- CHAPTER 11 -
CONCLUDING REMARKS

I grieve, when on the darker side
Of this great change I look; and there behold
Such outrage done to nature as compels
The indignant power to justify herself;
Yea, to avenge her violated rights.
For England's bane.
Wordsworth W. (1814) From *The Excursion* viii II: 151-5.

SUMMARY

1. The results of the Countryside Survey 2000 confirm that during the 1990s there has been a continuing deterioration in grassland biodiversity. In response most conservation organisations are lobbying to change the Common Agricultural Policy. They argue that the CAP must be at least 'greened', if not totally rebuilt. In concord, opinion of the restoration ecology community seems to be that the conservation of extant biodiverse habitats still remains of the highest priority. However, restoration does seem to be becoming an important instrument in meeting biodiversity targets obliged by international treaties. Important grassland restoration initiatives are formulated within two key, broadly compatible, national landscape conservation strategies: the UK Biodiversity Action Plan, and agri-environment schemes.

2. The results from LHF can be approximately divided into restoration strategies relying mostly on *natural* processes, and strategies utilising artificial enrichment methods. In comparison to the effort expended, the restoration gains in plant biodiversity at LHF seem rather minimal. The main environmental and botanical indicators used throughout the research indicate only minor positive changes in 2001 compared to the start of the project in 1994. Therefore, in contrast, a g*round zero* approach seems to have many immediate attractions. Turf and soil stripping massively reduce fertility, and undesirable weedy and competitive species are largely removed. However, such conservation reseeds are of very limited agricultural use due to very low productivity. The other route is to continue managing improved permanent grasslands without intensive inputs. Trends suggest that the concept of low-input permanent grasslands may be having a renaissance. The soils sequester carbon which would otherwise contribute to climate change. Permanency may also allow the build-up of more complex ecosystems, perhaps leading to higher resilience to perturbations, and greater long-term sustainable productivity.

3. With hindsight would we have altered our choice of restoration methods? In terms of the *natural* strategy, there is no doubt that reinstating traditional management is an essential mechanism. All other diversifying approaches must be organised within this framework. However, the findings strongly suggest that the seed banks of improved grasslands do not have much restorative value. Equally, attempting to diversify swards by creating herbicide gaps to manipulate recruitment seems basically ineffectual. As for artificial enrichment, the evidence suggests that at the beginning of the project in 1994, *all* the improved grasslands were probably too productive to attempt inoculation using a broad-spectrum of species. If the scheme had been purely to diversify the swards rather than to establish an integrated research project, then this aspect of enrichment must be seen as somewhat of a failure. Even though there were some anomalies, choice of transplant species should be probably restricted to a core selection, and a few additional intermediates.

4. Taking cost into account, inoculating the core species using seed rather than transplants seems to be the most efficient method. However, unlike the transplants, seed needs to be sown into artificial gaps to be successful. However, stripping a relatively small area, equivalent to the size of the plots (between 10-20% of the fields) and re-seeding with appropriate species-rich seed mixtures, would seem the acceptable compromise. The majority of the productive capacity of the grasslands would be ensured, whilst the lesser proportion would establish a nutrient impoverished, species-rich hub from which desirable species could disperse.

5. The overall, but qualified goal of the research, was to restore the improved grasslands to an approximation of the reference community. Obviously, the maximal goal of reassembling grassland typical of the species-rich MG 5 meadow type has clearly not been achieved, and even the *medial* goals have not been accomplished. The final account after seven years of restoration management allows that only minimal goals have been achieved e.g. non-significant reductions in productivity and increases in floristic diversity. Even with considerable resource input and relatively long timescales, restoration of eutrophied grasslands may involve periods of developmental stasis. These systems seem to be surprisingly resistant to conservation management. There was never a simple assumption that the experimental grasslands necessarily could be restored to reference community equivalence after five, seven, or even ten years. The established facts suggest that the LHF restoration project is still in the *initial restoration phase*. Far from a linear process, restoration after agricultural improvement may develop in unexpected ways. Consequently, it is important in the planning stage to develop multiple or ranked restoration objectives.

Keywords: Countryside Survey 2000, biodiversity deterioration, conservation, grassland restoration BAP, agri-environment scheme, natural, artificial, soil-stripping, *ground zero*, permanency, goal, reference community, timescale, initial restoration phase, restoration planning, multiple objectives.

GRASSLAND RESTORATION IN THE UK: FUTURE TRENDS

The results of the Countryside Survey 2000 (Haines-Young *et al* 2000) confirm that during the 1990s there has been a continuing deterioration in biodiversity, not only in the national stock of unimproved neutral grassland, but even agriculturally improved grasslands have experienced significant declines in plant diversity. Between 1990-98, two-fifths of neutral grassland was agriculturally improved (Haines-Young *et al* 2000). Though a degree of these losses were made up by transfers from other broad habitats to neutral grassland, the Council for the Protection of Rural England (CPRE) calculate the net national reductions in permanent grassland to be 126,000ha, and in the south-west alone 40,300ha (Everett 2001). The Countryside Survey also indicates that there have been widespread deleterious changes at a more indirect level, with many grasslands remaining safe from immediate cultivation to arable, but undergoing declines in biodiversity 'quality' due to continued over-use of artificial fertilisers, leading to dominance of common vigorous productive plants over more stress-tolerant species. In particular, the data show that the remaining pockets of less improved grassland within intensively managed pastoral landscapes are especially threatened by continuing eutrophication.

Within this stark context, the struggle to halt declines in biodiversity seems almost as unforgiving as any period since Britain joined the Common Agricultural Policy (CAP) in 1972 (Blackstock *et al* 1999). Pressure to change the CAP regime developed into a groundswell throughout the 1990s, such that now even the more conservative conservation organisations seem to agree that the fundamental strategy to halt and reverse declines in species and habitat biodiversity is through the 'greening' of the Common Agricultural Policy, if not its total rebuild (Green & Burnham 1989; Green 1990; Nixon 1999; Lyster 2000, 2002; BCG 2001; RSPB 2001; UKBAP 2001). In accordance, the common opinion of the restoration ecology community seems to be that the conservation of extant biodiverse habitats still remains of the highest priority (Ehrenfeld 2000; Flora Locale 2000; Macdonald & Johnson 2000; SER 2000; Young 2000; Jones 2001). However, restoration does seem to be becoming an important instrument in meeting targets for environmental and sustainable land use obliged by national and international treaties (Young 2000). Within the present structure, there are two key, broadly compatible, government initiatives set up to encourage nature conservation – the UK Biodiversity Action Plan (1994), and agri-environment schemes

(DEFRA 2002). Summary evaluation of these initiatives identifies habitat restoration as a component of present and future UK biodiversity planning.

