AN INVESTIGATION INTO THE SUCCESSFUL ESTABLISHMENT OF A HAY MEADOW COMMUNITY ON CLAY CAPPED LANDFILL

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ABSTRACT

There is a pressing need to develop means of creating habitats that can contribute to the conservation of species threatened by loss of existing natural and semi-natural habitats. Decommissioned landfill presents excellent opportunities for the creation of a number of habitat types. However, landfills are often sealed with a compacted clay cap which not only creates harsh growing conditions and poor soil quality but also results in an impoverished soil fauna making it a challenging environment in which to effect rapid habitat creation. It is currently unclear to what extent the established principles of hay meadow creation are transferable to clay capped landfill. This thesis considers the creation of a hay meadow community and focuses on three properties that are influential in their development and that can be manipulated in experimental conditions: the soils, vegetation and micro-topography. It also explores methods of accelerating the establishment of a hay meadow community, thus encouraging a rapid green cover to provide shelter on the inhospitable bare soil to encourage seedling recruitment as well as addressing potential conflicts with public acceptability of landscape changes during restoration. Soil was identified as being the key factor in the successful establishment of a hay meadow community and the introduction of micro-topographic undulations as having the potential to increase the botanical diversity. Attempts to accelerate the colonisation of the clay cap through the introduction of earthworms were unsuccessful although a hypothesis to address the potential conflict between science and socio-cultural factors was made. This involves the swift establishment of a green cover across the site through the addition of topsoil and ameliorant, the sowing with fast-growing grass species and the creation of small patches of habitat with sown assemblages of species based on semi-natural counterparts.

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AUTHOR'S DECLARATION

I hereby declare that I am the sole author of this thesis and that this is a true copy of this thesis.

1.0 INTRODUCTION

1.1 Habitat loss and restoration

Habitat loss is the primary environmental cause of biodiversity decline at local, regional and global scales (Balmford *et al.* 2005). The extent of habitat loss is clearly demonstrated by the decrease observed in temperate low-intensity farming systems that formed much of the semi-natural landscapes of Europe (Bakker and Berendse 1999). For example, unimproved species-rich grasslands, which were once widespread over much of Western Europe (van Dijk 1991), underwent a remarkable decline in the 20th Century. It was estimated that by 1984 semi-natural grassland had declined by 97% in lowland England and Wales (Fuller 1987; Parker 1995; Dryden 1997). Losses continued during the 1980s and 1990s and were recorded at 2-10% per annum in some parts of England (JNCC 1995). Similarly, by the end of the 20th Century only one sixth remained of the area of heathland which existed in 1800 (Farrell 1993) and only 62% of semi-natural woodland that existed in 1930 (Kirby *et al.* 1995). These losses in semi-natural habitats are linked both to development (Evans 1992) and changes in agricultural practices (Ratcliffe 1984) with grasslands being a particular focus of the latter (Crofts and Jefferson 1999). Agricultural intensification led to the extensive development of nutrient-demanding, productive *Lolium perenne* (perennial rye-grass) grasslands which are managed for grazing and silage production, which widely replaced practices such as traditional hay making (JNCC 1995). Townsend *et al.* (2004) identify nutrient enrichment, particularly through the use of artificial fertilisers, under-grazing, the lack of aftermath grazing and inappropriate herbicide use as being responsible for the degradation of hay meadows. The resulting loss in the botanical diversity of these grasslands (Blackstock *et al.* 1999) has been found to be key to the decline in the diversity of fauna such as birds (Siriwardena *et al.* 1998) and invertebrates (Woodcock *et al.* 2005; Woodcock *et al.* 2008).

In the last twenty years stronger protection for wildlife, combined with changes in grants and support payments have significantly reduced the rate of ongoing habitat loss, and indeed promoted some restoration and re-creation (Townsend *et al.* 2004). In addition, the restoration of internationally rare and threatened ecosystems has long formed part of the UK´s contribution to the international commitment to meeting the challenge posed by the Convention of Biological Diversity, Rio, 1992 (Department of the Environment 1995). For example, in the UK the Biodiversity Action Plan requires the recreation of 2448 hectares of various types of species-rich grassland in lowland England by 2010 (JNCC 2006). Consequently there has been a rapid increase in research into restoration ecology. In the Journal of Applied Ecology alone the number of papers on restoration ecology published has increased from 0% in 1982 to 12% in 2003 (Omerod 2003). As much of habitat restoration is and will be achieved through agri-environment schemes (Ovenden *et al.* 1998) such as the Environmentally Sensitive areas (ESAs) (Pywell *et al.* 2002), much research to date has focused on restoring or re-creating habitats on agriculturally improved land. This research has investigated factors such as the effect of soil nutrients on species-richness (Gough and Marrs 1990a, Gough and Marrs 1990b, Janssens *et al.* 1998), methods of reducing the nutrient status of soils to improve success in grassland (McCrea *et al.* 2001) and heathland (Allison and Ausden 2004) creation, the use of sulphurous soil amendments in the restoration of heathlands and acid grassland (Tibbett and Diaz 2005; Diaz *et al.* 2008) and multi-site experiments comparing soil pre-treatments, natural regeneration, nurse crops and seed mixes (Wells *et al.* 1986; Wells *et al.* 1989; Pywell *et al*. 2002).

1.2 Opportunities for habitat restoration and creation on post-operative landfill

Landfill is currently the most prevalent method of waste management in many countries. In the UK 60% of municipal waste went to landfill in 2006 (Department for the Environment, Food and Rural Affairs 2008). The number of active landfill sites in England and Wales declined from approximately 3,400 in 1994 to 2,300 in 2001 covering an estimated 28,000 hectares of land (Surrey County Council 2008). Decommissioned landfill sites therefore cover significant areas of land, often in or near urban centres (Handel *et al.* 1997). Although changes in waste management practices are likely to reduce the quantity of waste, there is likely to be a continuing need for landfill at least for materials that cannot be recycled or dealt with cost-effectively in other ways. Furthermore, many recycling and composting processes produce by-products such as ash which then require disposal (ETSU 1998). Although agricultural grassland is still the most common after-use for landfill sites across the world, the legislative framework that has developed in the UK over the last 20 years increasingly promotes ecological restoration (Townsend *et al.* 2004; Bradley *et al.* 2006). Obligations set out under legislation and policy such as The Town and Country Planning (EIA) (England and Wales) Regulations 1999 (Council Directive 1999/31/EC 1999), The Town and Country Planning Act 1990 and the Planning and Compensation Act 1991, Planning Policy Statement 9 on Biodiversity and Geological Conservation (PPS9) (Office of the Deputy Prime Minister 2005) and Biodiversity Action Plans (Department of the Environment 1995) significantly increase opportunities available for habitat creation on decommissioned landfill sites.

1.3 Habitat creation on clay capped landfill

Landfill engineering essentially consists of four components: an impermeable basal seal, the landfill cap, landfill gas and leachate control systems and an extraction system (ETSU 1998). The landfill cap is the most influential factor in habitat creation due to the highly impervious substrates, usually clay, that are used and the way in which they are deliberately compacted to create a seal (Butt *et al.* 2004). This means that habitats characteristic of distinctly acidic, calcareous or free draining conditions such as heathland, acid grassland and calcareous grassland are not suitable for creation on clay capped landfill. Habitats which are more suited to the soil conditions created by the clay cap are neutral pasture, hay meadows, woodlands and wetlands. Woodlands and wetlands do, however, require forward planning. A review of the world literature on this subject indicates that maximum depths achieved by tree roots are usually between 1-2 m (Moffatt 1995). This means the clay cap would need to exceed 1 metre in depth (Watson and Hack 2000), preferably 1.2 m (Department for Communities and Local Government 2008), to avoid penetration and thus the release of toxic gases and leachate. Neutral pastures and hay meadows are the habitats which are perhaps the most suited to the conditions as they are not affected by the depth of the clay cap. Consequently hay meadows have been chosen as the focus of this study.

1.3.1 Hay meadow creation and restoration

1.3.1.1 Hay meadows

Neutral hay meadows are probably best described as grassland found within the lowlands of Britain consisting of swards of drought-free mesotrophic soils of pH 4.5 to 6.5 (Parker 1995). Historical management of cutting the vegetation for hay is the main factor which separates hay meadows from mesotrophic lowland pasture (Sutherland and Hill 1995). The constant removal of nutrients in the form of the hay cut maintains a low-nutrient status which allows the species-rich plant community associated with neutral hay meadows to develop (Sutherland and Hill 1995). Hay meadows were traditionally managed to produce a hay crop with which to feed stock over-winter. These meadows are also sometimes grazed by stock over the wintering months but a crucial factor in the development of these species-rich communities is the absence of grazing from early spring (February or March in the lowlands) until late summer when the hay is cut (Crofts *et al.* 1999). The precise timing of cutting and details of after-grazing vary between sites and according to local customs (Sutherland and Hill 1995).

The National Vegetation Classification (NVC), a system by which all vegetation types found within Britain are classified, is a useful tool in describing typical plant communities that are associated with neutral hay meadows. The table below summarises these communities:

Table 1: Neutral hay meadow NVC communities (based on Rodwell 1992)

NVC Community	***Brief description***	***Status***
MG3	Sweet vernal-grass (*Anthoxanthum odoratum*) – wood crane's-bill (*geranium sylvaticum*) grassland. An upland grassland community where traditional hay management treatment has been applied in harsh, sub-montane conditions. Typically supports a dense growth of grasses and herbaceous dicotyledons up to 80cm in height.	Rare, restricted to a few upland valleys in the north of England
MG4	Meadow foxtail (*Alopecurus pratensis*) - great burnet (*Sanguisorba officinalis*) grassland. Characteristic of areas of traditional hay meadow management on seasonally flooded land with alluvial soils. Typically supports a species-rich and varied sward of grasses and dicotyledons.	Restricted distribution
MG5	Crested dog's-tail (*Cynosurus cristatus*) – black knapweed (*Centaurea nigra*) grassland. Characteristic of grazed traditional hay-meadows on circumneutral brown soils in lowland Britain. Typically rich in dicotyledons but variable in character ranging from a tight, low-growing sward to lush growth up to 60cm.	Declined although remains widespread but fragmented

1.3.1.2 Techniques of hay meadow creation

Much research has been conducted on methods of grassland creation (e.g.s Wells *et al.* 1986; Wells *et al.* 1989; Firbank *et al.* 1992; Ash *et al.* 1992; Pywell *et al.* 2002; Edwards *et al.* 2006). A major constraint in grassland restoration is the impoverishment of lowland species-pools (and seed sources) through habitat loss and fragmentation which has meant that grassland re-establishment has usually been seed-limited (Poschlod *et al.* 1998; Bullock *et al.* 2002). Important focuses have therefore been how to best introduce seeds and how to manage systems to get maximum establishment of sown species (Stevenson *et al.* 1997; Bakker and Berendse 1999; Pywell *et al.* 2003; Jongepierová *et al.* 2007). Literature on hay meadow creation and restoration identifies the following techniques for sowing seed and encouraging recruitment of herbaceous plants within the grassland sward:

1. Hay spreading i.e. taking a hay cut from an established grassland and spreading directly over the surface of the grassland creation site (Manchester *et al.* 1999; Mortimer *et al.* 2002; Edwards *et al.* 2006). This method has the advantages of sourcing the seeds from a known NVC grassland type and it is relatively inexpensive (Parker 1995; Manchester *et al.* 1999). Research in the 1980's suggested hay bales to be a significant source of seed (Wells *et al.* 1981; Wells *et al.* 1986).
2. Direct sowing: This method has advantages in terms of sourcing the seed. Seed can be harvested from established nearby grassland using a variety of methods, including hand collection, vacuum harvesting (Stevenson *et al.* 1997), brush harvesting (Edwards *et al.* 2006) or sourced commercially, preferably from a supplier of seeds of local provenance (Jones and Hayes 1999). A seed mix of 85% grass and 15% forb species at a typical sowing rate of 30kg/hectare sown in the autumn is recommended (Dryden 1997).
3. Direct transfer of pot-grown herbs: This method is often used in the restoration of improved pasture (Parker 1995) although Wells *et al.* (1989) reported mixed success using this technique in their research, with survival rates ranging from 0 to 87.8%.
4. Turf translocation: This is a method whereby soils are moved with their vegetation and any animals in order to rescue or salvage habitats that would be lost due to changes in land-use (Box 2003). However, this method has rarely been used (Good *et al.* 1999) and in many cases a net loss in diversity and habitat quality has been observed, largely due to unsuitable management of the donor site (Box 2003).
5. Introducing hemi-parasitic plants: Research during the 1990's also highlighted the benefits of introducing plants such as *Rhinanthus minor* (yellow rattle) and *Euphrasia spp.* (eye-brights).

These plants suppress the productivity of grasses which facilitates the development of a more forb-rich sward (Crofts *et al.* 1999; Bullock and Pywell 2005). This method is most frequently used on restoration of improved pastures to suppress the growth of aggressive competitive grasses, allowing the more desirable herbaceous plants to establish (Dryden 1997).

6. Introducing nurse plants: The establishment of hay meadow species is facilitated through creating a swift green cover by sowing plants such as *L. perenne* or *L. multiflorum* which provide shelter for seedlings as well as suppressing excessive growth of competitive species (Mitchley *et al.* 1996). However, their use in grassland restoration is relatively poorly studied (Pywell *et al.* 2002) with mixed results. Mitchley *et al.* (1996) encountered problems with the initial establishment of the nurse crop on soils of low fertility and in one study the abundance of the nurse crop, *L. multiflorum*, was detrimental to the early establishment of a lowland wet grassland sward (Manchester *et al.* 1997). Investigation into alternative methods of swiftly achieving a green cover to provide shelter on the inhospitable bare soil of creation sites to encourage seedling recruitment would therefore be of use and is a topic explored in this thesis.

1.3.2 Constraints associated with hay meadow creation on clay capped landfill

Although a wide range of literature on habitat creation is available (Baines and Smart 1991; Parker 1995; Dryden 1997; Gilbert and Anderson 1998) landfill sites present a unique environment requiring the modification of standard techniques (Watson and Hack 2000). The availability of soil is a major limitation on successful habitat creation (Bradley *et al.* 2006). Topsoil, often soil which has been scraped from the original site prior to landfill operations and stored, is usually laid over the cap (Watson and Hack 2000). There are a number of problems associated with the associated stripping, storage and handling processes which are echoed in habitat creation on opencast coal mines (Corbett *et al.* 1996). Some of these were highlighted at an opencast mine at Bryngwyn where reinstated soil was extremely impervious resulting in surface water logging after even light rainfall and the land becoming dry and cracked in periods of dry weather making it hostile for plants and animals alike (Scullion 1994). Earthworm populations at Bryngwyn were drastically reduced with Martin *et al.* (1998) finding a mortality rate of >90%. Similarly, during opencast mining, soil stripping, storage and replacement result in major disturbance of microbial communities (Harris *et al.* 1993; Scullion 1994). These problems are even more acute in the extreme case of landfill due to the deliberate compaction of a dense clay substrate to serve as a cap (Butt *et al.* 2004).

An additional consideration in the restoration of landfill is that of public acceptability of the outcome following restoration (Watson and Hack 2000) and of landscape changes during restoration (Pfadenhauer 2001). Higgs (2005) criticises ecological restoration practitioners for failing adequately to address these conflicts between science and socio-cultural factors. The consequence of these issues is that the primary objective of standard practice following capping remains to provide a green cover as rapidly as possible to avoid eyesores within the landscape, with the creation of habitats of high nature conservation being a secondary objective. Therefore the challenge is rapidly to establish a green cover whilst maintaining ecological integrity within the site by creating habitats that reflect the composition of their semi-natural or unimproved counterparts. One solution explored in this thesis may be to establish a green cover across the whole site and then create more diverse communities on part of the site.

1.3.2.1 Soil physical and chemical properties

Soil type, soil structure, soil water relations and soil fertility are perhaps the most difficult and underestimated aspects of successful habitat creation (Bradley *et al.* 2006). Soil structure and chemistry are recognised as parameters that strongly influence the structure of vegetation (Svenning and Skov 2002). The physical properties of soil are the production of continued interactions between soil biota and their abiotic milieu (Coleman and Crossley 2003). Reinstated soils differ from undisturbed soils which are composed of layers or horizons. Undisturbed soils extend from the parent material, or C horizon, which is usually bedrock, through the subsoil, or B horizon, to the topsoil, or A horizon to the soil surface which is usually an organic litter layer, also known as the O horizon (Courtney and Trudgill 1984; Bradley *et al.* 2006).

The soil horizons as described above do not exist in clay capped landfill. Watson and Hack (2000) acknowledge that although the clay cap could be considered to represent the B horizon and imported topsoil the A horizon, they form distinct layers and lack the gradation between horizons seen in naturally formed soils. The arrangement of particles into aggregates, and the size, shape and distribution of pores both within and between aggregates determines the soil structure (Courtney and Trudgill 1984). This is of vital relevance to the ability of soils to support plant, animal and microbial life (Bradley *et al.* 2006). These pores hold water, allow drainage, allow entrance of oxygen and removal of carbon dioxide from the soils and allow roots to penetrate (Rowell 1994). Clay can be described as a soil type composed of a small particle size, typically less than 2 millimetres (Courtney and Trudgill 1984; Rowell 1994). Clay soils are frequently poorly drained but often very rich in nutrients. They can hold large amounts of water, although much of it is held in a form unavailable to plants (Rowell 1994). They are also prone to drought and shrinkage during dry weather when large cracks can form (Watson and Hack 2000).

It has long been recognised that the stripping, storage and handling processes associated with reinstated soils and the deliberate compaction of the clay to form a cap on the landfill are of detriment to the soil's physical structure (Pringle 1958; Davies 1973; Lloyd 1984; McCrae 1989; Scullion 1994; Butt *et al.* 2004). Scullion (1994) found handling processes to have adverse effect on micro-aggregate and macro-aggregate stability at the Glyn Glas opencast mine. The results from Scullion's study indicted that this damage was cumulative and recovery was slow, particularly at depth. This breakdown in aggregate stability leads to soils having a higher bulk density which in turn leads to poor water infiltration and gas exchange. Lipeic *et al.* (1998) found that pore size pattern significantly alters many soil properties affecting water movement and root growth. Gemtos and Lellis (1997) found that induced compaction decreased the volume of soil, thus increasing bulk density, which delayed emergence and reduced root growth in cotton and sugar beet. This creates a vicious circle in that a continuous replenishment of organic matter is required for macro-aggregate stability (Ashman and Puri 2002). This is aggravated further by compaction making soils prone to water-logging which discourages the decomposition and recycling of nutrients (Scullion 1994).

The availability and relative concentrations of chemical elements, or nutrients, found within the soil plays a large role in dictating plant germination, growth and competition within terrestrial ecosystems (Courtney and Trudgill 1984; Walker *et al.* 2004). The availability of nutrients in large quantities promotes high plant productivity, which in an agricultural system for example, is of benefit. These conditions would not, however, be favourable in a grassland ecosystem of high nature conservation value as it would promote the vigorous growth of a small number of species that are more competitive than other more desirable plant species (Janssens *et al.* 1998; Simmons 1999). Much research has therefore focused on the role of nutrients in habitat creation on ex-arable farmland (Gough and Marrs 1990a; Gough and Marrs 1990b; Snow *et al.* 1997; McCrea *et al.* 2001; Allison and Ausden 2004; Walker *et al.* 2004; Tibbett and Diaz 2005; Diaz *et al.* 2008). Phosphorus appears to be the key nutrient determining soil fertility in agricultural soils and distinguishing them from species-rich grassland soils (Gough and Marrs 1990a; Janssens *et al.* 1998). Although nitrogen additions to formerly species-rich grassland are associated with the decline in diversity (Thurston 1969; Harper 1971; Berendse *et al.* 1992) research has not found this to be a key factor in habitat creation schemes (Gough and Marrs 1990a). The role of potassium is less clear (McCrea *et al.* 2001) but research indicates that unlike phosphorus, moderately high potassium levels are compatible with high levels of floristic diversity (Janssens *et al.* 1998). Marrs (1993) identified possible soil fertility reduction methods as grazing, arable cropping, hay sileage cropping, topsoil removal and inversion by deep ploughing.

Since research into the management of former opencast coal mined land commenced in the 1950's there has been an emphasis on the nutrient deficiencies associated with replaced soils (Scullion 1994). Although theory suggests that this low nutrient status would be beneficial in habitat creation, the addition of organic amendments is commonly recommended to improve the structure of the clay soil-forming prior to vegetation establishment. Watson and Hack (2000) recommend that soil substitutes should not be used on their own but mixed well with true soil. In a study on the effects of amendments used in landfill restoration Gregory and Vickers (2003) concluded that organic amendments might improve the structure of the clay soil-forming material prior to vegetation establishment. Gilbert and Anderson (1998) recognised that a high quality tilth is required to create grassland of nature conservation value as with any other grassland. In contrast, Simmons (1999) suggests that topsoil should not be used on areas where wildflower seed is to be sown as the fertility of the topsoil will encourage aggressive competitive species to grow at the expense of the more desirable forb species and the diversity will be lost. Existing literature on the use of soil amendments, and therefore the manipulation of soil's fertility and structure, is therefore conflicting and it is currently unclear as to whether the accepted principles of grassland creation on ex-arable are transferable to clay capped landfill.

1.3.2.2 Earthworm abundance and species

The soil biological community is key to the sustainable function of soils (Bradley *et al.* 2006), and earthworms are of particular importance for their role in generating and maintaining structure (Davidson and Grieve 2006). In fact earthworms are regarded as ecosystem engineers due to many species' ability to actively change and ameliorate their soil environment (Lavelle *et al.* 1997). Several studies have shown that earthworms improve the structure of the soil through increasing pore space, particularly macropore volume (Teotia *et al.* 1950; Ehlers 1975; Stewart 1981; Davidson and Grieve 2006), thus increasing soil aeration and drainage. Furthermore, earthworms play an important role in the recycling and distribution of nutrients within the soil. They make nitrogen available for plant growth by feeding on organic matter within the soil then voiding casts with a low carbon : nitrogen ratio, the latter often containing fragmented litter which in turn is readily broken down by micro-organisms (Edwards and Bohlen 1996; Sims and

Gerard 1999). Bouché (1977) divided earthworms into three major groups based on their feeding and burrowing characteristics, although not all species can be readily assigned to just one group (Sims and Gerard 1999). The characteristic behaviour of these groups of earthworms (Table 2) means they have differing functions within the soil. The presence of earthworms in soils can therefore be considered to almost always be beneficial and their absence detrimental to soil status (Butt 1999). This is illustrated in a study by Clements *et al.* (1991) where soils from which earthworms had been absent for twenty years showed increased bulk density and reduced penetrability, infiltration rate and organic matter compared with similar earthworm-rich soils. Studies have also shown earthworms to have positive effects on the growth of vegetation. An interesting example of this is a study where Stockdill (1982) recorded a 70% increase in grass production in the first year following the introduction of a European species of earthworm into a New Zealand pasture where earthworms were previously absent. Similar responses of increased plant growth to the presence of earthworms in glasshouse experiments were found in relation to wheat and clover (Baker *et al.* 1997) and ryegrass (Baker *et al.* 2003).

These accounts are therefore indicative that earthworms may be a useful tool in increasing the rapidity of the establishment of forb-rich grassland sward. However, research at an opencast mine at Bryngwyn (Scullion 1994) found that earthworm numbers are low in newly replaced soils. Where their recovery depends on natural re-colonisation, it may take ten years or longer for a population to develop which is typical of similarly managed, undisturbed land. Scullion reported little evidence of any change in soil conditions below a 10 centimetre depth for periods up to seven years. Scullion also found that the rate of earthworm population spread on replaced land ranged from five to fifteen metres per year, depending on management and weather conditions, which is consistent with other similar studies (Van Rhee 1969; Stockdill 1982; Hoogerkamp *et al.* 1983; Marinissen and van den Bosch 1992). Both naturally colonised earthworm populations and introduced earthworms and their effect on the created hay meadow's floral community has consequently been addressed in this thesis.

Table 2: The behavioural groups of earthworms

Behavioural group (Bouché 1977)	***Behaviour / functional charactersitics***	***Physical characteristics***	***Examples of species***
Epigeic	Litter dwelling species that live outside of the mineral substrata in litter, decomposing tree-trunks and compost.	Highly mobile, an adaptation to evade predators; Small in size; Have relatively high respiration rate; and usually red, vinaceous or rosy in colour	*Lumbricus castaneus* (chestnut worm) and *Lumbricus rubellus* (red worm)
Endogeic	Horizontal burrowing species that live permanently in horizontal burrows in the mineralo-organic sub-stratum of the soil. Their food consists of dead roots and both microflora and fauna ingested within the soil. Endogeic species cast below ground in burrows. Effect structure of soils through burrow-forming	Most of this group's characteristics indicate adaptation to living continuously within the soil, for example they have no pigmentation	*Octolasion* family
Anecic	Deep burrowing species, are soil dwelling earthworms with vertical burrows. They are characterised by their intense burrowing abilities and production of burrow complexes. They commonly emerge at night to feed on the surface litter Effect structure of soils through burrow-forming	Typically of a moderate to large size and have some pigmentation, often brown to brown black.	Larger species of the genus *Aporrectodea*. *Lumbricus terrestris* (lob worm)

1.3.2.3 Environmental heterogeneity

All natural environments are heterogeneous (Caldwell and Pearcy 1994) and this heterogeneity occurs in terms of both biotic and abiotic environmental factors (Wilson 2000). Topographic heterogeneity was identified by early ecologists to affect the biotic communities (e.g. Cowles 1899) and there are many examples of species diversity increasing with heterogeneity (Huston 1994; Rosenzweig 1995; Brose 2001; Tessier *et al*. 2002). Micro-topographic differences (<1m) can result in variability in the physical environment and chemical and biological processes over small spatial scales (Larkin *et al.* 2006). Rodwell (1992) acknowledges this effect by identifying clear patterning in grassland swards where distinctive grassland types have developed over drier ridges and within damper furrows. The manner in which a clay cap is laid and compacted on post-operative landfill results in a somewhat homogenous micro-topography lacking the micro-habitats associated with species diversity. Topographic heterogeneity is a relatively easy parameter to manipulate in restoration ecology and Larkin *et al.* (2006) highlight the requirement for studies into techniques of creating topographic variation at a range of scales and habitats.

1.4 Rationale for the current study

The combination of the above problems means that more established and traditional methods of habitat creation which are successfully employed in other situations may be less effective in the unique situation of landfill. It is currently unclear to what extent the established principles of hay meadow creation are transferable to clay capped landfill. This thesis considers the creation of a hay meadow community and focuses on three properties that are influential in their development that can be manipulated in experimental conditions: the soils, vegetation and micro-topography. It also explores whether the establishment of a hay meadow community can be accelerated without the addition of ameliorant via the effect of incolated earthworms on the clay cap, thus attempting to address the issue of increasing the rapidity of establishing a green cover to provide shelter on the inhospitable bare soil to encourage seedling recruitment as well as potential conflicts with public acceptability of landscape changes during restoration.

1.5 Aims and objectives

The overall aim of this study is to investigate the factors involved in the successful establishment of a hay meadow community on a clay capped landfill site and mechanisms by which restoration processes can be accelerated.

Three objectives were set. These are to investigate:

1. The effect of manipulation of soil and vegetation on the development of the target plant community, edaphic factors and earthworm populations on clay capped landfill that has been stabilised with green cover.
2. The use of earthworms as ecological engineers in speeding up the establishment phase of the grassland sward via their impact on the soil's physical and chemical properties.
3. The effect of manipulation of micro-topography to create ridges and furrows on the development of the target plant community, edaphic factors and earthworm populations on clay capped landfill.

2.0 METHODS

2.1 The study site

The experimental plots for this study are located at Dimmer Waste Disposal Site near Castle Cary in Somerset (ST618319; lat 51º04’38.10’’ N; long 2º32’59.93’’ W) as illustrated on Figure 1. Prior to use for waste disposal the site was a munitions store during the Second World War. The original substrate and topsoil was excavated to accommodate the munitions store and the soil was stored on adjacent farmland. The underlying substrate was blue-lias clay which formed in the Jurassic, characteristically with limestone bands resulting in a slightly calcareous characteristic (Findlay 1965). The original topsoil from the site can be described as a fine textured silty clay soil.

