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The ecological restoration of slate waste tips in Wales, UK

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The ecological restoration of slate waste tips in Wales, UK.

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by

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Abstract

Many hundreds of hectares of slate waste have accumulated in north Wales as a result of hundreds of years of quarrying. Due to its resistance to weathering and susceptibility to flushing, no fines or organic materials accumulate on slate waste tips and they remain largely devoid of vegetation cover.

Various methods aimed at restoring vegetation to slate waste tips have been tested but no single method can claim to have achieved results that could be applied on a large-scale. To address this problem an integrative experimental approach to the ecological restoration of slate waste was taken, focussing on the reassessment of past experiments, the sourcing of planting stock and the identification of the best mixture of materials for applying to slate waste to increase plant-available nutrients and water.

The reassessment of past restoration experiments at Penrhyn Quarry, north Wales, showed that only by introducing large quantities (for example, a 75 cm depth of boulder clay) of materials to provide nutrients and increase water-holding capacity could acceptable levels of plant establishment be achieved (e.g. survival in LIFE experiment A ≥96.43 %). The greatest levels of success were achieved by the hydraulic distribution of peat-based slurry followed by planting of tree saplings. The surviving trees in this treatment show sustained growth (e.g. highest growth rate recorded between 2002 and 2007 = 391.26 % increase in crown area (tree 23)) more than 30 years after planting.

To test the potential of using very local (quarry) provenances in quarry restoration activities, shoot cuttings of willow (*Salix caprea* × *Salix cinerea* = *Salix reichardtii*) and seed from seven species of tree and shrub (*Acer pseudoplatanus, Betula pendula, Cytisus scoparius, Fagus sylvatica, Sorbus aucuparia, Quercus petraea* and *Ulex europaeus*) were collected from both quarry (Penrhyn Quarry) and non-quarry populations. Survival and growth rates showed no clear differences between quarry and non-quarry populations

when grown in slate waste. Although it cannot be advised that all planting stock for quarry restoration is collected from quarry provenances, further research is recommended.

Field application of green waste compost (GWC) (produced by Conwy County Council) and hydrated polyacrylamide (PAM) gel in a 50:50 (v/v) mixture produced significantly greater species richness, vegetation cover and dry biomass over 18 months than a control treatment. Vegetation cover in the GWC-PAM treatment showed an increase of more than 90 % over that prior to applications of soil-forming materials.

In conclusion, rapid, sustainable, low-cost methods of successfully restoring vegetation cover to slate waste tips are achievable without the need for costly site preparation or disturbance of established successional assemblages. Development of nurse crops and early colonising vegetation cover of this type is an important step in the sequential ecological restoration of slate waste tips. And with careful consideration of site-specific requirements, past restoration successes and sources of planting stock, the efficiency and success of ecological restoration projects can be greatly increased.

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Abbreviations and Acronyms

Several abbreviations and acronyms have been used throughout the text of this thesis; these are alphabetically listed below for reference.

AD	-	*Anno Domini*
ADW	-	Ashed Dry Weight
AFLP	-	Amplified Fragment Length Polymorphism
AG	-	Above Ground
AGDW	-	Above Ground Dry Weight
AGFW	-	Above Ground Fresh Weight
ANOVA	-	Analysis Of Variance
AONB	-	Area of Outstanding Natural Beauty
BA	-	Stem Basal Area
BAP	-	Biodiversity Action Plan
BG	-	Below Ground
BGDW	-	Below Ground Dry Weight
BGFW	-	Below Ground Fresh Weight
BSH	-	British Seed Houses
C:N	-	Carbon : Nitrogen (ratio)
CA	-	Crown Area
CRoW	-	Countryside Rights of Way Act 2000
C_S	-	Stem Circumference
d_c	-	Average of two crown diameter measurements
DEFRA	-	Department of Food and Rural Affairs
DOE	-	Department Of Environment
d_s	-	Stem Diameter
DW	-	Dry Weight
EC	-	Electrical Conductivity
EIA	-	Environmental Impact Assessment
ESE	-	East south east
EU	-	European Union
FW	-	Fresh Weight
GPS	-	Global Positioning System
GWC	-	Green Waste Compost
ha	-	Hectare (10 000 m^2)
ISTA	-	International Seed Testing Association
LIFE	-	European Commission Directorate-General for the
LOI	-	Loss On Ignition
MASL	-	Metres Above Sea Level
MC	-	Moisture Content
MPA	-	Mineral Planning Authority
MPG	-	Mineral Planning Guidance
MPS	-	Minerals Planning Statement
Na-PA	-	Sodium Polyacrylate
nc	-	No Correlation
NGR	-	National Grid Reference
NNR	-	National Nature Reserve
NoF	-	No nutritional amendment
NoW	-	No addition of water-holding materials

NPK	-	Nitrogen Phosphorus Potassium (fertiliser)
NQ	-	Non-quarry
NVC	-	National Vegetation Classification
NW	-	North west
p	-	Probability
PAM	-	Polyacrylamide
PAS	-	Publicly Available Specification
PDOs	-	Potentially Damaging Operations
PPG	-	Planning Policy Guidance
PPS	-	Planning Policy Statement
Q	-	Quarry
r	-	Coefficient of correlation
r^2	-	Coefficient of determination
RFLP	-	Restricted Fragment Length Polymorphism
RIGS	-	Regionally Important Geological/Geomorphological Sites
SAP	-	Super Absorbent Polymer
SE	-	Standard error of the mean
SER	-	Society for Ecological Restoration
SSSI	-	Site of Special Scientific Interest
TCA	-	The Composting Association
TWIRLS	-	Treating Waste for Restoring Land Sustainability Environment Life Environment Programme
UK	-	United Kingdom (of Great Britain and Northern Ireland)
UKBAP	-	United Kingdom Biodiversity Action Plan
UKCIP	-	United Kingdom Climate Impacts Programme
USA	-	United States of America
USEPA	-	United States Environment Protection Agency
UV	-	Ultraviolet
WHC	-	Water Holding Capacity
WRAP	-	Waste and Resources Action Programme

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In writing this passage it marks the last piece of the Ph. D. shaped jigsaw on which I have been working for what seems an eternity. The completion of this thesis is a testament to the powers of absolute perseverance, determination and stubbornness; demonstrating that anything in life is achievable if you are willing to work hard enough.

1 Introduction

1.1 Background of research

In 2003, quarrying and mineral extraction in the UK employed in excess of 34,000 people at more than 2300 sites and produced more than 290 million tonnes of quarry products, which were estimated to exceed £2 billion in value (Hillier *et al.*, 2005). Total UK sales of building stone in 2003 amounted to £49 million, 65 % of which was sandstone. However, the total value of worked or monumental (i.e. cut, shaped, or finished) stone and slate was £455 million, showing the added value of worked or processed products (Lott *et al.*, 2005). The annual UK sales value of roofing and cladding slate is estimated to be in the region of £27 million (Lott *et al.*, 2005).

Today, slate extraction forms only a relatively small proportion of the UK's total mineral production. There are several slate rich regions in the UK; these are the Lake District, Cornwall (and Devon to a lesser extent), the Scottish Highlands, parts of south Wales, and north Wales. The slate in these regions was formed in five geological periods: the Devonian (Devon and Cornwall), Silurian (Cumbria and Denbighshire), Ordovician (Cumbria, Carmarthenshire, and Pembrokeshire), Cambrian (Gwynedd) and Precambrian (Scottish Highlands) (Lott *et al.*, 2005). In 2005, slate was being extracted from 44 sites in these regions, under permits to work a total of 471 hectares of land (Hillier *et al.*, 2005).

The quarrying and processing of slate to produce roofing slates involves, along its path, the creation of many waste fractions: overburden and development rock (removed to expose the slate vein); quarrying waste (from working the slate vein or face, removing poor quality rock, or rock damaged by blasting and the reduction of block size); sawn ends (off-cuts from block from the working quarry face); trimming waste (arising from splitting block into slates, including reject slates, splinters, and dressing waste); and mill fines (produced as a result of sawing and cutting slate block) (Richards, 1991;

DOE, 1995). The overburden and development rock fraction contributes most greatly to waste generation, comprising an estimated 90 % of total waste (Ove Arup, 2001).

Many SSSIs (Sites of Special Scientific Interest), and some SPAs (Special Protection Areas) and SACs (Special Areas of Conservation), have their origins in quarrying, because the quarrying processes have provided a range of habitats and ecological niches, for example ponds, rock faces and scree, that are relatively rare in the UK (DOE, 1995). Even whilst they are actively worked, quarries can be ecologically valuable, providing nesting sites for birds and a range of habitats for flora and fauna, including virgin surfaces for rare colonists. Quarries can therefore make a significant contribution to UK bio- and geo-diversity (BGS, 2005).

Slate waste tips, unlike other mining and quarrying substrates such as limestone or metalliferous mine wastes, are extremely resistant to natural colonisation. They do not have a typical and characteristic associated vegetation type, and closed herbaceous cover is rarely formed (DOE, 1995). This, however, is not to say that slate waste tips are ecologically uninteresting or unimportant. Species tolerant of drought and low nutrient levels, such as birch (*Betula* spp.), rowan (*Sorbus aucuparia*) and goat willow (*Salix caprea*), and occasionally oak (*Quercus* spp.) and hawthorn (*Crataegus monogyna*) may establish and form open woodland. Where significant spontaneous natural colonisation has occurred on waste tips, care should be taken to avoid disturbance by removing slate for aggregates processing; demand for aggregate should be satisfied from current arisings and waste tips (DOE, 1995).

Ecological restoration of slate waste is justified not only by a desire to rectify damage caused by industrial activities, but also by the need to protect interesting and valuable new habitats that are home to previously uncommon species.

1.2 Research objectives

This thesis aims to address practical and scientific short-fallings in current restoration practices and in doing so further the understanding of restoring severely disturbed landscapes and increase rates of restoration success on slate waste tips. This will be achieved by focussing on several specific objectives:

- To identify those past methods that have proved most successful for initiating ecological restoration on slate waste tips (chapter 3)

- To determine whether planting stock of very local (quarry) provenance survives and grows better than planting stock of non-quarry provenance in the conditions found on slate waste tips (chapter 4)

- To identify the potential constituents of an artificial soil that could be applied to slate waste tips to increase plant available water and nutrients (chapter 5)

- To identify the combination of these constituents (i.e. the artificial soil) that gives the most successful vegetation establishment on slate waste tips (chapter 5)

- To devise a low-input means of applying the artificial soil *in situ* with minimal disturbance of pre-existing site conditions (chapter 5 and appendix 8)

1.3 Thesis outline

Chapter 2 – Background to quarrying and restoration ecology

Presented within this chapter are introductions to quarrying and mineral extraction in the UK, indicating the importance of these activities and their associated impacts on the landscape. Particular attention is paid to slate, including a brief review of its chemistry, formation, physical properties and uses. This is followed by an overview of restoration ecology: what it is, what role it plays, why it is necessary within the context of minerals workings, and details of legislation relevant to the impacts of quarrying and mineral extraction in the UK.

Chapter 3 – A review of restoration work carried out at Penrhyn Quarry

A review of past restoration projects carried out at Penrhyn Quarry, Bethesda, investigating the efficacy of a number of different methods for restoring vegetation and promoting habitat development on slate waste tips formed by quarrying activities.

Chapter 4 – The importance of planting material source/genotype in post-industrial ecological site restoration

Cuttings and seeds were collected from tree and woody shrub species commonly found growing on slate waste tips at Penrhyn Quarry. This reproductive material was compared in a series of comparative germination, growth and survival studies with material of the same species collected from non-quarry locations in the vicinity of Penrhyn Quarry.

Chapter 5 – Material applications to post-industrial land for restorative purposes

Studies were conducted to determine the effectiveness of three super absorbent polymers and six green waste composts in increasing availability of water and nutrients to establishing plants. Mixtures of these materials (artificial soils) were applied to slate waste tip surfaces at Penrhyn Quarry, Bethesda, to facilitate re-vegetation and habitat development.

Chapter 6 – General discussion and conclusions

A general discussion of the findings of chapters 3 - 5 is presented, highlighting experimental limitations and areas for further research. The chapter concludes with an explanation of how the experimental findings can influence the success of ecological restoration work on slate waste tips in the UK.

7 – Appendices

8 – References

2 Background to quarrying and restoration ecology

2.1 Slate quarrying

The extraction of slate in the UK has been carried out for many hundreds of years; it is reported that extraction has taken place intermittently in some form since Roman times (North, 1946; Richards, 1991). Purple slate, probably originating from Bethesda or Nantlle, Gwynedd, was used in the construction of the Segontium Roman fort in Caernarfon, Gwynedd, dated 77 AD (North, 1946).

Early slate extraction in Wales took place on a small-scale, worked purely to satisfy local demand. However, records demonstrate that by the late fifteenth century substantial exports of roofing slates from north Wales to Ireland were taking place (North, 1946). The slate quarrying industry began to flourish in Wales during the eighteenth century, it was restricted by poor infrastructure to local and coastwise usage until the middle of the century, but with the onset of widespread road construction, it became a major industry in the latter half of this century (North, 1946; Richards, 1991). The further development of the road network, the dissemination of the properties and quality of Welsh slate, and the later development of tramways, canals and the rail network during the nineteenth century, coupled with a rapidly expanding population, saw the demand for Welsh slate boom (North, 1946). The industry reached a pinnacle during the 1890s (Richards, 1991; DOE, 1995), when production in north Wales exceeded 500,000 tons *per annum* and employed more than 16,000 men (DOE, 1995). From this point the industry went into decline, aided by two world wars, to reach an eventual low point in the 1960s (Richards, 1991).

It is reported (Hillier *et al.*, 2005) that the total UK production of slate products in 2001 totalled 551,000 tonnes. This comprised: 45,000 tonnes architectural, cladding, roofing and damp proof courses, 27,000 tonnes powder and granules, 39,000 tonnes crude blocks, and 440,000 tonnes fill and other uses. Despite this substantial production there were still considerable imports, amounting to 148,257 tonnes (exports equalled 13,400 tonnes) (Harrison *et al.*, 2002). The annual waste generation (England and Wales) associated with the production of this quantity of finished slate products was approximately 6 million tonnes (Harrison *et al.*, 2002; Hillier *et al.*, 2005) in 2001; 580,000 tonnes of this was used as aggregate (Hillier *et al.*, 2005) and the remaining 5.7 million tonnes was disposed of by land-filling and waste tip formation (Harrison *et al.*, 2002).

North Wales is the main slate producing area in the UK, with slate from Penrhyn Quarry (Bethesda, Gwynedd) accounting for more than 50 %, and slate from the combined Blaenau Ffestiniog (Gwynedd) area contributing 35-40 %, of the total UK roofing slate production (DOE, 1995). Various figures for the total number (combined operational and expired) of slate extraction sites are given in the literature, for example, Davidson *et al.* (1994) and DOE (1995) respectively identify 462 and 141 individual slate working sites in Gwynedd. However, the most detailed assessment of the slate industry in Wales is provided by Richards (1991) in a review of working sites ranging from the very large Penrhyn Quarry to the smallest exploratory pits never destined for commercial exploitation. In total Richards (1991) lists 638 slate extraction sites in Wales. North (1946) emphasises the fact that many of these quarries never passed the experimental stage, often because the slate was too thick, too soft or not sufficiently durable, or because the slate was of good quality but was inaccessible.

Extraction of slate is carried out by open, pit or underground quarrying, determined by the proximity of the slate vein to the land surface and the angle at which it lies (DOE, 1995). Open quarries extract slate close to the surface of a hill or mountain, whereas pit quarries extract slate that dips almost vertically, creating large pits such as the one at Penrhyn Quarry. Slate veins that dip less severely might require underground quarrying to follow the slate into the hill or mountainside; examples of this method of extraction are common throughout the Blaenau Ffestiniog area of Gwynedd.

2.2 Slate chemistry and formation

Slate is a hard, very fine-grained (< 32 μm), low-grade metamorphic rock that is non-gritty, can be scratched with a metal implement (due to the high content of soft clay minerals) and can be split with a chisel into sharp flakes or tiles (Merriman *et al.*, 2003). It is the metamorphic equivalent of siltstone, clay, shale and mudstone, and therefore contains a high percentage of quartz-silt and clay minerals (i.e. hydrous alumino-silicates with iron and magnesium), and is grouped with shale and mudstone as "mudrocks", the most plentiful sedimentary rock types in the Earth's crust. Unlike slate however, these other mudrocks degrade freely to produce earthy fine-grained matter. In contrast, slate is highly resistant to weathering and degradation; as a result its breakdown products consist of scree containing sharp-edged tiles and flakes (generally >20-30 mm) with little fines material (Merriman *et al.*, 2003).

The most notable feature of slate is its well-developed cleavage, resulting from the alignment of well-crystallised platy clay minerals arranged along a single set of micron-spaced parallel planes. Metamorphism causes all previous sedimentary features of the mudstone (e.g. bedding and fissility) to be replaced and slaty cleavage to be formed (Merriman *et al.*, 2003). Slate lacks porosity, and is therefore impervious to fluid flow (Lott *et al.*, 2005), thus making it extremely efficient at withstanding weathering.

Slate can occur in several colours, for example green, blue, red, and purple, although grey is the most common. Colouration results from the presence and different oxidation status of various iron-rich minerals. For example, red and purple colours result from disseminated hematite (Fe_2O_3), grey and blue result from variable quantities of non-oxidised pyrite (FeS_2), green slate may contain volcanic ash, and rusty intrusions and edges result from oxidised disseminated pyrite (Merriman *et al.*, 2003). Frequent patches of green colouration are common in slate from certain regions; these form as spheres around small organic particles existing within the shale or mudstone. Compression during orogenesis (mountain building) causes these spheres to become elongated parallel to the cleavage, giving an indication of the degree of compression encountered (Anon, undated).

Shale and mudstone is transformed into slate when a sedimentary basin is compressed and deformed during orogenesis, when metamorphic conditions of 250 – 300°C temperature and 3 kilobars pressure are created. During orogensis clay crystals from shales or mudstones are mechanically rotated and chemically re-crystallised forming white mica (e.g. illite, sericite or muscovite mica) and chlorite crystals grow and thicken along micron-spaced parallel planes orientated with the direction of the greatest stress (Merriman *et al.*, 2003).

2.3 History of Penrhyn Quarry

Penrhyn Quarry, the location for the majority of research carried out in this study, is located in Bethesda, Gwynedd (see figure 2.1). It is likely that the quarry was worked for slate in Roman times, as alluded to by North (1946), but the workings as they appear today date back to the late eighteenth century (North, 1946; Richards, 1991). Richard Pennant (the first Baron of Penrhyn (BBC, 2007)) bought out the existing leases for Penrhyn Quarry in 1782 (Richards, 1991) with a fortune gained from the slave plantations of Jamaica (BBC, 2007). Pennant developed the previous slate workings, which produced 1800 tons of slates in 1780, into a large-scale industrial operation with an output of 20,000 tons in 1800 and nearly 74,000 tons in 1836 (North, 1946). By 1882, 100 years after Pennant's purchase of Penrhyn Quarry, the production capacity had increased to a staggering 111,166 tons, requiring a workforce of 2089 men (Richards, 1991).

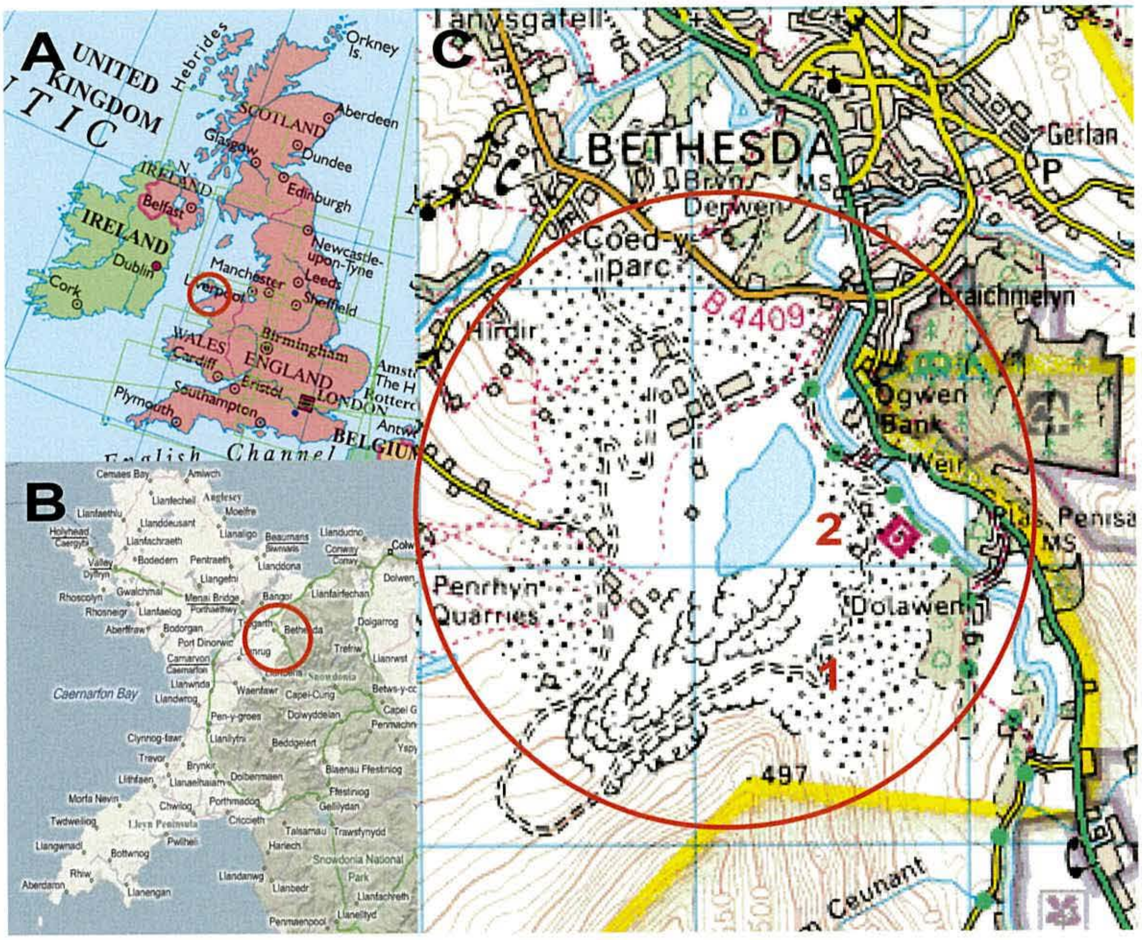


Figure 2.1 Location of Penrhyn Quarry at UK (A), regional (B) and local (C) scales. Approximate locations of experimental sites 1 and 2 (see chapter 5) at Penrhyn Quarry are indicated (C)

Maps (A and C) from Memory-Map Navigator (2002) and (B) Bing Maps (2010)

2.4 Slate quarry waste

During slate extraction and production, as little as 5 % of the material extracted might end up being used for quality slate products (Lott *et al.,* 2005). It has been estimated that nearly 6 million tonnes of slate waste is generated *per annum* in Gwynedd alone. In the UK the present proportion of waste to slate production varies greatly, from 20:1 to at least 100:1, and can be even higher where significant volumes of overburden have to be removed (Ove Arup, 2001). Some examples from slate quarries in Gwynedd are reported by Ove Arup (2001):

Penrhyn Quarry	115:1	2.4 million tonnes waste *per annum*
Oakeley Quarry	145:1	2.2 million tonnes waste *per annum*
Llechwedd Quarry	500:1	500,000 tonnes waste *per annum*

A report by the DOE (1995) found the following amounts of waste in the main slate-producing regions of Britain.

