

This chapter investigates methods for determining what should be conserved in the catchment (the core of grasslands or 'source fields' for restoration) and where expansion could best be achieved (suitable sites for restoration or 'sink fields'). This is achieved by examining:

1. The use of regional survey data in deriving targets in the context of the study area:
   i. GIS is used to investigate the current distribution of the target communities (section 6.3);
   ii. Alternatives to the use of the NVC for determining the 'core' of vegetation of conservation value are examined (section 6.4).
2. Having defined source fields, expansion of the wet grassland resource requires that the most appropriate sink fields be identified (section 6.5). These could be improved grassland or arable sites, but in the context of this study only set-aside fields will be considered as fields entered into set-aside are more likely to become available for nature conservation than are arable fields.

6.2 Datasets used

The two main species distribution datasets for the study area, derived from the results of the botanical surveys (described in Chapter 2), are:

1. COVdata: this dataset contains quantitative information for 178 species (including bryophytes) in 194 fields (within $1m^2$ quadrats, presence and abundance (percentage cover) were recorded). These quantitative data were also classified into NVC community types using Tablefit (NVCdata).
2. PAdata: this is a more comprehensive listing of species within fields: 229 species (including bryophytes) in 212 fields. However, this is a qualitative dataset, indicating presence or absence at the field-scale only.

In addition to the species distribution data, this chapter also draws upon the hydrological model results and management information summarised at the field scale (described in Chapter 2, section 2.4.1).

6.3 Identifying core areas for conservation ('source' fields for restoration) using data local to the study area

6.3.1 Target NVC community types

The area of occupancy of the target communities may be determined using local information collected within the study area and present in the GIS (Chapter 2).

Three different methods for identifying the distribution of the target communities within the study area were compared. Fields were identified as supporting the target communities if they either contained:

a quadrats assigned to the community type by Tablefit, with a goodness-of-fit value $\geq$ 60%.

b $\geq$ 60% of the community constituent species recorded in the study area; or

c $\geq$ 90% of the most constant species (constancy V and IV) of the community, as defined in the published constancy tables of the NVC (Rodwell, 1992b).

6.3.2 Approaches to the selection of source fields

If the nationally defined NVC cannot necessarily be used as a framework for regional and local restoration, then alternative methods for determining the core of vegetation of conservation value may be more appropriate.

6.3.2.1 Reserve selection analyses

Optimal reserve selection algorithms enable the identification of the smallest set of sites that are needed to represent a group of natural features in a given region (Pressey, Possingham and Margules, 1996). For the purposes of this study, the natural features in question are plant species or NVC community types and the region is the River Ray study area. A reserve selection algorithm (using nonlinear optimization code) was used to identify the smallest number of fields required to represent *all* species or communities. The data matrix (species versus fields) was exported from the GIS into a Microsoft Excel spreadsheet for the calculations. The algorithm (Microsoft Excel

Solver) uses the simplex method with bounds on the variables, and the branch-and-bound method. The optimisation was performed on three sets of data for the study area: PAdata and COVdata (both minus bryophytes and arable weeds) (**PAarea** and **COVarea** respectively), and NVCdata (33 community types were recorded with a goodness-of-fit over 60%) (**NVC60**).

6.3.2.2 Alternative selections

The performance of the 'reserve selections' was compared with a number of alternative field selections, each arbitrarily of 20 fields, that best fulfil the following criteria:

1. On the basis of species-richness.
   i. The target communities are all species-rich. Select fields on the basis of *the total number of species present*, i.e. fields that contain the greatest number of species from the complete qualitative dataset (**PA20**) data were compared.
   ii. Target vegetation is species-rich at a small-scale. Select fields with the highest number of species m^{-2}. Simpson's Index of Diversity was calculated for all quadrats and the 20 fields containing the highest ranked quadrats were selected (**divind**). Simpson's Index is calculated as follows: $D = 1 - \sum(n/N)^2$ where n is the number of individuals of a particular species and N is the total number of individuals. Simpson's Index of Diversity = 1 – D.
2. Target community types (**MG4com**, **MG5com**: composites). Select fields according to the communities they contain. This could be done either on the basis of the greatest number of constituent species present or, using TABLEFIT, by selecting fields that contain quadrats assigned to the community type. In practice, fields were selected as containing the target community types if they contained quadrats assigned to that community type (with goodness of fit > 60%) OR if they contained > 60% of the community constituent species recorded within the catchment. This was performed for MG4 and MG5, as these are the main target communities.

6.3.3 Results - Selecting source fields

6.3.3.1 Target NVC community types

The area of occupancy (spatial distribution) appears to differ depending upon which method is used (figures 6.3-6.5). Relatively few quadrats are actually assigned to MG4 and MG5a (with a goodness of fit value of at least 60%) and so this method suggests few fields support these target vegetation types (Figure 6.Xa). The greatest number of fields is selected when distribution is mapped using the co-occurrence of the most constant species of the community type as a guide (Figure 6.Xc). In addition, several fields are common to all three of the target communities defined this way. This is because many species of high constancy are common to more than one community and also a number of the higher constancy species are generally ubiquitous throughout the study area, e.g. *Holcus lanatus, Lolium perenne, Poa trivialis* and *Ranunculus repens* all occur in over 90% of fields.

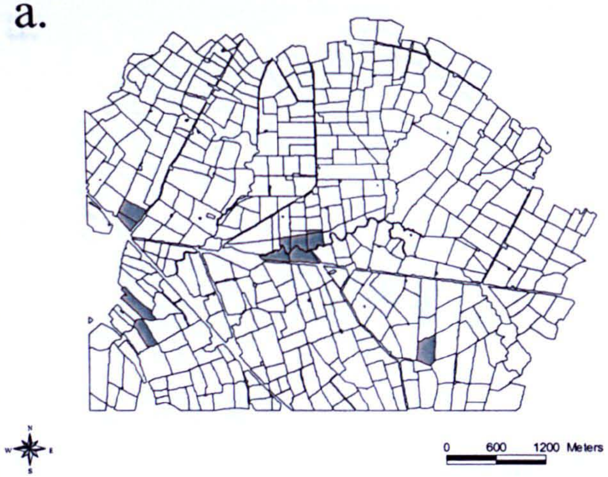
6.3.3.2 Alternative selection of source fields

Results are presented in terms of: the number of fields, area, number of species, and efficiency of selections (Table 6.1), and the species protected and their frequency (Table 6.2 and Appendix 6.1). The spatial placement of fields selected within PA20, Divind, NVC60 and PAarea (Figure 6.6) are presented for comparison with the target community composite distributions of MG4com and MG5com (figures 6.7a, b). Fields within the ESA scheme (**ESA**) are used as a baseline for comparison (figure 6.8).

Table 6.1 Efficiency of species representation by the source field selections

	PA20	divind	MG4com	MG5com	PAarea	COVarea	NVC60	ESA
Species count	187	145	137	111	212	187	141	151
No.fields	20	20	17	10	23	19	27	47
Area (ha)	129	109	98	65	133	97	162	224
Area (% of total)	11	10	9	6	12	9	14	20
Species field^{-1}	9.4	7.3	8.1	11.1	9.2	9.8	5.2	3.2
Species ha^{-1}	1.5	1.3	1.4	1.7	1.6	1.9	0.9	0.7
Species (% of total)	82	63	60	49	93	82	62	66

a.



b.

c.

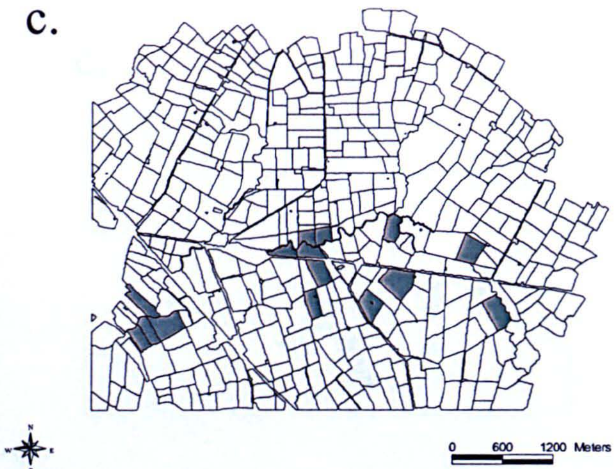


Figure 6.3 Legend

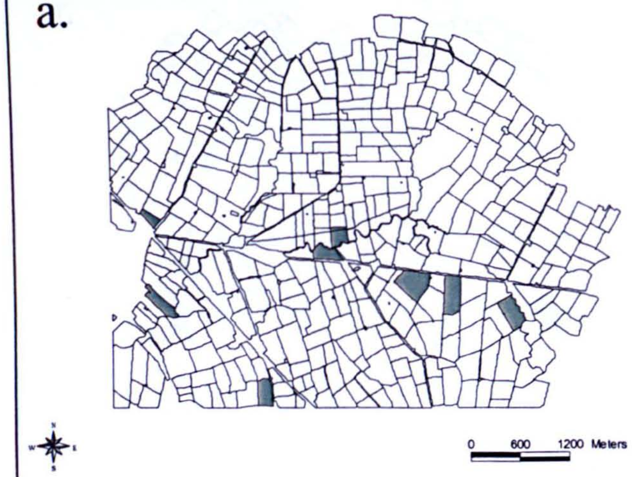
MG4 *Alopecurus pratensis-Sanguisorba officinalis* grassland

a. **Fields containing quadrats assigned to target community MG4 (goodness of fit > 60%)**

b. **Fields containing over 60% of MG4 constituent species present within the catchment**

c. **Fields containing >= 90% of the higher constancy (IV or V) species of MG4**

a.



b.

c.

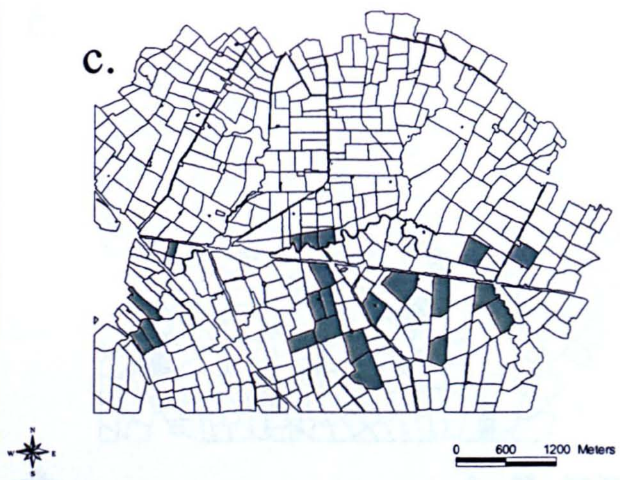


Figure 6.4 Legend

MG5 *Cynosurus cristatus-Centaurea nigra*

a. **Fields containing quadrats assigned to MG5 (goodness of fit > 60%)**

b. **Fields containing > 60% of constituent species of MG5 present in the catchment**

c. **Fields containing >= 90% of the higher constancy (IV or V) species of MG5**

a.

b.

c.

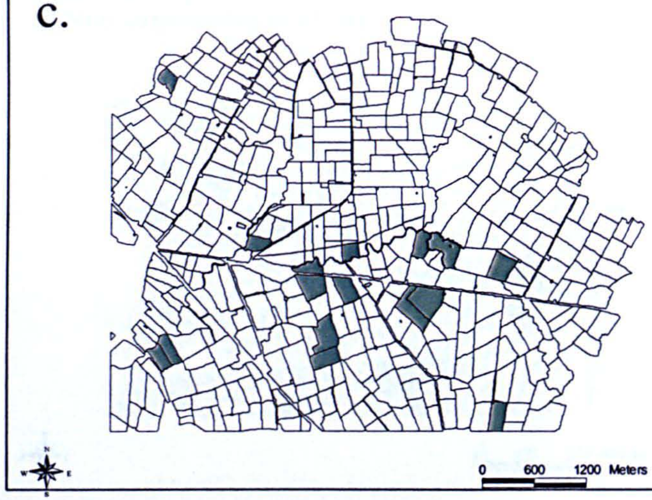


Figure 6.5 Legend

MG8: *Caltha palustris-Cynosurus cristatus* grassland

a. **Fields containing quadrats assigned to MG8 (goodness of fit > 60%)**

b. **Fields containing > 60% of MG8 constituent species present within the catchment**

c. **Fields containing >= 90% of the higher constancy (IV or V) species of MG8**

Figure 6.6a The 20 richest fields (PA20)

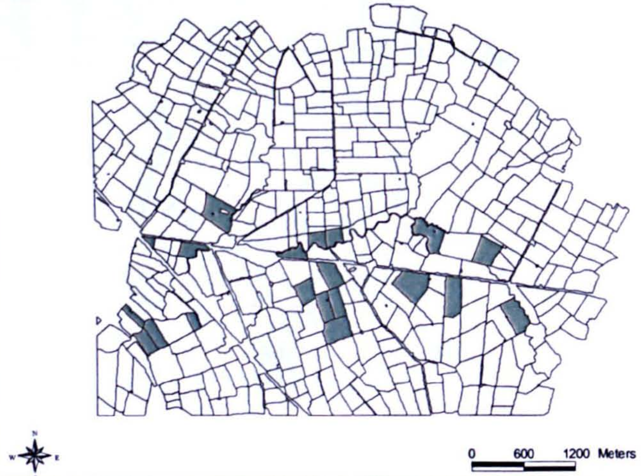
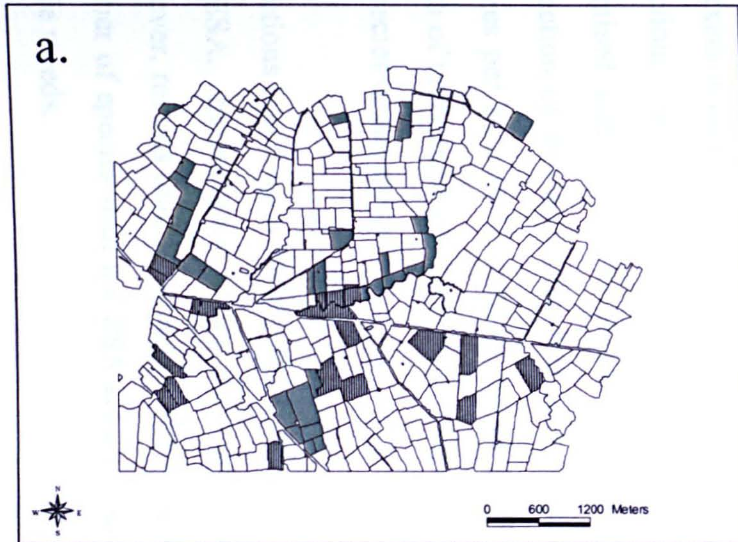


Figure 6.6b Quadrat diversity index (Divind)

Figure 6.6c Optimal selection of fields to represent all NVC communities (NVC60)

Figure 6.6d Optimal selection of fields to represent all species at least once (PAarea)



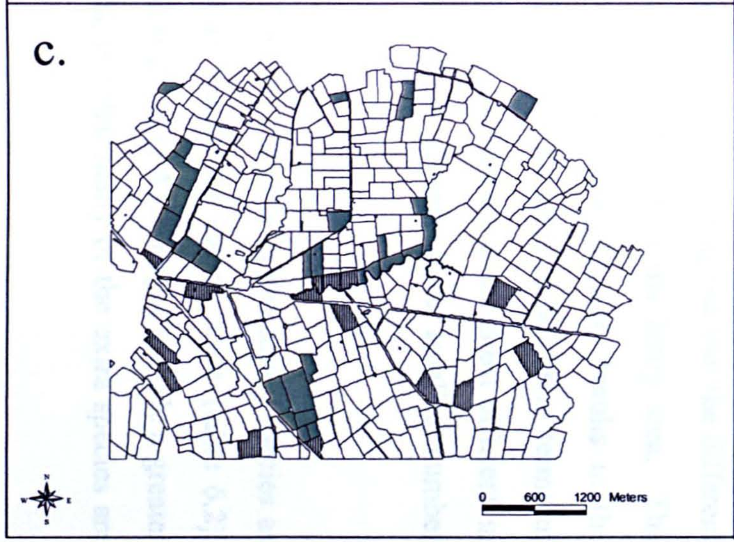


Figure 6.7 Legend

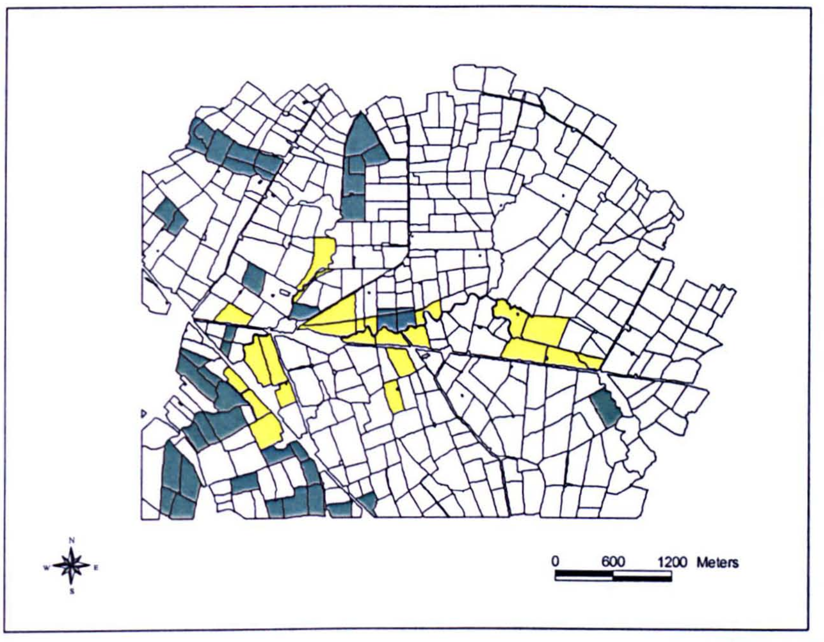
Composite maps of target community 'source' fields with set aside fields

a. *Alopecurus pratensis-Sanguisorba officinalis* (MG4)

b. *Centaurea nigra-Cynosurus cristatus* (MG5)

c. *Caltha palustris-Cynosurus cristatus* (MG8)

Figure 6.8 Fields within the ESA (green) and Countryside Stewardship (yellow) schemes



An examination of the summary efficiency results (table 6.1) suggests that the different selections vary in their efficiency at protecting species within the study area. The optimised selection of fields based on the qualitative data (PAarea) results in the protection of the largest number of species, but is not the most efficient in terms of species per field (MG5com) or per ha (COVarea). MG5com is the most efficient in terms of representing species per unit area, but actually represents the smallest number of species of any selection.

Selections based upon species-richness are at least as effective in protecting species as the ESA. Examination of the summary species results for source areas (table 6.2), however, reveals that, for example, the 20 richest fields (PA20) might include a greater number of species than the ESA scheme fields, but that many of the extra species are arable weeds.

Table 6.2 Summary of species protected by the various reserve selection methods

Selection	Total number	Grassland	Arable	Wetland	Grassland/ Wetland
PA20	187	122	51	42	33
DivInd	145	113	20	37	29
MG4com	137	112	21	32	28
MG5com	111	96	14	26	23
PAarea	212	138	56	45	35
COVarea	187	125	47	41	32
ESA	151	120	26	37	31
NVC60	141	112	21	34	31

Within any selection, the most frequent species in the study area that are not represented are arable weeds (Appendix 6.1).

Examination of the maps of selected fields reveals that several fields are common to all the target community composite selections (figures 6.7a-c), and a number of fields are common to two but not three of the communities. PA20 (figure 6.6a) and Divind (figure 6.6b) also have a large number of fields in common with the composite selections.

6.3.4 *Discussion - Selecting source fields*

The aim of this part of the study was to identify those fields within the study area that contained the grassland resource of highest conservation value. This 'core' of grassland should be the focus for conservation effort in the area, representing the very minimum that should be protected to ensure the survival of the best of the wet grassland resource. Rehabilitation of degraded grasslands and grassland re-creation on ex-arable land should be targeted to buffer and expand the core resource.

The identification of the minimum to be protected was approached using methods based on the NVC, species-richness and reserve selection.

Target communities

Whilst the target communities for conservation and restoration are MG4, 5 and 8, it is difficult to identify distinct source areas for each, as they do not necessarily occur as distinct entities within the study area. Relatively few fields contain quadrats that approximate well to MG4 and MG5 particularly, and those quadrats that do often co-occur within the same fields. Widening the method of definition of the target communities to include fields that contain a large number of the community constituent species again results in similar distributions and further emphasises the similarities between these communities.

Although MG5 is not confined to the floodplain (unlike MG4), fields that contain the highest number of constituent species of either community are common to both community types. This suggests that much of the MG5 vegetation does not occur in areas optimal for the community, instead developing in higher lying areas within the floodplain such as the ridges of ridge-and-furrow fields. This is borne out by the fact that fields are comprised of a number of community types (chapter 2, section 2.3.2). Indeed, large stands of lowland wet grassland vegetation tend to be comprised of a matrix of different communities, reflecting differences in the water table regime at the within-field scale (Gowing and Spoor, 1998).

For each community, the fields identified as containing quadrats assigned with goodness of fit > 60% were combined with those fields containing greater than 60% of the community constituent species as present within the study area. This composite distribution thus includes all areas that do support the target community or that have the potential to, based on the presence of constituent species.

Species-richness

The overlap between the most species-rich fields (figure 6.6a) and those containing the target communities (figure 6.7) is unsurprising since the target vegetation is species-rich. What is surprising though is that there is not more overlap. Species-rich fields that are not target community fields may be candidates for rehabilitation, e.g. they contain swards that are poor approximations to target community types. Fields with a large total number of species unevenly distributed could be managed to increase sward diversity.

Several of the most species-rich fields are some of the smallest in the study area. Larger fields might potentially be expected to contain more environmental variation and thus more species than smaller ones and, if this were so, could be selected as source fields for restoration. Presumably this would be the case if there were not overriding factors at work such as the increased ease of agricultural operations within larger fields. Larger fields do not necessarily contain more species, and certainly do not contain more species per unit area.

Reserve selection algorithms

Analyses were performed on both quantitative and qualitative data for the purposes of comparison. The quantitative dataset (COVarea) is less species-rich than the qualitative dataset (PAarea) as it represents, for each field, the mean of 5 (or 6) $1m^2$ quadrats rather than a complete record of all species present. For the purposes of selecting the minimum resource necessary to protect all species, the use of the quantitative data appears less efficient as it results in the protection of fewer species overall. However, it is a smaller dataset in terms of the number of species – the analysis was performed to select the minimum number of fields required to represent 117 species at least once, compared to the 139 species in the analysis of the presence-absence dataset. In fact, both the optimised selections are more efficient than most of the selections, in terms of the number of species per unit area.

The reserve selection procedure used here was very simple; those used in conservation often include a constraint that areas selected are close to one another. However, entry into the ESA scheme does not operate on a proximity basis. The target community distributions are not continuous and so none of the selections derived here required fields to be close to one another.

The minimum set of fields required to protect at least one example of each NVC community type (NVC60) is interesting. The fields are fairly remote from one another and many are high lying, out of the floodplain. This reflects the fact that mesotrophic grasslands range from inundated, unimproved swards to drier, agriculturally improved grasslands. Whilst this may represent the smallest number of fields necessary to preserve all community types, it is not efficient at species protection, representing only 62% of species. It also protects communities that are more characteristic of improved agriculture (e.g. MG6, MG7).

ESA fields

The current ESA coverage includes many more fields than any of the source field selections identified. Although not all the selections contain the same number of fields, the number of species protected is not dissimilar to that within ESA fields. The 20 richest fields protect 20% more species in less than half the number of fields present within the ESA scheme. *However, the aim of the ESA is not to represent as many species as possible but to protect the wet meadows and pastures.* The fields currently within the ESA scheme do contain more of the MG4 and MG8 grassland resource than of MG5 (57% of the fields that contain quadrats assigned to MG4 are within the ESA; 52% for MG8, 33% for MG5). This perhaps reflects the fact that the ESA aims to conserve the wet meadow resource rather than the drier grassland types, although it is more likely that fields more marginal for agriculture are entered into the scheme and that these fields contain MG4 and MG8. The ESA scheme is generally inefficient in protecting species, however, despite the fact that many of the 'better' areas of vegetation are within the scheme. One factor in favour of the current ESA coverage is that fields within the scheme tend to be in contiguous blocks, and although ESA fields do not necessarily link the blocks, many are linked by Countryside Stewardship Scheme (CSS) fields, which are themselves often contiguous (figure 6.8).

6.4 Identification of suitable sites to be restored ('sink' fields)

As there may be more suitable source fields than that used for the restoration experiment, similarly there may be more suitable ex-arable fields to be restored. These may be fields with a greater potential for reversion to the target community, or that better increase the overall viability of the target communities within the study area.

The major constraints on ecological restoration are: biological (propagule availability), chemical (soil nutrients) and physical (water availability).

Results of experimental studies (chapters 4, 5) suggest that the choice of site to restore may govern the level of success of ecological restoration. The seed banks of fields cultivated as arable land are generally unsuitable for the restoration of all species of a community. Even the seed banks beneath grasslands are tremendously variable, both within and between fields. Thus *in situ* sources of propagules will generally be too depauperate to be used as the sole source of colonisation. In the absence of a suitable soil seed bank, establishment of 'target' species is dependent upon immigration (and establishment) of propagules from other areas (chapter 1, section 1.5.1.2). Despite the poor dispersal, germination and establishment capabilities of many species, the importance of seed dispersal to reverting grasslands (and hence the value of areas of contiguous grassland and vegetation gaps for germination) cannot be overestimated.

The review of the literature suggests that the issue of soil nutrient availability may be less important for the mesotrophic grasslands than for other community types (chapter 1, section 1.5.1.1). However, an appropriate hydrological regime will be essential to the maintenance of wet grassland flora (chapter 1, section 1.5.1.2).

Assessment of site suitability in terms of the constraints of propagule and water availability should enable the prioritisation of set aside fields according to their ease of restorability.

6.4.1 Method

Potential restoration sites were determined using the following approaches:

1. Sites in close proximity to source fields are likely to be most suitable for restoration. These fields were visually identified from the maps of 'source' field distribution. If restored, such sites should buffer (and aid expansion of) the valued grassland resource. For the purposes of this section, NVC target community source areas (figure 6.7a-c) were defined as fields that contain quadrats assigned to the community with goodness of fit > 60% OR that contain ≥ 60% of the community constituent species present within the study area. Fields with large numbers of constituent species were included because, although they do not necessarily approximate at the m^2 scale to the community, they contain many of the 'building blocks' (species) necessary to assemble the community.
2. There are 22 ex-arable (set-aside) fields within the catchment (figure 6.9). The potential for natural regeneration of each was determined by considering: surrounding land use (grassland or arable), NVC communities present in adjacent grasslands and propagule availability issues (table 6.3). Propagule availability was assessed by investigation of the species present in grassland fields surrounding each set-aside field. Dispersal distances are not known for the majority of species, but in general most seed disperses close to the parent plant and so, although some seeds do disperse long distances, for the purposes of this study it is assumed that only species in the immediate area (i.e. adjacent fields) are available to colonise naturally. Species lists for surrounding grasslands were pooled to determine the total number of species that could potentially colonise each set-aside field. Species were also classified as those of 'grassland', 'grassland and wetland' or 'wetland'. Analysis of variance was used to test differences between set asides in terms of the mean number of species, and the mean species-richness of swards, in the immediate vicinity.
3. The characteristics of grassland fields that support target vegetation can be used to develop criteria for selecting those sites most likely to be successfully restored. Target communities were characterised according to:
   - the management of fields that support the target vegetation;
   - the modelled hydrological regime of fields containing the communities (chapter 2, section 2.4.1). The modelled hydrological regime of any field in

the study area can be compared to those that support the target communities in order to judge the suitability of a particular field for restoration to wet grassland. The hydrological regimes of those set-asides in proximity to source areas, as identified from the composite source field maps for the target communities, were compared with the target community hydrology (appendix 6.3). Because there is variation between fields that contain the target communities, for the purposes of comparison the minimum and maximum values recorded were used. If the hydrological regimes of the set aside fields fell within the upper and lower bounds for the target community, then the set aside was considered potentially suitable.); and

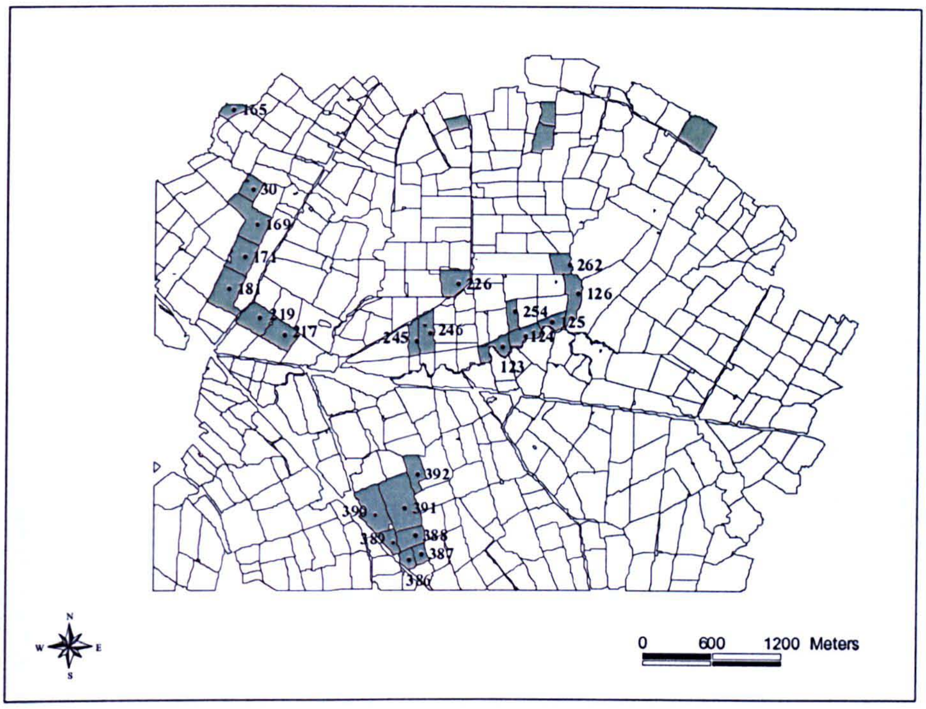
- Community indicator Scores. Mean indicator values (mF, mN and mean number of species, mS) were derived for individual quadrats by averaging the Ellenberg F and N values of all species recorded in the quadrat, or by summing the number of species present. Community indicator Scores were calculated by taking the mean of all quadrats assigned to each community.