Biodiversity Action Planning

Biodiversity Action Planning is the process developed after 1992 when the then Conservative UK Government signed '*The Convention on Biological Diversity*' treaty at the "Earth Summit" in Rio de Janeiro. The tangible outcome of this accord for the UK was the publishing in 1994 of "*Biodiversity – the UK Action Plan (the UK BAP)*" a government paper which outlined how the UK would contribute to global biodiversity conservation over the following 20 years (Entec 2000). The core statement of the UK BAP appropriate to restoration ecology was the commitment:

"*To conserve and enhance biodiversity within the UK and to contribute to conservation of global biodiversity through all appropriate mechanisms*."

The inclusion of the term "enhance" can be reasonably taken as a directive for the broad panoply of UK environmental organisations to investigate and undertake restoration works to increase the national biodiversity resource. The UK Biodiversity Steering Group (1995) report '*Meeting the Rio Challenge*' increased clarification as to how biodiversity action planning should meet the targets for key species and habitats. The checklist included both "special habitat management and restoration", and "habitat creation". In terms of grassland restoration, the 1998 UKBG report set out guidance for habitat restoration and expansion. The main techniques identified to meet these broad objectives were cessation of nutrient inputs; seeding and "turfing" with wild species; and arable reversion. In concrete terms, the objective set out in 1998 was the "attempt to re-establish 500-ha of lowland hay meadow of wildlife value at carefully targeted sites by 2010." The overriding concern was to use restoration as a means to counteract the negative effects of small patch size, fragmentation and isolation. These grand statements of intent are meant to be implemented at the regional and local scale through the tiers of local government. LHF and environs are encompassed within the South Somerset District Council's (1999) "*Strategy for Nature Conservation in South Somerset*" report, which specifies five key broad habitats for special consideration, of which lowland neutral hay meadows and pastures are one. The SSDC *Biodiversity Action Plan* (1998) is the working document guiding the specific approaches to the conservation hay meadows and pastures. Restoration seems to have a key place in the management directives, with explicit mention that restorative management on semiimproved neutral grasslands should be promoted, with the target to expand the area of semi-natural grassland by 20% by 2008, in line with the regional plan which declares the ambition to "restore and expand the quantity and quality of the neutral grassland resource by linking and buffering existing sites.

Agri-environment schemes

The second major conduit of restoration activity is through the distribution of financial incentives to encourage environmentally sensitive forms of agriculture. Mitchley *et al* (1998) consider agri-environmental schemes as the, "key mechanism for reaching biodiversity objectives…[and] restoration of species and habitats in the wider countryside." Clark & Baldock (1995) on behalf of the RSNC (The Wildlife Trusts) proposed that an extension of agri-environment schemes was the only way to achieve the goals set by the UK Biodiversity Action Plan (1994). However, Bignal *et al* (2001) consider that agri-envionment measures have had only slim successes, largely in introducing an element of biological diversity into already impoverished landscapes such as arable *agri-deserts*.

The main agri-environment schemes are Environmentally Sensitive Area (ESA) Countryside Stewardship (CS) designations (Pakeman *et al* 2002). Manchester *et al* (1999) assert that these schemes offer an "unprecedented opportunity for the rehabilitation, restoration or re-creation of threatened habitats". The Environmentally Sensitive Areas scheme was introduced in 1987 by MAFF in order to encourage farmers to protect areas of particular landscape, wildlife and historic interest. It is a voluntary, delivering payments to landowners who agree to undertake beneficial management agreements. Under the ESA schemes, farmers and agricultural land managers with land within one of the 22 designated ESA boundaries are able to enter 10 year management agreements with DEFRA. In Somerset there are three designated areas: the Somerset Levels and Moors, Exmoor, and the Blackdown Hills (DEFRA 2002). Each ESA has one or more tier of entry, of which Tier 3 payments are made for habitat re-creation and restoration. Each ESA has one or more tiers of entry and each tier requires different agricultural practices to be followed. Typically, higher tiers have higher payment rates than the base tier, but impose more conditions on farmers and achieve greater environmental benefits (DEFRA 2002). Tier 1 payments impose environmental restrictions on management of permanent grasslands, such as fertiliser reductions. Payment schemes, usually Tier 2-4, pertinent to grassland restoration are summarised below (DEFRA 2002):

- Arable reversions to permanent grassland.
- Regeneration to extensive pastures and hay meadows.
- Reversions of improved grassland to extensive permanent grassland and/or rough grazings.

As apparent from the above payment descriptions, restoration management seems to be more implied in ESA designations rather than explicitly directed. Notably, the RSPB (2001) is concerned that most of the uptake is in the lower tiers, avoiding restoration prescriptions. One is that biodiversity objectives are not promoted strongly enough within ESAs. Also they criticise the fact that ESAs objectives are not specifically linked to the Biodiversity Action Plan.

In comparison, the Countryside Stewardship Scheme takes a more prescriptive approach, and is considered by the Government as its "principal scheme for conserving and improving the countryside", including the clear purposes to "enhance, restore and recreate targeted landscapes, their wildlife habitats and historical features" (DEFRA 2001). The stated objectives of CS (MAFF 2001) include:

- Improve and extend wildlife habitats.
- Restore neglected land or feature.
- Create new habitats and landscapes.

Working within the BAP framework, each county has prioritised landscape conservation areas and targets, and, as usual, allocated limited funds. This means that all those that manage land are effectively eligible to apply, but only those which meet priority schedules can expect to receive the 10-year payment agreements. Two landscape categories, "chalk and limestone grassland", and "old meadows and pastures" are particularly relevant to grassland restoration.