Clywd	3-5 million tonnes
Cornwall and Devon	20 million tonnes
Cumbria	25-30 million tonnes
Dyfedd and Powys	10 million tonnes
Gwynedd	350-400 million tonnes
Scotland	50 million tonnes

The only significant current slate waste production is taking place in Gwynedd and at Delabole Quarry in Cornwall, which produces approximately 50,000 tonnes *per annum* (DOE, 1995).

North (1946) points out that much of the waste generated at slate quarries is unavoidable. Rock that is exposed at the surface suffers as a result of atmospheric conditions such as wetting and drying, and freeze-thaw cycles over extended periods of many hundreds or even thousands of years. Such processes affect slate along its joints, effectively preventing blocks of any useful size being hewn from the slate face or vein. Even slate that has not suffered from atmospheric exposure may be of poor quality due to poor cleavage in the slate bedding planes; this poorer material has to be extracted in order to reach material of commercial quality. Poor or impaired cleavage can also occur as a result of igneous intrusions into the slate veins, resulting in rock that cannot be split and is not therefore useable. These fractions of waste can be considered as overburden or development waste.

The prominence and starkness of slate waste tips is regularly commented on in the literature (North, 1946; Richards, 1991; DOE, 1995), reflecting the negative impact they are often perceived to have upon landscape quality.

Richards (1991) observes that waste tips may be the only evidence of quarrying activities at long abandoned extraction sites. Several additional factors combine to exacerbate the inherent problems associated with slate waste. These are that extraction processes rarely proceed in a systematic fashion, and that waste tips are viewed as potential future assets by quarry managers (for aggregate and rough block). This ensures that quarry operations continue for many decades, and almost entirely rules out the possibility of initiating restoration programmes (DOE, 1995). However, these issues could be easily addressed and potentially open up pathways for step-by-step programmes of restoration at UK slate quarry sites.

Because of the landscape impact of tipping waste slate and the financial potential of using slate waste, many quarrying operations have increased their efficiency and now sell a greater proportion of extracted slate than at any period in the past (DOE, 1995). Delabole Slate Ltd. in Cornwall has demonstrated this, and now, by improved resource utilisation, the company is able to sell virtually all materials extracted from its quarry (DOE, 1995; Lott *et al.,* 2005). This has minimised the waste arising from poor quality slate or processing materials, and removes the need to develop suitable disposal options.

The quantity of slate waste produced during the initial stages of extraction is largely determined by the method employed to cleave the rock from the slate vein. Early methods of extraction, as used in the eighteenth and nineteenth centuries, relied upon selective blasting with black powder and manual handling of slate blocks and waste. The quantities of waste generated by this method depended on the slate quality and the skill of the quarrymen; the waste generally contained very low quantities of fines (DOE, 1995). During the 1960s and 1970s "high blast" methods of slate extraction were introduced, with massive sections of quarry faces being shattered by blasting and removed by large quarry vehicles (DOE, 1995). This technique is still used at Penrhyn Quarry today. Although both rapid and inexpensive, this method is hugely wasteful, generating large volumes of mixed-size waste slate, including blocks weighing up to 15 tonnes (DOE, 1995). Modern techniques are now being integrated to increase the efficiency of the slate extraction process. Examples of these techniques are: more careful blasting with minimum charges to exploit natural faults and weaknesses in the slate veins, therefore minimising damage and waste generation; use of wire and chain sawing technology capable of extracting rectangular block up to 300 tonnes in size, with minimal waste; and computer control systems incorporating laser guidance to maximise the number of slates cut from quarried block (DOE, 1995).

Of the slate producing regions, some have fewer waste disposal issues than others. Waste from quarries located in coastal situations, for example the Scottish "slate islands" Easdale and Ellanabeich, north Pembroke and north Cornwall, can be tipped directly into the sea (DOE, 1995). This is not to suggest that tipping of slate waste into the sea is problem-free, but it does avoid visual intrusion upon the landscape.

The total impact of slate quarrying in the UK was quantified in a report by the DOE (1995), which stated that 300 hectares of former slate workings have been reclaimed (by all schemes, not just ecological restoration), 600 hectares remain derelict, and 1500 hectares have the potential to become derelict.

2.5 Restoration ecology

Ecosystems and habitats that suffer any form of disturbance resulting in conditions different to those present prior to the disturbance event can be called damaged. To make good this damage, ameliorative activities are often carried out; this may consist of rehabilitation, reclamation or even creation. Each method is used to achieve slightly different objectives; for instance, rehabilitation involves the reparation of ecosystem processes, services, and productivity; reclamation tends towards land stabilisation, public safety concerns, aesthetic improvement, and reinstatement of land-uses; and ecosystem creation is focussed more on mitigation after severe or complete disturbance to ecosystems or habitats (SER Int., 2004). All of these forms of land amelioration involve the application of theory derived from the science of ecology (Huttl and Bradshaw, 2000). An understanding of the principles of ecological theory will increase the success of restoration schemes (Young *et al.*, 2005).

The roots of ecological restoration date back many decades (Young *et al.*, 2005); an example is the publishing of "Restoration and Management Notes" by the University of Wisconsin more than 26 years ago (Davis and Slobodkin, 2004). Early restoration practice was more focussed on specific land management issues, for example, erosion control and range improvement (Young *et al.*, 2005). Later on, in 1988, the Society for Ecological Restoration (SER) was founded, and in 1994 the first issue of the journal "Restoration Ecology" was published by SER (Davis and Slobodkin, 2004). Within the last 15 years the discipline of restoration ecology has become a recognised academic field, with the regular publication of peer-reviewed research articles, representing a move away from the work of early "restorationists" who identified this novel field as an ideal opportunity to test ecological theory in real-world settings (Young *et al.*, 2005).

Ecological restoration is "the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed" (SER Int., 2004). Other definitions of a similar kind can be found throughout the relevant literature. Ecosystems can be degraded, damaged, transformed or entirely destroyed by both human activities, such as minerals exploitation, and by naturally occurring processes such as volcanism, landslides, or glacial retreat (Huttl and Bradshaw, 2000). Areas affected by these processes may suffer severe damage. Common features of such devastated ecosystems and landscapes are the absence of vegetation, biocoenosis and ecological interactions (Huttl and Bradshaw, 2000).

At the very simplest level, restoration ecology aims to remove or modify the factors causing disturbance, allowing independent ecosystem recovery (SER Int., 2004; Hobbs and Harris, 2001). Restoration strategies involving a higher degree of intervention are also used to reintegrate disturbed and fragmented ecosystems into whole landscapes and re-establish pre-existing links. Such strategies include the intentional implementation of remedial measures to initiate or accelerate recovery and return damaged ecosystems to their historic trajectory of succession and development (SER Int., 2004). The implementation of restoration works can be a long-term commitment, potentially requiring aftercare and management for many decades before an acceptable level of ecosystem recovery is achieved.

The uptake of ecological restoration strategies is sometimes problematic, being dependent on many factors such as the current political climate and, therefore, government backing and availability of funds (Hobbs and Harris, 2001). A perceived barrier to the uptake of ecological restoration methods is the disconnection between these and practical every-day land management activities. Hobbs and Harris (2001) report that there appear to be clear distinctions forming between restoration ecology and land management and conservation, when in reality they should complement one another. Young *et al.* (2005) suggest that land and resource managers require guidance from restoration ecologists on ways of practically and effectively achieving restoration goals. As pointed out by Hobbs and Harris (2001), restoration ecologists can do this by explaining conceptual frameworks detailing how ecosystems work and how restoration activities will influence the onset of restoration. Much real-world restoration is carried out in the absence of academic input, but increased consultation between practitioners and ecologists can provide useful information on the relative merits of different restoration schemes and restoration options, their potential costs and the probability of their success. Young *et al.* (2005) are optimistic; reporting that there is increasing evidence of practitioners employing greater levels of ecologically-driven strategies.

Ecosystem recovery can be signified by the development and demonstration of certain functions and attributes. The SER set out nine attributes of restored ecosystems (SER Int., 2004). Restored ecosystems will: contain sufficient biotic and abiotic resources to allow continual development in the absence of management processes; contain characteristic indigenous species and assemblages; be self-sustaining in structure and function; be resilient to normal environmental stresses; be capable of sustaining reproducing populations; interact with adjacent habitats and ecosystems within a landscape, allowing uninterrupted biotic and abiotic flows; have had the causatives of degradation or damage removed or eliminated.

To judge how restoration processes are progressing, or to establish how successful different restoration methods have been, the use of reference sites is important. Reference sites (or just “the reference”) provide model conditions from which restoration projects can be planned, monitored, and evaluated (SER Int., 2004). The reference may be provided by un-degraded and intact habitats or ecosystems directly adjacent to the damaged site in need of restoring, thus offering direct comparison of structure and dynamics (Hobbs and Harris, 2001). It is suggested (SER Int., 2004) that reference conditions are best assembled from a range of relevant sites and resources, for example local species lists, maps, aerial photos, habitat remnants, herbaria and local knowledge, providing as comprehensive a set of information as possible.

Small areas of undisturbed land supporting vegetation typical of the site are often abundant at mineral extraction sites, for example in areas near site boundaries and between extraction sites. These habitat remnants are valuable, providing examples of the reference habitat and vegetation type, and sometimes creating links with undisturbed land adjacent to the extraction sites (DOE, 1995).

It is also suggested (SER Int., 2004) that studies of the historic trajectory of succession and development of an ecosystem can provide good knowledge of baseline conditions from which restoration projects can be planned. If restoration projects attempt to restore damaged ecosystems to their historic trajectories, knowledge of past conditions can identify the point on the trajectory that restoration projects should aim for. Information on historic trajectories can be gained from pre-existing ecosystem structure and composition, studies of comparable intact sites, and other ecological, cultural, and historic reference data (SER Int., 2004). However, caution is advised when using historic ecosystem trajectories as models for restoration projects, as it is not always the case that ecosystems will recover to pre-disturbance conditions. Awareness of the fact that alterations in ecosystem trajectories are probable is required.

Hobbs and Harris (2001) consider that too much emphasis is placed upon producing clear explanations of what restoration ecology is. Rather than focus on defining the finer points of this field of research, Hobbs and Harris (2001) suggest that it is far more important at the outset of any restoration project for ecologists to set clear restoration goals. These are detailed targets that restoration works should aim to reach in order to achieve successful habitat or ecosystem restoration. These goals will relate to the severity of disturbance and the features and condition of the reference. However, the setting of restoration goals brings with it some problems, for instance: what goals should be set? Should a number of separate goals be set, or should there be a broad outline relevant to the situation and circumstances? An example of these problems relates to the restoration timescale; if the broad goal is to accelerate site restoration over that occurring naturally through regeneration, how fast must restoration be, to be considered successful? Hobbs and Harris (2001) also give some guidelines for setting restoration goals, including: consultation with stakeholders to determine specific expectations and requirements; the availability of finances and resources; consideration of the nature of the degraded site and the factors leading to, or causing, degradation; and the key attributes or functions as represented by the reference site, for example, ecosystem function, structure, composition, vigour, organisation and resilience. It is important to consider all available restoration options whilst goal setting and before the onset of ill-considered and inappropriate practices.

Hobbs and Harris (2001) emphasise that ecosystems are dynamic systems, and it is important to allow a degree of flexibility when putting together goals for restoration projects. It should also be recognised that restoration efforts should aim to produce and protect conditions suitable for and characteristic of the future, ensuring ecosystems are correctly positioned on their relevant developmental trajectory. Emphasis should not merely be directed at restoring past conditions and producing a static system susceptible to future degradation and failure. In addition, very strict restoration goals may prove to be very expensive to implement, in which case the designation of alternative goals is advisable (Hobbs and Harris, 2001).

The use of performance standards derived from reference sites will provide a benchmark for regular monitoring and evaluation, allowing determination of progress towards restoration goals. The SER (SER Int., 2004) propose three methods for effective evaluation of restoration works: direct comparison with reference sites; attribute analysis, assessing components in relation to typical ecosystem functions and qualities possessed by a fully restored ecosystem; and trajectory analysis, comparing observed progression with established trends demonstrated by other restored sites and reference sites. Hobbs and Harris (2001) add to this list of performance standards by recommending the use of monitoring features such as ecosystem productivity and standing crop, cycling of nutrient and size of mineral pools to determine post-restoration establishment success.

Other important, but slightly less controllable, factors may play significant roles in determining restoration success. Young *et al.* (2005) point out that increased importance is now being placed on the role of soil microbes in restoration work. At some restoration sites the soil microbial population may be completely absent, and measures to promote microbial colonisation are required. In contrast, some sites may simply have reduced soil microbial activity, in which case ameliorative activities to stimulate greater activity might be used. The soil microbial population plays an important role in the health of higher plants and numerous ecosystem functions. Further research into the importance of soil microbes may benefit the success of future ecological restoration practices.

Other processes are completely out of the restoration practitioner's control, but may dictate whether some projects succeed or fail. For instance, projects relying upon seed collection are susceptible to year effects, and it is not uncommon to be faced with total reproductive failure in some years (Young *et* *al.*, 2005), an issue that cannot be easily overcome if that species is an important part of the designated restoration goals.

Some researchers do not agree with some of the sentiments and arguments put forward by the restoration ecology community, highlighting terms and concepts such as ecosystem "health" and "integrity" with regard to setting restoration goals. Davis and Slobodkin (2004) claim that restoration ecology focuses more on defining "value-based" goals than on scientifically-based ones, arguing that the designation of ecosystems or communities as either "healthy" or "damaged" has little scientific justification. These authors suggest that the term "ecological architecture" is a more appropriate description of many ecological restoration works. They (Davis and Slobodkin, 2004) also give an alternative definition of ecological restoration ("the process of restoring one or more valued processes or attributes of a landscape"), which avoids any questionable or ambiguous terms involving "health" or "integrity" of ecosystems or communities. It is suggested that this new definition allows restoration practitioners greater scope for setting a range of restoration objectives, for example high levels of diversity and/or productivity as well as improved aesthetic qualities, whilst not merely accepting any type of landscape improvement or transformation as an "ecological restoration".

Increasingly, modern approaches to ecological restoration focus more attention on the social and cultural aspects of restoring ecosystems and landscapes, clearly important given the diverse nature of man's long-term history of living and working on the land. Naveh (1998) states that restoration should broaden its scope to encompass not just ecological but also cultural issues, learning from disciplines other than the natural sciences, such as the arts and humanities. It is claimed that this would lead to a post-industrial symbiosis between human society and nature, allowing more efficient ecological and cultural restoration. As part of effective cultural (landscape) restoration, historic and cultural values of ancient and traditional landscapes should be restored, recognising the importance of relationships between natural and human systems (Naveh, 1998). To facilitate effective cultural restoration in the context of landscape restoration, information on the land history and of its flora and fauna, in both ecological and ethno-historical and anthropological-cultural contexts, must be considered.

The implementation of ecological restoration practices is seen as of great importance. Both Naveh (1998) and Hobbs and Harris (2001) concur, stating that greater integration of culture and nature, and the restoration of Earth's industrially damaged ecosystems and their associated services, will divert the trajectory of both biological and cultural evolution away from extinction, and maximise our potential for continued survival.

2.6 Some examples of restoration methods

Many different methods have been tested for their suitability in promoting habitat restoration in a multitude of situations and environments. However, some treatments appear to be environmentally undesirable, for example the application of tar and seed to china clay waste surfaces in southwest England (Noble, pers. comm. 2005). Perhaps, therefore, the maxim for restoration methods should be that until tested and the results analysed, all restoration techniques should be considered.

Prior to excavating an area for mineral extractive purposes, it is good practice to salvage and stockpile topsoil and over-burden substrates. Set-aside topsoil and substrates can then be applied to areas destined for restoration and re-vegetation when disturbance by extraction stops. This approach allows plant development from the seed bank and no additions of foreign soils, substrates, or composts to sites (Griffith and Toy, 2001; Sarrailh and Ayrault, 2001; Petersen *et al.,* 2004). Topsoil and substrates may also be sourced locally (e.g. construction sites where foundations have been dug), with the intention of using them for restorative application to disturbed sites. Paradelo *et al.* (2007) state that the practice of importing and spreading topsoil is quite common in restoration projects, although topsoil may be in short supply, has high transport costs, and is of variable quality. Martinez-Ruiz *et al.* (2007) report on an example of importing topsoil; a 30 cm layer of fine textured material, collected at 10 – 150 cm depth from a local extraction pit, was spread over waste from mined uranium ore. It was anticipated that vegetation would establish from seed and plant material already present within the topsoil (i.e. the seed-bank and living propagules).

There is increasing interest in the restoration potential of waste fractions of minerals processed at extractive sites. Fraser and McBride (2000) point out that aggregate producers could substantially lower both waste disposal and rehabilitation costs, as well as environmental impacts, if waste minerals, substrates and by-products were incorporated into onsite rehabilitation schemes. The authors describe work on the use of processing fines, collected from crushing and washing activities, for restoration of disturbed areas at dolomite quarries. This approach is also discussed by Dudeney *et al.* (2004), who state that the effective management of bulk secondary aggregates and wastes is a key component of sustainable development. They have studied the co-utilisation of non-toxic bulk mineral wastes and organics to create artificial topsoils which can replace soils lost through industrial processes, promote initial vegetation establishment and facilitate site restoration.

Dudeney *et al.* (2004) also comment upon the use of materials such as sewage sludge, ash, limestone and other materials in mine-site re-vegetation. Materials such as these, and others derived from plant residue, rock fragments, woodchips or gravel, may be added in mulch form (Agassi and Ben-Hur, 1992) or can be combined to form artificial topsoil. Gao *et al.* (2007) describe an application method adapted from hydro-seeding technology (see chapter 5), called outside soil spray seeding (OSSS). This method involves spreading a thin, continuous layer of a soil, wood mulch, fertiliser, stabilising agent, cement and seed mixture on to bare rock slopes, in a highly efficient and fast, mechanised fashion, as with hydro-seeding. The authors report on OSSS in China, suggesting that the technique has the potential for restoration of steep cut-slopes, although observations show that after approximately a year, deficiencies in material integrity and available plant nutrients cause the mixture to flake from the rock surface, and remaining vegetation to degrade. The authors increased the percentage of cement included in the mixture to increase the viscidity and aid it's adherence to steep rock slopes and added a water absorbent polymer. The inclusion of cement as an ameliorant is interesting; it successfully increased the viscidity of the applied mixture, but cement is well known for its high alkalinity and could pose a hazard to plant matter if present in high concentrations.

Paradelo *et al.* (2007) describe the development of a sustainable restoration treatment utilising only recycled constituent materials. Work was carried out at slate quarries in south-eastern Galicia, Spain, where it is estimated 120,000 tonnes of slate processing fines are produced annually as a result of sawing and cutting. By utilising this substantial resource of fine-grained waste material to cover slate waste tips, it was hypothesised that conditions more conducive to vegetation establishment, and therefore the first steps towards restoration, might be achieved. Despite offering a suitable physical substrate for vegetation development, and despite having appreciable concentrations of calcium and potassium, slate processing fines contain little organic matter, nitrogen and phosphorus and show little microbial activity. To overcome these limitations the researchers investigated the addition of another locally available resource, vermi-composted spent grape marc, a waste product from the wine production industry. This material successfully increased total nitrogen, available calcium, magnesium, potassium and phosphorus, significantly enhanced biological activity, decreased bulk density and increased water-holding ability. Plant productivity in this mixture was significantly higher than in slate processing fines alone. It was therefore concluded that a mixture of this nature, with components that are by-products of local industrial practices, could be successfully employed to negate the environmental impacts resulting from quarrying and mining activities. Paradelo *et al.* (2007) state that the slate processing fines and grape marc vermi-compost mixture could help to establish permanent vegetation cover, as a first step towards reclamation.

Other options involve some form of engineering; both hard and soft engineering techniques have produced successful results. Griffith and Toy (2001) report on work carried out by Silva (1993) (cited in Griffith and Toy, 2001), who achieved successful re-vegetation of steep slopes at a Brazilian iron ore mine by staking jute sacks filled with soil, fertiliser, organic material and seeds to the pit slopes. Development of this technique led to the establishment of a cottage industry producing biodegradable mats intended for re-vegetating steep slopes, a technique which is still commonly used by highway and rail departments in Brazil. A recent adaptation of this technique has been the inclusion of seeds into the matting, producing sophisticated restoration geo-textiles. Petersen *et al.* (2004) report on a similar fabricated restoration material, nylon-netted excelsior erosion control cloth. This material consists of a 6 – 7 mm thick layer of aspen fibre mulch, held together, and to land surfaces, by decomposable nylon netting, applied over ground previously seeded and raked. This technique can be adapted so that the aspen fibre mulch is held together with tackifying glue rather than nylon netting.

Hard engineering techniques encompass a range of activities. There are stark contrasts between the work of Lee *et al.* (1996), for example, in which a high level of engineering intervention was utilised to create concrete drainage and planting trenches to protect steep mudstone slopes in Taiwan, and less invasive/intrusive techniques such as those employed by Wheater and Cullen (1997) and Cullen *et al.* (1998). The latter authors report on work carried out at abandoned limestone quarries in Derbyshire, UK, where simple blasting processes were carried out to create scree and other landforms conducive to restoration. Restoration blasting utilises selective drilling and blasting of quarry faces, aiming to replicate natural features, in this case Daleside landforms, more rapidly than they would form naturally. Blast piles were subsequently covered with a shallow layer of limestone tailings, and hydro-seeded with a seed mixture of Daleside species. This demonstrates the potential for co-utilisation of technologies when designing restoration projects, effectively blending damaged landforms into the landscape, and maximising the potential for successful re-vegetation. Jim (2001) reported on similar restoration activities in granite quarries in Hong Kong. Controlled restoration blasting of vertical quarry faces was used to produce debris for scree slope formation at the foot of rock faces. Scree slopes were capped with a layer of fine earth amended with organics (unspecified), followed by a thick (300 – 450 mm) application of decomposed granite enriched with organics (unspecified), providing a suitable substrate for restoration planting. It is suggested by this author that pulverised fuel ash from coal-fired electricity generation could be utilised to amend the physical properties of decomposed granite scree. This again demonstrates the combination of technologies and techniques when undertaking restoration activities on severely damaged sites. It is this combined and considered approach that is likely to produce the greatest success in restoration ecology, and is strongly advised for practitioners.