6.4.2 Results

6.4.2.1 Ex-arable fields in proximity to source fields

Set-aside fields (SA) adjacent to target community source fields can be identified from figures 6.9 and 6.7a-c. SA123 appears suitable for restoration using most of the source field selections. This is wholly because field 122 (the SSSI) is the most species-rich grassland in the catchment (total numbers of species, small-scale species-richness) with a sward that approximates to a number of NVC community types. SA246 also appears suitable if source fields are those identified by composite target community distributions for MG4 and MG8. SA392 is adjacent to a grassland that contains greater than 60% of constituent species of MG4 and >90% of constancy IV and V species of MG5, although the sward of this grassland does not contain quadrats that approximate to the target communities, being species poor and instead approximating to the *Lolium perenne* leys (MG7) and to the *Holcus lanatus-Deschampsia cespitosa* (MG9) grasslands that occur in moist hollows within pastures and are also characteristic of agricultural abandonment (Rodwell, 1992b).

Figure 6.9 Set-aside fields



6.4.2.2 Set-aside fields with the greatest potential for natural regeneration

Some set-aside fields can be ruled out as contenders for restoration due to their relative isolation, e.g. SA254 and SA262, which are surrounded by arable land or improved grassland (table 6.3).

Examination of the species present in the vicinity of set-aside fields suggests that the study site (SA123) is in one of the most species-rich areas (table 6.3). It could potentially receive propagules of a greater number of species than other set-aside fields, followed by SA217, SA246 and SA245. Classification of species by broad habitat indicates that SA123 (then SA246 and SA217) has the potential to receive the greatest number of grassland species and wet grassland species at the wetter end of the spectrum (occur in both grassland and wetland). SA123, SA246 and SA245 are also in the most species-rich area to receive propagules of all the target communities (table 6.3).

There were no significant differences between the mean total number of species in fields surrounding set-aside fields, but the swards in proximity to SA123 are consistently richer at the m^{-2} scale ($P < 0.001$) (table 6.3).

Table 6.3 Set aside fields.
Surrounding land use. **G1**: extensive grass; **G2**: intensive grass; **NR**: nature reserve; **CSS**: Countryside Stewardship; **SA**: set aside; **WW**: winter wheat; **WB**: winter barley; **OSR**: oil seed rape.

	Surrounding land use								Number of quadrats	Major NVC communities in
Field	**G1**	**G2**	**NR**	**CSS**	**SA**	**WW**	**WB**	**OSR**	**(G1, NR or CSS)**	**surrounding fields**
30	4				1				3 G1 (15)	MG6, MG9, MG10, MG11
169	3	2			2				2 G1 (10)	MG7, MG10
123	3		1	1	2	2			3 G1, 1 NR, 1 CSS (21)	MG4, MG7, MG11, MG8, MG9, MG10
124	2				3	3			2 G1 (9)	MG11, MG6
125	2	1			2	2			2 G1 (11)	MG7, MG6, MG9
126	1	1			2	3			1 G1 (5)	MG6, MG11, MG9
171	1	1			2	1			1 G1 (5)	MG7
181	3				2	1			2 G1 (10)	MG7, MG4, MG8
217	2			1	1	1			2 G1, 1 CSS (13)	MG4, MG10, MG13, MG6, MG7
219	4			1	2	1			4 G1, 1 CSS (24)	MG7, MG4, MG6, MG10, MG8
226	3					2	1	2	2 G1 (10)	MG9, MG11, MG6
245				4	1	1	1		4 CSS (20)	MG7, MG6, MG4
246	3		1	2	1		1		3 G1, 2 CSS, 1 NR (36)	MG6, MG9, MG8, MG11, MG7, MG4
254		1			2	3			-	-
262					1	4			-	-
386	3				3				3 G1 (15)	MG7, MG8
387	2				2				2 G1 (10)	MG6, MG7, MG8, MG9, MG11, MG5
388	2				4				2 G1 (10)	MG7, MG6, MG11, MG5
389	3				4				3 G1 (15)	MG7, MG6, MG10, MG11, MG5, MG9
390	2	1			2				2 G1 (10)	MG11, MG6, MG10, MG7
391	2	2			4				2 G1 (10)	MG7, MG9, MG6, MG11
392	2	2			1				2 G1 (9)	MG9, MG6, MG10, MG11

Table 6.3 (continued). **Absolute number of species**: GW grassland and wetland; G grassland; W wetland; **Number of NVC species**: number of species (as listed in NVC constancy tables) in surrounding fields; **Mean number of species**: in surrounding fields or small-scale (m^2).

	Absolute species numbers				Number of NVC species			mean number of species	
Field	**Total**	**GW**	**G**	**W**	**MG4**	**MG5**	**MG8**	**field (± S.E.)**	**m^2 (± S.E.)**
30	79	13	60	16	37	38	29	39.33 ± 6.67	10.73 ± 0.63
169	55	5	38	6	23	22	17	25.67 ± 10.65	7.54 ± 0.69
123	123	25	94	27	53	54	46	35.80 ± 11.41	16.91 ± 1.79
124	69	12	48	13	30	29	24	29.00 ± 3.89	9.06 ± 0.69
125	57	13	52	15	40	36	29	40.00 ± 2.00	13.36 ± 0.87
126	42	10	35	12	31	27	25	-	15.60 ± 1.29
171	10	0	8	0	4	4	3	-	6.20 ± 0.49
181	38	4	35	5	29	28	21	23.00 ± 13.00	12.36 ± 1.87
217	117	17	71	21	35	41	30	48.75 ± 6.52	12.57 ± 1.12
219	76	12	61	14	40	40	30	33.00 ± 5.63	11.72 ± 1.12
226	37	4	34	5	28	27	20	29.00 ± 5.00	12.00 ± 0.89
245	84	14	67	15	42	41	37	46.20 ± 1.99	11.55 ± 0.71
246	89	19	77	21	46	48	41	41.86 ± 4.90	14.70 ± 0.60
254	-	-	-	-	-	-	-	-	-
262	-	-	-	-	-	-	-	-	-
386	42	3	37	4	30	31	22	28.33 ± 2.91	11.40 ± 0.59
387	59	3	52	4	40	40	28	36.00 ± 1.73	12.90 ± 0.69
388	67	10	58	11	39	40	29	37.67 ± 0.88	11.39 ± 0.75
389	51	6	45	7	37	38	27	31.67 ± 5.93	12.20 ± 1.40
390	35	7	33	8	24	23	16	27.50 ± 1.50	9.40 ± 0.78
391	60	14	53	16	35	35	32	42.00 ± 4.00	12.44 ± 1.12
392	59	11	54	12	41	40	33	49.00 ± 3.00	13.00 ± 1.35

Table 6.4 Composition of set aside fields: total numbers of species present; numbers of species that occur in grassland (grsld), wetland (wtld), grassland and wetland, or arable habitats.

Field	Total	Grsld	Wtld	grsld+wtld	Arable
SA30	21	16	5	4	8
SA123	57	26	11	7	33
SA124	60	33	12	9	28
SA125	44	27	9	8	22
SA126	55	28	9	7	33
SA171	33	20	11	5	8
SA217	47	37	6	4	9
SA219	50	35	9	7	12
SA245	39	36	7	6	6
SA246	51	47	8	7	8
SA386	39	34	4	3	7

Examination of the species composition of the set-aside fields reveals that SA124, SA123 and SA126 contained the greatest number of species at the time of survey (table 6.4), but species typical of arable habitats made up roughly half of the total number. Approximately 90% of species found within SA245, SA246 and SA286 were typical of grasslands. However, species of wetland (and those that occur in both grassland and wetland) are represented in greater numbers in SA123 and SA124.

6.4.2.3 Conditions that support the target community types

Management: However the target communities are defined, the management regime that sustains them is similar (Table 6.5). For comparison between communities, the regime adopted was that of fields containing quadrats assigned to the community, since even newly reseeded grasslands may have many of the species present, but are unlikely to closely resemble unimproved grassland swards. The target communities cannot be separated on the basis of management alone: all three developed under traditional extensive management. In this study, areas that support the target vegetation are grazed, both by cattle and sheep, although cattle are the predominant grazers. Swards are not cut for silage and do not receive applications of slurry or herbicide. They are permanent grasslands (not reseeded), with a stocking rate lower than 1.25 livestock units, and generally do not receive high rates of nitrogen application and are not drained.

Table 6.5 Summary of management regime of fields supporting target communities. 'Quadrat': fields containing vegetation assigned to the community type by Tablefit with a goodness of fit over 60%. 'Constit': fields containing >60% of constituent species of the community (from the NVC constancy tables). 'VandIV': fields containing species of constancy IV or V (for MG4, 14-16 species; MG5a, 13; MG8, 10-11). Values for stocking rate, altitude and flood refer to the mean ± standard error (the range in parentheses).

Community	mown	Silaged	grazed	sheep	cattle	Stocking rate	Altitude (m)	flood	drained	reseed	N	Pk	Fym	Slurry	Herbicide
MG4															
Quadrat	1,0	0	1	1,0	1,0	0.5 ± 0.2 (0-1.0)	61.7 ± 0.3 (61.2-63.0)	2.7 ± 0.7 (0-4)	1,0	0	1,0	0	1,0	0	0
Constit	1,0	0	1	1,0	1,0	0.7 ± 0.2 (0-2.5)	62.5 ± 0.4 (61-65)	2.4 ± 0.4 (0-4)	1,0	0	1,0	1,0	1,0	0	0
VandIV	1,0	0	1	1,0	1,0	0.7 ± 0.2 (0-2.5)	62.2 ± 0.3 (61-64)	2.8 ± 0.4 (0-4)	0	0	1,0	1,0	1,0	0	1,0
MG5															
Quadrat	1,0	0	1	1,0	1,0	0.8 ± 0.1 (0.6-1.0)	63.4 ± 0.6 (61.5-65.0)	1.0 ± 0.6 (0-3)	1,0	0	1,0	0	1,0	0	0
Constit	1	0	1	1,0	1	1.0 ± 0.4 (0-2.5)	63.6 ± 0.8 (61.2-67)	1.7 ± 0.7 (0-4)	1,0	0	1,0	1,0	1,0	0	0
VandIV	1,0	0	1	1,0	1,0	0.7 ± 0.3 (0-2.5)	62.8 ± 0.5 (61-65)	2.3 ± 0.5 (0-4)	1,0	0	1,0	1,0	1,0	0	0
MG8															
Quadrat	1,0	0	1	1,0	1,0	0.9 ± 0.2 (0-2.5)	63.1 ± 0.6 (61-68)	1.0 ± 0.5 (0-4)	1,0	0	1,0	1,0	1,0	0	0
Constit	1, 0	0	1	1,0	1	0.3 ± 0.1 (0-0.5)	62.0 ± 0.5 (61-63.5)	3.0 ± 0.7 (0-4)	0	0	0	0	0	0	0
VandIV	1,0	0	1	1,0	1,0	0.8 ± 0.2 (0-2.5)	62.4 ± 0.4 (61-67)	2.4 ± 0.4 (0-4)	1,0	0	1,0	1,0	1,0	0	1,0

Hydrology: It might be expected that the target communities would be differentiated on the basis of their hydrological regime (see chapter 2), particularly MG5 from the wet meadows. In fact, the water regimes (characterised at the field-scale) that support the target communities are similar (figure 6.10) although the supplementary information on field altitude and subjective classification of flooding frequency does indicate that MG4 occurs at lower altitudes and floods more frequently than MG5.

Only SA123 (out of SA217, SA386-SA389, SA391, SA392, SA245 and SA246) experienced a water regime that apparently could not maintain the target vegetation. Within SA123, the depth to the water table is consistently greater and the number of days flooded fewer than are necessary for the survival of wet grassland.

Mean indicator values

The target communities can be distinguished from one another in terms of the mean number of species m^{-2} and also by the community Ellenberg indicator Scores of mean Nitrogen (mN) and mean soil moisture (mF) values (Table 6.6). MG4 and MG5 are the most species-rich, with the lowest mN values, whilst MG5 has the lowest mF value.

Table 6.6 Summary of mean species-richness (mS), mmN and mmF for NVC communities (derived from quadrats assigned to community with goodness-of-fit > 60%).

Community	mS ± s.e.	mmF ± s.e.	mmN ± s.e.
MG 4	21.3 ± 1.27	5.92 ± 0.09	5.08 ± 0.11
MG 5	22.9 ± 0.77	5.39 ± 0.13	4.93 ± 0.19
MG 6	14.3 ± 0.28	5.81 ± 0.02	5.69 ± 0.03
MG 7	8.7 ± 0.30	5.86 ± 0.03	6.30 ± 0.04
MG 8	17.1 ± 0.80	5.80 ± 0.05	5.61 ± 0.08
MG 9	13.1 ± 0.41	6.24 ± 0.04	5.42 ± 0.05
MG10	10.9 ± 1.26	6.37 ± 0.09	6.12 ± 0.07
MG11	10.4 ± 0.26	6.04 ± 0.03	5.96 ± 0.03
MG13	9.4 ± 2.27	6.30 ± 0.21	6.10 ± 0.11
P	<0.001	<0.001	<0.001
F	35.98	19.79	53.59

Figure 6.10a Modelled hydrological regime - mean depth to the water table

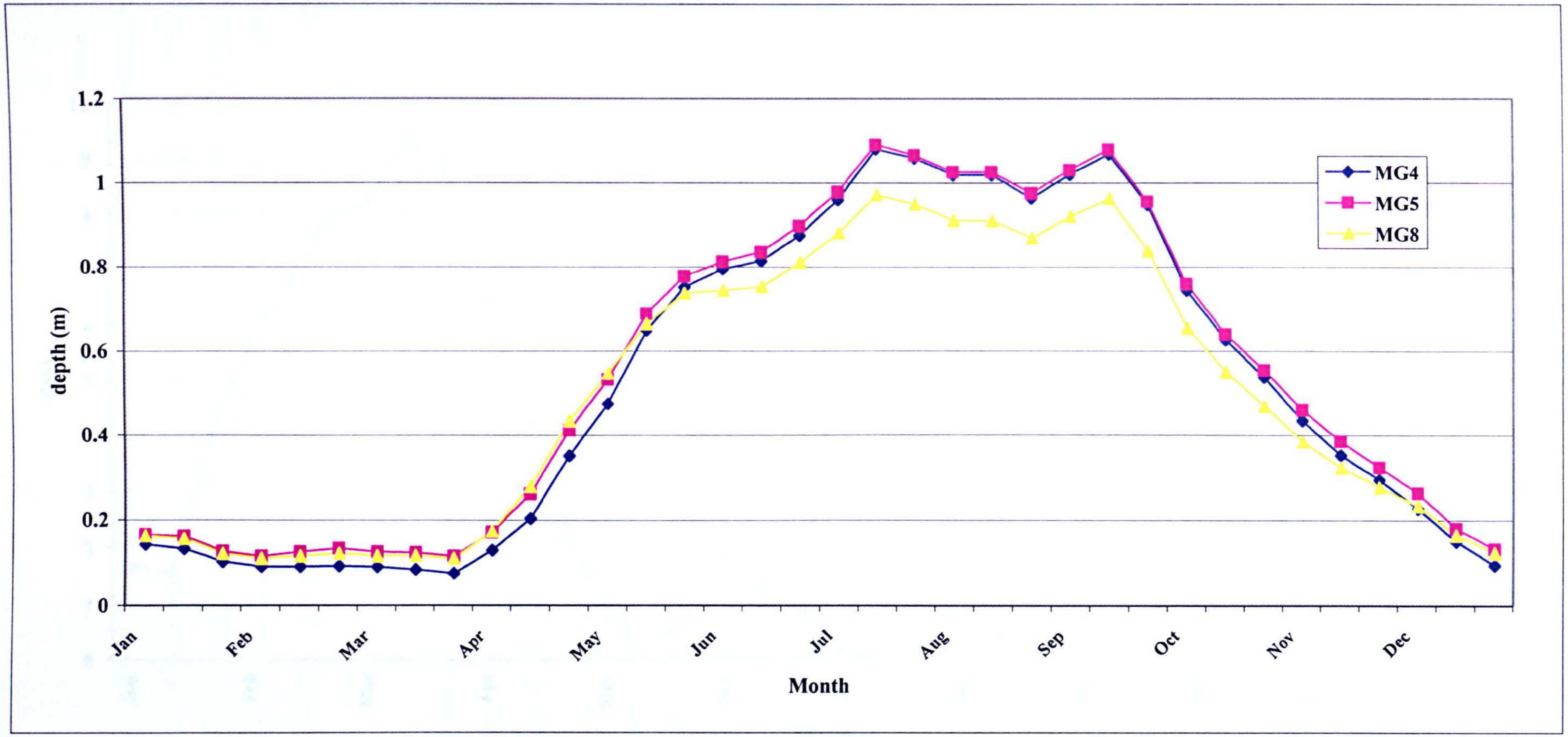
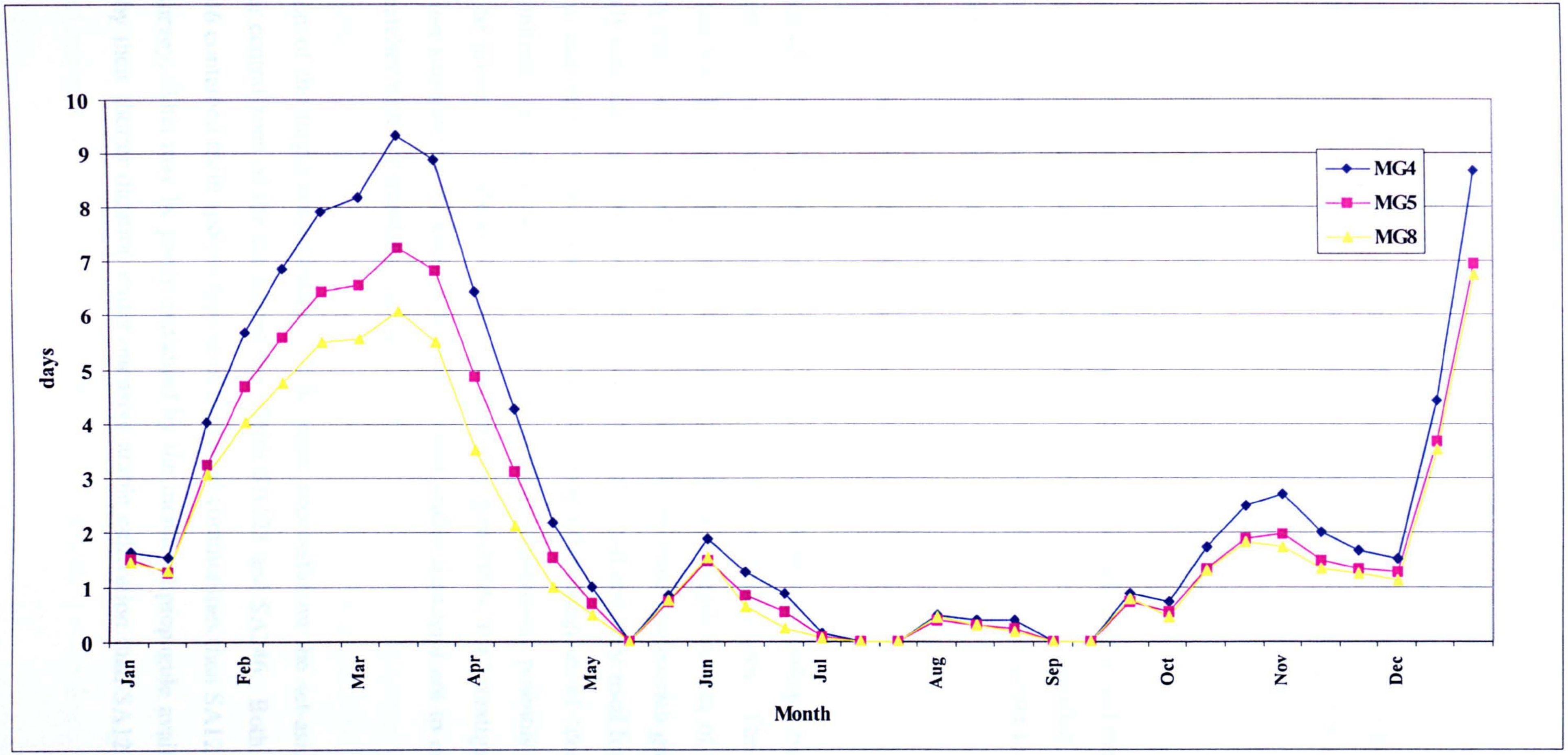


Figure 6.10b Modelled hydrological regime – mean days flooded in 10-day hydroperiods



6.4.3 Discussion - identification of sink areas

Within the study area, the predominant land cover is grass. However, there is somewhere of the order of 27 set-aside fields, with another 80 or so fields used to grow oil seed rape, winter wheat or winter barley. This section considered methods for prioritising the set aside fields on the basis of ease of restoration to the target vegetation types.

Selecting restoration sites on the basis of proximity to source fields

Selecting sites for restoration adjacent to source fields would undoubtedly aid expansion of the resource. On the basis of proximity to source fields, however identified, SA123 would be considered suitable. SA246 also appears suitable, as it is adjacent to source fields of MG4 and MG8 vegetation.

Prioritising sites on the basis of propagule availability

The potential for natural regeneration of target vegetation can perhaps be better determined by assessing the surroundings of potential restoration sites. The results suggest that SA123 will potentially receive propagules of a larger number of species than many other set-aside fields. Although the SSSI is the most species-rich grassland in the study area, the fields to the north and east of SA123 had also been used for arable production and thus the potential for natural dispersal (of propagules of 'desirable' species) initially appeared somewhat limited. Indeed, restoration potential would probably be greater in an ex-arable site surrounded by grasslands, but investigation of the land use surrounding set aside fields reveals that arable sites tend not to occur as isolated patches within a grassland matrix.

Restoration of the target communities will be more successful on the set-aside land within the central zone of the catchment, e.g. fields SA123 and SA246. Both SA245 and SA246 contained more species typical of the target communities than SA123 at the time of survey. This may be partly explained by the increased propagule availability, but also by their shorter duration under intensive arable cultivation than SA123. The species composition of SA245 and SA246 is indistinguishable from that of many grassland fields, unlike the composition of SA123 in its first year set-aside.

Identifying site conditions suitable to support the target vegetation

Of the information available to characterise the target communities, only the modelled hydrological regimes are potentially of use in selecting ex-arable fields for restoration.

The management regime that sustains the target vegetation is essentially the same for all communities and, following the reversion of arable land to grassland, should be reinstated to encourage the restoration of species-rich swards. Similarly, the reinstatement of less intensive grassland management practices on improved grasslands (particularly those in proximity to source fields) should facilitate the rehabilitation of degraded wet grassland habitat.

Community-indicator Score cannot be satisfactorily used to identify suitable arable fields for restoration, because scores derived for the flora of ex-arable fields will tend to be based on transient assemblages of annual arable weeds which perhaps say more about the agricultural management than the abiotic conditions. The scores could be used to identify degraded grasslands for rehabilitation. For example, they could be used to identify swards that have similar moisture and nitrogen values to the target vegetation but differ on the basis of species-richness. Such fields may be isolated from source areas and so propagule immigration is minimal, or closed swards may prevent the establishment of additional species.

The modelled hydrological information is potentially useful in the selection of suitable arable sites since the restoration of wet grasslands is often constrained by inappropriate hydrological regimes. However, the measured and modelled hydrological information for the study area may be of limited practical use in targeting sites for restoration and should be used with caution.

The water regimes of fields that support the target communities are largely similar. Whilst the mean depth to the water table is within 20cm of soil surface during the late winter and spring, this drops during the summer months to over a metre deep. Surface flooding can occur at any time of year, but is more frequent and more prolonged during the spring months for all three communities. MG5 has a lower Ellenberg community mean indicator value for soil moisture than either MG4 or MG8, indicating that it is actually developed under drier conditions than the hydrological model suggests. The supplementary information on altitude and flooding frequency indicates that MG4 occurs at lower altitudes than MG5, and in more frequently flooded areas. Both MG4 and MG5 do co-occur at lower altitudes (e.g. in ridge and furrow fields), but their relative distributions are at least partially determined by the duration of flooding and so only MG5 occurs out of the floodplain on the Denchworth series soil (Treweek *et al.*, 1996). The flooding events apparently experienced by MG5 emphasize that this community is increasingly limited to the higher lying areas of the floodplain (ridges) rather than the higher lying areas of the catchment where one would intuitively expect it to occur. In reality there are differences between the communities (Gowing and Spoor, 1998), as MG4 and MG5 can withstand longer periods of drought stress/ low water table than MG8, but MG4 and MG8 are able to survive longer periods of aeration stress (high water table/ waterlogging) than MG5.

Therefore, the measured and modelled regime characterised for the target communities may be optimal for the persistence of MG4 and MG8, but is most certainly not optimal for MG5. By characterising vegetation according to current site factors, we are assuming that the observed management and site-physical factors are optimal for the maintenance of these communities. This is probably true for the management regime, but the hydrological regime may already be degraded, i.e. the regime that is currently in place is not the one that historically maintained these communities (see Manchester *et al.*, 1999 for discussion of hydrology). Results of hydrological and botanical monitoring between 1993 and 1996 (Chapters 3, 8) suggest that the hydrological regime may not be stable, with vegetation becoming more typical of drier conditions. Certainly, differences in the sward of the SSSI reference habitat between 1993 and 1996 suggest a drying trend (chapter 3). If species distributions are altering in response to hydrological changes, then the regime may no longer be suitable for the continued existence of wet grassland vegetation. It would be unwise to select fields for restoration to the target communities based upon the identification of a sub-optimal hydrological regime.

Even if the regime is stable, water regime characterised at the field scale is too coarse. Swards of unimproved fields are not homogenous in composition. Several community types may co-occur at a relatively small spatial scale, with differences in micro-topography responsible for small-scale changes in the sward. This emphasises again the floristic similarities between the communities and the importance of within field micro-topographical variation for determining small-scale changes in sward composition. Many grassland fields in the study area share an apparently similar hydrological regime to that identified for the target communities, indicating that many fields could potentially support these communities. The water regime information summarised at the field-scale is too coarse to be of value in predicting precisely where the individual target communities may occur, but the fact that the target communities share similar hydrological regimes that are widespread throughout the floodplain of the river Ray suggests that hydrology is not necessarily the chief limiting factor in the development and maintenance of the target communities.

Whilst the hydrological regime that supports the target communities is widespread throughout the study area, not all of the set aside fields examined are subject to this regime. In particular, SA123 appears to be subject to a regime that would not support any of the target vegetation types. However, the hydrological model did not attempt to model the decay in functionality of mole drainage (Armstrong *et al.*, 1996) and assumes that the depth to the water table within drained fields is close to the depth of the mole drains. The degree of dryness experienced will depend on the status of the mole drains. Without regular maintenance, mole drainage will decline in effectiveness with the result that, each year, the level of the water table will drop more slowly following rainfall events (Armstrong, per.comm.) and the soil will gradually revert to a condition equivalent to an undrained soil. Indeed, Armstrong *et al.* (1996) found that set aside fields within the catchment that had reverted from arable cultivation, and where mole drainage had not been renewed, were effectively undrained. It is probable that the mole drainage system within SA123, having not been renewed for at least 10 years, is now totally non-functional and the site will have the same hydrological characteristic as undrained fields, with the dominant effect being surface flooding (Armstrong, pers.comm.).

6.5 Conclusions

Identification of source fields

Data held at 10km resolution cannot be used alone to influence planning restoration at the local scale: co-occurrence mapping of the constituent species of NVC communities results in the apparent co-existence of a greater number of species than will actually occur in any one site. Also, such coarse data cannot be used to determine the area of occupancy of any particular vegetation community type.

With the focus of the ESA on wet grasslands, an amalgamation of several source field selections would satisfy the aims of protecting this resource. For example, combining the target community distributions with the fields of greatest species-richness results in the linking of source fields into continuous blocks of resource.

National conservation and biodiversity targets are referenced to the NVC. Within the study area, however, it would be inappropriate to focus on one community alone when no one field contains a uniform sward of one target community. Micro topographic variation within fields results in the co-occurrence of a number of vegetation communities within several metres of one another. Investigation of the distribution of the target communities revealed considerable overlap, however they were defined. Amalgamation of the target community distributions with those based on species-rich fields and swards results in a composite map of source fields that represent the core of wet grassland and would ideally receive protection through the agri-environment scheme.

If MG4 and MG5 had not been defined as conservation targets, the 20 most species-rich fields identified from the presence-absence data could be considered the 'source' areas for grassland restoration in the study area. A number of these fields were also selected in the optimised selection, in addition to being those that contain the target community types. The fact that a number of these rich fields do contain target vegetation, determined by co-occurrence of constituent species, emphasizes that the target vegetation is species-rich and also that many of the higher constancy species are common to a number of grassland communities.

The use of a reserve selection algorithm to identify the minimum set of fields necessary to represent all species (except arable weeds and bryophytes) at least once results in increased emphasis on species that are rare within the study area.

If agri-environment schemes really are intended to be a key mechanism for delivering biodiversity targets, then some method for targeting valuable sites must be implemented. Currently, entry of fields into agri-environment schemes is on an ad hoc basis, with no targeting of those sites that would confer extra value if added to the scheme. The efficiency of the ESA scheme coverage in protecting the valued grassland resource within the study area could be improved upon. The total number of species represented could be higher, and within a smaller number of sites, e.g. if the 20 richest fields were selected or those identified as MG5 source fields. Similarly, targeting fields known to contain examples of the target vegetation could increase the level of protection afforded to the target communities.