Major non-governmental organisations

In terms of a combination of resources, clout and commitment, the RSPB has set the agenda for restoration. The organisation is not only pushing the Government to attain BAP targets, but also has the ambition to substantially increase the targets (RSPB 2001), even while the present BAP objectives have not been accepted by central government as statutory goals. The RSPB state that habitat restoration "is vital to replace lost habitat and to reduce fragmentation…provide habitat 'stepping stones' to allow wildlife to adjust to the effects of climate change." Not surprisingly the restoration effort is concentrated on habitats which are profitable to bird conservation, and lowland hay meadows and pastures which are not 'wet', are not of strategic importance for lowland birds. However, the RSPB is disposed towards restoration of interstitial hay and pasture grasslands to enhance and buffer primary bird habitats (RSPB 2001).

Whilst not specifically mentioning restoration, the National Trust subscribes to a commitment to "maintaining a high quality environment, particularly in terms of landscape and environment" (Nixon 1999). The Trust considers agri-environment schemes as absolutely vital to sustaining biodiversity in the countryside; "farmers should be encouraged and rewarded for managing the land for a variety of benefits [including] enhancement of nature conservation [and] maintenance of high quality landscapes" (Nixon 1999). This is a particular consideration to the National Trust which has a landholding of nearly ¼ million hectares, most of which is farmed under tenancy agreements and commoner rights. Apart from the occasional controversial statement (Jarman 1995), The National Trust does not hold a particularly pro-active stance to restoration, and is most keen on the coalescence of the agri-environment schemes (ESAs and CS) into a single fund with greater, sustained funding to advance conservation priorities including grassland restoration.

The Wildlife Trusts, as a loose federation, have a considerable landholding and have through the central mouthpiece – The Wildlife Trusts – have an influential voice. The Wildlife Trusts, both at a central level and at the individual county level, have been instrumental along with statutory bodies such as English Nature, JNCC, MAFF (now DEFRA) and Countryside Commission in the numerous builds of the UK BAP since the signing of the 'Rio' convention. Because the county trusts are so bound up with BAP objectives, as described above, they have solid commitments to grassland restoration targets. As far back as 1988 (Clark & Baldock 1995), the then RSNC stated that, "there should be a more adventurous approach to the creation and restoration of habitat…[there] should be a programme of creative conservation aimed at the establishment of additional habitat and natural features within farms, the restoration of regionally important tracts of countryside and the resurgence of 'ecological halos' around towns and cities." More recently, the president of the Wildlife Trusts, Simon Lyster (2000) made it clear that the commitment of the organisation(s) to restoration is comprehensive: "Throughout the UK, Wildlife Trusts are restoring grasslands, wetlands, woodlands and other habitats. This work will, in many ways, be the focus for conservation in this century." However, because funds are prescribed and limited, the understandable necessity is for protection of the remaining semi-natural, species-rich grasslands rather than spreading monies thinly to enhance agriculturally improved grasslands as well, even if buffering priority grasslands. If not through major CAP reforms, the governmental agri-environment schemes have to have substantial funding augmentations for the Wildlife Trusts to be able to proceed with extensive grassland restoration objectives identified in the UK BAPs.

The above summary of the main national organisations and schemes driving landscape conservation shows that grassland restoration is considered not only by conservation NGOs, but also government departments, as a key component in meeting international environmental obligations. Grassland restoration is locked into all the major conservation systems operating across the country with varying levels of commitment, methodology and available resources. Nevertheless, at present there seems to be a discontinuity between the concrete objectives and targets expressed in Biodiversity Action Planning - which do not constitute statutory obligations - and the administration of the main agri-environment schemes which disperse government funds (David Westbrook pers. comm.). Whilst influential, mass appeal landowning conservation bodies such as the RSPB (2001), and The National Trust (Nixon 1999), have the capacity and finances to declare grassland restoration aims and actually see them through to fruition, whereas most county Wildlife Trusts depend on government departmental sponsorship for extensive restoration projects. Because the majority of the UK landscape is under private farmed ownership, the key to wholesale restoration of biodiversity to grassland is contingent on comprehensive changes to the CAP, and at least in Britain, the meshing together of BAPs with agri-environment schemes in order to guarantee the security of species-rich grasslands, while upgrading the biodiversity of grasslands nationally. The Curry Report (PC 2002), commissioned ostensibly due to the 2001 foot-and-mouth crisis, proposes fundamental CAP reform in order to "encourage best practice [in farming] and pay for environmental benefits." The benefits envisaged include increased spending on agri-environment schemes through "modulation", eventually leading to the creation of a new, simplified, single agri-environment stewardship scheme which has the putative aim of expanding scheme uptake substantially by using a, "broad and shallow" approach. Initial reactions from conservation organisations have been generally positive, though the Report's concentration on expanding landholder uptake without direct integration with Biodiversity Action Planning, certainly curtails the inferred restoration potentiality of the report if implemented. The immediate response of Friends of the Earth (30th Jan 2002) to the Report is that it "is only a first step towards a greener future for farming and the countryside. But it still falls short of the 20% which, under current EU rules, the Government could direct towards rural development, organic farming, and other environmental schemes." If one of the longer-term objectives of the Report is that the conventional [CAP] national farming perspective on grassland management is to harmonize with biodiversity planning, "modulation", and eventual "degression", these economic policies must be interwoven with biodiversity objectives.

STRATEGIES FOR GRASSLAND RESTORATION: LESSONS FROM THE LHF EXPERIMENT

The above review of governmental policies appertaining to grassland restoration suggests that the findings of the research work carried out at LHF should have a ready audience. Grassland restoration seems to be promoted as an important secondary tool in securing national biodiversity obligations. The UKBAP (1998) policy statement includes the intent to "collate, distribute, and apply the results from national research into restoration techniques." More recently, the Biodiversity Research Working Group on pastoral systems (2000) also identified priority research requirements in "habitat creation techniques; pooling and dissemination of best practice; and understanding the effects of restoration management at the landscape scale." So, what are the key findings that need to be disseminated to grassland restorationists? Perhaps, the results can be approximately divided into strategies relying mostly on *natural* restoration processes, and strategies utilising artificial enrichment methods.