Work carried out by Lee *et al.* (1996) involved far greater levels of engineering intervention than simply blasting rock faces. The whole technique is referred to as "green-coating"; mudstone slopes were first sprayed with liquid asphalt to 2 – 3 mm thickness, and then coated in a 5 mm thick green (for aesthetic purposes) geo-textile asphalt sheeting. These measures were designed to prevent water infiltration into the unstable mudstone slopes. Following the waterproofing work was the installation of a 15 cm thick concrete slab at the top of each slope, increasing the stability of treated slopes. Formations were created in the concrete slabs, providing trenches for drainage and vegetation planting. The cost of this project was reported as $77,000; the total area treated was only 630 m^2, but the results are claimed to be very good. This is a very good example of the positive association between restoration cost and extent of engineering. Cost is usually a limiting factor for many restoration projects, and expensive engineering, unless deemed absolutely necessary for safety reasons, will probably not be carried out unless large budgets are available.

In contrast to work carried out by Lee *et al.* (1996), apart from one clear similarity, is a form of restoration carried out on Mount Laoshoushan, in Kunming, China, where the mountainside was sprayed with green paint (Fei, 2007). Following quarrying, and in the absence of any restoration initiatives, ten workers covered several thousand square metres of hillside with the paint. This was primarily carried out to adjust the "Feng shui" of the area, and cost approximately $210, a low price indeed! However, if that money had been utilised for tree seedlings, several more disturbed mountain sites might have been restored as well.

2.7 Legislation on mineral extraction and restoration

Quarrying and minerals extraction, by their very nature, are hugely destructive practices, often having major impacts at species, habitat, and landscape levels, and with the potential to completely decimate habitats and irreversibly alter landscapes. Because of these significant impacts, quarrying is the subject of extensive legislative and policy guidance (Cripps *et al.*, 2007); no working of minerals can legally be carried out in the UK without planning consent (Harrison *et al.*, 2002).

International law, in the form of the Convention of Biological Diversity as implemented at the 1992 Earth Summit in Rio de Janeiro, provides the main impetus of governmental policy on nature conservation, and dictates that nation states rehabilitate and restore degraded ecosystems (Cripps *et al.*, 2007; Williamson *et al.*, 2003).

In the UK, one of the major outcomes from the Earth Summit was the publication of "Biodiversity: The UK Action Plan (UKBAP)", and later the UK Biodiversity Steering Group Report. These focus upon a number of national action plans for target habitats and species, describing the UK's natural resources and providing plans for their protection (Cripps *et al.*, 2007). The UK Biodiversity Steering Group was succeeded in 2002 by the UK Biodiversity Partnership, following the Millennium Biodiversity Report. The partnership comprises private individuals, business, government and non-government representatives, providing services from funding to expert advice (UKBAP, 2007). The Countryside Rights of Way (CRoW) Act 2000 provides a legislative basis for preparation of Biodiversity Action Plans (BAPs), which are subsequently used to inform policy on biodiversity issues. In 2007 a list of proposed UK Priority Species and Habitats was presented and formally adopted; it lists 1149 species and 65 habitats identified as priorities for conservation action under the UKBAP (UKBAP, 2007). Williamson *et al.* (2003) recommend that all information regarding local or regional BAPs is sourced from local authorities prior to the onset of public consultation and planning for proposed restoration works on quarried sites.

Cripps *et al.* (2007) report that the Government is keen to encourage the private sector to provide species or habitat "champions"; these are companies or businesses that wish to provide support for a BAP species or habitat, in the form of funding, materials or man-power for Action Plan work (UKBAP, 2007). It is suggested by Cripps *et al.* (2007) that quarry restoration projects provide an ideal opportunity for minerals operators to contribute to this process, and that they in turn will receive positive publicity.

Many other laws relating to the British countryside are derived from the National Parks and Access to the Countryside Act 1949. The Act provided the designation for National Parks and Areas of Outstanding Natural Beauty (AONB), and introduced the concept of National Nature Reserves (NNRs) and Sites of Special Scientific Interest (SSSIs). These designations were devised to protect areas of particular importance for floral, faunal, geological or physiographical (landscape) features, and underline national policy with regard to all landscape conservation issues (Cripps *et al.*, 2007). Indeed, it is actually central to UK Government policy that any land damaged as a result of quarrying or extractive operations is restored to a condition suitable for the intended after-use at the earliest possible point in time (Cripps *et al.*, 2007).

Many quarries exist within areas of the UK covered by national landscape and nature conservation designations, such as National Parks. They include quarries for granite on Dartmoor, slate in the Lake District, and sandstone in the Peak District (Lott *et al.*, 2005). The area around Blaenau Ffestiniog in Gwynedd was deliberately omitted from the Snowdonia National Park because of the effects of large-scale slate quarrying (Sheldon and Bradshaw, 1975; DOE, 1995). Any new quarry or mineral extraction developments within protected areas are the subject of rigorous examination, and it is increasingly difficult to obtain planning permission for them (Lott e*t al.*, 2005).

Proposals for mineral workings covering more than 25 hectares, or those adjacent to or located within sensitive areas, must carry out formal Environmental Impact Assessments (EIAs) (and very often supporting environmental studies for the Minerals Planning Authorities (MPA)) in line with the 1999 Town and Country Planning (Environmental Impact Assessment - England and Wales) Regulations (Lott *et al.*, 2005). The limit of 25 hectares is intended to act as a disincentive for smaller operators to expand their workings (Lott *et al.*, 2005).

Two key pieces of European legislation relevant to biodiversity are applicable to land disturbance by quarrying; these are the conservation of wild birds under the Birds Directive 1979 and the conservation of natural habitats and wild fauna and flora under the Habitats Directive 1992. The Habitats Directive advises that by maintaining, managing and, where appropriate, developing features of the landscape that are important for flora and fauna, EU member states can enhance biodiversity (Williamson *et al.*, 2003). These directives are implemented in the UK under the Wildlife and Countryside Act 1981 (and its amendments), which allows designation of NNRs and SSSIs for the protection of areas containing nationally and internationally important species (Cripps *et al.*, 2007). It is an offence, under the 1981 Act to either intentionally or recklessly disturb or harm European protected species.

Also protected under the Wildlife and Countryside Act 1981 are sites designated as SSSIs for their geological or geomorphological importance. More than 2200 such sites have been designated in the UK, including numerous slate quarries (DOE, 1995). Further protection for sites of geological importance can be granted if they are considered to be an example of a Regionally Important Geological/Geomorphological Sites (RIGS) (DOE, 1995). SSSI and RIGS designations for geological features should guarantee that proposed quarrying activities will be assessed for their potential to affect important on-site features.

The 1992 EU Habitats Directive is implemented domestically in the UK by the 1994 Habitats Regulations, under which species listed in Annex II are given strict protection. This Regulation and the 1981 Wildlife and Countryside Act are the main pieces of legislation providing protection for habitats and species. Supplementary protection to areas under these designations is given by the CRoW Act 2000, which bolsters the protection of SSSIs (Cripps *et al.*, 2007).

SSSI status does not, by itself, guarantee protection; what it does do, however, is ensure that due consideration is given to the site during the planning process for proposed developments (DOE, 1995). An important feature of SSSI designation is that a list of Potentially Damaging Operations (PDOs) is identified for each site (DOE, 1995). These might, for example, include activities such as quarrying and minerals extraction; if any such operation were to be planned by the landowner or occupier, application for consent has to be made to the relevant administrative body four months in advance. Applications may be approved, or they may be rejected if regarded as damaging to important features (DOE, 1995).

Details of the legislation applicable to quarries and mineral extraction sites are laid out in Land-use Planning and Development Control notes issued by the Department of Transport, Local Government and the Regions (DTLR), the Scottish Parliament, and the National Assembly for Wales. Guidance is provided in the form of Planning Policy Guidance (PPG) notes and Mineral Planning Guidance (MPG) notes in England, National PPGs in Scotland, Technical Advice Notes in Wales, and Government White Papers that are applicable across the UK (Williamson *et al.,* 2003). PPGs (and their equivalents) offer guidance on both general and specific aspects of planning policy, whilst MPGs deal specifically with the control of minerals developments. Updating and replacement of PPGs and MPGs has recently been carried out, with PPSs (Planning Policy Statements) and MPSs (Minerals Planning Statements) now providing current guidance (Cripps *et al.*, 2007).

Various pieces of legislation deal with the after-use and aftercare of quarry and extraction sites. For example, the Town and Country Planning (Minerals) Act 1981, which has been superseded by the 1990 act of the same name, and amended by the Planning and Compensation Act 1991, is concerned with the enforcement of aftercare conditions. Planning authorities, for example, Mineral Planning Authorities (MPAs), are empowered to impose aftercare conditions that ensure that land is fit for agriculture, forestry or amenity purposes (Cripps *et al.*, 2007).

Section 106 of the 1990 Town and Country Planning Act states that Local Planning Authorities can enter into legally binding agreements or obligations, including restoration agreements, with land developers. These "Section 106 Agreements" formalise the details of what is proposed, required and expected (Nason *et al.*, 2007; Anon, 2006a), and can be used as an instrument for restricting developers, minimising impacts and stipulating mitigation measures.

Aftercare agreements have a statutory running period of five years, commencing from the point at which a soil profile is established, during which the land must be restored to a satisfactory state (Cripps *et al.*, 2007). However, legislation on after-use agreements does not currently specify any standards to be met in aftercare works. As a result, reclamation targets must be prescribed on a site-by-site basis. It is a requirement of quarry planning proposals that they set out an appropriate after-use; the last person or company using the land for mineral extraction is responsible for both leaving the site in a safe and stable state (highlighted in the Quarries Regulation 1999) and the financial obligations of aftercare commitments (Cripps *et al.*, 2007). Leaving a quarry site in a safe and stable state is important regardless of the intended after-use.

A comprehensive list of the legislation applicable to mineral extraction and quarrying is provided by Cripps *et al.* (2007). Some of the acts, not mentioned above include the Mines and Quarries Act 1954, the Protection of Badgers Act 1992, the Water Environment (Water Framework Directive) (England and Wales) Regulations 2003, and the Scottish Planning Advice Note PAN64 (Reclamation of Surface Mineral Workings) 2002.

2.8 Slate waste as aggregate

The majority of aggregate used in England and Wales is derived from primary sources, namely quarried or dredged sand and gravel and crushed quarried rock. The annual total of these primary materials won for aggregate use in the UK is approximately 205 million tonnes, a figure that has been fairly consistent for the past three decades (Ove Arup, 2001). It is estimated that the crushed rock fraction of this total is approximately 116 million tonnes. The market for secondary aggregates and recycled materials is considerably lower, annually totalling approximately 35 million tonnes, and is dominated by recycled materials (Ove Arup, 2001). Annual primary aggregate demand is predicted to increase in the coming years to 370-440 million tonnes (DOE, 1995).

Of the accumulated slate waste in north Wales, the amount available as a source of secondary aggregates is estimated to be 270-371 million tonnes, located primarily at quarries in Bethesda and Blaenau Ffestiniog (Gwynedd) (Ove Arup 2001). The balance of the waste slate resource in Gwynedd, estimated to be 360-460 million tonnes, is considered unavailable for exploitation due to environmental and/or access or transport constraints. However, waste generation is described as "continuing at a faster rate than it can be beneficially used", with annual waste slate generation in Gwynedd of 6 million tonnes (Ove Arup, 2001). Presently, only 275,000 tonnes of this total (less than 5 %) is utilised as aggregate, the remainder being tipped.

The majority of slate waste produced, remains within the vicinity of the producing region because of restrictive transport costs (Ove Arup, 2001). For example, most waste produced in Gwynedd is used in north Wales for road building and general fill applications, such as substrate for the infrastructure of a new industrial estate in Bangor.

Slate waste use for aggregate has perceived benefits, such as the reduction of large and prominent stockpiles that are visually intrusive (Lott *et al.*, 2005). This same large-scale removal of slate waste, however, has the potential to change the character of a site in a detrimental fashion. Assessment of site features important to the local landscape character should be carried out prior to any such alterations (DOE, 1995). Another problem to consider when proposing the removal of slate from waste tips is the environmental impacts associated with the increased movement of traffic (Lott *et al.*, 2005).

2.9 Aggregates levy

The Aggregates Levy was introduced in April 2002 and applies to sand, gravel (including marine dredged resources) and crushed rock (extracted with the primary objective for use as aggregate), and is charged at a rate of £1.60 / tonne and is intended to reduce demand for virgin aggregates and promote the use of recycled alternatives and secondary aggregates usage (BGS, 2005).

A key intended function of the Aggregates Levy was the reduction of the environmental costs associated with mineral workings, for example, noise, dust, vibrations, pollution, impacts of dewatering and discharge, stability, visual intrusion, loss of amenity and damage to biodiversity, in compliance with the governmental statement of intent regarding environmental taxation (BGS, 2005).

The minerals industry has reported that the levy has had an adverse effect upon sales of low-grade aggregates; these are now no longer regarded as being competitive with recycled or secondary aggregates. This has resulted in disposal issues at extraction sites and has seen the stockpiling of huge quantities of mineral wastes (Lott *et al.,* 2005). It has also been suggested that the production of untaxed materials, shale for example, has increased as a result of the introduction of the levy (Gribben, 2005).

Crushed slate waste is exempt from the Aggregates Levy, a decision largely based on the huge stockpiles of waste slate already available, for example in north Wales. The exclusion of slate waste from the levy has seen an increase in its use as aggregate from 440,000 tonnes in 2001 to 728,000 tonnes in 2003 (Lott *et al.,* 2005).

The minerals industry claims that although the levy is worth approximately £350 million annually, it has largely failed to meet its desired objectives of producing environmental improvements. For example, in a newspaper report focusing on the Aggregates Levy, Gribben (2005) states that DEFRA have been accused of using monies generated from the Levy that were destined for environmental projects for other spending commitments. However, the government insists that environmental objectives in areas affected by quarrying are being met, and that of an initial £35 million set aside for England, £29.3 million was specifically ring-fenced to fund 200 environmental projects (Gribben, 2005). Major criticisms aimed at the Aggregates Levy include the fact that, because it is seen as a sales tax and is not based on environmental impacts, there are no added incentives to improve environmental standards and performance at minerals processing sites. In response, the Treasury claims that funding provided by the levy has led to reduced noise, vibrations, dust, and other emissions. But the industry claims that there is no evidence for these improvements, and state that good practice, appropriate legislation and regulation are the main drivers of any observed reduction in environmental impact (Gribben, 2005). The Transport Department appears to agree, claiming that whilst the Aggregates Levy has made recycled materials more viable, it fails to directly address the environmental impacts produced as a result of minerals extraction and quarrying.

2.10 Uses of slate

2.10.1 Finished slate

North (1946) provides an account of why slate is such a valuable material, and the ways in which it can be used. Slate is virtually impervious to water; North (1946) describes a study in which a typical roofing slate was found to absorb less than one six-thousandth of its weight following 24 hours submersion in water; as a result slate is resistant to frost, freeze-thaw effects, and wetting and drying cycles. Slate is both incombustible and a poor conductor of heat and electricity; it does not corrode, is resistant to the action of acids, does not shrink and does not warp. This combination of properties, coupled with its strength (North (1946) states that a "small duchess" (a roofing slate measuring 12" × 22" and one-sixth of an inch (<5 mm) thick), supported weights of 157-178 lbs (71-81 kg), flexing more than half an inch (>12 mm) before it broke) and hardness when in a solid state, allows slate to be employed in a wide range of uses. These include billiards tables, skin-dressers tables, school blackboards, electrical switchboards, vats to contain acid, brewery tanks, aquaria, cisterns, dairy tables, laboratory tables, post-mortem tables, sterilising benches, battery charging benches and monumental work such as headstones; it can also be enamelled and used as a substitute for marble in wash-stand tops, mantelpieces, or similar domestic fittings.

The most widespread use of slate, however, remains as a roofing material. Its resistance to water and weathering, its hardness and its durability ensure that the best quality slate can remain effective and provide a barrier to the atmosphere for many hundreds of years, often outlasting many original roof structures. For example, the St. Asaph Cathedral, north Wales, was re-roofed some 250 years after its creation due to failing timbers; the slates, however, could be re-used (North, 1946).

2.10.2 Slate "waste"

In an unprocessed form, waste slate can be employed as rough block for walling and paving stone, armour-stone for river and coastal protection, and in Snowdonia and some areas of Devon and Cornwall for traditional field walls and earth-stone banks respectively (DOE, 1995).

It could be argued that the most beneficial use of waste slate is as backfill for quarry pits, thus eliminating the need for huge waste tips. Backfilled areas could be developed, to accommodate industrial units or car parks. Slate waste is a very suitable material for backfilling due to its inert nature and resistance to degradation and settlement, as observed at sites where fill depths up to 15 metres have been used (DOE, 1995).

Crushed slate, produced from recovered waste fractions, is a similarly versatile material as when used in a solid form. One of the most large-scale uses of slate waste is in the construction industry; crushed slate can be used to create reconstituted roofing tiles and blocks (North, 1946), for bulk fill to cap landfills or embankment construction (and general capping and blinding substrate), for DOT 1 and Type 2 sub-base (terms for aggregate types used to provide structural support beneath roads and buildings as defined by the Specification for Highway Works, clause 803 (WRAP, 2010)) for road construction, drainage blankets, pipe trench backfill, French-drain fill (DOE, 1995), in ready-mix concrete, and for decorative aggregate (Lott *et al.,* 2005). A DOE (1995) report states that slate waste can be used to produce "expanded lightweight aggregates" and mineral wool; however, these processes are energy demanding and these materials have not proven to be economically viable.

When processed to a very fine degree, slate waste becomes a powder with particles comparable in size to water vapour. In this form it can produce a colloidal suspension with water, and is able to absorb oils and grease (North, 1946). There are many other applications for fine slate powder: as an inert filler in cosmetics such as lipstick, foundation and eye shadow (DOE, 1995; Rendell, pers. comm., 2005); in the manufacture of pills (i.e. to bulk out small amounts of active pharmaceutical ingredients into an easily manageable and consumable sized pill or pillule) (Rendell, pers. comm., 2005); to bulk out rubberoid products such as tyres, mats and shoe heels (North, 1946; Lott *et al.,* 2005); as fillers in linoleum, abrasive soaps, disinfecting powders, and in paint (North, 1946); filler for asphalt and bitumastic coatings for road surfaces, roofing felt, and undersea pipe coatings (North, 1946; DOE, 1995; Lott *et al.,* 2005). Due to the rich clay mineral nature of slate it has a high alumina concentration (17-18 %), and aluminium recovery from slate powder has been considered (North, 1946).

This multitude of methods for using slate waste suggests that there are effective options for utilising the material resource efficiently and sustainably, so that the quantity of material being landfilled or tipped can be significantly reduced.

2.11 Alternative uses for slate quarry sites

Once extractive and operational processes have ceased at slate quarries, numerous activities other than ecological restoration can be observed taking place, these range from functional through educational and recreational to unlawful.

The educational value of former slate workings is considerable and covers fields from geology and industrial archaeology to natural history, ornithology and botany. For school and youth groups, abandoned quarries can serve as valuable outdoor classrooms in which to demonstrate important aspects of industrial history, geography and ecology.

Recreational activities that are commonplace in many former quarries are mountain and motor biking, rock climbing, abseiling, caving and mine exploration, sub-aqua diving, and rescue training (both rope and water based) (DOE, 1995). Quarry based rock climbing and sub-aqua diving are in fact almost synonymous with the abandoned quarries of north Wales, representing, as they do, internationally important centres for these past-times (DOE, 1995). The British Sub-Aqua Federation regard Dorothea Quarry, Nantlle (Gwynedd) as a key location for the sport, as it provides one of only a few locations in the UK that enables winter deep water driving for training purposes (DOE, 1995).

Quarry pits that are not flooded and used for sub-aqua activities provide a potential resource for waste disposal. However, given that the location of slate quarries is often in areas of important environmental quality and sensitivity, this is a highly questionable practice, especially where the waste intended for disposal is leachate-producing. Any type of lining or geo-membrane designed to prevent leaching would be rendered ineffectual by the jagged nature of slate quarry pit sides (DOE, 1995). A more realistic commercial end-use for abandoned quarry sites is for noisy or unsightly practices such as car breaking, which can be largely concealed in quarries, reducing their negative impact upon other land users and residential areas (DOE, 1995).

Various proposals have been put forward for utilising former slate quarries as sites for tree plantations and even energy crops (MIRO, 2004). In the north-west Wales, Cumbrian and Scottish slate regions the slate is far too hard to enable such commercial planting activities to take place. Slate waste tips commonly have slope angles approaching 45°, while the maximum slope angle for viable commercial forestry is 27° (DOE, 1995). However, where softer slate occurs, for example, in the Silurian beds of Llangollen and Glyn Ceiriog, Clwyd, sufficient weathering has occurred to create soft, clay-rich soils in which softwood tree species have been successfully established, demonstrating both good growth and survival (DOE, 1995). Similar findings were reported by Crompton (1967, cited in DOE, 1995) for Corsican and mountain pine, planted in 1954 at Rhyd Ddu by the Forestry Commission; they had largely failed on the most inhospitable areas, but where the slate had weathered sufficiently, trees were found to be both healthy and growing steadily ten years after planting. For effective large-scale commercial planting to realistically take place on former quarry sites, substantial degrees of site preparation would be essential (DOE, 1995).

The Minerals Industry Research Organisation (MIRO) and Cranfield University have published reports on the "Quarries2energy" project (MIRO, 2004). The project was done to investigate the possibility of using former quarry sites to grow crops of short rotation coppice willow (*Salix* spp.) or elephant grass (*Miscanthus* spp.) to provide biomass for power generation from co-firing or combined heat and power combustion. This would be a novel restoration approach and would "enhance socio-economic and environmental benefits and secure sustainable restoration outcomes" (MIRO, 2004). However, concerns are raised in the report about crop yields where there are shallow soils, soils with low water holding capacity, or low concentrations of plant-available nutrients. It is considered that many hard rock quarries, slate quarries in particular, are completely unsuitable for growing bio-energy crops.

Problem activities that can occur in abandoned slate quarries include the illegal fly-tipping of rubbish, builders waste and waste soil. The last is perhaps the greatest problem because of the risk of introducing non-native and/or invasive plant species, for example, Japanese knotweed (*Fallopia japonica*) (DOE, 1995).

Following closure, a few quarries may be used for scaled-down operations to provide material with particular properties (of specific colour, for example), for the historic restoration of buildings or monuments of regional or national importance (DOE, 1995).

3 A review of restoration work carried out at Penrhyn Quarry

Abstract

A number of restoration projects carried out at Penrhyn Quarry (Bethesda, Gwynedd) between the early 1970s and 2002 were re-assessed during 2007 to determine the success and restoration potential of the various techniques and methods employed.

Water-holding (boulder clay and PAM hydrogel) and nutritional (NPK mineral fertiliser and sewage sludge) amendment of slate waste to promote tree and woody shrub survival, establishment and growth by Williamson *et al.* (2009) achieved limited success except where substantial (75 cm depth) applications of boulder clay were used (for example mean tree height (cm)- NPK = 111.65, sewage = 125.52, PAM = 71.37, clay = 193.42).