Identification of sites for restoration

As previously stated the best sites for restoration or recreation of wet grassland habitats will, in general, be those that:

- have not been in arable usage for prolonged periods;
- have not been intensively improved, or are marginal for agriculture;
- are sufficiently close to existing species-rich vegetation for it to act as a source of propagules;
- link up existing areas of semi-natural vegetation; and
- need minimal 'engineering' to restore the appropriate hydrological regime.

The hydrological regime that sustains the target vegetation types appears essentially the same for all three communities, although in reality at a small-scale this is not the case (Gowing and Spoor, 1998). In the study area, the hydrological regime was characterised at the field-scale and so can not take into account the micro-topographical variation and within-field changes in water table and soil wetness that allow swards of differing composition to develop within the same field (e.g. dry grassland on ridges with swards typical of inundation grasslands in furrows) and so communities are apparently sustained by the same hydrological regime although in fact their soil moisture preferences are very different. This is reflected in the differences in mean Ellenberg moisture values between communities because these were calculated using $1m^2$ (quadrat) data.

Hydrology should not be assumed to be the over-riding constraint on the development and maintenance of species-rich (wet) grassland within the study area. The apparently similar hydrological regimes mean that it is difficult to predict community occurrence, since it is a combination of micro topographical variation and species distribution, dispersal and establishment that determine the interchange of vegetation types within fields. Within the study area, and the floodplain especially, the choice of sites to restore should be guided by the availability of desirable propagules. The hydrological regime, although modified is not controlled directly by man, and a large number of fields are subject to the regime that sustains the target communities. Even previously drained fields should not be discounted, as mole drainage will degrade with time. There are thus a large number of fields that would need no 'engineering' to restore appropriate hydrology.

The majority of arable sites within the floodplain are likely to be marginal for agriculture due to the difficulties of working and improving the impermeable soils. If this were not the case, many more fields would be under arable production. The level of improvement has more bearing on the restoration of nutrient-poor grasslands than mesotrophic grasslands. Floodplain grasslands receive nutrients with floodwater, in the form of sediment and organic material. However, the more remote fields are less likely to have been as intensively improved, and thus ease of access should perhaps also be a consideration. In this respect, fields SA245 and SA246 may be less suitable than SA123.

The choice of the study site (SA123) for restoration to lowland wet grassland appears to be valid despite an initially apparently inappropriate hydrological regime. SA123 has the added advantage of potentially receiving flood-dispersed propagules. Its position adjacent to the river should also benefit the development of wet grassland vegetation, as it does flood more frequently than fields SA245 and SA246. 'Reversion' of SA123 to grassland should aid in buffering the adjacent nature reserve. Establishment of perennial grassland cover will reduce the seed rain of arable weeds to the reserves. Cessation of field drainage will, to some extent, protect the hydrological regime of the SSSI. The cessation of other practices associated with arable cultivation (e.g. fertilisation, herbicide application) will also reduce inputs of damaging chemicals to the unimproved fields.

7 DERIVING SPECIES TARGETS

7.1 Introduction

Within the Upper Thames Tributaries ESA, Tier 3B (the reversion of arable land to wet grassland) is targeted at arable land in the floodplain, particularly areas where the arable land adjoins existing wet grassland. At the time of this study, under Tier 3B, a permanent grass sward had to be established using at least five species chosen from an approved list of grass species (*Agrostis capillaris, Alopecurus pratensis, Anthoxanthum odoratum, Cynosurus cristatus, Festuca arundinacea, F.pratensis, F.rubra, Holcus lanatus, Phleum pratense*). Seed of wild flower species typical of wet grassland could be included in the seed mixture in addition to the specified grass species if desired. Within the Countryside Stewardship Scheme the prescription for the regeneration of semi-natural vegetation on cultivated land was similar. A minimum of four grass species should be chosen, appropriate to locality and soils, from the approved list (*Agrostis capillaris, Alopecurus pratensis, Anthoxanthum odoratum, Bromus commutatus, Cynosurus cristatus, Dactylis glomerata, Deschampsia cespitosa, Festuca arundinacea, F.ovina, F.pratensis, F.rubra, Hordeum secalinum, Phleum pratense bertolonii, Poa pratensis, Trisetum flavescens*).

These prescriptions provide a basis for reverting arable land to grassland, but do not provide the more detailed floristic targets necessary for full and effective restoration. The species on the approved lists are not necessarily appropriate for evaluating restoration success. The field experiment (Chapter 5) compared the success of grassland restoration using a seed mixture that approximated to the agri-environment prescriptions with two more species-rich mixtures. The presence of sown species (from a species-poor seed mixture) within the restored vegetation merely indicates that sown species have established and not that the target habitat has been restored. Specific objectives that can be used to monitor success are required.

For the purposes of deriving floristic targets for the restoration experiment, the adjacent SSSI was considered as the reference or 'source' field and a subset of the species recorded within this habitat were selected as target species (chapter 4). The use of a SSSI as the target community set site-specific, extremely high standards. The survey of the wider catchment revealed that some of the target species selected are rare elsewhere within the study area. For example, *Thalictrum flavum* and *Cirsium dissectum* each occur in only 4 fields, with one field common to both species (the SSSI itself). They occur in unimproved swards only, and as such are unlikely to establish within an ex-arable field during the early years of reversion. The inclusion of these species in a seed mix for ex-arable land is likely to be wasted effort (e.g. *T.flavum*, Chapter 4). As longer-term goals for natural regeneration, the inclusion of these species will only be appropriate to a limited number of fields in the immediate vicinity of the SSSI. These species could perhaps be more widespread in the study area, but as target species for habitat restoration on arable land they are unrealistic. Whilst the composition of the SSSI may be the ideal for all fields, it is unrealistic to expect vegetation, on what is essentially set-aside land, to approach that of a nature reserve, certainly in the short term.

Restoration experiments reported in the literature have derived floristic targets in a number of ways. Some authors advocate the use of seed harvested from high quality sites (Gilbert 1995; McDonald, 1993; Porter, 1994) or the use of hay, similarly sourced (Jones *et al.*, 1989, 1995). Others have based seed mixtures on the vegetation composition of local sites (Stevenson *et al.* 1995; Mitchley *et al.* 1996; Cullen *et al.*, 1998). Anderson (1995) suggests that the published NVC accounts (Rodwell 1991 *et seq.*) can be used to guide preparation of the species lists and determine relative abundances, but Box (1996) warns that the NVC cannot be used simplistically to draw up species lists or derive planting frequencies. Pywell *et al.* (1997a) based seed mixtures on the botanically diverse grassland communities described in the NVC, selecting species which were considered appropriate to soil type, drainage and the location of each restoration site. Mountford *et al.* (2000) suggested that, for wet grassland species, those that have declined significantly in an area could be considered priority species for restoration, but that these species may not necessarily form part of assemblages in need of restoration in the same area. Wells (1983) developed criteria for choosing species for seed mixes:

- species should be regular members of the grassland community;
- they should be relatively abundant in a variety of grasslands and be widely distributed;

- they should be perennial with effective means of spread and should have readily germinable seed; and
- highly competitive species and those that form single species stands in the wild should be avoided.

The survey data collected for the study area afford an opportunity to investigate the derivation of species targets appropriate to the whole catchment, and which could be applied to other ex-arable sites available for restoration. As previously mentioned (chapter 1, section 1.6; chapter 6. section 6.1), success of conservation and restoration management cannot be evaluated without reference standards. The determination of reference conditions for a valued grassland resource enables an assessment of progress of restoration or rehabilitation. In addition, it is clear from the literature (chapter 1, sections 1.6.1) and the experimental studies (chapter 5) that many species will not arrive and establish within ex-arable sites of their own accord. The field experiment demonstrated that a species-poor grass mixture is relatively unsuccessful at re-establishing a species-rich sward (chapter 5). Diverse seed mixtures can be expensive and not all species establish successfully from seed (poor value for money). The determination of local reference conditions may enable development of prescriptions for diverse seed mixtures that are based on appropriate, relatively common species that form the matrix of 'good' grassland locally.

7.2 Methods

7.2.1 Using nationally-referenced information

Using 10km data, Mountford *et al.* (1997) derived status statistics to quantify changes in the frequency of individual species between 1952-1960 and 1987-1988. These statistics were derived for only a subset of the British flora, plant species markedly more common in wet grasslands than in other biotopes. These data were examined for those 10km squares containing the study area.

7.2.2 Using regional survey data to derive floristic targets – core species

The targets for restoration are those vegetation communities characteristic of wet grassland locally. Is there a common core of species that could not only form the basis for seed mixtures, but also aid in the evaluation of success?

A number of methods were compared for the derivation of target species:

1. Using species lists and recorded frequency to derive floristic targets, species occurring with constancy V (81-100% of samples) or IV (61-80%) were determined from:
   - i. the published NVC constancy tables for the target communities. Target species were those remaining following removal of those that did not occur locally, bryophytes, occasional species (constancy I and II) and species of constancy III (unless occurring as constancy IV or V in another of the target communities);
   - ii. quadrats assigned to the target communities (with goodness of fit $\geq$ 60%);
   - iii. quadrats present within fields identified as 'source fields' from the composite maps of target community distribution; and
   - iv. the qualitative data for fields included within the alternative source field selections of PA20 and Divind.
2. Using Ellenberg indicator values to select species. Ellenberg indicator values were used to determine targets for the field experiment, selecting species from the adjacent reference habitat, the SSSI. A number of species selected as targets failed to establish in the re-seeding experiment (chapter 5). Can the approach to the derivation of target species be improved?
   - i. Following the approach used in Chapter 4, species were selected using Ellenberg indicator values. Species with a preference for above average soil moisture ($F \geq 5$) and below average nitrogen availability ($N \leq 5$) were selected from the quantitative data (includes only species that have been recorded within swards, rather than those present casually. Species present in less than 5% of fields in the study area were discounted as they were considered to be too rare and localised to be catchment-wide targets.

ii. Quadrat mean indicator values (mF, mN) for Ellenberg F and N were calculated by averaging the indicator values of the species which occur in the quadrat. These values were then used to calculate: (1) Community indicator scores (C-mF, C-mN) by averaging the mF/ mN values of quadrats assigned to each community type; and (2) Species indicator scores (S-mF and S-mN) by averaging the mF/ mN values of the quadrats within which a species was recorded. In addition, communities (and individual species) were assigned a score (mS), which indicates the average species-richness (at the m^2 scale) of the community or indicates the species-richness of the swards where a species is present.

Target species are those with similar indicator scores to those of the target communities.

7.3 Results

7.3.1 Status statistics for wet grassland species

Statistics for wet grassland species present in the 10km squares that contain the study area (Table 7.1) reveal that whilst true wetland species have suffered some of the worst declines, e.g. *Hydrocotyle vulgaris*, *Juncus subnodulosus*, *Valeriana dioica* and *Carex* species, other groups of species have fared somewhat better e.g. the agricultural grasses: *Agrostis capillaris* and *A.stolonifera*, *Festuca rubra* and *F.pratensis* and *Phleum pratense*. With the exception of *Carex* species, however, wetland species are absent from the study area (Appendix 6.1).

7.3.1 Using published NVC constancy tables

When species of higher constancy (V or IV) from the published NVC constancy tables for the target communities are examined, 19 species are found to be common to all three (Table 7.2), although not all occur at constancy IV or V in each community.

Table 7.2. Species occurring with constancy IV or V in the target communities.

Species	MG4	MG5a	MG8
Agrostis capillaris	II	IV	I
Alopecurus pratensis	IV	I	
Anthoxanthum odoratum	III	IV	IV
Caltha palustris	I		V
Centaurea nigra	III	IV	I
Cerastium fontanum	IV	III	IV
Cynosurus cristatus	V	V	V
Dactylis glomerata	III	IV	I
Festuca rubra	V	V	V
Filipendula ulmaria	V	I	III
Holcus lanatus	IV	IV	V
Lathyrus pratensis	IV	III	
Leontodon autumnalis	IV	II	IV
Lolium perenne	IV	IV	II
Lotus corniculatus	III	V	I
Plantago lanceolata	V	V	III
Poa trivialis	I	II	IV
Ranunculus acris	V	IV	V
Rumex acetosa	V	III	IV
Sanguisorba officinalis	V		II
Taraxacum officinale agg.	V	III	II
Trifolium pratense	V	IV	III
Trifolium repens	IV	IV	V

7.3.2 Deriving constancy classes from reserve selections

If constancy is determined from the field-scale data, a large number of species appear to occur at high constancy in a great number of fields (table 7.3). Use of presence within quadrats to determine constancy greatly reduces the number of species occurring at high constancy, even when the fields considered are the same (e.g. PA20 versus PA20(quad)). Grasses occur more consistently at higher constancy.

7.3.3 Ellenberg indicator values

As expected, grassland species within the study area with F ≥ 5 and N ≤ 5 are generally typical of wet grasslands (Table 7.4). Comparison of these species with Class II (Appendix 3.1) target species (those present in the reference community) reveals considerable similarity, despite the fact that the current selection was drawn from a larger number of species (the region).

Table 7.4 Species with F value ≥ 5, N value ≤ 5, OR generalist species (47 species).

Grasses	Forbs	Seges/rushes
Agrostis canina	*Achillea millefolium*	*Carex disticha*
Agrostis capillaris	*Cardamine pratensis*	*Carex flacca*
Anthoxanthum odoratum	*Centaurea nigra*	*Carex hirta*
Briza media	*Cerastium fontanum*	*Carex nigra*
Bromus hordeaceus agg.	*Cirsium palustre*	*Carex riparia*
Bromus racemosus	*Filipendula ulmaria*	*Juncus acutiflorus*
Cynosurus cristatus	*Galium palustre*	*Juncus conglomeratus*
Deschampsia cespitosa	*Leontodon autumnalis*	*Juncus effusus*
Festuca arundinacea	*Leontodon saxatilis*	*Juncus inflexus*
Festuca rubra	*Lotus pedunculatus*	
Holcus lanatus	*Lychnis flos-cuculi*	
Hordeum secalinum	*Oenanthe fistulosa*	
Poa subcaerulea	*Oenanthe silaifolia*	
Trisetum flavescens	*Persicaria amphibia*	
	Plantago lanceolata	
	Potentilla reptans	
	Prunella vulgaris	
	Ranunculus acris	
	Ranunculus flammula	
	Sanguisorba officinalis	
	Silaum silaus	
	Stellaria graminea	
	Trifolium pratense	
	Vicia cracca	

Table 7.3. Species of constancy V (81-100% of samples) or IV (61-80%). For **Study area-Field**, **PA20** and **Divind**, samples are fields.

	Study area		PA20			Composite			Quadrat data			Published NVC		
Species	Field	Quad	PA20	(quad)	Divind	MG4	MG5	MG8	MG4	MG5	MG8	MG4	MG5A	MG8
Agrostis capillaris	IV		V		V					V			IV	
Agrostis stolonifera	V	IV	V	V	V	V	V	V	IV	V	V			
Alopecurus geniculatus	IV		V		V									
Alopecurus pratensis	V		V		V	IV	IV	IV	IV		IV	IV		
Anthoxanthum odoratum	IV		V		V	IV	IV	IV	V	V	V		IV	IV
Bellis perennis			IV											
Caltha palustris														V
Cardamine pratensis			V		IV									
Carex hirta			IV											
Carex riparia			IV											
Centaurea nigra			V		V								IV	
Cerastium fontanum	IV		V		V					IV	IV	IV		IV
Cirsium arvense	V		V		V									
Cynosurus cristatus	IV		V		V	IV	IV	IV	V	V	V	V	V	V
Dactylis glomerata	IV		IV		IV								IV	
Deschampsia cespitosa	IV		V		V									
Elytrigia repens			V		IV									
Festuca pratensis			IV		IV									
Festuca rubra	IV		V		V	IV	IV		V	V	V	V	V	V
Filipendula ulmaria			IV									V		
Galium palustre			IV											
Holcus lanatus	V	IV	V	IV	V	V	V	V	V	V	V	IV	IV	V
Hordeum secalinum	IV		V		V					V				
Juncus effusus			IV											

Table 7.3 (continued)

	Study area		PA20			Composite			Quadrat data			Published NVC		
	Fields	Quad	PA20	(quad)	Divind	MG4	MG5a	MG8	MG4	MG5	MG8	MG4	MG5A	MG8
Juncus inflexus			IV											
Lathyrus pratensis			V		V							IV		
Leucanthemum vulgare			IV											
Leontodon autumnalis												IV	II	IV
Lolium perenne	V	V	V		V	IV	IV	IV		V	V	IV	IV	
Lotus corniculatus			IV		IV					IV			V	
Phleum pratense	IV		V		IV									
Plantago lanceolata			IV									V	V	
Poa trivialis	V	V	V	IV	V	V	V	V	IV	IV	V			IV
Potentilla reptans			IV		IV									
Ranunculus acris	V		V	IV	V	IV	IV	IV	V	V	V	V		V
Ranunculus bulbosus										IV				
Ranunculus flammula			IV											
Ranunculus repens	V	IV	V		V			IV						
Rumex acetosa			V		V					IV	IV	V		IV
Rumex conlgomeratus			IV											
Rumex crispus			V		IV									
Sanguisorba officinalis			IV						V			V		
Silaum silaus			IV											
Taraxacum agg.	IV		IV		V							V		
Trifolium dubium			IV		IV									
Trifolium pratense	IV		V		V					V	IV	V	IV	
Trifolium repens	V		V	IV	V	IV	IV	IV		V	V	IV	IV	V
Vicia cracca			V		V									

7.3.4 Mean indicator scores

Species with similar indicator scores (S-mF, S-mN, S-mS) to those calculated for all three target communities (C-mF, C-mN, C-mS: Table 7.5) differ from those selected using actual Ellenberg values (Table 7.4). For example, using this method *Oenanthe* species, *Ranunculus flammula* and *Carex* species are not selected as they have higher soil moisture (mF) values than the target community values (C-mF).

Table 7.5 Species with similar indicator scores (S-mF, S-mN, S-mS) to those of all target communities.

Grasses	Forbs
Anthoxanthum odoratum	*Achillea millefolium*
Arrhenatherum elatius	*Achillea ptarmica*
Briza media	*Centaurea nigra*
Bromus commutatus	*Leucanthemum vulgare*
Festuca rubra	*Filipendula vulgaris*
Poa pratensis	*Heracleum sphondylium*
Trisetum flavescens	*Hypochaeris radicata*
	Leontodon autumnalis
	Leontodon saxatilis
	Luzula campestris
	Ophioglossum vulgatum
	Plantago lanceolata
	Potentilla reptans
	Prunella vulgaris
	Ranunculus bulbosus
	Rhinanthus minor
	Rumex acetosa
	Senecio erucifolius
	Silaum silaus
	Trifolium pratense

A large number of species, whilst not approximating to *all* target communities, do have similar indicator scores to at least one of the communities, and particularly MG8 (table 7.6). Many species that are not typical of species-rich grassland approximate to MG8, including *Cirsium arvense*, *Sonchus asper*, *Trifolium hybridum* and *Elytrigia repens*.

7.4 Discussion

While the arable reversion prescriptions of the agri-environment schemes suggest suitable grass species for reintroduction as seed, there is little guidance on overall floristic targets or how to evaluate restoration success. For the experimental re-establishment of wet grassland (chapter 5), species targets were derived using Ellenberg indicator values to select species from an adjacent SSSI reference habitat (chapter 3). Whilst these species were appropriate for the area, they were rigorous targets to set, as the SSSI is the only field to contain all target species identified. Many of the species selected are too 'rare' and localised in the study area to be realistic targets for arable reversion, certainly in the short-term. This chapter further investigated the derivation of objective targets for restoration, to enable the formulation of appropriate seed mixtures and to aid evaluation.

Use of National datasets

Whilst broad targets for conservation may be defined in terms of NVC community types, there are several reasons why the published constancy tables should not be used to define target species without local refinement. Not all species listed in the published constancy tables will occur within any one region, let alone any one site. Presence of community constituent species within individual sites cannot be extrapolated from presence within a 10km square. The NVC and data held at 10km resolution (e.g. figures 6.1, 6.2) should not be used alone to influence restoration planning at the local scale.

The status statistics (table 7.1) indicate that local restoration effort should focus upon wetland habitats. Examination of Appendix 6.1, however, reveals that many of the declining wetland species recorded regionally do not occur within the study area. In the study area, the wet grassland resource has been identified as the focus for restoration effort despite the fact that many of the species associated with this habitat have apparently not declined. Some species of wet grassland, however, have declined alarmingly between 1952-1960 and 1987-1988 (Mountford *et al.* 1997). For example, *Fritillaria meleagris* (a valued component of MG4 grasslands in Oxfordshire and the UTT ESA) suffered a 71% decline nationally between the 1950s and the 1980s. This species has been recorded within the 10km square containing the study area, and would seem an obvious target for restoration. However, it has not been recorded *within* the study area and so would be an inappropriate target. Many species occur within the 10km square that contains the study area, but are absent from the study area itself. The reasons for their absence locally are not known, although these could probably be explained piecemeal. It is possible that declining species are contracting their ranges and may previously have been present or the species may never have occurred locally. Regardless of the reason for their absence, such species are best avoided by restoration schemes with limited resources. If conditions are not suitable, then efforts to reintroduce such species will be wasted. It will be better to focus on the representative aspects of the habitat rather than the rare elements alone.

The majority of species that have suffered the most severe declines at larger scales, and that are present in the study area (e.g. *Cirsium dissectum, Thalictrum flavum, Serratula tinctoria*) are relatively infrequent locally. The available national status statistics are based on the presence of species at a relatively coarse spatial scale and include no information about the frequency or abundance of a species. It is therefore possible be that the populations of these species have not actually declined in the study area. It is possible that field conditions are becoming less suitable for these species, with the result that they are declining. If that is the case, then current conditions cannot be assumed to be suitable for these species and certainly should not be used to inform habitat restoration, i.e. fields with similar site-physical characteristics and management to those that 'support' these species are not necessarily suitable sites for reintroduction. Longer-term monitoring would be required to evaluate the status of local populations and determine optimum conditions for survival. Alternatively, it is possible that these species have always been scattered at low density through the landscape. They would thus be vulnerable, being readily lost from 10km squares if the few sites they occupied were perturbed. Historically, declines would be more readily recoverable due to the more continuous nature of land cover and the shorter duration of alternative land use (propagules were more widespread and transient seed banks would be persistent enough to ensure recovery). Although their restoration may be desirable, in most cases the limited resources available to conservation will be better spent promoting the 'representative' rather than the 'rare'.

Selection of 'constant' species

Although not all species listed in the published NVC constancy tables for the target communities are present in the study area, the selection of species of higher constancy (table 7.2) as floristic targets for restoration (provided they do occur relatively frequently locally) may be appropriate. The use of the constancy tables of MG4, 5 and 8 to derive target species resulted in a selection that could form the basis of any of the target communities, depending on environmental conditions. The exception to this is *Caltha palustris*, a constituent of MG8 only, and absent from the study area. Since the target communities are so similar floristically, selecting a core of species common to all communities would seem to be more appropriate than focussing on one community alone.

The derivation of species constancies from their presence at the field-scale within the (non-NVC) source field selections reveals a large core set of grass species present at high constancy (regardless of the selection process) (table 7.3). The use of field-scale presence-absence (qualitative) data to determine constancy results in a larger number of species at higher constancy than the use of quadrat data, even when the number of fields considered is the same (e.g. PA20 versus PA20(quad)). Constancy determined from the quadrat data, rather than from presence within fields, will result in the definition of more appropriate targets as it will ensure that species selected do occur regularly in grassland swards (Wells, 1983). For example, *Cirsium arvense* occurs in 81-100% of fields in the study area but does not achieve that level of constancy in quadrats (25% of all quadrats recorded) and, as an injurious weed, would certainly never be selected as a target for restoration (e.g. Gilbert and Anderson, 1998). Examination of target community quadrats reveals a very limited number of forb species present at high constancy, e.g. *Trifolium pratense* and *T.repens*, *Rumex acetosa*, *Ranunculus acris* and *Cerastium fontanum*.

Selecting species using Ellenberg indicator values

Using Ellenberg indicator values to select species with a preference for above average soil moisture and below average nitrogen availability results in a suite of species typical of wetter swards locally (table 7.4). Despite the fact that the 'Ellenberg filter' ($F \geq 5$, $N \leq 5$) was applied to a far greater number of species (all those recorded during the quadrat survey of the study area), the species selected are very similar to those recorded in the SSSI reference habitat (Appendix 3.1) and designated Class II target species for the field experiment (chapter 3, section 3.3.2.2). The removal of species present in fewer than 5% of fields, however, does ensure that rare and localised species are not selected for (re-) introduction, e.g. *Thalictrum flavum*, which was selected from the reference community using the Ellenberg indicator value criterion (chapter 3) and was sown in the field experiment (chapter 5) but failed to establish. Certain species that are apparently integral to the formation of the target communities, and that occur in >5% of fields, are not selected, however, most obviously MG4: *Alopecurus pratensis* (N value = 7), MG4, MG5, MG8: *Trifolium repens* (N value = 6). For the purposes of the field experiment, *A. pratensis* was included despite not meeting the criterion. Despite being a fairly constant constituent of the target communities, *T. repens* is generally not considered suitable for inclusion in 'conservation' seed mixtures as its aggressive growth may be detrimental to the establishment and maintenance of species-rich grassland (Chapman *et al.*, 1996; Warren, 2000). It appears that 'expert' opinion will always be required to determine the appropriateness of certain species as targets for restoration or conservation.

Using derived 'indicator scores' to select species

Selecting species with similar 'indicator scores' to those of *all* the target communities resulted in a different set of species (table 7.5), ranging from those obviously more typical of drier grasslands (*Filipendula vulgaris*, *Ranunculus bulbosus*) to species of wetter swards. Many of the 'matrix' species are missing, grasses particularly, e.g. *Alopecurus pratensis*, *Holcus lanatus*. This is because such species occur widely in grassland swards and are not particularly associated with high species-richness in the study area. Mountford *et al.* (2000) reported that many grass species were not preferential for wet grasslands because of their widespread distribution in other grassland types. The grasses that are present within this selection never are abundant in unimproved swards themselves and although relatively frequent, would nevertheless be inappropriate at high abundance (e.g. as core species for seed mixtures).

A large number of species approximated to the indicator scores of MG8 swards (table 7.6), many of them 'undesirable'. Examination of the indicator scores of these species (table 7.6) reveals that the lower species-richness of MG8 swards resulted in the selection of a number of unsuitable species. As restoring species-richness is one of the aims, it will be better to remove species that approximate to MG8 only, thus removing species more typical of species-poor swards and which are not desirable components of the vegetation, e.g. *Elytrigia repens*, *Cirsium arvense*, *Rumex crispus*, *Sonchus asper*, *Stellaria media*.

The derivation of indicator scores was based simply on the presence of species within quadrats. No weighting was given to account for frequency or abundance of species. It is possible that infrequent species with extreme values will have biased the mean values. Ertsen (1998), however, suggested that this is unlikely as mean indicator values based solely on the presence of plant species differ little from the mean values weighted to cover abundance. In the present study, the results for infrequent species should be treated with caution as they are based on a small number of samples.

Ellenberg indicator values were developed for the vegetation of central Europe and, although widely used in other areas of Europe, are subject to some debate as to their applicability elsewhere. Indeed, Hill *et al.* (2000) have now recalculated Ellenberg values for the British flora. However, the Ellenberg values used within this study were the original values. Comparison of community indicator scores derived using the original and the new values of Hill *et al.* (2000) did reveal differences between values (table 7.7). In all cases, re-calculation with Hill's new values resulted in a lowering of the score, but the communities largely maintained their positions relative to one another.

When indicator scores are re-calculated for individual species using the values as calculated for the British flora (Hill *et al.*, 2000), and compared to the Community indicator scores derived using the recalculated values of Hill et al. (2000), a slightly different set of species emerges (table 7.8). A common core of 23 species is selected using old or new indicator scores. However, four species with similar scores to the target communities (based on the original values) are not selected using the recalculated values (*Achillea ptarmica*, *Bromus commutatus*, *Centaurea nigra* and *Silaum silaus*), whilst four new species are identified (*Cirsium palustre*, *Galium verum*, *Lotus corniculatus* and *Poa humilis*).

Selecting appropriate species

The more common and generalist species should be selected as short-term targets for restoration, and for re-introduction to ex-arable land. Such species are more likely to be suited to local conditions and can generally be expected to germinate and establish successfully as well as being likely to benefit invertebrate feeders. Some ubiquitous species, however, are likely to colonise naturally and so will not need re-introduction, e.g. *Taraxacum* agg. (Gilbert and Anderson, 1998). Less frequent, more discerning species are unlikely to establish in modified (ex-) arable soils. Species may be rare because conditions are unsuitable for survival and they are declining, they may be poor dispersers and colonisers, or simply because not all species are widespread and common. Whatever the reasons for their rarity, such species appear only occasionally within swards and are not appropriate short-term targets for community restoration. Moreover, if the hydrological regime is not suitable for the maintenance of the target communities within he study area, i.e. the regime that is currently in place is not the one that historically maintained these communities, then species distributions may already be degraded, and contracting. The period of study coincided with a number of drought years and differences in the sward of Long Herdon SSSI between 1993 and 1996 certainly suggest a drying trend (chapter 4). As many of the characteristic species of these grasslands are not specialist wetland species, it would be appropriate to set targets in terms of 'generalist' rather than 'specialist' species. If conditions are optimal for wet grasslands, natural processes should ensure that at least some wetter species colonise naturally. If conditions are not optimal for wet grassland, introduction of wetter species will fail, but more generalist species should survive.

Comparison of the ESA approved list of species for re-introduction (section 7.1) with the present results suggests that several approved species may be inappropriate for the re-establishment of species-rich wet grassland in the study area. For example, *Festuca arundinacea* is selected as a target using Ellenberg values (table 7.4), but is not a constant member of any of the target communities and only occurs at constancy I (1-20% of fields) in the study area (appendix 6.1). Although *Phleum pratense* occurs more frequently (61-80% of fields), it is not a constant member of the target community swards (table 7.3) and is not selected as a target using Ellenberg values. Neither of these species were sown in the field experiment (chapter 5) and so, although it is not clear how their inclusion would have affected grassland establishment, it seems likely that the sward developed would not be similar to those that occur naturally.