Efficacy of natural restoration strategies

This is an approach that Manchester *et al* (1999) call "wait-and-see", and Clewell *et a*l (2000) term "passive". By reinstating traditional management the intention is to facilitate diversification through natural regeneration and colonisation (Newbold 1989; Crofts 1999; Willems 2001). For LHF, this has meant the cessation of fertilisers, and reintroduction of spring and autumn grazing with summer hay cut. It was hoped that export of nutrients through hay and livestock would lead to significant impoverishment of soil nutrient capital, and concomitant reductions in peak phytomass. Additionally, this would reduce the dominance of competitive species, increase diversity, and make the swards more receptive to re-colonisation of deleted species. This was the most straight-forward approach, requiring the least resources and effort, and in fact hay sales provided a useful supplement to the author's university bursary! On top of the traditional management regime, other techniques can be employed in order to enhance and speed up the natural diversification processes. In the case of LHF, the main intervention was the creation of herbicide gaps in an attempt to promote re-colonisation from soil seed bank and seed rain (Hillier 1990).

However, after seven years of restoration management, on the whole there have not been significant reductions in either soil phosphorus or potassium. While slightly lower, neither total P nor K levels have been significantly reduced by nutrient export through cropping. Phytomass production still seems to be positively related to these macronutrients, suggesting they are still the key variables controlling resistance to diversification. Equally, peak phytomass may at last be indicating a sustained downturn, though the average is still relatively high and subject to high variation across the improvement gradient. In 1994 mean peak phytomass was $642g/m^{-2}$, decreased to $481g/m^{-2}$ in 2001, but the dataset shows wide annual fluctuations such that there cannot be real confidence that productivity has necessarily reached a lower band.

In the grasslands of LHF, the diversification value of creating herbicide gaps seems limited, if not essentially ineffectual. Because artificial gaps, even relatively large ones, seem to be re-filled chiefly by dominant field-layer species, significant gains in overall diversity are probably minimal. The evidence suggests that activating seed banks after agricultural improvement will usually establish a vegetation matrix at most neutral in character, as both seed rain and seed banks are generally dominated by the same grass species abundant in the field-layer. Thus, under productive conditions, the size of artificial gap used (30-cm diameter) had little positive influence on restoration as they were quickly occluded, leaving little scope for re-colonisation of less mobile, stress-tolerant species from nearby species-rich propagule exporting sites.

So, *natural* restoration management has so far not produced clearly identifiable progress towards attaining substantial attributes of the reference/target community. The swards are nearly as dominated by common grass species at the end of the study period as at the beginning, and *Lolium perenne* is still overall the most abundant species. Diversity, as measured using Shannon's Index, was also almost exactly the same after seven years, and species richness was even in some cases marginally down. Nevertheless, an encouraging perspective is provided by the multivariate analyses, suggesting that vegetation changes in Plots 1-3 are at least following a roughly linear succession trajectory towards the MG5 reference community (Plot 5). Unfortunately, the vegetation of Plot 4 seems to be developing at a tangent to the main successional trend.

Overall then, in accord with many researchers, the *natural* restoration approach to agriculturally improved grasslands seems to suffer from the debilitating effects of: a) the very long time scale involved in depleting eutrophied soils through management (Marrs & Gough 1989; Marrs 1993); b) the paucity of desirable species present as persistent seed in the soil (Dutoit & Alard 1995; Bekker *et al* 1997; Davies & Waite 1999); c) and the lack of substantial seed rain of desirable species as there are no immediately contiguous species-rich propagule exporting sites (Huxel & Hastings 1999; Manchester *et al* 1999; Bischoff 2002).

Efficacy of artificial enrichment strategies

The clear alternative to the relatively passive approach presented above is the application of artificial enrichment techniques, notably inoculation with seed and transplants (Wells *et al* 1989; Davies *et al* 1999; Hopkins *et al* 1999). Within the UK context, this approach could be pejoratively termed as "gardening" (K. Thompson pers. comm.). Such an argument is perhaps founded on the idea that domestic landscapes should be delineated from agricultural landscapes in both ecological and conservation terms. Certainly, urban landscapes such as back gardens can be simply viewed as extensions of domestic space, whereas agricultural landscapes are products of important natural, historical and cultural lineage (Marren 1995). However, the deleterious effects of modern industrial farming within the UK has meant that for some species urban green spaces are refuges from the vast areas of biologically depleted agricultural land (Ash *et al* 1993; Baines 1995). Thus, utilising intensive "gardening" techniques such as transplant inoculation to restore biological deserts seems all the more acceptable. Perhaps the core of this semantic debate is the *scale* of intervention. Gardening is ultimately associated with small scale but intensive management, whereas agricultural management impacts on whole landscapes. Yet modern agriculture is in many ways the macro-scaling of horticultural techniques used in the average back-garden, such as sowing, transplanting, and encouraging propagation. Equally, the use of intensive artificial enrichment techniques in grassland restoration is paralleled by industrial agricultural operations such as resowing, oversowing, slot seeding, and modular plug transplantation (Wells *et al* 1989; Bisgrove & Dixie 1996; Gilbert & Anderson 1998). In such an intensively managed country as the UK, the difference between gardening and agricultural/industrial landscape management is essentially about scale. Whether an operation is undertaken by hand or by machine, the operation is still the same.

Unlike countries such as the United States with positively ascribable areas of "wilderness", the land surface of the UK is one of the most human influenced of all countries. Beyond the accusations of gardening, there is an ongoing, somewhat heated debate concerning the importance of species' provenance in restoration schemes, and whether artificial enrichment is an acceptable practice at all. However, a strong caucus of researchers conclude that additive treatments may be the only way to guarantee the re-colonisation of characteristic species (Pywell *et al* 1997; Strykstra *et al* 1998; Manchester *et al* 1999; Bischoff 2002). At LHF we utilised three main inoculant media; 9-cm pot transplants, 2-cm plug transplants, and seed doses. These were all sown into three diameters of competition-free gap. We wanted to gain some idea of comparative enrichment efficacy. The results can be compared in a number of ways: numerical differentials, statistically significant differentials, and cost differentials.