Hydroseeding, mulching with PAM and the use of various pocket planting techniques (e.g. different sized planting pockets, and planting pocket amendment with slate sand or PAM) tested by Rowe *et al.* (2005 and unpublished) failed completely, achieving 0 % survival within seven years of the experimental onset.

Fertilisation of spontaneously regenerated tree pairs (one of each pair receiving soluble and granular slow-release fertiliser) by Rowe *et al.* (2006) demonstrated only short-term benefits to tree growth. Initial records (October 2001) demonstrated significant growth responses in all but one measurement (tree height percentage increase (p = 0.1234); whereas records for the period October 2001 – January 2007 show no significant growth differences between fertilised and non-fertilised trees. This method is not therefore considered to be an effective option for successful long-term ecological restoration on slate waste tips.

The method developed and used by Sheldon and Bradshaw in 1973/74 for the hydraulic distribution of peat-based slurry onto slate waste tip surfaces,

followed by planting of tree saplings, was considered the most successful in enhancing plant establishment (22 trees surviving 33–34 years after planting), sustained growth (mean crown area percentage change 2002 – 2007 = 76.22 %) and habitat restoration on slate waste.

3.1 Introduction

3.1.1 Penrhyn Quarry

Penrhyn Quarry is situated in Bethesda, Gwynedd, UK, centred at NGR (national grid reference) SH 61950,65275 (53°9'59.8" N, 4°3'59.5" W) (see figure 2.1). Adjacent to the quarry are the Snowdonia National Park and two sites of special scientific interest (SSSI) (Glyderau and Cwm Idwal) (Williamson *et al.,* 2003). The quarry complex ranges from 150 m to approximately 425 m. a. s. l. (metres above sea level) and has a total land area greater than 265 hectares (Rowe *et al.,* 2005).

Within the quarry the land is predominated by sloping slate waste tips and cliffs of quarried rock faces. Slate waste has an angle of repose of approximately 45°. Extractive work within Penrhyn Quarry has focussed mainly upon two large open pits, roughly central in the quarry complex, that back onto each other in a figure-of-eight fashion. This has produced scree slopes and rock faces of all aspects; therefore regardless of prevailing wind direction, different areas exist within the quarry that will always be impacted by wind-stress and associated environmental conditions. Mean annual precipitation is 2472 mm (UKCIP, 2007), mean annual air temperature is 9.3°C, and mean annual soil temperature at 10 cm depth is 9.9°C (Rowe *et al.,* 2005). The geology of the site is quite complex, but is obviously dominated by slate from the Cambrian range (c. 590 million years old), but there are also deposits of boulder clay, Fron Clwyd grit (DOE, 1995) and igneous intrusions of quartz-porphyry (North, 1946). Slate on the quarry site is found in the underlying geology, on exposed quarry faces and in waste tips, where it occurs in very coarse, coarse, and fine size fractions. It is estimated that from 0-30 cm depth, slate waste tips typical of Penrhyn Quarry consist of >83 % void volume (Williamson, pers. comm., 2005).

Penrhyn Quarry has little vegetation cover of any significance. However, small populations of naturally colonised / spontaneously regenerated birch (*Betula* *pendula*), willow (*Salix caprea* × *Salix cinerea* = *Salix* × *reichardtii*) and heather (*Calluna vulgaris* and *Erica* spp.) have begun to establish in un-worked sections of the quarry complex, and species such as foxglove (*Digitalis purpurea*), broom (*Cytisus scoparius*), gorse (*Ulex europaeus* and some *Ulex gallii*), wood sage (*Teucrium scorodonia*), ragwort (*Senecio* spp.), and bramble (*Rubus* spp.) for example, establish freely. Despite these populations, total vegetation cover in the quarry complex is estimated to be less than 5 %. Plant communities that border the quarried area are dominated by dry acid dwarf shrub heath (NVC communities H8 *Calluna vulgaris* – *Ulex galii* heath; H9 *Deschampsia flexuosa* heath; and H10 *Erica cinerea* heath) (upland zone) and semi-natural lowland broad-leaved woodland (W17 *Quercus petrea* – *Betula pubescens* – *Dicranum majus* woodland) (lowland zone) (Rodwell, 1991).

3.1.2 Rationale and objectives

Since the early 1970s numerous researchers (Sheldon and Bradshaw, 1975; Rowe *et al.*, 2005; Rowe *et al.*, 2006; Williamson *et al.*, 2009) have attempted to develop successful restoration strategies to promote vegetation establishment and habitat development on the slate waste tips of Penrhyn Quarry. By re-assessing the experimental work carried out by these researchers it will be possible to determine the long-term success and potential for use in future restoration works of each of the restoration methods used.

Information available from published articles regarding the proposed outcomes, methods and techniques used, modes of monitoring success and the results achieved for each of the restoration methods employed at Penrhyn Quarry will be scrutinised. By repeating the monitoring and analysis methods used in each of the studies it will be possible to evaluate their long-term success. This will, therefore, allow conclusions to be drawn on which restoration method or methods provide the most potential for use in future restoration works at Penrhyn Quarry.

3.1.3 The reclamation of slate waste tips by tree planting (Sheldon and Bradshaw, 1975)

Sheldon and Bradshaw (1975) observed that trees were able to grow without the intervention of man on even the most hostile slate waste tips at Penrhyn Quarry. It was hypothesised that the main reason for the limited numbers of trees present was establishment failure, and that if a seedling was able to successfully establish, subsequent root growth into the underlying slate waste tips would provide access to sufficient water and nutrient resources for reasonable vegetative growth.

It was therefore proposed that the planting of tree saplings into slate waste tips offered a realistic and economic method of restoration. Thereafter, attention was focussed on developing a simple method to overcome the pressures that limit rates of natural colonisation.

The key problems facing natural colonisation were identified as the lack of fines material in the surface of the slate waste tips and their blocky and void-rich nature. This substrate provides little moisture and plants are at high drought risk. It was decided that the planting method should involve minimal movement/activity upon, and therefore disturbance of, the slate waste tip surfaces, thus minimising the risk of burial, damage or removal of establishing saplings. Additional requirements were that tree planting should involve minimal levels of soil amelioration, reducing both economic and practical problems, and that planting sites should be fenced off to exclude grazing animals.

A suite of birch (*Betula* spp., unspecified) growth studies using different substrates including soil, soil-free compost, chemical foams and John Innes Base slow release fertiliser, were carried out in 1973/74. Vegetative growth and survival were greatest in peat-based compost, which is suitable for land application because of its good water-holding capacity and low bulk density.

It was found that 240 g of magnesium, ammonium, phosphate (5:24:10) Enmag slow-release fertiliser and 150 g ground limestone per 36.4 litres of sphagnum peat compost was the optimum planting medium for early winter planting of tree saplings. Three-point-six (3.6) litres of planting medium was found to be the smallest quantity capable of supporting a two-year-old sapling through the first growing season following early winter planting. In this relatively small volume of planting medium, trees showed root extension out of their original planting pocket (a depression created in the slate waste surface into which tree saplings were transplanted) into the surrounding slate waste, decreasing their reliance upon the planting medium. Careful positioning of slates so they were angled around the planting pocket was found to increase the interception of precipitation and, therefore, increase the plant-available water for the planted saplings.

Following these initial trials further experiments were carried out in 1973/74 to overcome the problems associated with hand application of the peat-based planting medium to the surface of slate waste tips. A distribution method using a hydraulic seeding tanker was devised. It was found that if the peat-based medium was made up as a wet slurry and wood cellulose was added to increase the density and flow of the mixture, it could be pumped from the tanker through a series of pipes directly on to the waste tips. Several days (unspecified) after application, to allow for substrate settling, the treated areas were planted (saplings were planted into planting pockets dug into the slurry/waste tip surface) with birch (*Betula* spp., unspecified), oak (*Quercus petraea* and *Quercus rubra*), mountain ash (*Sorbus aucuparia*), goat willow (*Salix caprea*), lodgepole pine (*Pinus contorta*), and alder (*Alnus glutinosa*) saplings. Alder was considered an important restoration species because of its nitrogen-fixing qualities and was inter-planted amongst the other species.

This method allowed rapid distribution of moist planting media with minimal substrate disturbance, and was effective in filling void spaces in the upper surface of the slate waste tips, resulting in a degree of connectivity between the surface and reserves of fines deep within the waste tip.

Early in 2006 dialogue with Dr. E. Rowe revealed that he had carried out some basic measurements (tree height, stem basal circumference and crown diameter) on a small group of trees growing at Penrhyn Quarry in January 2002. These were later confirmed by one of the authors (A. D. Bradshaw) of the initial restoration report (Sheldon and Bradshaw, 1975) to be some of the trees planted as part of the works carried out in 1973/4.

Twenty-three (23) trees were measured by Dr. E. Rowe in January 2002, seven of which were alder and the rest birch. All trees were less than 5 metres in height despite being approximately 30 years old. It was noted that the alder did not demonstrate any superiority over birch in terms of tree height despite the presence of many well-formed root nodules. As a result, it was hypothesised by Dr. E. Rowe that nitrogen was not the limiting nutrient for plant growth in slate waste tips and that another nutrient, possibly phosphorus, might be restricting plant growth.

Re-measurement of the 1973/74 plantings was done to supplement the observations made in 2002 and assess the long-term success of the method used.

3.1.4 LIFE experiment A: Alleviation of both water and nutrient limitations is necessary to accelerate ecological restoration of waste rock tips (Williamson et al., 2009)

Between April and May 2000, 810 native tree and shrub seedlings, and un-rooted cuttings of six native woody species, were planted at Penrhyn Quarry. This planting was part of a three-year European Commission Directorate-General for the Environment LIFE-Environment Programme project carried out by a research team from Bangor University.

The main research objectives of this study were to test methods for improving native woody species establishment rates in the quarry by amelioration of the slate waste with materials that increase water and nutrient availability to plants. The materials used to improve water-holding capacity were cross-linked anionic polyacrylamide (hydro-) gel (Vitagrow, Lancashire, UK) (referred to as PAM) and boulder clay (referred to as clay), which was present on site as overburden overlying the rich reserves of slate. Sewage sludge cake mixed with paper mill sludge (referred to as sewage) and slow release mineral fertiliser (referred to as NPK) were used to increase the concentrations of plant-available nutrients. The organic fertiliser (sewage) was processed to a target C:N ratio of 15:1; it comprised digested sewage cake sourced from Dŵr Cymru / Welsh Water (Caernarfon, UK) and paper mill sludge supplied by the Bridgewater Paper Company (Ellesmere Port, UK). The mineral fertiliser had a N:P:K ratio of 15:10:10 (with 2MgO plus trace elements) (supplied by Osmocote, Ohio, USA). Non-amendment treatments of no additions of water-holding materials (referred to as NoW) and no nutritional additions (referred to as NoF) were also included. The experiment, therefore, incorporated three water-holding amendment treatments (PAM, clay and NoW), three nutritional-amendment treatments (sewage, NPK and NoF), and six native woody species. It used a split-plot design with three blocks. Water-holding treatments were allocated randomly to main plots. Within each main plot there were 18 sub-plots, and the other treatment combinations (three nutritional amendments and six species) were allocated randomly to these sub-plots.

The experimental area, located on an approximately 40-year-old slate waste tip, was prepared by a mechanical backacter excavator, which created a flat space measuring approximately 3000 m^2, situated amongst surrounding large-fraction (≥100 cm diameter) slate waste. The experimental area was fenced to exclude both sheep and rabbits. The experimental site is situated approximately 275 m above sea level with an approximately north-eastern aspect and some moderate sheltering provided by the surrounding waste tips.

All amendments except clay (i.e. PAM, sewage, NPK) were mixed (quantities unspecified) with slate waste in the bottom 5 cm of planting pockets of approximately 3.4 litres (150 × 150 × 150 mm) capacity dug into the prepared area of slate waste. The clay amendment was provided in the form of a 75 cm deep pad sitting on top of the slate waste, into which similarly-sized planting pockets were dug. Nutritional amendments were mixed into the bottom of these pockets as with slate waste pockets.

One seedling (1 to 2-year-old nursery-raised, unspecified local provenance) or hardwood cutting (120-200 × 10-30 mm cuttings of five naturally-occurring quarry willow trees) of six native woody species (alder (*Alnus glutinosa*), birch (*Betula pendula* × *Betula pubescens* = *Betula* × *aurata*), gorse (*Ulex europaeus*), oak (*Quercus petraea*), rowan (*Sorbus aucuparia*) and willow (*Salix caprea* × *Salix cinerea* = *Salix* × *reichardtii*)) was planted or inserted into each planting pocket (five pockets per sub-plot arranged in a quincunx fashion) and assessed over two years; at each assessment, survival, tree height, crown diameter and stem basal diameter were recorded. Other measurements and sampling carried out in 2000 are described by Williamson *et al.* (2009).

The main findings from these early observations were that first year survival and shoot growth were greatest in the clay treatment, while PAM did not improve tree survival or growth. NPK and sewage gave similar growth responses in the first three growing seasons. Clay (without nutritional amendments) was more effective than either sewage or NPK without water-holding amendments in increasing growth.

This experiment was re-measured in May 2007 as part of a subsequent research project (*Treating Waste for Restoring Land Sustainability (TWIRLS)*, funded by the European Commission Directorate General for the Environment, under the LIFE-Environment III programme). A summary of this re-assessment is presented in Nason *et al.* (2007), and a more detailed evaluation is reported here (see section 3.2.2).

3.1.5 LIFE experiment B: initial tree establishment on blocky quarry waste ameliorated with hydrogel or slate processing fines (Rowe et al., 2005); plant establishment following PAM mulching of slate waste tips (Rowe, unpublished)

Rowe *et al.* (2005) investigated the potential of cross-linked anionic co-polymer polyacrylamide (PAM) and slate-processing fines (slate sand) for improving survival rates of trees planted into slate waste tips at Penrhyn Quarry. Unlike the previous LIFE study (see section 3.1.4), in which the experimental area was mechanically prepared prior to restoration, this study focussed upon slate waste tips in an unaltered state. The authors stress the importance of this fact, stating that many sites are difficult to access with large machinery, and that site manipulation may not be possible for reasons of pollution, safety issues, economic constraints, aesthetic considerations or conservation impacts. The authors therefore attempted to develop a method involving minimal surface substrate disturbance and using easily-transportable restoration materials.

The study area identified for restoration planting comprised three penclips (elongate mounds formed by forward tipping of quarried waste) and the large gully areas adjoining them. The penclips were formed in the 1950s and consisted of free-draining, blocky slate waste with minimal fines accumulations. Electric fencing surrounded the study site to exclude sheep. The study site was divided into nine blocks, which were sub-divided into three blocks for each of the three slope directions (approximately south, west and east facing) on the site. In each of the nine blocks, eight plots measuring either 4 m × 5 m or 7 m × 3 m were marked out. The experiment used a randomised complete block design.

Only three of the eight treatments originally implemented are reported by Rowe *et al.* (2005); the results of the remaining five treatments are unpublished and are discussed in further detail below.

The three treatments described by Rowe *et al.* (2005) are as follows. One-year-old, nursery-raised common alder (*Alnus glutinosa*) and Italian alder (*Alnus cordata*), and 12 $\times$ 3 cm (approximate) fresh hardwood cuttings of willow (*Salix cinerea* $\times$ *Salix caprea* = *Salix* $\times$ *reichardtii*), of local provenance (willow cuttings were sourced from a single Penrhyn Quarry donor tree, and alder were of unspecified local provenance), were individually planted/inserted into 3-litre hessian sacks filled with peat-free compost amended with 2.8 g slow-release fertiliser (Osmocote (N:P:K $\cong$ 15:10:10 + 2MgO + trace elements), Ohio, USA) per litre of compost (see Rowe *et al.*, 2005 for details). Two or three of these hessian sacks per species, per plot were then planted into crevices or pockets dug into the surface of the slate waste tips. These pocket-planted trees were subjected to three treatments; no further amendments (control); 50 litres of slate sand placed under the hessian sack-root ball (plus an additional 13.1 g Osmocote per tree); 150 g dry PAM placed under the hessian sack-root ball (plus an additional 13.1 g Osmocote per tree). This experiment was established during March 2001.

Prior to planting, a point survey of the experimental area found only 1.3 % vegetation cover, mostly made up of mosses and small crassulacean acid metabolism (CAM) plants (e.g. *Sedum* spp.).

Measurements of tree survival, tree height and stem basal diameter (see Rowe *et al.*, 2005 for details) were made for three years after the initial planting. Chemical and physical analysis was conducted of newly-applied PAM, slate sand and the compost included in the tree root balls, and on samples collected 42 months after application (see Rowe *et al.*, 2005 for details). Phosphorus content of leaves recovered from the planted Italian alder trees (and of Italian alders established at a deep soil site) was determined.

Results showed that by the second growing season, more trees survived when grown with slate sand than with PAM or without any amendment, and that both species of alder grew better than the willow. Following some periods of drought in 2002-2003 all trees growing without either PAM or slate sand amendments died, and mean survival for all species when grown with slate sand (17%) was higher than when grown with PAM (4 %).

Plants grown with slate sand had higher stem basal area and height than those grown with either PAM or no amendment.

The treatments that were not reported on by Rowe *et al.* (2005) investigated the effect of the size of planting pocket, species and reproductive material types, and hydroseeding and mulching of surface substrates. All treatments except the control (plots that received no amendment and no planting), involved the use of peat-based compost filled hessian bags. Treatments were differentiated by volume of the hessian bags used and the species sown into them. Species used were birch (*Betula* spp., unspecified), oak (*Quercus* spp., unspecified), gorse (*Ulex* spp., unspecified), ivy (*Hedera helix*) and heather (unspecified). Birch was used as both seeds and seedlings (6-month-old), oak, gorse and ivy were all sown as seeds, and heather was included in the hessian planting bags as duff (equivalent to leaf litter). Three sizes of hessian planting bags were used (1, 3 and 10 litres); the 1-litre bags were used for sowing trials, 3-litre bags were used for heather duff and birch seedlings, and the 10-litre bags were used only for birch seedlings. The final treatments were surface mulching orientated: hydroseeding with a coir-and-composted-cotton-waste mixture; and application of PAM gel (anhydrous) at a rate of 100 g m^{-2}. Each mulching treatment was followed by planting of 3-litre, root-balled birch seedlings.

The hydroseeding mixture was applied at a rate equivalent to 28 g seed, 390 g coir, 167 g composted cotton waste, 2.2 g nitrogen, 1.9 g phosphorus, 1.9 g potassium and 8 litres of water per square metre. The seed mixture was designed for amenity and land reclamation use, and contained the grass species *Agrostis tenuis*, *Festuca longifolia*, *Festuca ovina*, *Festuca rubra* and *Poa compressa*, and the legumes *Trifolium repens* and *Lotus corniculatus*.

The application of these treatments was done at the same site and in the same manner as the treatments described in Rowe *et al.* (2005). All treatments were carried out between April and September 2000.

Monitoring was carried out until July 2002, and was done using five permanent 1 m^2 quadrats placed in each plot on three occasions (April, July and September) each year. Variables recorded were plant species, vegetation cover, and invertebrate species presence and abundance (one baited litter trap per plot).

No results of these observations are available.

3.1.6 Fertiliser application during primary succession changes the structure of plant and herbivore communities (Rowe et al., 2006)

Rowe *et al.* (2006) investigated the benefits of fertilisation on the restoration of nutrient-poor primary successional vegetation on slate quarry waste tips. Previous research (Helm, 1995; Elmarsdottir *et al.*, 2003; both cited in Rowe *et al.*, 2006) had shown that fertilisation increases plant establishment and diversity. Substrates that have little organic content have limited early successional plant development because of low associated nitrogen availability (Walker and Syers, 1976; Vitousek *et al.*, 1993; Chapin *et al.*, 1994; Vitousek and Farrington, 1997; all cited in Rowe *et al.*, 2006). It has also been demonstrated that nutrient-deprived plants are likely to have associated afflictions, for instance poor root development and therefore drought stress (Fitter and Bradshaw, 1974, cited in Rowe *et al.*, 2006). Therefore, it was hypothesised that increasing the supply of plant nutrients in fertilisers would enhance plant establishment and development on slate waste tips.

However, some species of high conservation value cannot cope with the greater competition resulting from increased nutrient availability (Davis *et al.*, 1993, cited in Rowe *et al.*, 2006). It has also been shown that fertilisation can affect the structure of vegetation, which may in turn increase the total associated arthropod diversity, but may reduce the diversity of species that are more common in shorter vegetation types (Morris, 2000, cited in Rowe *et al.*, 2006). The nutritional value of fertilised vegetation will be markedly different from that of previously nutrient-deprived plants; this is likely to favour herbivorous invertebrates, but it may change the nature of plant-invertebrate interactions and assemblages; some species might, therefore, be lost (Rowe *et al.*, 2006).

Ten pairs of naturally-established birch (*Betula pendula* × *Betula pubescens* = *Betula* × *aurata*), ten pairs of naturally-established willow (*Salix caprea* × *Salix cinerea* = *Salix* × *reichardtii*) and ten pairs of heath-land plots (≥50 % *Calluna vulgaris* ground cover), each pair occupying similar locations and altitudes, were identified across a range of situations (altitudes, substrate sizes, tip ages, and aspects) at Penrhyn Quarry (see Rowe *et al.*, 2006 for full details). Circular plots of 1.5 m radius centred on study trees, and circular heath-land plots of 3 m diameter were created. One of each pair of plots (randomly assigned) was fertilised on May 23rd 2000 and May 23rd 2001 by evenly distributing soluble (Phostrogen, Bayer Corp., Newbury, UK) and granular slow-release (Osmocote, Scotts UK Ltd., Surrey, UK) fertiliser (50:50 based on nitrogen content), at a rate of 175 kg nitrogen ha^{-1}, 53 kg phosphorus ha^{-1} and 188 kg potassium ha^{-1}, over the plot area.

Records of shoot extension of the longest shoot present on the tree (measured as the growth from the base of the previous year's bud scar to the tip of the terminal bud), tree top height, stem basal area and crown area (calculated from circumference and diameter measurements respectively, and based on assumption of elliptical stem and crown) were made on the birch and willow plots prior to fertilisation in May 2000 and after leaf drop in 2000 and 2001 to assess changes resulting from fertiliser addition. Surveys of ground flora and invertebrates (invertebrate sampling only conducted on tree species), and analysis of soil (birch plots only) and plant samples (tree species only) were also carried out (see Rowe *et al.*, 2006 for full details).

The results of the tree growth measurements showed that fertilised plants had significantly better growth than non-fertilised plants. No measurements were recorded for the heath-land plots.

Leaf nitrogen was significantly higher at all sample occasions in fertilised trees than non-fertilised trees. Extractable polyphenol concentrations decreased in fertilised trees of both birch and willow. No leaf quality analysis was carried out on heath-land plots. Species richness was not altered in either tree or heath-land plots, but the change in total ground cover from 2000 to 2001 was significantly greater in fertilised than in non-fertilised plots for all species. Total invertebrate abundance was greatest in fertilised trees, after both the initial and the second fertiliser application (no invertebrate surveys were carried out on heath-land plots).