The site is divided into a series of cells which are depicted in Figure 2. Carymoor Environmental Trust manages the habitat creation on the cells once de-commissioned and has encouraged research into habitat restoration. Once de-commissioned, all cells are capped with the blue-lias clay although its depth and addition of topsoil and/or green compost varies between cells according to availability and the requirement for planned land use following capping. Where topsoil is graded across the clay cap the stored soil from the original excavation of the site is used. Compost produced as part of a recycling initiative to deal with domestic green waste is also used as ameliorant. Green waste is collected, chipped and processed at temperatures exceeding 65 degrees centigrade. The current study established three experiments to investigate objectives 1, 2 and 3 as stated in section 1.5, and these were located on cells 18, 27 and 21 respectively. Figure 3 shows aerial views of the locations of the three experimental plots.

Figure 1: Location of the study site

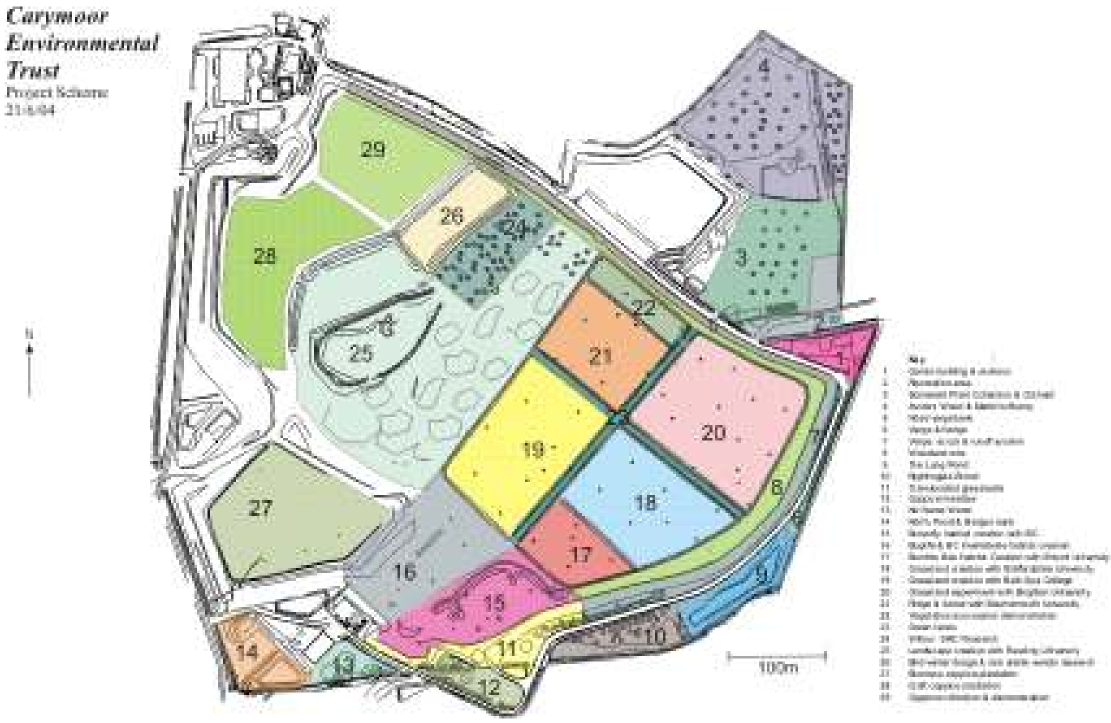


Figure 2: Site map of cells at Carymoor

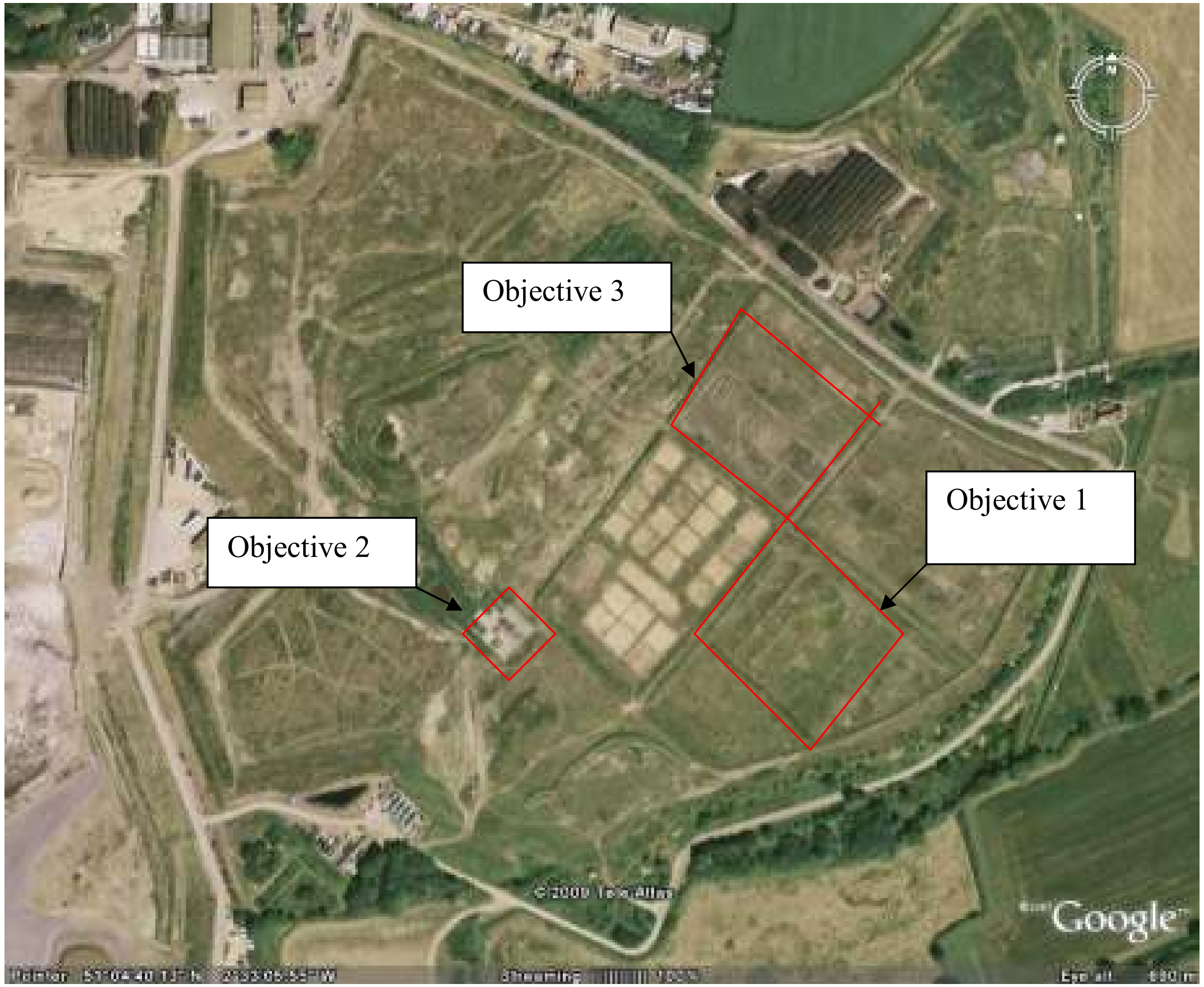


Figure 3: Aerial photograph of experimental plots investigating objectives 1, 2 and 3

2.2 General methods

2.2.1 Seeding methods

The experimental plots used for all four objectives comprised of a bare clay cap prior to any experimental manipulations. No ground preparation was undertaken, mainly due to the physical constraints posed by the clay cap. Disturbance or rotovation of the soil would have been useful but the clay at Carymoor is often either water-logged and not practical for accessing machinery or too cracked and dry to facilitate effective ground disturbance. Commercial seed mixes were used in all three experiments and seed was sown by hand at a rate of 1.8g per m^2. For each experiment, seed was weighed for each plot and placed in bags. Each plot was then walked in transects 2.5 m apart and a handful of seed was taken from the bag at a time and scattered in a swinging motion.

2.2.2 Vegetation sampling

The frequency of vegetation was sampled according to Sutherland (2002). Vegetation was recorded using $1m^2$ quadrats, sub-divided into 10cm squares. The locations of these were decided using randomly generated numbers, incorporating a 1m buffer zone around the perimeter of each plot to minimise edge effects. The presence of each rooted species found in each sub-square of the quadrat was recorded. The summation of the occurrences of each species in each quadrat provided frequency scores. The frequency of bare ground was also recorded using this method.

2.2.3 Soil analysis

2.2.3.1 Soil sampling

Soil samples were collected during the growth season of the vegetation in June using a core auger with a 3 cm diameter. Twenty samples were taken from each plot following a 'W-shaped pattern to a depth of approximately 15 cm. Soil from each plot was subsequently bulked and frozen at -18ºC immediately after collection to store it prior to analysis.

2.2.3.2 Moisture content

Soil moisture content was analysed in accordance with Rowell (1994). Soil samples were placed in plastic trays which were weighed with and without the soil. Samples were then placed in an oven set at 60°C overnight and weighed the following morning. Samples were placed back in the oven for another hour and re-weighed to ensure all moisture had evaporated. Percentage moisture loss was then calculated.

2.2.3.3 Organic matter

Organic matter was assessed using a standard loss on ignition method (Rowell 1994). The oven-dried samples were gently ground using an electric pestle and mortar and passed through a 2mm sieve. Crucibles were weighed and approximately 2g of each sample were placed in the crucibles. Samples were ashed in the furnace over night at a temperature of 400°C. Samples were weighed the following morning, re-ashed for an hour and weighed again to ensure the weights were constant indicating loss of all organic matter. Percentage loss of organic matter was then calculated.

2.2.3.4 Phosphate analysis

Phosphate levels of the soil were measured using the standard KCl extraction method according to Rowell (1994). Oven dried soil samples were gently ground and passed through a 2mm sieve. Phosphate was extracted from duplicate 2.5 g soil samples by shaking in 25 ml of 2M KCl for 60minutes. Extracts were filtered through Whatman no. 42 filter paper and stored at -18 ^{o}C until analysis. Phosphate concentrations in extracts were determined colourmetrically. A 20 ml sub-sample of each extract was taken and 1 ml of Ammonium molybdate solution (0.2 M $(NH_4)_6Mo_7O_{24}.4H_2O$ in 28% H_2SO_4) followed by 3 drops of Stannous chloride solution (0.11 M $SnCl_2$ in 100% glycerol) were added. Absorbance at 650nm was measured using a Varian Cary 50 UV/vis.

2.2.3.5 Nitrate analysis

Nitrate levels of the soil were measured by extracting nitrate using K_2SO_4 according to Rowell (1994) and determining concentrations colourmetrically according to Doane and Howarth (2003) and Miranda *et al.* (2001). As with the preparation of soils for the phosphate analysis, oven dried soil samples were gently ground and passed through a 2 mm sieve. Nitrate was extracted from duplicate 2.5g soil samples by shaking 25ml of 0.33M potassium sulphate for 60 minutes. Extracts were filtered through Whatman no. 42 filter paper and stored at -18 ^{o}C until analysis. Vanadium(III) in dilute acid solution was used to reduce nitrate to nitrite which was then captured using Greiss reagents (sulphanilamide and N-(1-naphthyl)-ethylediamine) to produce a red dye. Absorbance at 540nm was measured using a Varian Cary 50 UV/vis.

2.2.3.6 pH

pH was analysed according to Rowell (1994) whereby 10ml of 0.1M KCl was added to 4g of soil from each plot in sterilin tubes and placed on the electronic shaker for 15 minutes. The pH meter was calibrated using buffer agents of pH4, 7 and 9 and the probe inserted into each soil solution and a reading taken once it had settled. All readings were adjusted by pH 0.5 to allow for the KCl solution.

2.2.4 Earthworm sampling

Earthworm abundance was measured using the hot mustard extraction method as described by Lawrence *et al.* (2002). Colman's English mustard powder was mixed with water at a ratio of 1:2 allowing 50 g of mustard per 0.5 m^2 quadrat. This was left to settle over night to maximise the effectiveness of the irritant allyl isothiocyanate (AITC) in the mustard. Immediately prior to sampling 100 ml of mustard paste was diluted in 7 litres of water and any vegetation or leaf litter was cleared from within the quadrat. The mustard solution was then poured onto the ground in randomly placed (using randomly generated numbers) 0.25 m^2 quadrats and earthworms that emerged from the soil over a 20 minute period were collected and identified using the keys from Sims and Gerard (1999).

The hot mustard extraction method was developed as alternative to the use of formaldehyde due to its associated health and safety risks (Lawrence *et al.* 2002). In an investigation into the efficiency of the hot mustard extraction method of earthworm sampling Bartlett *et al.* (2006) found that it was biased towards large, sexually mature anecic earthworms. In an attempt to address this potential constraint, earthworm sampling on objective one's experimental plots in 2004 compared mustard extraction and hand sorting methods. Two 0.25 m^2 quadrats were taken in each plot using the mustard extraction and two using the time-limited hand-sorting method as developed by Schmidt (2001). Marginally greater numbers of earthworms were recorded using the mustard extraction sampling method than the time-limited hand-sorting method (Table 3) although these differences were not found to be significant. This similarity in the numbers of earthworms collected using these two methods is also indicative that the mustard extraction method is an effective method even on the unmodified clay cap where it may have been anticipated that the mustard solution would have had trouble penetrating. The mustard extraction method was subsequently adopted in this study due to its less destructive and less labour-intensive nature.

Table 3: Medians, ranges between minimum and maximum abundance and results of Mann-Whitney U-Test comparing the numbers of earthworms recorded using the mustard extraction (n = 18) and time-limited hand-sorting (n = 18) methods in 2004

Earthworm species	Mustard extraction		Hand-sorting		*U* value	*P* value
	Median	Range	Median	Range		
Lumbricus castaneus	1.00	7.00	0.50	4.00	135.0	0.377
Lumbircus rubellus	3.50	9.50	3.00	7.00	143.5	0.557
Lumbricus terrestris	0.50	3.50	0.00	2.00	121.0	0.159
Allolobophora chloritica	13.25	27.50	10.00	19.00	129.5	0.303
Apporrectodea caliginosa	7.50	29.50	6.50	12.50	133.0	0.358
Total number of earthworms	28.25	46.50	20.75	32.50	135.0	0.377

2.2.5 Data analysis

Data were analysed using the statistical packages SPSS version 15 and PRIMER version 6 (Clarke and Gorley 2006). Data sets for each variable were analysed for normality and homogeneity of variance using Kolmogorov-Smirnov and Levene's tests. Significant heteroscedasticity of variance was found in data sets from all three experiments and it was not possible to rectify this through attempts to transform the data. Consequently parametric tests were performed where the data were distributed normally and non-parametric tests when the data were not normally distributed. Mann-Whitney U-Tests and t-tests were used to compare data where two dependent variables were being compared. Kruskal-Wallis and ANOVA were performed where three or more dependent variables were involved. Nemenyi’s Test (Wheater and Cook 2000) was used as a post-hoc test to determine where any significant differences detected occurred. Spearman Rank Correlation Tests were used for correlation analysis of non-parametric data and Pearson’s Correlation to analyse parametric data. Analysis of similarities was conducted using the multivariate ANOSIM test (Clarke and Gorley 2006) to test the significance of difference between the overall vegetation and earthworm communities in different treatments. This computes a test statistic (r) which reflects the observed differences in overall community composition between treatments and is particularly useful as it takes into account all species recorded and their abundance. The r statistic falls within the range of -1 to 1. Where r = 1 or -1 this shows that all replicates within a site or treatment are more similar to eachother than any replicates from different sites. Where r = 0 this shows that similarities within and between treatments are, on average, the same.

3.0 THE EFFECT OF MANIPULATION OF SOIL AND VEGETATION ON THE DEVELOPMENT OF A HAY MEADOW COMMUNITY, EDAPHIC FACTORS AND EARTHWORM POPULATIONS ON CLAY CAPPED LANDFILL THAT HAS BEEN STABILISED WITH GREEN COVER

This Chapter investigates objective 1 of this thesis and is written in the format of a paper, which was accepted for publication in 'Restoration Ecology' on 27th November 2008.

3.1 Summary

This chapter investigates the effect that manipulation of soil and vegetation conditions has on plant community development during attempts to create neutral hay meadow communities on a clay capped landfill in Somerset, UK that had already been established with a green cover. The objectives are, (1) to determine the effect of manipulation of soil and vegetation on the development of the target plant community, (2) to identify whether these treatments had an effect on edaphic factors (physical and chemical properties, earthworm populations); and (3) to establish which, if any, of these edaphic parameters are underlying factors in determining the vascular plant community composition. In 2001 a commercial hay meadow seed mix was sown on three substrate treatments: (i) bare clay, (ii) a mixture of topsoil and compost ameliorant in equal proportions and (iii) over naturally colonised vegetation. Plant community development and edaphic factors were monitored between 2001 and 2007. Although initially the presence of ameliorant promoted germination and growth of seeded species, after 2004 non-seeded competitive grasses dominated the sward. Where ameliorant was removed the target community continued to develop proving this to be the most successful treatment. This was found to be a due to suppression of competition from *Elytrigia repens* (Common couch) which grew most abundantly on soils with high organic matter and high soil water levels within the soil. No evidence was found of earthworm abundance affecting plant community development.

3.2 Introduction

Landfill is currently the most prevalent method of waste management in many countries, for example, in 2006 60% of municipal waste went to landfill both in the US (Environmental Protection Agency 2008) and in the UK (Department for the Environment, Food and Rural Affairs 2008). Decommissioned landfill sites therefore cover significant areas of land, often in or near urban centres throughout the world (Handel *et al.* 1997). Although agricultural grassland is still the most common after-use for landfill sites across the world, the legislative framework that has developed in the UK over the last 20 years increasingly promotes ecological restoration (Townsend *et al.* 2004). Hay meadows are an appropriate habitat to create on capped landfill given the neutral substrate as well as problems associated with cap penetration from tree roots or in wetland creation (Handel *et al.* 1997; Watson and Hack 2000; Athy *et al.* 2006). In addition to this, grasslands have been a particular focus for agricultural intensification over the last 50 years (Crofts and Jefferson 1999) and this has been a major contributing factor to an estimated 97% loss in lowland unimproved grassland between 1930 and 1984 in England and Wales (Fuller 1987). Consequently, hay meadows are recognized as an internationally rare and threatened ecosystem and their restoration has long formed part of the UK´s contribution to the international commitment to meeting the challenge posed by the Convention of Biological Diversity, Rio, 1992 (Department of the Environment 1995).

Previous research on grassland creation has largely focused on conversion of agricultural land and has particularly investigated the effect of soil nutrients on species-richness (Gough and Marrs 1990a; Gough and Marrs 1990b; Janssens *et al.* 1998) methods of reducing the nutrient status of soils to improve success in grassland creation (McCrea *et al.* 2001) and multi-site experiments comparing soil pre-treatments, natural regeneration, nurse crops and seed mixes (Wells *et al.* 1986; Wells *et al.* 1989; Pywell *et al.* 2002). There are, however, additional problems associated with reinstated soils and studies specific to these, particularly on capped landfill, are sparse. Landfill, usually comprises a clay cap (Butt *et al.* 2004) which not only creates harsh growing conditions and poor soil quality (Athy *et al.* 2006) but also results in an impoverished soil fauna. For example, Scullion's research (1994) at Bryngwyn in Wales found reinstated soils to be deficient in nutrients and prone to water-logging even after light rainfall and prone to cracking in dryer weather. Martin *et al.* (1998) also recorded an earthworm mortality of >90% at this site. Existing literature concentrates on soil amendments and therefore the manipulation of soil fertility and structure, however the conclusions are conflicting. For example, Watson and Hack (2000) recommend that soil substitutes should not be used on their own but mixed well with true soil. By contrast, in a study on the effects of amendments used in landfill restoration, Gregory and Vickers (2003) concluded that organic amendments might improve the structure of the clay soil-forming material prior to vegetation establishment. As an important challenge facing many managers of landfill sites is the need to establish a green cover to stabilise soils, many sites have topsoil added to facilitate rapid plant growth. However Simmons (1999) suggests that topsoil should not be used on areas where wildflower seed is to be sown as the fertility of the topsoil can encourage aggressive competitive species to grow at the expense of the more desirable forb species, and the diversity can be lost. One solution may be to establish a green cover across the whole site and then create more diverse communities on part of the site.

This study was therefore established with the aim of identifying the optimum substrate treatment for creating a species-rich hay meadow community on clay capped landfill that has been stabilised with green cover and investigating whether the accepted principals of habitat creation on agricultural land are transferable to landfill. A grassland seed mix comprising key species from an MG5 *Cynosurus–Centaurea* community, as categorised by the British National Vegetation Classification (Rodwell 1992), was sown on three substrate treatments: bare clay, a mixture of topsoil and ameliorant in equal proportions and naturally colonised vegetation. The objectives were: to determine the effect of manipulation of soil and vegetation on the development of the target plant community, (2) to identify whether these treatments had an effect on edaphic factors (physical and chemical properties, earthworm populations); and (3) to establish which, if any, of these edaphic factors are underlying factors in determining the vascular plant community.

3.3 Methods

3.3.1 The Site

This study was conducted on cell 18 (see Section 2.1) at Dimmer Waste Disposal Site (lat 51º04’38.10’’ N; long 2º32’59.93’’ W). The site of this experiment lies at 38 metres above sea level on a south-east facing 4% slope. The site was capped with a 1m depth of blue-lias clay when it was decommissioned in 1998 and a mixture of 50% original topsoil from the site and 50% green compost was graded across the cap to produce a substrate depth of 15-20 cm. The topsoil, which was stored on adjacent farmland, was a fine textured silty clay soil. The compost was produced by the waste disposal site’s recycling initiative whereby domestic green waste is chipped and processed at temperatures exceeding 65ºC. Coarse grassland vegetation had naturally colonised the site between capping and this experiment being initiated in 2001. Typical species included *Elytrigia repens* (Common couch), *Holcus lanatus* (Yorkshire-fog), *Lolium perenne* (Perennial rye-grass) and *Phleum pratense* (Timothy).

3.3.2 Experimental Design

A randomised block design was adopted to account for the 4% slope of the study site. The design incorporated six blocks of 10 by 10 m plots resulting in 6 replicates of the following treatments: (1) topsoil removed (TR): the reinstated topsoil was removed using a mechanical digger set to remove a 15 cm depth of soil, exposing the bare blue-lias clay cap; (2) vegetation removed (VR): the vegetation that had naturally colonised the topsoil since it was reinstated was removed through the use of herbicide. The dead vegetation was then raked from the plots; (3) control (C): no pre-treatment was applied to these plots which therefore comprised the clay cap, reinstated topsoil and naturally colonised vegetation. There were six replicates per treatment, separated with a 1.5 m buffer zone around their perimeter. A commercial hay meadow seed mix (supplied by Emorsgate, King's Lynn) (Table 4) was sown at a rate of 1.8 g per m² on each plot in June 2001. All vegetation on the study site was mown on an annual basis in August and all cuttings removed from the site.

The frequency of occurrence of each plant species per plot was sampled annually in the third week of June between 2003 and 2007 using a 1 m² quadrat, sub-divided into 10 cm squares. The presence of each rooted species found in each sub-square was recorded and the sum of occurrences used to provide frequency scores. These were used to calculate relative abundance by dividing frequency of occurrence of the species by the sum frequency of occurrence of all species. Relative abundances were used rather than absolute abundances to enable plant community composition to be compared independently to differences in amount of bare ground. Two random quadrats were taken in each 100 m² plot and their locations determined using randomly generated numbers, avoiding a 1 m buffer zone around the margins.

Soils were sampled from each plot in June 2004 and 2006 using a core auger with a 3cm diameter to a depth of 15 cm. Twenty samples were taken from each plot following a 'W'-shaped pattern. Soil from each plot was subsequently bulked and frozen at -18°C immediately after collection prior to analysis. Soils were oven dried at 60ºC, during which soil water content was calculated, and then gently ground using an electric pestle and mortar and passed through a 2 mm sieve. Soil was analysed for organic matter content, pH and phosphate using standard methodologies according to Rowell (1994) and for nitrate using vanadium (II) according to Miranda *et al.* (2001) and Doane and Howarth (2003).

Earthworm species and abundance were sampled in 2004 and 2006 using the hot mustard extraction method according to Lawrence and Bowers (2002).Two 0.25 m quadrats were taken in each 100 m² plot. The locations of these were determined using randomly generated numbers.

3.3.3 Data Analysis

Mann-Whitney U-Tests were used where two independent variables were being compared. Kruskal-Wallis was used in conjunction with Nemenyi's (Wheater and Cook 2000) as a post-hoc test where three or more independent variables were involved. Pearson's bivariate and partial correlation analyses were conducted on selected variables. Analysis of similarity comparing overall plant and earthworm communities between treatments was performed using ANOSIM using PRIMER version 6 (Clarke and Gorley 2006), set to the recommended 999 permutations. All univariate analyses were performed using SPSS version 15.

Table 4: List of sown species

Sown species	**% weight**
Achillea millefolium (Yarrow)	0.5
Agrostis capillaris (Common bent)	10.0
Alopecurus pratensis (Meadow foxtail)	2.0
Anthoxanthum odoratum (Sweet vernal-grass)	2.0
Briza media (Common quaking-grass)	1.0
Centaurea nigra (Black knapweed)	2.0
Cynosurus cristatus (Crested dog's-tail)	35.0
Festuca rubra (Red fescue)	29.0
Galium verum (Lady's bedstraw)	1.5
Hordeum secalinum (Meadow barley)	1.0
Leucanthemum vulgare (Oxeye daisy)	1.5
Lotus corniculatus (Common bird's-foot trefoil)	1.0
Lychnis flos-cuculi (Ragged robin)	0.8
Plantago lanceolata (Ribwort plantain)	0.8
Primula veris (Cowslip)	1.5
Prunella vulgaris (Self-heal)	1.8
Ranunculus acris (Meadow buttercup)	3.0
Rhinanthus minor (Yellow rattle)	1.0
Rumex acetosa (Common sorrel)	1.5
Silaum silaus (Pepper-saxifrage)	1.0
Stachys officinalis (Betony)	1.0
Vicia cracca (Tufted vetch)	0.5

3.4 Results

3.4.1 Effect of Substrate on the Development of Vegetation

3.4.1.1 Overall Plant Community

Table 5 lists the relative abundance (% frequency of occurrence) of the 10 most frequent species within the grassland sward in each year. *A. millefolium, G. verum* and *L. vulgare* are shown to be the most prominent of the seeded species. Both *A. millefolium* and *L. vulgare* gradually decreased in relative abundance over time whilst relative abundance of *G. verum* was seen to steadily increase. Competitive species such as *E. repens* and *H. lanatus* that featured strongly within the community that had developed naturally on the clay cap prior to this experiment being established were also prominent within the developing plant community. A pioneer species associated with disturbed land, *Picris echioides* (Bristly ox-tongue), was abundant within the plant community until 2005 when a decline in its percentage frequency was observed. *Trifolium repens* (White clover) was also abundant throughout the study period.

Table 5: Mean relative abundance (% frequency of occurrence) of the 10 most frequent species (*n* = 6) within the plant community. Where TR = topsoil removed; VR = vegetation removed; C = control.