In terms of straight numerical differentials, pot transplant survivorship was the most successful. Of 1080 pot transplants inserted in 1994, 13.8% were still alive in 1999, compared to only 6.11% plug plants remaining. For the autumn cohorts, after three years, pot survival was 37.9% and 21.9% for plug implants. Pot plants then, show an immediate advantage for enrichment purely in terms of survival. However, there were few *statistically* significant differences between the performances of the two transplant sizes. There were also no end-state performance differentials between the two transplant sizes, indicating that whilst seedling-plugs have an immediate handicap, this disparity is rapidly made up by higher relative growth rate, and/or because pot plants can suffer proportionately greater retardation after implantation. Cost-wise, despite the greater overall survivorship of pot implants, plugs still work out cheaper in terms of unit cost/survival ratio.

In the case of inoculating with seed, it is difficult to compare transplant efficiency directly as one has to make an arbitrary decision as to how much seed equals one transplant. However, the dose rate used at LHF (50 seed per dose) appears to be far more efficacious in cost terms than using either pot or plug transplants for the same relative unit of enrichment. Though it must be noted that seed-derived seedlings do seem to have a greater potential of remaining indefinitely as juveniles or vegetative adults under these competitive, productive conditions.

Of course there were substantial survivorship, establishment and entrenchment differentials between species. In general though, irrespective of transplant medium, a clear set of 'core' transplant species can be identified. Using most parameters, *Primula veris*, *Malva moschata* and *Geranium pratense* performed most effectively. Apart from these robust, long-lived species, under productive conditions, inoculation success for intermediate and marginal species often seemed more of a question of postponing mortality risk rather than establishing viable populations. Adding non-core species to a scheme requires an in-depth knowledge of the inoculation resistance of the grassland and the competitive ability/resilience of each additional species. Nevertheless, Pakeman *et al* (2002) counsel against only using as small selection of "sure-fire winners" as this would threaten the broad aim of incrementing overall biodiversity. Indeed, a number of species have shown themselves to be more resilient than their natural distributions would suggest (Grime *et al* 1988; Green *et al* 1997; Pakeman *et al* 2002). *Filipendula vulgaris*, *Stachys officinalis*, and *Succisa pratensis* still survive from the 1994 cohort, whilst more common species such as *Leucanthemum vulgare* and *Prunella vulgaris* have become nearly extinct. Perhaps the only species to thoroughly live up to their strict stress-tolerator characteristics (Grime *et al* 1988) were the *Campanula* spp., and *Scabiosa columbaria* which rapidly became extinct in both spring and autumn implantations.

The time of year in which implantation takes place does seem to pose ramifications for transplant survival. Autumn planting led to higher overall survivorship (29.6%) than spring (21.7%), and autumn survivorship was higher for all species except *Malva moschata*. However, the outstanding species-differential was that of *Centaurea* which had very significantly higher survival than for the spring cohort. This analysis seems to indicate that most transplant species will establish most successfully when planted in early autumn as soil moisture deficit is usually low, and light and temperature conditions are still sufficient for plants to establish strong root systems in preparation for more droughty conditions during the following summer.

While, as stated previously, the creation of herbicide gaps did not seem to have a positive diversifying effect, they were also mostly ineffectual for aiding transplant establishment. The sizes of gap used (15 and 30-cm diameter) were insufficient to ameliorating the effects of sward resistance to transplants at the point of inoculation. The occlusion of the gaps was so rapid that competition was not postponed long enough to enhance transplant survivorship. However, there were positive interactions between the artificial competition-free gaps and seed enrichment. The 30-cm diameter herbicide gaps encouraged highest germination and seedling survival. A key finding is that whilst transplants were largely unassisted by the gap treatments, seed enrichment seems to be comparatively heavily dependent. In fact, gap creation in productive swards seems to be an essential treatment for successful seed inoculation.

If cost is not taken into account, then there are a number of positive things to say about the artificial enrichment techniques. Although there were large variations in survival and establishment success, mostly as a function of the broad spectrum of species used, the use of transplants was evidently successful in re-introducing species deleted by intensive agricultural operations. Some of these species are still present in sizeable numbers. Nevertheless, the only transplant species to actually integrate fully with the recipient swards was *Hordeum secalinum*. *Primula veris* and *Malva moschata* have produced offspring which have reached adulthood, though not throughout the improvement gradient, and the original implants are still conspicuous mastheads in the sea of grass.

Attempt restoration or start again?

Perhaps, in comparison to the effort expended, the restoration gains in plant biodiversity at LHF do seem to be rather minimal. The main environmental and botanical indicators used throughout the research indicate only minor positive changes in 2001 compared to the start of the project in 1994. However, does this suggest that a *ground zero* approach would have been more appropriate than trying to improve biodiversity using the range of restoration measures? *Ground zero* has many immediate attractions for floristic restoration. Turf and soil stripping massively reduce fertility, and undesirable weedy and competitive species are largely removed from both the field-layer and seed bank (Tallowin & Smith 2001). Reseeding after seedbed optimisation will usually result in a sparse and diverse sward, with most sown, relatively generalist stress-tolerators managing to establish, flower and regenerate (Anderson 1995; Marrs 1993), at least in the short term. Conspicuous invertebrates such as mobile butterflies will readily colonise the sparse vegetation mosaics after re-seeding. The only apparent downside of *ground zer*o restoration is the loss of agriculturally useful grassland. Turf-stripped re-seeds are essentially artefacts of conservation and cannot be readily integrated into agricultural systems (Wells *et al* 1989; Hopkins *et al* 1999; Manchester *et al* 1999). While this is acceptable where nature conservation is of the highest priority in order to extend or buffer reserves, what this approach cannot do is necessarily maintain economically appropriate levels of production within the farmed landscape. In addition, it is very difficult to evaluate the productive worth of reasonably mature soil microcosms (Merryweather 2001). The agricultural value of grassland at LHF is still attracting commercial rates from the neighbouring non-organic farm for hay and grass keep (grazing). This proves that hay production and grazing are still economically viable even while active restoration is taking place. Plot 1 has a particular interest in that it is equally the most intensive managed grassland, but is the least productive without continued intensive management. Because of this, even though Plot 1 has produced some surprisingly positive floristic results, and some inoculations have been more effective than for the 30+ year-old grasslands (e.g. *Hordeum* and *Malva*), no data has been collected on invertebrate, microbial or fungal diversity (Hambler & Speight 1995; Merryweather 2001). The only relevant observations are that common meadow butterflies such as meadow brown *Maniola jurtina*, skippers *Thymelicus sylvestris* and *Ochlodes venata*, even marbled white *Melanargia galathea*, seem as abundant as adults in the *Lolium* ley as Plot 4. Also, grasshoppers *Chorthippus* spp. seem to be equally abundant in Plot 1. However, one may assume that the soil ecology of the ley must be highly simplified compared with that of permanent grasslands (Merryweather 2001 etc). In addition, strategies for combating climate change may include the encouragement of no tillage in the management of grasslands (García Torres 1999). At present, conventional pastoral farming includes the regular ploughing and reseeding of grasslands, sometimes on a very short frequency, or even every year. Even many organic dairy farms utilise this intensive form of grass management (van Elsen 2000). Regular ploughing prevents the build up of non-sown species, relies on re-seeding with a few very productive but short-lived *Lolium* and *Trifolium* cultivars, and releases any stored nutrient capital. However, regular ploughing also releases the soil carbon store to the atmosphere, possibly exacerbating global warming (Brady & Weil 1999; Sauerbeck 2001). If pastoral management reverts back to maintaining permanent grasslands, the soils will at least sequester carbon which would otherwise contribute to climate change (García Torres 1999). Permanency of grasslands in biodiversity terms may also allow the build up of more complex ecosystems, perhaps leading to higher resilience to perturbations (Holmes 1998; Aarsen 1997; Naeem *et al* 2000), and perhaps greater long-term sustainability in production (Smith & Alcock 1985; Smith *et al* 1997; García Torres 1999; Hector *et al* 1999).