Rowe *et al.* (2006) conclude that the application of mineral fertiliser to naturally-regenerated vegetation on slate waste represents both a cheap (at the time of publication) and straightforward (no associated disturbance) method of accelerating succession. Although the use of organic fertiliser or nutrient-rich soils might have reduced environmental impact from nutrient leaching, mineral fertiliser is easier to transport over difficult terrain and poses no threat from introduced weed species. The authors do however point out that the results of this research demonstrate only short-term effects, observable for two years following fertiliser application.

3.1.7 Hypotheses

Re-assessment of restoration methods employed at Penrhyn Quarry from the early 1970s to 2001 will highlight the areas where the greatest levels of long-term success have been achieved. This will:

- Identify which restoration methods have the most potential for taking forward and building upon to develop novel restoration methods to meet current restoration targets as set out by the local planning authorities; and

- Highlight the relative benefits of increasing the water holding capacity and plant available nutrient concentration for plant establishment and habitat development on slate waste tips.

3.2 Methods

3.2.1 Re-measurement of Sheldon and Bradshaw trees, October 2007

The 23 alder and birch trees measured in a January 2002 survey were re-measured on 2nd October 2007. The presence of each tree and its NGR (National Grid Reference) and altitude were recorded by GPS (Global Positioning System).

The original work carried out by Sheldon and Bradshaw was done as a demonstration not a designed experiment. Therefore, the measurements of tree size in 2007 represent a rather *ad-hoc* re-assessment of the success of the methods developed by Sheldon and Bradshaw.

Measurements were made of tree total height (cm) using a telescopic tree measuring pole, stem basal circumference (cm) at 30 cm above ground level (on all stems if the tree was multi-stemmed) using a standard centimetre tape measure, and crown diameters tip to tip (cm) along the longest diameter and at right angles to this, again using a standard centimetre tape measure.

Stem diameter (d_s) was calculated as $d_s=C_s/\pi$, where C_s is the stem circumference, and stem basal area (BA) as $BA=\pi d_s^2/4$. Crown area (CA) was calculated as $CA=\pi d_c/4$, where d_c is the average of the two crown diameter measurements. Calculations for area were based on the assumption that both stem and crown were circular in cross section.

3.2.2 Re-assessment of LIFE experiment A, April 2007

Tree height (cm), crown diameter (cm) and stem basal diameter (mm) were measured using a five metre telescopic tree measuring pole, standard centimetre tape measure and digital vernier callipers respectively. Measurements were carried out during April 2007.

Stem basal diameter was measured in two directions at approximately 10 cm above ground level, or if considerable stem basal thickening had occurred, just above this point. The point measured in initial observations (2000), marked by a dot of white paint, was often still visible. The first measurement was made in the direction indicated by the white dot and the second at right angles to this. If plants had more than one stem all stems were measured. Two measurements of crown diameter were made, the first of the longest crown diameter tip to tip, and the second at right angles to this.

Stem basal area (BA) was calculated as $BA=\pi d_s^2/4$, where d_s is the average of the two stem diameter measurements. Crown area (CA) was calculated as $CA=\pi d_c/4$, where d_c is the average of the two crown diameter measurements.

Tree height was determined by aligning the telescopic measuring pole at ground level with the base of the tree stem and raising the telescopic sections to meet the tip of the main stem. The height was then observed and recorded.

Data presented for this re-assessment work are based on the total number of survivors as recorded during the last experimental observations carried out in 2002. This therefore excludes all trees that died or were destructively sampled during the period 2000 to 2002.

Soil samples were collected from 0 – 10 cm depth in November 2007. One sample, of approximately 100 g (wet weight), was collected from the centre of each plot (162 samples in total). Soil samples were placed into labelled foil trays and placed into a 105°C oven for air-drying for 72 hours. Approximately 10 g of each oven-dried soil sample was then weighed into labelled crucibles. These 10 g sub-samples were then sieved using a standard 2 mm soil science sieve, removing the medium to large fraction mineral content. The remaining soil was re-weighed (DW) into the labelled crucibles and placed into a 450°C muffle furnace for a period of 12 hours. After removal from the furnace, ashed soil samples were allowed to cool for a period of approximately 3 hours, and then re-weighed (ADW). Percentage loss on ignition (LOI) was calculated as LOI = 100 × (DW-ADW) / DW.

3.2.3 Re-assessment of LIFE experiment B, November 2007

Plots containing the three experimental treatments described by Rowe *et al.* (2005) were surveyed for tree survival and growth during November 2007. However, no measurements were taken as there were no survivors.

Of the five additional treatments not reported on by Rowe *et al.* (2005), only plots treated with PAM mulching (100 g m^{-2}) between April and September 2000 were assessed during November 2007. This decision was taken because PAM gel was also used in LIFE experiment A (Williamson *et al.*, 2009), therefore providing continuity of test materials, and also because it is considered that PAM gel offers great potential for use in future slate quarry restoration practices.

Vegetation surveys were conducted by recording species richness and total vegetation cover in each of the nine plots treated with PAM mulching. The number of surviving trees planted into each experimental plot was also recorded.

From these plots, soil samples were also collected, taken from 0 – 10 cm depth where possible. Due to the blocky nature of this site it was not possible to collect the same number of samples from all plots. Sampling was carried out systematically at 1-metre intervals, starting 1 m in from the top edge and 1 m in from the side edge of the plot. However, as shown in table 3.1, very few samples could be collected due to the absence of any fines material in the surface layers of this site.

Table 3.1 Soil samples collected from the LIFE experiment B site, Penrhyn Quarry, November 2007

Block	Plot No.	Dimensions	Potential sample count	Soil samples available
1	13	3 x 7	12	0
2	21	4 x 5	12	0
3	36	4 x 5	12	0
4	46	4 x 5	12	0
5	54	4 x 5	12	0
6	61	4 x 5	12	0
7	76	4 x 5	12	0
8	85	3 x 7	12	5
9	96	4 x 5	12	0

Soil samples were analysed using the method described in section 3.2.2.

3.2.4 Re-measurement of fertilised and un-fertilised tree pairs, January 2007

Using information provided by Dr. E. Rowe and Dr. M. Nason (Rowe *et al.*, 2006) and a handheld GPS device, 57 of the 60 original trees were located. One pair of birch trees and a single willow tree could not be located, probably as a result of disturbance by quarry traffic.

Using the method described by Rowe *et al.* (2006) measurements were carried out in January 2007. Tree height (cm) was measured using a 15-metre telescopic tree measuring pole; the longest crown diameter (cm) tip to tip and the diameter at right angles to this were measured using a standard centimetre tape measure; and stem circumference (cm) was also measured using a standard centimetre tape measure at approximately 10 cm above the ground surface level or just above the area of basal thickening, if considerable.

Stem diameter, stem basal area and crown area were calculated as described in section 3.2.1.

Re-assessment of this experiment was done in collaboration with Miss Tara Froggatt who reported some of the experimental findings in her final year honours project (Froggatt, 2007).

3.2.5 Statistical analysis

No statistical analysis of data collected from reassessment of restoration works carried out by Sheldon and Bradshaw (1975) was conducted due to the non-experimental nature of the initial works.

Data collected from the LIFE experiment A trees in 2007 were analysed using analysis of variance procedures (split-plot analysis of variance and unbalanced design analysis of variance) and chi-squared tests in Genstat version 8.0 (VSN Int. Ltd., 2007). Log transformation of stem basal area and crown area data was carried out, and missing data points resulting from mortalities and destructive sampling were removed from the analysis. With the exception of the three-way interaction (water-holding amendment × nutritional amendment × restoration species) for stem height, crown area and stem basal area, all statistical tests consistently demonstrated the same levels of significance. Significance is only indicated in the results where the three different methods of analysis (split-plot analysis of variance, unbalanced design analysis of variance and chi-squared test) used all gave the same qualitative result.

No data were collected from the three treatments in LIFE experiment B described by Rowe *et al.* (2005) (see section 3.2.3). Data collected from plots treated with PAM mulching (Rowe *et al.*, unpublished) were not statistically analysed because no comparative data were available.

Analysis of data collected from the fertiliser application experiment (Rowe *et al.*, 2006) was carried out using the one-way analysis of variance (ANOVA) and independent samples T-test procedures in SPSS version 14.0 (SPSS Inc., 2007). All data were normally distributed and of equal variance.

3.3 Results

3.3.1 Survival and growth of Sheldon and Bradshaw trees

Table 3.2 shows that 22 trees remained from those measured in January 2002, a loss of just one alder in the five-year period between measurements. Location data are provided so that future measurements can be easily facilitated.

(N.B. In tables 3.2 – 3.5 "*" indicates that tree 6 was missing.)

Table 3.2 Information (UK NGR and altitude (metres above sea level)) for surviving trees planted by Sheldon and Bradshaw 1973/74

Tree	Species	UK National Grid Ref.	Altitude (m.a.s.l.)
1	Alder	SH 61717,65204	223
2	Alder	SH 61709,65187	223
3	Alder	SH 61693,65178	223
4	Birch	SH 61700,65182	226
5	Alder	SH 61709,65191	227
6	Alder	*	*
7	Birch	SH 61700,65177	227
8	Alder	SH 61713,65187	228
9	Alder	SH 61690,65141	228
10	Birch	SH 61704,65171	229
11	Birch	SH 61712,65192	229
12	Birch	SH 61717,65188	229
13	Birch	SH 61713,65191	229
14	Birch	SH 61703,65183	224
15	Birch	SH 61706,65184	224
16	Birch	SH 61709,65191	225
17	Birch	SH 61713,65192	229
18	Birch	SH 61719,65193	230
19	Birch	SH 61721,65192	230
20	Birch	SH 61720,65189	230
21	Birch	SH 61710,65175	228
22	Birch	SH 61710,65180	228
23	Birch	SH 61711,65185	229

Stem basal area, crown area and height of each tree in 2007 are presented in table 3.3, and the mean for each species in table 3.4. There was substantial variation in growth among individuals. Tree height in alder ranged from 254 cm to 488 cm, and from 86 cm to 681 cm in birch. Ranges were similar for stem basal area and crown area. In general, as demonstrated in table 3.4, alder trees had larger stem and crown areas than birch, but birch trees were taller than alders.

Table 3.3 Stem basal area, crown area and tree height in 2007 of surviving trees planted by Sheldon and Bradshaw 1973/4

Tree	Species	Stem basal area (cm^2)	Crown area (m^2)	Tree height (cm)
1	Alder	207.70	9.27	301
2	Alder	207.70	16.12	416
3	Alder	53.79	2.76	254
4	Birch	42.10	3.78	310
5	Alder	282.10	14.86	427
6	Alder	*	*	*
7	Birch	81.49	7.00	460
8	Alder	325.95	36.42	488
9	Alder	138.86	7.55	312
10	Birch	81.49	4.51	415
11	Birch	53.79	3.70	305
12	Birch	13.45	1.70	227
13	Birch	91.99	5.62	415
14	Birch	71.62	4.95	495
15	Birch	49.74	4.15	361
16	Birch	43.95	4.36	415
17	Birch	108.94	13.33	572
18	Birch	133.77	12.16	464
19	Birch	23.00	2.53	176
20	Birch	3.36	1.07	86
21	Birch	45.84	3.51	361
22	Birch	23.00	2.22	271
23	Birch	81.49	12.85	681

Table 3.4 Mean stem basal area, crown area and height in 2007 of surviving birch and alder trees planted by Sheldon and Bradshaw 1973/4

Species	Measurement		
	Stem basal area (cm^2)	Crown area (m^2)	Tree height (cm)
Alder	202.68	14.50	366.33
Birch	59.31	5.46	375.88

Table 3.5 shows the change in stem basal area, crown area and height of individual trees between 2002 and 2007. Table 3.6 shows the mean values for the two species. The majority of trees performed at least reasonably well over this period; for example tree 10 showed an increase of 93.57 % in stem basal area, 110.69 % in crown area and 10.37 % in tree height. Some trees showed substantial increases in some variables and not in others; for example tree 23 showed a 0 % change in stem basal area but a 391.26 % increase in crown area and a 48.04 % increase in tree height (table 3.5). Some trees are recorded as being smaller in size than in the previous measurements

recorded in 2002. For example tree 21 is 7.84 % smaller in terms of stem basal area and 2.17 % smaller in tree height; this is likely to result from operator error associated with the recording of measurements by different researchers.

There was little difference between alder and birch in terms of percentage change in stem basal area and tree height from 2002 to 2007. However, the percentage change in crown area was much higher in birch (76.22 %) than alder (48.24 %) (table 3.6).

Table 3.5 Percentage change (2002 – 2007) in stem basal area, crown and height of trees planted by Sheldon and Bradshaw 1973/74

Tree	Species	Stem basal area % change	Crown area % change	Tree height % change
1	Alder	16.05	-3.13	3.79
2	Alder	29.06	60.56	37.75
3	Alder	-7.27	21.65	1.60
4	Birch	4.49	4.23	-3.13
5	Alder	75.56	38.22	22.00
6	Alder	*	*	*
7	Birch	21.76	17.82	12.20
8	Alder	60.61	145.08	18.16
9	Alder	19.68	27.07	20.00
10	Birch	93.57	110.69	10.37
11	Birch	8.16	37.59	3.39
12	Birch	0.00	50.06	12.38
13	Birch	16.50	47.84	18.57
14	Birch	70.13	131.41	23.13
15	Birch	4.12	50.47	-7.20
16	Birch	38.06	19.98	25.38
17	Birch	0.00	95.05	25.71
18	Birch	49.79	234.98	16.29
19	Birch	37.46	-0.55	13.55
20	Birch	164.06	-61.39	26.47
21	Birch	-7.84	46.06	-2.17
22	Birch	0.00	44.00	4.23
23	Birch	0.00	391.26	48.04

Table 3.6 Mean percentage change (2002 – 2007) in stem basal area, crown area and height of trees planted by Sheldon and Bradshaw 1973/74

Species	Measurement		
	Stem basal area % change	Crown area % change	Tree height % change
Alder	32.28	48.24	17.22
Birch	31.27	76.22	14.20

3.3.2 LIFE experiment A- survival, growth and LOI

Survival in 2007 of planted tree and shrub species is presented in table 3.7. For the five-year period 2002 – 2007 survival rates were very good, not dropping below 96 % for any individual treatment or species. The most successful amendments were clay (water-holding amendment) and sewage (nutrition amendment), with two and three mortalities across all species respectively. The highest survival rate among the test species was shown by alder (100 %).

Table 3.7 Percentage survival in 2007 of trees present in 2002 in three water-amendment treatments, three nutritional amendment treatments and six species. NoF - no nutritional amendments; NoW - no addition of water-holding materials; PAM - polyacrylamide gel

Amendment	Treatment	% survival
Water	NoW	97.04
	PAM	96.67
	Clay	98.95
Nutrition	NoF	96.47
	NPK	97.85
	Sewage	98.37
Species	Alder	100.00
	Birch	96.61
	Gorse	98.40
	Oak	96.69
	Rowan	97.26
	Willow	96.43

Tables 3.8 and 3.9 show that none of the interactions were particularly detrimental to survival. The lowest survival rate (90.48 %) was observed when rowan was grown in the NoF (no nutritional amendments) treatment.

Table 3.8 Percentage survival in the three nutritional-amendment treatments of the three water-amendment treatments. Abbreviations as in table 3.7.

Water treatment	Nutrition treatment	% survival
NoW	NoF	94.12
	NPK	100.00
	Sewage	96.77
Clay	NoF	98.41
	NPK	98.51
	Sewage	100.00
PAM	NoF	96.43
	NPK	95.24
	Sewage	98.36

Table 3.9 Percentage survival of six species in (A) three water-holding and (B) three nutritional-amendment treatments. Abbreviations as in table 3.7.

A

Species	Water-holding treatment	% survival
Alder	NoW	100.00
	Clay	100.00
	PAM	100.00
Birch	NoW	95.00
	Clay	100.00
	PAM	94.44
Gorse	NoW	97.62
	Clay	97.50
	PAM	100.00
Oak	NoW	97.44
	Clay	100.00
	PAM	92.11
Rowan	NoW	95.45
	Clay	96.00
	PAM	100.00
Willow	NoW	95.45
	Clay	100.00
	PAM	92.86

B

Species	Nutrition treatment	% survival
Alder	NoF	100.00
	NPK	100.00
	Sewage	100.00
Birch	NoF	93.75
	NPK	95.45
	Sewage	100.00
Gorse	NoF	100.00
	NPK	97.67
	Sewage	97.50
Oak	NoF	92.31
	NPK	100.00
	Sewage	97.56
Rowan	NoF	90.48
	NPK	100.00
	Sewage	100.00
Willow	NoF	100.00
	NPK	93.33
	Sewage	96.55

Figures 3.1 and 3.2 show the mean tree height, crown area and stem basal area in the three water-holding and three nutritional-amendment treatments respectively. Trees in the NoW and PAM treatments showed very similar, poor growth (figure 3.1), and trees in the NoF and NPK treatments showed similar growth (figure 3.2). Of the water-holding treatments, amendment with clay resulted in far better growth than non-amendment (NoW) and amendment with PAM. Although the effect was not as great, nutritional amendment with sewage produced better growth than NoF and NPK treatments. There were significant differences among treatments for all growth variables measured (water-holding treatments all $p \leq 0.001$; nutritional treatments all $p \leq 0.001$).

Figure 3.3 shows the mean tree height, crown area and stem basal area of each species. No one species was consistently superior in all growth variables, although alder and birch had higher values than all other species. There were significant differences among species in tree height ($p \leq 0.001$), crown area ($p \leq 0.001$) and stem basal area ($p \leq 0.001$).

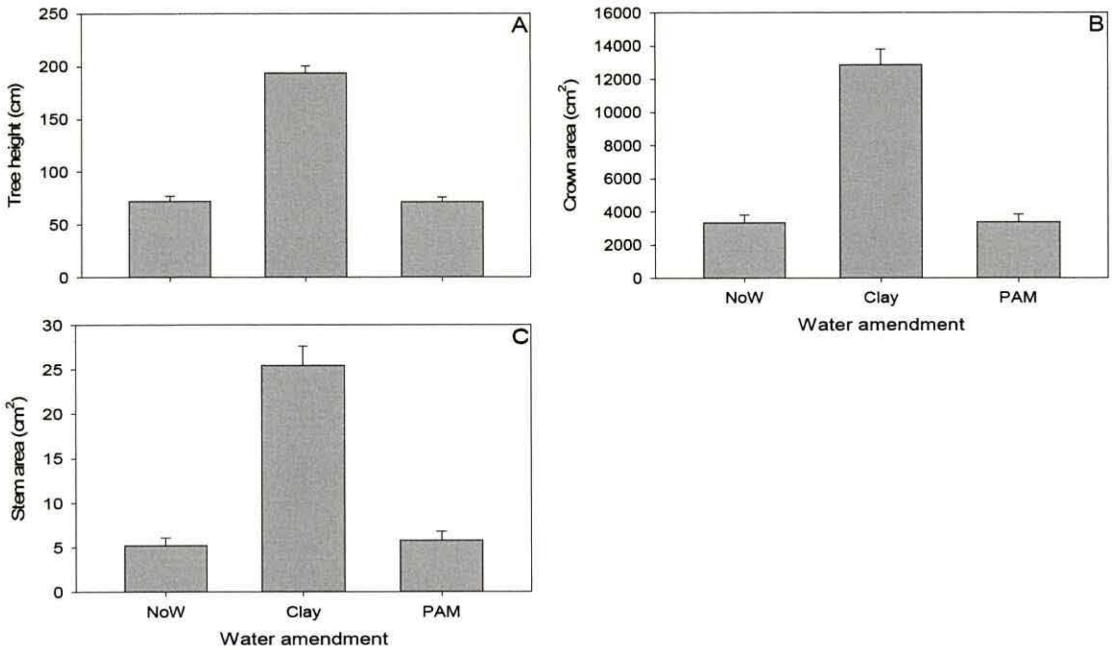


Figure 3.1 Effect of water-amendment treatments on mean tree height (A), crown area (B) and stem basal area (C) (PAM = polyacrylamide gel, Clay = boulder clay, NoW = no amendment with water-holding materials) (bars = standard error of the mean)

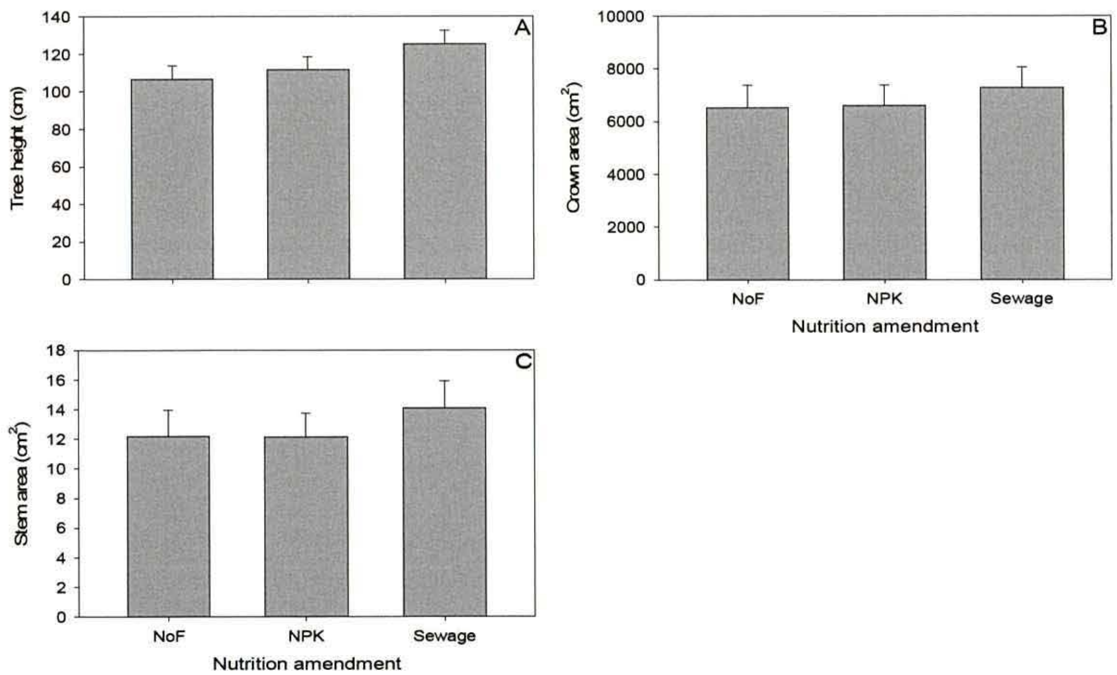


Figure 3.2 Effect of nutritional-amendment treatments on mean tree height (A), crown area (B) and stem basal area (C) (NPK = Nitrogen:Phosphorus:Potassium mineral fertiliser, Sewage = sewage sludge cake mixed with paper mill sludge, NoF = no amendment with nutritional materials) (bars = standard error of the mean)