Species	Treatment	2003	2004	2005	2006	2007
Achillea millefolium (Yarrow)	TR	15.0	7.7	9.7	4.3	1.6
	VR	15.7	17.6	8.6	7.6	1.1
	C	4.6	6.2	6.5	5.0	2.4
Elytrigia repens (Common couch)	TR	0.0	0.1	0.0	0.0	0.2
	VR	3.0	5.3	0.1	14.1	54.5
	C	6.7	19.0	15.2	25.3	54.8
Galium verum (Lady's bedstraw)	TR	5.7	7.9	7.2	9.8	10.4
	VR	4.7	5.9	7.7	11.1	4.9
	C	0.6	1.5	2.0	1.5	0.0
Holcus lanatus (Yorkshire-fog)	TR	0.5	1.6	4.1	6.6	5.3
	VR	1.0	3.5	11.2	9.9	11.9
	C	1.0	3.5	11.2	9.9	11.9

Species	Treatment	2003	2004	2005	2006	2007
Leucanthemum vulgare (Ox-eye daisy)	TR	17.0	13.1	10.7	2.1	2.4
	VR	20.3	23.6	8.8	1.9	0.1
	C	19.6	15.2	16.3	8.6	1.3
Lolium perenne (Perennial rye-grass)	TR	13.8	17.2	10.3	20.6	14.0
	VR	3.7	3.4	4.14	6.6	5.8
	C	6.1	5.6	0.0	2.7	0.0
Picris echioides (Bristly ox-tongue)	TR	1.1	6.0	1.4	0.4	0.6
	VR	2.5	6.3	4.5	2.0	0.3
	C	4.9	4.5	7.4	3.5	1.1
Poa trivialis (Rough meadow-grass)	TR	0.0	3.2	1.9	0.6	0.0
	VR	0.1	2.0	6.8	4.7	0.0
	C	0.2	7.6	8.3	8.7	0.0
Ranunculus repens (Creeping buttercup)	TR	8.8	9.6	1.9	3.9	0.7
	VR	1.7	0.7	0.7	1.8	0.1
	C	9.2	3.1	1.9	0.9	0.0
Trifolium repens (White clover)	TR	10.6	10.7	15.2	17.3	16.8
	VR	3.8	3.2	9.3	6.3	3.2
	C	0.7	2.0	0.0	0.0	0.0

Whilst the three treatments support communities containing similar species, the analysis of similarity (Table 6) shows that the overall plant communities in all three treatments are significantly different in their composition between all treatments in 2003 and 2004. By 2006 the difference between plant communities in VR and C were no longer significantly different. The vegetation on the TR plots, however, continued to develop into a distinct community, significantly different to that on both VR and C plots.

Table 6: Results of ANOSIM analysis of similarities test result (*r)* and *P* values comparing overall plant community composition within substrate treatments. Where TR = topsoil removed; VR = vegetation removed; C= control.

	2003		**2004**		**2005**		**2006**		**2007**	
	r	***P***	***r***	***P***	***r***	***P***	***r***	***P***	***r***	***P***
TR & VR	0.252	*0.037*	0.522	*0.006*	0.243	*0.002*	0.363	*0.004*	0.944	*0.002*
TR & C	0.672	*0.002*	0.609	*0.002*	0.678	*0.002*	0.654	*0.002*	0.893	*0.002*
VR & C	0.433	*0.002*	0.219	*0.048*	0.42	*0.002*	0.08	0.208	0.219	0.071

3.4.1.2 Target Plant Community

The success of the target community was measured using two parameters, the number of seeded species (Figure 4) and the relative abundance of seeded species (Figure 5) within the plots. In 2003 the greatest success was seen in the VR treatment, and this difference was significant between the VR and C plots (H=12.08; df = 5; $P=0.002$). By 2004 the TR treatment showed the greatest success with significant differences in the mean number of seeded species between TR and C in both 2004 ($H=7.09$; df = 5; $P=0.029$) and 2005 (H=11.36; df = 5; $P=0.003$). In 2006 a reversal in this pattern was seen in TR plots when a decrease in the mean number of seeded species was seen in the TR plots, although this was not significant. By 2007 the greatest numbers of seeded species were again found in TR, and differences were significant compared with C plots (H=9.21; df = 5; $P=0.01$).

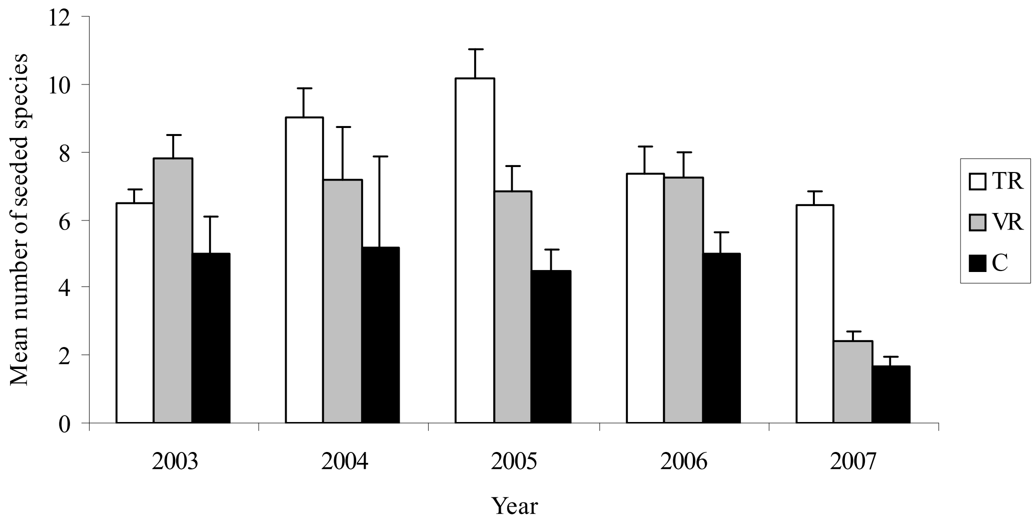


Figure 4: Means and standard errors of the number of seeded species (n = 6) recorded within each treatment between 2003 and 2007. Where TR = topsoil removed; VR = vegetation removed; C = control.

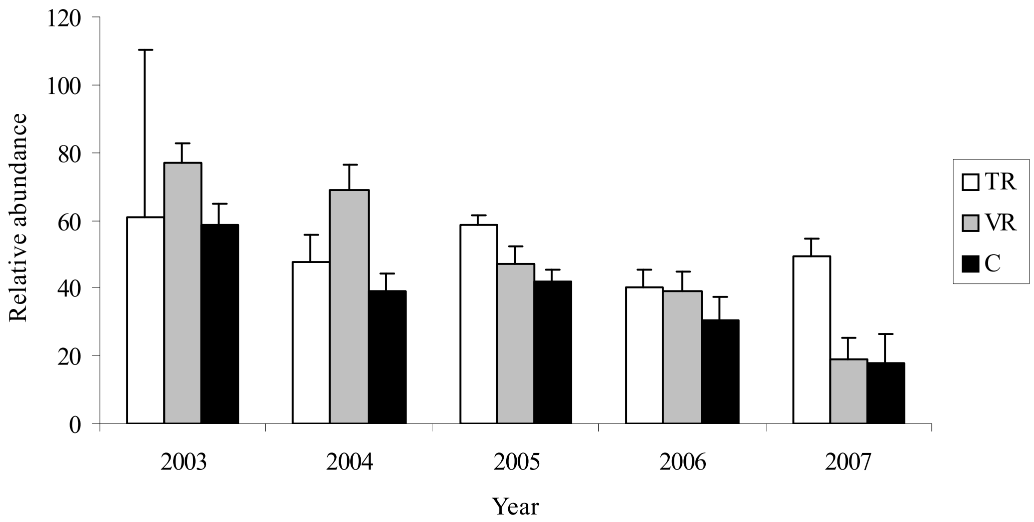
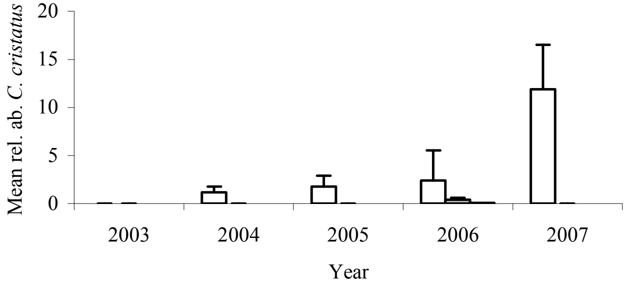


Figure 5: Means and standard errors of relative abundance (% frequency of occurrence) of seeded species (n = 6) within each treatment between 2003 and 2007. Where TR = topsoil removed; VR = vegetation removed; C = Control.

A similar trend to that observed in the number of seeded species was also seen in the response of individual seeded species to the substrate treatments (Figure 6 and Table 7). In 2003, whilst the frequency of seeded species was generally evenly distributed between the treatments, significant differences were detected for *Galium verum* and *L. corniculatus* which were both significantly more abundant in TR and VR plots than in C plots. In 2004 and 2005 *G. verum* and *L. corniculatus* responded similarly in TR and VR treatments, increasing in abundance but showing little increase in frequency in the C plots. In 2006 a decline of these species was noted in all treatments with the greatest decline being observed in VR and C plots. In 2007 the greatest recovery of these species was also noted in TR.

A difference in the frequency of non-seeded competitive species between treatments was also observed. *E. repens* and *H. lanatus* both occurred at higher frequencies in the VR and C plots. *E. repens* was significantly more abundant in C plots than TR in 2004 and 2005, after which this difference was found to be significant between TR and both VR and C. *H. lanatus* was only found to be significantly more abundant in C plots than TR in 2003. *T. repens* was significantly more abundant in TR plots than C throughout the study period, and than in VR plots in 2004 and 2007.



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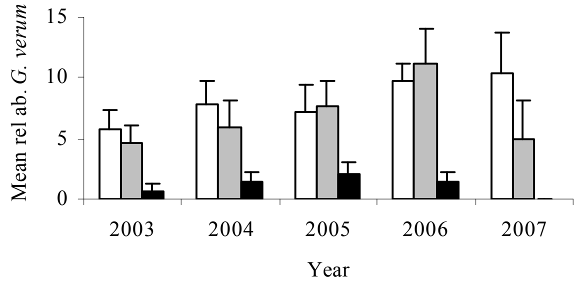

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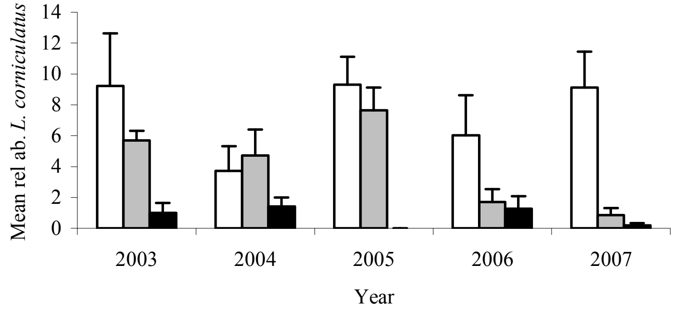

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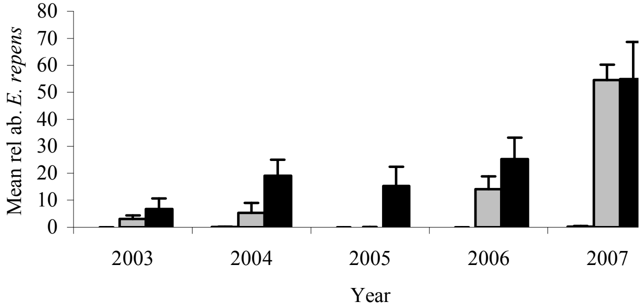

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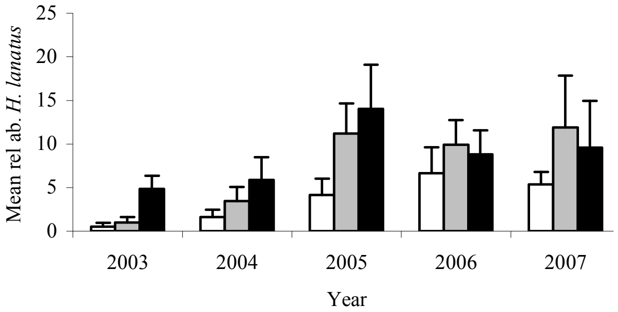

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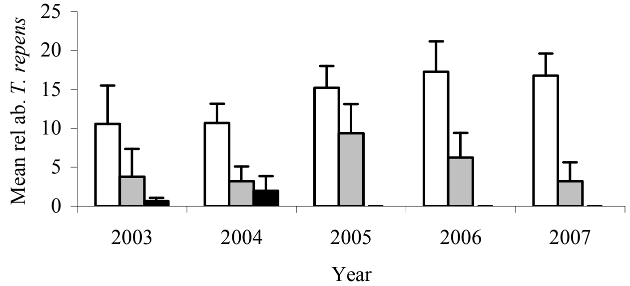

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Figure 6: Means and standard errors of relative abundance (% frequency of occurrence) of selected seeded (n = 6) and non-seeded species (n = 6) recorded within the treatments from 2003 to 2007: (a) *Cynosurus cristatus;* (b) *Galium verum;* (c) *Lotus corniculatus*; (d) *Elytrigia repens;* (e) *Holcus lanatus*; (f) *Trifolium repens.*

Where: ☐ Topsoil Removed ▩ Vegetation Removed ■ Control

Table 7: Kruskal-Wallis Test results (*H*) and *P* values comparing the mean relative abundance (see Figure 6) of selected seeded and non-seeded species between treatments. Where differences are significant, Nemenyi's Test indicates which pairs of treatments show significant differences. TR = topsoil removed; VR = vegetation removed; C = control.

	Species	***H***	**Df**	***P***	**Nemenyi's**
2003	*C. cristatus*	0.000	5	1.000	
	G. verum	8.634	5	*0.013*	TR&C/VR&C
	L. corniculatus	9.892	5	*0.007*	TR&C/VR&C
	E. repens	5.311	5	0.070	
	H. lanatus	6.054	5	*0.048*	TR&C
	T. repens	5.780	5	0.056	TR&C
2004	*C. cristatus*	6.733	5	*0.034*	TR&VR/TR&C
	G. verum	9.313	5	*0.009*	TR&C
	L. corniculatus	1.811	5	0.404	
	E. repens	7.341	5	*0.025*	TR&C
	H. lanatus	2.023	5	0.364	
	T. repens	8.785	5	*0.012*	TR&VR/TR&C
2005	*C. cristatus*	6.733	5	*0.035*	TR&VR/TR&C
	G. verum	4.765	5	0.092	
	L. corniculatus	11.843	5	*0.003*	TR&C/VR&C
	E. repens	7.662	5	*0.022*	TR&C
	H. lanatus	2.549	5	0.280	
	T. repens	11.259	5	*0.004*	TR&C
2006	*C .cristatus*	4.555	5	0.103	
	G. verum	9.003	5	*0.011*	TR&C/VR&C
	L. corniculatus	2.942	5	0.230	
	E. repens	9.364	5	*0.009*	TR&VR/TR&C
	H. lanatus	0.852	5	0.653	
	T. repens	11.922	5	*0.003*	TR&C
2007	*C. cristatus*	12.645	5	*0.002*	TR&VR/TR&C
	G. verum	10.194	5	*0.006*	TR&C / VR&C
	L. corniculatus	11.823	5	*0.003*	TR&C
	E. repens	11.620	5	*0.003*	TR&VR/TR&C
	H. lanatus	0.375	5	0.829	
	T. repens	12.306	5	*0.002*	TR&C

3.4.2 Effect of Substrate on Soil Attributes

3.4.2.1 Soil Physical and Chemical Properties

Significant differences were found between treatments in pH, soil water content and organic matter in both 2004 and 2006 and in phosphate in 2006 only (Table 7). These differences were all found to lie between the C plots and TR plots. The TR plots were more alkaline whilst the C plots had significantly higher levels of soil water, organic matter and phosphate. No differences were found in nitrate levels between treatments.

Table 8: Means, standard errors and results of Kruskal Wallis Test (*H*) and *P* values comparing soil physical (*n* = 6) and chemical characteristics (*n* = 6) recorded between treatments.

Soil characteristic	**Year**	**Mean (Standard error)**			***H* value**	**Df**	***P* value**
		TR	**VR**	**C**			
pH	2004	7.875 (0.038)	7.667 (0.044)	7.618 (0.038)	17.80	5	0.003
	2006	7.795 (0.067)	7.540 (0.050)	7.613 (0.097)	13.73	5	0.017
Water content (%)	2004	26.170 (1.252)	32.893 (0.283)	35.462 (0.569)	28.60	5	<0.001
	2006	26.645 (1.550)	34.447 (0.899)	36.973 (0.683)	14.57	5	0.012
Organic matter (%)	2004	7.104 (0.852)	12.127 (1.475)	16.042 (15.639)	25.42	5	<0.001
	2006	9.045 (0.877)	17.086 (0.754)	18.260 (0.504)	22.98	5	<0.001
Phosphate (Mg / Kg^{-1})	2004	3.646 (1.617)	2.813 (0.213)	12.917 (9.452)	6.41	5	0.268
	2006	0.000 (0.000)	3.958 (1.308)	3.750 (0.853)	21.75	5	<0.001
Nitrate (Mg / Kg^{-1})	2004	12.167 (1.563)	10.417 (1.136)	8.750 (1.389)	2.87	5	0.721
	2006	11.750 (1.553)	13.000 (2.412)	11.667 (1.476)	1.87	5	0.868

3.4.2.2 Earthworm Communities

Some differences were found in the number of earthworms recorded between the three treatments (Figure 7), although differences were not found to be significant. Similarly, ANOSIM analysis found no significant difference in earthworm community composition (Table 9).

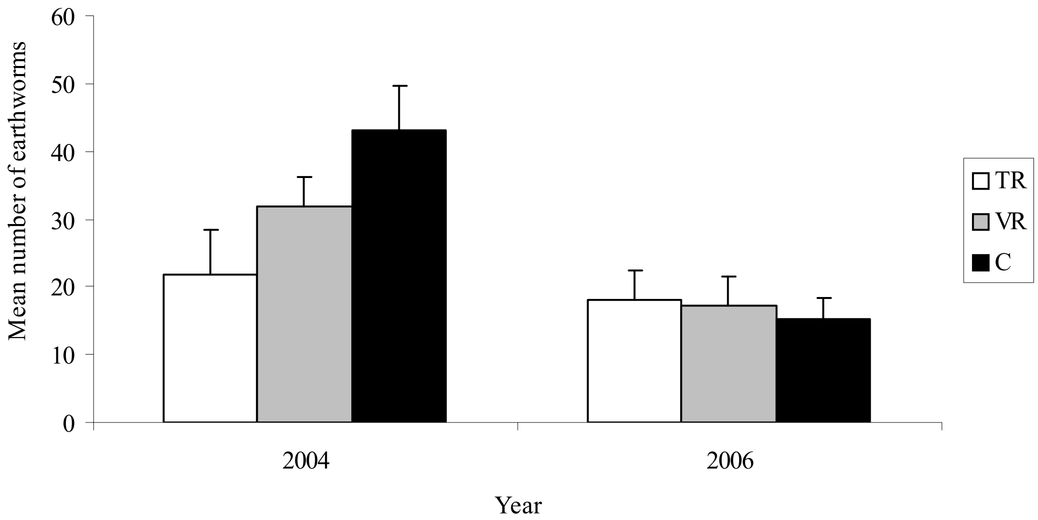

Figure 7: Means and standard errors of the number of earthworms (n = 6) recorded within each treatment in 2004 and 2006. Where TR = topsoil removed; VR = vegetation removed; C = control. 2004: H=7.48; df=5; P=0.187; 2006: H=4.19; df=5; P=0.523.

Table 9: ANOSIM similarity test (r) results and P values comparing the extent to which earthworm communities differ between treatments. Where TR = topsoil removed; VR = vegetation removed; C = control.

		TR& VR	**TR& C**	**VR& C**
2004	*r* statistic	-0.046	0.061	-0.17
	P	0.719	0.232	0.968
2006	*r* statistic	-0.033	-0.109	0.039
	P	0.595	0.883	0.574

3.4.3 Effect of soil attributes on the target plant community

Pearson's bivariate correlation analysis (Table 10) reveals that the only factor showing a significant negative correlation with the mean number and relative abundance of seeded species is the abundance of *E. repens*. The analysis also shows significant positive correlations between both high organic matter and high soil water content with the mean frequency and relative abundance of *E. repens* (Table 11). This implies that these edaphic features may have an indirect effect on the mean number of seeded species via their influence on the growth of *E. repens* even though their direct individual contibutions are not significant. Partial correlation analysis of the relationship between organic matter and water content with *E. repens* found no significant effect when either variable was individually controlled for (r = 0.011; df = 17; P = 0.966 for control of organic matter; r= 0.267; df = 17; P = 0.300 for control of water content). This indicates that these two variables are entirely interdependent and have a joint effect.

Table 10: Pearson's test result (r) and P values for correlation analysis between the mean number (n = 18) and relative abundance of seeded species (n = 18) and variables for which significant differences were detected between treatments (n = 18).

Mean number of seeded species			**Relative abundance of seeded species**		
Variable	***r***	***P***	**Variable**	***r***	***P***
pH	-0.045	0.859	pH	-0.22	0.931
Water content	-0.286	0.249	Water content	-0.035	0.889
Organic matter	-0.264	0.290	Organic matter	-0.121	0.632
Phosphate	-0.189	0.453	Phosphate	-0.402	0.098
Elytrigia repens	-0.480	0.044	*Elytrigia repens*	-0.677	0.002
Holcus lanatus	0.325	0.188	*Holcus lanatus*	0.050	0.844

Table 11: Results of Pearson's correlation analysis (*r*) and *P* values between i) the mean frequency of occurence (*n* = 18) ii) relative abundance of *E. repens* (*n* = 18) and variables for which significant differences were detected between treatments (*n* = 18).

Mean frequency of *E. repens*			**Relative abundance of *E. repens***		
Variable	***r***	***P***	**Variable**	***r***	***P***
pH	-0.175	0.487	pH	-0.125	0.620
Water content	0.596	0.009	Water content	0.580	0.012
Organic matter	0.533	0.017	Organic matter	0.562	0.015
Phosphate	-0.348	0.157	Phosphate	0.407	0.094
Holcus lanatus	-0.099	0.697	*Holcus lanatus*	-0.239	0.340

3.5 Discussion

3.5.1 Response of the Plant Communities to the Treatments

During the first three years following sowing of the grassland the treatments and the control developed plant communities that differed significantly in their frequency of seeded species. This is likely to reflect the greater suitability of the VR (vegetation removed) plots for seeds from a wide range of species to geminate due to the mesic water and organic matter conditions of the soil (Parker 1995; Gregory and Vickers 2003). Community development was slowest on the TR (topsoil removed) plots, apparently also due to soil physical conditions. Whilst a clay substrate with particles typically less than <2mm in size such as those in the TR plots would normally be able to hold large quantities of water (Watson and Hack 2000), in the case of landfill where the soil is deliberately compacted the soil's aggregate stability becomes broken down leading to a higher bulk density and poor water filtration (Scullion and Mohammed 1991). The retention of soil ameliorant on the VR and C (control) plots promotes soil water which Parker (1995) suggests is key in the germination success of seeded species. This also provides a high quality tilth which has been documented as being essential in the creation of hay meadows of nature conservation value (Gilbert and Anderson 1998; Watson and Hack 2000; Gregory and Vickers 2003).

Although the VR treatment promoted good initial seedling germination and establishment, a distinct shift in the development of the plant communities was seen from 2004 onwards. At this point seeded species continued to germinate and increase in abundance within the TR plots, whilst a decline in their abundance was seen in the C plots, and the number of seeded species within the initially successful VR plots reached a plateau. It is notable that a decrease in the number and abundance of seeded species was seen in 2006 in all treatments which can be attributed to drought conditions during which many of the seeded species died. The negative effect of competitive grass species on floristic diversity is well documented (Rodwell 1992; Department of the Environment 1995; Gilbert and Anderson 1998) and in this study *E. repens*, probably due to its rhizomatous nature (Grime *et al.* 2007), was less affected by the drought, and went on to almost entirely dominate the grassland sward in the VR and C plots.

By contrast, the TR plots showed greater recovery both in terms of the number of seeded species and in the abundance of certain species such as *C. cristatus* and *L. corniculatus*. This may be due to the inhibited growth of competitors such as *E. repens* resulting in significant areas of bare ground on this treatment. Box (2003) highlights the importance of bare ground in providing colonization gaps. Interestingly, although the non-seeded competitive species, *T. repens*, was most abundant in the TR plots, the effect documented by Warren (2000) of *T. repens* reducing forb abundance and diversity by promoting the growth of competitive grasses through its nitrogen fixation abilities, was not detected in the present study. Indeed, the present study found that species diversity increased with increased abundance of *T. repens*. It may be that the significantly lower levels of phosphate, soil water and organic matter within the TR plots are limiting factors suppressing the tall and vigorous growth of grasses, irrespective of levels of available nitrogen.

3.5.2 Edaphic Factors

The chemical composition of the substrate would be expected to play a key role in the establishment of the target plant community as data from numerous studies suggest that floristic diversity of low-input grassland systems are closely linked with phosphate availability (Lloyd and Pigott 1967; Rorison 1971; Gough and Marrs 1990a; Janssens *et al.* 1998). Whilst a significant difference between phosphate levels was recorded between TR and C plots, the only significant (negative) correlation with the number of seeded species was found with *E. repens*. This suggests that the decrease in the success of seeded species in the VR and C plots from 2004 onwards may be a direct effect of competition from this species of grass. The increased competitive strength of *E. repens* may be associated with the higher levels of organic matter, soil water and phosphate within the soil of these plots. By contrast, the levels of phosphate within the TR plots fall within the optimum range of phosphorus of 0.5-1.0 mg P 100g-1 as identified by Gough and Marrs (1990) and Snow *et al.* (1997) in their research into grassland creation on arable land. Nitrate levels were not found to be significantly different between the treatments, although nitrogen additions to formerly species-rich grassland are associated with the decline in diversity (Thurston

1969; Harper 1971; Van der Bergh 1979; Berendse *et al.* 1992). Consequently the research presented here supports the findings of Gough and Marrs (1990a) that nitrogen was not found to be a key factor determining the germination and growth of seeded species.

It is interesting that significant differences in the soils were only detected between the TR and C treatments as essentially the same soil treatment was applied to VR and C plots. Whilst VRs' soil characteristics showed greater affinity in their levels to C plots they essentially formed an intermediate between TR and C. One explanation for this is that the removal of the vegetation prior to sowing the seed mix on the VR plots had enough of an effect on the soil characteristics to cause these differences. In their study on ex-arable land McCrea *et al.* found that a cropping treatment significantly reduced the pH levels of the soil in just one year. However, the removal of vegetation would not have been expected to have this impact on soil characteristics such as phosphate, soil water content and organic matter. Gough and Marrs (1990a,b) found that it took between five and fifteen years of hay cropping to make significant differences in phosphate levels. Whilst the management of the VR and C plots did involve an annual hay cut they were subject to the same treatment and so it is unlikely that the single additional removal of vegetation on the VR plots would have caused these differences. It is possible that this is due to a type 2 error and that a greater intensity of soils analysis would have in fact confirmed the TR soils to differ significantly to the soil of VR treatment as well as the Controls.

This study recorded a total of five species of earthworm, with species representative of the three major groups based on burrowing and feeding characteristics (Bouché 1977). In accordance with Scullion's findings (1994) the small shallow-working species, *Allolobophora chloritica* and *Lumbricus rubellus* were most numerous in all treatment plots. Anecic species such as *Lumbricus terrestris* were considerably less numerous with only one or two individuals being recorded in each treatment. Whilst a greater number of earthworms were recorded in the VR and C plots than in TR plots these differences were not significantly different (although numbers were lower overall in 2006, probably due to the drought conditions that year). The representation of the three functional groups of earthworms and relatively even distribution of earthworms across all three treatments is interesting as the processes of soil stripping, storage and reinstatement contribute to a large reduction in earthworm populations (O'Flanagan *et al.* 1963; Scullion *et al.* 1988). Research has long documented both the benefits of earthworms to soil development (Teotia *et al.* 1950; Ehlers 1975; Stewart 1981) and the benefits of the development of plant communities to earthworms (Mato *et al.* 1989; Doube and Brown 1998; Pizl 1999). The presence of earthworms is likely to be of benefit in improving soil conditions, which was expected to be of particular relevance to the physical soil conditions within the TR treatment. However, this study did not find any differences in earthworm abundance between treatments and therefore there is no evidence that they are factors in determining plant community in this case.