Grass species not recommended for sowing, but that are constant members of the target communities include *Lolium perenne* and *Poa trivialis*. Results of the seed bank investigation (chapter 4) indicated that seed of *Poa* and *Lolium* species was present in the soil and indeed, both these species did arrive naturally at the restoration site (chapter 5). Early abundance of *P. trivialis* was high in experimental plots and, although abundance has declined in subsequent years, augmentation by sowing might have been detrimental to the establishment of other species. *L. perenne*, however, has only ever been present at low abundance (and frequency) and so it can be assumed that *Lolium* seed in the soil was largely of *L. multiflorum*. A higher abundance and frequency of *L. perenne* may therefore be desirable but, even had the seed mixture been formulated following seed bank investigation, this species would not have been included.

7.5 Conclusions

The findings of this study (chapters 5, 6 and 7 particularly) suggest that guidance for land owners entering land into the Upper Thames Tributaries ESA could be improved, both to aid in establishing a species-rich sward on ex-arable land and to enable progress towards the desired endpoint to be evaluated. Realistic species targets can only be determined with local knowledge.

The different NVC community types do not occur in isolation within the study area and so a suitable overall aim might be to restore species-rich grassland vegetation that is characteristic of the study area, i.e. a mosaic of community types including the species-rich, drier MG5 community (both out of the floodplain and on higher-lying land within the floodplain), the diverse wet grassland communities of MG4 and MG8, elements of improved grassland (MG6) and species-poor swards more typical of inundation, such as might be found in furrows (e.g. MG9, MG10, MG11).

Where natural regeneration is not appropriate, a seed mixture comprised of a greater number and diversity of species than the recommended 5 (or 4) grass species mix should be formulated. Although the composition of the seed mix could be derived in a number of ways, ubiquitous species are more likely to establish successfully on ex-arable land than the less frequent but more discerning, habitat-specific species of wet grassland and so species for (re-) introduction as seed should be selected from the listing of species present at reasonably high constancy locally within grassland swards (e.g. within quadrats identified by Tablefit as belonging to the target communities (table 7.3)). Caution should be used, however, as certain species present at high constancy locally are unlikely to need introducing, even to arable land, e.g. *Poa trivialis* and *Trifolium repens*.

The use of Ellenberg indicator values (as used to derive targets for the field experiment; chapter 3) resulted in the selection of species indicative of wet grasslands, whilst the use of the 'indicator score' approach selects species indicative of MG4, MG5 and MG8. These species are targets for community restoration, but not without the 'matrix' grassland species, which are identified using the constancy species approach, but not the 'indicator score' approach. Restoration targets for (ex-) arable land should initially be to re-establish the matrix species and, in the longer-term, to establish the indicator species.

When monitoring the success of restoration, the species present at high constancy in the study area should be short-term goals, whilst the species selected using the 'indicator scores' approach could be used as a longer-term evaluation criterion 'indicator species'. For example, if the experimental results from the different seed treatments (Chapter 5) are re-examined, once again the sward derived from the most species-rich seed mixture contains the greatest number of indicator species (as listed in table 7.4 and 7.5) and also the greatest number of high constancy species (table 7.2 and 7.3). The use of data local to the study area, rather than data local to an individual field, when formulating restoration targets should ensure that targets are appropriate to the region rather than to one field only.

Table 7.1 Percentage change in frequency of individual species within Britain, and within two BSBI regions, south-east and south-west England (from Mountford *et al.*, 1997).
Species listed are those occurring within the 10km squares containing the study area.

Species	national	SE Eng	SW Eng
Achillea ptarmica	-8	16%	-23
Agrimonia eupatoria	-15	4	-5
Agrostis capillaris	5	8	12
Agrostis stolonifera	12	8	7
Ajuga reptans	-8	-7	-9
Alopecurus geniculatus	4	21	2
Alopecurus pratensis	5	1	27
Angelica sylvestris	2	9	5
Anthoxanthum odoratum	2	-1	3
Bellis perennis	2	4	3
Berula erecta	-8	7	-37
Briza media	-17	-19	-11
Bromus hordeaceus	4	6	3
Caltha palustris	-9	-21	-16
Cardamine pratensis	1	1	1
Carex acutiformis	-10	3	-24
Carex disticha	-26	-1	-32
Carex echinata	-14	-56	-30
Carex hirta	-1	3	-3
Carex nigra	-11	-33	-28
Carex otrubae	-8	3	-1
Carex ovalis	-10	-11	-43
Carex panicea	-11	-17	-30
Centaurea nigra	0	-3	3
Cerastium fontanum	3	4	3
Cirsium palustre	0	-1	-3
Cirsium dissectum	-41	-43	-53
Cynosurus cristatus	1	1	7
Dactylorhiza fuchsii	3	6	-10
Dactylorhiza praetermissa	20	106	-5
Deschampsia cespitosa	1	-1	1
Epilobium obscurum	4	49	-11
Epilobium palustre	-2	-30	-16
Epilobium parviflorum	-10	1	-2
Festuca rubra	12	8	17
Festuca arundinacea	36	44	40
Festuca pratensis	-8	-2	-13
Filipendula ulmaria	-2	-3	-1
Fritillaria meleagris	-71	-74	-79
Galium palustre	1	-6	1
Galium uliginosum	-27	-41	-36
Glyceria fluitans	3	9	7
Holcus lanatus	3	6	3

Table 7.1 (continued)

Species	national	SE Eng	SW Eng
Hordeum secalinum	-19	14	-18
Hydrocotyle vulgaris	-25	-48	-21
Hypericum tetrapterum	-9	6	-5
Hypochaeris radicata	2	4	7
Iris pseudacorus	-3	11	-3
Juncus acutiflorus	7	3	-1
Juncus articulatus	0	-13	-10
Juncus conglomeratus	2	-10	-10
Juncus effusus	3	6	3
Juncus inflexus	-2	8	-2
Juncus subnodulosus	-32	-6	-54
Lathyrus pratensis	2	-1	3
Leontodon autumnalis	3	8	-1
Leucanthemum vulgare	-8	1	3
Lolium perenne	0	6	5
Lotus pedunculatus	2	9	3
Luzula campestris	-1	-2	5
Lychnis flos-cuculi	-9	-12	-8
Mentha aquatica	0	1	5
Myosotis laxa	-10	-22	-30
Persicaria hydropiper	-4	0	-7
Phalaris arundinacea	0	8	22
Phleum pratense	13	11	21
Phragmites australis	-5	3	-7
Plantago lanceolata	2	2	3
Poa trivialis	9	8	7
Potentilla anglica	-21	-58	-34
Potentilla anserina	0	1	-1
Potentilla reptans	-6	4	3
Potentilla erecta	-6	-34	-11
Primula veris	-22	-6	-13
Prunella vulgaris	1	2	3
Ranunculus acris	2	-1	3
Ranunculus bulbosus	-13	-3	5
Ranunculus flammula	-10	-18	-24
Ranunculus repens	2	4	3
Rhinanthus minor	-5	-21	-9
Rumex conglomeratus	4	17	30
Rumex acetosa	2	11	3
Sanguisorba officinalis	-17	4	1
Senecio aquaticus	-19	-24	-8
Serratula tinctoria	-34	-61	-22
Sieglina decumbens	-14	-13	-24
Silaum silaus	-25	-4	13
Succisa pratensis	-11	-22	-15
Thalictrum flavum	-37	-28	-42

Table 7.1 (continued)

Species	national	SE Eng	SW Eng
Trifolium pratense	1	4	3
Trifolium repens	4	4	5
Trifolium dubium	-1	6	5
Triglochin palustre	2	-45	-32
Trisetum flavescens	-15	3	-11
Valeriana dioica	-27	-39	24
Veronica beccabunga	-3	1	1
Vicia cracca	1	6	10

Table 7.6 Mean indicator scores (**S-mN, S-mF, S-mS**), the target communities (**MG4, MG5, MG8**) to which they approximate (*), & the percentage frequency within quadrats (**%F**).

Species	%F	S-mN Mean ± S.E.	S-mF Mean ± S.E.	S-mS Mean ± S.E.	MG4	MG5	MG8
Achillea millefolium	1	5.41 ± 0.14	5.54 ± 0.07	18.20 ± 1.26	*	*	*
Achillea ptarmica	1	5.20 ± 0.25	6.20 ± 0.20	22.44 ± 1.95	*	*	*
Elytrigia repens	15	5.87 ± 0.04	6.00 ± 0.03	13.64 ± 0.39			*
Agrostis stolonifera	73	5.71 ± 0.02	6.07 ± 0.02	13.74 ± 0.17			*
Agrostis capillaris	20	5.53 ± 0.03	5.97 ± 0.03	16.20 ± 0.32		*	*
Alopecurus pratensis	47	5.73 ± 0.02	6.07 ± 0.02	14.46 ± 0.21			*
Anthoxanthum odoratum	34	5.39 ± 0.03	6.04 ± 0.02	17.31 ± 0.22	*	*	*
Arrhenatherum elatius	1	5.64 ± 0.19	5.92 ± 0.06	20.73 ± 1.19	*	*	*
Bellis perennis	6	5.67 ± 0.05	5.64 ± 0.03	14.34 ± 0.49		*	*
Briza media	1	4.67 ± 0.12	5.37 ± 0.14	26.43 ± 1.39	*	*	*
Bromus commutatus	3	5.68 ± 0.10	5.80 ± 0.04	15.56 ± 0.67	*	*	*
Bromus hordeaceus	9	5.91 ± 0.04	5.88 ± 0.04	13.52 ± 0.38			*
Bromus racemosus	7	5.58 ± 0.05	6.11 ± 0.04	16.27 ± 0.56	*		*
Cardamine pratensis	16	5.30 ± 0.04	6.33 ± 0.04	17.58 ± 0.33			
Carex flacca	3	4.83 ± 0.09	6.18 ± 0.09	19.11 ± 0.79	*		
Carex hirta	4	5.41 ± 0.07	6.26 ± 0.06	15.20 ± 0.77			*
Centaurea nigra	10	5.02 ± 0.04	6.00 ± 0.05	20.66 ± 0.41	*	*	*
Cerastium fontanum	16	5.61 ± 0.03	5.74 ± 0.02	15.98 ± 0.30		*	*
Leucanthemum vulgare	2	5.12 ± 0.11	5.57 ± 0.06	18.07 ± 1.33	*	*	*
Cirsium arvense	25	5.85 ± 0.03	5.91 ± 0.02	13.25 ± 0.27			*
Cirsium palustre	1	4.67 ± 0.12	6.00 ± 0.09	20.20 ± 1.94	*	*	
Cirsium vulgare	5	5.98 ± 0.06	5.79 ± 0.04	14.12 ± 0.53			*
Cynosurus cristatus	51	5.54 ± 0.02	5.91 ± 0.02	15.44 ± 0.19		*	*
Dactylis glomerata	11	5.71 ± 0.04	5.74 ± 0.03	15.07 ± 0.43		*	*
Deschampsia cespitosa	29	5.38 ± 0.03	6.17 ± 0.02	14.90 ± 0.28			*
Festuca pratensis	7	5.63 ± 0.06	6.00 ± 0.04	15.40 ± 0.61		*	*
Festuca rubra	36	5.50 ± 0.03	6.03 ± 0.02	16.52 ± 0.23	*	*	*
Filipendula ulmaria	5	4.92 ± 0.07	6.35 ± 0.08	21.26 ± 0.61	*		
Filipendula vulgaris	1	4.92 ± 0.07	5.56 ± 0.08	24.56 ± 0.77	*	*	*
Galium palustre	1	4.96 ± 0.18	6.40 ± 0.22	20.29 ± 1.78	*		*
Galium verum	1	4.90 ± 0.09	5.43 ± 0.07	24.67 ± 1.78	*	*	
Geranium dissectum	3	5.92 ± 0.09	6.69 ± 0.08	13.70 ± 0.98			*
Heracleum sphondylium	1	5.61 ± 0.14	5.91 ± 0.11	18.50 ± 1.95	*	*	*
Holcus lanatus	78	5.67 ± 0.02	6.02 ± 0.01	13.87 ± 0.16			*
Hordeum secalinum	30	5.72 ± 0.02	5.91 ± 0.02	14.63 ± 0.23		*	
Hypochaeris radicata	1	4.89 ± 0.25	5.63 ± 0.15	25.17 ± 2.09	*	*	*
Juncus inflexus	2	5.44 ± 0.09	6.31 ± 0.10	14.04 ± 0.89			*
Lathyrus pratensis	10	5.30 ± 0.05	6.09 ± 0.04	19.58 ± 0.40	*		*
Leontodon autumnalis	3	5.29 ± 0.10	5.81 ± 0.09	19.72 ± 1.09	*	*	*
Leontodon saxatilis	1	5.33 ± 0.10	5.64 ± 0.10	15.60 ± 0.93	*	*	*

Table 7.6 (continued)

Species	%F	S-mN Mean ± S.E.	S-mF Mean ± S.E.	S-mS Mean ± S.E.	MG4	MG5	MG8
Lolium perenne	85	5.89 ± 0.02	5.92 ± 0.01	12.51 ± 0.15			*
Lotus corniculatus	6	4.90 ± 0.06	5.98 ± 0.06	20.88 ± 0.55	*	*	
Lotus pedunculatus	1	4.79 ± 0.14	6.41 ± 0.12	21.13 ± 0.95	*		
Luzula campestris	3	4.99 ± 0.06	5.69 ± 0.08	21.28 ± 0.85	*	*	*
Ophioglossum vulgatum	1	4.85 ± 0.22	5.55 ± 0.14	25.43 ± 1.95	*	*	*
Phleum bertolonii	5	5.60 ± 0.04	5.72 ± 0.05	12.77 ± 0.49			*
Phleum pratense	39	6.00 ± 0.02	5.90 ± 0.02	12.68 ± 0.21			*
Plantago lanceolata	2	5.22 ± 0.11	5.65 ± 0.06	22.44 ± 1.00	*	*	*
Plantago major	4	6.26 ± 0.07	5.79 ± 0.05	12.04 ± 0.47			*
Poa pratensis	10	5.70 ± 0.14	6.03 ± 0.19	17.50 ± 1.29	*	*	*
Poa trivialis	87	5.88 ± 0.02	6.02 ± 0.01	12.74 ± 0.15			*
Potentilla anserina	1	5.83 ± 0.16	6.38 ± 0.24	9.60 ± 0.93			*
Potentilla reptans	9	5.34 ± 0.04	5.89 ± 0.04	17.41 ± 0.43	*	*	*
Prunella vulgaris	3	5.18 ± 0.11	5.74 ± 0.07	19.03 ± 0.90	*	*	*
Ranunculus acris	50	5.57 ± 0.02	6.00 ± 0.02	15.63 ± 0.19		*	*
Ranunculus bulbosus	11	5.37 ± 0.03	5.53 ± 0.03	16.47 ± 0.45	*	*	*
Ranunculus repens	58	5.86 ± 0.02	6.11 ± 0.02	13.24 ± 0.18			*
Rhinanthus minor	2	5.12 ± 0.09	5.61 ± 0.08	23.05 ± 1.00	*	*	*
Rumex acetosa	19	5.58 ± 0.03	5.87 ± 0.02	17.08 ± 0.31	*	*	*
Rumex crispus	7	6.10 ± 0.05	6.10 ± 0.04	12.82 ± 0.48			*
Sanguisorba officinalis	16	5.25 ± 0.04	6.13 ± 0.04	18.55 ± 0.35	*		*
Senecio erucifolius	1	5.75 ± 0.35	5.61 ± 0.17	11.20 ± 0.58	*	*	*
Silaum silaus	3	5.14 ± 0.08	6.06 ± 0.09	21.41 ± 0.72	*	*	*
Sonchus asper	1	6.14 ± 0.17	5.72 ± 0.16	13.38 ± 1.60			*
Stellaria media	2	6.35 ± 0.10	5.67 ± 0.16	10.39 ± 0.86			*
Taraxacum agg.	28	6.02 ± 0.03	5.87 ± 0.02	12.87 ± 0.28			*
Trifolium dubium	7	5.65 ± 0.05	5.65 ± 0.04	14.92 ± 0.50		*	*
Trifolium hybridum	1	6.19 ± 0.10	5.71 ± 0.10	13.20 ± 0.80			*
Trifolium pratense	23	5.57 ± 0.03	5.81 ± 0.02	16.81 ± 0.29	*	*	*
Trifolium repens	65	5.83 ± 0.02	5.90 ± 0.01	13.13 ± 0.18			*
Trisetum flavescens	5	5.45 ± 0.06	5.72 ± 0.04	18.23 ± 0.63	*	*	*
Vicia sativa	2	5.45 ± 0.15	6.28 ± 0.11	17.76 ± 1.07	*		*
Vicia cracca	11	5.27 ± 0.05	6.23 ± 0.05	18.49 ± 0.44	*		*

Table 7.7 Re-calculated C-mN and C-mF indicator scores for NVC communities (values re-calculated using the recalculated values of Hill *et al.* (2000)).

Community	N	mmF ± S.E.	mmN ± S.E.
MG 4	20	5.60 ± 0.07	4.81 ± 0.11
MG 5	7	5.24 ± 0.04	4.61 ± 0.19
MG 6	144	5.56 ± 0.02	5.23 ± 0.03
MG 7	93	5.56 ± 0.03	5.75 ± 0.04
MG 8	14	5.49 ± 0.03	5.07 ± 0.08
MG 9	84	5.84 ± 0.03	5.20 ± 0.05
MG10	8	5.96 ± 0.09	5.67 ± 0.08
MG11	94	5.66 ± 0.02	5.66 ± 0.03
MG13	5	5.86 ± 0.21	5.71 ± 0.11
P value		<0.001	<0.001
F		16.30	45.81

Table 7.8 Species whose re-calculated indicator scores (**S-mN**, **S-mF**) approximate to *all* three target communities (**MG4**, **MG5**, **MG8**).

Species	S-mN: new Mean ± S.E.	S-mF: new Mean ± S.E.
Achillea millefolium	5.01 ± 0.09	5.37 ± 0.04
Anthoxanthum odoratum	4.97 ± 0.02	5.75 ± 0.02
Arrhenatherum elatius	5.47 ± 0.13	5.60 ± 0.03
Briza media	4.46 ± 0.11	5.30 ± 0.11
Cirsium palustre	4.56 ± 0.06	5.79 ± 0.08
Festuca rubra	5.14 ± 0.02	5.70 ± 0.02
Filipendula vulgaris	4.73 ± 0.05	5.48 ± 0.05
Galium verum	4.70 ± 0.07	5.36 ± 0.06
Heracleum sphondylium	5.33 ± 0.14	5.62 ± 0.07
Hypochaeris radicata	4.69 ± 0.23	5.40 ± 0.07
Leontodon autumnalis	4.95 ± 0.08	5.72 ± 0.09
Leontodon saxatilis	4.75 ± 0.08	5.37 ± 0.08
Leucanthemum vulgare	4.95 ± 0.12	5.42 ± 0.05
Lotus corniculatus	4.65 ± 0.04	5.74 ± 0.05
Luzula campestris	4.69 ± 0.05	5.52 ± 0.05
Ophioglossum vulgatum	4.51 ± 0.12	5.49 ± 0.08
Plantago lanceolata	4.79 ± 4.79	5.45 ± 0.04
Poa pratensis	5.37 ± 0.13	5.69 ± 0.09
Poa humilis	4.95 ± 0.05	5.76 ± 0.10
Potentilla reptans	5.06 ± 0.04	5.59 ± 0.03
Prunella vulgaris	4.78 ± 0.08	5.54 ± 0.05
Ranunculus bulbosus	5.10 ± 0.03	5.39 ± 0.02
Rhinanthus minor	4.89 ± 0.09	5.49 ± 0.06
Rumex acetosa	5.08 ± 0.02	5.57 ± 0.02
Senecio erucifolius	5.68 ± 0.25	5.64 ± 0.09
Trifolium pratense	5.16 ± 0.02	5.55 ± 0.02
Trisetum flavescens	5.02 ± 0.04	5.38 ± 0.02

CHAPTER 8 FINAL DISCUSSION

Lowland wet grassland occurs within a number of the UK's Environmentally Sensitive Areas (ESA). The Upper Thames Tributaries ESA, and associated prescriptions for arable reversion, provided the environmental and policy context for this investigation of re-creation of lowland wet grassland on ex-arable land.

8.1 Constraints on wet grassland re-creation

A review of the literature (chapter 1) identified which vegetation communities were constituents of lowland wet grassland swards in the UK and summarised the factors responsible for their ongoing decline. Lowland wet grassland tends to develop on the least workable land (i.e. low-lying, varying from seasonally inundated to permanently waterlogged and often on heavy soils) and therefore often remained unimproved longer than grassland in more workable situations. However, the second half of the twentieth century saw the introduction of economic techniques for the 'reclamation' of even such previously unproductive land for intensive agriculture. Land drainage and changes in farming practices then resulted in the deterioration (in conservation terms) of existing grasslands and the conversion of permanent grasslands to arable cultivation. Constraints on the preservation and promotion of lowland wet grassland include hydrology, soil nutrient availability and the availability and suitability of propagule sources. Whilst these factors affect restoration of any grassland type, their relevance to wet grassland differs in detail and degree from other habitat types. Thus, possible techniques for overcoming these constraints were considered both in the context of their relevance to the communities under consideration and also in the context of agri-environment schemes and continued agricultural production from this land.

8.1.1 Hydrology

The maintenance of wet grassland depends on a suitable hydrological regime. The ex-arable study site lies on a stretch of the River Ray that currently supports species-rich wet grassland and is subject to the same hydrological regime as the adjacent SSSI. It was decided that hydrological conditions in the study area could be assumed to be suitable, as the regime remains largely 'natural' despite some modifications to the watercourse (chapter 2). However, the results of the hydrological monitoring and modelling work (Armstrong *et al.*, 1996; Rose & Armstrong, 1996) suggested that the hydrological regime of the experimental study site was not appropriate for the maintenance of wet grassland. This field experiences a much greater variation in water levels than many undrained fields. The presence of mole drains means that the depth to the water table drops greatly during the summer months when the water level in the River Ray is low. However, its location adjacent to the River means that is in a high-risk area for flooding. The hydrological model did not address the effects of the ongoing decay of mole drainage, but by now the hydrological profile of the study site is likely to be approaching that of undrained sites locally.

The fields in this part of the ESA are not hydrologically distinct units and so, even if the water regime had been considered unsuitable, there was limited scope for hydrological manipulation without fairly major engineering works.

Following the establishment of a permanent grassland sward, ex-arable land entered into the ESA scheme wet grassland reversion Tier must be managed in accordance with Tier 2 guidelines for wet grassland. These Tier 2 prescriptions require that water levels are maintained in ditches and watercourses to within 30cm of mean field level from 1st April to 31st May and for as long as possible thereafter. Throughout the year there should be at least 30cm of water in the bottom of ditches and watercourses. This is similar to the water management prescriptions in operation in other English ESAs that support lowland wet grassland. Whilst appropriate to areas on peat soils where soil wetness is maintained by a high water table, it is debatable whether this is actually an appropriate prescription for the study area. As discussed in chapter 2, impermeable soils and surface inundation maintain wet grassland in the study area. The implementation of a 'new' hydrological regime may result in damage to the character of existing wet grasslands. Maintenance of water levels in drainage channels could benefit the re-establishment of characteristic vegetation on previously drained sites, as existing field drainage structures would enable water to travel to the field centres. However, the low hydraulic conductivity of the soils prevents rapid lateral movement of water and thus undrained soils would be unlikely to benefit from the maintenance of water levels in surrounding watercourses.

Although the continued existence of wet grasslands in proximity to the arable reversion study site certainly suggested that the hydrological regime might be suitable, during the early years of the field experiment hydrological conditions were not conducive to the maintenance of this community. Rainfall was particularly low in the spring of 1995, resulting in part in the drastically reduced ground cover and number of species recorded in the restored vegetation (chapter 5). Results of vegetation monitoring of the adjacent SSSI in 1993 and 1996 further suggested a sub optimal hydrological regime, e.g. the ground cover of sedges and rushes, as a group, had declined by 15% over the three years (chapter 4). Whilst this regime has undoubtedly been detrimental to the establishment of wet grassland, the selection of the study site for restoration is still valid especially given the Government's responsibilities for biodiversity and the publication of the Habitat Action Plan lowland species-rich meadows, particularly MG4, MG5 and MG8 (Anon, 1995, 1998).

During the period of this study, there has been increasing concern that Long Herdon SSSI and the adjacent meadow (owned by Plantlife and Timotei) should be protected from major alterations to the hydrological regime that might adversely affect their characteristic vegetation. The Environment Agency has identified the need to enhance the wetland and river channel habitat of the study area and has been considering a number of options to enhance the ecological value of the site (Armstrong *et al.*, 1999). Any hydrological manipulation that benefits the unimproved meadows can only be beneficial to the study site also.

8.1.2 Soil Nutrients

Low levels of soil phosphorus are generally required to sustain high levels of species co-existence in grasslands in the longer term (Marrs *et al.*, 1991; Janssens *et al.*, 1998). Janssens *et al.* (1998) suggest that, at least on soils rich in organic matter, sites for restoration of high diversity should have extractable phosphorus concentrations below 5mg/100g or it will be necessary to decrease the quantity of soil phosphorus. However, methods for phosphorus reduction are costly, impractical and unlikely to be unacceptable. Agri-environment scheme agreements are not made in perpetuity and the possibility that land may be required for more intensive agricultural production in the future means that actions which might 'adversely affect' the productivity of the land are not likely to be popular. Moreover, reduction of soil nutrients may not always be necessary. Not all species-rich grasslands are inherently nutrient-poor systems, e.g. *Alopecurus pratensis-Sanguisorba officinalis* grassland (MG4), routinely receives nutrients with floodwater (Rodwell, 1992b). Indeed, Gilbert (1995) points out that the majority of wild flower communities are developed on fertile brown earth soils and thus are not low fertility systems.

Under conditions of high water tables, many species are shallow-rooting, which restricts the volume of soil within which a plant can search for nutrients (Gowing and Spoor, 1998). Under wet conditions, mineral nitrogen is lost from soil through denitrification, depleting reserves of nitrogen available to plants (Van Oorschot *et al.*, 2000). Hence, nutrient availability is generally lower under conditions of high water tables. Species that are able to capitalise on the high nutrient availability of drier soils and so gain a competitive advantage are not similarly advantaged in wet grasslands where high water tables may limit nutrient availability and the species itself is poorly adapted to conditions of high soil wetness. Nutrient enrichment may well not be such an obstacle to the restoration of wet grassland as it is to more nutrient poor communities.

Gilbert *et al.* (1996) sampled soils beneath a range of semi-natural communities classified according to the NVC methodology (Rodwell, 1991 *et seq.*). The results for available phosphorus further demonstrated that the species-rich wet grassland communities are not particularly low in terms of measured available phosphorus. Soils under the dry hay meadow community MG5a were lower in phosphorus (5 mgl^{-1}) than both MG4 (10-16 mgl^{-1}) and MG8 (~15 mgl^{-1}). This further emphasizes the influence of flooding on nutrient levels within floodplain grassland.

Levels of available soil phosphorus were measured within the study area (Rose, *pers.comm.*). Concentrations (Olsen P; mgl^{-1}) were determined for a number of different levels of grassland intensification and for the ex-arable reversion site itself. Concentrations measured within the reversion site (6 mgl^{-1}) were found to compare favourably with those recorded for semi-natural grasslands (7 mgl^{-1}) unlike semi-improved (13 mgl^{-1}) and improved (17 mgl^{-1}) grasslands.

It had been decided prior to determination of the nutrient status of the ex-arable study site that measures to reduce soil nutrients would not be implemented. The subsequent soil analysis revealed that the phosphorus status of the field was within acceptable levels of enrichment to support semi-natural mesotrophic grassland. Thus soil phosphorus concentrations are unlikely to present a problem to the restoration and maintenance of species-rich grassland on this site. It is also possible that inundation will prevent those species with a competitive advantage in nutrient-rich *dry* situations from becoming dominant in flooded sites.

8.1.3 Sources of propagules for restoration

The literature revealed that soil seed banks of arable fields are generally unsuitable for the restoration of many habitats including chalk grassland and heathland communities. When this study began, very little work had been carried out on the communities typical of lowland wet (floodplain) grassland and it seemed possible that soil seed banks of floodplain grasslands (and those converted to arable) might have a greater potential to contribute to 'desirable' vegetation.

It had been suggested that seeds might be retained in a viable state for longer in waterlogged soils (Chippendale & Milton, 1934). However, the particular grassland communities of interest are generally developed on free-draining soils (Rodwell, 1992b). Hydrological monitoring of fields within the study area revealed that the depth to the water table was greater during the summer months (e.g. Rose and Armstrong, 1996). The ex-arable study site and the Reverting field were subject to largely similar water regimes, with the mean depth to the water table during summer recorded at over 100 cm (compared to 95 cm for undrained meadows). The soils are not permanently waterlogged, certainly near to the surface. If seeds are maintained in a viable state for longer, it will be at greater depths only. It is thus unlikely that there would be increased densities of viable seeds of wet grassland species within soils of the study area.

The wet meadows studied do still flood, particularly over the winter months although heavy rain at any time of year can result in flash floods, and so there is the potential for seed dispersal by floodwater. There are, however, considerable difficulties associated with determining whether hydrochory does in fact play a role in dispersal, although inferences may be drawn from the present study. The seed bank of the ex-arable (SA123) field was very different to those of the grasslands, largely because of the increased density of arable weed seeds. In terms of the numbers of seeds of 'grassland' species the ex-arable seed bank was not dissimilar to the improved grassland seed banks, suggesting that whilst seed densities of arable weeds were increasing within the ex-arable field, those of grassland species may not have been declining as markedly as within other habitats. The study site was regularly cultivated when in arable production, and so it is unlikely that seeds were maintained in the soil, rather that fresh input augmented numbers. It would be difficult, without modelling or further experimentation, to determine patterns of seed movements within the area. The ground level is highest adjacent to the river and falls away towards the new drain. A higher mean number of species and seeds were recorded in the seed bank of the study site nearest the new drain. This increased number within the lowest-lying area of the field was true for both grassland and weed species and does suggest that larger numbers of seeds have been deposited in this area than elsewhere in the field, possibly by floodwater.