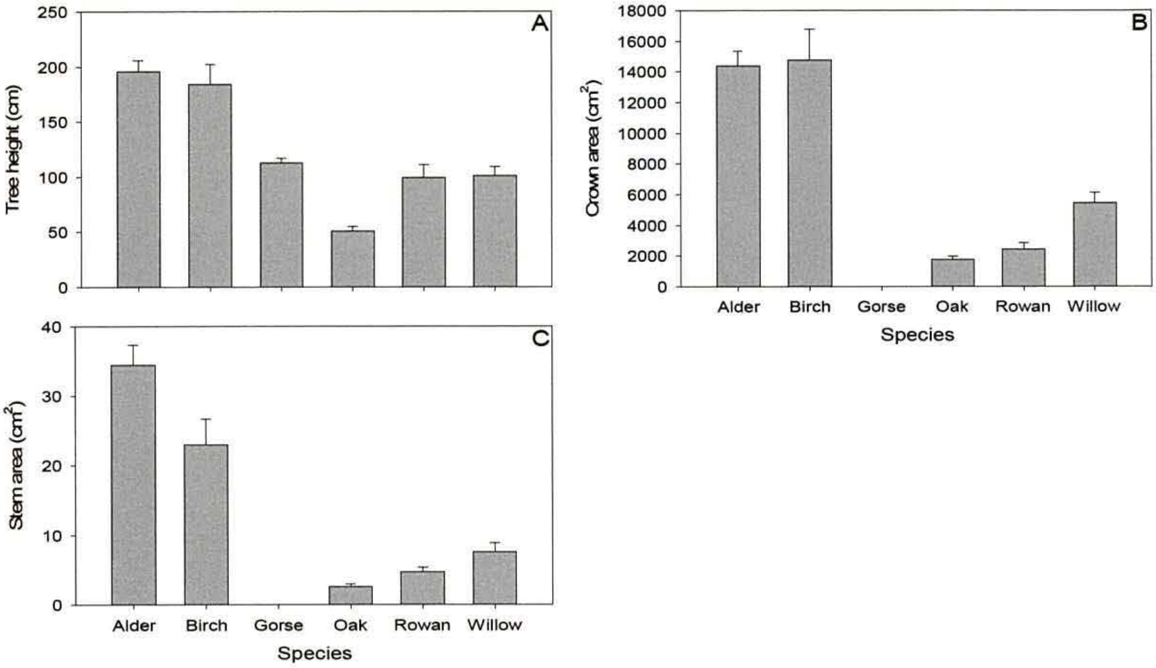


Figure 3.3 Species differences in mean tree height (A), crown area (B) and stem basal area (C) (bars = standard error of the mean). N.B. gorse was not measured for either stem or crown diameter

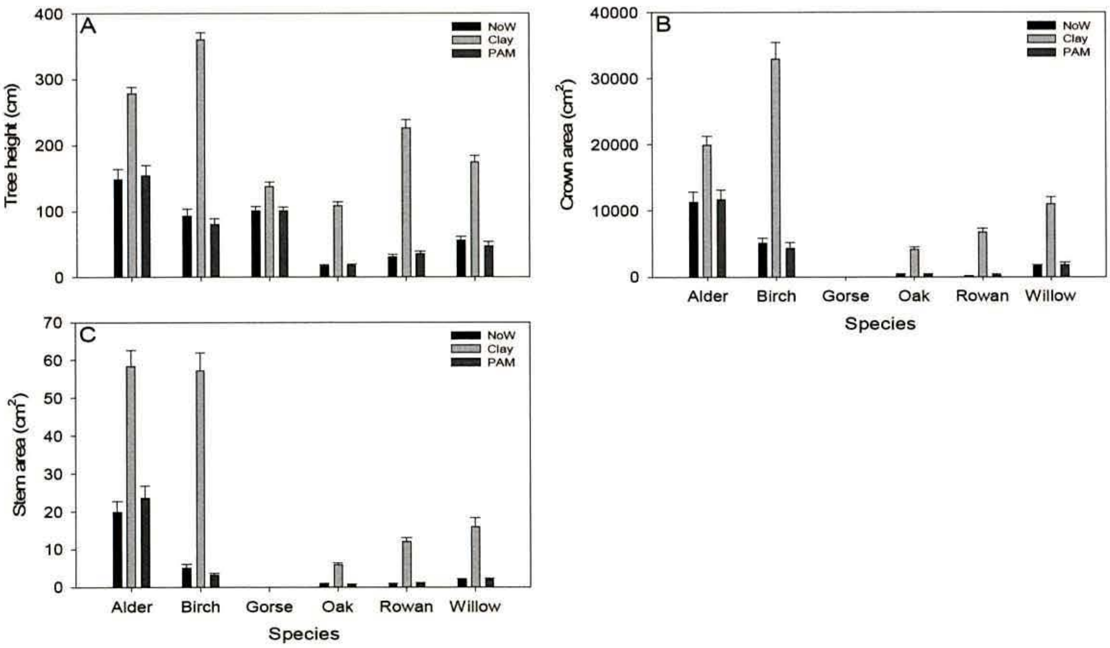


Figure 3.4 Effect of water amendment treatments on the mean tree height (A), crown area (B) and stem basal area (C) of the six tree and shrub species (bars = standard error of the mean). N.B. gorse was not measured for either stem or crown diameter. Abbreviations as in figure 3.1.

Figures 3.4 and 3.5 show the effects of water-holding (figure 3.4) and nutritional (figure 3.5) treatments on each of the six species. The interaction between species and water-holding amendment treatment was significant for all growth measurements (all $p \leq 0.001$). Oak, rowan and willow showed very little growth in any treatment except clay. In contrast, gorse performed quite well in the NoW and PAM treatments, where it was taller than birch, and closely rivalled oak in the clay treatment.

The interaction between species and nutritional amendment treatment was significant for all growth measurements (tree height $p \leq 0.001$; crown area $p \leq 0.01$; and stem basal area $p \leq 0.001$). In all species except rowan, amendment with sewage produced greater tree height than both no amendment (NoF) and NPK amendment. For crown area, ranking of nutritional treatments varied among species. The sewage treatment resulted in the greatest stem basal area in three species, but both birch and rowan had the greatest stem basal area in the NPK treatment.

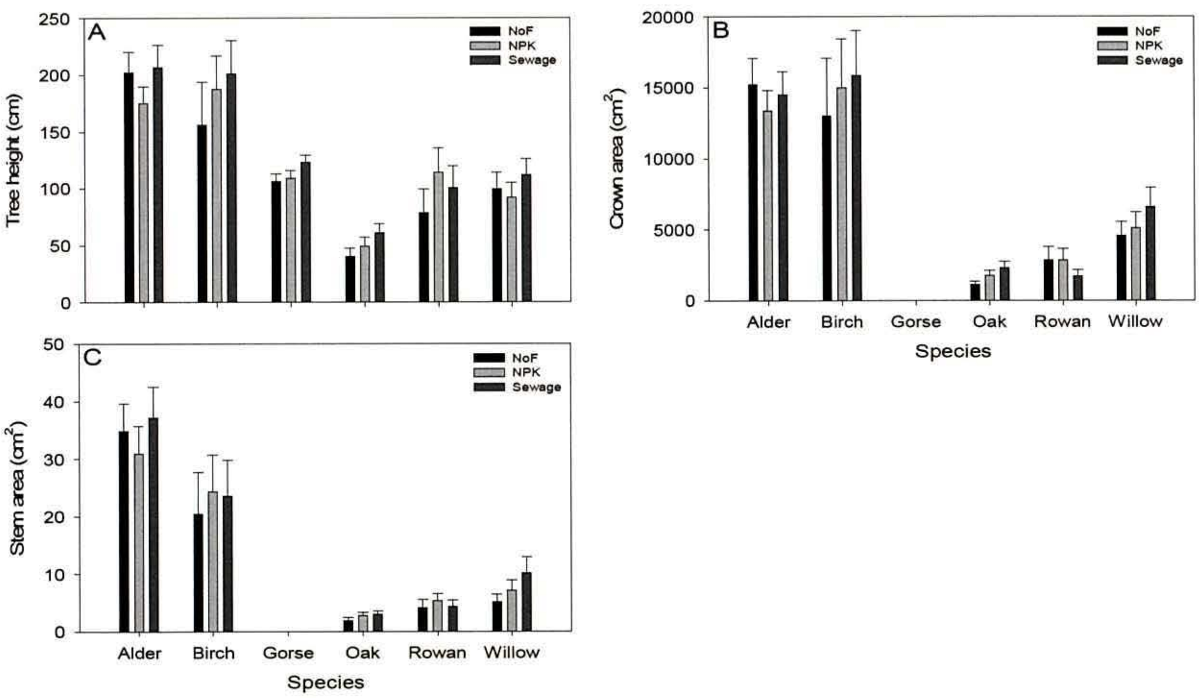


Figure 3.5 Effect of nutrition amendment treatments on the mean tree height (A), crown area (B) and stem basal area (C) of six tree and shrub species (bars = standard error of the mean). N.B. gorse was not measured for either stem or crown diameter. Abbreviations as in figure 3.2.

Percentage changes in tree height over the five-year period 2002 – 2007 are shown in figures 3.6 and 3.7. The greatest overall change was shown in the clay water-holding amendment (175.74 %); significant differences among water-holding treatments were observed ($p \leq 0.001$). Figure 3.6 (B) shows that height changes in the NPK and sewage treatments were similar, but overall differences among nutritional treatments were also significant ($p \leq 0.001$). Figure 3.6 C shows that alder, birch and, to a lesser extent, gorse generally grew better than other species. Significant among-species differences were found ($p = 0.000$).

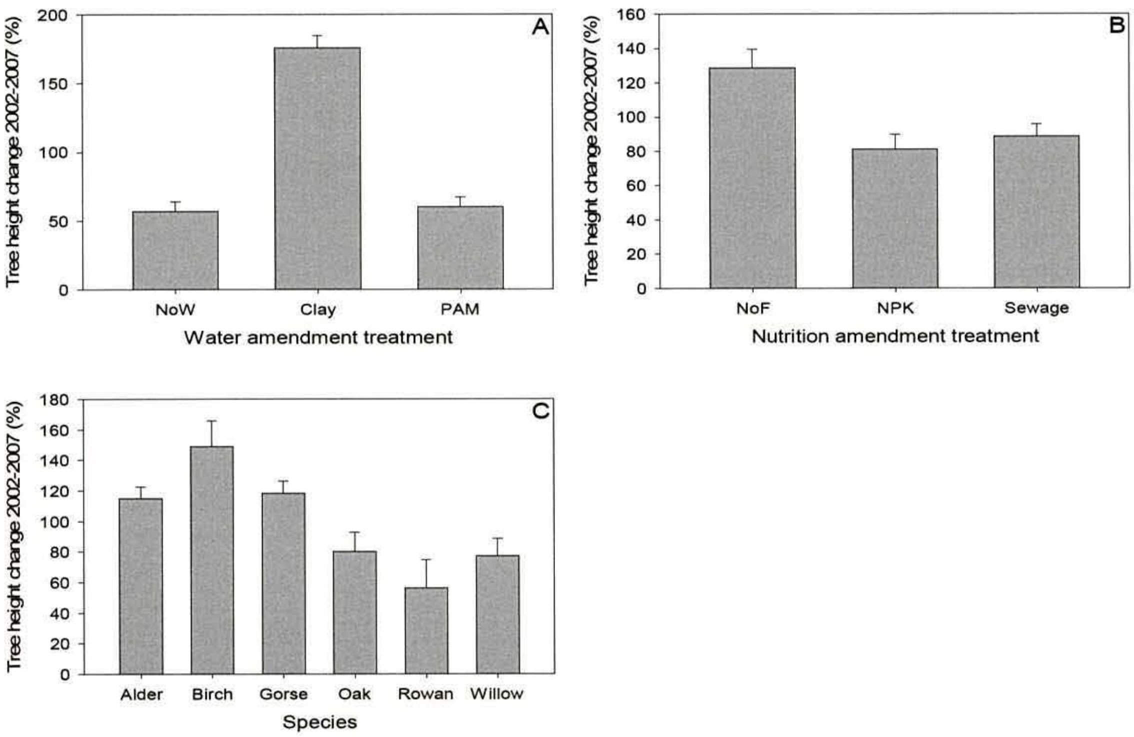
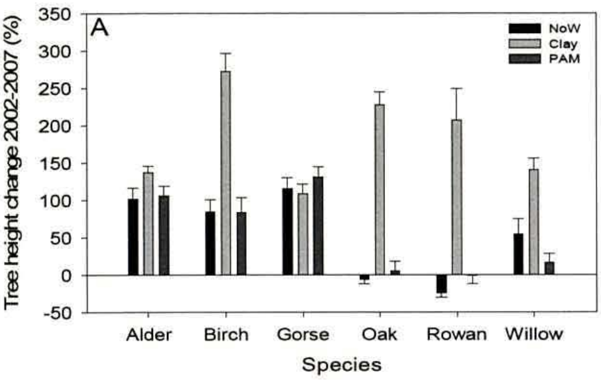


Figure 3.6 Mean percentage change in tree height measurements from 2002 – 2007 in slate waste amended with water-holding (A) and nutritional (B) treatments, and by species (C) (bars = standard error of the mean). Abbreviations as in figures 3.1 and 3.2.

Figure 3.7 shows individual species responses to water-holding and nutritional amendment treatments. With the exception of alder and gorse, all species exhibited significant differences (all $p \leq 0.001$) in tree height among the three water-holding amendment treatments for the period 2002 –2007. Significant differences in percentage tree height change for the same period among nutritional amendment treatments were found in alder (p = 0.001), birch (p = 0.025) and gorse (p = 0.018) (figure 3.7 B).



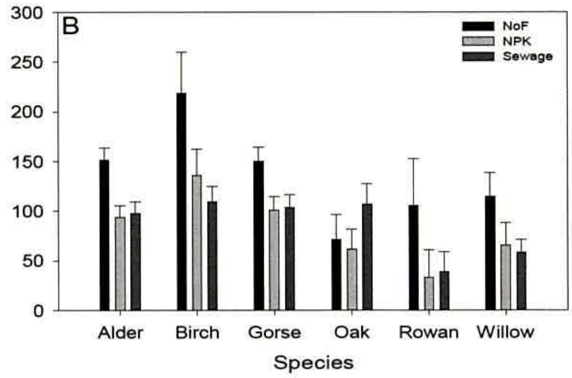
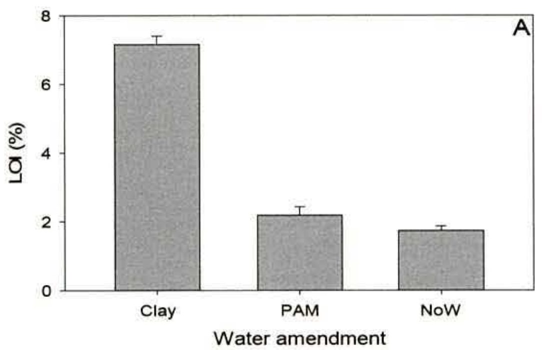


Figure 3.7 Mean percentage change in tree height from 2002 – 2007 of six species amended with water-holding (A) and nutritional (B) treatments (bars = standard error of the mean). Abbreviations as in figures 3.1 and 3.2.

Loss on ignition (LOI) values for both water-holding (figure 3.8 A) and nutritional (figure 3.8 B) amendment treatments of slate waste were generally low, clay (7.16 %) having the greatest value. There were significant differences in LOI among water amendment treatments (p = 0.000). Differences in LOI among nutritional amendment treatments were not significant.



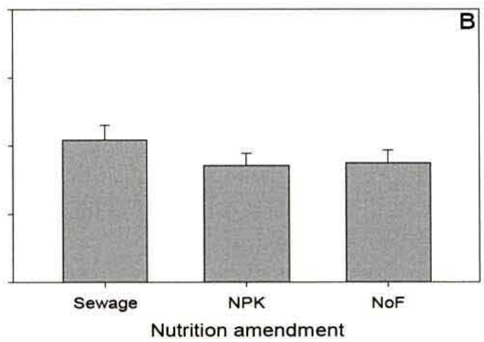


Figure 3.8 Mean loss on ignition (LOI %) of slate waste amended with water-holding (A) and nutritional (B) treatments (bars = standard error of the mean). Abbreviations as in figures 3.1 and 3.2.

3.3.3 LIFE experiment B- tree survival, vegetation establishment and LOI

None of the trees planted as part of work carried out by Rowe *et al.* (2005) were alive in 2007 (i.e. survival was 0 %).

Very few soil samples were collected from plots mulched with PAM. The small number of samples that were collected had very low LOI values (observed range = 0.68 % - 2.91 %).

Table 3.10 shows the number of species and total vegetation cover per plot. A maximum of nine species (see table 3.11 for species list) and 5 % total cover were found in any one 20 m^2 plot (plot 85).

Table 3.10 Species richness and vegetation cover (%) on plots treated with polyacrylamide (PAM) mulching

Block	Plot	No. of spp.	% total veg. cover
1	13	7	1
2	21	7	5
3	36	7	1
4	46	7	1
5	54	5	1
6	61	6	1
7	76	3	1
8	85	9	5
9	94	3	3

Table 3.11 Species lists for plots treated with PAM mulching

Plot	Botanical name	Common name
13	*Amphidium mougeotii*	Meugeot's yoke-moss
	Cardamine hirsuta	Hairy bittercress
	Cladonia coccifera	Cup lichen
	Huperzia selago	Fir clubmoss
	Orthotrichum lyellii	Lyell's bristle-moss
	Polytrichum commune	Clubmoss
	Sedum anglicum	English stonecrop
21	*Agrostis capillaris*	Common bent
	Cladonia coccifera	Cup lichen
	Epilobium augustifolium	Rosebay willowherb
	Huperzia selago	Fir clubmoss
	Leucobryum glaucum	White cushion-moss
	Sedum acre	Biting stonecrop
	Sedum anglicum	English stonecrop
36	*Amphidium mougeotii*	Meugeot's yoke-moss
	Andreaea rothii	Dusky rock-moss
	Cardamine hirsuta	Hairy bittercress
	Cladonia coccifera	Cup lichen
	Huperzia selago	Fir clubmoss
	Leucobryum glaucum	White cushion-moss
	Sedum anglicum	English stonecrop
46	*Bellis perennis*	Daisy
	Campylopus introflexus	Campylopus moss
	Cardamine hirsuta	Hairy bittercress
	Cladonia coccifera	Cup lichen
	Leucobryum glaucum	White cushion-moss
	Sedum acre	Biting stonecrop
	Sedum anglicum	English stonecrop
54	*Agrostis capillaris*	Common bent
	Parmelia saxatilis	Lichen (Crottle)
	Pleurozium schreberi	Schreber's feather-moss
	Rhytidiadelphus triquetrus	Rough gooseneck-moss
	Sedum anglicum	English stonecrop
61	*Amphidium mougeotii*	Meugeot's yoke-moss
	Cardamine hirsuta	Hairy bittercress
	Cladonia coccifera	Cup lichen
	Festuca rubra	Creeping red fescue
	Huperzia selago	Fir clubmoss
	Sedum anglicum	English stonecrop
76	*Agrostis capillaris*	Common bent
	Polytrichum commune	Clubmoss
	Sedum anglicum	English stonecrop
85	*Andreaea rothii*	Dusky rock-moss
	Cirsium arvense	Thistle
	Deschampsia flexuosa	Wavy-hair grass
	Epilobium penduncularе	New Zealand willowherb
	Festuca ovina	Sheeps fescue
	Huperzia selago	Fir clubmoss
	Sedum anglicum	English stonecrop
	Senecio jacobaea	Common ragwort
	Teucrium scorodonia	Woodsage
94	*Deschampsia flexuosa*	Wavy-hair grass
	Polytrichum commune	Clubmoss
	Sedum anglicum	English stonecrop

3.3.4 Fertilised and un-fertilised tree pairs; growth responses

Figures 3.9 - 3.11 show the mean changes in stem basal area, tree height and crown area of birch and willow for the periods May 2000 – January 2007 (pre-application – 80 months post-application), January 2001 – January 2007 (8 months post-application – 80 months post-application) and October 2001 – January 2007 (17 months post-application – 80 months post-application).

For the periods January 2001 - January 2007 and October 2001 - January 2007 there were no significant differences in growth of either birch or willow between fertilised and non-fertilised trees. For the whole period of observations from pre-fertilisation in May 2000 to the latest measurement in January 2007 the only significant difference was in tree height (p = 0.0202) and crown area (p = 0.0290) of birch trees (see table 3.12).

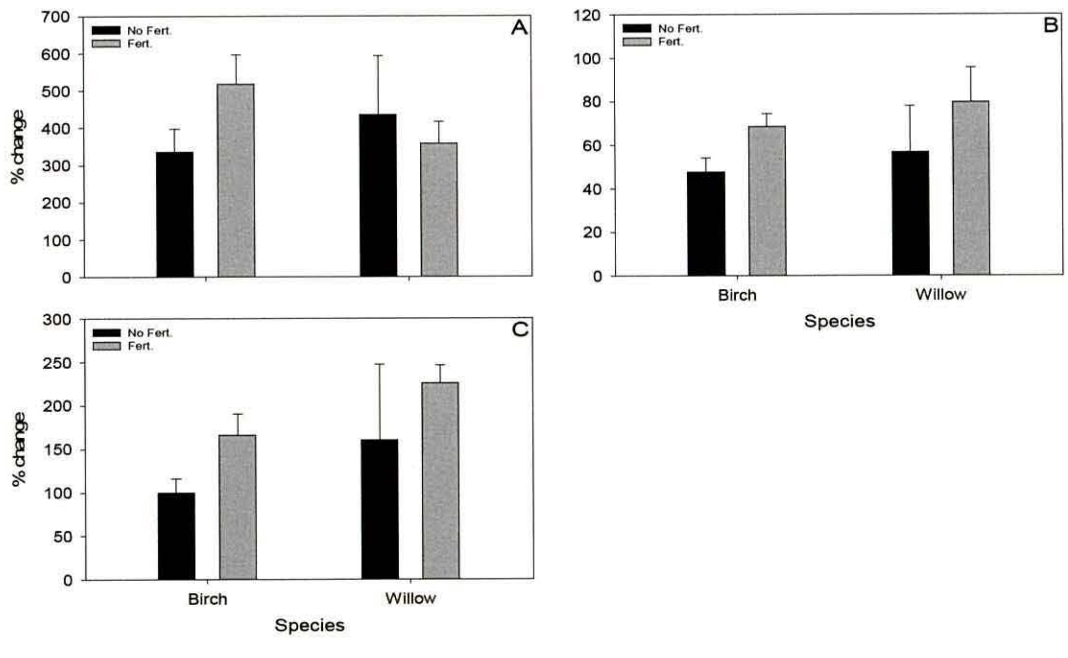


Figure 3.9 Percentage changes in stem basal area (A), tree height (B) and crown area (C) of birch and willow trees between May 2000 and January 2007 (pre-application – 80 months post-application of fertiliser) (bars = standard error of the mean). Fert. - fertilser; No Fert. - no fertiliser.