If experimental design was not a priority, and we had the hindsight of the above research findings, would we have altered our choice of restoration methods? In terms of the *natural* strategy, there is no doubt that reinstating traditional management is an essential, though slow, restoration mechanism. All other diversifying approaches must be organised within this framework. However, the research findings strongly suggest that the seed banks of improved grasslands do not have much restorative value. Equally, attempting to diversify swards by creating herbicide gaps so as to manipulate both seed bank and seed rain recruitment seems basically ineffectual. The ranking of species abundances in gaps may be altered in comparison with the sward, but the component species are roughly the same i.e. domineering grasses.

But what of the artificial techniques? The evidence from this study suggests that at the beginning of the project in 1994, *all* the improved grasslands were probably too productive to attempt inoculation using a broad-spectrum of species. Overall, whether pot or plug, the transplant losses were huge, and in cost terms, probably cannot be adequately warranted for most restoration schemes. For instance, the implantation of *Scabiosa columbaria* was a complete failure for both spring and autumn cohorts, as well as for seed inoculation. A limited literature review would suggest that it was an inappropriate species to attempt (Schenkeveld & Verkaar 1984; Grime *et al* 1988). However, the restricted ecological range of some species may be more to do with regeneration constraints than lack of robustness in the face of persistent neighbour competition, as the continuing survival of *Filipendula* would seem to indicate (Pakeman *et al* 2002). However, even though there are some anomalies, choice of transplant species should be probably restricted to a core selection, and a few additional intermediates. Was it worth attempting artificial enrichment? If the scheme had been purely to diversify the swards rather than an integrated research project, then this aspect of enrichment must be seen as somewhat of a failure, and an expensive one at that. The wide range of species used was for scientific purposes, whereas for a restoration scheme it would be unlikely that the more stress-tolerant species would have been selected. In a similar vein, van Andel (1998) comments that, "transplant experiments leave much room for a trial and error approach in restoration projects. However, such an approach can at the same time be used to obtain more adequate experimental data". For the core species both transplant and seed inoculations have managed to establish small populations of these species, some of which are starting to make recruitment inroads. For the intermediate and marginal survivors, the hope lies in the longevity strategy, of remaining alive until sward conditions become favourable for effective regeneration. In fact it is hoped that when productivity does start to significantly decline, the survivors will be released from the 'straight-jacket' of the competitive dominants, and manage to effectively reproduce. Furthermore, it will be interesting to see how long these individual plants can live (Tamm 1956; Tamm 1991; Ehrlén & van Groenendael 1998).

But what if we were attempting to diversify the whole field within which the plots stand? Taking cost into account, inoculating the core species using seed rather than transplants seems to be the most efficient method. However, unlike the transplants, seed needs to be sown into artificial gaps to be successful. As stated above, the destruction of a whole permanent, though improved grassland, through turf stripping and/or soil stripping in order to re-seed with a species-rich seed mixture, makes little ecological or economic sense. However, stripping a relatively small area, equivalent to the size of the plots (between 10-20% of the fields) and re-seeding with appropriate species-rich seed mixtures, would maybe seem the acceptable compromise (Tallowin & Smith 2001). Up to 80% of the productive capacity of the grasslands would be ensured, whilst the lesser proportion would establish a nutrient impoverished, species-rich hub from which desirable species can disperse. Considering many researchers consider proximity to species rich, propagule export sites crucial to restoration success (Hodgson & Grime 1990; Baudry *et al* 1994; MacDonald & Smith 1994; Pegtel *et al* 1996; Bischoff 2002), establishing floristically diverse patches within agriculturally improved grasslands may accelerate recolonisation, obviously in tandem with traditional management, whilst maintaining acceptable levels of agricultural productivity. This is hardly a novel approach, but the technique possibly needs further investigation. For instance, what are the optimal parameters? There is a need to understand to what extent depth of soil removal, and diffusion block size, shape and pattern, impact on the efficiency of recolonisation? For example, using the same area of clearance, what strategy would be best: a) single large diffusion hub; b) multiple smaller blocks; c) or long strips that traverse the whole grassland? Following the results of artificial gap occlusion at LHF, the large block design seems to have the key advantage over smaller blocks and strips in that there is a lower edge to area ratio. This reduces the interface for recolonisation, but perhaps also increases the propagule export longevity of the hub, as infill by competitive species may be slower, or markedly reduced. Certainly the indications are that relatively thin strips (<50-cm width) would probably be rapidly re-invaded by the dominant grasses and stoloniferous species (Bullock *et al* 1995; Tallowin & Smith 2001), undermining the enrichment value of the method. However, strips that traverse the restoring grasslands have the advantage over a single block as far as the potential for dispersion coverage, and taking into account environmental factors such as variations in hydrology and soil fertility. Arguments over what shape or shapes the patches should be, and their distribution patterns has obvious island-biogeography connotations, as the species-rich diffusion hubs are surrounded by a sea of relatively inhospitable improved grassland matrix. And whilst created to disperse propagules into these interstices, the diffusion hubs are also subject to constant re-invasion pressure themselves. This aspect smacks a little of the 'single large or several small' debate concerning optimal nature reserve size and number (Shafer 1990), though essentially in reverse. Further experimentation on the subject is required, and as far as the author knows, no systematic research has been conducted.