3.6 Conclusions

This study found that, despite the problems with depleted soil conditions associated with clay caps, sowing seed onto bare clay was the most successful method in the long-term for establishing a hay meadow community. The retention of topsoil and ameliorant initially resulted in a greater success in seeded species, probably through creating more suitable soil conditions for germination. However, three years after sowing the abundance of sown species significantly decreased as a direct result of competition from *E. repens*. The success of *E.repens* on treatments where topsoil and ameliorant were added directly correlated to the higher levels of organic matter and soil water content associated with this substrate mix. The success of *E. repens* also appears to be linked to the higher levels of phosphate in these treatments, thus implying that this key factor in hay meadow creation on ex-arable is also key to successful establishment of this habitat on landfill despite the additional problems associated with its reinstated soil.

3.7 Implications for Practice

- Forb-rich grassland communities can be created on clay capped landfill even after initial development of a green cover if the topsoil is removed.
- This means that patches of species-rich habitats could be created on sites with an established green cover thus increasing the nature conservation value of the site without compromising on aesthetic landscape issues.
- The retention of ameliorant and/or topsoil promotes swift colonization and germination of seeded species, but also provides ideal conditions for competitive grass species which soon out-compete the desired forbs.
- Hay meadow seed mix sown directly onto bare clay is slow to develop but continues to support the desired community in the long-term.
- The principle of low phosphate levels being key to hay meadow creation appears to transfer to landfill and problems relating to depleted soil structure specific to reinstated soils.

4.0 INOCULATION OF EARTHWORMS AS A MECHANISM TO ACCELERATE THE ESTABLISHMENT OF A HAY MEADOW COMMUNITY ON A CLAY CAPPED LANDFILL

4.1 Summary

This chapter investigates whether the rate of establishment of a hay meadow community can be accelerated through the inoculation of earthworms on a clay capped landfill in Somerset, UK. The objectives are (1) to compare the survival of inoculated earthworms on unmodified clay cap with that where compost had been added on top of the clay cap; (2) to compare the rate of natural colonisation of earthworms on the unmodified clay cap with that where compost has been added; and (3) to determine whether there is a relationship between abundance of earthworms and the extent of the development of the target plant community. In 2006 a commercial hay meadow seed mix was sown over plots subjected to four treatments: clay cap with and without inoculated earthworms, and compost with and without inoculated earthworms. Natural colonisation of earthworms was found to be successful whereas earthworm inoculation failed. No significant differences were found between clay cap and compost in success of natural colonisation by earthworms. This indicates that earthworm populations can develop on the hostile environment created by the clay cap without ameliorant, thus allowing for retention of soils of low nutrient status which are conducive to hay meadow creation. However, earthworms were not found to have an accelerating affect on the development of the overall plant community.

4.2 Introduction

Landfill is currently the most prevalent method of waste management in many countries, for example, in 2006 60% of municipal waste went to landfill both in the US (Environmental Protection Agency 2008) and in the UK (Department for the Environment, Food and Rural Affairs 2008). Decommissioned landfill sites therefore cover significant areas of land, often in or near urban centres throughout the world (Handel *et al.* 1997). It is essential for the continued acceptability of the waste management industry that, following restoration, landfill developments are able to demonstrate a beneficial after-use (Watson and Hack 2000). Landfill sites have therefore traditionally been restored to grassland, either for agriculture or public amenity space (Simmons 1999). However, the legislative framework that has developed in the UK over the last 20 years increasingly promotes ecological restoration (Townsend *et al.* 2004) which has led to a growing interest in creating or restoring habitats for nature conservation on capped landfill. In practice, however, insufficient social acceptance of landscape changes during restoration is a limiting factor (Pfadenhauer 2001) resulting in many managers of landfill sites adding topsoil to the clay cap to facilitate rapid plant growth. This practice is in conflict with goals to create habitats for nature conservation as the fertility of the topsoil encourages competitive species to grow at the expense of more desirable forb species (Simmons 1999; Carrington and Diaz 2009). It would therefore be of benefit to find alternative methods to accelerate the development of habitats.

Many earthworm species actively change and ameliorate their soil environment and are regarded as ecosystem engineers (Lavelle *et al*. 1997). Several studies have shown that earthworms improve the structure of the soil through increasing pore space, particularly macropore volume (Teotia *et al.* 1950; Ehlers 1975; Stewart 1981), thus increasing soil aeration and drainage. Furthermore, earthworms play an important role in the recycling and distribution of nutrients within the soil (Sims and Gerard 1999). However, studies have shown that earthworm numbers are low in reinstated soils (Scullion 1994; Martin *et al.* 1998). Natural re-colonisation and recovery of earthworm populations can be slow (Van Rhee 1969; Stockdill 1982; Hoogerkamp *et al*. 1983) taking up to ten years or longer for a population typical of similarly managed, undisturbed land to develop (Scullion 1994). Rushton (1986) identifies the adverse physical conditions prevailing in reinstated soils as a limiting factor in natural recovery of earthworm populations. Measures to accelerate earthworm colonisation of reinstated land are now recognised as being an important factor in their rehabilitation (Scullion 1994; Watson and Hack 2000).

Unfortunately, whilst anecdotal accounts of earthworm inoculation projects are plentiful, the results of these are little documented and, where documented, post-inoculation monitoring rarely exceeds six months (Butt 1999). Where trials on reinstated soils have been documented the outcomes are mixed. For example, whilst Butt *et al.* (1999) successfully inoculated two species of earthworm onto a clay capped landfill site no significant changes were observed in the physical conditions of the soil during the five year period of monitoring. In contrast, within two years following inoculation of earthworms on reinstated soils at Bryngwyn open cast coal mine Scullion (1994) found evidence of increased porosity and water-stable aggregation. These effects were recorded to depths of 15 centimetres compared with little or no change to physical properties in the control areas. Nonetheless, in accordance with numerous studies both on reinstated land (Rushton 1986; Marinissen 1994) and arable (Leroy *et al.* 2008) both Scullion (1994) and Butt *et al.* (1999) found significantly higher earthworm survival where organic amendments had been added. Successful inoculation of earthworms into reclaimed peat at

Conclast resulted in 25% greater yield of grass compared to controls two years following inoculation and 49% in the third year (Curry 1988). These accounts show that it is possible to successfully inoculate earthworms onto the clay cap and the positive effects of earthworms on both the physical characteristics of the soil and on plant productivity indicates that they may be a useful tool in increasing the rapidity of the establishment of forb-rich grassland sward.

This study was established with the aim of investigating whether the rate of establishment of a hay meadow community could be accelerated through the inoculation of earthworms on a clay capped landfill. The objectives were: (1) to compare the survival of inoculated earthworms on the unmodified clay cap with that where compost had been added; (2) to compare the rate of natural colonisation of earthworms on the unmodified clay cap with that where compost has been added; and (3) to determine whether there is a relationship between abundance of earthworms and the extent of the development of the target plant community.

4.3 Methods

4.3.1 Study site

This study was conducted on cell 27 at Dimmer Waste Disposal Site near Castle Cary in Somerset (lat 51º04'38.10'' N; long 2º32'59.93'' W). The site of this experiment lies at 38 metres above sea level on level land. The site was capped with a 1.5m depth of blue-lias clay when it was decommissioned in 2005. No existing vegetation was present at the time this experiment was established in 2006.

4.3.2 Experimental design

A fully randomised factorial design was adopted to compare the development of a hay meadow community on a total of twenty 5 by 5 m plots subjected to four treatments: the unmodified clay cap, with and without inoculated earthworms, and plots where ameliorant was added, with and without inoculated earthworms. A 1 m buffer zone was incorporated around the margins of each 25 m^2 plot. The ameliorant used was compost produced by the waste disposal site's recycling initiative which chips and processes green waste at temperatures exceeding 65ºC. This was graded across ten of the plots at a depth of 10 cm.

Two species of commercially sourced earthworms (supplied by Tommy Topsoil, Sowerby Bridge, West Yorkshire, UK), *Lumbricus terrestris* and *Lumbricus rubellus*, were used to inoculate plots in April 2006. These species of earthworm were chosen due to their availability for purchase commercially from a UK source, *L. terrestris'* intense burrowing activities (Bouché 1977) and the potential benefits this could have on the soil structure (Butt 1999) and for the previous success in inoculating *L. rubellus* (e.g.s Hallows 1993, Robinson *et al.* 1992). Earthworms were supplied in batches of 100 and were of mixed age and size, provided in the medium in which they were bred which contained eggs and cocoons. These were inoculated in the centres of five plots of unmodified clay cap and five where compost had been added by digging a shallow hole into which the 'batch' of earthworms was directly broadcast and then covered with loose soil. The earthworm inoculation took place in April to optimise chances of survival as the greatest success has been observed in previous studies with spring inoculations (e.g.s Butt *et al.* 2004, Butt *et al.* 2007).

A commercial hay meadow seed mix (supplied by Emorsgate, King's Lynn, Norfolk, UK) comprising key species from an MG5 *Cynosurus–Centaurea* community (Table 12), as categorised by the British National Vegetation Classification (Rodwell 1992) was sown at a rate of 1.8 g per m² on all plots in April 2006. All vegetation on the study site was mown annually in August and all cuttings removed from the site.

The presence and abundance of earthworms was measured using the 'hot mustard' extraction method according to Lawrence and Bowers (2002). This was first done prior to inoculation in March 2006 and the absence of earthworms was confirmed. Subsequent earthworm sampling was undertaken annually in October until 2008. Two 0.25 m² quadrat was taken from random positions in each 25 m² plot.

The frequency of occurrence of each plant species was sampled annually in the fourth week of June between 2006 and 2008 using a 0.25 m² quadrat, sub-divided into 5 cm squares. The presence of each rooted species found in each square was recorded and the sum of occurrences used to provide frequency scores. Two random quadrats were taken in each 25 m² plot. The percentage volume of above ground vegetation was calculated using the sum of ten percentage cover estimates taken at 5 cm height intervals within each quadrat

Soils were sampled from each plot in June 2008 using a core auger with a 3 cm diameter to a depth of 15 cm. Twenty samples were taken from each plot following a 'W'-shaped pattern according to Rowell (1994). Soil from each plot was subsequently bulked and frozen at -18°C immediately after collection prior to analysis. Soils were oven dried at 60ºC, during which soil water content was calculated, and then gently ground using an electric pestle and mortar and passed through a 2 mm sieve. Soil was analysed for organic matter content, pH and phosphate using standard methodologies according to Rowell (1994) and for nitrate using vanadium (II) according to Miranda *et al.* (2001) and Doane and Howarth (2003). Bulk density was calculated by dividing the weight of oven dried peds of soil by their volume which was determined through submerging paraffin-covered peds in a known volume of water (after Johnston 1945).

4.3.3 Data analysis

Kruskal-Wallis was used to compare earthworm abundance between the four treatments and Mann-Whitney U-Tests were used to compare plant abundance and soil characteristics between plots on the unmodified clay cap and those where compost was added. Spearman Rank correlation analysis was used to assess whether earthworm abundance had an accelerating effect on the development of the target plant community by 3 years post-inoculation. All analyses were performed using SPSS version 15.

Table 12: List of sown species

Type of seed	***% weight***	***Type of seed***	***% weight***
Achillea millefolium	0.5	*Lychnis flos-cuculi*	0.4
Agrostis capillaris	5.0	*Malva moschata*	1.0
Centaurea nigra	1.0	*Plantago media*	0.6
Cynosurus cristatus	35.0	*Primula veris*	1.0
Daucus carota	0.5	*Prunella vulgaris*	1.0
Festuca rubra	40.0	*Ranunculus acris*	2.0
Filipendula ulmaria	0.6	*Rhinanthus minor*	1.0
Galium mollugo	0.4	*Rumex acetosa*	1.0
Galium verum	1.0	*Sanguisorba minor*	1.5
Knautia arvensis	1.0	*Silene dioica*	1.0
Leontondon hispidus	0.4	*Silene vulgaris*	0.5
Leucanthemum vulgare	0.8	*Stachys officinalis*	0.5
Lotus corniculatus	0.8	*Vicia cracca*	0.5

4.4 Results

4.4.1 Abundance of inoculated and naturally colonised earthworms

A high fatality of earthworms was observed on both the unmodified clay cap and plots where compost was added during the first six months following inoculation. Only one *L. terrestris* was recorded at all, and the median number of *L. rubellus* was consistently 0, with ranges from 4 on plots with added compost to 10 on the unmodified clay cap (Table 13). By the third year after inoculation no *L. terrestris* were recorded but *L. rubellus* and the naturally colonised *Allolobophora chlorotica* were recorded at similar densities across both treatments with no significant differences being detected (H = 2.821, P = 0.420 and H = 2.597, P = 0.458 respectively). The *L. rubellus* were recorded both on plots where earthworms been inoculated and where they had not.

Table 13: Medians and ranges between minimum and maximum numbers of earthworms recorded (n = 5) in years 1 and 3

Treatment effect	***Year***	***Compost - worms***		***Compost + worms***		***Clay cap - worms***		***Clay cap + worms***	
		Median	**Range**	**Median**	**Range**	**Median**	**Range**	**Median**	**Range**
Number of *L. terrestris*	1	0	1	0	0	0	0	0	0
	3	0	0	0	0	0	0	0	0
Number of *L. rubellus*	1	0	4	0	0	0	1	0	10
	3	2	8	1	5	2	3	4	5
Number of *A. chlorotica*	1	0	0	0	0	0	11	0	1
	3	1	6	1	7	0	2	1	4
Total number of earthworms	1	0	4	0	0	0	12	0	10
	3	4	14	2	11	2	5	5	8

4.4.2 Earthworms and the acceleration of the development of the target plant community

The ten most frequent species recorded within the grassland sward three years after sowing are listed in Table 14. Four of these are sown species. No differences were observed in their abundance between treatments. However, higher frequencies of occurrence of the highly competitive non-sown *Agrostis stolonifera* and *Poa trivialis* were found on plots where compost had been added. This is reflected by the greater vegetation volume that was also found in this treatment and the correspondingly lower levels of bare ground. Pioneer species *Picris echioides* and *Polypogon monspeliensis* were more frequent on the unmodified clay cap.

Evidence of earthworms having an accelerating effect on the development of the target plant community was limited. Where compost was added the total number of earthworms correlated positively with the abundance of the sown species *F. rubra* (Table 15). In contrast, a negative correlation was found between the total number of earthworms and the abundance of the sown *C. cristatus.* On the unmodified clay cap a positive correlation was detected between *A. chlorotica* and the sown *L. corniculatus.* No negative correlations were found between earthworms and sown species on this treatment but the abundance of *L. rubellus* was found to correlate positively with the non-sown competitor *A. stolonifera*. Correspondingly, a positive correlation was also found between the total number of *L. rubellus* and volume of vegetation on the unmodified clay cap. In contrast the total number or earthworms and *A. chlorotica* both correlated negatively with the volume of vegetation on plots where compost had been added. This study found no evidence of earthworms influencing the soil physical or chemical soil properties measured in this study.

Table 14: Medians, ranges between minimum and maximum abundance scores and the results of Mann-Whitney U-Test comparing the abundance of the ten most frequent species (n = 10) within the plant community and soil properties (n = 10) between treatments in the 3rd year after sowing

Variable	***Compost***		***No compost***		***Mann Whitney U-test***	
	Median	**Range**	**Median**	**Range**	**U**	**P**
Sown species						
Cynosurus cristatus	0.0	40.0	0.0	1.0	49.5	0.964
Festuca rubra	20.0	100.0	6.0	16.5	28.0	0.096
Lotus corniculatus	0.0	1.0	0.0	7.5	33.0	0.091
Prunella vulgaris	0.0	1.0	0.0	6.5	33.5	0.101
Non-sown species						
Agrostis stolonifera	100.0	35.0	17.5	97.5	5.5	0.001
Deschampsia cespitosa	0.0	15.0	0.0	2.0	38.5	0.214
Lolium perenne	0.0	21.5	0.0	6.0	42.5	0.522
Picris echioides	0.0	0.0	0.75	12.0	15.0	0.002
Poa trivialis	5.0	62.5	0.0	3.0	18.5	0.007
Polypogon monspeliensis	0.0	0.0	4.0	32.5	5.0	<0.001
Total vegetation volume & bare ground						
Vegetation volume	38.9	34.8	5.1	15.4	2.0	<0.001
Bare ground	0.0	0.0	71.5	94.5	0.0	<0.001
Soil properties						
Organic matter	15.8	10.2	8.0	8.9	4.0	0.001
Water content	35.7	11.2	23.2	6.0	0.0	<0.001
Bulk density	0.9	0.4	1.9	0.7	0.0	<0.001
pH	7.3	0.3	7.2	0.4	41.5	0.520
Phosphate	6.2	6.1	2.5	3.4	52	<0.001
Nitrate	2.5	12.7	5.5	18.0	27.5	0.088

Table 15: Results of Spearman Rank correlation test (r) between the number of earthworms and the frequency scores of the ten most frequent species (n = 10) of plant and soil properties (n = 10) in the 3rd year after sowing

Variable	***Compost***						***No compost***					
	A. chlorotica		***L. rubellus***		**Total earthworms**		***A. chlorotica***		***L. rubellus***		**Total earthworms**	
	r	***P***	**r**	***P***	**r**	***P***	**r**	***P***	**r**	***P***	**r**	***P***
Sown species												
Cynosurus cristatus	-0.387	0.269	-0.545	0.103	-0.654	0.04	-0.101	0.780	-0.049	0.892	-0.021	0.954
Festuca rubra	0.511	0.131	0.448	0.195	0.665	0.036	0.070	0.847	-0.059	0.871	0.037	0.919
Lotus corniculatus	0.241	0.502	-0.530	0.115	-0.239	0.505	0.656	0.040	0.339	0.338	0.466	0.174
Prunella vulgaris	0.422	0.225	0.530	0.115	0.539	0.108	0.057	0.876	0.108	0.767	0.118	0.747
Non-sown species												
Agrostis stolonifera	0.152	0.674	-0.489	0.152	-0.331	0.350	0.264	0.462	0.703	0.02	0.633	0.050
Deschampsia cespitosa	-0.062	0.865	0.182	0.615	0.169	0.640	0.000	1.000	0.355	0.315	0.236	0.512
Lolium perenne	0.184	0.610	0.163	0.653	0.106	0.771	0.139	0.701	0.494	0.147	0.444	0.199
Picris echioides	0.000	0.000	0.000	0.000	0.000	0.000	0.315	0.375	0.404	0.246	0.419	0.228
Poa trivialis	-0.248	0.489	-0.044	0.905	-0.272	0.447	-0.425	0.221	-0.473	0.168	-0.530	0.115
Polypogon monspeliensis	0.000	0.000	0.000	0.000	0.000	0.000	-0.337	0.341	-0.359	0.308	-0.324	0.361

Variable	Compost						No compost					
	A. chlorotica		***L. rubellus***		**Total earthworms**		***A. chlorotica***		***L. rubellus***		**Total earthworms**	
	r	***P***	**r**	***P***	**r**	***P***	**r**	***P***	**r**	***P***	**r**	***P***
Total vegetation volume and bare ground												
Vegetation volume	-0.663	0.037	-0.583	0.077	-0.693	0.026	0.399	0.253	0.624	0.054	0.591	0.072
Bare ground	0.000	0.000	0.000	0.000	0.000	0.000	0.051	0.889	-0.223	0.536	-0.250	0.486
Soil properties												
Organic matter	-0.453	0.188	0.037	0.919	-0.156	0.666	-0.336	0.343	-0.247	0.492	-0.277	0.439
Water content	0.378	0.282	0.560	0.092	0.594	0.070	-0.405	0.245	-0.488	0.153	-0.388	0.268
Bulk density	-0.019	0.959	0.468	0.173	0.275	0.442	0.785	0.425	0.216	0.584	0.351	0.370
pH	-0.113	0.755	0.031	0.933	0.138	0.705	-0.213	0.555	-0.384	0.273	-0.333	0.347
Phosphate	-0.271	0.449	-0.074	0.839	-0.163	0.654	-0.247	0.491	-0.198	0.584	-0.295	0.407
Nitrate	0.215	0.552	-0.315	0.376	0.003	0.993	0.118	0.746	0.046	0.899	0.099	0.786

4.5 Discussion

4.5.1 Is the survival of inoculated earthworms affected by substrate?

No difference in earthworm survival was found between the unmodified clay cap and plots where compost had been added, with a high rate of mortality being observed across all plots. Survival of *L. terrestris* was particularly poor with no individuals of this species being recorded by the third year following inoculation. A similar fate has been observed for this species in other inoculation trials. Craven (1995) and Butt *et al.* (1993) found very low survival rates at Hallside colliery and Hillingdon capped landfill respectively. Butt (1999) attributed this to unsuitable conditions at these sites for this deep-burrowing species that requires deep, mature soils with a supply of surface organic matter (Bouché 1977; Edwards and Bohlen 1996). Studies on the natural colonisation of earthworms on reclaimed land have found *L. terrestris* to be restricted to sites of greater age and development (Curry and Cotton 1983; Butt 2008) with deeper, mature soils and a supply of surface organic matter (Butt *et al.* 2004) adding further evidence to the unsuitability of this species to the freshly laid clay cap.

The second inoculated species, *L. rubellus*, survived on both treatments and was the most abundant species recorded in this study. The survival of this species agrees with the only documented directly comparable previous inoculation trial conducted by Hallows (1993) on a clay capped landfill at Stockley Park. Hallows found *L. rubellus,* alongside *Dendrobaena veneta*, to be the most successful of the four species inoculated. Unfortunately, Hallows does not specify the density at which *L. rubellus* was recorded during post-inoculation monitoring although all life stages are documented as being present after two years. Similarly, high survival of *L. rubellus* was reported by Robinson *et al.* (1992) in their inoculation trial within a spruce plantation with high levels of organic matter in Cumbria. In the current study *L. rubellus* ws recorded at extremely low densities ranging from only 4 to 24 m^{-2} by year 3 although unfortunately it is not possible to gauge how this compared to survival in previous studies such as Hallows (1993) or Robinson *et al.*'s (1992) as this level of detail does not appear to have been documented for this species. Two species for which inoculation trials have been better documented are *A. chlorotica* and *A. longa*. In Butt *et al.*'s (1999) trial at Calvert landfill site in Buckinghamshire these species were recorded at densities of up to 35.8 and 22.1 m^{-2} respectively six years following inoculation. Butt *et al.* (2004) recorded respective densities of 108 and 70 m^{-2} at the same site 11 years following inoculation. Presuming population growth continues at a steady rate on the current study's site, the abundance of *L. rubellus* recorded is therefore comparable to inoculated species at Calvert. However, in contrast to the current study, significantly higher numbers of inoculated earthworms were recorded where organic matter had been added at the Calvert site (Butt *et al.* 1999).

Interestingly, the distribution of the surviving inoculated earthworm species in the present study, *L. rubellus*, did not correspond with the sites of inoculation. This species was mainly recorded in plots around the margins of the experiment and appeared as regularly in plots that were not inoculated as those that had been. The rate of dispersal from the points of inoculation to the locations at which they were recorded by far exceeds previously recorded typical rates of between 4 and 6 metres per year (Marinissen and van den Bosch 1992). This may be indicative of 100% mortality of all inoculated earthworms which questions the quality of the commercially sourced inoculums. Butt (1999) advises caution in the use of the commercially sourced earthworms and surmised that poor quality of inoculum may be responsible for the failure of Craven's (1995) inoculation trial in the Hallside Steelworks Project. Practices such as long periods of refrigeration impact on their likelihood of survival and this may have also played a part in the scale of fatality observed in this trial. However, Hallows (1993) has successfully used commercially sourced earthworms in a number of projects including Stockley Park, Worcester Park and Rayleys Field at Sevenoaks. The survival of the commercially sourced earthworms in this study remains uncertain although the presence of *L. rubellus* and absence of *L. terrestris* is indicative of the importance of suitable species selection. The fact that there were no significant differences between the numbers of inoculated earthworms in plots with or without compost shows that earthworms can survive in the freshly laid clay cap with no specific requirement for organic amelioration.

4.5.2 Is the rate of natural colonisation of earthworms affected by substrate?

One species of earthworm that was not inoculated, *A. chlorotica*, was recorded as being co-dominant with *L. rubellus* in this study. Other studies have found *A. chlorotica*, alongside *L. rubellus* and *Apporectodea calignosa*, as the most successful early colonists of restored land largely due to their high reproductive rates and strong powers of dispersal (Brockmann *et al.* 1980; Curry and Cotton 1983; Judd and Mason 1995; Butt *et al.* 1999). However, the lack of preference of these earthworms for either the unmodified clay cap or the plots where compost had been added is of note. Higher densities of naturally colonised earthworms would have been expected where compost had been added due to more favourable conditions as not only are the significantly higher levels of organic matter in this treatment vital to earthworms (Butt *et al.* 2004) but the more humid conditions are also of great importance for respiration (Sims and

Gerard 1999). In a similar study Butt *et al.* (1999) recorded significantly greater numbers of earthworms where dung had been applied and surmised that the general lack of organic matter on restored landfill sites may be a limiting factor in the development of natural populations. The results of the current study suggest that the availability of organic matter may not be of such importance in the colonisation of *A. chlorotica*.

A. chlorotica had colonised the site of the current study within a year following capping and continued to increase in number and distribution over the duration of this experiment. This is in contrast with Butt *et al.* (1999) who failed to locate any natural earthworm colonists after three years in their study on a partially restored landfill site. Similarly, Judd and Mason (1995) did not record any earthworms on the one year old trial site in their experiment. This higher rate of colonisation may be due to the proximity of existing earthworm populations. For instance, there is a series of banks and hedgerows that separate the site of this experiment at Carymoor from surrounding, more established areas of capped landfill. Further study on the rate of earthworm colonisation on clay capped landfill in relation to existing stable populations would be of interest.

4.5.3 Do earthworms accelerate the development of the plant community?

Some evidence linking earthworm abundance and the rate of development of plant species and their abundance was found in the present study although where relationships were detected it is not possible to ascribe causality. The results show positive relationships between earthworms and vegetation volume, the abundance of the sown species *L. corniculatus* and of the non-sown competitor *A. stolonifera* on the unmodified clay cap. Previous studies have recorded similar relationships between vegetation and earthworms in which it is suggested that the development of a mat of grass roots particularly favours earthworms (Davis 1986) through providing protection from predators and higher levels of humidity (Mather and Christensen 1992). Similarly, previous studies have found earthworm abundance to have a positive effect of plant productivity (Stockdill 1982; Baker *et al.* 1997) which is thought to relate to earthworms critical influence on soil structure, nutrient recycling and distribution (Edwards and Bohlen 1996). In contrast, the current study detected negative correlations between total earthworm abundance and abundance of *A. chlorotica* and vegetation volume on plots where compost was added. This would imply that this relationship between earthworms and vegetation is of greater importance on the unmodified clay cap than where ameliorant has been added. However, this may be misleading as *F. rubra*, a dominant species on plots where compost has been added, was found to correlate positively with the total number of earthworms. This may be due to the tendency of *F. rubra* to form a dense, low-lying mat of vegetation (Grime *et al*. 2007) which has meant that its cover is not directly reflected in the measure of vegetation volume.

4.5.4 Implications for management and conservation

In Butt's (1999) review of earthworm inoculation trials in the UK there are only four that relate to capped landfill which highlights a requirement for the documentation of such studies. The current study raises a number of valuable implications, not only for future earthworm inoculation trials, but also grassland restoration, particularly relating to reinstated soils.

Firstly, the absence of *L. terrestris* from the site of the present study three years following inoculation is indicative of the importance of species selection for inoculation. Whilst deep-burrowing species such as this may be seen as having the greatest potential to improve the soil conditions of the compacted clay cap (Butt 1999), they are also unsuited to these conditions. This makes chances of survival slim, as demonstrated in this study, and therefore unsuitable for inoculation on immature reinstated soils. The result of this study also indicates the importance of the quality of the earthworm inoculum. Commercially sourced earthworms were used and these were supplied in the medium in which they were grown, in an attempt to replicate a starter culture as developed by Butt *et al.* (1995; 1997) as part of the Earthworm Inoculation Unit (EIU) inoculation method. The Butt method is one of the most successful methods of earthworm inoculation that has been used and was developed to address problems such as lack of cocoon transfer, desiccation and damage to the collection site that are associated with more traditional inoculation techniques. However, it is relatively labour intensive and time consuming with a requirement not only to collect earthworms but to maintain these to form a 'starter culture' for several months prior to inoculation. The failure of cheaper commercially sourced earthworm cultures to survive clearly shows that time and labour cannot be saved in this way.