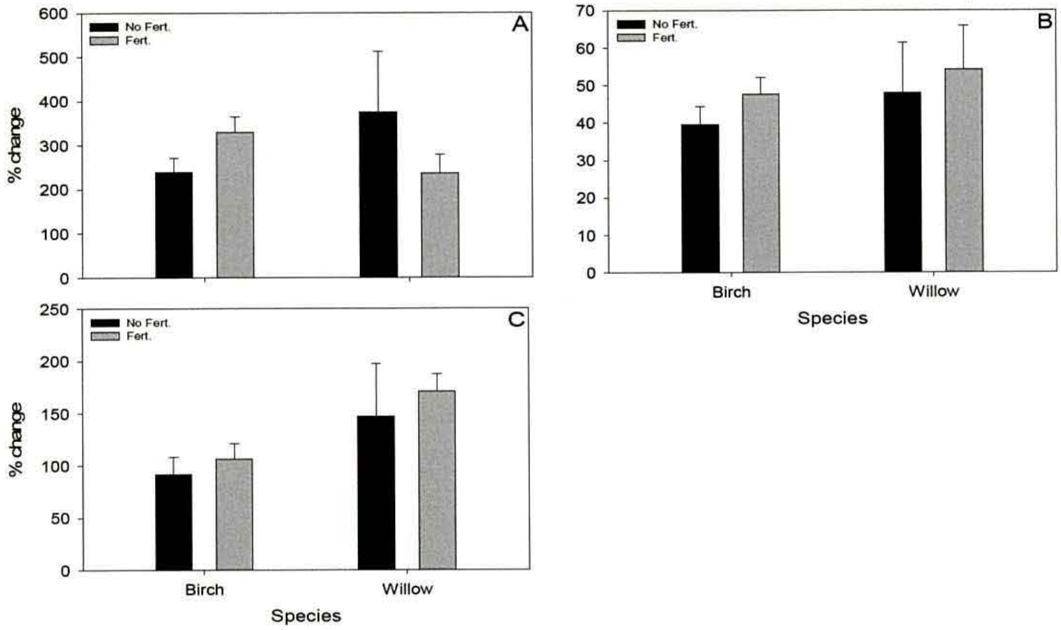


Figure 3.10 Percentage changes in stem basal area (A), tree height (B) and crown area (C) of birch and willow trees between January 2001 and January 2007 (8 months post-application – 80 months post-application of fertiliser) (bars = standard error of the mean). Fert. - fertilser; No Fert. - no fertiliser.

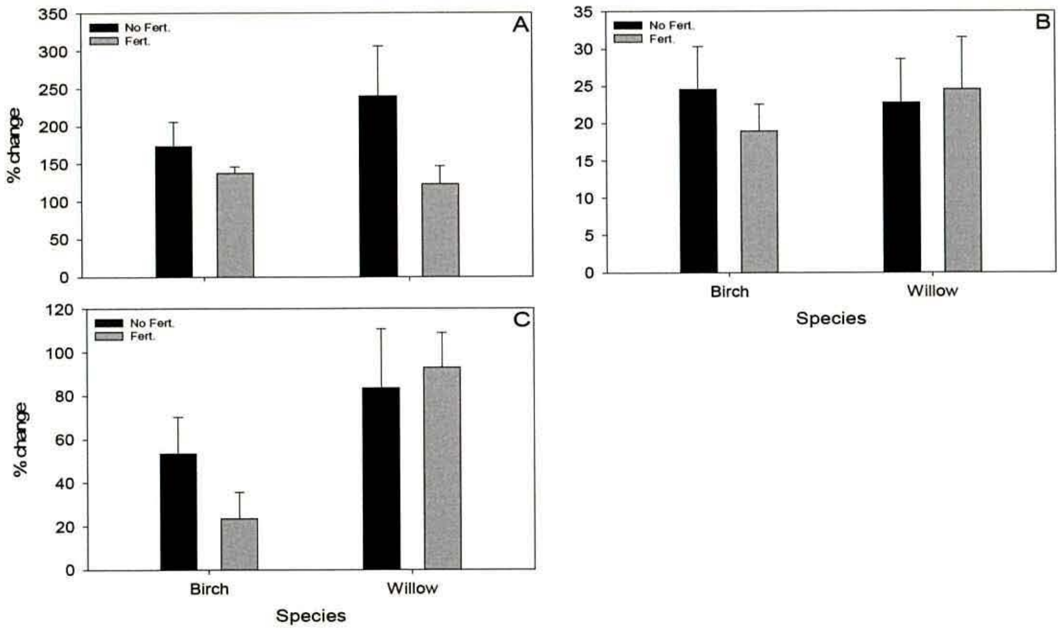


Figure 3.11 Percentage changes in stem basal area (A), tree height (B) and crown area (C) of birch and willow trees between October 2001 and January 2007 (17 months post-application – 80 months post-application of fertiliser) (bars = standard error of the mean). Fert. - fertilser; No Fert. - no fertiliser.

Table 3.12 Significant differences (p values) between fertilised and non-fertilised birch and willow growth responses (figures in bold represent significantly different growth responses)

Period	Measurement	Birch	Willow
May 2000 - October 2001	Stem basal area % inc.	**0.0392**	**0.0014**
	Tree height % inc.	**0.0121**	0.1234
	Crown area % inc.	**0.0034**	**0.0215**
May 2000 - January 2007	Stem basal area % inc.	0.0772	0.6531
	Tree height % inc.	**0.0202**	0.4016
	Crown area % inc.	**0.0290**	0.4749
January 2001 - January 2007	Stem basal area % inc.	0.0660	0.3530
	Tree height % inc.	0.2340	0.7330
	Crown area % inc.	0.5210	0.6570
October 2001 - January 2007	Stem basal area % inc.	0.3034	0.1200
	Tree height % inc.	0.4177	0.8469
	Crown area % inc.	0.1706	0.7712

3.4 Discussion

Of the trees planted by Sheldon and Bradshaw in 1973/74, 22 of 23 trees measured in 2002 by Dr. E. Rowe remained in 2007, and all of them had shown good levels of growth over this period. It appears, as suggested by Sheldon and Bradshaw (1975), that the main hindrance to re-vegetation of slate waste tips occurs at the establishment phase; if successful establishment is facilitated high survival rates and continued growth follow.

Hydraulic distribution of a peat-based slurry as carried out by Sheldon and Bradshaw has produced clear evidence of its efficacy for aiding vegetation establishment and development on slate waste. The technique was reported (Sheldon and Bradshaw, 1975) to be an effective way of rapidly distributing the growth substrate whilst causing low levels of disturbance, and the observations recorded in 2007 demonstrate that the substrate used is capable of facilitating long-term plant growth. This is probably due in part to the increased connectivity of surface and inner portions of the slate waste tips resulting from pumping the peat-based slurry onto and into the tip matrix. Good levels of species recruitment and development have occurred in the area amended with pumped slurry, with woody shrub species such as *Erica cinerea*, *Calluna vulgaris* and *Cotoneaster microphyllus* growing in abundance.

The hypothesis put forward by Dr. E. Rowe that perhaps nitrogen is not the nutrient limiting plant growth in slate waste, following observations carried out in 2002, noting that alder did not appear to demonstrate any dominance over birch in terms of tree height, is supported by the data shown in table 3.4. However, in the context of stem and crown area observations, this statement does not hold true; there are stark differences evident between alder and birch in stem basal area and crown area, alder being larger on both accounts. This hypothesis can therefore be rejected upon consideration of the additional information gathered. The substantially greater stem and crown development in alder plants suggest that non-nitrogen fixing plants suffer as a result of the inherently low concentrations of plant available nitrogen present in slate waste.

Percentage growth increases calculated for the period 2002 to 2007 (table 3.6) demonstrate that alder and birch performed similarly in terms of stem basal area and tree height growth, but that birch has developed at a far greater rate than alder in this period with respect to crown area. The figures indicate approximately 30 % increases in stem basal area, 14-17 % increases in tree height, and crown area increases of ~48 % (alder) and ~76 % (birch). These figures suggest that there is little between-species difference in growth rates, with birch in fact performing slightly better than alder in the case of crown development. This would seem to suggest that there is no benefit to be gained from nitrogen fixing, and that plant-available nitrogen is present in non-inhibiting concentrations; if any plant nutrient is limiting, it might be phosphorus as suggested by Dr. E. Rowe.

Unlike any other study carried out at Penrhyn Quarry, data collected from the trees planted by Sheldon and Bradshaw truly represent long-term results. The continued survival and growth of this small stand of trees demonstrates that the methods developed by the authors have great potential for promoting vegetation establishment, habitat development and long-term ecological restoration of slate waste resulting from quarrying activities. Restoration work carried out more recently at Penrhyn Quarry, for example the LIFE A and B experiments, has been unable to demonstrate other methods showing similar levels of potential for restoration on slate waste tips.

Two general trends emerge from the re-assessment of LIFE experiment A: water-holding amendment with clay and nutritional amendment with sewage offer the most benefit in terms of promoting woody plant growth on slate waste.

The single most effective treatment was amendment with clay. However, this treatment cannot be considered comparable to the others utilised in this experiment. The clay amendment consisted of laying a complete continuous

75 cm thick slab of boulder clay on top of the slate waste, and creating planting pockets in the clay; other amendments consisted of small additions of materials into planting pockets dug into the slate waste. If similar quantities of sewage, for example, were applied in the same manner, equivalent levels of success might be expected.

Additionally, the use of clay in this fashion cannot be considered a realistic or viable option for promoting vegetative establishment, growth or ecological restoration of disturbed sites. The clay used for this experiment was present on site at Penrhyn Quarry as overburden laying on top of slate deposits, and the creation of an area approximately 300 m^2 and 75 cm deep (equivalent to 225 m^3) was therefore unproblematic. However, for use on a wider scale (e.g. the derelict area of Penrhyn Quarry which covers approximately 100 hectares), there are not sufficient reserves of clay. Furthermore, the bulky and dense nature of boulder clay means its transportation costs are high, limiting its potential uptake.

Amendment with inorganic materials (e.g. PAM and NPK) did not differ substantially from the "no amendment" treatments (NoW and NoF), therefore offering very little benefit to woody plant establishment and growth, and these treatments are probably not worth further consideration.

However, excluding the clay treatment reveals that plants in the nutritional amendment treatments in general perform slightly better than those in water-holding amendment treatments, suggesting that it is the availability of plant nutrients that is most limiting when establishing plants in slate waste.

The experimental area for LIFE experiment A was heavily machined prior to planting, creating a level area free from large fraction slate waste and higher than average (for typical conditions found at Penrhyn Quarry) quantities of fines material; these conditions are far more amenable to plant growth than that found in the vast majority of locations within Penrhyn Quarry. Far higher levels of success might have been expected from this project, and it could be considered that the techniques utilised are far from adequate.

In contrast, for LIFE experiment B (as reported by Rowe *et al.*, 2005) there was no site preparation prior to the experimental onset; consequently tree planting failed completely. Mulching with PAM also failed to promote any significant vegetation establishment, with none of the nine plots treated in this manner exceeding 5 % total vegetation cover seven years after the initial restoration works. At the point of re-assessment in 2007 there was virtually no soil present at this experimental site, providing a clear explanation for the observed failure of tree establishment and vegetation development. Mulching with 100 g m^{-2} dry (anhydrous) PAM gel cannot therefore be considered a suitable method for promoting ecological restoration of slate waste tips of this nature. The use of a bulkier material would have greater potential for restoration success, such that voids and gaps within the surface layers of the slate waste tips would effectively be plugged, creating pockets into which root growth could be extended. Small quantities of PAM applied to waste tip surfaces only provided a thin coating susceptible to weathering, drying, UV breakdown and flushing into the deep inner tip, offering little potential for windblown seeds, for example, to colonise.

In contrast to the above restoration methods, whereby tree planting and additions of materials were made to the surface of the slate waste tip or to the tree planting pockets, Rowe *et al.* (2006) tested the potential of promoting the growth of naturally-colonised trees as a restoration technique.

Re-assessment of the fertilised and un-fertilised tree pairs established by Rowe *et al.* (2006) showed that fertilised trees out-grew non-fertilised trees in all but one variable (willow stem basal area) during the whole experimental period (May 2000 – January 2007). However, the observed between-treatments difference was only significant for two variables (birch tree height and crown area).

In the period October 2001 to January 2007, only two variables (willow tree height and crown area) indicated greater growth in fertilised trees than in their non-fertilised counterparts. No significant differences between fertiliser treatments were found for this six-year period. It appears, therefore, that the long-term response of birch and willow to fertiliser application has been negligible.

The overall findings from this set of studies is that despite initial observations of growth acceleration resulting from the fertilisation of naturally-colonised trees growing at Penrhyn Quarry, the growth advantage ceases within seven years of initial fertilisation. The potential of this technique as an effective restoration tool is therefore not clear, and a cautious approach to its uptake is recommended.

It should be noted that there are some miscalculations in the paper by Rowe *et al.* (2006). For example, it is stated that "fertiliser was evenly distributed over the whole 2.25 m^2 plot area", when it had previously been stated that the experimental plots measured 1.5 m radius centred around the tree, giving a plot area of 7.07 m^2. It is possible that fertiliser was applied at a far lower rate than intended.

Additionally, it is claimed that stem and crown areas were calculated assuming elliptical shapes, though it is apparent that both areas were calculated using the equation $\pi d^2/4$ (the calculation for a circle). Calculations based on measurements made in 2007 assumed circular stems and crowns, therefore allowing direct comparison with previous results.

3.5 Conclusions

Re-investigation of four restoration projects carried out at Penrhyn Quarry, dating back to 1973, has provided useful information on the long-term effectiveness of the techniques and methods employed to promote vegetation establishment and ecological restoration of slate waste tips. Some of these techniques have resulted in partial successes and have shown early results indicating their potential for use in restoration projects, whereas other techniques failed completely.

All of the restoration techniques (except for work carried out by Rowe *et al.*, 2006) tested at Penrhyn Quarry have involved the provision of materials to increase either water or nutrient availability for plants. Despite the varying levels of success achieved this effectively highlights the two most important factors to consider when planning restoration processes on slate waste tips. However, both moisture and nutrients need to be provided in such a way that they will be effective in the challenging conditions presented by slate waste, whilst also being sustainable, economically viable and effective in the long-term.

The main emphasis of three of the re-assessed restoration techniques was on tree planting into slate waste tip surfaces with accompanying amendments. The only successes of this approach were planting into large quantities of hydraulically applied peat-slurry and planting into a substantial depth of boulder clay. Numerous other restoration techniques involving tree planting, also tested at Penrhyn Quarry, effectively failed; these include PAM, NPK and slate sand additions to planting pockets. It appears that nutritional amendment has a slightly greater impact on planted tree growth than water-holding amendments when used in similar quantities (as demonstrated by growth of sewage amended plants compared to those treated with PAM in LIFE experiment A), and that organic sources of nutrition are more beneficial (as demonstrated by growth of sewage amended plants compared to those treated with NPK in LIFE experiment A) than mineral inorganic forms. Additionally, surface mulching with PAM failed to promote planted tree survival, tree growth or any significant colonisation. It is also apparent that inorganic fertilisation of naturally colonised trees on slate waste tips has little or no potential as an effective restoration method.

The lack of widespread establishment of tree seedlings when not accompanied by substantial amendment suggests the possibility that other factors influence success rates in restoration projects. It is possible that the initial levels of intervention in restoration projects could be reduced by using planting stock of ecotypes adapted to conditions on quarry waste. Although both LIFE experiments used local provenance tree seedlings, only willow cuttings were sourced from quarry trees; all other species used were of unspecified local provenance. By incorporating exclusively quarry-sourced planting stock into restoration projects it is possible that establishment rates could be increased.

Nason *et al.* (2007) and Williamson *et al.* (2009) suggest that although the use of boulder clay had a markedly greater impact on plant growth than any other individual treatment amongst those used in the LIFE experiment A trial, its wider uptake is severely limited by its localised availability and its high bulk density. There is, therefore, scope for testing a wider variety of materials for their potential inclusion in future methods for the restoration of slate waste. Necessary qualities for potential restoration materials are their sustainability, availability, organic (rather than inorganic) nature and ability to provide both water-holding capacity and nutritional improvement to slate waste. Recent moves towards the widespread production of high quality green waste compost in large quantities offers a potentially invaluable resource capable of fulfilling this function. Williamson *et al.* (2009) state that "neither nutrient supply nor water-holding capacity, alone, was of paramount importance" for promoting vegetation establishment on slate waste tips, alluding to the fact that only materials capable of providing both attributes will have a useful effect on plant establishment.

Sheldon and Bradshaw (1975) highlighted the need for an effective method of applying restoration materials to slate waste tips surfaces. They demonstrated that hydraulic distribution was an efficient method of distributing substantial quantities of peat-based slurry whilst causing little associated disturbance upon the integrity and stability of the slate waste tips surfaces. Rowe *et al.* (2005) also recognised this as being one of major hurdles to be overcome if restoration projects are to be successful, and accordingly developed the concept, employing techniques using easily transportable materials such as PAM gel and hydroseeding equipment to apply organic matter and seeds to slate waste tip surfaces in remote locations within Penrhyn Quarry. However, no useful restoration was achieved with either PAM gel or hydroseeding equipment; there remains, therefore, the need to develop an effective method for applying water-holding and nutritional materials to slate waste tip surfaces to promote vegetation establishment and habitat restoration.

Bradshaw (1997) states that mined lands often exhibit extreme soil conditions inhibitory to plant growth; to overcome these problems is to initiate the whole restoration process. To achieve this Bradshaw (1997) proposes implementing carefully-considered interventions to improve soil quality, such as those outlined here, so that self-sustaining, long-term restoration of severely disturbed sites can be accomplished.

3.6 Further work and recommendations

Studies carried out by Williamson *et al.* (2009) and Rowe *et al.* (2006) relied quite heavily upon analysis of soil and plant samples. Time constraints did not allow for such analysis to be done in this study. If more time and resources had been available, soil and leaf analysis, as done by Rowe *et al.* (2006), would have added substantially to the results.

With regard to the work of Sheldon and Bradshaw (1975) and Rowe (unpublished), some information was incomplete or unavailable. As a result some direct comparisons and cross references to initial findings and observations were not possible. For instance, no records of species richness or vegetation cover of plots treated with PAM mulching by Rowe *et al.* (unpublished) are available for the period from the experimental onset in April 2000 to the end of monitoring in July 2002.

The initial intention of this part of the study was to carry out a fairly intensive evaluation of a small number of restoration methods used on slate quarries in the Gwynedd area of north Wales. This would have identified the most successful methods of restoration and helped to provide a guide for future restoration projects. A small number of local quarries that had received some form of restoration work was identified (see table 3.13). However, it became apparent that although it was known whether or not restoration work had been carried out at a site, the details of specific restoration activities and project objectives were not always available. It was, therefore, decided to focus on the projects carried out at Penrhyn Quarry for which information has been published and made publicly available.

Table 3.13 Proposed study sites for investigation of past restoration techniques

Site/Quarry name	UK NGR
Abercwmeiddaw	SH 746, 093
Bowydd	SH 708, 464
Braichgoch	SH 748, 079
Bus stop quarry/Blue Peris/Dinorwic	SH 588, 613
Cwm Penmachno	SH 750, 471
Dorothea	SH 494, 532
Fotty	SH 706, 465
Gelli Iago	SH 632, 484
Glan y don	SH 697, 467
Padarn Country Park	SH 583, 607
Talysarn	SH 488, 528
Tanyrallt	SH 492, 524
Y Fron	SH 515, 547

4 The importance of planting material source/genotype in post-industrial ecological site restoration.

Abstract

The potential for increasing restoration success on slate waste tips was investigated by comparing the survival and development of very locally-sourced plants (i.e. quarry plants) versus locally-sourced plants (i.e. non-quarry plants).

Shoot cuttings of willow (*Salix caprea* × *Salix cinerea* = *Salix reichardtii*) were taken from quarry and non-quarry trees. Cuttings were inserted into artificial slate waste tips as pegs 250 mm in length. Thirteen monthly observations were made of shoot initiation and development. Cuttings from non-quarry trees produced shoots more rapidly, in greater quantities (e.g. mean shoot count one-month after insertion: NQ = 5.69, Q = 4.26) and of larger overall size (e.g. mean total shoot diameter five months after insertion: NQ = 12.37 mm, Q = 10.05 mm) than those produced by quarry trees.

Three watering regimes (representing ambient, ambient −20 %, and ambient +20 % rainfall levels) were applied to the rooted quarry and non-quarry willow cuttings. It was found that quarry clones exhibited a significantly (p = 0.000) superior ability to cope with varied water availability. This was demonstrated by the change in total shoot diameter recorded after six months (December 2007), whereby quarry clones increased by 11.16 % and non-quarry clones decreased by 2.64 %.

Seed from seven species (*Acer pseudoplatanus, Betula pendula, Cytisus scoparius, Fagus sylvatica, Sorbus aucuparia, Quercus petraea* and *Ulex europaeus*) growing on slate waste tips at Penrhyn Quarry was collected from both quarry and non-quarry provenances. Observations of germination, development and survival were recorded over a 12-month period. Provenance

effects varied across the species, with neither quarry nor non-quarry sourced plants exhibiting consistently better performance. Non-quarry sycamore performed significantly better than quarry sycamore in germination ($p \leq 0.0209$), shoot growth ($p < 0.001$) and most biomass variables ($p < 0.001$). The reverse was found with broom, in which quarry plants consistently performed significantly better than non-quarry plants; for instance, all biomass variables were greater ($p \leq 0.0025$), as were the majority of shoot length measurements ($p \leq 0.0328$) and germination records ($p \leq 0.0421$).

Although there were no definite signs of adaptation of plants sourced from quarry populations to quarry conditions, it remains unclear if, over long timeframes, very local provenances might offer the most effective regenerative material for the restoration of slate waste tips. Further research is recommended, using a wider range of species and longer experimental periods.

4.1 Introduction

4.1.1 Source of planting material and restoration ecology

Planting often plays a major part in restoration activities, and the main focus may simply be achieving a particular species composition and planting density. With careful consideration of the desired project outcomes and intelligent exploitation of plant properties and species composition to suit site-specific conditions, planting can be carried out in such a way to maximise success. For example, Good *et al.* (1985) recommend the use of natural pioneer species and the sourcing of intra-specific variants of species to maximise rates of survival, establishment and development.

Use of specified planting stock for site restoration activities is becoming more prevalent. Harris *et al.* (2006), for example, state that restoration practitioners, where possible, nowadays tend towards the utilisation of locally-derived plant material. Other researchers, for example Stevensen *et al.* (1997), Jim (2001), Wilkinson (2001), Krauss and Koch (2004), McKay *et al.* (2005), Bischoff *et al.* (2006a) and O'Brien *et al.* (2007) state that it is best practice to use locally-sourced planting material, rather than introduce exotic or non-local seed for habitat re-creation, restoration and rehabilitation projects, particularly when the desired outcome is conservation value rather than amenity usage. This practice is often prescribed because it is considered that locally-sourced planting material will be adapted to conditions at the site of restoration.