Despite the theoretical potential for these seed banks to be of increased utility to habitat restoration, that of the ex-arable study site did not contain sufficient seed of suitable species to enable restoration of species-rich wet grassland vegetation solely from *in situ* propagule sources (chapter 3). Moreover, it became clear that even seed reserves under old, species-rich grasslands were limited and would be unsuitable for restoration of the whole community were the aboveground vegetation to be lost.

The results of this study emphasize the importance of preserving the remaining species-rich grassland resource. Not only are seed banks of ex-arable fields unsuitable for the restoration of a whole community, seed banks of grasslands are generally too dissimilar from the above-ground vegetation, and too species-poor, to be of use in restoration (e.g. Bekker *et al.*, 1997; Bekker, 1998). McDonald *et al.* (1996) had concluded that few species of the *Alopecurus pratensis-Sanguisorba officinalis* could persist in the seed bank for longer than 20 years. This study further revealed that few species indicative of species-rich, unimproved wet grassland possess short-term, let alone long-term, persistent seed banks. If these species are to form part of restored vegetation communities they will need reintroducing from elsewhere, either through natural dispersal or by deliberate reintroduction.

8.2 Arable reversion within the UTT ESA scheme

Within the Upper Thames Tributaries ESA, arable land is eligible for entry into one of two different options. Tier 3A is for the reversion of arable land to 'extensive permanent grassland for the benefit of wildlife and the landscape', with Tier 3B for the reversion of arable land to 'wet grassland for increased benefit to the wildlife'. Both Tiers are targeted on arable land in the floodplain, Tier 3B particularly at areas where the arable land adjoins existing wet grassland. Although the study site had been entered into the Countryside Stewardship scheme prior to the designation of the ESA, had its entry into an agri-environment scheme been delayed by a year, it would have entered Tier 3B of the ESA scheme.

Under the reversion Tiers, a permanent grass sward must be established using at least five species chosen from an approved list of grass species, although seed of wild flower species typical of wet grassland could be included in the seed mixture. The approved list of grasses is similar, with a number of species common to both Tiers 3A and 3B (*Festuca pratensis*, *Festuca arundinacea*, *Festuca rubra*, *Anthoxanthum odoratum*, *Alopecurus pratensis*, *Cynosurus cristatus*, *Agrostis capillaris*, *Phleum pratense*, *Holcus lanatus*). Tier 3A differs, however, in that *Poa pratensis*, *Briza media* and *Trisetum flavescens* may also be chosen.

Natural processes of dispersal and colonisation do not function effectively in the modern fragmented landscape. To ensure the arrival of species at restoration sites, deliberate introduction will be necessary. The introduction of a species-poor seed mixture, such as that recommended by the ESA scheme, would be unlikely to facilitate the establishment of a species-rich sward (chapter 5). Experimental results demonstrated that the sowing of a wider range of species from the target community will be the most successful method for restoring lowland wet grassland to ex-arable sites. According to MAFF (1992), reversion to both extensive permanent grassland and wet grassland would encourage a gradual recolonisation of the characteristic wildlife of river valley grassland. This study suggests that recolonisation of characteristic wildlife will indeed be extremely gradual in the absence of propagule introduction.

When the experiment was begun, the prescription for Tier 3B stated that a permanent grass sward should be established using only suitable species from the approved list, and that 'where practicable, the indigenous grass seed mixture should be of British origin' (MAFF, 1992). However, there was no requirement for seed used to be of native (or local) provenance. The scientific debate over the importance of local provenance has not been resolved. A consultation document (MAFF, 1998), which sets out proposed changes for the ESA scheme within the Upper Thames, now recognises that this is an important question, particularly because this is the only ESA to contain extensive MG4 grasslands. Re-creation of this habitat and the protection of existing sites could be compromised by the introduction of non-native (non-local) varieties. It is now suggested that the seed mixture used in arable reversion schemes is agreed with the Project Officer and, in some cases, 75% of the cost of a seed mixture should be met to ensure native seed of local provenance is used. Another problem with the original prescriptions was that the inclusion of wild flower species in seed mixtures was optional. The consultation document acknowledged that the scheme did little to require the purchase of appropriate native seeds, particularly wildflowers which are an important part of the traditional grassland types that the ESA aims to promote. The suggested added financial incentive (cost of seed mixtures) is thus also aimed at encouraging the inclusion of forb species in the seed mixture. The initial establishment and monitoring of the current field experiment was partly funded by MAFF. Early results were used to underpin the prescriptions for arable reversion within the ESAs and have thus contributed to these changes.

Although the sowing of increased numbers of species has been the most successful method in re-establishing lowland wet grassland on the study site (chapter 5), deliberate re-introduction of species is clearly not the whole story. After six years of reversion, numbers of species in all treatments are still increasing. This suggests continuing immigration of propagules. The introduction of seeds at this relatively late stage in the development of perennial vegetation on the study site would be irrelevant but for the continued presence of bare ground. The maintenance of bare ground has undoubtedly played its part in the continued provision of niches for regeneration, as has the ongoing decline of *Lolium multiflorum*.

The sowing rate employed in the experiment was higher than those currently recommended for arable reversion to grassland (e.g. FRCA, 1999; MAFF, 1992). Stevenson *et al.* (1995) found that lower seeding rates enhanced establishment of calcareous grassland and reduced weed cover, whilst higher rates more rapidly eliminated weeds, but developed a closed sward more quickly. In this study, initial establishment of sown species was poor, partly due to the adverse weather conditions during the early years but probably also due to the poor condition of the seedbed at the time of sowing. In addition, competition and shading by *Lolium multiflorum* was intense in the earlier years. It is unlikely that lower sowing rates would have been successful in this study and may have resulted in species that actually established at low densities failing to establish at all.

The utility of the nurse crop is still open to debate. Following early monitoring, the conclusion was that the nurse crop was unnecessary and generally detrimental to the establishment of a species-rich sward. Previous work had indicated that the nurse crop would not persist, but this was not the case at the study site. Longer-term monitoring has now raised some doubts as to how detrimental the nurse crop has actually been. If the study site had been isolated from sources of propagules, species introductions would have been essential in restoring species-rich grassland vegetation since natural dispersal would be negligible. The detrimental effects of the nurse would then have been more harmful to the developing vegetation. In such cases it would be desirable to have all target species established immediately at densities suitable for persistence in the absence of further immigration. However, the study site is not isolated and does apparently receive propagules through natural dispersal. The early establishment of a closed sward would have prevented further diversification of the restored grassland. If initial establishment of sown species had been better, the use of a nurse crop that declined gradually over a number of years releasing sites for regeneration would have been particularly desirable.

Despite the continued presence of bare ground and the ongoing diversification of experimental plots, it is still the vegetation developed from the seed mixture containing the largest number of species that most closely approaches the desired targets.

8.3 The derivation of targets and objectives

As part of this study, the suitability of different sources and scales of data to inform habitat restoration was assessed (chapter 6). The appropriateness of national species distribution data to the study area was determined by comparison with the results of local survey. Whilst national data have a role to play in determining the geographical limits of occurrence of species and communities (thus ensuring that natural ranges are respected), and hence in planning conservation and restoration, the resolution of the data is too coarse for use alone at the local scale. This was demonstrated in a number of ways. For example, the use of current and historical data can help determine which species (and communities) have declined in frequency and where these declines have been most severe. This knowledge will ensure effective targeting of those communities in need of promotion and of those regions where restoration is both appropriate and necessary. However, the implementation of this approach for wet grasslands (Mountford *et al.*, 1997) appears to be complicated by the ubiquitous nature of many of the species characteristic of this biotope. This could result in areas that have never supported wet grasslands being falsely identified as in need of restoration of this habitat. Conversely, regions that have suffered severe declines in the extent and quality of wet grasslands may go largely undetected if the constituent species still occur within the 10km squares. The problems associated with using data based on presence at such a coarse scale have already been discussed (chapters 1 and 6). Another problem of using coarse resolution data in this study was that species might be present in the 10km square that contains the study area but absent from the study area itself. Without local historical information it will not be possible to determine whether such species were ever present, particularly if they are not on the limits of their natural ranges. Closer examination of the distribution of such species will be required to ensure that conservation effort is appropriate, both with respect to natural ranges but also to frequency and abundance within that range.

Local data were examined prior to restoration of the study site. Site-specific restoration targets were influenced by the fact that the wet meadows local to the study site contained elements of a number of NVC communities. Environmental monitoring of the Upper Thames Tributaries ESA (ADAS, 1998) confirmed this finding: monitored stands of semi-natural grassland in the Ray valley do not match any one NVC community type closely, instead showing affinities to MG4, MG5, MG9 and mire communities. Variation in topography, drainage and flooding frequency can result in the formation of swards containing mosaics of different NVC community types (Killick *et al.*, 1998). Semi-natural meadows and pastures monitored in the Thames and Windrush valleys were apparently a better match to NVC communities than stands in the Ray valley (ADAS, 1998). It is not clear why the Ray valley differs, but there could be a number of contributory reasons. For example, the floodplain of the Ray is wider than the floodplains of the other tributaries. Differences in flood depth between tributaries and increased micro-topographical variation on the floodplain of the Ray could result in this mosaic effect.

Targets set for the experimental restoration of wet grassland of conservation interest within the study area were based upon this 'blurring' of community composition. If the grasslands in the study area had been more uniform in terms of community composition, the use of a local reference habitat to derive seed mixtures would have been unnecessary. Species for (re-) introduction could have been chosen from the published NVC constancy table for the community, selecting only those known to occur within the area and adjusting abundance to approximate to those recorded locally. This does suggest that in some areas, regionally (or locally) refined NVC species lists may be appropriate as the basis of restoration targets.

As already noted, the hydrological regime within the study area was not optimal for the establishment (or maintenance) of species-rich lowland wet grassland during the period of this study. Targets were based on the communities currently present within the study area, but these may actually reflect the effects of drying out. When fields are drained to increase productivity, they are generally also intensified in other ways and the character of the vegetation may change rapidly. For example, fields may be fertilized, stocking rates may be increased or the timing and/or duration of grazing may be altered. In unimproved wet grassland, vegetation change may be much slower, particularly when traditional agricultural management practices are continued. In a species-rich sward, the first casualties of a sub-optimal hydrological regime are likely to be those species adapted to conditions of increased wetness, e.g. sedges and rushes. Because the species-rich mesotrophic communities are so similar floristically, it may be possible for species-rich vegetation to be maintained whilst 'wetland' species are being lost. On impermeable soils with minor variations in surface topography, it might be expected that vegetation approximating to MG4 (and more species-poor wet grassland communities) would persist in depressions as the site dried out while vegetation more similar to MG5 developed in more elevated areas. If this were the case, the reinstatement of an optimal regime in this area might result in the loss of MG5-type communities on the more elevated areas as MG4 swards replace them. Concurrently, the lower-lying areas that currently do support wet grassland communities would change also, possibly becoming less species-rich as the duration of waterlogging or inundation increased. Without further long-term monitoring of the area it will not be possible to determine whether this is the case or not.

Since the target communities have been found to co-occur within individual fields, it is not surprising that they are all maintained in the study area by a similar management regime of mowing and aftermath grazing (with sheep and, more commonly, cattle). Swards are not used for silage production, nor do they receive applications of slurry or herbicide. They are permanent grasslands (not reseeded), with a stocking rate lower than 1.25 livestock units. Whilst fertiliser may be applied, it is generally at low rates of nitrogen application and fields are not drained.

Species targets

Floristic targets for the field experiment were defined using extremely localised knowledge of species assemblages within a reference habitat. Following more extensive vegetation monitoring within the study area, it became apparent that the reference habitat selected is indeed a unique assemblage of plant species within the area. In addition to comprising elements of a number of NVC community types, it also contains species that are not listed within the NVC association tables for mesotrophic grasslands (e.g. *Oenanthe silaifolia*, *Juncus subnodulosus*, *Carex riparia*). When the field experiment was initiated, limited survey results were available for the study area. Species considered potentially suitable for inclusion in treatments were selected from the reference habitat on the basis of their requirements for both soil moisture and available nitrogen. Ellenberg indicator values were used to determine these requirements, despite the fact that values were derived for species within central Europe rather than Britain itself. Despite the recalculation of values for the British flora (Hill *et al.*, 2000), this study continued to use the original Ellenberg indicator values for consistency.

The use of Ellenberg indicator values to derive floristic targets was further investigated using the more extensive species distribution data available for the study area (chapter 7). The selection of species based upon individual indicator values (as used to derive experimental target species) resulted in the selection of species indicative of wet grasslands, whilst the use of a derived 'community mean value' enabled the selection of species indicative of both species-rich wet (MG4) and dry (MG5) grasslands. Although MG8 is a Biodiversity Action Plan target community, the use of the MG8 community mean indicator values is not recommended as it selects species which could be considered 'negative indicators' of successful restoration (i.e. their presence is undesirable), e.g. *Elytrigia repens* and *Cirsium arvense*. The use of indicator values does not identify the 'matrix' grassland species (grasses particularly), which can be determined by calculating the constancy of species within reference habitats or the target communities. The short-term aim for restoration on ex-arable land must be the re-establishment of the appropriate grassland matrix and, in the longer-term, to establish the 'indicator' species within this matrix. A combination of local constancy and indicator species seems appropriate to the definition of targets, the derivation of seed mixtures and the evaluation of success.

8.4 Evaluating restoration

Assessment of restoration success depends upon the identification of objective, measurable targets. There is increasing emphasis on developing rigorous, objective, repeatable methods for the evaluation of success of conservation and restoration management (e.g. Critchley, 2000; Mitchley *et al.*, 2000). Mitchley *et al.* have developed a methodology for monitoring restored sites as part of English Nature's Habitat Restoration Project, based upon measurable attributes for site and habitat characteristics in relation to overall targets. Critchley's method is based on the presence of species suited to the specific biophysical conditions that define the target vegetation. The current evaluation criteria (chapter 4) to some degree encompass both these approaches, although they have not used the same terminology. For example, Mitchley *et al.* have a criterion 'low infestation of pernicious weed species'. Within this study, high ground cover of Class II species is desirable and, if high cover of target species is achieved, then infestation of pernicious weeds will be low. Mitchley *et al.* set yearly targets and, whilst this study did not, an increase in value for each criterion is desirable until the restored vegetation is indistinguishable from the target vegetation.

Since originally defining the evaluation criteria for this study, additional measures have suggested themselves. For example, the target community types are all species-rich, with no single species constantly dominant. In addition to criteria for total species-richness and the contribution of target species, some assessment of the spread of abundance of species may be desirable. Rather than comparing restored abundance to target abundance for each species, a more realistic assessment may be to construct dominance-diversity curves for the restored vegetation. Such graphical representation of the data allows an assessment of progress towards the reference habitat in terms of general character of the vegetation.

Another desirable characteristic of restored vegetation, certainly during early years of establishment, is the continued presence of bare ground as vegetation gaps are generally required for regeneration from seed. The findings of this study further demonstrate that vegetation can continue to diversify with time. The establishment of a dense sward will be detrimental to the subsequent establishment of immigrating propagules.

The importance of selecting appropriate evaluation criteria should not be underestimated. By determining the general characteristics of the target vegetation, criteria may be developed that allow an estimation of the success of habitat restoration. Perhaps more importantly, such criteria may also help identify factors contributing to the failure of restoration. Restoration management can then be tailored to achieve the particular desirable characteristics of the target community.

General habitat measures appear to be more appropriate as objectives and evaluation criteria than the measure of similarity of restored vegetation to some reference habitat or target community in terms of specific species and their abundance.

8.5 Targeting sites

8.5.1 Selection of source fields

The Upper Thames Tributaries ESA is one of two remaining strongholds for MG4 and also contains important areas of MG5 grassland (MAFF, 1998). This suggests that sites chosen for restoration should primarily be those that adjoin areas of MG4 grassland. This study compared different methods for the objective definition of 'source' fields (chapter 6). Since the target communities of MG4 and MG5 are species-rich, the selection of the core of fields for protection in the study area on the basis of absolute numbers of species might be appropriate here particularly since the species-rich swards are a mosaic of community types.

The use of a reserve selection algorithm to select sites for protection is a more systematic approach that the ad hoc selections of reserves in the past (Pressey *et al.*, 1994). Such algorithms can be used to identify the smallest set of sites (number or total area) that represent a defined group of species, communities or landscape attributes. Reserve selection algorithms have been used extensively by other authors (e.g. Pressey *et al.*, 1994, 1996; Csuti *et al.*, 1997) and problems have been widely discussed (e.g. see Prendergast *et al.*, 1999).

Identification of the minimum number of fields necessary to support all species at least once resulted in increased emphasis on rare species, i.e. those known to have substantially declined in frequency are better represented by an optimised selection. If the aim of the ESA scheme were to protect the rare rather than the representative, the use of an optimisation routine to target areas for restoration would be appropriate. If the hydrological regime were changing within the study area, and with the possibility of future climate change, however, the designation (protection) of sites based upon the presence of increased numbers of infrequent and isolated species would not be sensible.

It will be better to focus limited conservation resources on the more achievable goals of conservation, rehabilitation and restoration of representative grassland.

In the current study, the reserve selection algorithm was used very simply – to protect all species (except arable weeds and bryphytes) or all communities present. There are, however, many alternative possibilities for analysis. For example,

- It could be further refined following the identification of species targets to consider certain species only, e.g. rare species, arable weeds, *Carex* species;
- Species occurring with constancy IV or V are likely to be protected, whichever (or however) fields are selected. The selection algorithm could be used to identify the minimum set of sites that would result in the protection of all 'indicator' species;
- Sites could be selected based on protecting species at a defined level of abundance or frequency (approximating to 'healthy' populations of species).

In the current context, the source field selections arising could be considered in one of several ways. If the ESA scheme were to incorporate some element of targeting specific sites rather than accepting the choice of the landowner, then reserve selection algorithm could be used to inform these selections. All species present within the study area can be protected more efficiently than at present and the selections presented in this thesis are the very minimum that should be preserved, but could be considered 'core' areas for conservation in this area. Alternatively, these core areas can be considered as source fields for restoration and, providing these fields are protected, additional fields entering the ESA should add to these areas. The ESA scheme already recognises that arable reversion to wet grassland should be targeted at sites adjacent to existing wet grassland. If the existing valuable resource were protected, particular sites for reversion could be selected, not on an ad hoc basis as at present, but according to pre-defined objectives. Protection of watercourses, promotion of declining species, even aesthetic value could be incorporated into the selection procedure.

8.5.2 Selecting fields for restoration ('sink' fields)

In terms of selecting sites to be restored, the choice will be somewhat arbitrary; certainly while agri-environment schemes operate on a voluntary basis as at present. Given the choice of any (ex-) arable field in the study area, the study site would still be chosen for restoration. Despite constraints imposed by the long duration of arable cultivation, the benefits will be great. Whilst other set aside fields apparently had a more suitable hydrological regime than the study site during the period 1993-1996, the mole drainage within SA123 is degrading and this field does have the greatest potential to receive propagules of a diverse range of species. The restoration of perennial grassland vegetation on this site will buffer the adjacent nature reserve, preventing further input of arable weed seeds and spray drift from agricultural operations. In addition, the cessation of arable agricultural management will be beneficial to the watercourses. Although floodwater cannot return to the River Ray over-bank, maintained mole drainage allows the return of water to the river once the main river levels have dropped below the level of the mole drains. In this way, arable weed seeds and leachate of inorganic inputs would be able to enter the main watercourse.

8.6 Socio-economic considerations

The cost effectiveness of the different methods for the re-establishment of wet grassland was assessed following three years of sward development (Manchester *et al.*, 1999). The assessment was based on the success of the different propagule treatments in re-establishing wet grassland at one site only. Had a number of potential sites been available, costs other than those of the propagules would need to have been considered. For example, costs of water management or agricultural operations may have differed between fields.

The earlier assessment of the effectiveness of the seed treatments at re-establishing species-rich vegetation relative to one another is still largely appropriate following six years of sward establishment. If the experimental vegetation is again ranked on the basis of which treatment performs best according to the criteria used by Manchester *et al.* (1999), the results are very similar. SM3 remains the most effective treatment, followed by SM2 and then HB. The vegetation developed from all treatments is still changing, with numbers of species (including target species) and small-scale species-richness increasing within all treatments, but treatments have largely maintained their position relative to one another.

The general conclusion (after the first three years of the experiment) was that, despite the most comprehensive seed mixture outperforming the remaining propagule treatments, the cost of introducing such a wide range of species was likely to be prohibitively expensive.

The compensatory payment for entering land into the reversion Tier differs between the two options (reversion to extensive permanent grassland or to wet grassland). When the ESA was first designated, landowners received £260 ha^{-1} for reverting arable land to extensive permanent grassland compared to £310 ha^{-1} for reversion to wet grassland. The higher payment for reversion to wet grassland reflects the requirement on Tier 3B to follow Tier 2 wet grassland prescriptions in addition to Tier 1 permanent grassland prescriptions. Under Tier 2, ditches and watercourses must be managed and water levels within them maintained. In addition, livestock grazing is also prohibited from the 1st April to 15th May to avoid disturbance to ground-nesting birds.

Even the higher payment for entering land into Tier 3B only covered the cost of the most basic seed mixture used within the current field experiment (and which approximates to the mixtures recommended at that time by the agri-environment schemes). Manchester *et al.* (1999) suggested that the compensatory payment would need to be increased, or a one-off payment made, to cover the cost of a more diverse seed mixture were it to be proved that increased environmental benefits would accrue. Without additional incentives landowners would not voluntarily include wild flowers in seed mixtures, as the cost would be too high relative to the standard compensatory payment. Even the inclusion of an extremely limited number of commonly available wildflower species which are known to establish from seed (e.g. *Leucanthemum vulgare*, *Ranunculus acris*, *Trifolium pratense*) is likely to double the cost of a seed mixture (e.g. treatment SM2 relative to SM1). However, the current experimental work has demonstrated that the addition of a limited number of common forb species does not result in appreciably better performance than the lower cost options in the medium term in any case. Following changes to the ESA scheme, 75% of the cost of a seed mixture for arable reversion may be met in cases where native seed of local provenance is required to help meet biodiversity enhancement targets (MAFF, 1998). This basically reflects the findings of the current, and other, experiments.

By the year 2000, the payment rates within the UTT ESA had risen to £290 ha^{-1} for the reversion of arable land to extensive permanent grassland (Tier 3A) and £435 ha^{-1} for reversion to wet grassland (Tier 3B). Existing agreement holders, however, with land already in Tier 3B receive payment of £330 ha^{-1}. Similarly, land now entering Tier 2 (wet grassland) will receive £270 ha^{-1}, compared to a rate of £155 ha^{-1} for existing agreements. This rate increase for the wet Tiers largely reflects changes in the prescriptions for the management of existing wet grassland with respect to water management and increased duration of stock exclusion to further benefit breeding waterfowl.

8.7 Further research

This study has raised a number of questions that would benefit from further research:

- The approach to targeting of restoration developed by Mountford *et al.* (1997) may be of greater application to habitat types whose constituent species are largely restricted to a smaller number of communities, i.e. declines in suites of regionally restricted species may better indicate community/biotope decline. Supplementing the national species data with other datasets that further delimit the extent of occurrence of wet grasslands, however, would further refine the approach. For example, climatic, geological and hydrological information could be examined in those regions where national data indicates potential community declines to determine whether in fact the area could support such communities.
- Assessment of the dispersal abilities of target species will aid in determining the necessity of species introductions. Further research on the dispersal and establishment of target species that apparently do not possess even a transient seed bank is necessary if they are to form part of restored vegetation.
- Additional study of the establishment requirements of certain species that are integral to target communities, but that appear difficult to establish (e.g. *Sanguisorba officinalis*) is required. *S. officinalis* germinated within the study site, but never reached maturity and did not persist past the short-term. Determining the reasons for this failure will help improve restoration in both ecological and cost-effectiveness terms.

- Further research at a greater number of sites to determine optimal sowing rates. Lower sowing rates would reduce the cost of seed mixtures and potentially enable the inclusion of a more diverse range of species, but can lower rates be consistently effective at establishing species-rich vegetation?
- Management techniques for promoting diversity and increasing species immigration should be investigated, for example, gap creation in closed swards or the movement of stock from source to 'sink' fields.
- Can the effectiveness of hay as a source of propagules be improved? For target species, what is the optimal timing of the hay cut? What is the optimal age of the bale? Is immediate transfer of hay to the restoration site necessary or can it be left to lie on the donor site (and thus return seed) yet still be effective? What is the optimal 'preparation' of the hay to increase the spatial distribution of diversity?

8.8 Conclusions

This study has demonstrated that the constraints on the restoration of lowland wet grassland are not necessarily those experienced by other grasslands. In particular, residual soil nutrient availability may not be as detrimental to the establishment of floodplain grasslands as it is to species-rich dry grassland or heathland communities.

For the study site, the potential for restoration of species-rich grassland is high. Soil nutrient levels are not too elevated to preclude the maintenance of species-rich vegetation. The use of a diverse seed mixture has facilitated the development of a diverse sward. The site is not isolated from extant species-rich vegetation and receives propagules by natural dispersal, enabling continued diversification. The derivation of floristic targets based upon a local reference habitat resulted in the introduction of species that were typical of both wet (MG4) and dry (MG5) grassland communities. Depending upon the future hydrological regime, the introduction of species common to both means that the restored vegetation can thus develop towards seasonally inundated floodplain swards or towards dry hay meadow vegetation.

This study did not aim to (and has not been able) to determine the hydrological regime required to maintain the target communities, but has established that restoration of ex-arable land to species-rich grassland will be possible. Suitable sites for restoration of wet grassland, certainly in terms of hydrology and soil conditions, will often largely suggest themselves by virtue of being difficult to improve for agriculture. In areas that support wet grassland and wetland communities, land entered into agri-environment schemes will tend to be marginal for agriculture. In the study area, impermeable soils, microtopography and over bank flooding from the River Ray control the hydrological regime. This largely natural regime is widespread in the study area and thus reinstatement of appropriate hydrological regimes is not a constraint on the restoration of wet grasslands.

The most suitable sites for restoration will be those with the greatest potential to receive a wide range of propagules through natural immigration. Arable cultivation depletes reserves of grassland species within soil seed banks such that natural regeneration from *in situ* propagules will result only in a species-poor and weedy sward. If the restoration site is not isolated from extant habitat, the more ubiquitous grassland species may be expected to colonise naturally. However, in the majority of cases of arable reversion, species will need to be deliberately reintroduced either to ensure their arrival or to augment population numbers.

Current and historical national species distribution data can be used to indicate communities and species that have declined, and where conservation effort should be focussed, but realistic targets for restoration at the local-scale require local knowledge. The National Vegetation Classification provides an invaluable framework for the conservation and restoration of vegetation. However, the community definitions were based on a relatively limited number of samples and cannot take into account all regional variants of a community. Particularly where species introductions are necessary, local assessment of vegetation composition and species associations will be required to determine appropriate restoration.

Following identification of the local composition (i.e. species complement) of the target vegetation for restoration, a general assessment of the character of the vegetation will be required for the formulation of specific objectives and the derivation of appropriate criteria for the assessment of progress towards the desired goal.

Where seed mixtures are to be used, and particularly where a more diverse mixture would be appropriate, species chosen should be those that are known to establishment successfully from seed. Early species reintroduction should not focus on habitat-specific, indicator species but on the matrix species, as these are largely common to all target communities. Precise conditions of microtopography and position relative to the main river will determine which community develops. The introduction of MG4 or MG5 specialist species is not likely to be cost-effective, as they will fail where conditions are not appropriate. Even in fields that are not ridge and furrow, microtopographical variation is sufficiently large to enable the development of a number of different communities: only improved grasslands have relatively uniform swards, unimproved swards are mosaics. Restoration should focus initially on re-establishing species-rich vegetation comprised of ubiquitous species whilst maintaining an open sward to allow the establishment of immigrating propagules. Provided the sward is kept open, later introductions of specialist species could be targeted at specific areas within fields once it is clear in which direction the vegetation is developing or, as in the case of SA123, once mole drainage has decayed and the field has regained undrained status. Once this has occurred, the range of water table variation will be reduced and although flooding frequency will not increase, the duration of individual flood events will be extended resulting in conditions more suitable for the maintenance of wetter swards.

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APPENDICES

APPENDIX 3.1 Class II target species.