Natural colonisation of earthworms was a more successful approach than inoculation with *A. chlorotica* being recorded just six months after the experiment was initiated, and one year after the clay cap was laid. In addition, earthworms were recorded at higher densities in the current study than has been found in previous studies on similar aged clay caps. This success may relate to the proximity of the experimental plots to an established earthworm population. This warrants further investigation as it may be crucial in determining the speed of natural earthworm colonisation and thus whether it is an appropriate approach. The lack of affinity of earthworms for plots where compost had been added in this study is of key importance to both earthworm inoculation and habitat creation on clay capped landfill. It implies that earthworm populations can develop, or survive if inoculated, in the hostile environment created by the compacted clay cap and other reinstated soils. This means that the addition of ameliorant is not required thus allowing for the optimal soil conditions for hay meadow creation.

No evidence was found to suggest that earthworms had an accelerating effect over three years either on the development of the overall plant community or the soils physical or chemical character. A relationship was, however, detected between earthworms and vegetation. In the absence of compost, earthworm abundance positively correlated with vegetation volume and the abundance of two plant species and further investigation is required to determine whether this is a causal relationship and what is the direction of causality. For example it would be useful to establish whether earthworms do accelerate the establishment of certain semi-natural plant species on clay caps and, conversely, if certain plant species that are readily associated with semi-natural habitats were found to accelerate earthworm colonisation on unmodified clay caps or other reinstated soils, their inclusion in seed mixes would be of benefit.

5.0 THE EFFECT OF MICRO-TOPOGRAPHIC MANIPULATION OF A CLAY CAP TO CREATE RIDGES-AND-FURROWS ON THE DEVELOPMENT OF A HAY MEADOW COMMUNITY ON A DECOMMISSIONED LANDFILL SITE

5.1 Summary

This chapter investigates whether the botanical diversity can be increased in a hay meadow creation scheme on a clay capped landfill through the introduction of micro-topographic variation in the form of ridges and furrows. The objectives are to determine: (1) whether introducing micro-topographic heterogeneity in the form of ridges and furrows affects the plant community composition and whether this is related to edaphic factors including earthworm abundance; (2) whether the height of the ridges and furrows affects plant community composition; and (3) whether there is a difference in plant community composition on the ridges compared to in the furrows and whether this is related to edaphic factors including earthworm abundance. A field scale experiment was established in 2002 comprised of three treatments: (1) ridge and furrow with a 25 cm height differential between the bottom of the ridge and the top of the ridge; (2) ridge and furrow with a 50 cm height differential; and (3) control plots which were level. Overall, results from this study indicate that creating ridge and furrow micro-topographical variation does influence plant diversity and community composition. Whilst significantly different plant communities were not detected where micro-topographic variations had been introduced compared to where they had not, differences in the community that developed on the ridges and in the furrows, and in the furrows of the different ridge and furrow heights were recorded. This means that incorporation of micro-topographic heterogeneity in the restoration of habitat patches within the landscape could positively contribute to Beta diversity, particularly where species selection within the seed mix considered the differing conditions created by these micro-topographical features. It is also concluded that had a more appropriate seed mix been applied that greater differences between the treatments would have been recorded and that alpha diversity would have been promoted where micro-topographic variation had been introduced. Nonetheless, the overall the findings of this study indicate that the ridge and furrow formations that are associated with species-diverse grassland communities in the UK are intrinsic to their nature conservation value and that their incorporation in restoration and creation projects would be of value.

5.2 Introduction

Topographic heterogeneity is defined as pattern in elevation over a specific area (Larkin *et al.* 2006) and this was identified by early ecologists to affect biotic communities, such as the differences in flora observed by Cowles (1899) on dunes of varying structure along the shores of Lake Michgan. It can be considered at a range of scales from broad-scale differences in the landscape (Mladenoff *et al*. 1993; Foster *et al.* 1998) to fine-scale differences, or micro-topography, which describes soil surface variation within an elevation range from roughly one centimetre to as much as one metre (Moser *et al.* 2007). Micro-topography creates variations in the physical environment and chemical and biological processes, such as soil and micro-climatic resources, over small, plant level, spatial scales (Brandani *et al.* 1988, Núñez-Farfan and Dirzo 1998, Clark 1990; Larkin *et al.* 2006). For example, low micro-topographical positions tend to have a greater accumulation of litter, lower moisture loss and temperature variability, greater soil moisture levels, and greater levels of bacteria than level of higher micro-topographical positions (Dwyer and Merriam 1981; Dwyer and Ivarson 1983; Larkin *et al.* 2006; Stenrod *et al.* 2006).

Theory about the relation of this heterogeneity to ecosystem structure and function comes largely from theory on habitat diversity (Larkin *et al.* 2006) and factors that are widely recognised as influencing diversity such as disturbance and resource diversity (Brewer 1994). This suggests that local, or alpha (Whittaker 1960), diversity increases with micro-topographic heterogeneity (Huston 1994) through the creation of a greater diversity of distinct niche spaces (Jeltsch *et al.* 1998). Through its effect on the environment, the coexistence of species is facilitated by the difference in resource exploitation strategies between species (Tilman 1982, Pacala and Tilman 1994) such as germination micro-site preferences (Huenneke and Sharitz 1986; Flinn 2007) and differences in growth and mortality at different micro-topographic positions (Eldridge *et al*. 1991; Vivian-Smith 1997; Ewing 2002). Jeltsch *et al.* (1998) and Tessier *et al*. (2002) expand on this to propose that it is the disturbances causing the micro-topographic heterogeneity that is promoting species diversity through maintaining habitats in a non-equilibrium state. In turn, inter-patch differentiation in species assemblages, or beta (Whitaker 1960) diversity is also increased through the creation of transitional zones, or ecotones, between these micro-habitats (Huston 1994).

Land that is the subject of habitat restoration or creation, such as landfill, open-cast coal mines or intensively farmed agricultural land, often lacks this micro-topographic variation that occurs in natural ecosystems (Moser *et al.* 2007; Thiere *et al.* 2009). Consequently there is a need for greater understanding of how heterogeneity can be introduced to enhance the ecological portfolio of restored sites (Tilman *et al.* 1998). There has been substantial research exploring the restoration of heterogeneity at the landscape level (e.g. Howe 1994; Gore and Shields 1995; Stanford *et al.* 1996; Rundlöf and Smith 2006; Thiere *et al.* 2009). However, micro-habitat manipulations at the local scale to restore heterogeneity have largely focused on wetlands (e.g. Vivian-Smith 1997; Moser *et al*. 2007). Studies have found floristic diversity to be consistently greater in experimental wetland communities with induced heterogeneous micro-topographic elevations, at a scale of as little as 1 to 3 cm (Vivian-Smith 1997; Brose 2001; Moser *et al*. 2007). By comparison, surprisingly little research has extended to the effect of introduced heterogeneity in restored terrestrial systems given that many micro-topographic variations occur in these naturally and that micro-topography is a relatively easily manipulated parameter. Furthermore, where studies have been conducted the results are promising. This is demonstrated by Ewing (2002) and Reader and Buck (1991) who found increased alpha and beta floristic diversity in their studies where mounding was incorporated into prairie creation on capped landfill and restoration of abandoned pasture respectively. The present study builds on this limited resource of research by exploring the effect of induced terrestrial micro-topography on floristic diversity on restored landfill.

Landfill covers significant areas of land throughout the world (Handel *et al.* 1997) and, once decommissioned, presents an important land resource for habitat restoration. Hay meadows are an internationally rare and threatened ecosystem and their restoration and creation has long formed part of the UK's contribution to the international commitment to meeting the challenge posed by the Convention of Biological Diversity, Rio (Department for the Environment 1995). Ridge-and-furrow, which are micro-topographical wave-like undulations deriving from medieval farming practices (Rackham 1986), are now readily associated with species-rich examples of hay meadow communities (Rodwell 1992) and are apparent in many parts of the British lowlands (Beresford 1948; Yelling 1977). While much research has been conducted on hay meadow creation (e.g. Gough and Marrs 1990a, Gough and Marrs 1990b, Janssens *et al.* 1998; McCrea *et al.* 2001; Pywell *et al*. 2002) surprisingly little attention has been given to studying the impact of ridge-and-furrow on soils and vegetation (Cannon and Reid 1993) or the replication of these particular micro-topographic undulations in habitat restoration. It consequently remains unclear whether ridge-and-furrow promotes diversity or whether their association with species-rich grassland communities is purely circumstantial.

Ridge-and-furrow was therefore re-created on cell 21 of Dimmer Waste Disposal Site with the overall aim of investigating the effect of manipulating the micro-topography of a decommissioned landfill site's clay cap to create ridges and furrows on the development of a hay meadow community. A grassland seed mix comprising key species from an MG5 *Cynosurus–Centaurea* community, as categorised by the British National Vegetation Classification (Rodwell 1992), was sown over plots subjected to three treatments: two heights of ridge and furrow, 50 and 25 cm, and level control plots. The objectives were to determine: (1) whether introducing micro-topographic heterogeneity in the form of ridges and furrows affects the plant community composition and whether this is related to edaphic factors including earthworm abundance; (2) whether the height of the ridges and furrows affects plant community composition; and (3) whether there is a difference in plant community composition on the ridges and to in the furrows and whether this is related to edaphic factors including earthworm abundance.

5.3 Methods

5.3.1 Study site

This study was conducted on cell 21 at Dimmer Waste Disposal Site (lat 51º04'42.40'' N; long 2º33'01.73'' W). The site of the experiment lies at 47 m above sea level on a south-east slope with a 3 % gradient. The site was capped with a 1.5 m depth of blue-lias clay in 1998. No existing vegetation was present at the time this experiment was established.

5.3.2 Experimental design

This experiment was set up in April 2002. A randomised design was adopted (Figure 6) and this incorporated nine 20 m by 20 m plots. These were subjected to three treatments: (1) ridge and furrow with a 25 cm height differential between the bottom of the ridge and the top of the ridge; (2) ridge and furrow with a 50 cm height differential; and (3) control plots which were level. Six ridges and six furrows were included in each plot subjected to one of these micro-topographical treatments, and these were created using a mechanical digger. Depressions were scraped out and placed immediately adjacent resulting in furrows that lie lower than level ground and ridges that lie higher. Commercially sourced seed (supplied by Herbiseed, Twyford, Berkshire, UK) with equal proportions of species listed in Table 16 was sown at a rate of 1.8 g per m^2 on all plots in April 2002. All vegetation on the study site was mown on annually in August and all cuttings removed from the site.

Table 16: List of sown species

Species
Agrostis capillaris (Common bent)
Bromus hordaceus (Soft brome)
Centaurea nigra (Black knapweed)
Cynosurus cristatus (Crested dog's-tail)
Daucus carota (wild carrot)
Leucanthemum vulgare (Oxeye daisy)
Phleum pratense (Timothy grass)
Plantago laceolata (Ribwort plantain)
Ranunculus acris (Meadow buttercup)
Trifolium pratense (Red clover)
Vicia sativa (Common vetch)

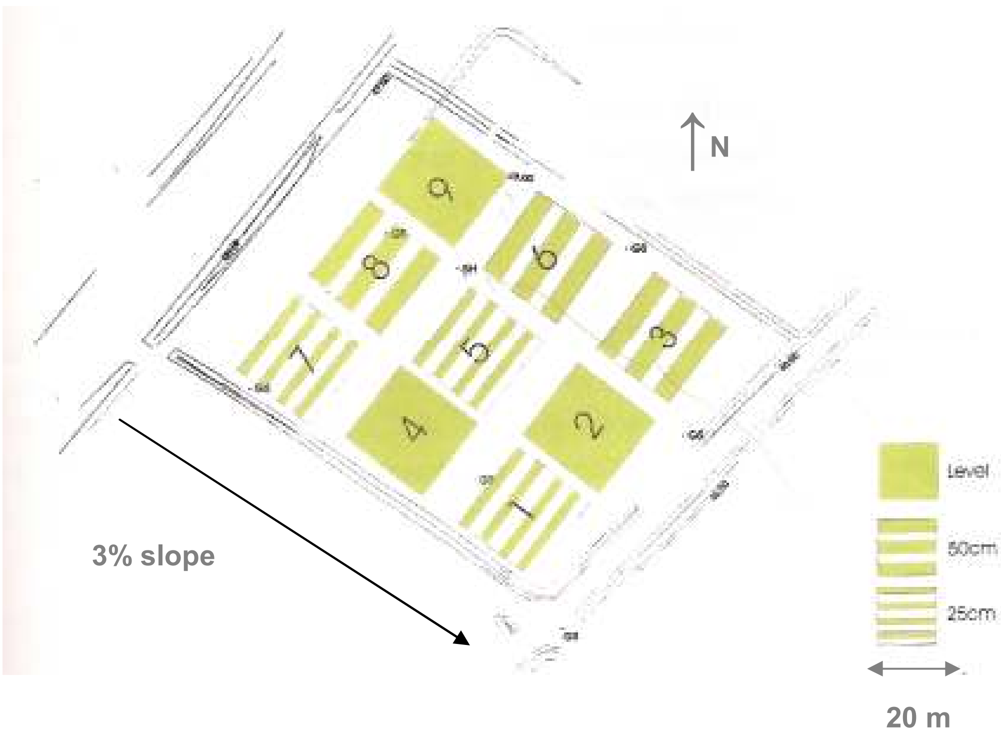

Figure 8: Diagram illustrating randomised experimental design

Vegetation was recorded in July 2006 by estimating percentage cover of each plant using a 1 m^2 quadrat. A total of twenty four quadrats were taken on each plot, half of which were taken on the furrows and half in the ridges where appropriate. The locations of the quadrats were determined using randomly generated numbers. Shannon-Weaver Diversity values (H) were calculated for each quadrat.

Soils were sampled from each plot in July 2006 using a core auger with a 3cm diameter to a depth of 15 cm (Rowell 1994). Four samples were taken from each plot comprising of soils taken in randomly located transects, sampling ridge and furrow positions in duplicate. These were subsequently bulked and frozen at -18°C immediately after collection prior to analysis. Soils were oven dried at 60ºC, during which soil water content was calculated, and then gently ground using an electric pestle and mortar and passed through a 2 mm sieve. Soil was analysed for organic matter content, pH and phosphate using standard methodologies according to Rowell (1994) and for nitrate using vanadium (II) according to Miranda *et al.* (2001) and Doane and Howarth (2003).

Earthworm species and abundance were also sampled in 2006 using the hot mustard extraction method according to Lawrence and Bowers (2002). Twenty four randomly located 0.25 m^2 quadrats were taken in each plot, half taken on the ridges and half in the furrows where appropriate.

5.3.3 Data Analysis

An ANOSIM test was performed to determine similarities in plant communities between treatments using PRIMER version 6 (Clarke and Gorley 2006). All univariate analyses were performed using SPSS version 15. A Kolmorogorov-Smirnov test found that the vegetation and earthworm data were not normally distributed, and it was not possible to transform the data into parametric form. Kruskal-Wallis was therefore used to compare variables across the treatments and Mann -Whitney U-Tests were used to compare variables on the ridges and furrows within and between treatments. Soils data was analysed using ANOVA and paired t-tests.

5.4 Results

5.4.1 Does induced micro-topographic heterogeneity affect the plant community?

The overall plant communities that developed on the two ridge and furrow treatments and the topographically homogeneous controls (Table 17) were not found to be significantly different. This was reflected in both the results of the analysis of similarity test (r = -0.259, P = 0.918) and statistical comparison of the measures of success of the target plant community and abundances of selected seeded and unseeded species (Table 18). The lack of significant differences may relate to the contrasting responses of species within the plant community to the treatments in their three replicates. For example, where median values for each variable are ranked, both the highest and the lowest median rankings for the number of seeded species were recorded on control plots. Both the highest and second lowest abundance of seeded species were also recorded in the control plots. A similar lack of consistency was also observed in the abundances of individual seeded and unsown species. This is exemplified by *L. vulgare* for which the second and third gretest median abundance was recorded in the 25 cm treatment in which the lowest abundance of this species was also recorded. *Agrostis stolonifera* was the most abundant unsown species, and both the highest and the second lowest medians occurred on the controls, and the second highest and lowest medians of abundance were on the 25 cm treatment. Bare ground was also a prominent feature of all treatments. No significant differences in earthworm abundance were detected between treatments (Table 19) and analysis of the soils data (Table 20) revealed pH to be only soil characteristic that differed significantly between the treatments.

Table 17: Medians and the ranges between minimum and maximum percentage cover scores of vegetation (n = 24) recorded within the three replicates for all all three treatments

Variable	***50 cm treatment***			***25 cm treatment***			***Control***		
	Replicate 1 Median (Range)	**Replicate 2 Median (Range)**	**Replicate 3 Median (Range)**	**Replicate 1 Median (Range)**	**Replicate 2 Median (Range)**	**Replicate 3 Median (Range)**	**Replicate 1 Median (Range)**	**Replicate 2 Median (Range)**	**Replicate 3 Median (Range)**
Measures of success									
Number of seeded species	2.0 (5.0)	3.0 (6.0)	1.0 (3.0)	3.0 (6.0)	0.5 (4.0)	3.0 (5.0)	3.0 (4.0)	4.0 (4.0)	1.0 (5.0)
Total abundance of seeded species	11.5 (80.0)	21.0 (84.0)	4.5 (31.0)	26.0 (83.0)	0.5 (28.0)	11.5 (62.0)	32.5 (64.0)	9.0 (55.0)	2.0 (17.0)
Shannon-Weaver Diversity Index	0.8 (1.9)	1.3 (1.9)	0.9 (1.8)	1.4 (1.6)	0.2 (2.0)	1.5 (1.2)	1.0 (1.2)	1.4 (1.7)	0.9 (1.9)
Seeded species									
A. capillaris	0.0 (2.0)	0.0 (10.0)	0.0 (10.0)	0.0 (40.0)	0.0 (0.0)	0.0 (10.0)	0.0 (1.0)	0.0 (1.0)	0.0 (3.0)
B. hordaceus	0.0 (10.0)	0.0 (1.0)	0.0 (0.0)	0.0 (15.0)	0.0 (2.0)	0.0 (2.0)	0.0 (2.0)	0.0 (1.0)	0.0 (1.0)
C. nigra	0.0 (0.0)	0.0 (3.0)	0.0 (0.0)	0.0 (2.0)	0.0 (0.0)	0.0 (1.0)	0.0 (3.0)	0.0 (1.0)	0.0 (1.0)
D. carota	0.0 (6.0)	0.0 (25.0)	0.0 (5.0)	0.0 (10.0)	0.0 (1.0)	0.0 (2.0)	0.0 (2.0)	1.0 (4.0)	0.5 (5.0)
L. vulgare	0.0 (80.0)	10.0 (55.0)	0.0 (30.0)	3.5 (50.0)	0.0 (10.0)	5.0 (55.0)	1.0 (15.0)	3.0 (50.0)	0.0 (10.0)
P. pratense	1.0 (10.0)	1.0 (1.0)	0.0 (3.0)	1.5 (10.0)	0.0 (3.0)	1.0 (2.0)	1.0 (10.0)	1.0 (5.0)	0.0 (2.0)
P. lanceolata	0.0 (5.0)	0.0 (1.0)	0.0 (0.0)	0.0 (4.0)	0.0 (0.0)	0.0 (0.0)	0.0 (1.0)	0.0 (1.0)	0.0 (0.0)
T. pratense	1.5 (30.0)	3.6 (6.0)	0.0 (0.0)	10.0 (60.0)	0.0 (20.0)	1.0 (8.0)	25.0 (65.0)	1.5 (8.0)	0.0 (5.0)

Variable	50 cm treatment			25 cm treatment			Control		
	Replicate 1	**Replicate 2**	**Replicate 3**	**Replicate 1**	**Replicate 2**	**Replicate 3**	**Replicate 1**	**Replicate 2**	**Replicate 3**
	Median (Range)	**Median (Range)**	**Median (Range)**	**Median (Range)**	**Median (Range)**	**Median (Range)**	**Median (Range)**	**Median (Range)**	**Median (Range)**
Unseeded species									
A. stolonifera	40.0 (100.0)	10.0 (100.0)	47.5 (80.0)	20.0 (90.0)	50.0 (99.0)	9.0 (50.0)	22.5 (95.5)	10.0 (98.0)	60.0 (90.0)
L. perenne	0.5 (15.0)	0.0 (5.0)	0.0 (0.0)	4.0 (50.0)	0.0 (5.0)	2.0 (20.0)	0.0 (10.0)	0.0 (10.0)	0.0 (20.0)
T. repens	0.0 (35.0)	0.0 (20.0)	0.0 (20.0)	0.0 (80.0)	0.0 (3.0)	0.0 (20.0)	0.0 (10.0)	0.0 (40.0)	1.0 (50.0)
Bare ground	15.0 (75.0)	20.0 (90.0)	10.0 (70.0)	30.0 (77.0)	25.0 (95.0)	55.0 (80.0)	15.0 (63.0)	50.0 (93.0)	17.5 (7.0)

Table 18: Results of the Kruskal-Wallis tests (*H*) and *P* values comparing the abundance of vegetation across all 3 treatments

Variable	***H***	**Df**	***P***
Measures of success			
Number of seeded species	0.685	2	0.710
Total abundance of seeded species	0.022	2	0.989
Shannon-Weaver Diversity Index	0.881	2	0.644
Seeded species			
A. capillaris	0.000	2	1.000
B. hordaceus	0.000	2	1.000
C. nigra	0.000	2	1.000
D. carota	0.000	2	1.000
L. vulgare	0.461	2	0.794
P. pratense	0.333	2	0.846
P. lanceolata	0.000	2	1.000
T. pratense	0.162	2	0.922
Unseeded species			
A. stolonifera	0.291	2	0.864
L. perenne	3.231	2	0.199
T. repens	2.000	2	0.368
Bare ground	4.101	2	0.129

Table 19: Medians and the ranges between maximum and minimum numbers of earthworms recorded (n = 24) and Krusk-Wallis test (H) test results and P treatments

Variable	***50 cm treatment***			***25 cm treatment***			***Control***			***Kruskal-Wallis test result and P values***		
	Replicate 1	**Replicate 2**	**Replicate 3**	**Replicate 1**	**Replicate 2**	**Replicate 3**	**Replicate 1**	**Replicate 2**	**Replicate 3**			
	Median (Range)	**Median (Range)**	**Median (Range)**	**Median (Range)**	**Median (Range)**	**Median (Range)**	**Median (Range)**	**Median (Range)**	**Median (Range)**	***H***	**Df**	***P***
Total number of earthworms	2.0 (8.0)	3.0 (8.0)	9.5 (24.0)	1.0 (10.0)	0.0 (10.0)	8.0 (14.0)	2.0 (12.0)	5.0 (14.0)	0.5 (10.0)	1.195	2	0.550
Number of *A. chlorotica*	2.0 (6.0)	1.0 (7.0)	8.0 (23.0)	1.0 (7.0)	0.0 (3.0)	5.0 (15.0)	2.0 (12.0)	3.0 (13.0)	0.0 (9.0)	0.638	2	0.727
Number of *L. rubellus*	0.0 (3.0)	1.0 8.0)	0.0 (5.0)	0.0 (3.0)	0.0 (8.0)	1.0 (8.0)	0.0 (3.0)	3.0 (7.0)	0.0 (3.0)	0.095	2	0.953
Number of *L. terrestris*	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (1.0)	0.0 (3.0)	0.0 (0.0)	0.0 (0.0)	0.000	2	1.000

Table 20: Means of soil characteristics (n = 4) recorded within the three replicates of the three treatments and results of ANOVA (F) across treatments

Soil characteristic	***50 cm treatment***			***25 cm treatment***			***Control***			***ANOVA***		
	Replicate 1	**Replicate 2**	**Replicate 3**	**Replicate 1**	**Replicate 2**	**Replicate 3**	**Replicate 1**	**Replicate 2**	**Replicate 3**	**F**	**Df**	***P***
Water content (%)	19.91	23.37	19.01	22.13	25.03	20.04	20.01	20.73	21.12	0.741	2	0.516
Organic matter (%)	17.25	4.88	5.81	5.43	6.87	5.35	4.65	5.88	5.07	0.901	2	0.455
pH	7.81	7.85	7.86	7.68	7.72	7.72	7.82	7.75	7.84	11.083	2	*0.010*
Phosphate (Mg / Kg^{-1})	0.10	0.26	0.11	0.15	0.16	0.18	0.14	0.13	0.08	0.702	2	0.532
Nitrate (Mg / Kg^{-1})	4.14	5.19	3.49	3.61	0.44	21.97	10.63	0.19	2.17	0.344	2	0.722

5.4.2 Does ridge and furrow height affect the plant community?

The overall plant communities that were recorded on the ridges of the two different treatment heights were not found to be significantly different while, interestingly, the communities within the furrows were. This is reflected by the ANOSIM results where $r = -0.185$, $P = 0.07$ where the similarity of the communities on the ridges were compared and $r = 0.407$, $P = 0.01$ where the similarity of the communities in the furrows were compared. Little variation in the abundance of seeded and unseeded species (Table 21) was observed between the ridges of the two treatment heights and none of these differences were found to be significant (Table 22). Whilst the greater number and abundance of seeded species was generally found in the furrows of the 25 cm treatment none of these differences were statistaically significant. Statistical analysis of individual species recorded in the furrows infers that it is the significantly greater abundance of *A. stolonifera* in the furrows of the 50 cm treatment to be the driver for the difference in overall community detected by the ANOSIM test. Signficantly greater levels of bare ground were also present in the furrows of the 25 cm treatment. However, these differences are not reflected in the success of establishment of the target plant community with no significant differences being detected between the furrows of the two treatments. No differences in in earthworm abundance (Tables 23 and 24) between the two ridge and furrow heights were found whilst a single difference in edaphic factors was detected, with significantly higher pH (Table 25) on the ridges of the 50 cm treatment ($t = 3.095$ $P = 0.036$).