THE LONG ROAD AHEAD

There is a singular, yet complex and crucial question to be eventually addressed in any restoration scheme: when has restoration been achieved? Evidently, it is fundamental to any restoration programme that goals/objectives/targets are identified before restorative activity takes place (White & Walker 1997; Clewell 2000; SER 2000; Willems 2001). Ehrenfeld (2000) notes that the specification of goals is often considered the most important component of a project, because it lays down expectations, impels detailed action plans, and decides the kind and extent of monitoring protocols. In systematic terms, the Society for Ecological Restoration (SER 2000) considers that a restoration has been successful, "when an ecosystem's biodiversity, functions, ability to accommodate normal disturbances, and abiotic services closely compliment these same attributes in the conceptualised reference ecosystem." For the LHF project, these principal goals equate with the general restoration category determined by Clewell *et al* (2000) as, "repair of a damaged ecosystem." The damage to biodiversity inflicted on the LHF grasslands has been carried out through intensive agricultural management. Although there is no absolute botanical, historical or verbal evidence that the study fields were species-rich MG5 grasslands (Rodwell 1992) prior to 1945, the cumulative data suggests that this was probably the case (see **Chapter 2**). Thus, in accordance with Clewell *et al*'s (2000) guidelines, the most proximal agriculturally unimproved reference community was identified in the pre-restoration phase as Plot 5, and closely monitored in order to establish an acceptable reference *gold-standard* for evaluating the effects of restoration treatments on the study grasslands. Thus, the qualified goal has been to restore the improved grasslands to an ecological semblance as close as possible to the reference community (White & Walker 1997; Willems 2001). However, debate can still range over how close is "close enough". To seriously attempt the replication of the reference community would normally be seen as an essentially unfeasible proposition, especially after so much damage has taken place. This is backed up by Parker & Pickett (1997) who contend that, "for an array of different sites to be restored, restricting restoration objectives to a single ideal reference state can create fundamental problems because the environmental context may differ among sites…[we] must strongly disagree with the use of 'arbitrary' reference states; instead a reference system should be based upon a range of what is possible." This accords with Ehrenfeld (2000), who suggests that specific goals would be more easily and more appropriately set if restorationists would set forth at the outset the true scope and limitations of what is possible in a given project." Appropriately then, there has to be pragmatism in goal-setting as to what is realistically achievable, and the establishment of appropriate timescales for achieving the goals. Attempting to achieve maximal targets is not in itself a fanciful enterprise. For example, there are some accounts of grassland restoration schemes that have produced remarkable results in relatively short periods (Gibson 1995; Shared Earth Trust 2000; Barbaro *et al* 2001). However, each site presents specific biotic and abiotic constraints on restoration, some of which are clearly apparent at the outset so as to be included in the planning process, and others that can only be revealed over time. Therefore a more flexible approach to goal-setting accepts the reference community (or communities) as the optimal target, but also accepts lower tiers of attainment, especially as timescale parameters are fitted around most, if not all schemes. This approach accords with the directives of the Society for Ecological Restoration (2000) which is convinced that restoration should not be treated as a product, but rather an ongoing process. This allows the establishment of what SER term *performance standards* prior to the inception of restoration activities. These standards represent milestones of accomplishment, whose attainment can be determined objectively through a monitoring program. The inference is that once performance standards are attained, there is a reasonably good chance that other ecosystem functions have also been revived. Performance standards should be considered as indications of the degree to which restoration has been achieved. For LHF, whilst using the reference community as an essential totem, in the light of seven years of restoration management, goal-setting may be usefully reset to reflect the overall results, and also to establish performance standards for the next seven or more years. As far as appraising the results from the first seven years, perhaps three basic tiers can be defined as:

- Maximal target(s) - achieving ecological equivalence to the *reference community*. In this case productivity, diversity and species-richness typical of MG5 hay meadows.
- Medial target(s) – clear development towards MG5 grassland; significant decreases in productivity, and increases in diversity and species-richness.
- Minimal target(s) – indications of development towards MG5; non-significant reductions in productivity and increases in diversity and species-richness.

Obviously, the maximal goal of reassembling grasslands typical of the species-rich MG5 grassland type (Rodwell 1992) has clearly not been achieved. Unfortunately, it seems even the *medial* goals have also not been accomplished. The multivariate analyses do not, as yet, identify clear successional developments towards the target community. Indeed, basic significant increments in biodiversity are still not apparent. Thus, the final account after seven years of restoration management allows that only minimal goals have been achieved e.g. non-significant reductions in productivity and increases in floristic diversity. Even with considerable resource input and relatively long timescales, restoration may involve periods of developmental stasis, a time-lag between restoration effort and tangible results (Tallowin *et al* 1995; Huxel & Hastings 1999). Eutrophied grassland systems seem to be surprisingly resistant to conservation management (Mountford *et al* 1994; Anderson 1995; Kirkham *et al* 1996; Smith *et al* 2000). Consequently, it is important in the planning stage to develop multiple or ranked restoration objectives, including the eventuality that null or negative change could even take place. In view of the range of possible outcomes, The Society for Ecological Restoration (2000) assert, “there are no hard-and-fast rules conditioning biological systems, comparisons of pragmatic goals and what is actually achieved.”