Species recorded in the reference habitat (June 1993)

Herbaceous species

Achillea ptarmica
Cardamine pratensis
Centaurea nigra
Cerastium fontanum
Cirsium arvense
Cirsium dissectum
Filipendula ulmaria
Galium aparine
Galium palustre
Geranium dissectum
Heracleum sphondylium
Lathyrus pratensis
Leontodon autumnalis
Leucanthemum vulgare
Lotus corniculatus
Lotus pedunculatus
Lychnis flos-cuculi
Lysimachia nummularia
Myosotis discolor
Myosotis laxa ssp. cespitosa
Oenanthe fistulosa
Oenanthe silaifolia
Ophioglossum vulgatum
Plantago lanceolata
Persicaria amphibia
Potentilla reptans
Prunella vulgaris
Ranunuculus acris
Ranunculus flammula
Ranunculus repens
Rhinanthus minor
Rumex acetosa
Rumex conglomeratus
Rumex crispus
Rumex X pratensis
Sanguisorba officinalis
Serratula tinctoria
Silaum silaus
Stellaria graminea
Succisa pratensis
Taraxacum agg.
Thalictrum flavum
Trifolium pratense
Trifolium repens
Vicia cracca
Vicia sativa

Grasses

Agrostis canina
Agrostis capillaris
Agrostis stolonifera
Alopecurus geniculatus
Alopecurus pratensis
Anthoxanthum odoratum
Arrhenatherum elatius
Briza media
Bromus commutatus
Bromus hordeaceus
Bromus racemosus
Cynosurus cristatus
Dactylis glomerata
Deschampsia cespitosa
Festuca pratensis
Festuca rubra
Holcus lanatus
Hordeum secalinum
Lolium perenne
Phleum pratense
Poa annua
Poa trivialis
Trisetum flavescens

Sedges and rushes

Carex disticha
Carex flacca
Carex hirta
Carex nigra
Carex panicea
Juncus acutiflorus
Juncus conglomeratus
Juncus effusus

APPENDIX 3.2 Class I target species, Ellenberg 'F' (moisture indicator value) and 'N' (fertility indicator value) **values**

Species Name	F	N
Achillea ptarmica	8	2
Agrostis capillaris	x	3
Alopecurus pratensis	6	7
Anthoxanthum odoratum	x	x
Briza media	x	2
Cardamine pratensis	7	x
Carex disticha	9	5
Carex nigra	8	2
Carex panicea	7	3
Centaurea nigra	5	?
Cirsium dissectum	8	2
Cynosurus cristatus	5	4
Festuca pratensis	6	6
Festuca rubra	x	x
Filipendula ulmaria	8	4
Holcus lanatus	6	4
Hordeum secalinum	6	5
Lathyrus pratensis	6	6
Leucanthemum vulgare	4	3
Lotus corniculatus	4	3
Lotus pedunculatus	8	4
Lychnis flos-cuculi	6	x
Lysimachia nummularia	6	x
Oenanthe fistulosa	9	5
Oenanthe silaifolia	8	5
Ranunculus acris	x	x
Ranunculus flammula	9	2
Rhinanthus minor	x	2
Rumex acetosa	x	5
Sanguisorba officinalis	7	3
Serratula tinctoria	x	5
Silaum silaus	7	2
Thalictrum flavum	8	2
Trifolium pratense	x	x
Trisetum flavescens	x	5
Vicia cracca	5	x

APPENDIX 3.3 SEED MIXTURES

(Commercially unavailable species are in parentheses)

Seed mixture 1 (Basic)
Alopecurus pratensis, (Anthoxanthum odoratum), Cynosurus cristatus, Festuca rubra, Phleum bertolonii

Seed mixture 2 (intermediate)

Grasses
Agrostis capillaris
Alopecurus pratensis
(Anthoxanthum odoratum)
Cynosurus cristatus
Festuca pratensis
Festuca rubra
Holcus lanatus

Herbs
(Cardamine pratensis)
Filipendula ulmaria
Leucanthemum vulgare
Lotus corniculatus
Ranunculus acris
Trifolium pratense

Seed mixture 3 (comprehensive)

Grasses
Agrostis capillaris
Alopecurus pratensis
(Anthoxanthum odoratum)
Briza media
Cynosurus cristatus
Festuca rubra
Holcus lanatus
Hordeum secalinum
Trisetum flavescens

Herbs
Achillea ptarmica
(Cardamine pratensis)
(Carex disticha or C. panicea)
(Carex nigra)
Centaurea nigra
(Cirsium dissectum)
Filipendula ulmaria
(Juncus acutiflorus)
Lathyrus pratensis
Leucanthemum vulgare
(Lotus pedunculatus)
Lychnis flos-cuculi
(Lysimachia nummularia)
Oenanthe fistulosa
(Oenanthe silaifolia)
Ranunculus acris
(Ranunculus flammula)
Rhinanthus minor
Rumex acetosa
Sanguisorba officinalis
(Serratula tinctoria)
Silaum silaus
Thalictrum flavum
Trifolium pratense
Vicia cracca

APPENDIX 3.4 Comparison of mean percentage cover of Class I target species within the 'target' community between 1993 and 1996. Change: no change (=), increase (+), decrease (-).

Species	1993	1996	change
Achillea ptarmica	<2	<2	=
Agrostis capillaris	<2	2-4	+
Briza media	<2	<2	=
Cardamine pratensis	<2	<2	=
Carex disticha	<2	<2	=
Carex nigra	<2	<2	=
Centaurea nigra	<2	2-4	+
Cirsium dissectum	<2	<2	=
Festuca pratensis	<2	<2	=
Hordeum secalinum	<2	2-4	+
Lathyrus pratensis	<2	<2	=
Leucanthemum vulgare	<2	<2	=
Lotus corniculatus	<2	<2	=
Lotus uliginosus	<2	<2	=
Lychnis flos-cuculi	<2	<2	=
Lysimachia nummularia	<2	<2	=
Oenanthe fistulosa	<2	<2	=
Oenanthe silaifolia	<2	<2	=
Ranunculus flammula	<2	<2	=
Rhinanthus minor	<2	<2	=
Rumex acetosa	<2	<2	=
Serratula tinctoria	<2	<2	=
Silaum silaus	<2	<2	=
Thalictrum flavum	<2	<2	=
Trifolium pratense	<2	<2	=
Trisetum flavescens	<2	<2	=
Vicia cracca	<2	2-4	+
Alopecurus pratensis	2-4	4-6	+
Carex panicea	2-4	<2	-
Filipendula ulmaria	2-4	4-6	+
Ranunculus acris	2-4	4-6	+
Cynosurus cristatus	4-6	4-6	=
Juncus acutiflorus	4-6	2-4	-
Sanguisorba officinalis	4-6	6-8	+
Festuca rubra	6-8	10-12	+
Anthoxanthum odoratum	10-12	8-10	-
Holcus lanatus	14-16	6-8	-

APPENDIX 3.5 Cost of seed (£ kg^{-1}) for Class I species available commercially. Prices as of 1993.

Species	**Cost (£ kg^{-1})**
Agrostis capillaris	6.56
Alopecurus pratensis	9.95
Briza media	149.00
Cynosurus cristatus	2.18
Festuca pratensis	1.15
Festuca rubra	2.31
Holcus lanatus	4.50
Hordeum secalinum	39.95
Phleum bertolonii	3.95
Trisetum flavescens	13.50
Achillea ptarmica	159.00
Centaurea nigra	99.00
Filipendula ulmaria	99.00
Lathyrus pratensis	320.00
Leucanthemum vulgare	39.00
Lotus corniculatus	99.00
Lychnis flos-cuculi	135.00
Ranunculus acris	59.00
Rhinanthus minor	99.00
Rumex acetosa	77.50
Silaum silaus	399.00
Trifolium pratense	75.00
Vicia cracca	399.00

APPENDIX 3.6 Price of Seed Mixtures. Figures in parentheses refer to cost calculated using abundance as recorded in 1996 but seed prices as of 1993.

Treatment	Cost (£) plot (18m x 38 m)	Hectare (ha)
Seed Mix 1 – nurse	£10.69	£156.29 (£163.89)
Seed Mix 1 + nurse	£11.54	£168.71
Seed Mix 2 – nurse	£42.97	£628.22 (£685.38)
Seed Mix 2 + nurse	£43.82	£640.64
Seed Mix 3 – nurse	£77.49	£1132.89 (£1311.40)
Seed Mix 3 + nurse	£78.84	£1152.63

Appendix 4.1 Mean numbers of seeds m^{-2} ($\pm$ standard error) in the soil seed bank.
Individual species listed are those with counts greater than 10 seeds per field

Species	IE	IW	PL	LH	REV	SA123	Significance
Whole field	**15978 ± 2922**	**13194 ± 1454**	**35074 ± 3066**	**50497 ± 3977**	**10027 ± 703**	**19852 ± 1654**	
Agrostis sp.	0	207 ± 83	1124 ± 288	1384 ± 255	174 ± 49	0	
Alopecurus geniculatus	0	0	0	0	0	1538 ± 345	
Alopecurus myosuroides	0	0	0	0	0	3549 ± 714	
Alopecurus pratensis	0	334 ± 119	2576 ± 437	1453 ± 171	774 ± 234	53 ± 16	
Anthoxanthum odoratum	1466 ± 506	1512 ± 317	2656 ± 443	6306 ± 654	0	0	
Avena fatua	0	0	0	0	0	488 ± 76	
Bromus sp.	377 ± 181	0	159 ± 71	0	0	0	
Cynosurus cristatus	0	0	0	277 ± 103	0	0	
Deschampsia cespitosa	0	0	239 ± 76	571 ± 157	0	0	
Festuca pratensis	0	0	418 ± 146	0	0	0	
Holcus lanatus	921 ± 491	1035 ± 200	7242 ± 1027	15172 ± 1790	116 ± 43	74 ± 21	
Lolium sp.	880 ± 600	223 ± 65	587 ± 190	450 ± 91	1092 ± 177	4207 ± 580	
Poa trivialis	10052 ± 1927	5602 ± 1064	13707 ± 1563	7698 ± 1000	5650 ± 582	5263 ± 757	
Triticum aestivum	0	0	0	0	0	647 ± 111	
All Poaceae	**14135 ± 2581**	**9183 ± 1211**	**28907 ± 2876**	**33613 ± 3002**	**8016 ± 716**	**15947 ± 1442**	**P 0.000**
Carex disticha	0	0	0	147 ± 56	0	0	
Juncus conglomeratus	440 ± 146	955 ± 263	786 ± 187	10605 ± 2451	470 ± 70	414 ± 88	
All Juncaceae/Cyperaceae	**461 ± 147**	**1066 ± 319**	**855 ± 189**	**10829 ± 2468**	**557 ± 84**	**467 ± 97**	**P 0.000**

Appendix 4.1 (continued). Mean numbers of seeds m^{-2} ($\pm$ standard error) in the soil seed bank.
Individual species listed are those with counts greater than 10 seeds per field

Species	IE	IW	PL	LH	REV	SA123	Significance
Atriplex prostrata	0	0	0	0	0	64 ± 30	
Brassica sp.	0	0	0	0	0	149 ± 44	
Cardamine pratensis	0	271 ± 99	1542 ± 340	2007 ± 371	0	0	
Centaurea nigra	0	0	0	112 ± 34	0	0	
Chenopodium album	0	0	0	0	0	53 ± 22	
Chenopodium polyspermum	0	0	0	0	101 ± 33	223 ± 66	
Cirsium vulgare	0	0	0	0	260 ± 57	228 ± 62	
Epilobium ciliatum	0	0	0	0	94 ± 27	292 ± 62	
Lychnis flos-cuculi	0	239 ± 100	119 ± 41	251 ± 66	0	0	
Myosotis discolor	0	0	0	0	80 ± 80	0	
Persicaria amphibia	0	0	0	0	0	138 ± 42	
Persicaria lapathifolia	0	0	0	0	0	69 ± 44	
Plantago major	0	0	0	0	0	69 ± 28	
Polygonum aviculare	0	0	0	0	0	69 ± 30	
Ranunculus flammula	0	0	0	943 ± 416	0	101 ± 46	
Ranunculus sp.	482 ± 210	796 ± 198	2825 ± 603	1963 ± 337	0	0	
Rumex acetosa	0	0	0	208 ± 125	0	0	
Rumex sp.	0	0	0	0	0	58 ± 26	
Senecio jacobea	0	0	0	0	94 ± 59	0	
Senecio vulgaris	0	366 ± 317	0	0	0	74 ± 45	
Sonchus asper	0	0	0	0	116 ± 34	605 ± 133	
Stellaria media	314 ± 293	462 ± 199	209 ± 62	0	195 ± 100	334 ± 160	
Trifolium repens	0	0	0	95 ± 35	0	0	
Tripleurospermum inodorum	0	0	0	0	0	658 ± 530	
All dicotyledons	**1382 ± 411**	**2944 ± 508**	**5312 ± 756**	**6055 ± 885**	**1454 ± 184**	**3438 ± 660**	**P 0.000**

Appendix 4.2 Mean numbers of seeds m^{-2} (± s.e.). Species listed are those with counts of less than 10 seeds per field

Species	IE	IW	PL	LH	REV	SA123
Agrostis capillaris	125.65 ± 61.25	0	0	0	0	0
Agrostis stolonifera	0	0	0	0	0	37.14 ± 18.19
Alopecurus geniculatus	0	0			7.23 ± 7.23	0
Alopecurus myosuroides	0	15.92 ± 15.92	19.89 ± 13.89		65.11 ± 38.31	0
Alopecurus pratensis	188.47 ± 82.59	0	0	0	0	0
Anthoxanthum odoratum	0	0	0	0	0	21.22 ± 10.05
Atriplex patula	20.94 ± 20.94	15.92 ± 15.92	29.84 ± 16.78	17.30 ± 12.10	14.47 ± 10.14	0
Avena fatua	0	0	0	8.65 ± 8.65	0	0
Brassica sp.	0	0	9.95 ± 9.95	0	7.23 ± 7.23	0
Bromus sp.	0	143.24 ± 50.75	0	60.55 ± 32.65	0	0
Capsella bursa-pastoris	0	0	0	0	0	5.31 ± 5.31
Carex sp.	20.94 ± 20.94	95.49 ± 66.10	9.95 ± 9.95	0	28.94 ± 14.06	26.53 ± 15.42
Carex riparia	0	0	59.68 ± 22.75	51.90 ± 29.29	0	0
Centaurea nigra	0	15.92 ± 15.92	59.68 ± 30.39	0	0	5.31 ± 5.31
Cerastium fontanum	0	31.83 ± 22.03	0	25.95 ± 19.17	0	10.61 ± 10.61
Chenopodium album	0	0	0	0	7.23 ± 7.23	0
Chenopodium polyspermum	0	31.83 ± 22.03	0	0	0	0
Cirsium arvense	0	0	9.95 ± 9.95	8.65 ± 8.65	36.17 ± 18.68	0
Cirsium vulgare	188.47 ± 167.68	127.32 ± 85.09	59.68 ± 33.56	17.30 ± 12.10	0	0
Conium maculatum	0	0	9.95 ± 9.95	8.65 ± 8.65	0	0
Cynosurus cristatus	83.77 ± 83.77	15.92 ± 15.92	89.52 ± 41.51	0	0	0
Dactylis glomerata	0	15.92 ± 15.92	49.74 ± 29.15	34.60 ± 24.19	14.47 ± 14.47	5.31 ± 5.31
Daucus carota	0	0	0	0	0	5.31 ± 5.31
Deschampsia cespitosa	20.94 ± 20.94	0	0	0	65.11 ± 28.77	26.53 ± 13.40
Elytrigia repens	0	0	9.95 ± 9.95	34.60 ± 24.19	14.47 ± 14.47	21.22 ± 12.62
Epilobium ciliatum	104.71 ± 51.30	63.66 ± 49.70	29.84 ± 22.01	0	0	0
Epilobium hirsutum	20.94 ± 20.94	0	29.84 ± 16.78	8.65 ± 8.65	36.17 ± 15.57	21.22 ± 12.62
Festuca pratensis	20.94 ± 20.94	79.577 ± 39.79	0	60.55 ± 30.21	7.23 ± 7.23	15.92 ± 8.87
Filipendula ulmaria	0	15.92 ± 15.92	0	0	0	0
Geranium dissectum	20.94 ± 20.94	0	9.95 ± 9.95	17.30 ± 12.10	0	37.14 ± 22.49
Hordeum secalinum	0	0	0	25.95 ± 19.17	0	0
Juncus articulatus	0	15.92 ± 15.92	0	25.95 ± 14.65	21.70 ± 12.30	15.92 ± 8.87

Appendix 4.2 (continued). Mean numbers of seeds m^{-2} ($\pm$ s.e.). Species listed are those with counts less than 10 seeds per field

Species	IE	IW	PL	LH	REV	SA123
Juncus bulbosus	0	0	0	0	36.17 ± 18.68	5.31 ± 5.31
Leontodon autumnalis	0	0	0	25.95 ± 25.95	0	5.31 ± 5.31
Leucanthemum vulgare	0	0	9.95 ± 9.95	17.30 ± 12.10	7.23 ± 7.23	0
Lotus corniculatus	0	0	9.95 ± 9.95	0	0	0
Luzula campestris	0	0	0	0	0	5.31 ± 5.31
Lychnis flos-cuculi	0	0	0	0	7.23 ± 7.23	0
Myosotis discolor	41.88 ± 28.78	143.24 ± 64.49	19.89 ± 13.89	25.95 ± 14.65	0	0
Persicaria amphibia	0	0	0	0	7.23 ± 7.23	0
Phleum pratense	0	0	29.84 ± 16.78	77.85 ± 29.32	36.17 ± 15.57	0
Picris echioides	20.94 ± 20.94	63.66 ± 29.78	49.74 ± 21.07	0	50.64 ± 18.05	47.75 ± 15.55
Plantago lanceolata	20.94 ± 20.94	0	0	0	0	0
Plantago major	20.94 ± 20.94	15.92 ± 15.92	19.89 ± 13.89	51.90 ± 26.55	28.94 ± 14.06	0
Polygonum aviculare	0	0	0	0	7.23 ± 7.23	0
Polygonaceae spp.	0	0	0	0	7.23 ± 7.23	0
Potentilla erecta	0	0	0	17.30 ± 17.30	0	0
Prunella vulgaris	0	0	0	17.30 ± 17.30	0	5.31 ± 5.31
Ranunculus sp.	0	0	0	0	0	21.22 ± 16.60
Ranunculus ficaria	0	0	9.95 ± 9.95	0	0	0
Ranunculus flammula	0	0	0	0	7.23 ± 7.23	0
Ranunculus sceleratus	0	47.75 ± 34.99	0	0	7.23 ± 7.23	0
Rubus fruticosus	0	0	9.95 ± 9.95	0	0	0
Rumex acetosa	0	31.83 ± 22.03	29.84 ± 16.78	0	0	0
Rumex crispus	0	0	0	8.65 ± 8.65	0	0
Rumex obtusifolius	0	0	9.95 ± 9.95	0	0	0
Scrophularia auriculata	0	0	0	0	0	5.31 ± 5.31
Senecio aquaticus	0	0	0	0	21.70 ± 21.70	0
Senecio vulgaris	0	0	49.74 ± 21.07	34.60 ± 20.79	0	0
Silaum silaus	0	0	19.89 ± 13.89	17.30 ± 17.30	7.23 ± 7.23	5.31 ± 5.31
Sinapis arvensis	0	0	0	0	0	5.31 ± 5.31
Sonchus asper	0	47.75 ± 34.99	49.74 ± 21.07	60.55 ± 24.63	0	0
Stellaria media	0	0	0	25.95 ± 19.17	0	0
Taraxacum agg.	20.94 ± 20.94	31.83 ± 22.03	0	25.95 ± 19.17	43.41 ± 24.59	21.22 ± 12.62

Appendix 4.2 (continued). Mean numbers of seeds m^{-2} ($\pm$ s.e.). Species listed are those with counts less than 10 seeds per field

Species	IE	IW	PL	LH	REV	SA123
Trifolium pratense	0	31.83 $\pm$ 22.03	19.89 $\pm$ 13.89	25.95 $\pm$ 14.65	0	0
Trifolium repens	41.88 $\pm$ 41.88	47.75 $\pm$ 26.39	9.95 $\pm$ 9.95	0	21.70 $\pm$ 16.06	21.22 $\pm$ 10.05
Urtica dioica	41.88 $\pm$ 28.78	47.75 $\pm$ 26.39	49.74 $\pm$ 25.44	17.30 $\pm$ 12.10	21.70 $\pm$ 16.06	5.31 $\pm$ 5.31
Vicia cracca	20.94 $\pm$ 20.94	0	0	0	0	0
Viola arvensis	0	0	0	0	7.23 $\pm$ 7.23	0

Appendix 4.3 Species recorded in the vegetation of SA123 prior to experimental establishment in 1993.

Forbs

Anthriscus sylvestris
Atriplex prostrata
Brassica rapa
Calystegia sepium
Cardamine pratensis
Centaurea nigra
Chenopodium album
Cirsium arvense
Cirsium vulgare
Conium maculatum
Coronopus squamatus
Crepis biennis
Dipsacus fullonum
Epilobium ciliatum
Epilobium hirsutum
Epilobium montanum
Galeopsis tetrahit
Geranium dissectum
Lactuca serriola
Leontodon autumnalis
Matricaria discoidea
Matricaria recutita
Persicaria amphibia
Persicaria hydropiper
Persicaria maculosa
Picris echioides
Plantago major
Ranunculus flammula
Ranunculus repens
Rorippa palustris
Rumex conglomeratus
Rumex crispus
Senecio vulgaris
Sonchus asper
Stellaria media
Taraxacum agg.
Trifolium dubium
Trifolium repens
Veronica catenata
Vicia faba

Grasses

Agrostis stolonifera
Alopecurus geniculatus
Alopecurus myosuroides
Alopecurus pratensis
Anisantha sterilis
Avena fatua
Bromus commutatus
Bromus hordeaceus
Dactylis glomerata
Festuca pratensis
Lolium multiflorum
Lolium perenne
Poa annua
Poa trivialis
Triticum aestivum

Sedges and rushes

Juncus effusus

Appendix 4.4 Vegetation survey: Mean percentage ground cover of species within study fields

	1993					1996				
Species	**LH**	**PL**	**IW**	**IE**	**REV**	**LH**	**PL**	**IW**	**IE**	**REV**
Achillea ptarmica	0.02	0.1	0.2			0.01	0.02	0.01		
Agrostis canina	2.1	0.3	0.1		0.02	3.57	0.62	0.04		
Agrostis capillaris	0.1	1.4	0.2	0.1		3.7	0.78	1.72	0.58	
Agrostis stolonifera	2.8	8.3	3.4	1.4	0.2	2.64	6.31	1.48	0.43	0.84
Alopecurus geniculatus	0.1	0.03			2.2			0.01		0.44
Alopecurus pratensis	2.5	8.9	2.2	0.7	0.1	7.82	16.24	2.63	3.73	0.36
Anthoxanthum odoratum	10.1	9.6	7.8	6.8		9.68	5.81	2.28	4.12	0.45
Anthriscus sylvestris					0.02	0.01				0.01
Arrhenatherum elatius	0.6	0.4				0.31	0.1		0.01	
Bromus commutatus	0.3	0.7	1.4	1.2	0.1	1.62	3.02	4.77	0.04	1.27
Bromus h. hordeaceus		0.3	2	6.5				2.86	4	0.01
Bromus racemosus	0.1	0.1	0.04	0.1		1.82	3.1	7.21	32.23	0.39
Cardamine hirsuta										0.01
Cardamine pratensis	1.5	1	0.4		0.4	1.56	1.27	0.38	0.02	0.02
Carex disticha	0.7	0.4			0.8	0.37	0.93			
Carex flacca	0.7	0.1				0.01				
Carex hirta	0.02					0.08				
Carex nigra	1.3	1.1				0.43	0.93			
Carex panicea	2.9	0.4				0.39	0.19			
Carex riparia	3.3	2.2				1.46	0.95			
Carex spicata		0.1								
Centaurea nigra	2	0.8	0.2	0.2		3.8	0.38	0.04	0.32	
Cerastium fontanum	0.04	0.1		0.1	0.1	0.01	0.07	0.12	0.01	0.03
Cirsium arvense	0.1	0.2	0.8	1.5	0.1	0.06		0.68	4.32	0.54
Cirsium dissectum	1.2					0.9				

Appendix 4.4 (continued)

	1993					1996				
Species	**LH**	**PL**	**IW**	**IE**	**REV**	**LH**	**PL**	**IW**	**IE**	**REV**
Crataegus monogyna							0.01			0.01
Cynosurus cristatus	4.9	2.9	2.9	2.4	0.1	4.12	3.34	5.34	4.5	0.07
Dactylis glomerata	0.6	0.8		0.2		0.29	0.33		0.01	0.04
Deschampsia cespitosa	2.5	1.7	0.6	0.1		2.24	1.11	0.01		
Dipsacus fullonum					0.02					
Elytrigia repens	0.4	3.3	3.2	9.4	0.5	0.16	0.6	1.29	7.97	1.02
Festuca arundinacea						0.02	0.02			
Festuca pratensis	8	3	1			0.85	0.04	0.04		0.01
Festuca rubra	7.1	8.8	6.7	1.7	0.1	10.55	15.5	15.12	1.7	
Filipendula ulmaria	2.4	1.1	0.5			5.42	1.21	0.12		
Galium aparine					0.02	0.02	0.01			
Galium palustre	0.1									
Geranium dissectum	0.02			0.2	0.1	0.98	2.08	0.01	0.01	
Glyceria maxima		0.5								
Heracleum sphondylium	0.3	0.1				0.18	0.29			
Holcus lanatus	15.4	14.2	15.8	13.2	0.3	7.29	5.55	3.73	6.27	0.38
Hordeum secalinum	1.6	2.8	2.9	0.9	0.2	3.63	6.81	2.28	1.35	0.64
Juncus acutiflorus	4.7	0.9				3.21	0.6			
Juncus articulatus	0.4									
Juncus conglomeratus	6.7	1.1				0.22	0.02			
Juncus effusus	0.1					0.27				
Lathyrus pratensis	0.7	1.6	0.5	0.1		1.56	2.86	4.65	0.13	
Leontodon autumnalis	0.1				0.04	0.19				0.01
Lolium multiflorum			1.5	6.3	8.2			0.01	0.62	0.14
Lolium perenne	0.6	2.7	8.4	13.6	26.5	0.56	1.93	12.89	16.81	33.5
Lotus corniculatus	0.7	0.5	0.3			0.23	0.7	0.12	0.04	

Appendix 4.4 (continued)

	1993					1996				
Species	**LH**	**PL**	**IW**	**IE**	**REV**	**LH**	**PL**	**IW**	**IE**	**REV**
Lotus pedunculatus	0.3	0.03				0.12				
Luzula campestris						0.04				
Lychnis flos-cuculi	0.6	0.2	0.5			0.19	0.01	0.05	0.01	0.02
Lysimachia nummularia	0.02					0.06				
Myosotis discolor						0.01		0.01		0.04
Myosotis laxa	0.04					0.02				0.01
Oenanthe fistulosa	0.3		0.1		0.1	0.09		0.72	0.01	
Oenanthe silaifolia	0.3	0.02				1.31				0.04
Ophioglossum vulgatum	0.04	0.02				0.01	0.01	0.01		
Persicaria amphibia	0.1	0.2	0.2				0.01	0.01	0.01	
Phalaris arundinacea	0.02	0.1				0.01	0.19			
Phleum pratense	0.8	0.4	0.3	0.9	16.1	0.48	0.12	0.17	0.23	12
Picris echioides					0.02					
Plantago lanceolata	0.1	0.1		0.4		0.04	0.02		0.08	
Plantago major					0.2					
Poa annua	0.02		0.04		0.7			0.02		0.15
Poa humilis							0.13	0.01	0.16	
Poa pratensis		0.1					0.05	0.04	0.08	
Poa trivialis	5.7	13.7	9.4	13.2	17.2	2.64	3.55	5.65	5.58	21.2
Potentilla reptans	0.1	0.2	0.2			0.02	0.12	0.64	0.01	
Prunella vulgaris	0.02				0.04	0.04				
Prunus spinosa										
Ranunculus acris	3.1	3.6	7	2.8	0.04	5.17	6.36	5.8	4.41	0.24
Ranunculus bulbosus		0.1								
Ranunculus ficaria									0.04	
Ranunculus flammula	0.5				0.3	0.04				

Appendix 4.4 (continued)

	1993					1996				
Species	**LH**	**PL**	**IW**	**IE**	**REV**	**LH**	**PL**	**IW**	**IE**	**REV**
Ranunculus repens	1.1	3.2	1.2	0.5	1.7	2.51	3.22	4.69	0.69	6.98
Rhinanthus minor	0.1	0.2				0.02	0.88			
Rosa canina s.l.			0.04							
Rumex acetosa	0.4	0.2	1.2		0.02	0.23	0.12	1.02	0.04	
Rumex conglomeratus					0.1	0.01		0.04		0.01
Rumex crispus	0.1	0.5	0.3	0.2	0.2	0.24	0.19	0.17	0.5	0.49
Rumex obtusifolius										0.04
Sanguisorba officinalis	5.6	2.5	5.5	1.9		7.72	1.42	13.76	0.51	
Senecio erucifolius					0.1					0.02
Serratula tinctoria	0.1					0.01				
Silaum silaus	0.4	0.1	0.1	0.1		0.44	0.23	0.13	0.04	
Sonchus asper					0.02					
Stellaria graminea	0.1									
Succisa pratensis	0.02					0.06				
Taraxacum agg.		0.2	0.9	1.6	1.1	0.04	0.1	0.61	1.32	1.57
Thalictrum flavum	0.04					0.02			0.08	
Trifolium dubium				0.3	1.7	0.04	0.19	0.96	2.26	11.7
Trifolium pratense	0.2	0.7	1.7	0.9	0.6	0.19	0.38	3.17	1.67	0.05
Trifolium repens	0.5	0.6	1	0.3	18.6	0.09	0.1	2.46	2.66	6.42
Trisetum flavescens	0.1	0.02				0.2				
Veronica serpyllifolia		0.02								
Vicia cracca	1	0.9	0.04	0.05		2.6	1.77	1.65	0.35	0.02
Vicia hirsuta						0.08				
Vicia sativa ssp nigra	0.3	0.4	0.4	0.3		0.47	0.71		0.24	0.01

APPENDIX 5.21. NATURAL REGENERATION (NR). Treatment means (± s.e.)

	1994	1995	1996	1997	1999
Class I target species					
Achillea ptarmica					
Agrostis capillaris		0.007 ± 0.004	0.07 ± 0.04		0.34 ± 0.30
Alopecurus pratensis	0.60 ± 0.28			0.007 ± 0.007	0.003 ± 0.003
Anthoxanthum odoratum					
Briza media					
Cardamine pratensis					
Centaurea nigra					
Cynosurus cristatus				0.003 ± 0.003	0.31 ± 0.11
Festuca pratensis	0.033 ± 0.033				
Festuca rubra	0.003 ± 0.003				0.81 ± 0.51
Filipendula ulmaria					
Holcus lanatus	0.80 ± 0.31	0.78 ± 0.37	2.34 ± 1.04	3.98 ± 1.43	26.80 ± 4.95
Hordeum secalinum					0.003 ± 0.003
Lathyrus pratensis					
Leucanthemum vulgare				0.003 ± 0.003	
Lotus corniculatus					
Lotus pedunculatus					
Lychnis flos-cuculi	0.01 ± 0.01			0.033 ± 0.033	0.13 ± 0.13
Oenanthe silaifolia					
Ranunculus acris	0.04 ± 0.03		0.0003 ± 0.0003		
Ranunculus flammula	0.39 ± 0.15	0.013 ± 0.007	0.001 ± 0.0004		
Rhinanthus minor					
Rumex acetosa		0.033 ± 0.033		0.033 ± 0.033	
Sanguisorba officinalis					

APPENDIX 5.21. NATURAL REGENERATION (NR). Continued.