Table 21: Medians and ranges between minimum and maximum percentage cover scores of vegetation (n = 24) recorded on the ridges and the furrows on the three replicates of the 25 cm and 50 cm

Variable	***50 cm treatment***						***25 cm treatment***					
	Replicate 1		**Replicate 2**		**Replicate 3**		**Replicate 1**		**Replicate 2**		**Replicate 3**	
	Ridge	**Furrow**	**Ridge**	**Furrow**	**Ridge**	**Furrow**	**Ridge**	**Furrow**	**Ridge**	**Furrow**	**Ridge**	**Furrow**
	Median (Range)	**Median (Range)**	**Median (Range)**	**Median (Range)**	**Median (Range)**	**Median (Range)**	**Median (Range)**	**Median (Range)**	**Median (Range)**	**Median (Range)**	**Median (Range)**	**Median (Range)**
Measures of success												
Number of seeded species	3.0 (4.0)	2.0 (3.0)	3.5 (5.0)	2.5 (6.0)	2.0 (2.0)	1.0 (1.0)	3.0 (6.0)	3.0 (4.0)	2.0 (4.0)	0.0 (4.0)	3.0 (5.0)	4.0 (4.0)
Total abundance of seeded species	29.0 (76.0)	7.5 (31.0)	25.5 (84.0)	8.0 (44.0)	7.5 (30.0)	1.0 (5.0)	18.5 (83.0)	28.5 (60.0)	5.0 (28.0)	0.0 (4.0)	17.0 (59.0)	8.5 (16.0)
Shannon-Weaver Diversity Index	1.5 (1.3)	0.5 (1.1)	1.4 (1.3)	0.7 (1.9)	1.1 (1.3)	0.4 (0.9)	1.7 (1.6)	1.4 (1.2)	0.9 (1.8)	0.0 (2.0)	1.4 (1.0)	1.7 (1.2)
Individual seeded species												
A. capillaris	0.0 (2.0)	0.0 (0.0)	2.0 (10.0)	0.0 (0.0)	2.5 (10.0)	0.0 (0.0)	0.0 (40.0)	0.0 (8.0)	0.0 (0.0)	0.0 (0.0)	0.0 (10.0)	0.0 (1.0)
B. hordaceus	0.5 (10.0)	0.0 (0.0)	0.0 (1.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (15.0)	0.0 (0.0)	0.0 (2.0)	0.0 (0.0)	0.5 (2.0)	0.0 (2.0)
C. nigra	0.0 (0.0)	0.0 (0.0)	0.0 (3.0)	0.0 (3.0)	0.0 (0.0)	0.0 (0.0)	0.0 (2.0)	0.0 (1.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (1.0)

Variable	50 cm treatment						25 cm treatment					
	Replicate 1		Replicate 2		Replicate 3		Replicate 1		Replicate 2		Replicate 3	
	Ridge	Furrow	Ridge	Furrow	Ridge	Furrow	Ridge	Furrow	Ridge	Furrow	Ridge	Furrow
	Median (Range)	Median (Range)	Median (Range)	Median (Range)	Median (Range)	Median (Range)	Median (Range)	Median (Range)	Median (Range)	Median (Range)	Median (Range)	Median (Range)
D. carota	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0
	(6.0)	(3.0)	(0.0)	(25.0)	(0.0)	(0.0)	(10.0)	(5.0)	(0.0)	(1.00	(1.0)	(2.0)
L. vulgare	15.0	0.0	22.5	1.0	3.0	0.0	7.5	0.5	3.0	0.0	15.0	3.5
	(80.0)	(20.0)	(55.0)	(15.0)	(30.0)	(5.0)	(50.0)	(10.0)	(10.0)	(1.0)	(53.0)	(8.0)
P. pratense	1.0	0.5	1.0	1.0	0.5	0.0	1.5	1.5	1.0	0.0	0.0	1.0
	(3.0)	(10.0)	(1.0)	(1.0)	(1.0)	(3.0)	(10.0)	(5.0)	(3.0)	(1.0)	(1.0)	(1.0)
P. lanceolata	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	(5.0)	(1.0)	(0.0)	(1.0)	(0.0)	(0.0)	(4.0)	(1.0)	(0.0)	(0.0)	(0.0)	(0.0)
T. pratense	3.0	2.0	0.0	1.5	0.0	0.0	1.0	20.0	0.0	0.0	0.0	2.0
	(4.0)	(30.0)	(28.0)	(30.0)	(0.0)	(0.0)	(15.0)	(60.0)	(20.0)	(1.0)	(5.0)	(8.0)
Unseeded species												
A. stolonifera	15.0	90.0	5.0	77.5	20.0	52.5	5.0	20.0	37.5	50.0	5.0	12.5
	(100.0)	(85.0)	(90.0)	(100.0)	(70.0)	(80.0)	(90.0)	(90.0)	(85.0)	(99.0)	(50.0)	(43.0)
L. perenne	7.5	0.0	0.5	0.0	0.0	0.0	1.5	5.0	0.0	0.0	1.5	3.0
	(15.0)	(15.0)	(5.0)	(3.0)	(0.0)	(0.0)	(10.0)	(50.0)	(5.0)	(0.0)	(20.00	(10.0)
T. repens	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	(3.0)	(35.0)	(20.0)	(3.0)	(20.0)	(0.0)	(80.0)	(5.0)	(3.0)	(2.0)	(20.0)	(1.0)
Bare ground	20.0	4.5	25.0	12.5	10.0	10.0	25.0	40.0	10.0	50.0	40.0	70.0
	(73.0)	(70.0)	(80.0)	(70.0)	(50.0)	(70.0)	(35.0)	(77.0)	(37.0)	(95.0)	(65.0)	(50.0)

Table 22: Results of Mann-Whitney U-Tests comparing vegetation abundances on the ridges and in the furrows of the 50 and 25 cm treatments

Variables	***Between ridges***			***Between furrows***		
	U	**Df**	***P***	**U**	**Df**	***P***
Measures of success						
Number of seeded species	3.5	2	0.637	3.0	2	0.513
Total abundance of seeded species	2.0	2	0.275	3.0	2	0.513
Shannon-Weaver Diversity Index	4.5	2	1.000	3.0	2	0.513
Seeded species						
A. capillaris	1.5	2	0.121	4.5	2	1.000
B. hordaceus	4.5	2	1.000	4.5	2	1.000
C. nigra	4.5	2	1.000	4.5	2	1.000
D. carota	4.5	2	1.000	3.0	2	0.317
L. vulgare	3.0	2	0.500	3.0	2	0.487
P. pratense	4.0	2	0.817	3.0	2	0.500
P. lanceolata	4.5	2	1.000	4.5	2	1.000
T. pratense	4.0	2	0.796	3.0	2	0.500
Unseeded species						
A. stolonifera	4.0	2	0.817	0.0	2	0.05
L. perenne	4.5	2	1.000	1.5	2	0.121
T. repens	4.5	2	1.000	4.5	2	1.000
Bare ground	3.0	2	0.500	0.0	2	0.050

Table 23: Medians and ranges between the minimum and maximum numbers of earthworm abundance (n = 24) recorded on the ridges and the furrows in the three replicates of the 25 cm and 50 cm treatments

Variable	***50 cm treatment***						***25 cm treatment***					
	Replicate 1		**Replicate 2**		**Replicate 3**		**Replicate 1**		**Replicate 2**		**Replicate 3**	
	Ridge	**Furrow**	**Ridge**	**Furrow**	**Ridge**	**Furrow**	**Ridge**	**Furrow**	**Ridge**	**Furrow**	**Ridge**	**Furrow**
	Median (Range)	**Median (Range)**	**Median (Range)**	**Median (Range)**	**Median (Range)**	**Median (Range)**	**Median (Range)**	**Median (Range)**	**Median (Range)**	**Median (Range)**	**Median (Range)**	**Median (Range)**
Total number of earthworms	2.0 (6.0)	2.5 (8.0)	2.0 (7.0)	3.0 (8.0)	1.0 (10.0)	0.0 (9.0)	1.0 (10.0)	1.5 (5.0)	1.0 (10.0)	0.0 (2.0)	4.0 (14.0)	9.5 (14.0)
A. chlorotica	1.5 (3.0)	2.0 (6.0)	1.0 (7.0)	2.0 (3.0)	1.0 (9.0)	0.0 (4.0)	1.0 (7.0)	1.0 (5.0)	0.0 (3.0)	0.0 (0.0)	4.0 (13.0)	6.0 (15.0)
L. rubellus	0.0 (3.0)	0.0 (2.0)	1.0 (4.0)	1.0 (7.0)	0.0 (1.0)	0.0 (5.0)	0.0 (3.0)	0.0 (1.0)	0.0 (8.0)	0.0 (2.0)	1.0 (5.0)	1.5 (8.0)
L. terrestris	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (1.0)

Table 24: Results of Mann-Whitney U-Tests comparing the abundance of earthworms between the rides and furrows of the 50 and 25 cm treatments, and the test results comparing earthworm abundance between ridges and furrows within the two treatments

Variable	***Between ridges***			***Between furrows***			***Between ridges and furrows within 50 cm treatment***			***Between ridges and furrows within 25 cm treatment***		
	U	**Df**	***P***	**U**	**Df**	***P***	**U**	**Df**	***P***	**U**	**Df**	***P***
Total number of earthworms	4.0	2	0.814	4.5	2	1.000	3.0	2	0.507	4.0	2	0.825
A. chlorotica	4.0	2	0.817	4.5	2	1.000	3.0	2	0.500	4.0	2	0.822
L. rubellus	4.5	2	1.000	4.0	2	0.796	4.5	2	1.000	4.0	2	0.796
L. terrestris	0.0	2	1.000	0.0	2	1.000	0.0	2	1.000	0.0	2	1.000

Table 25: Means of soil characteristics recorded on the ridges (n = 2) and furrows (n = 2) of the three replicates within the 50 and 25 cm ridge and furrow treatments

Soil characteristic	***50 cm treatment***						***25 cm treatment***					
	Replicate 1		**Replicate 2**		**Replicate 3**		**Replicate 1**		**Replicate 2**		**Replicate 3**	
	Ridge	**Furrow**	**Ridge**	**Furrow**	**Ridge**	**Furrow**	**Ridge**	**Furrow**	**Ridge**	**Furrow**	**Ridge**	**Furrow**
Water content (%)	18.67	21.16	19.99	26.74	18.64	19.39	21.16	23.10	23.67	26.40	19.71	20.37
Organic matter (%)	29.21	5.30	5.42	4.35	5.99	5.63	5.53	5.34	5.45	8.29	5.89	4.81
pH	7.72	7.84	7.80	7.90	7.84	7.88	7.66	7.72	7.75	7.79	7.70	7.74
Phosphate (Mg / Kg^{-1})	0.07	0.14	0.38	0.13	0.09	0.13	0.13	0.18	0.14	0.17	0.17	0.20
Nitrate (Mg / Kg^{-1})	1.49	6.79	0.48	9.91	5.98	0.99	6.67	0.56	0.14	0.73	7.42	36.52

5.4.3 Do different plant communities develop on the ridges to in the furrows?

The overall plant community supported on the ridges and in the furrows of the 50 cm height ridge and furrow treatment was found to be significantly different with an ANOSIM result of $r = 0.889$, $P = 0.01$. Significantly greater species diversity (H) was also recorded on the ridges compared to in the furrows of this treatment (Tables 23 and 26) and this difference in species diversity is likely to be linked to the significantly greater abundance of the unseeded *A. stolonifera* in the furrows. In addition *L. vulgare* was also found to be significantly more abundant on the ridges than in the furrows on this treatment. These clear differences in plant community on the ridges and furrows are not reflected by significant differences in soil characteristics except that the pH of the furrows was found to be significantly higher ($t = 4.000$ $P = 0.057$) (Table 25). Earthworms were also recorded at similar densities on the ridges to in the furrows (Table 24).

The overall plant community supported on the ridges compared to in the furrows of the 25 cm treatment plots were also found to be significantly different with an ANOSIM result of $r = 0.037$, $P = 0.05$. However, comparison of the measures of success and abundances of key seeded and unseeded species did not reveal any significant differences. *B. hordaceus* and *L. vulgare* were more abundant on the ridges as was *T. pratense* in the furrows and bare ground was a prominent feature of the furrows. Although these differences were not significantly different when compared individually the cumulative effect of this pattern is likely to have determined the significant differences found in the analysis of similarity of the overall plant community composition. No significant differences were detected in earthworm numbers between treatments (Tables 24 and 25) and as with the 50 cm treatment, few differences in soil characteristics were detected; the only significant differences were higher pH and phosphate levels in the furrows ($t = 6.455$ $P = 0.023$; $t = 4.650$ $P = 0.043$ respectively).

Table 26: Results of Mann-Whitney U-Tests comparing vegetation abundances on the ridges and in the furrows within the 50 cm and 25 cm treatments

Variables	***Within the 50 cm treatment***			***Within the 25 cm treatment***		
	U	**Df**	***P***	**U**	**Df**	***P***
Measures of success						
Number of seeded species	1.5	2	0.184	4.0	2	0.817
Total abundance of seeded species	1.5	2	0.184	4.0	2	0.827
Shannon-Weaver Diversity Index	0.0	2	0.050	4.0	2	0.822
Seeded species						
A. capillaris	1.5	2	0.121	4.5	2	1.000
B. hordaceus	3.0	2	0.317	3.0	2	0.317
C. nigra	4.5	2	1.000	4.5	2	1.000
D. carota	4.5	2	1.000	3.0	2	0.317
L. vulgare	0.0	2	0.046	1.0	2	0.127
P. pratense	2.5	2	0.346	4.5	2	1.000
P. lanceolata	4.5	2	1.000	4.5	2	1.000
T. pratense	4.0	2	0.817	2.0	2	0.246
Unseeded species						
A. stolonifera	0.0	2	0.050	2.0	2	0.268
L. perenne	1.5	2	0.121	2.5	2	0.369
T. repens	4.5	2	1.000	4.5	2	1.000
Bare ground	1.5	2	0.184	0.5	2	0.077

5.5 Discussion

5.5.1 Does induced micro-topographic heterogeneity affect the plant community?

Overall introducing micro-topographic heterogeneity in the form of ridges and furrows was not found to cause significant changes in hay meadow community composition. These findings conflict with the widely accepted principle that increased alpha diversity is associated with increased variation in micro-topography, through providing greater opportunities and micro-site preferences for flora (Pacala and Tilman 1994; Vivian-Smith 1997; Grace *et al.* 2000; Ewing 2002; Svenning and Skov 2002; Flinn 2007; Moser *et al.* 2007) and fauna (Cromsigt and Olff 2006; Stenrod *et al.* 2006; Wang *et al.* 2006). However, differences in certain plant species abundances between replicates but within the same treatment were observed. This variation infers that the plant community was being influenced by the treatments but that the treatment effect was different within the different replicates. Whilst statistical analysis of edaphic factors does not explain this apparent effect, on average water levels were highest in one replicate of both ridge and furrow heights than in the other two. As water relations and soil moisture are recognised as one of the main influences affecting species diversity in topographically heterogeneous environments (Larkin *et al*. 2006; Hokkanen *et al*. 2006; Leuschner and Lendzion 2009) it is possible that this is the factor responsible for this variation that was found in the plant community. It is notable that the differences in water content between plots were only apparent for the plots with ridge and furrow treatment, with the water content of the control plots being consistent within all three replicates. It is therefore suggested that small inconsistentces in engineering and its effect on water retention in the furrows are largely responsible for the variation that was observed in plant community between treatment replicates.

This sensitivity can be both advantageous and disadvantageous in restoration ecology and highlights the importance in field scale trials in furthering knowledge. An advantage is that small differences can produce a wide range of micro-topographic conditions, so promoting diversity. A disadvantage is that it is difficult to predict exact outcomes. The value of micro-topographic manipulation in restoration ecology is a subject that warrants further investigation, but some clear lessons regarding the effect of engineering of the ridges and furrows on plant communities and soil properties and development can be drawn from the present study to inform future research. It is also important that further investigation is conducted into differences in faunal diversity, particularly invertebrate populations, small mammals and nesting waders such as snipe.

5.5.2 Does the size of the ridges and furrows affect the plant community?

Ridge and furrow height was not found to affect the overall plant community on the ridges although the community that developed in the furrows of the two treatment heights was found to be significantly different. However, these differences in overall community composition were not reflected by differences in either the measures of success of the target community nor in the abundances of seeded species. This is surprising given that numerous previous studies have found that micro-topographic variations of as little as a centimetre height can affect plant diversity (Hunt *et al.* 1997; Vivian-Smith 1997; Moser *at al.* 2007). Gradients of increasing moisture, substrate pH and exchangeable calcium, and decreasing inorganic nitrogen and total phosphorus have been shown to be associated with a micro-topographic gradient from higher to lower elevations (Stoeckel and Miller-Goodman 2001; Bruland and Richardson 2005). This, coupled with Bedford *et al.*'s (1999) findings that plant community composition and diversity was linked to species-level responses to differences in nutrients in the soil, means that differences in overall diversity would have been expected and that these would be directly linked to differences caused by the differential between ridge top and furrow bottom. According to the widely accepted theory of species interaction (Tilman 1982) the greatest diversity would have been expected on the 50 cm treatment where the greatest level of micro-topographic variation had been introduced through the facilitation of different resource exploitation strategies of species.

This effect was, however, apparent in that *A. stolonifera* was significantly more abundant in the furrows of the 50 cm ridge and furrow treatment and a significantly greater proportion of bare ground was present in the furrows of the 25 cm furrows. It is these factors that are likely to have influenced the differences that were found in the overall plant community composition of the furrows. Again, this may relate to water retention in the deeper furrows of the 50 cm treatment making the environment here more suitable for the competitive grass species *A. stolonifera* and less suitable for the range of typical hay meadow plant species that were included in the seed mix.

5.5.3 Do different plant communities develop on the ridges to the furrows?

Differences in plant community were found on the ridges compared to in the furrows, and these differences were most pronounced in the 50 cm treatment. Overall species diversity was greater on the ridges than in the furrows in this treatment whereas no differences were detected in the measures of success of the target plant community in the 25 cm treatment. It is therefore surprising that no differences in edaphic factors that are likely to be of ecological significance between the ridges and the furrows were revealed to explain these differences in plant community. In a study on the influence of ridge and furrow on the soil water regime

Cannon and Reid (1993) found soils in the furrows to have significantly higher moisture content, exchangeable potassium and pH. Although statistically pH was found to be higher in the furrows than on the ridges, the differences are likely to be too minor to have affected plant distribution or growth given that all sown species tolerate pHs' between 5.5 and 7.5 (Grime *et al.* 2007). However, although statistically insignificant, soil water content was consistently found to be higher in the furrows than on the ridges. Failure to detect a statistical difference here may be due a type 2 error or may relate to the time at which soils were sampled. Incorporation of greater frequency and intensity of soil water sampling may, therefore, have provided useful information in determining whether this is the parameter that has shaped these differences in plant community on the ridges and furrows.

In contrast to the lack of detection of significant edaphic differences between ridge and furrow in both the 25 cm and 50 cm treatments, the abundance of individual species of plant does help to explain differences in diversity and infers differences in edaphic factors. The lower diversity of the furrows compared to the ridges of the 50 cm treatment appears to be linked to the dominance of the furrows by *A. stolonifera*. This is species is recognised as having a high tolerance of seasonally inundated and moist soils (Grime *et al*. 2007) and so may be responding to the higher water content in the furrows of the 50 cm treatment. It is a procumbent mat-forming species that has been found to out-compete other grasses and forbs (Carrington and Diaz 2009), and may account for these difference in plant community composition and lower diversity in the furrows of the 50 cm treatment. Lower diversity in the furrows of the 50 cm treatment may also be linked to selection of sown species, where all except one species, *T. pratense*, favour the drier environment provided by ridges. The less pronounced differences in community between the ridges and furrows of the 25 cm treatment, and particularly between numbers and abundances of sown species infers less differential in water content in this treatment. Whilst greater abundances of species such as *Pulicaria dysenterica* and *T. pratense* which favour the damper furrows (Grime *et al*. 2007) were present here, greater abundance of species suited to drier environments such as *L. vulgare* and *B. hordaceus* (Grime *et al.* 2007) were also present.

Some of the most botanically diverse examples of hay meadow communities and permanent pastures are associated with areas of land that bear the wave like undulations of ridge and furrow in the UK (Rodwell 1992; Cannon and Reid 1993). These often characteristically support mosaics of drier hay meadow community species on the ridges and a community comprising species more suited to damper conditions in the furrows, such as *Lychnis flos-cuculi* (ragged robin), *Juncaceae* (rushes) and *Fillipendula ulmaria* (meadowsweet), in the furrows (Rodwell 1992). This indicates that if some of these species characteristic of damper conditions had been included in the seed mix it is likely that the 50 cm treatment may have supported the greatest botanical diversity. However, the commercial hay meadow mix used in the present study contained very few of these species and the most moisture tolerant sown species present in its experimental plots, *T. pratense*, was either absent or occurred at low abundances in the 50 cm furrows. This coupled with the abundance of *A. stolonifera*, infers that the deeper furrows of the 50 cm treatment are considerably wetter than those of the 25 cm treatment and that periods of inundation are not conducive to the target plant community that can be obtained from sowing at least one commercial MG5 grassland mix.

5.5.4 Implications for management and conservation

Previous accounts of restoration attempts that incorporate ridge and furrow are extremely sparse and inconclusive. For example, Box (2003) reviews the translocation of part of Brampton Meadows Site of Special Scientific Interest (SSSI) in Cambridgeshire in which ridge-and-furrow topography was created on an area of land using sub-soil from the donor site. Unfortunately in this case the re-located grassland degenerated into a species-poor coarse grassland community due to a lack of subsequent management. Surprisingly little attention has been paid to studying the impact of ridge and furrow on the soil conditions and plant communities (Cannon and Reid 1993) and it is unclear whether ridge and furrow contribute to this diversity or whether they are simply a consequence of land that has not been subject to mechanised agricultural practices. Overall, results from the present study indicate that creating ridge and furrow micro-topographical variation can clearly influence diversity and plant community composition. The effects detected are also likely to apply to hay meadow creation on systems other than the clay capped landfill on which the current study was conducted, such as agricultural restoration, brown-field sites or other re-instated lands.

Whilst significantly different plant communities were not detected where micro-topographic variations had been introduced compared to where they had not, differences in the community that developed on the ridges and in the furrows, and between different ridge and furrow heights were recorded. This means that incorporation of micro-topographic heterogeneity in the restoration of habitat patches within the landscape could positively contribute to Beta diversity, particularly where species selection within the seed mix considered the differing conditions created by these micro-topographical features. It is also likely that a more appropriately selected seed mix to reflect the different conditions created by these micro-topographic undulations would have resulted in greater alpha diversity compared to the topographically homogeneous plots. Nontheless, the overall the findings of this study indicate that the ridge and furrow formations that are associated with species-diverse grassland communities in the UK are intrinsic to their nature conservation value and that their incorporation in restoration and creation projects would be of value.

6.0 DISCUSSION AND FINAL CONCLUSIONS

6.1 Discussion

This thesis aimed to investigate factors involved in the successful establishment of a hay meadow community on clay capped landfill and how restoration processes might be accelerated. Whilst much literature is available on habitat restoration and creation, advice specific to post-operative landfill and the extent to which the widely accepted principles of habitat restoration are transferable to landfill is sparse. This may be due to landfill sites having traditionally been restored to agricultural or amenity grassland (Simmons 1999), thus demonstrating a socially beneficial after-use for the continued public acceptability of the waste management industry (Watson and Hack 2000). Nonetheless, it is now 17 years since the Convention of Biological Diversity, Rio-de-Janeiro, during which a multi-national commitment to halting and indeed reversing the loss of internationally threatened ecosystems was made (Department of the Environment 1995). This, combined with the global extent of post-operative landfill and thus its potential as a land resource for habitat restoration, highlights the importance for research in this area.

Restoration ecology has traditionally focused on restoration of habitats and ecosystems at a site scale (Falk *et al.* 2006) and on the influence of regional processes, operating at large temporal and spatial scales and how these can affect and even constrain local community structure (Menninger and Palmer 2006). More recently the importance of understanding species interactions and the underlying ecological processes in shaping communities or species assemblages is being acknowledged (Temperton *et al.* 2004). This thesis therefore considered the effect of the manipulation of three parameters, soils, vegetation and micro-topography on underlying ecological processes and species interactions shaping a target hay meadow community. These parameters are recognised as being influential in the development and maintenance of semi-natural hay meadow systems (Rodwell 1992; Sutherland and Hill 1995), and can be manipulated both in experimental conditions and more extensive ecological restoration scenarios (Dryden 1997; Bradley *et al.* 2006; Larkin *et al.* 2006). This thesis also acknowledges that in many circumstances there is a need to accelerate the rate of the restoration process to swiftly achieve a green cover. It has been suggested that this may be beneficial in habitat restoration to provide shelter for sown seedlings as well as suppressing excessive growth of competitive species (Mitchley *et al.* 1996) and to address any discrepancies that lie between the high ideals of restoration ecology goals and reality (Pfadenhauer 2001). In particular, Pfadenhauer identifies insufficient social acceptance of landscape changes during restoration processes as a limiting factor. Higgs (2005) criticises ecological restoration practitioners for failing to adequately address these conflicts between science and socio-cultural factors.

Three field scale experiments were consequently established with the objectives of:

1. Investigating the effect of manipulation of soil and vegetation on the development of the target plant community, edaphic factors and earthworm populations on clay capped landfill that has been stabilised with green cover;
2. Investigating the use of earthworms as ecological engineers in speeding up the establishment phase of the grassland sward via their impact on the soil's physical and chemical properties; and
3. Investigating the effect of manipulation of micro-topography on the development of the target plant community, edaphic factors and earthworm populations on clay capped landfill.

In addition to addressing these objectives, through establishing these experiments this study has demonstrated that a plant community supporting the key species associated with the internationally threatened hay meadow habitat can be successfully created on clay capped landfill despite the additional constraints posed by the hostile environment created by the clay cap.

The experiment that was established to address the first objective clearly demonstrated that increased plant diversity can be promoted through the manipulation of soil and vegetation, even once the clay cap has been stabilised with a green cover. Despite the problems with depleted soil conditions associated with the clay cap (Scullion and Mohammed 1991; Athy *et al.* 2006), the mechanical removal of topsoil and vegetation and sowing of the seed mix onto the bare clay was the most successful method in the long-term for establishing a hay meadow community. The retention of topsoil and ameliorant initially resulted in a greater success in seeded species, probably through creating more suitable soil conditions for germination (Parker 1995; Gilbert and Anderson 1998; Watson and Hack 2000; Gregory and Vickers 2003. However, three years after sowing the abundance of sown species significantly decreased as a direct result of competition from *Elytrigia repens*. This agrees with Simmon's (1999) who suggested that topsoil should not be used on areas where wildflower seed is to be sown on clay capped landfill as the fertility of the topsoil can encourage aggressive competitive species to grow at the expense of the more desirable forb species, and the diversity can be lost. The success of *E. repens* on treatments where topsoil and ameliorant were added directly correlated to the higher levels of organic matter, phosphate and soil water content associated with this substrate mix. This finding supports previous studies that suggest that floristic diversity of low-input grassland systems are closely linked with phosphate availability (Lloyd and Pigott 1967; Rorison 1971; Gough and Marrs 1990a; Janssens *et al.* 1998).

Although greatest plant diversity was observed in the absence of topsoil and ameliorant, the development of the plant community was slow with a mean of 7 % bare ground even six years after the experiment was established. Whilst this is largely conducive with the goals of ecological restoration, which can be viewed as an attempt to recover a natural range of ecosystem composition, structure and dynamics (Falk 1990; Allen *et al.* 2002; Palmer *et al.* 2005), there is potential for conflict with social perception and public acceptability (Watson and Hack 2000; Pfadenhauer 2001; Higgs 2005). One solution would be to find a mechanism by which the rate of colonisation of vegetation can be accelerated without the addition of topsoil or ameliorant.

Research has long documented both the benefits of earthworms to soil development (Teotia *et al.* 1950; Ehlers 1975; Stewart 1981) and the benefits of the development of plant communities to earthworms (Mato *et al.* 1989; Doube and Brown 1998; Pizl 1999). A total of five species of earthworm, with species representative of the three major groups based on burrowing and feeding characteristics (Bouché 1977), were recorded on experimental plots investigating the effect on soil and vegetation on the development of the plant community. This occurrence of earthworms from different functional groups and what was a relatively even distribution of earthworms across plots with and without added ameliorant or existing vegetation is interesting as the processes of soil stripping, storage and reinstatement contribute to a large reduction in earthworm populations (O'Flanagan *et al.* 1963; Scullion *et al*. 1988). This clearly demonstrates that despite being subjected to these processes and additional problems associated with depleted soil conditions (Butt *et al.* 2004; Athy *et al.* 2006) the clay cap can sustain earthworm populations. However, this study found no evidence that earthworm abundance was a factor in determining plant community composition. This may be due to the low population densities of earthworms in this objective's experimental plots compared to natural systems where many of these benefits have been observed. Mean total numbers of earthworms recorded ranged from 87.2 to 174 m^2 compared to numbers ranging from 390 to 470 m^2 recorded by Svendsen (1955) and 400 to 500 m^2 as recorded by Cotton and Curry (1980) in old pastures in the UK. It was therefore hypothesised that, where introduced in sufficient numbers, earthworms may have the potential to accelerate the establishment of the hay meadow community. Furthermore, in a review of earthworm inoculation trials Butt (1999) identified only four that relate to capped landfill, highlighting the requirement for further work and documentation on this subject.

An experiment investigating the use of earthworms as ecological engineers in speeding up the establishment phase of the grassland sward via their impact on the soil's physical and chemical properties was consequently established as part of this thesis in 2005. Unfortunately limited success with the earthworm inoculation itself was observed in this experiment although a number of valuable implications for future earthworm inoculation trials and grassland restoration, particularly relating to reinstated soils, were identified. A high mortality of inoculated earthworms was observed, particularly with *L. terrestris* which was no longer recorded and was assumed absent three years following inoculation. This highlights the importance of species selection for inoculation. Whilst deep-burrowing species such as *L. terrestris* may be seen as having the greatest potential to improve the soil conditions of the compacted clay cap (Butt 1999; Butt 2008), they are also unsuited to these conditions.