From the start, attempting ecological parity with the reference community, for example, in terms of composition, was not a necessarily practicable objective. Plot 5 still endures as a species-rich remnant community, the survival of which was assured by its location on a steep, un-cultivatable bank. Its species assemblage may be broadly representative of the LHF grasslands before agricultural improvement, but cannot be strictly assumed as a hay meadow archetype inclusive of all the environmental variables of each restoration field (Rodwell 1992; Gibson 1998). There was never a simple assumption that the experimental grasslands necessarily could be restored to reference community equivalence after five, seven, or even ten years. Especially, as the short-term monitoring endpoint was not predicated on goal attainment, rather it was based on the arbitrary termination of doctoral research programming (Pakeman *et al* 2002). Perhaps, at least, it was hoped and expected that the successional trajectory of the grasslands would align roughly with the reference community. Certainly, the study grasslands are simply categorised by Rodwell (1992) as MG6, and as such are immediate affiliates of the target community, indicating a short step to reassembly. Critchley *et al* (2002) suggest that relatively species-rich examples of MG6 sub-communities offer "the best opportunities for restoration." However, they note that, whereas unimproved species-rich communities have relatively narrow ecological amplitudes, in contrast the MG6 vegetation type spans not only a huge national land surface, but also a very broad successional nexus, with equally broad restoration pathways (Rodwell 1992; Hodgson *et al* 1995). The evidence so far presented suggests that, far from a linear process, restoration after agricultural improvement may develop in unexpected ways. There have not been discernable linear increments in plant species richness/diversity with time, facilitated by restoration management. However, there may be so far undefined soil fertility/productivity thresholds, which need crossing before evident linear increments or step progressions in diversity can take place. Irrespective of forecasting and theorising, after seven years, the successional trajectories of the improved grasslands are still very uncertain. The pessimistic inference is that after artificial eutrophication, restoration to a pre-improvement state may be impossible (Gibson & Brown 1991; Gilbert & Anderson 1998). Indeed, many researchers have both identified, and projected, that attempting grassland restoration after agricultural improvement using passive means is commonly a very protracted process, or even impossible given the fact that landscape biodiversity has been massively degraded and fragmented (Parker 1995; Tallowin & Smith 2001; Willems 2001). Bakker & Berendse (1999) comment that, "until recently, it was thought that restoration of these communities would be straightforward.

However, abiotic constraints (with respect to eutrophication and acidification) have hampered restoration more than previously thought.” Gibson & Brown (1991) offer an even more pessimistic forecast that, “after heavy fertilisation…on deeper soils, the outlook [for restoration] may be bleak. Even with time, ‘ancient grassland’ communities will not develop on the wrong soil.” An added dimension to this prognosis is that contiguity with species-rich export sources is counted as essential for effective restoration (Gibson & Brown 1991; Baudry *et al* 1994; Bakker & Berendse 1999; Willems 2001). Unfortunately, none of the LHF grasslands are strictly adjacent to unimproved grassland. Perhaps we can speculate whether contiguity with species-rich export sources would have made a significant difference, and restoration to MG5 equivalence would seem more achievable in the long-term.

So what can we hope to happen over the next seven years of restoration management? Unless further funding is found, it is unlikely that any more artificial enrichment schemes will be undertaken. However, if further enrichment were conducted without research funding, informed by the findings presented in this thesis, they would centre on seed inoculation experimentation. Otherwise, gains in biodiversity will therefore be reliant on: a) recruitment from the surviving introduced plants; b) natural recruitment from nearby sources of propagules, the nearest being Plot 5 and Sheepwash meadow; c) diversification, particularly through reductions in grass dominance with continuing traditional management. Clearly the gains through a) and b) could not take place without c), though major gains in species-richness cannot occur without substantial new recruitment. The procedures of grassland management may also facilitate diversification such as transport of propagules between grasslands on hay-making machinery or by livestock (Poschlod *et al* 1998; Bakker *et al* 2002). Nevertheless, the established facts suggest that the LHF restoration project is still in the *initial restoration phase* (Willems 2001), and minimal performance standards have been reached thus far. The as yet unanswered question is whether conservation management alone can attain significant biodiversity augmentation. Natural immigration streams may not be adequate enough to be relied on due to the fragmentation of landscape biodiversity, thus restoration goals may be stymied unless there are repeated artificial inputs. Because the grasslands of LHF will probably not be applied with further enrichment schemes, future restorative progress is reliant on the regenerative potential of the established transplant species hubs, and the interaction between conservation management and the multiple evolving start-points derived from the systematic or arbitrary differentials in agricultural improvement. Edwards *et al* (1997) suggest that, "another lesson for restoration ecologists is that we must accept many different endpoints and not strive too doggedly to achieve a particular, precisely defined community." But to follow this precept now would be to acknowledged that maybe grasslands such as those studied in this project cannot be returned to an approximate pre-improvement state, and have an as yet unresolved successional terminus (Parker 1995). It is far too early for such a conclusion to be accepted. Even so, the research at LHF was established and conducted with the principal objectives of exploring the key ecological factors behind grassland restoration, and to use any significant findings to advance grassland restoration practice (Anderson 1995; UKBRWG 2000). Clewell *et al* (2000) contend that, "restoration ecologists improve their craft by becoming familiar with how restoration objectives were accomplished." Whilst pertinently, The Society for Restoration Ecology support the notion that, "even falling well short provides useful lessons for other restorationists." Therefore, the implication is that equally important information is accrued by investigating how and why restoration targets are *not* achieved. The conclusions from this response-data can then be incorporated to revise management practice, and fine-tune the restoration. This feedback approach can be broadly encompassed within the concept of *adaptive restoration* (van Duren *et al* 1998; Gunderson 1999; Zedler & Callaway 2002). Zedler (2001; pers. comm.) uses the term to describe the design of restoration projects as large field experiments, structured so that restoration develops as phased modules, enabling experimental results from early modules to inform later approaches and experiments. From the LHF point of view, the intriguing aspect to adaptive restoration is the difference between the "reactive" and "proactive" feedback systems, which are not mutually exclusive, though do represent important conceptual divisions in methodology. Thus, restoration using traditional nature conservation management takes a relatively passive approach, using monitoring and experiential knowledge to identify ecological developments – whether positive or negative – in order to adjust management appropriately. In contrast, the more "proactive" model advanced by Zedler (2002) integrates scientific experimentation as a rolling programme, the results of which inform management adaptation, refines practice, and helps to develop new restoration paradigms.

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