	1994	**1995**	**1996**	**1997**	**1999**
Silaum silaus					
Trifolium pratense	0.003 ± 0.003	0.033 ± 0.033	0.17 ± 0.17	0.50 ± 0.39	0.97 ± 0.70
Trisetum flavescens				0.13 ± 0.13	0.003 ± 0.003
Vicia cracca					0.033 ± 0.033
Class II target species					
Agrostis canina			0.034 ± 0.034		0.31 ± 0.17
Agrostis stolonifera	0.003 ± 0.003	0.303 ± 0.26	0.17 ± 0.13	1.15 ± 0.79	5.74 ± 1.95
Alopecurus geniculatus	0.37 ± 0.22	2.70 ± 2.16	1.35 ± 1.11	0.003 ± 0.003	0.003 ± 0.003
Arrhenatherum elatius				0.35 ± 0.25	1.033 ± 0.82
Bromus commutatus	2.61 ± 1.0		0.11 ± 0.10		0.003 ± 0.003
Bromus hordeaceus					0.003 ± 0.003
Bromus racemosus		0.133 ± 0.133		0.003 ± 0.003	0.72 ± 0.31
Cerastium fontanum					
Cirsium arvense	0.38 ± 0.16	0.60 ± 0.33	2.40 ± 1.43	3.50 ± 1.88	6.0 ± 0.70
Dactylis glomerata					0.003 ± 0.003
Deschampsia cespitosa	0.07 ± 0.05	0.04 ± 0.03	0.13 ± 0.13	0.033 ± 0.033	
Galium aparine					
Geranium dissectum	0.23 ± 0.15		0.17 ± 0.08	0.21 ± 0.20	0.01 ± 0.007
Heracleum sphondylium					
Juncus conglomeratus					
Lolium perenne		0.003 ± 0.003	0.34 ± 0.13		0.51 ± 0.31
Persicaria amphibia			0.0003 ± 0.0003		
Phleum pratense		0.66 ± 0.39	0.27 ± 0.12	2.44 ± 1.14	6.91 ± 3.00
Plantago lanceolata					
Poa annua	0.003 ± 0.003	0.003 ± 0.003	0.0003 ± 0.0003		

APPENDIX 5.21. NATURAL REGENERATION (NR). Continued.

	1994	1995	1996	1997	1999
Poa trivialis	22.43 ± 1.96	9.23 ± 1.92	2.73 ± 0.79	2.85 ± 1.07	4.97 ± 1.05
Prunella vulgaris	0.04 ± 0.03				
Ranunculus repens	4.88 ± 1.91	2.19 ± 0.99	5.27 ± 2.56	1.21 ± 0.47	4.17 ± 1.70
Rumex conglomeratus					
Rumex crispus	0.20 ± 0.15	0.03 ± 0.03	0.13 ± 0.13	0.07 ± 0.07	0.04 ± 0.03
Taraxacum agg.	0.21 ± 0.09	0.52 ± 0.15	0.97 ± 0.46	0.95 ± 0.28	0.51 ± 0.18
Trifolium repens	1.35 ± 0.30	1.80 ± 1.10	9.27 ± 4.55	37.0 ± 11.8	1.51 ± 0.50
Vicia sativa					0.003 ± 0.003
<u>Other species</u>					
Alopecurus myosuroides	5.37 ± 1.86		0.27 ± 0.23		
Anisantha sterilis					
Avena fatua	0.003 ± 0.003		0.54 ± 0.29	0.35 ± 0.25	0.003 ± 0.003
Barbula unguiculata					
Brachythecium rutabulum	0.003 ± 0.003				0.33 ± 0.21
Brassica rapa					
Bryum sp.					
Calliergon cuspidatum					0.37 ± 0.33
Ceratodon purpureus					
Chenopodium album					
Cirsium vulgare	0.08 ± 0.07		0.11 ± 0.07	0.14 ± 0.07	0.10 ± 0.10
Crataegus monogyna					0.007 ± 0.004
Dicranella varia					
Elytrigia repens				0.033 ± 0.003	0.27 ± 0.18
Epilobium ciliatum	0.24 ± 0.09				
Epilobium hirsutum	0.11 ± 0.06				

APPENDIX 5.21. NATURAL REGENERATION (NR). Continued.

	1994	1995	1996	1997	1999
Eurhynchium praelongum					
Fissidens taxifolius					0.84 ± 0.75
Galium verum					
Geranium molle					
Geum urbanum					
Hypochaeris radicata					
Lactuca serriola	0.04 ± 0.03				
Lamium purpureum					
Lolium multiflorum	54.0 ± 4.74	40.70 ± 5.35	50.73 ± 6.41	35.77 ± 3.52	27.63 ± 4.23
Lythrum salicaria					
Matricaria discoidea	0.033 ± 0.033				
Matricaria recutita					
Persicaria maculosa					
Phleum bertolonii			0.47 ± 0.39	0.08 ± 0.05	
Picris echioides					0.033 ± 0.033
Plantago major	0.04 ± 0.03				
Poa pratensis			0.04 ± 0.04		
Prunus spinosa					
Rosa canina					0.003 ± 0.003
Rumex obtusifolius	0.003 ± 0.003			0.07 ± 0.07	
Sambucus nigra					
Senecio aquaticus					
Senecio erucifolius				0.033 ± 0.033	0.07 ± 0.07
Senecio jacobea					
Solanum dulcemara					

APPENDIX 5.21. NATURAL REGENERATION (NR). Continued.

	1994	1995	1996	1997	1999
Sonchus asper	0.14 ±0.07			0.033 ±0.033	
Sonchus oleraceus				0.003 ±0.003	
Stellaria media	0.48 ±0.14				
Trifolium dubium	0.24 ±0.11	0.03 ±0.03	0.54 ±0.29	0.18 ±0.13	0.013 ±0.01
Trifolium hybridum					
Trifolium medium					
Triticum aestivum	3.80 ±0.50				
Urtica dioica					
Veronica beccabunga					
Veronica serpyllifolia					

APPENDIX 5.22. HAY BALES (HB), treatment means (± s.e.)

	1994	1995	1996	1997	1999
Class I target species					
Achillea ptarmica					
Agrostis capillaris			0.07 ±(0.04	0.11±0.10	0.07 ±0.07
Alopecurus pratensis	0.51 ±0.25			0.003 ±0.003	0.033 ±0.033
Anthoxanthum odoratum	0.27 ±0.17	0.033 ±0.033	0.17 ±0.10	0.17 ±0.17	1.87 ±0.99
Briza media					
Cardamine pratensis	0.003 ±0.003				
Centaurea nigra					0.133 ±0.13
Cynosurus cristatus				0.23 ±0.16	0.54 ±0.19
Festuca pratensis			0.0003 ±0.0003		
Festuca rubra					0.20 ±0.20
Filipendula ulmaria					
Holcus lanatus	0.31 ±0.15	0.77 ±0.20	1.80 ±0.54	5.61 ±0.99	34.53 ±4.71
Hordeum secalinum	0.003 ±0.003		0.07 ±0.07	0.23 ±0.15	0.27 ±0.27
Lathyrus pratensis					
Leucanthemum vulgare					0.13 ±0.13
Lotus corniculatus					0.033 ±0.033
Lotus pedunculatus					0.13 ±0.13
Lychnis flos-cuculi		0.04 ±0.03	0.001 ±0.0004		0.10 ±0.07
Oenanthe silaifolia					
Ranunculus acris	0.02 ±0.01				0.03 ±0.03
Ranunculus flammula	0.09 ±0.05	0.003 ±0.003	0.03 ±0.03		
Rhinanthus minor					
Rumex acetosa	0.007 ±0.005		0.0003 ±0.0003		
Sanguisorba officinalis					

APPENDIX 5.22. HAY BALES (HB), Continued

	1994	1995	1996	1997	1999
Silaum silaus					
Trifolium pratense	0.07 ± 0.05	0.003 ± 0.003	0.10 ± 0.10	0.07 ± 0.07	0.27 ± 0.16
Trisetum flavescens					
Vicia cracca					
Class II target species					
Agrostis canina		0.003 ± 0.003	0.04 ± 0.04		0.033 ± 0.033
Agrostis stolonifera	0.03 ± 0.03	0.20 ± 0.13	0.30 ± 0.26	0.77 ± 0.39	6.91 ± 2.74
Alopecurus geniculatus	3.27 ± 1.22	2.17 ± 1.21	0.53 ± 0.32	0.11 ± 0.08	0.007 ± 0.007
Arrhenatherum elatius				0.003 ± 0.003	0.133 ± 0.133
Bromus commutatus	0.98 ± 0.25		0.17 ± 0.17	0.20 ± 0.20	0.17 ± 0.17
Bromus hordeaceus					
Bromus racemosus		0.07 ± 0.07			0.81 ± 0.34
Cerastium fontanum	0.08 ± 0.07	0.003 ± 0.003			
Cirsium arvense	0.24 ± 0.14	0.42 ± 0.19	2.74 ± 1.66	2.37 ± 1.11	4.87 ± 1.07
Dactylis glomerata					
Deschampsia cespitosa					0.33 ± 0.26
Galium aparine					
Geranium dissectum	0.07 ± 0.07		0.17 ± 0.13	0.22 ± 0.13	0.04 ± 0.03
Heracleum sphondylium					
Juncus conglomeratus					
Lolium perenne	0.10 ± 0.07	0.01 ± 0.005	0.002 ± 0.001	0.07 ± 0.07	0.007 ± 0.007
Persicaria amphibia			0.0003 ± 0.0003	0.003 ± 0.003	
Phleum pratense		0.42 ± 0.39	0.83 ± 0.33	1.54 ± 0.79	2.24 ± 0.54
Plantago lanceolata			0.0003 ± 0.0003		
Poa annua	0.04 ± 0.03		0.0003 ± 0.0003	0.003 ± 0.003	

APPENDIX 5.22. HAY BALES (HB), Continued

	1994	1995	1996	1997	1999
Poa trivialis	17.77 ± 1.77	6.24 ± 2.95	4.14 ± 0.78	3.01 ± 0.93	5.03 ± 1.44
Prunella vulgaris			0.033 ± 0.033	0.003 ± 0.003	0.13 ± 0.13
Ranunculus repens	2.11 ± 0.63	1.85 ± 1.05	3.38 ± 1.98	0.52 ± 0.28	3.47 ± 1.08
Rumex conglomeratus					
Rumex crispus	0.04 ± 0.03			0.003 ± 0.003	
Taraxacum agg.	0.21 ± 0.09	0.65 ± 0.25	1.70 ± 0.48	1.69 ± 0.48	1.007 ± 0.40
Trifolium repens	1.36 ± 0.56	1.41 ± 1.20	7.77 ± 5.98	16.47 ± 9.18	1.18 ± 0.64
Vicia sativa					
Other species					
Alopecurus myosuroides	6.34 ± 1.56	0.57 ± 0.57	0.17 ± 0.08	0.04 ± 0.04	
Anisantha sterilis	0.10 ± 0.10			0.033 ± 0.033	
Avena fatua	0.21 ± 0.10		0.34 ± 0.26	0.48 ± 0.22	0.003 ± 0.003
Barbula unguiculata				0.07 ± 0.04	
Brachythecium rutabulum		0.17 ± 0.13	0.10 ± 0.10		
Brassica rapa	0.003 ± 0.003				
Bryum sp.				0.003 ± 0.003	
Calliergon cuspidatum		0.003 ± 0.003		0.07 ± 0.07	6.98 ± 5.37
Ceratodon purpureus		0.003 ± 0.003			
Chenopodium album					
Cirsium vulgare	0.31 ± 0.17	0.37 ± 0.37	0.001 ± 0.0007	0.67 ± 0.55	
Crataegus monogyna				0.003 ± 0.003	0.04 ± 0.03
Dicranella varia		0.07 ± 0.07			
Elytrigia repens		0.04 ± 0.04	0.10 ± 0.10	2.0 ± 1.37	0.17 ± 0.17
Epilobium ciliatum	0.37 ± 0.14			0.033 ± 0.033	
Epilobium hirsutum	0.14 ± 0.06				

APPENDIX 5.22. HAY BALES (HB), Continued

	1994	1995	1996	1997	1999
Eurhynchium praelongum		0.003 ± 0.003		0.003 ± 0.003	0.07 ± 0.07
Fissidens taxifolius	0.04 ± 0.03	0.14 ± 0.09		0.04 ± 0.03	0.58 ± 0.32
Galium verum	0.07 ± 0.07				
Geranium molle					0.007 ± 0.007
Geum urbanum					
Hypochaeris radicata					
Lactuca serriola	0.04 ± 0.03				
Lamium purpureum					
Lolium multiflorum	54.63 ± 5.67	41.37 ± 6.11	41.53 ± 4.46	43.73 ± 3.54	22.07 ± 3.14
Lythrum salicaria					
Matricaria discoidea	0.003 ± 0.003				
Matricaria recutita			0.0003 ± 0.0003		
Persicaria maculosa					
Phleum bertolonii			0.13 ± 0.13		
Picris echioides					
Plantago major			0.07 ± 0.07	0.033 ± 0.033	
Poa pratensis			0.001 ± 0.001		
Prunus spinosa	0.003 ± 0.003			0.002 ± 0.002	
Rosa canina					0.003 ± 0.003
Rumex obtusifolius					
Sambucus nigra					
Senecio aquaticus					
Senecio erucifolius				0.003 ± 0.003	
Senecio jacobea					
Solanum dulcemara					

APPENDIX 5.22. HAY BALES (HB), Continued

	1994	1995	1996	1997	1999
Sonchus asper	0.023 ± 0.008			0.04 ± 0.03	
Sonchus oleraceus					
Stellaria media	0.01 ± 0.005				
Trifolium dubium	0.10 ± 0.07		0.30 ± 0.19	0.04 ± 0.04	0.27 ± 0.15
Trifolium hybridum	0.13 ± 0.13				
Trifolium medium					
Triticum aestivum	5.60 ± 0.98				
Urtica dioica					
Veronica beccabunga					
Veronica serpyllifolia					

APPENDIX 5.23. SEED MIX 1 (SM1). Treatment means (± s.e.). Shading indicates that species was sown.

	1994	1995	1996	1997	1999
Class I target species					
Achillea ptarmica					
Agrostis capillaris		0.04 ± 0.04	0.07 ± 0.07	0.003 ± 0.003	1.64 ± 0.99
Alopecurus pratensis	0.40 ± 0.34	0.04 ± 0.03	2.30 ± 1.67	2.90 ± 0.74	4.48 ± 1.53
Anthoxanthum odoratum					0.17 ± 0.17
Briza media					
Cardamine pratensis	0.08 ± 0.07				0.003 ± 0.003
Centaurea nigra					
Cynosurus cristatus	1.82 ± 0.39	0.25 ± 0.10	2.97 ± 0.60	4.83 ± 1.03	5.24 ± 0.72
Festuca pratensis					0.003 ± 0.003
Festuca rubra	0.03 ± 0.03	0.22 ± 0.17	0.20 ± 0.14	0.67 ± 0.28	1.86 ± 0.44
Filipendula ulmaria					
Holcus lanatus	0.65 ± 0.28	0.34 ± 0.26	1.33 ± 0.80	1.18 ± 0.56	15.83 ± 7.07
Hordeum secalinum					
Lathyrus pratensis					
Leucanthemum vulgare	0.003 ± 0.003				
Lotus corniculatus	0.033 ± 0.033				
Lotus pedunculatus					
Lychnis flos-cuculi	0.003 ± 0.003		0.033 ± 0.033	0.007 ± 0.004	
Oenanthe silaifolia					
Ranunculus acris	0.14 ± 0.10		0.03 ± 0.03	0.003 ± 0.003	
Ranunculus flammula	0.21 ± 0.07	0.02 ± 0.01	0.002 ± 0.001		
Rhinanthus minor					
Rumex acetosa		0.003 ± 0.003		0.07 ± 0.07	0.10 ± 0.10
Sanguisorba officinalis	0.003 ± 0.003				

APPENDIX 5.23 (continued). SEED MIX 1 (SM1).

	1994	1995	1996	1997	1999
Silaum silaus					
Trifolium pratense				0.14 ± 0.14	0.44 ± 0.23
Trisetum flavescens					0.11 ± 0.10
Vicia cracca					0.003 ± 0.003
Class II target species					
Agrostis canina			0.07 ± 0.07	0.07 ± 0.04	0.91 ± 0.28
Agrostis stolonifera	0.31 ± 0.17	0.033 ± 0.033	0.13 ± 0.13	0.84 ± 0.39	1.58 ± 0.87
Alopecurus geniculatus	0.87 ± 0.31	0.71 ± 0.43	0.27 ± 0.14	0.04 ± 0.03	0.53 ± 0.53
Arrhenatherum elatius				0.007 ± 0.007	
Bromus commutatus	2.74 ± 0.86		0.001 ± 0.001		0.37 ± 0.37
Bromus hordeaceus				0.003 ± 0.003	
Bromus racemosus					1.57 ± 0.82
Cerastium fontanum	0.003 ± 0.003				
Cirsium arvense	0.84 ± 0.35	0.11 ± 0.07	0.83 ± 0.36	1.60 ± 0.64	3.40 ± 1.0
Dactylis glomerata					
Deschampsia cespitosa	0.04 ± 0.03				0.003 ± 0.003
Galium aparine	0.033 ± 0.033				
Geranium dissectum	0.003 ± 0.003		0.07 ± 0.07	0.11 ± 0.05	0.13 ± 0.08
Heracleum sphondylium					
Juncus conglomeratus					
Lolium perenne	0.07 ± 0.05	0.17 ± 0.13	1.07 ± 0.71	0.10 ± 0.10	0.75 ± 0.42
Persicaria amphibia					
Phleum pratense		0.07 ± 0.07			5.24 ± 1.61
Plantago lanceolata					
Poa annua				0.003 ± 0.003	

APPENDIX 5.23 (continued). SEED MIX 1 (SM1).

	1994	1995	1996	1997	1999
Poa trivialis	20.37 ± 2.37	3.27 ± 0.78	1.97 ± 0.45	3.78 ± 1.04	5.04 ± 1.47
Prunella vulgaris					
Ranunculus repens	3.31 ± 0.90	0.92 ± 0.40	3.17 ± 1.63	1.01 ± 0.34	1.71 ± 0.54
Rumex conglomeratus				0.003 ± 0.003	
Rumex crispus		0.04 ± 0.03			0.003 ± 0.003
Taraxacum agg.	0.48 ± 0.12	0.39 ± 0.06	0.90 ± 0.26	1.16 ± 0.43	0.35 ± 0.30
Trifolium repens	0.71 ± 0.16	0.82 ± 0.52	6.83 ± 3.78	16.77 ± 9.99	1.08 ± 0.46
Vicia sativa					
<u>Other species</u>					
Alopecurus myosuroides	4.57 ± 1.05	0.15 ± 0.07	0.10 ± 0.05	0.01 ± 0.01	
Anisantha sterilis	0.11 ± 0.10				
Avena fatua	0.48 ± 0.23		0.34 ± 0.24	0.55 ± 0.36	0.01 ± 0.007
Barbula unguiculata				0.13 ± 0.13	
Brachythecium rutabulum		0.04 ± 0.04			1.54 ± 1.49
Brassica rapa					
Bryum sp.				0.01 ± 0.01	0.14 ± 0.14
Calliergon cuspidatum					0.41 ± 0.33
Ceratodon purpureus					
Chenopodium album					
Cirsium vulgare	0.007 ± 0.005		0.07 ± 0.04	0.04 ± 0.04	0.17 ± 0.17
Crataegus monogyna					0.033 ± 0.033
Dicranella varia					
Elytrigia repens				0.07 ± 0.07	0.07 ± 0.07
Epilobium ciliatum	0.41 ± 0.13	0.003 ± 0.003			
Epilobium hirsutum	0.07 ± 0.07	0.003 ± 0.003			

APPENDIX 5.23 (continued). SEED MIX 1 (SM1).

	1994	1995	1996	1997	1999
Eurhynchium praelongum	0.003 ± 0.003	0.003 ± 0.003			0.04 ± 0.04
Fissidens taxifolius	0.21 ± 0.11	0.05 ± 0.04		0.05 ± 0.05	1.27 ± 0.56
Galium verum					
Geranium molle					
Geum urbanum					
Hypochaeris radicata					
Lactuca serriola	0.10 ± 0.06				
Lamium purpureum		0.003 ± 0.003			
Lolium multiflorum	57.10 ± 4.2	41.47 ± 2.89	46.03 ± 5.57	39.37 ± 4.15	35.17 ± 5.41
Lythrum salicaria	0.003 ± 0.003				
Matricaria discoidea					
Matricaria recutita					
Persicaria maculosa		0.003 ± 0.003			
Phleum bertolonii		0.48 ± 0.20	3.47 ± 0.82	7.41 ± 2.52	3.23 ± 3.23
Picris echioides					0.003 ± 0.003
Plantago major				0.003 ± 0.003	0.003 ± 0.003
Poa pratensis			0.10 ± 0.07		
Prunus spinosa	0.003 ± 0.003				
Rosa canina					
Rumex obtusifolius				0.10 ± 0.10	0.003 ± 0.003
Sambucus nigra	0.033 ± 0.033				
Senecio aquaticus					
Senecio erucifolius					0.003 ± 0.003
Senecio jacobea				0.033 ± 0.033	
Solanum dulcemara					

APPENDIX 5.23 (continued). SEED MIX 1 (SM1).

	1994	1995	1996	1997	1999
Sonchus asper	0.09 ± 0.05		0.03 ± 0.03	0.003 ± 0.003	
Sonchus oleraceus					
Stellaria media	0.07 ± 0.07				
Trifolium dubium	0.20 ± 0.17		0.10 ± 0.05	0.01 ± 0.01	0.27 ± 0.13
Trifolium hybridum					
Trifolium medium					
Tripleurospermum inodorum			0.0003 ± 0.0003		
Triticum aestivum	5.74 ± 1.01				
Urtica dioica					
Veronica beccabunga	0.003 ± 0.003				
Veronica serpyllifolia	0.003 ± 0.003				

APPENDIX 5.24. SEED MIX 2 (SM2), treatment means (± s.e.). Shading indicates that species was sown.

	1994	1995	1996	1997	1999
Class I target species					
Achillea ptarmica					
Agrostis capillaris		0.77 ± 0.65	0.30 ± 0.23	0.10 ± 0.07	2.30 ± 1.46
Alopecurus pratensis	0.23 ± 0.14		1.23 ± 0.46	2.54 ± 0.75	2.74 ± 0.42
Anthoxanthum odoratum	0.033 ± 0.033			0.033 ± 0.033	1.14 ± 0.74
Briza media					
Cardamine pratensis	0.007 ± 0.005				0.003 ± 0.003
Centaurea nigra					0.003 ± 0.003
Cynosurus cristatus	0.84 ± 0.33	0.32 ± 0.26	2.13 ± 0.48	2.57 ± 0.35	4.30 ± 0.61
Festuca pratensis	0.07 ± 0.07		0.001 ± 0.001		0.04 ± 0.03
Festuca rubra	0.13 ± 0.10	0.04 ± 0.03	0.10 ± 0.10	0.44 ± 0.24	3.24 ± 1.44
Filipendula ulmaria	0.017 ± 0.007				
Holcus lanatus	1.91 ± 0.53	1.51 ± 0.69	3.33 ± 1.31	2.77 ± 0.42	19.63 ± 4.22
Hordeum secalinum					0.003 ± 0.003
Lathyrus pratensis					
Leucanthemum vulgare	0.22 ± 0.10	0.31 ± 0.14	1.27 ± 0.71	2.71 ± 1.14	2.64 ± 0.76
Lotus corniculatus	0.10 ± 0.07	0.113 ± 0.105	0.03 ± 0.03	0.70 ± 0.24	0.51 ± 0.32
Lotus pedunculatus					0.07 ± 0.07
Lychnis flos-cuculi	0.10 ± 0.10		0.033 ± 0.033	0.003 ± 0.003	
Oenanthe silaifolia					
Ranunculus acris	0.20 ± 0.07		0.40 ± 0.20	0.42 ± 0.25	0.70 ± 0.21
Ranunculus flammula	0.34 ± 0.14	0.003 ± 0.003	0.07 ± 0.07		
Rhinanthus minor					
Rumex acetosa					0.003 ± 0.003
Sanguisorba officinalis					

APPENDIX 5.24 (continued). SEED MIX 2 (SM2). Continued.

	1994	1995	1996	1997	1999
Silaum silaus					
Trifolium pratense	0.24 ±0.09	0.72 ±0.44	3.17 ±1.38	6.64 ±2.45	4.04 ±2.51
Trisetum flavescens				0.003 ±0.003	
Vicia cracca					
Class II target species					
Agrostis canina			0.07 ±0.07	0.037 ±0.033	0.17 ±0.13
Agrostis stolonifera	0.003 ±0.003	0.07 ±0.07	1.04 ±0.53	2.50 ±1.00	4.57 ±1.96
Alopecurus geniculatus	2.54 ±0.74	2.63 ±1.46	1.0 ±0.73	0.22 ±0.11	
Arrhenatherum elatius				0.037 ±0.033	
Bromus commutatus	3.01 ±0.75		0.003 ±0.003	0.04 ±0.04	0.007 ±0.007
Bromus hordeaceus					
Bromus racemosus					0.22 ±0.13
Cerastium fontanum	0.10 ±0.07				
Cirsium arvense	1.14 ±0.34	0.78 ±0.43	2.93 ±1.52	1.37 ±0.72	2.10 ±0.72
Dactylis glomerata					
Deschampsia cespitosa	0.30 ±0.22	0.10 ±0.10			0.20 ±0.16
Galium aparine	0.01 ±0.006				
Geranium dissectum	0.11 ±0.06		0.27 ±0.27	0.04 ±0.03	0.04 ±0.03
Heracleum sphondylium					
Juncus conglomeratus			0.07 ±0.07		
Lolium perenne	0.47 ±0.26	0.07 ±0.07	0.40 ±0.29		0.003 ±0.003
Persicaria amphibia	0.013 ±0.006		0.0003 ±0.0003	0.003 ±0.003	
Phleum pratense	0.37 ±0.18	0.69 ±0.18	1.93 ±0.65	1.88 ±0.63	6.34 ±1.47
Plantago lanceolata					
Poa annua					

APPENDIX 5.24 (continued). SEED MIX 2 (SM2). Continued.

	1994	1995	1996	1997	1999
Poa trivialis	24.54 ± 2.67	6.47 ± 1.31	3.00 ± 0.60	3.47 ± 0.93	3.40 ± 1.18
Prunella vulgaris					
Ranunculus repens	3.58 ± 0.87	3.13 ± 1.52	5.67 ± 2.32	1.64 ± 0.60	1.72 ± 0.75
Rumex conglomeratus	0.033 ± 0.033				
Rumex crispus		0.033 ± 0.033	0.033 ± 0.033	0.033 ± 0.033	
Taraxacum agg.	0.49 ± 0.15	0.49 ± 0.20	0.83 ± 0.18	1.18 ± 0.40	0.92 ± 0.40
Trifolium repens	1.99 ± 0.47	3.41 ± 1.58	14.0 ± 5.0	28.80 ± 11.7	1.26 ± 0.68
Vicia sativa					
Other species					
Alopecurus myosuroides	5.04 ± 1.10	0.003 ± 0.003	0.10 ± 0.07		
Anisantha sterilis					
Avena fatua	0.34 ± 0.13		0.27 ± 0.13	0.12 ± 0.04	0.003 ± 0.003
Barbula unguiculata					
Brachythecium rutabulum	0.007 ± 0.005	0.007 ± 0.007			0.35 ± 0.20
Brassica rapa					
Bryum sp.				0.18 ± 0.13	
Calliergon cuspidatum					0.10 ± 0.04
Ceratodon purpureus					
Chenopodium album					
Cirsium vulgare	0.12 ± 0.10	0.007 ± 0.004	0.001 ± 0.001	0.20 ± 0.20	
Crataegus monogyna	0.003 ± 0.003			0.007 ± 0.007	0.007 ± 0.004
Dicranella varia					
Elytrigia repens		0.07 ± 0.07		0.23 ± 0.15	0.37 ± 0.29
Epilobium ciliatum	0.34 ± 0.12				
Epilobium hirsutum	0.17 ± 0.08				

APPENDIX 5.24 (continued). SEED MIX 2 (SM2). Continued.

	1994	1995	1996	1997	1999
Eurhynchium praelongum		0.003 ± 0.003			0.04 ± 0.04
Fissidens taxifolius	0.24 ± 0.11	0.25 ± 0.23		0.24 ± 0.17	0.17 ± 0.10
Galium verum					
Geranium molle					0.003 ± 0.003
Geum urbanum	0.003 ± 0.003				
Hypochaeris radicata					0.033 ± 0.033
Lactuca serriola	0.04 ± 0.03				
Lamium purpureum					
Lolium multiflorum	45.30 ± 4.56	33.80 ± 5.57	35.83 ± 5.96	40.77 ± 7.99	32.97 ± 4.22
Lythrum salicaria					
Matricaria discoidea					
Matricaria recutita	0.007 ± 0.005		0.0003 ± 0.0003		
Persicaria maculosa					
Phleum bertolonii		0.033 ± 0.033	0.27 ± 0.27	0.20 ± 0.20	
Picris echioides					
Plantago major		0.033 ± 0.033	0.033 ± 0.033		
Poa pratensis			0.002 ± 0.001		
Prunus spinosa				0.003 ± 0.003	
Rosa canina					
Rumex obtusifolius					
Sambucus nigra					
Senecio aquaticus					
Senecio erucifolius					
Senecio jacobea					
Solanum dulcemara					

APPENDIX 5.24 (continued). SEED MIX 2 (SM2). Continued.