Natural colonisation of earthworms was a more successful approach than inoculation with *A. chlorotica* being recorded just 6 months after the experiment was initiated, and only one year after the clay cap was laid. However, no evidence was detected within the three year study to suggest that earthworms had an accelerating affect on the development of the hay meadow community or the soils physical or chemical character. It is postulated that this is because earthworm numbers were too low to have an effect on vegetation growth in such a short timescale. Where positive responses of vegetation to earthworm inoculation have been previously observed earthworm density was considerably higher (Stockdill 1982; Baker *et al.* 1997). The density of naturally colonised earthworms in the present study were, however, higher than those observed on other similar sites. For example, Butt *et al.* (1999) failed to locate any natural earthworm colonists after three years in their study on a partially restored landfill site. Similarly, Judd and Mason (1995) did not record any earthworms on the one year old trial site in their experiment. This higher rate of colonisation may be attributed to the proximity of existing earthworm populations. For instance, there is a series of established banks and hedgerows within 15 m of the site of this experiment. Further study on both on the rate of natural earthworm colonisation of clay capped landfill in relation to existing stable populations and on earthworm inoculation would be of interest. However, at present results from this thesis suggest that the use of earthworms to accelerate the establishment of a species-rich green cover is not viable.

This conflict between the requirement to swiftly establish a green cover and to create species-rich grassland habitat for nature conservation interest is not unique to landfill. It has also been identified as a major problem associated with the rapid stabilisation of spoil created during the construction of the Channel Tunnel (Kershaw *et al.* 1995; Mitchley *et al.* 1996) as well as in the reversion of intensive agricultural land (Walker *et al.* 2004). One solution has been to sow nurse crops such as *Lolium perenne* or *L. multiflorum* with species mixtures to establish a swift green cover and improve the establishment of target species by providing shelter for seedlings as well as suppressing excessive weed growth (Mitchley *et al.* 1996). Field trials have shown that on low fertility soils a decline in the abundance of the nurse crops is observed in the second season, allowing slower-growing wildflower species to expand their cover (Kershaw *et al.* 1995; Mitchley and Buckley 1995). In contrast, however, most studies have recorded few beneficial effects of nurse crops (Pywell *et al.* 2002). In addition, problems associated with the initial establishment of the nurse crop on soils of low fertility have been encountered (Mitchley *et al.* 1996) and in one study the abundance of the nurse crop, *L. multiflorum*, was detrimental to the early establishment of a lowland wet grassland sward (Manchester *et al.* 1997). The present research has shown that an alternative solution may be to swiftly establish a green cover across the site through the addition of topsoil and ameliorant and then create patches of more diverse communities.

The experiments documented in this thesis not only indicate that higher diversity within habitat patches could be achieved through the removal of topsoil but also that botanical diversity can be enhanced through introducing micro-topographic heterogeneity. Theory relating to micro-topographic heterogeneity suggests that alpha diversity increases with micro-topographic heterogeneity through the creation of a greater variety of distinct niche spaces (Jeltsch *et al.* 1998). Overall, the results of this experiment indicated that creating micro-topographic heterogeneity in the form of ridge and furrow directly influenced plant diversity and community composition. Whilst significantly different plant communities were not detected where micro-topographic variations had been introduced compared to where they had not, differences in the community that developed on the ridges and in the furrows, and between different ridge and furrow heights were recorded. This means that incorporation of micro-topographic heterogeneity in the restoration of habitat patches within the landscape could positively contribute to Beta diversity, particularly where species selection within the seed mix considered the differing conditions created by these micro-topographical features. It is also likely that a more appropriately selected seed mix to reflect the different conditions created by these micro-topographic undulations would have resulted in greater alpha diversity compared to the topographically homogeneous plots.

Interpretation of the effects of micro-topographic heterogeneity on diversity was impeded by the variation that was seen in the replicates within the treatments, most likely caused by inconsistent engineering of the micro-topographic variations and its effect on water retention in the furrows. This sensitivity can be both advantageous and disadvantageous in restoration ecology and highlights the importance of field scale trials in furthering the knowledge of ecological restoration. An advantage is that small differences can produce a wide range of micro-topographic conditions to promote diversity. A disadvantage is that it is difficult to predict exact outcomes. Nonetheless, the results of this experiment clearly show that the introduction of micro-topographic variations does affect the development of the plant community, can enhance biodiversity and has the potential to be a useful tool in creating habitat patches.

The creation of habitat patches could be considered to be an extension at the small spatial scale of modern day thinking that landscape heterogeneity is a key factor promoting biodiversity (Wagner *et al.* 2000; Rundlőf and Smith 2006). Heterogeneous landscapes are characterised by semi-natural habitat patches such as wetlands, hedgerows, and grasslands bordering cultivated fields (Thiere *et al.* 2009). Since the second half of the 20th Century the intensification of land-use practices and the associated decline of semi-natural habitats have been major drivers of biodiversity loss at local, regional and global scales (Norris 2008). Creation of habitat patches in homogeneous agricultural landscapes is being increasingly promoted in Europe to address the loss of multi-functionality in agricultural landscapes in general (Otte *et al.* 2007). Thiere *et al.* (2009) found that both local and regional diversity was increased through the creation of 300 ha of dual-purpose wetlands in the intensive agricultural lowlands of south-west Sweden.

Previous studies that have found benefits in creating semi-natural habitat patches appear to have largely concentrated on the landscape or regional scales (Moser *et al.* 2009; Thiere *et al.* 2009), and the question of whether these principles will transfer to smaller scales of heterogeneity such as those possible within single landfill sites is currently unanswered. As an indication, typical covered landfill sites in Worcestershire range in size from 1.26 to 23.4 hectares (Worcestershire County Council 2009). This therefore raises the important question of at what spatial and temporal scales can habitat patches of communities of high nature conservation value be created and sustained within the wider decommissioned landfill site. Preliminary results from this thesis suggests that a sown hay meadow community can be sustained, and indeed is continuing to develop, six years after creation on plots as small as 100 m² that are surrounded by grassland dominated by competitive grasses,. This success depends on providing and maintaining a low nutrient status edaphic environment.

Previous studies of grassland re-creation on ex-arable land suggest that persistence in the longer term, and the ability to colonise surroundings stabilised with green cover, is also linked to dispersal ability (Pywell *et al.* 2003) and to the specific germination traits of species (Burke and Grime 1996; Pakeman *et al.* 2002; Pywell *et al.* 2002). Both persistence and colonisation ability appear to be correlated to species that have higher germination rates in the light than dark (Pakeman *et al.* 2002) and larger seeds (Burke and Grime 1996; Pakeman *et al.* 2002; Pywell *et al.* 2002). For example *Centaurea nigra* and *Trisetum flavescens* (yellow oat grass) have good dispersal abilities and are able to persist within a closed grassland sward (Pakeman *et al.* 2002). Further investigation of the interaction between species responses to specific abiotic conditions associated with landfill and other species, and of species responses to the heterogeneity, size and distribution of habitats, may be key to the successful landscape design of capped landfill.

6.2 Limitations of the current study and recommendations for further work

One of the main limitations encountered during this study are those relating to field scale experiments. Ecological systems are naturally heterogeneous and there is growing evidence that the processes which generate natural variability operate over difference scales of space and time (Thrush *et al.* 2000). Field scale trials benefit in representing this heterogeneity by having a high degree of realism (Diamond 1986), and extrapolations require fewer assumptions when experimental scales are similar to the scale at which predictions are needed (Carpenter 1999). The problems associated with extrapolating results from laboratory or microcosm experiments means that these types of experiments have limited relevance for community ecology and the study of ecosystem processes (Carpenter 1999). However, there are also a number of disadvantages associated with field scale trials. Pakeman *et al.* (2002) identify issues such as competition from invading species, dispersal of sown species between plots and invasion from persistent seed banks as having the potential to reduce the integrity of conclusions drawn from trials. Problems associated with the seed bank were not a problem with the freshly laid clay cap and the potential of dispersal of plants between experimental plots was minimised by incorporating buffer zones between plots. Nonetheless, the dispersal of earthworms and their suspected colonisation from adjacent habitat into objective two's experimental plots made it difficult to draw conclusions on the survival of inoculated earthworms here. It did, however, highlight the fact that earthworms can readily colonise even the most inhospitable soils such as the freshly laid compacted clay cap of this study's site relatively quickly. It is likely that in this instance the network of hedgebanks that had been established for approximately ten years that surrounded the study site may have acted as a source for this natural colonisation. Further investigation would therefore be useful to ascertain whether it was these features that aided the rapidity of natural earthworm colonisation. If this was the case these features could easily be incorporated into the design of post-operative landfill sites to enhance the rate of natural earthworm colonisation which, in theory, would ultimately increase the rate of development of a soil structure more conducive to supporting a greater diversity of plants and soil flora and fauna (Davis 1986; Walker *et al.* 2004).

Disadvantages in field scale trials were also particularly highlighted in objective three's experimental plots, investigating the use of induced micro-topography in hay meadow creation. Firstly, the size of the plots required to incorporate the ridge-and-furrow meant that the possible rate of replication was low due to constraints in the area of land available, with only three plots of each treatment. This not only meant that it was difficult to draw conclusions due to the low rate of replication but there also appeared to be substantial variation in the results and this is suspected to have been caused by small differences in the engineering of the ridge-and-furrow. With greater replication it is likely that a more consistent pattern in differences caused by ridge and furrow height would have been detected. Importantly, this would also have provided greater statistical power for analysis. The results of this study certainly infer that where the area of available land permits that a similar experiment with greater replication would yield interesting results and would be likely to reveal greater differences between treatments. In hindsight, a greater intensity of sampling could have been achieved without psuedoreplication by recording the variables, such as plant abundance and soil characteristics, at regular intervals throughout the year. This is likely to have been of greatest use for soil characteristics, particularly soil water content which is likely to have fluctuated considerably on a seasonal basis and was identified as being the most likely parameter to have been affecting the differences in plant community compostition of the ridges and furrows. In fact, one of the main conclusions that were drawn from this experiment was the inappropriate selection of plant species that was sown which almost entirely comprised species suited to dry conditions. Given that low micro-topographical positions tend to have greater soil moisture levels (Dwyer and Merriam 1981; Dwyer and Ivarson 1983; Larkin *et al.* 2006) it is not entirely surprising that lower plant diversity was recorded in the furrows than on the ridges of the 50 cm treatment. Had the seed mix contained hay-meadow species suited to damp conditions it is likely that plots with indroduced micro-topographical variation would have supported a significantly more diverse plant community than the micro-topographically homogeneous ones. This result would have reflected previous research where increased alpha diversity was associated with increased variation in micro-topography, through providing greater opportunities and micro-site preferences for flora (Vivian-Smith 1997; Grace *et al.* 2000; Ewing 2002; Svenning and Skov 2002; Flinn 2007;

Moser *et al.* 2007). Further investigation of this on objective three's experiment has consequently been planned in which the plots will be over-sown in April 2010 with a seed mix comprising typical hay meadow species that favour both dry and damp conditions. The effect this has on plant community compostion and species diversity within the micro-topographically homogenous and heterogenous plots will be monitored.

Another limitation associated with the current study is the narrow focus of variables measured and / or monitored. This study focused on target floral plant communities, soil physical and chemical characteristics, which have traditionally been the basis of habitat restoration (Gough and Marrs 1990a; Gough and Marrs 1990b; Janssens *et al.* 1998; Bakker and Berendse 1999). However, given the importance of microbial, fungal and invertebrate communities, both as functional components of grasslands and in terms of structuring plant communities (Walker *et al.* 2004), the consideration of these other trophic levels should be an integral part of all restoration programs (Young 2000). Earthworms are widely accepted as being of particular importance due to their role in generating and maintaining soil structure (Davidson and Grieve 2006) and as such these were included as a component in all three experiments conducted as part of this thesis. Naturally, limits on resources restricted the scope of other faunal and floral communities that could be included in this study, although further investigation into the interactions of groups such as arbuscular mycorrhizal fungi, soil bacteria, *Collembola* and mites, with the parameters studied in this thesis (soils, vegetation and micro-topography) would be of benefit.

Perhaps the most profound constraint associated with this study is the fact that it remains unclear how transferable the principles that were determined in the current study are to other sites. Although it needs to be tested on multiple sites, it is likely that principles derived from the current study would apply to other clay capped landfill sites as this is essentially a man-made substrate that involves the application of clay using standard methods of engineering. However, the effectiveness of the application of these principles to other scenarios of restoration ecology such as ex-arable or pasture, opencast coal mines or other brownfield sites is even less clear. This was particularly highlighted in objective one's experiment where the effect documented by Warren (2000) of *T. repens* reducing forb abundance and diversity by promoting the growth of competitive grasses through its nitrogen fixation abilities, was not detected in the present study. Indeed *T. repens* was most abundant in the plots with bare clay where ameliorant had been removed and species diversity increased with increased abundance of this species. It was suggested that it was the depleted soil conditions specific to the clay cap that was preventing this effect which may mean that this treatment of top-soil stripping would not prove as successful on other more natural soils. Pywell (2002) highlights the benefits of multi-site experiments where he was able to test the generality of treatment effects by applying them to five different sites distributed across different regions of Southern England. Ideally this thesis would have incorporated a similar variety of sites to add substance to the findings although unfortunately, as with much research, financial and time constraints did not allow for this in the current study. An alternative may have been to run glasshouse or laboratory experiments alongside the field scale trials. Useful parameters to have measured in this way are the effect of substrate on phosphate levels and the relationship between this and plant nutrient content in objective one's experiment. Likewise, to have inoculated the commercially sourced earthworms of objective two's experiment in containers within the laboratory would have helped to assess more accurately whether it was the poor sourcing of incolum that was responsible for the level of mortality that was observed.

Finally, the other factor that may limit the applicability of this work to other hay meadow creation schemes is the lack of aftermath grazing. This type of disturbance is intrinsic to the development of semi-natural hay meadow communities (Sutherland and Hill 1995; Crofts *et al.* 1999) and where comparisons have been made it is the combination of taking a hay crop and aftermath grazing that has been the more successful approach than either cutting or grazing alone (Hayes *et al.* 2000; Smith *et al.* 2000; Hayes and Sackville Hamilton 2001). Smith (2000) found that the aftermath grazing created germination gaps which allowed for the colonisation of sown and naturally dispersed forbs. In contrast, hay cutting without aftermath grazing has been shown to favour coarse grasses (Hayes and Sackville Hamilton 2001) which is the effect observed in both objective one and two's experimental plots. However, there are more often than not constraints associated with after-use of landfill as there are with other reclaimed land such as open cast coal mines or urban brown field sites. On many of these sites, as is the case at Dimmer waste disposal site where the current study was undertaken, grazing is simply not possible. The challenge for these sites is therefore to adapt more traditional and widely accepted methods of habitat creation to work within the bounds of the constraints. It may have therefore been useful to compare possible management alternatives to attempt to re-create some of the disturbance effects of grazing. This could have included different intensities or timing of hay cropping, causing disturbance mechanically to mimic that caused by cattle or even to compare sites where other forms of grazers such as deer and rabbits are active to those where they are not. Again, it was not possible to include this in the current study due to limited resources, however it is a subject that clearly warrants further investigation and is likely to be key in improving the success and quality of habitat created in the restoration of reclaimed land.

6.3 Final conclusions

This thesis identified soil as being the key factor in the successful establishment of a hay meadow community on clay capped landfill and the introduction of micro-topographic undulations as having the potential to increase the botanical diversity. Attempts to accelerate the colonisation of the clay cap through the introduction of earthworms were unsuccessful although a suggestion to address the potential conflict between science and socio-cultural factors was made. This involves the swift establishment of a green cover across the site through the addition of topsoil and ameliorant and sowing with fast-growing grass species and the creation of small patches of habitat with sown assemblages of species based on semi-natural counterparts.

7.0 REFERENCES

Allen, C.D., Savage, M., Falk, D.A., Suckling, K.F., Swetnam, T.W., Schulke, T., Stacey, P.B., Morgan, P., Hoffman, M. and Klingel, J.T., 2002. Ecological restoration of southwestern Ponderosa pine ecosystems: A broad perspective. *Ecological Applications,* 12 (5), 1418-1433.

Allison, M., and Ausden, M., 2004. Successful use of topsoil removal and soil amelioration to create heathland vegetation. *Biological Conservation*, 120 (2), 221-228.

Ash, H.J., Bennett, R., and Scott, R., 1992. *Flowers in the grass. Creating and managing grasslands with wildflowers.* Peterborough: Nature Conservancy Council.

Ashman, M.R. and Puri, G., 2002. *Essential soil science: a clear and concise introduction to soil science*. Oxford: Blackwell Science.

Athy, E.R., Keiffer, C.H., and Stevens, M.H., 2006. Effect of mulch on seedlings and soil on a closed landfill. *Restoration Ecology*, 14 (2), 233-241.

Baines, C., and Smart, J. 1991. *A guide to habitat creation (2nd edition)*. London: London Ecology Unit.

Baker, G.H., Williams, P.M.L., Carter, P.J., and Long, N.R., 1997. Influence of lumbricid earthworm on yields and quality of wheat and clover in glasshouse trials*. Soil Biology and Biochemistry,* 29 (3/4), 599-602.

Baker, G.H., Chan, K.Y., and Munro, K., 2003. Influences of native (Megascolecidae) and exotic (Lumbricidae) earthworms on ryegrass production in acidic pasture soils from south-eastern Australia. *Pedobiologia*, 47, 830-834.

Bakker, J.P., 1989. *Nature management by grazing and cutting*. London: Kluwer Academic.

Bakker, J.P., and Berendse, F., 1999. Constraints in the restoration of ecological diversity in grassland and heathland communities. *Trends in Ecological Evolution,* 14 (2), 63-68.

Balmford, A., Bennun, L., Brink, B.T., Cooper, D., Cōte, I.M., Crane, P., 2005. The convention on biological diversity's 2010 target. *Science*, 307, 212-213.

Berendse, F., Oomes, M.J.M., Altena, H.J., and Elberse, W.T., 1992. Experiments on the restoration of species-rich meadows in the Netherlands. *Biological Conservation*, 62 (11), 59-65.

Beresford, M.W., 1948. Ridge and furrow and the open fields. *Economic History Review*, 1 (1), 34-45.

Blackstock, T.H., Rimes, C.A., Stevens, D.P., Jefferson, R.G., Robertson, H.J., Macintosh, J., and Hopkins, J.J., 1999. The extent of semi-natural grassland communities in lowland England and Wales: a review of conservation surveys 1978-1996. *Grass Forage Science*, 54, 1-18.

Bouché, M.B., 1977. Strategies lombriciennes. *In:* Lohm, U., Persson, T., eds. *Soil organisms as components of ecosystems*. Copenhagen: Munksguaard, 122-132.

Box, J., 2003. Critical factors and evaluation criteria for habitat translocation. *Journal of Environmental Planning and Management*, 46 (6), 839-856.

Bradley, I., Clarke, M., Cooke, H., Harris, J., Leeds Harrison, P., Mayr, T., Towers, W., Rodwell, J., and Gowing, D., 2006. *Guidance on understanding and managing soils for habitat restoration projects*. Peterborough: English Nature Research Projects, Number 712.

Brandani, A.G., Hartshorn, G.S., and Orians, G.H., 1988. Internal heterogeneity of gaps and species richness in Costa Rican tropical wet forest. *Journal of Tropical Ecology*, 4, 99-119.

Brewer, R., 1994. *The Science of Ecology, Second Edition*. Saunders College Publishing., Florida.

Brockmann, W., Koehler, H., and Schriefer, T., 1980. Reculivation of refuse tips: soil ecological studies. *In:* Dindal, D.L. ed., *Soil biology as related to land-use practices*. Environmental Protection Agency., Washington DC, pp. 200-207.

Brose, U., 2001. Relative importance of isolation, area and habitat heterogeneity for vascular plant species richness of temporary wetlands in East-German farmland. *Ecography,* 24 (6), 722-730.

Bullock, J.M., Moy, I., Pywell, R.F., Coulson, S.J., Nolan, A.M. and Caswell, H., 2002. Plant dispersal and colonization processes at local and landscape scales. In: Bullock, J.M., Kenward, R.E. and Hails, R., eds. *Dispersal Ecology*, Blackwell Scientific Publications, Oxford, 279–302.

Bullock, J.M., and Pywell, R. F., 2005. Rhinanthus: A tool for restoring diverse grassland? *Folia Geobotanica*, 40 (2/3), 273-288.

Burke, M.J.W., and Grime, J.P., 1996. An experimental study of plant community invisibility. *Ecology,* 77 (3), 776-790.

Butt, K.R., 1999. Inoculation of earthworms into reclaimed soils: the UK experience, *Land Degradation and Development*, 10 (6), 565-575.

Butt, K.R., 2008. Earthworms in soil restoration: Lessons learned from united Kingdom case studies of land reclamation. *Restoration Ecology*, 16 (4), 637-641.

Butt, K.R., Frederickson, J., and Morris, R.M., 1993. Investigations of an earthworm inoculation experiment, London Borough of Hillingdon. *Waste Planning*, 7, 9-12.

Butt, K.R, Frederickson, J., and Morris, R.M., 1995. An earthworm cultivation and soil inoculation technique for land restoration. *Ecological Engineering*, 4 (1), 1-9.

Butt, K.R, Frederickson, J., and Morris, R.M., 1997. The earthworm inoculation unit (EIU) technique, an integrated system for cultivation and soil-inoculation of earthworms. *Soil Biology and Biochemistry*, 29 (3/4), 251-257.

Butt, K.R., Frederickson, J., and Lowe, C.N., 1999. Colonisation, survival and spread of earthworms on a partially restored landfill site. Pedobiologia 43, 684-690.

Butt, K.R., Lowe, C.N., Frederickson, J., and Moffat, A.J., 2004. The development of sustainable earthworm populations at Calvert Landfill Site, UK. *Land Degradation and Development*, 15 (1), 27-36.

Caldwell, M.M., and Pearcy, R.W., 1994. *Exploitation of environmental heterogeneity in plants: Ecophysical process above and below ground*. Burlington, MA: Academic Press.

Cannon, P. M., and Reid, I., 1993. The influence of relic Ridge-and-Furrow on the soil water regime and the plant community of an ancient meadow, Cumbria, England. *Global Ecology and Biogeographical Letters*, 3 (1), 18-26.

Carpenter, S.R., 1999. Microcosm experiments have limited relevance for community and ecosystem ecology: Reply. *Ecology*, 80 (2), 1085-1088.

Carrington, L.P., and Diaz, A., 2009. An investigation into the effect of soils and vegetation on the successful establishment of a hay meadow on a clay capped landfill. *Restoration Ecology*, in press.

Clark, D.B., 1990. The role of disturbance in the regeneration of tropical moist forests. *In:* Bawa, K.S., and Hadley, M., eEds. *Reproductive ecology of tropical forest plants*. UNESCO., Paris, 291-315.

Clarke, K.R., and Gorley, R.N., 2006. *PRIMERv6*. PRIMER-E Ltd, Plymouth.

Clements, R.O., Murray, P.J., and Sturdy, R.G., 1991. *The* impact of earthworms and three levels of N fertiliser on a grassland soil environment. *Agriculture Ecosystem and Environment*, 36 (1/2), 75-85.

Coleman, D.C., and Crossley, D.A., 2003. *Fundamentals of soil ecology*. London: Academic Press.

Corbett, E., Anderson, R., and Rodgers, C., 1996. Prairie re-vegetation of a strip mine in Illinois: Fifteen years after establishment. *Restoration ecology*, 4 (4), 346-354.

Cotton, D.C.F., and Curry, J.P., 1980. The effects of cattle and pig slurry fertilisers on earthworms (Oligochaeta, Lumbircidae) in grassland managed for sileage production. *Pedobiologia*, 20, 181-188.

Courtney, F.M., and Trudgill, S.T., 1984. *The soil: An introduction to soil study*. Hodder Arnold, London.

Cowles, H. C., 1899. The ecological relations of the vegetation on the sand dunes of Lake Michigan. *Botanical Gazette*, 27 (3), 167-202.

Craven, D.R.J., 1995. The Hallside Steelworks Project. *Land Contamination and Reclamation*, 3, 31-38.

Crofts, A., and Jefferson, R.G., 1999. *The lowland grassland management handbook, 2nd Edition*. Peterborough: English Nature.

Cromsigt, J.P.G.M., and Olff, H., 2006. Resource partitioning among savanna grazers mediated by local heterogeneity: an experimental approach. *Ecology*, 87 (6), 1532-1541.

Curry, J.P., and Cotton, D.C.F., 1983. Earthworms and land reclamation. *In:* Satchell, J.E. ed., *Earthworm Ecology*. Chapman and Hall., London, pp. 167-175.

Curry, J.P., 1988. The ecology of earthworms in reclaimed soils and their influence on soil fertility. *In:* Edwards, C.A., Neuhauser, E.F., eds. *Earthworms in waste and environmental management*. The Hague: SPB Academic, 251-261.

Davidson, D.A., and Grieve, I.C., 2006. The influence of soil fauna on soil structural attributes under a limed and untreated upland grassland. *Land Degradation and Development*, 17 (4), 393-400.

Davies, C., 1973. Soil restoration after industrial exploitation. *In: Aspects of current use and misuse of soil resources*, Welsh Soils Discussion Group, 10, 109-115.

Davis, B.N.K., 1986. Colonisation of newly created habitats by plants and animals. *Journal of Environmental Management*, 22, 361-371.

Department for Communities and Local Government, 2008. *Woodland establishment on landfill sites, ten years of research*. Wetherby: Communities and Local Government Publications.

Department for Environment, Food and Rural Affairs, 2008. *Waste statistics*. Available from: http://www.defra.gov.uk/news/2008. [Accessed 10 July 2008].

Department of the Environment, 1995. *Biodiversity: UK steering group report. Action Plans, volume 2.* HMSO, London.

Diamond, J., 1986. Overview: Laboratory experiments, field experiments and natural experiments. *In:* Diamond, J., and Case, T.J., eds. *Community Ecology*. Harper and Row, New York, 3-22.

Diaz, A., Green, I., and Tibbett, M., 2008. Recreation of heathland on improved pasture using top soil removal and sulphur amendments: Edaphic drivers and impacts on ericoid mycorrhizas. *Biological Conservation*, 141 (6), 1628-1635.

Doane, T.A., and Horwarth, W.R., 2003. Spectophotometric determination of nitrate with a single reagent. *Analytical Letters*, 36 (12), 2713-2722.

Doube, B.M., and Brown, G.G., 1998. Life in a complex community: functional interactions between earthworms, organic matter, micro-organisms and plants. *In:* Edwards. C.A., ed. *Earthworm Ecology*. Boca Raton: St. Lucie Press.

Dryden, R., 1997. *Habitat restoration project: Fact sheets and bibliographies*. Peterborough: English Nature.

Dwyer, L.M., and Merriam, G., 1981. Influence of microtopographic hererogeneity on deciduous litter decomposition. *Okios*, 37, 228-237.

Dwyer, L.M., and Ivarson, K.C., 1983. Influence of woodlot floor topography on microbial decomposer distribution. *Holartic Ecology*, 6, 101-106.

Edwards, A., Mortimer, S.R., Lawson, C.S., Westbury, D.B., Harris, S.J., Woodcock, B.A., and Brown, V.K., 2006. Hay strewing, brush-harvesting of seed and soil disturbance as tools for the enhancement of botanical diversity in grasslands. *Biological Conservation*, 134 (3), 372-382.

Edwards, C.A., and Bohlen, P.J., 1996. *Biology and ecology of earthworms, 3rd edition*. London: Chapman and Hall.

Ehlers., W., 1975. Observations on earthworm channels and filtration in tilled and untilled loess soil. *Soil Science*, 119 (3), 242-249.

Eldridge, D.J., Westoby, M., and Holbrook, K.G., 1991. Soil surface characteristics, microtopography and proximity of mature shrubs: effects on survival of several cohorts of *Atriplex vesicaria* seedlings. *Journal of Ecology*, 79 (2), 357-364.

Environmental Protection Agency., 2008. *Garbage facts.* Available from: www.epa.gov/garbage/facts [accessed on 10 July 2008].

ETSU., 1998. *An introduction to household waste management*. London: Department of Trade and Industry.