	1994	1995	1996	1997	1999
Sonchus asper	0.39 ±0.19		0.001 ±0.001	0.03 ±0.01	
Sonchus oleraceus					
Stellaria media	0.05 ±0.03				
Trifolium dubium	0.24 ±0.16		0.33 ±0.13	0.09 ±0.04	0.37 ±0.16
Trifolium hybridum					
Trifolium medium					
Triticum aestivum	5.10 ±0.84				
Urtica dioica	0.003 ±0.003				
Veronica beccabunga					
Veronica serpyllifolia					

APPENDIX 5.25. SEED MIX 3 (SM3). Treatment means (± s.e.). Shading indicates that species was sown.

	1994	1995	1996	1997	1999
Class I target species					
Achillea ptarmica	0.007 ± 0.05	0.007 ± 0.004	0.10 ± 0.10		0.14 ± 0.13
Agrostis capillaris		0.07 ± 0.07	0.04 ± 0.03	0.30 ± 0.17	0.80 ± 0.65
Alopecurus pratensis	0.003 ± 0.003		1.53 ± 0.54	2.68 ± 1.04	1.95 ± 0.23
Anthoxanthum odoratum	0.08 ± 0.05				0.21 ± 0.11
Briza media				0.003 ± 0.003	0.007 ± 0.004
Cardamine pratensis					
Centaurea nigra	0.003 ± 0.003		0.17 ± 0.11	0.08 ± 0.04	0.67 ± 0.38
Cynosurus cristatus	0.92 ± 0.19	0.45 ± 0.28	2.40 ± 0.32	2.11 ± 0.46	5.17 ± 0.90
Festuca pratensis	0.28 ± 0.13			0.003 ± 0.003	0.033 ± 0.033
Festuca rubra	0.04 ± 0.03	0.54 ± 0.54	0.17 ± 0.13	0.74 ± 0.38	0.943 ± 0.81
Filipendula ulmaria	0.07 ± 0.03				
Holcus lanatus	2.05 ± 0.36	1.68 ± 0.29	4.20 ± 1.24	3.61 ± 0.42	24.90 ± 1.72
Hordeum secalinum			0.43 ± 0.27	0.82 ± 0.32	3.88 ± 1.13
Lathyrus pratensis	0.05 ± 0.03	0.01 ± 0.007	0.03 ± 0.03	0.17 ± 0.08	0.07 ± 0.04
Leucanthemum vulgare	0.38 ± 0.15	0.013 ± 0.007	0.53 ± 0.42	1.87 ± 0.74	4.14 ± 1.38
Lotus corniculatus					
Lotus pedunculatus					
Lychnis flos-cuculi	0.007 ± 0.005		0.001 ± 0.001	0.04 ± 0.03	0.04 ± 0.03
Oenanthe silaifolia			0.10 ± 0.10	0.013 ± 0.010	0.41 ± 0.33
Ranunculus acris	0.08 ± 0.05	0.09 ± 0.07	0.30 ± 0.19	0.38 ± 0.29	0.34 ± 0.09
Ranunculus flammula	0.30 ± 0.13	0.04 ± 0.04	0.001 ± 0.001		
Rhinanthus minor	1.64 ± 0.34				
Rumex acetosa	0.04 ± 0.01	0.007 ± 0.004	0.03 ± 0.03	0.013 ± 0.007	0.14 ± 0.09
Sanguisorba officinalis	0.02 ± 0.007	0.007 ± 0.004	0.0003 ± 0.0003	0.003 ± 0.003	

APPENDIX 5.25 (continued). SEED MIX 3 (SM3).

	1994	1995	1996	1997	1999
Silaum silaus	0.12 ± 0.04	0.01 ± 0.004	0.13 ± 0.08	0.07 ± 0.07	0.27 ± 0.13
Trifolium pratense	0.45 ± 0.14	0.14 ± 0.09	1.60 ± 0.60	4.77 ± 2.36	1.77 ± 0.79
Trisetum flavescens			0.10 ± 0.07	0.78 ± 0.28	3.04 ± 1.58
Vicia cracca	0.08 ± 0.05		0.07 ± 0.04	0.12 ± 0.10	0.34 ± 0.14
Class II target species					
Agrostis canina			0.13 ± 0.13	0.10 ± 0.05	0.18 ± 0.10
Agrostis stolonifera	0.11 ± 0.06	0.033 ± 0.033	1.37 ± 0.57	2.60 ± 1.02	11.63 ± 2.97
Alopecurus geniculatus	2.81 ± 1.10	3.51 ± 2.28	2.40 ± 1.96	0.24 ± 0.10	0.003 ± 0.003
Arrhenatherum elatius				0.07 ± 0.07	0.11 ± 0.10
Bromus commutatus	2.37 ± 0.52				0.007 ± 0.007
Bromus hordeaceus	0.003 ± 0.003				
Bromus racemosus			0.003 ± 0.003	0.10 ± 0.10	0.45 ± 0.23
Cerastium fontanum	0.05 ± 0.03				
Cirsium arvense	0.74 ± 0.24	0.39 ± 0.21	1.67 ± 0.61	2.34 ± 1.02	3.34 ± 0.39
Dactylis glomerata					0.103 ± 0.099
Deschampsia cespitosa	0.007 ± 0.005	0.04 ± 0.03	0.07 ± 0.07	0.13 ± 0.10	0.07 ± 0.07
Galium aparine					
Geranium dissectum	0.34 ± 0.15		0.17 ± 0.08	0.22 ± 0.07	0.04 ± 0.03
Heracleum sphondylium	0.13 ± 0.13				
Juncus conglomeratus					
Lolium perenne	1.08 ± 0.31	0.003 ± 0.003	0.0007 ± 0.0004	0.003 ± 0.003	0.04 ± 0.03
Persicaria amphibia					
Phleum pratense		0.49 ± 0.23	1.44 ± 0.53	2.10 ± 0.57	6.23 ± 1.67
Plantago lanceolata				0.007 ± 0.004	0.003 ± 0.003
Poa annua	0.003 ± 0.003		0.001 ± 0.0005		

APPENDIX 5.25 (continued). SEED MIX 3 (SM3).

	1994	1995	1996	1997	1999
Poa trivialis	23.83 ± 1.60	7.13 ± 2.08	4.43 ± 0.23	2.29 ± 0.74	3.68 ± 0.84
Prunella vulgaris					
Ranunculus repens	3.76 ± 1.15	1.41 ± 0.71	5.24 ± 2.38	2.32 1.0	2.28 0.68
Rumex conglomeratus					
Rumex crispus		0.003 ± 0.003		0.10 0.10	
Taraxacum agg.	0.15 ± 0.06	0.25 ± 0.08	0.77 ± 0.29	0.88 0.23	0.45 0.16
Trifolium repens	1.49 ± 0.26	1.78 ± 0.73	8.707 ± 2.66	25.0 8.68	1.58 0.59
Vicia sativa	0.007 ± 0.005				
<u>Other species</u>					
Alopecurus myosuroides	2.92 ± 1.10	0.17 ± 0.17	0.30 ± 0.23	0.003 ± 0.003	
Anisantha sterilis					
Avena fatua	0.35 ± 0.15		0.24 ± 0.13	0.19 ± 0.09	0.04 ± 0.03
Barbula unguiculata				0.033 ± 0.033	
Brachythecium rutabulum	0.003 ± 0.003			0.10 ± 0.07	0.007 ± 0.004
Brassica rapa					
Bryum sp.					
Calliergon cuspidatum				0.033 ± 0.033	0.10 ± 0.10
Ceratodon purpureus					
Chenopodium album			0.003 ± 0.003		
Cirsium vulgare	0.17 ± 0.14	0.013 ± 0.004	0.04 ± 0.03	0.18 ± 0.11	0.033 ± 0.033
Crataegus monogyna			0.003 ± 0.003	0.003 ± 0.003	0.003 ± 0.003
Dicranella varia					
Elytrigia repens			0.13 ± 0.13		0.23 ± 0.23
Epilobium adenocaulon					
Epilobium ciliatum	0.07 ± 0.03	0.007 ± 0.007			

APPENDIX 5.25 (continued). SEED MIX 3 (SM3).

	1994	1995	1996	1997	1999
Epilobium hirsutum	0.05 ±0.03				
Eurhynchium praelongum					0.007 ±0.004
Fissidens taxifolius	0.10 ±0.07	0.11 ±0.11		0.003 ±0.003	0.22 ±0.14
Galium verum					
Geranium molle					
Geum urbanum					
Hypochaeris radicata					
Lactuca serriola	0.01 ±0.006				
Lamium purpureum					
Lolium multiflorum	45.47 ±3.53	39.13 ±6.82	38.23 ±4.51	31.43 ±5.02	15.93 ±2.47
Lythrum salicaria					
Matricaria discoidea					
Matricaria recutita					
Persicaria maculosa					
Phleum bertolonii					
Picris echioides					0.07 ±0.07
Plantago major	0.003 ±0.003	0.003 ±0.003		0.003 ±0.003	
Poa pratensis			0.001 ±0.001		
Prunus spinosa	0.003 ±0.003				
Rosa canina					0.003 ±0.003
Rumex obtusifolius					
Sambucus nigra					
Senecio aquaticus		0.003 ±0.003	0.033 ±0.033		
Senecio erucifolius					
Senecio jacobea	0.07 ±0.07				

APPENDIX 5.25 (continued). SEED MIX 3 (SM3).

	1994	1995	1996	1997	1999
Solanum dulcemara	0.007 ±0.005				
Sonchus asper	0.11 ±0.06		0.0007 ±0.0004	0.01 ±0.005	
Sonchus oleraceus	0.003 ±0.003				
Stellaria media	0.16 ±0.08				
Trifolium dubium	0.41 ±0.33	0.003 ±0.003	0.53 ±0.20	0.10 ±0.04	0.34 ±0.16
Trifolium hybridum					
Trifolium medium					0.003 ±0.003
Triticum aestivum	4.70 ±0.70				
Urtica dioica					
Veronica beccabunga					
Veronica serpyllifolia	0.07 ±0.07				

APPENDIX 5.3. Germination results for sown species after 20 weeks. **% GERM^N^**: laboratory germination; **% QUADRATS**, **% PLOTS**: observed germination in the field experiment, as a percentage of the total number of quadrats (or plots) that each species was sown in; iii) species marked '*' were not recorded in the vegetation at any time.

Species	% GERM^{N}	% QUADRATS					% PLOTS				
		1994	1995	1996	1997	1999	1994	1995	1996	1997	1999
Lolium multiflorum	92	100	100	100	100	100	100	100	100	100	100
Phleum bertolonii	92	0	56.7	80.0	93.33	15.6	0	83.3	100	100	16.67
Alopecurus pratensis	87	6.7	2.2	46.7	70.00	86.67	33.3	11.1	83.3	100	100
Cynosurus cristatus	82	64.4	41.1	92.2	91.11	98.89	88.9	94.4	100	100	100
Briza media	82	0	0	0	3.33	6.67	0	0	0	5.56	11.11
Festuca rubra	75	3.3	16.7	17.8	31.11	63.33	27.8	33.3	44.4	66.67	88.89
Vicia cracca	69	20.0	0.0	16.7	23.3	30.0	33.3	0	33.3	66.67	66.67
Leucanthemum vulgare	69	36.7	21.7	28.3	38.33	55.0	66.7	66.7	66.7	83.33	100
Trisetum flavescens	66	0	0	20.0	60.0	66.67	0.0	0	33.3	83.33	100
Festuca pratensis	58	3.3	0	6.7	0	10.0	16.7	0	16.7	0	50.0
Trifolium pratense	58	36.7	35.0	45.0	53.33	45.0	91.7	75.0	83.3	91.67	75.0
Holcus lanatus	52	80.0	78.33	80.0	85.0	93.33	100.0	100.0	91.7	100	100
Lychnis flos-cuculi	52	6.7	0	6.7	6.7	6.7	16.7	0	16.7	33.33	33.33
Achillea ptarmica	52	3.33	3.33	6.7	0	3.33	33.3	33.3	33.3	0	33.33
Hordeum secalinum	51	0	0	20.0	50.0	73.33	0	0	66.7	100	100
Lotus corniculatus	50	6.7	20.0	6.67	23.33	20.0	33.3	50.0	33.3	66.67	66.67
Lathyrus pratensis	48	36.67	10.0	6.67	10.0	13.33	50.0	33.3	33.3	50.0	66.67
Oenanthe fistulosa	43*										
Rumex acetosa	38	40.0	6.67	6.67	13.33	13.33	83.3	33.3	33.3	50.0	50.0
Agrostis capillaris	26	3.33	15.0	8.33	11.67	26.67	0.0	41.7	50.0	41.67	33.33
Ranunculus acris	12	31.7	11.7	25.0	33.33	33.33	66.7	25.0	50.0	66.67	91.67
Filipendula ulmaria	4	28.3	0	0	0	0	58.3	0	0	0	0
Centaurea nigra	3	3.3	0	16.67	16.67	16.67	16.7	0	66.7	66.67	66.67
Sanguisorba officinalis	3	20.0	6.7	3.3	3.3	0	50.0	33.3	1.7	66.67	66.67
Silaum silaus	0	63.3	13.3	20.0	10.0	23.33	100	66.7	50.0	33.33	66.67
Rhinanthus minor	0	66.7	0	0	0	0	100	0	0	0	0
Thalictrum flavum	0*										

Appendix 6.1 Frequency of species protected within 'source field/reserve' selections

Species	total	PA20	divind	MG4com	MG5com	ESA	NVC60	PAarea	COVarea
Achillea millefolium	26	9	6	6	5	6	4	4	5
Achillea ptarmica	7	4	5	3	3	5	3	2	3
Agrimonia eupatoria	3	2	2	1	2	1	0	1	1
Agrostis canina	43	10	12	7	4	14	10	11	8
Agrostis capillaris	133	17	19	14	10	34	20	15	12
Agrostis stolonifera	202	20	20	16	10	45	25	21	18
Ajuga reptans	1	0	1	0	0	1	1	1	1
Allium vineale	1	0	0	0	0	1	0	1	0
Alopecurus geniculatus	160	17	16	14	9	37	21	17	15
Alopecurus myosuroides	10	1	0	0	0	0	0	2	1
Alopecurus pratensis	176	20	19	16	10	39	23	18	17
Anagallis arvensis	2	2	0	1	0	0	1	1	0
Angelica sylvestris	4	2	1	0	0	2	1	1	3
Anisantha sterilis	2	0	0	0	0	0	0	1	0
Anthoxanthum odoratum	133	19	20	16	10	29	20	17	15
Anthriscus sylvestris	11	1	0	0	0	0	2	1	3
Apium nodiflorum	1	1	1	0	0	0	0	1	1
Arctium minus spp. minus	1	0	0	0	0	0	0	0	0
Arrhenatherum elatius	22	5	4	2	2	5	3	6	4
Atriplex patula	3	1	0	0	0	0	1	1	1
Atriplex prostrata	2	1	0	0	0	0	0	0	0
Avena fatua	3	1	0	0	0	0	0	2	1
Bellis perennis	97	9	6	6	6	23	8	8	8
Betonica officinalis	1	1	1	1	0	1	0	1	0
Brachythecium rutabulum	132	18	18	16	10	32	21	19	14

Appendix 6.1 (continued)

Species	total	PA20	divind	MG4com	MG5com	ESA	NVC60	PAarea	COVarea
Brassica napus	1	0	0	0	0	0	0	0	0
Briza media	11	4	3	4	4	4	1	2	3
Bromus commutatus	27	6	5	4	2	4	8	6	5
Bromus hordeaceus agg.	55	5	6	7	4	10	9	9	6
Bromus racemosus	70	9	11	13	8	13	14	10	7
Bryum spp	4	2	1	1	1	1	1	2	1
Calliergon cuspidatum	30	9	9	7	5	13	9	9	8
Calystegia sepium	2	0	0	0	0	0	0	1	0
Capsella bursa-pastoris	2	0	0	0	0	0	0	0	0
Cardamine pratensis	78	17	16	14	9	18	15	12	12
Carex acutiformis	16	2	1	1	0	2	1	1	3
Carex disticha	27	12	8	7	4	9	5	5	8
Carex flacca	26	7	6	4	4	13	3	4	7
Carex hirta	53	14	9	8	5	18	8	8	8
Carex nigra	17	8	5	5	3	7	2	3	6
Carex otrubae	26	7	5	4	1	5	4	7	7
Carex ovalis	21	7	5	2	0	6	3	4	6
Carex panicea	10	4	4	3	3	5	3	2	3
Carex riparia	23	12	9	8	2	8	5	8	6
Carex spicata	14	3	3	1	1	5	2	3	3
Centaurea nigra	61	17	17	14	7	12	11	12	12
Cerastium fontanum	158	18	16	14	9	39	20	19	15
Cerastium glomeratum	3	0	1	1	1	1	1	0	0
Chamerion angustifolium	1	1	0	0	0	0	0	1	1
Chenopodium album agg.	1	1	0	0	0	0	0	1	1
Chenopodium polyspermum	3	1	0	0	0	0	0	2	1

Appendix 6.1 (continued)

Species	total	PA20	divind	MG4com	MG5com	ESA	NVC60	PAarea	COVarea
Cichorium intybus	1	0	0	0	0	1	0	1	0
Cirsium arvense	188	18	17	14	9	43	23	21	15
Cirsium dissectum	3	2	2	1	1	2	1	1	2
Cirsium palustre	13	4	3	3	0	2	2	2	2
Cirsium vulgare	85	11	5	7	5	18	9	13	7
Conium maculatum	3	2	0	0	0	0	0	2	2
Convolvulus arvensis	5	1	0	0	0	1	0	3	1
Coronopus squamatus	4	0	0	0	0	2	0	1	0
Crataegus laevigata	1	1	0	0	0	0	0	1	1
Crataegus monogyna	8	2	1	1	0	0	1	2	1
Crepis biennis	4	1	0	0	0	0	0	2	1
Crepis capillaris	1	1	0	0	0	0	0	1	1
Crepis vesicaria	7	1	0	0	0	1	0	3	1
Cynosurus cristatus	159	17	19	16	10	39	22	17	13
Dactylis glomerata	116	14	11	12	10	23	16	11	9
Dactylorhiza fuchsii	3	1	1	1	0	1	1	2	2
Deschampsia cespitosa	155	19	20	15	9	38	22	19	18
Dipsacus fullonum	3	1	0	0	0	0	0	1	1
Drepanocladus aduncus	3	2	0	1	0	0	1	1	1
Eleocharis palustris	13	6	2	2	0	1	1	5	4
Elytrigia repens	78	17	14	13	6	16	15	13	13
Epilobium ciliatum	1	1	0	0	0	0	0	1	1
Epilobium hirsutum	6	2	0	0	0	1	0	2	2
Epilobium obscurum	1	0	1	0	0	1	0	0	1
Epilobium parviflorum	1	1	0	0	0	0	0	1	1
Epilobium tetragonum	3	1	0	0	0	0	0	1	1

Appendix 6.1 (continued)

Species	total	PA20	divind	MG4com	MG5com	ESA	NVC60	PAarea	COVarea
Equisetum arvense	3	1	1	1	0	1	0	2	0
Euphorbia helioscopia	1	1	0	1	0	0	0	0	0
Eurhynchium praelongum	56	9	11	9	6	14	13	11	7
Festuca arundinacea	13	3	3	2	1	4	1	2	2
Festuca ovina agg.	4	0	0	0	0	1	1	1	0
Festuca pratensis	79	15	14	14	10	19	15	9	9
Festuca rubra	132	17	17	16	10	32	20	15	15
Filipendula ulmaria	20	8	10	7	4	9	6	7	6
Filipendula vulgaris	10	4	3	2	0	5	1	2	3
Fissidens taxifolius	10	1	2	2	1	2	2	1	1
Galeopsis tetrahit	1	1	0	0	0	0	0	1	1
Galium aparine	4	2	0	0	0	0	0	3	2
Galium mollugo	1	0	0	0	0	1	0	1	1
Galium palustre	24	12	8	4	2	9	5	9	9
Galium verum	20	9	9	9	6	7	4	6	5
Geranium dissectum	45	7	5	7	3	14	5	11	6
Geranium molle	1	1	1	0	0	0	0	1	1
Glechoma hederacea	1	1	0	0	0	0	0	1	1
Glyceria fluitans	44	8	6	5	2	6	6	6	4
Glyceria maxima	4	1	1	1	1	2	0	1	1
Gnaphalium uliginosum	1	0	0	0	0	1	0	0	0
Hedera helix	1	0	0	0	0	0	0	1	0
Heracleum sphondylium	12	4	3	4	3	2	2	4	3
Holcus lanatus	203	20	20	17	10	46	26	21	19
Holcus mollis	1	1	0	0	0	0	0	1	1
Hordeum secalinum	149	16	18	16	10	31	22	15	15

Appendix 6.1 (continued)

Species	total	PA20	divind	MG4com	MG5com	ESA	NVC60	PAarea	COVarea
Hordeum vulgare	2	1	0	0	0	0	0	1	1
Hypochaeris radicata	10	4	3	3	1	3	2	3	4
Iris pseudacorus	1	1	0	0	0	0	0	1	1
Juncus acutiflorus	23	10	8	7	3	6	5	7	8
Juncus articulatus	10	4	3	1	1	4	2	3	1
Juncus bufonius	4	0	1	0	0	2	0	1	1
Juncus conglomeratus	38	14	10	9	4	15	7	10	10
Juncus effusus	27	11	7	4	2	8	4	7	9
Juncus inflexus	52	13	10	6	3	15	7	9	11
Juncus x diffusus	2	0	0	0	0	0	0	1	1
Lactuca serriola	4	2	0	0	0	0	0	2	2
Lathyrus nissolia	3	0	0	1	0	1	0	2	0
Lathyrus pratensis	75	19	18	15	8	23	13	12	12
Leontodon autumnalis	33	8	6	6	3	7	6	8	7
Leontodon hispidus	6	1	2	2	2	0	0	1	0
Leontodon saxatilis	15	3	4	3	2	4	2	4	4
Lepidum campestre	1	0	0	0	0	0	0	1	0
Leucanthemum vulgare	35	9	9	5	4	10	5	7	9
Linum catharticum	1	0	1	0	0	1	1	1	1
Linum usitatissimum	1	0	0	0	0	0	0	0	0
Lolium multiflorum	20	1	1	1	0	0	3	2	1
Lolium perenne	196	18	20	16	10	42	25	20	16
Lolium x hybridum	2	0	0	0	0	0	0	1	0
Lotus corniculatus	60	16	14	11	8	18	9	10	12
Lotus pedunculatus	14	4	4	3	2	8	3	3	4
Luzula campestris	30	7	8	10	5	9	8	6	4

Appendix 6.1 (continued)

Species	total	PA20	divind	MG4com	MG5com	ESA	NVC60	PAarea	COVarea
Lychnis flos-cuculi	23	8	8	7	3	9	7	10	8
Lysimachia nummularia	5	4	4	1	1	3	1	2	2
Matricaria maritima	6	1	0	0	0	0	0	2	1
Matricaria recutita	6	1	0	0	0	0	0	1	1
Medicago lupulina	1	0	0	0	0	0	0	1	0
Melilotus altissima	1	0	0	0	0	0	0	1	0
Mentha arvensis	1	1	0	0	0	0	0	1	1
Myosotis arvensis	1	1	0	0	0	0	0	1	1
Myosotis discolor	2	0	1	1	0	2	0	1	1
Myosotis laxa ssp. cespitosa	9	3	2	2	1	1	4	4	3
Odontites verna	1	0	0	0	0	1	0	0	0
Oenanthe fistulosa	32	11	9	6	3	7	8	7	9
Oenanthe silaifolia	35	7	5	5	3	13	5	7	10
Ononis repens	2	2	1	1	0	0	0	1	0
Ophioglossum vulgatum	6	3	2	4	2	3	2	3	3
Orchis morio	1	0	0	1	0	1	0	1	0
Papaver rhoeas	1	0	0	0	0	0	0	0	0
Persicaria amphibia	20	9	8	5	2	8	5	8	4
Persicaria maculosa	5	1	1	0	0	1	0	1	2
Petroselinum segetum	2	0	0	0	0	0	0	1	0
Phalaris arundinacea	21	11	8	6	2	7	5	8	7
Phleum bertolonii	71	10	7	9	5	18	9	9	4
Phleum pratense	149	18	15	12	8	27	17	19	15
Phragmites australis	2	0	0	0	0	0	0	0	0
Picris echioides	9	3	1	0	0	2	1	2	3
Plantago lanceolata	31	12	10	13	8	7	8	7	7

Appendix 6.1 (continued)

Species	total	PA20	divind	MG4com	MG5com	ESA	NVC60	PAarea	COVarea
Plantago major	75	9	5	5	4	12	7	10	6
Poa annua	62	6	6	5	5	15	6	6	1
Poa pratensis	26	8	6	7	4	6	4	8	2
Poa subcaerulea	16	6	5	4	1	5	2	5	4
Poa trivialis	209	20	19	16	10	45	25	21	17
Polygonum aviculare	6	1	2	0	0	1	0	1	2
Potentilla anglica	2	1	1	1	0	1	1	1	1
Potentilla anserina	19	12	7	4	1	6	1	6	5
Potentilla erecta	3	1	1	0	0	1	0	0	1
Potentilla reptans	77	18	15	12	8	16	14	12	11
Primula veris	4	1	2	0	0	3	1	1	1
Prunella vulgaris	46	7	8	7	5	13	5	8	4
Prunus spinosa	13	2	2	0	0	4	2	2	4
Pulicaria dysenterica	1	0	0	0	0	0	0	0	0
Quercus robur	16	3	2	2	0	5	3	7	6
Ranunculus acris	179	19	20	16	10	39	25	20	17
Ranunculus bulbosus	83	11	11	12	8	19	12	12	7
Ranunculus flammula	28	13	8	7	2	7	5	9	8
Ranunculus repens	200	20	20	16	10	44	25	22	18
Ranunculus sceleratus	2	0	0	0	0	0	1	1	0
Rhinanthus minor	20	6	6	7	4	5	5	7	6
Rhynchostegium confertum	5	2	2	3	2	2	2	2	1
Rhytidiadelphus squarrosus	1	0	0	0	0	0	1	0	1
Rorippa nasturtium-aquaticum	1	1	1	0	0	0	0	1	1
Rorippa palustris	1	1	0	0	0	0	0	1	1
Rosa canina agg.	4	1	2	0	0	1	2	1	3

Appendix 6.1 (continued)

Species	total	PA20	divind	MG4com	MG5com	ESA	NVC60	PAarea	COVarea
Rubus fruticosus agg.	5	1	0	0	0	0	0	3	1
Rumex acetosa	123	19	19	15	10	29	17	16	15
Rumex acetosella	1	1	0	0	0	0	0	1	1
Rumex conglomeratus	55	12	10	9	6	14	8	10	8
Rumex crispus	120	15	15	12	9	22	16	17	15
Rumex obtusifolius	68	6	5	5	4	10	8	9	8
Salix fragilis	1	0	0	0	0	0	0	1	0
Sanguisorba officinalis	54	15	12	11	5	12	10	9	9
Senecio aquaticus	4	1	0	0	0	0	1	1	2
Senecio erucifolius	15	2	1	1	0	4	2	3	4
Senecio jacobea	1	1	0	0	0	0	0	1	1
Senecio vulgaris	4	2	1	1	1	1	1	2	1
Sherardia arvensis	1	0	0	0	0	0	0	1	0
Sieglingia decumbens	2	0	1	1	0	2	1	2	1
Silaum silaus	38	13	10	9	5	9	8	9	9
Solanum dulcemara	2	1	0	0	0	0	0	1	1
Sonchus arvensis	5	0	0	0	0	0	1	0	0
Sonchus asper	12	4	1	1	0	0	2	4	3
Sonchus oleracea	2	1	0	0	0	0	0	1	1
Stellaria graminea	13	5	4	2	1	5	2	6	4
Stellaria media	12	1	0	0	0	1	0	3	1
Succisa pratensis	5	2	2	2	1	3	2	2	3
Taraxacum agg.	154	16	16	15	9	29	21	18	13
Thalictrum flavum	4	3	1	2	1	1	1	2	2
Tragopogon pratensis	4	1	0	1	0	0	2	1	3
Trifolium campestre	1	0	0	0	0	0	0	1	0

Appendix 6.1 (continued)

Species	total	PA20	divind	MG4com	MG5com	ESA	NVC60	PAarea	COVarea
Trifolium dubium	73	11	10	11	6	15	12	12	9
Trifolium hybridum	4	1	0	0	0	1	0	0	1
Trifolium medium	1	1	1	1	0	0	0	1	0
Trifolium pratense	147	19	20	16	10	33	23	19	17
Trifolium repens	194	19	19	16	10	43	24	19	17
Trisetum flavescens	59	12	11	15	10	13	12	8	6
Triticum aestivum	7	1	0	0	0	0	0	3	1
Ulex europaeus	1	1	0	0	0	0	0	1	1
Ulmus procera	1	1	0	0	0	0	0	1	1
Urtica dioica	34	3	0	0	0	7	3	3	4
Veronica arvensis	4	1	0	1	1	0	1	2	1
Veronica catenata	3	2	1	0	0	0	1	2	1
Veronica persica	2	0	0	0	0	0	0	1	0
Veronica serpyllifolia	4	2	2	1	1	2	0	2	2
Vicia cracca	80	19	18	15	7	20	15	15	16
Vicia faba	7	3	1	1	1	0	1	3	0
Vicia hirsuta	3	0	0	1	0	2	0	1	0
Vicia sativa	30	5	8	7	5	8	5	8	4
Vicia tetrasperma	2	0	0	0	0	1	0	1	0
Viola arvensis	3	1	0	0	0	0	0	1	1
Viola canina	1	0	0	0	0	0	0	0	0
X Festulolium loliaceum	7	2	1	1	0	1	1	1	1