European Union., 1979. *On the Conservation of wild birds*, *Council Directive 79/409/EEC*. Brussels: European Union.

European Union., 1992. *On the Conservation of natural habitats and wild fauna and flora*, *Council Directive 92/43/EEC*. Brussels: European Union.

European Union., 1999. *On the landfill of waste*, *Council Directive 1999/31/EEC*. Brussels: European Union.

Evans, E., 1992. *A history of nature conservation in Britain.* Second Edition. London: Routledge.

Ewing, K., 2002. Mounding as a technique for restoration of prairie on a capped landfill in the Puget Sound lowlands. *Restoration Ecology*, 10 (2), 289-296.

Falk, D.A., 1990. Discovering the past, creating the future. *Restoration and Management Notes*, 8 (2), 71-72.

Falk, D.A., Palmer, M.A., and Zedler, J.B., 2006. Restoration ecology in context. *In*: Falk, D.A., Palmer, M.A., and Zedler, J.B., eds. *Foundations of Restoration Ecology*. Washington: Island Press, 257-259.

Farrell, L., 1993. *Lowland heathland: the extent of habitat change.* English nature Science No.1. Peterborough: English Nature.

Findlay, D.C., 1965. *Memoirs of the soil survey of Great Britain: The Mendip District of Somerset.* Harpenden: Rothamsted Experimental Station.

Firbank, L.G., Arnold, H.R., Eversham, B.C., Mountford, J.O., Radford, G.L., Telfer, M.G., Treweek, J.R., Webb, N.R.C., and Wells, T.C.E., 1992. *The potential uses of set-aside land to benefit wildlife*. Wallingford: Institute of Terrestrial Ecology.

Flinn, K.M., 2007. Microsite-limited recruitment controls fern colonization of post-agricultural forests. *Ecology,* 88 (12), 3103-3114.

Foster, D.R., Knight, D.H., and Franklin, J.F., 1998. Landscape patterns and legacies resulting from large, infrequent forest disturbances. *Ecosystems*, 1 (6), 497-510.

Fuller, R.M., 1987. The changing extent and conservation interest of lowland grasslands in England and Wales: a review of grassland surveys 1930-1984. *Biological Conservation*, 40 (4), 281-300.

Gemtos, T.A., and Lellis, T., 1997. Effects of soil compaction, water and organic matter on the initial growth of cotton and sugar beet. *Journal of Agricultural Engineering Research*, 66 (2), 121-134.

Gilbert, O.L., and Anderson, P., 1998. *Habitat creation and repair*. Oxford University Press.

Goldberg, E., 2003. Plantations on ancient woodland sites. *Quarterly Journal of Forestry*, 97 (2), 133-138.

Good, J.E.G., Wallace, H.L., Stevens, P.A., and Radford, G.L., 1999. Translocation of herb-rich grassland from a site in Wales prior to opencast coal extraction. *Restoration Ecology*, 7 (4), 336-347.

Gore, J. A., and Shields, F. D., 1995. Can large rivers be restored? *Bioscience*, 45 (3), 142-152.

Gough, M.W., and Marrs, R.H., 1990a. A comparison of soil fertility between semi-natural; and agricultural plant communities: Implications for the creation of species-rich grassland on abandoned agricultural land. *Biological Conservation*, 51 (22), 83-96.

Gough, M.W., and Marrs, R.H., 1990b. Trends in soil chemistry and floristics associated with the establishment of a low-input meadow system on an arable clay soil in Essex, England. *Biological Conservation*, 52 (2), 135-146.

Grace, J.B., Allain, L., and Allen, C., 2000. Vegetation associations in a rare community type – coastal tallgrass prairie. *Plant Ecology*, 147 (1), 105-115.

Gregory, A.S, and Vickers, A.W., 2003. Effects of amendments on soil structural development in a clay soil-forming material used as a clay cap for landfill restoration. *Soil Use and Management*, 19 (3), 273-276.

Grime, J.P., Hodgson. J.G., and Hunt, R., 2007. *Comparative plant ecology: A functional approach to common British species, Second edition*. Dalbeattie: Castlepoint Press.

Hallows, D.G., 1993. The worms that turn. *Mine and Quarry*, 22, 24-26.

Handel, S.N., Robinson, G.R., Parsons, W.F.J., and Mattei, J.H., 1997. Restoration of woody plants to capped landfills: Root dynamics in an engineered soil. *Restoration Ecology*, 5 (2), 178-186.

Harper, J.L., 1971. Grazing, fertilizers and pesticides in the management of grasslands. *In:* Duffey, E., and Watts, A.S., eds. *The scientific management of animal and plant communities for conservation.* Oxford: Blackwell Scientific Publications, 15-32.

Harris, J.A., Birch, P., and Short, K.C., 1993. The impact of storage of soils during opencast mining on the microbial community strategist theory interpretation. *Restoration Ecology,* 1 (2), 88-100.

Hayes, M.J., Sackville Hamilton, N.R., 2001. *The effect of sward management on the restoration of species-rich grassland: a reassessment of IGER's grassland restoration experiment, Trawsgoed*. Countryside Council for Wales Contract Science Report No. 438, Bangor.

Hayes, M.J., Sackville Hamilton, N.R., Tallowin, J.R.B., Buse, A., Davies, O., 2000. *Methods of enhancing diversity in upland environmentally sensitive area swards*. Report to the Ministry of Agriculture, Fisheries and Food (BD1424), Institute of Grassland and Environmental Research, Aberystwyth.

Heunneke, L.F., Sharitz, R.R., 1986. Microsite abundance of woody seedlings in a South Carolina Cypress-Tupelo swamp. *American Midland Naturalist*, 115 (2), 328-335.

Higgs, E., 2005. The two-culture problem: ecological restoration and the integration of knowledge. *Restoration Ecology*, 13 (1), 159-164.

Hokkanen, P.J., 2006. Environmental patterns and gradients in the vascular plants and bryophytes of eastern Fennoscandian herb-rich forests. *Forest Ecology and Management*, 229 (1), 73-87.

Hoogerkamp, M., Rogaar, H., and Eijsackers, H.J.P., 1983. Effect of earthworms on grassland on recently reclaimed polder soils in the Netherlands. *In:* Satchell, J.E., ed. *Earthworm Ecology: From Darwin to vermiculture*. London: Chapman and Hall, 85-105.

Howe, H.F., 1994. Managing species diversity in tallgrass prairie: assumptions and implications. *Conservation Biology*, 8 (3), 691-704.

Hunt, R.J., Krabbenhoft, D.P., and Anderson, M.P., 1997. Assessing hydrogeochemical heterogeneity in natural and constructed wetlands. *Biogeochemistry*, 39 (3), 271-293.

Huston, M.A., 1994. *Biological diversity. The co-existence of species on changing landscapes*. Cambridge University Press.

Janssens, F., Peeters, A., Tallowin, J.R.B, Bekker, J.P.R.M., and Fillat, F., 1998. Relationships between soil chemical factors and grassland diversity. *Plant and Soil*, 202 (1), 69-78.

Jetsch, F., Milton, S.J., Dean, W.R.J., Van Rooyen, N., and Moloney, K.A., 1998. Modelling the impact of small-scale heterogeneities on tree-grass coexistence in semi-arid savannas. *Journal of Ecology*, 86 (5), 780-793.

Johnston, J.R., 1945. An accurate method for determining the volume of soil clods. *Soil Science*, 59 (6), 449-452.

Joint Nature Conservation Committee., 1995. *Biodiversity: the UK steering group Report, Volume 2*. Peterborough: Joint Nature Conservation Committee.

Joint Nature Conservation Committee., 2006. *Habitat Targets: Revised 2006*. Available from: http://www.ukbap.org.uk/BAPGroupPage. [Accessed 25 August 2009].

Jongepierová, I., Mitchley, J., and Tzanopoulos, J., 2007. A field experiment to recreate species rich hay meadows using regional seed mixtures. *Biological Conservation*, 139, 297-305.

Judd, K.W., and Mason, C.F., 1995. Earthworm populations of a restored landfill site. *Pedobiologia*, 39, 107-115.

Kershaw, K.R., Mitchley, J., Buckley, G.P., and Helliwell, D.R., 1995. Slope protection and establishment of vegetation on Channel Tunnel spoil in an environmentally sensitive coastal site. *In:* Barker, D.H., ed. *Vegetation and slopes*. London: Institute of Civil Engineers, 117-126.

Kirby, K.J., and Thomas, R.C., 1995. Pasture woodland and its conservation in Britain. *Biological Journal of the Linnean Society*, 56, 135-153.

Jones, A.T., and Hayes, M.J., 1999. Increasing floristic diversity in grassland: the effects of management regime and provenance on species on species introduction. *Biological Conservation* 87 (3), 381-390.

Larkin, D., Vivian-Smith, G., and Zedler, J.B., 2006. Topographic heterogeneity theory and ecological restoration. *In:* Falk, D.A., Palmer, M.A., and Zedler, J.B., eds. *Foundations of Restoration Ecology*. Washington: Island Press, 142-164.

Lavelle, P., Bignell, D., Lepage, M., Wolters, V., Roger, P., Ineson, P., Heal, O. W., and Dhillion, S., 1997. Soil function in a changing world: the role of invertebrate ecosystem engineers. *European Journal of Soil Biology*, 33 (44), 159-193.

Lawrence, A.P., and Bowers, M.A., 2002. A test of the ''hot' mustard extraction method of sampling earthworms. *Soil Biology and Biochemistry*, 34 (4), 549-552.

Leroy, B.L.M., Schmidt, O., Van den Bossche, A., Reheul, D., and Moens, M., 2008. Earthworm population dynamics as influenced by the quality of exogenous organic matter. *Pedobiologia*, 52, 139-150.

Leuschner, C., and Lendzion, J., 2009. Air humidity, soil moisture and soil chemistry as determinants of the herb layer composition in European beech forests. *Journal of Vegetation Science*, 20 (2), 288-298.

Lipeic, J., Hatano, R., and Slowinska-Jurkiewicz, A., 1998. The fractal dimension of pore distribution patterns in variously compacted soil. *Soil and Tillage Research*, 47 (1/2), 61-66.

Lloyd, A., 1984. Soil management problems on opencast sites. *Proceedings of North of England Soil Discussion Group*, 19, 19-35.

Lloyd, P.S., and Pigott, C.D., 1967. The influence of soil conditions on the course of succession on the chalk of southern England. *Journal of Ecology*, 55 (1), 137-146.

Manchester, S.J., Treweek, J.R., Mountford, J.O., and Pywell, R.F., 1997. The success of nurse crops in the establishment of species-rich lowland wet grassland. *In:* Sheldrick, R.D., ed., *Grassland management in Environmentally Sensitive Areas British Grassland Society Occasional Symposium, Volume 32.* Reading: British Grassland Society, 260-261.

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.

Marinissen, J.C.Y., 1994. Earthworm populations and stability of soil structure in a silt loam soil of a recently reclaimed polder in the Netherlands. *Agriculture Ecosystems and Environment*, 51, 75-87.

Marinissen, J.C.Y., and Van den Bosch, F., 1992. Colonization of new habitats by earthworms. *Oecologia* 91, 371-376.

Marrs, R.H., 1993. Soil fertility and nature conservation in Europe: theoretical considerations and practical management solutions. *Advances in Ecological Research*, 24, 241-300.

Martin, A.D., Humphries, R.N., and Whittington, W.J., 1998. Midland research project – invertebrate studies. *In:* Land restoration investigation techniques. Mansfield: British Coal Opencast Executive, 47-56.

Mather, J.G., and Christensen, O., 1992. Surface migration of earthworms in grassland. *Pedobiologia*, 36, 51-57.

Mato, S., Trigo, D., and Diaz-Cosin, D. J., 1989. Influence of parent matter and vegetation on types on earthworms. *Agrobiol*, 48, 39-48.

McCrea, A., Trueman, I., and Fullen (2001), A comparison of the effects of four arable crops on the fertility depletion of a sandy sit loam destined for grassland habitat creation. *Biological Conservation*, 97 (2) 181-187.

McRae, S.G., 1989. The restoration of mineral workings in Britain – a review. *Soil Use and Management*, 5 (3), 135-142.

Menninger, H.L., and Palmer, M.A., 2006. Restoring ecological communities: from theory to practice. *In:* Falk, D.A., Palmer, M.A., Zedler, J.B., eds. *Foundations of Restoration Ecology*. Washington: Island Press, 88-112.

Mitchley, J., Buckley, G.P., and Helliwell, D.R., 1996. Vegetation establishment on chalk marl spoil: the role of nurse grass species and fertiliser application. *Journal of Vegetation Science*, 7, 543-548.

Miranda, K.M., Esprey, M.G., Wink, and D.A., 2001. A rapid, simple spectrophotometric method for simultaneous detection of nitrate and nitrite. *Nitric Oxide Biology and Chemistry*, 5 (1), 62-71.

Mladenoff, D.J., White, M.A., Pastor, J., and Crow, T.R., 1993. Comparing spatial pattern in unaltered old growth and disturbed forest landscapes. *Ecological Applications*, 3 (2), 294-306.

Moffatt, A.J., 1995. Minimum soil depths for the establishment of woodland on disturned ground. *Arboricultural Jounral*, 19, 19-27.

Mortimer, S.R., Booth, R.G., Harris, S.J., and Brown, V.K., 2002. Effects of initial site management on the Coleoptera assemblages colonising newly established chalk grassland on ex-arable land. *Biological Conservation*, 104 (3), 301-313.

Moser, K., Ahn, C., and Noe, G., 2007. The influence of microtopography on soil nutrients in created mitigation wetlands. *Restoration Ecology*, 17 (5), 641-651.

Moser, K.H, Ahn, C., and Noe, G.B, 2009. Characterisation of microtopography and its influence on vegetation patterns in created wetlands. *Wetlands*, 27 (4), 1081-1097.

Norris, K., 2008. Agriculture and biodiversity conservation: opportunity knocks. *Conservation Letters*, 1, 2-11.

Núñez-Farfan, J., and Dirzo, R., 1988. Within-gap heterogeneity and seedling performace in a Mexican tropical forest. *Okios*, 51, 174-284.

Office of the Deputy Prime Minister, 2005. *Planning Policy Statement 9: Biodiversity and Geological Conservation.* HMSO, London.

O'Flanagan, W.C.,Walker, G.J., Waller, M.W., and Murdoch, G., 1963. Changes taking place in topsoil stored in heaps on opencast sites. *National Agricultural Advisory Service Quarterly Review*, 62, 85-92.

Omerod, S.J., 2003. Restoration in applied ecology: editor's introduction. *Journal of Applied Ecology*, 40, 44-50.

Otte, A., Simmering, D., and Wolters, V., 2007. Biodiversity at the landscape level: recent concepts and perspectives for multifunctional land use. *Landscape Ecology,* 22 (5), 639-642.

Ovenden, G.N., Swash, A.R.H., and Smallshire, D., 1998. Agri-environment schemes and their contribution to the conservation of biodiversity in England. *Journal of Applied Ecology*, 35, 955-960.

Pacala, S.W., and Tilman, D., 1994. Limiting similarity in mechanistic and spatial models of plant competition in heterogeneous environments. *The American Naturalist*, 143 (2), 222-257.

Pakeman, R.J., Pywell, R.F., and Well, T.C.E., 2002. Species spread and persistence: implications for experimental design and habitat creation. *Applied vegetation science*, 5, 75-86.

Palmer, M.A., Bernhardt, E., Allan, J.D., Alexander, G., Brooks, S., Clayton, S., Carr, J., Dahm, C., Follstad-Shah, J., Galat, D.L., Gloss, S., Goodwin, P., Hart, D., Hassett, B., Jenkinson, R., Kondolf, G.M., Lake, S., Lave, R., Meyer, J.L., O'Donnell, T.K., Pagano, L., Srivastava, P., and Sudduth, E., 2005. Standards for ecologically successful river restoration. *Journal of Applied Ecology*, 42 (2), 208-217.

Parker, D.M., 1995. *Habitat creation – a critical guide, English Nature Science No. 21*. Peterborough: English Nature.

Paz Gonzàlez, A., Viera, S.R., and Taboada Castro, M.T., 2000. The effect of cultivation on the spatial ability of selected properties of an umbric horizon. *Geoderma*, 97 (3/4), 273-292.

Pfadenhauer, J., 2001. Some remarks on the socio-cultural background of restoration ecology. *Restoration Ecology*, 9 (2), 220-222.

Pizl, V., 1999. Earthworm succession in abandoned fields – a comparison of deductive and sequential approaches to study. *Pedobiologia*, 43, 705-712.

Pywell, R.F., Bullock, J.M., Hopkins, A., Walker, K.J., Sparks, T.H., Burke, M.J.W., and Peel, S., 2002. Restoration of species-rich grassland on arable land: assessing the limiting processes using a multi-site experiment. *Journal of Applied Ecology*, 39 (2), 294-309.

Pywell, R.F., Bullock, J.M., Roy, D.B., Warman, L.I.Z., Walker, K.J., and Rothery, P., 2003. Plant traits as predictors of performance in ecological restoration. *Journal of Applied Ecology*, 40 (1), 65-77.

Rackham, O., 1986. *The history of the countryside: The classic history of Britain's landscape, flora and fauna.* Pheonix, London.

Ratcliffe, D.A., 1984. Post-medieval and recent changes in British vegetation: the culmination of human influence. *New Phytologist*, 98 (1), 73-100.

Reader, R. J., and Buck, J., 1991. Community response to experimental soil disturbance in a mid-successional abandoned pasture. *Vegetatio*, 92 (2), 151-159.

Robinson, C.H., Piearce, T.G., Ineson, P., Dickson, D.A., and Nys, C., 1992. Earthworm communities of limed coniferous soils: field observations and implications for management. *Forest Ecology and Management*, 55 (1), 117-134.

Rodwell. J.S., 1992. *British Plant Communities, Volume 3: Grasslands and montane communities*. Cambridge University Press.

Rorison, I.H., 1971. The use of nutrients in the control of the floristic composition of grassland. *In*: Duffey. E., and Watt, A.S., eds*., The Scientific management of animal and plant communities for conservation.* Oxford: Blackwell Scientific Publications, 65-77.

Rosenzweig, M.L., 1995. *Species diversity in space and time*. Cambridge University Press.

Rowell, D.L., 1994. *Soil Science: Methods and Applications*. Longman Scientific and Technical, Harlow.

Rundlőf, M., and Smith, H.G., 2006. The effect of organic farming on butterfly diversity depends on landscape context. *Journal of Applied Ecology*, 43 (6), 1121-1127.

Rushton, S.P., 1986. The effects of soil compaction on Lumbricus terrestris and its possible implications on land reclaimed from open-cast coal mining. *Pedobiologia*, 29, 85-90.

Scullion, J., 1994. *Restoring farmland after coal, The Bryngwyn project.* Mansfield: British Coal Opencast.

Scullion, J., and Mohammed, A.R.A., 1991. The effect of cultivation on the structure in soils affected by opencast mining for coal. *Journal of the Science of Food and Agriculture*, 55 (2), 327-339.

Scullion, J., Mohammed, A.R.A., and Richardson, H., 1998. Effects of storage and reinstatement procedures on earthworm populations in soils affected by opencast mining. *Journal of Applied Ecology*, 25 (1), 223-240.

Simmons, E., 1999. Restoration of landfill sites for ecological diversity. *Waste Management Research*, 17 (6), 511-519.

Sims, R.W., and Gerard, B.M., 1999. *Earthworms*. Shrewsbury: Field Studies Council.

Siriwardena, G.M., Baillie, S.R., Buckland, S.T., Fewster, R.M., Marchant, J.H., and Wilson, J.D., 1998. Trends in the abundance of farmland birds: a quantitative comparison of smoothed Common Birds Census indices. *Journal of Applied Ecology*, 35, 24-43.

Smith, R.S., Shiel, R.S., Millward, D. and Corkhill, P., 2000. The interactive effects of management on the productivity and plant community structure of an upland meadow: an 8-year field trial. *Journal of Applied Ecology* 37, 1029–1043.

Snow, C.S.R., Marrs, R.H., and Merrick, L., 1997. Trends in soil chemistry and floristics associated with the establishment of a low-input meadow system on an arable clay soil in Essex. *Biological Conservation*, 79 (1), 35-41.

Spencer, J.W., and Kirby, K.J., 1992. An inventory of ancient woodland for England and Wales. *Biological Conservation*, 62 (1), 77-93.

Stanford, J.A., Ward, J.V., Liss, W.J., Frissell, C.A., Williams, R.N., Lichatowich, J.A., and Coutant, C.C., 1996. A general protocol for restoration of regulated rivers. *Regulated Rivers: Research and Management*, 12, 391-413.

Stenrod, M., Charnay, M., Benoit, P., and Eklo, O.M., 2006. Spatial variability of glyphosate mineralization and soil microbial characteristics in two Norwegian sandy loam soils as affected by surface topographical features. *Soil Biology and Biochemistry*, 38 (5), 962-971.

Stevenson, M.J., Ward, L.K., and Pywell, R.F., 1997. Re-creating semi-natural communities: vacuum harvesting and hand collection of seed on calcareous grassland. *Restoration Ecology*, 5, 66-76.

Stewart, V.I., 1981. *The productivity of restored land*. London: Land Decade Educational Council.

Stockdill, S.M.J., 1982. Effects of introduced earthworms on the productivity of New Zealand pastures. *Pedobiologia*, 24, 24-35.

Surrey County Council, 2008. *Landfill sites*. Available from: http://www.surreycc.gov.uk., [Accessed 12 September 2008].

Sutherland, W.J., 2002. *Ecological census techniques: A handbook*. Cambridge University Press.

Sutherland, W.J., and Hill, D.A., 1995. *Managing habitats for conservation*. Cambridge University Press.

Svenning, J.C., and Skov, F., 2002. Mesoscale distribution of understorey plants in temperate forest (Kalo Denmark): the importance of environment and dispersal. *Plant Ecology*, 160 (2), 169-185.

Svensden, J.A., 1955. Earthworm population studies, a comparison of sampling methods. *Nature*, 175, 184.

Temperton, V.M., Hobbs, R.J., Nuttle, T., Fattorini, M., and Halle, S., 2004. Introduction: Why assembly rules are important to the field of restoration ecology. *In:* Temperton, V.M., Hobbs, R.J., Nuttle, T., and Halle, S. eds. *Assembly rules and restoration ecology: Bridging the gap between theory and practice*. Washington: Island Press, 1-8.

Teotia, S.P., Dudley, F.L., and McCalla, T.M., 1950. The effect of stubble mulching on number and activity of earthworms. *Nebraska Agricultural Experimental Station Research Bulletin*, 165, 20.

Tessier, M., Gloaguen, J. C., and Bouchard, V., 2002. The role of spatio-temporal heterogeneity in the establishment and maintenance of Suaeda maritime in salt marshes. *Journal of vegetation science*, 13 (1), 115-122.

Thiere, G., Milenkovski, S., Lindgren, P., Sahlén, G., Berglund, O., and Weisner, S.E.B., 2009. Wetland creation in agricultural landscapes: Biodiversity benefits on local and regional scales. *Biological Conservation*, 142 (5), 964-973.

Thrush, S.F., Hewitt, J.E., Cummings, V.J., Green, M.O., Funnell, G.A., and Wilkinson, M.R., 2000. The generality of field experiments: Interactions between local and broad-scale processes. *Ecology*, 81 (2), 399-415.

Thurston, J.M., 1969. The effect of liming and fertilisers on the botanical composition of permanent grassland, on the yield of hay. *In:* Rorison, I.H., ed., *Ecological aspects of the mineral nutrition of plants*. Oxford: Blackwell Science Publications, 3-10.

Tibbett, M., and Diaz. A., 2005. Are sulphurous soil amendments ($S^{o,}$ $Fe^{(II)}$ SO_4, $Fe^{(III)}SO_4$) an effective tool in the restoration of heathland and acidic grassland after four decades of rock phosphate fertilisation? *Restoration Ecology*, 13 (1), 83-91.

Tilman, D., 1982. Resource competition and community structure. *Monographs in Population Biology*, 17, 1-296.

Tilman, D., 1989. Ecological experimentation: strengths and conceptual problems. *In:* Likens, G.E., ed. *Long-term studies in Ecology: Approaches and Alternatives*. New York: Springer-Verlag, 136-157.

Tilman, D., Lehman, C.L., and Bristow, C.E., 1998. Diversity-stability relationships: A Statistical inevitability or ecological consequence? *American naturalist*, 151(3), 277-282.

Toth, L.A., Obeysekera, J.T.B., Perkins, W.A., and Loftin, M.K., 1993. Flow regulation and restoration of Florida's Kissimmee River. *Regulated Rivers: Research and Management*, 8, 155-166.

Townsend, D., Stace, H., and Radley, D., 2004. *State of nature: Lowlands – future landscapes for wildlife*. Peterborough: English Nature.

Van den Bergh, J.P., 1979. Changes in thecomposition of mixed populations of grassland species. *In:* Werger, M.J.A., ed., *The study of vegetation*. The Hague: Junk, 59-80.

Van Dijk, G., 1991. The status of semi-natural grasslands in Europe. *In:* Goriup, P.D., Batten, L.A., and Norton, J.A., eds. *The conservation of lowland dry grassland birds in Europe*. Joint Nature Conservation Committee, Peterborough.

Van Rhee, J.A., 1969. Development of earthworm populations in polder soils. *Pedobiologia*, 9, 133-140.

Vivian-Smith, G., 1997. Microtopographic heterogeneity and floristic diversity in experimental wetland communities. *Journal of Ecology*, 85 (1), 71-82.

Wagner, H.H., Wildi, O., and Ewald, K.E., 2000. Additive partitioning of plant species diversity in an agricultural mosaic landscape. *Landscape Ecology*, 15 (3), 219-227.

Wang, G.M., Hobbs, N.T., Boone, R.B., Illius, A.W., Gordon, I.J., Gross, J.E., and Hamlin, K.L., 2006. Spatial and temporal variability modify density dependence in populations of large herbivores. *Ecology*, 87 (1), 95-102.

Warren, J.M., 2000. The role of white clover in the loss of diversity in grassland habitat restoration. *Restoration Ecology*, 8 (3), 318-323.

Waste Recycling Group, 2009. *Wellbeck landfill site: Facts and Figures*. http://www.wrg.co.uk/wellbeck/facts.asp [accessed 30 June 2009].

Watson, D., and Hack, V., 2000. *Wildlife management and habitat creation on landfill sites, A manual of best practice*. Cambridge: Ecoscope Applied Ecologists.

Wells, T.C.E., Cox, R., and Frost, A., 1986. *Wildflower grasslands from crop grown seed and hay banks*. Peterborough: Nature Conservancy Council.

Wells, T.C.E., Cox, R., and Frost, A., 1989. *The establishment and management of wildflower meadows*. Peterborough: Nature Conservancy Council.

Wheater, C.P., and Cook, P.A., 2000. *Using statistics to understand the environment*. London: Routledge.

Whittaker, R.H., 1960. Vegetation of the Siskiyou Mountains, Oregon and California. *Ecological Monographs*, 30 (3), 279-338.

Wilson, S.D., 2000. Heterogeneity, diversity and scale in plant communities. *In:* Hutchings, M.J., John, E.A., and Stewart, A.J.A., eds., *The Ecological consequences of environmental heterogeneity*. Oxford: Blackwell Science, 53-69.

Woodcock, B.A., Pywell, R., Roy, D.B., Rose, R., and Bell, D., 2005. Grazing management of calcareous grasslands and its implications for the conservation of beetle communities. *Biological Conservation*, 125, 192-202.

Woodcock, B.A., Edwards, A.R., Lawson, C.S., Westbusry, D.B., Brook, A.J., Harris, S.J., Brown, V.K., and Mortimer, S.R., 2008. Contrasting success in the restoration of plant and phytophagus beetle assemblages of species-rich mesotrophic grasslands. *Oecologia*, 154, 773-783.

Worcestershire County Council, 2009. *Waste management*. http://www.worcestershire.gov.uk/home/wcc-waste-landfill [accessed 30 June 2009].

Yelling, J.A., 1977. *Common field and enclosure in England, 1500-1850*. London: Macmillan.

Young, T.P., 2000. Restoration ecology and conservation biology. *Biological Conservation*, 92 (1), 73-83.