4.1.2 Definitions

Turesson (1922) introduced the term "ecotype" to refer to "groupings of populations (ecological races or sub-species) in relation to a type of habitat or climate", to be used in reference to a species' response to specific habitat types (Quinn, 1978). Gregor *et al.* (1936) (cited in Quinn, 1978) subsequently developed this definition, including specific reference to the morphological and physiological characteristics that distinguish populations from one another, stating also that these populations will remain inter-fertile with one another

(other ecotypes of the "eco-species"), yet the free exchange of genes will be prevented by ecological barriers.

Hufford and Mazer (2003) provide additional and complementary definitions for use in this subject; they state that "local" refers to a previously-existing genotype at a site (also referred to as existing, native, or indigenous); "local adaptation" refers to the process by which populations genetically change as a result of natural selection pressures within their habitat; and "ecotype" refers to distinct genotypes or populations within one species, formed as a result of adaptation to environmental conditions within the local habitat, whilst maintaining the ability to interbreed with other ecotypes of the same species. Smith *et al.* (2007) define "local ecotype" as plant material that is known to have been sourced from an ecologically defined region, for example a region of specific climatic conditions or perhaps very distinct soil chemistry. Macel *et al.* (2007) concur with the previous definitions, and state that "local adaptation" is a pattern resulting from divergent natural selection on populations that evolve in response to specific local environmental and ecological conditions.

It is important to understand that when the term "local" is used, this can refer to plant material sourced from a region that is either geographically close or ecologically similar to the site where the material is to be planted.

N.B. The definition of provenance, as set out by Hubert and Cundall (2006) is "the geographic locality of a stand of trees from where the seed was collected"; this can be used also to refer to any plant type collected from a defined geographic locality.

4.1.3 Theory of genetic adaptation and ecotype formation

The theory behind the formation of distinct ecotypes is important in the understanding of this subject matter. Several definitions worth highlighting to aid the understanding of ecotype formation are that of genetic drift (the change in allele frequency from one generation to the next, due to offspring inheritance of a random sample of the parents alleles) (Lande, 1989), divergent natural selection (selection acting in different directions on different populations, so that population means are pulled towards different adaptive peaks, each advantageous in one environment but not in others) (Anon, 2001; Schluter, 2009; Schluter, 2000, cited in Takahashi *et al.* 2009), and gene flow/migration (movement of genes from one population to another) (Jones, 2005).

Schluter (2009) states that, "the agents of divergent (natural) selection are extrinsic and can include abiotic and biotic factors such as food resources, climate, habitat, and interspecies interactions such as disease, competition, and behavioural interference."

Divergent natural selection, when gene flow potential is limited and where there are strong selective forces at work, is the main causative agent of local adaptation and the formation of multiple, genetically-distinct ecotypes located at different sites (Hufford and Mazer, 2003; Bischoff *et al.*, 2006a; Macel *et al.*, 2007). Other evolutionary processes, for instance mutation and random genetic drift, are equally as capable of causing divergence between populations (Steinger *et al.*, 2002), but in these cases population differences are likely to be adaptive.

Despite the fact that plant reproductive material can potentially be dispersed over huge distances, for example wind-borne pollen and water-borne seeds, it has been found that reproductive isolation between populations can occur over very small distances. For example, Aston and Bradshaw (1966), Antonovics and Bradshaw (1969) and Antonovics (1971) have demonstrated that where there are extreme environmental, geographical or habitat gradients present (e.g. salt tolerance in costal communities, and heavy metal tolerance in mine-site communities), very localised adaptation between populations and even speciation can occur over distances as small as 50 metres (Quinn, 1978).

Reproductive isolation (defined by Anon (2001) as the absence or severe restriction of gene flow between populations) between populations begins, according to Schluter (2001), under allopatric conditions (whereby geographic separation prevents two or more populations from encountering each other (Anon, 2001)), where populations "accumulate adaptations" to characteristics of their habitats. Such habitat characteristics will be contrasting for the different populations and might include resource type or availability, structural features or other biotic or abiotic features (Schluter, 2001). Rundle *et al.* (2000) concur with this, adding that reproductive isolation is more closely correlated with environmental gradients than with geographic or genetic distances. This local adaptation can be identified over distances as small as 50 metres.

This process of adaptation results in populations that are better adapted to their surrounding ecosystem conditions than are other populations. This can be described as home-site advantage, and under the home-site advantage theory it can be stated that the local population will be better suited to home-site conditions than non-local populations (Sackville Hamilton, 2001; Hufford and Mazer, 2003; Smith *et al.*, 2005). This process can be observed in widespread species that occupy a range of habitats. Localised adaptation and specialisation creates much within-species genetic diversity because the species encounters different environmental stimuli across their ranges (Bischoff *et al.*, 2006a).

It has also been shown that these population adaptations can occur over relatively small timescales. Wu *et al.* (1975) reported studies of *Agrostis stolonifera* populations that were adapted to atmospheric copper pollution produced by local refineries (British Insulated Callenders Cables Ltd., unspecified metal type). It was found that the evolution of heavy-metal tolerance could occur very quickly, even within a single generation, due to the strong selection conditions present.

If the incorporation of locally-sourced planting material into restoration projects is not possible due to, for example, limited seed availability, it is necessary to obtain non-local seed, often from commercial suppliers. Introduction of non-local plant material to restoration sites brings about a range of problems, not least, theoretically, their limited capacity to successfully exist away from their home-site (Keller and Kollmann, 1999; Keller *et al.*, 2000). Additional complexities arise as a result of cross breeding or hybridisation with local populations, disrupting genetic integrity (Keller *et al.*, 2000) and possibly disrupting established ecosystem processes such as herbivore predation (Keller and Kollmann, 1999). Progeny resulting from hybridisation between local and non-local populations may suffer from reduced fitness when compared to locally-adapted parent plants, due to disruption of epistatic gene complexes. Such loss of fitness can be described as out-breeding depression (Keller *et al.*, 2000), and is likely to reduce the potential for successful outcomes in restoration projects (McKay *et al.*, 2005). Out-breeding depression can cause reduced mean population fitness (Hufford and Mazer, 2003) by the expression of less-well-adapted phenotypes (McKay *et al.*, 2005) in F1 offspring or subsequent generations. A serious consequence of out-breeding depression is that it can accelerate the rate at which disturbed remnant populations at restoration sites move towards extinction (Sackville Hamilton, 2001).

Despite the potential problems of hybridisation between native and introduced populations at restoration sites, over time some natural processes occur to minimise negative effects. It is suggested that, providing there is no further disturbance or introduction of non-locally-adapted plant material, natural selection will remove less fit or poorly adapted individuals, especially if some degree of backcrossing with local plants takes place, leading to increases in locally-adapted gene complexes and mean fitness throughout the population (Keller *et al.*, 2000; Wilkinson, 2001; McKay *et al.*, 2005).

4.1.4 Utilisation of local provenances

Very limited information exists in the literature regarding the employment of planting material specifically sourced from particular provenances in the restoration of sites disturbed by quarrying and mining. Despite this, several examples, put forward by a small group of authors, do provide an insight into the problems, challenges, and considerations involved in carrying out such projects.

For instance, McKay *et al.* (2005) indicate that observing plant growth upon mine tailings has contributed to the understanding of plant adaptation to local site conditions, specifically, to very localised heavy-metal concentrations in mine spoils. It could therefore be suggested that severely-disturbed quarries and mine sites provide a model situation for testing the theory of local adaptation in plant populations, and therefore the importance of using planting material of suitable provenance in restoration activities.

Good *et al.* (1985) note the challenges, and indeed benefits, of focussing upon the provenance of planting material in mining restoration efforts. The authors suggest that tests utilising seed collected from young trees growing on old spoil heaps may provide interesting results. This is because these trees may represent the third or fourth generation subjected to challenging site conditions presented by spoil heaps. It can therefore be hypothesised that the young trees sampled may have a range of phenotypes and/or genotypes that have developed over time in response to the challenging site conditions. Natural selection may have resulted in an ecotype better adapted than non-local populations, and therefore showing more successful growth and survival under the local conditions. However, the authors also advise caution towards this style of sampling. In general it could be assumed that after several generations of natural seeding, plants should be at least partially adapted to growth in challenging circumstances. However, there is always a small chance that a number of individuals within the population might, by chance, have established and developed in favourable microclimates situated within a larger, more difficult site. The paper documents the survival of birch and willow when planted to restore opencast coal spoil throughout Britain. Seedlings were specifically selected from colliery spoil tips deficient in plant nutrients, and control plants were sourced from non-colliery provenances. Results consistently demonstrated greater survival of the colliery clones (e.g. *Betula pubescens* survival in unselected plants (non-colliery) = 39 %, and in selected (colliery) = 94 %), thus demonstrating a high degree of adaptation to the challenging conditions presented by the mining waste.

The findings of Good *et al.* (1985) are backed up by Rehounkova and Prach (2006), whose studies of spontaneous vegetation succession in gravel-sand pits in the Czech Republic demonstrated the importance of plant populations surrounding sites of restoration in providing a seed source for natural colonisation. The authors comment that findings of this nature are commonly reported in studies of restoration in gravel pits, sand pits and stone quarries. This implies that local seed distributed from nearby plant populations enables successful and important recruitment on severely disturbed land, possibly enhanced by ecotypic adaptations to local site conditions.

Several references to the importance of provenance in habitat restoration are provided from researchers working in Australia. Burgin *et al.* (2005) describe the restoration work carried out by mining organisations as "faking nature", often comprising half-hearted attempts at habitat restoration involving tree planting. The authors recommend that the mining industry should set itself restoration goals, focussing upon the species composition, functionality and ecological dynamics on a site-by-site basis. An important part of this process will be the selection and utilisation of suitable planting stock, for example locally-sourced ecotypes. O'Brien *et al.* (2007) report that bauxite mining in Western Australia is responsible for the destruction of 700 hectares of northern Jarrah forest (largely made up of *Eucalyptus marginata*) *per annum*. However, companies carrying out bauxite mining activities are actively sourcing local provenance seed (from within 20 km of restoration sites) where possible, in an effort to conserve local genotypes and maintain intra-specific genetic variation. This conscientious approach has been developed from the recognition by two mining companies (Alcoa World Alumina Australia, and Worsley Alumina Pty. Ltd.) in the early 1990s that conservation of genetic diversity in mine-site restoration schemes was important for maximising restoration potential and maintaining ecosystem functionality (Krauss and Koch, 2004). However, mining companies are likely to encounter problems in collecting sufficient quantities of seed of very local provenance to satisfy re-vegetation demands of disturbed sites. Krauss and Koch (2004) point out that mine-site restoration projects being carried out on a commercial scale and utilising local seed provide great opportunities to study the establishment and development of local provenance (when compared to previous planting activities using non-local seed).

Some reasons for caution in the utilisation of locally-sourced planting material in restoration projects are presented by Smith *et al.* (2007). It is suggested that excessive focus and dedication of time, resources and finance towards maintaining a population's genetic integrity in severely disturbed landscapes may be overlooking the most important restoration requirements. For example, at mine sites or quarry sites, where habitat disturbance may be such that all vegetation and soil have been completely removed, restoring any degree of ecosystem functionality whatsoever would be of greater importance and significance than concerns about genetic integrity.

Smith *et al.* (2005) describe the effect of provenance on the establishment of *Lotus corniculatus* in restoration experiments at a limestone quarry in Dorset, UK. Samples from ecologically distant populations, when planted into un-ameliorated bare clay substrate at the limestone quarry, developed into larger, more fecund plants than ecologically local plants. This, it is hypothesised, may be the result of adaptation of local plants to the limestone rich chalk grassland habitat within which the quarry is located. Several main conclusions were formulated from this study; firstly that there are regional differences between *L. corniculatus* populations which may affect restoration potential; that provenance may have a significant impact upon translocation success rates; and that no consistent home-site advantage in local genotypes was observed.

4.1.5 Habitat matching

Habitat matching is the practice of sourcing planting material from sites that are ecologically similar to the intended site of planting and habitat restoration.

Most authors discussing habitat matching in the context of their research state that it is an issue of great importance in terms of project success, possibly more so than concerns about locally-sourced plant material and maintaining local genotypes. For example, Wilkinson (2001) states that habitat matching is likely to be of greater importance than matching local provenances to restoration sites with different habitat conditions from those of the local population. These views are shared by Walker *et al.* (2004a), Bischoff *et al.* (2006a) and O'Brien *et al.* (2007). Krauss and Koch (2004) consider that it may be possible to apply habitat-matching practices alongside genetic ones, thereby maximising restoration potential. Examples of environmental factors to be considered in habitat matching, both biotic and abiotic, include elevation, soil chemistry, climate regime, and the impact of pathogens and predators (Hufford and Mazer, 2003).

4.1.6 Utilisation of specifically sourced planting material

The earliest investigations into plant provenance (and local adaptations) were carried out by the forestry industry in the early 19th century (O'Brien *et al.*, 2007), when researchers tested different populations of trees for economically important traits, such as survival, growth rate and stem form (Worrell, 1992). When the objectives of forest management were other than timber production, little emphasis was placed upon the importance of seed provenance (Worrell, 1992); thus there is a far smaller pool of knowledge regarding the importance of provenance in conservation practice. However, as Smith *et al.* (2007) indicate, re-vegetation practices have undergone a transition from the almost exclusive use of exotics for planting (or as the case may be, anything that could be obtained), to the wide incorporation and appreciation of the importance of local provenance planting stock.

The use of specific provenances can be successfully incorporated into other land management practices to develop optimised strategies of land care and organisation. For example, Renison *et al.* (2005) discuss the utilisation of selected provenances of *Polylepis australis* to reduce the period for which livestock need to be excluded from re-vegetated degraded woodland sites in central Argentina. The authors found that plants establishing from seed collected from well-preserved woodlands grew both taller and had a shrubbier form than those growing from seed sourced from degraded woodlands.

Incorporation of provenance trials into restoration projects allows valuable research to be conducted into the importance of sourcing and utilising local planting material. Krauss and Koch (2004) state that practical restoration, incorporating specifically provenanced plant material, is the ideal circumstance whereby the pros and cons of utilising both local and non-local populations can be tested, albeit in a non-experimental fashion.

Good *et al.* (1985) offer valuable advice on the practicalities of sourcing plant material for provenance studies. Firstly, it is stated that samples (seed or vegetative cuttings) should only be collected from individuals within populations that exhibit some clear indication that they originated via natural seeding; secondly, individuals within a population growing in atypically favourable conditions when compared to general site characteristics should be avoided; and finally, it is suggested that occasionally clonal sampling is more beneficial than seed collection, since it reduces plant variability.

4.1.7 Problems of local provenance planting

The greatest obstacle to the wider uptake and utilisation of local provenances in restoration and land management projects is that of financial viability. Collection, and propagation if required, of seed of local provenances requires substantially greater time and labour resources than purchasing planting stock. Nurseries that specialise in the collection of locally-sourced seed often have to rely upon volunteer workforces due to the labour-intensive nature of the process (Smith *et al.*, 2007). The costs associated with obtaining seed of specific provenance often far outweigh the potential perceived benefits of maintaining genetic integrity, and unless local adaptation is shown to be practically important, the utilisation of local seed will tend to be overlooked in the majority of restoration activities (McKay *et al.*, 2005).

Other barriers to the use of local provenance in restoration projects include dissemination of knowledge and delineation of ecotype boundaries. For example, Burgin *et al.* (2005) report findings of a survey directed at the land-care sector in Australia. The survey showed that one in five of all groups surveyed had no awareness of the issues and importance of local provenance planting. As a result, it is suggested that a large number of land-care groups may be indirectly inflicting damage upon habitats through their restoration and rehabilitation projects. This may result from a general lack of research on and knowledge of indigenous material performance in restoration (Petersen *et al.*, 2004), and the consequences of out-breeding depression and implications for survival in restored populations (Hufford and Mazer, 2003). It could be claimed that the seed industry is responsible for some of the lack of knowledge regarding local provenance plant material, as it has been reported by Smith *et al.* (2007) that there is significant animosity towards the promotion and use of local ecotypes for restoration work.

Collection of seed or other propagative material (e.g. cuttings) from locally-adapted populations firstly requires population boundaries to be identified. This is not necessarily a straightforward undertaking, as pointed out by Macel *et al.* (2007). Local adaptation may not always exist due to strong gene flow, lack of genetic variation, or temporal fluctuations. An added level of complexity may arise from very localised differentiation of site conditions. For example, Krauss and Koch (2004) point out the not-uncommon occurrence of mosaics of site conditions over small areas. Very localised on-site variation may reflect differences in soil chemistry or substrate type, both of which can alter substantially within a matter of metres. McKay *et al.* (2005) state that there is a pressing need for greater information on the scale of geographic and ecotypic specialisation. O'Brien *et al.* (2007) agree, and note that some of the techniques used to identify genetic differentiation among populations are inappropriate. For example, molecular markers such as amplified fragment length polymorphism (AFLP), restricted fragment length polymorphism (RFLP), allozymes and microsatellites have persistently failed to demonstrate between-population genetic differentiation and do not necessarily reflect patterns of variation in adaptive traits (O'Brien *et al.*, 2007). Wilkinson (2001) points out that there is often no correlation between geographical and molecular genetic distances between populations, and this can make the task of distinguishing populations impossible. However, some authors do advocate the use of molecular genetic techniques: for example, Krauss and Koch (2004) recommend AFLPs for their accuracy and efficiency. Hufford and Mazer (2003) state that molecular markers are particularly useful for identifying strong founder effects, genetic swamping and genetic divergence between populations, all of which may indicate ecotypic adaptation. Despite the perceived advantages of genetic marker techniques, it is likely that most conservation schemes will neither have the expertise or budget available to allow their usage (Wilkinson, 2001).

More often than not, seed mixtures for restoration activities will contain several species, either specifically selected for restoration processes such as phytoremediation, or to fit into local habitats. It is not uncommon for desired (local) seed sources of these species to be unavailable (Stevenson *et al.*, 1997; McKay *et al.*, 2005). An example that typifies the situation well is given by Jones *et al.* (2001); during 1997 80 % of all hawthorn (*Crataegus monogyna*) sold by the horticultural trade in the UK was of European Continental origin, a situation arising because collection of British seed was not financially viable.

Further complications that arise (and that are often overlooked) as a result of the collection of local seed are the impacts upon wild seed reserves and populations. Stevenson *et al.* (1997) state that seed collection, regardless of technique, will have an inevitable impact upon donor sites, reducing subsequent seed availability for dispersal and recruitment. Even non-intensive levels of reduction of regenerative material within populations may cause deleterious knock-on effects, although it is not certain to what extent this may be a problem. The greatest impact will be on populations of annual plant species which rely mostly (some recruitment occurs from soil seed banks) upon annual recruitment and establishment from the seed set of the preceding generation. This cost of restoration activities is hard to quantify (Smith *et al.*, 2007), but could be a serious negative consequence.

It can never simply be assumed that local provenances have greater restoration potential than non-local ones (Walker *et al.*, 2004a). Bischoff *et al.* (2006b) demonstrated better germination in non-local provenances of two (out of three) test species. Worrell *et al.* (2000) report that despite 25 years of provenance studies focussing upon silver birch (*Betula pendula*), understanding of adaptive genetic variation within the species is at best patchy. Many studies take place over a much shorter period, often not exceeding a year (Bischoff *et al.*, 2006a), and it will often be very difficult to draw any firm conclusions from them. Some adaptations arise as a response to infrequent climatic or environmental extremes; short-term experiments are unlikely to be able to account for these, and important differences between test populations may not be detected (Worrell *et al.*, 2000). It is also a possibility that, no matter the length of experimental observations, there is adaptive variation to environmental factors that do not vary across experimental sites. Differences between populations in variation of this kind may be overlooked (Macel *et al.*, 2007).

Bischoff *et al.* (2006a) discuss several points regarding the experimental design of provenance tests. Many such experiments are carried out with reduced competition, resulting from vegetation removal and weeding, although competition within plant communities may actually be a selective force leading to adaptation within a species. If species other than the main test species are removed, population responses may not be properly evaluated. It is inferred that provenance trials, if carried out in laboratory or greenhouse conditions, or at sites away from or non-representative of the restoration site, suffer from the lack of ability to replicate and simulate important environmental conditions, thus providing unrealistic test situations. The authors also point out the importance of observing the life stages of plant development that are applicable to the restoration techniques to be used. For example, where seeding is to be used, germination and emergence rates are important. Therefore, when planning provenance experiments for restoration activities, there should be careful consideration of experimental design to ensure all traits of interest and importance are recorded in sufficient detail.

Smith *et al.* (2007) detail several noteworthy points regarding the sourcing of seed material for utilisation in provenance studies. Restoration practitioners carrying out seeding activities upon disturbed land often have to rely upon seed processors and suppliers for seed resources. They assume that the seed being purchased is exactly as described, despite the fact that it is not uncommon for seed suppliers themselves to be supplied by a number of collectors of unknown experience and integrity. Local ecotype seed is sometimes multiplied to increase seed stocks, but is subject to both agricultural selection and hybridisation during the multiplication stage in seed fields. This creates a situation whereby population adaptations may be lessened, and the seed received by practitioners and other users may be of different quality and have different properties from those initially required and specified. One final point regarding seed sourcing is that across the seed industry, collection, processing and storage methods may vary widely. This non-uniformity is a potential source of error, and might negatively impact observations and results from experimental trials (Keller and Kollman, 1999).

If seed is collected by the practitioners involved in carrying out restoration activities, rather than purchased from specialised suppliers, it is important that seed is collected from as many donor plants as possible, not from one or two very productive individuals. This will maintain genetic diversity and decrease the risk of bottlenecking within the population (Hufford and Mazer, 2003).

A major unknown in the use of non-local provenances is the way in which they will interact with *in situ* remnant populations left after disturbance activities (Walker *et al.,* 2004a). Sackville Hamilton (2001) puts forward the hypothesis that the exclusive usage of locally-provenanced plant material may actually be detrimental to isolated remnant populations at restoration sites, whereby the potential of introduced populations out-competing remnants is high. Bischoff *et al.* (2006b) concur, stating that introduced plant material may be more competitive than *in situ* populations and become invasive. Because of this possibility Wilkinson (2001) states that it is sometimes hard to justify the use of introduced populations into planting schemes.

Wilkinson (2001) points out that quarternary climate change will have affected species' ability to adapt to local conditions. An example given by the author is that of a mature oak tree, the probability being that it germinated during the "little ice age" some 400 years before present. It is questionable if such a tree could produce seeds that could be considered "locally-adapted" in the present day, given the difference in mean annual temperatures of 1.5°C between these periods. In contrast, geographically isolated populations of short-lived species might be expected to show more local adaptation. Discussion of the likelihood of local adaptation should also consider other selection pressures exerted by factors such as soil chemistry and the underlying geology. Where intense selection pressures exist, for example those created by high concentrations of heavy-metals in soils, local adaptation can occur within relatively short timeframes (e.g. hundreds of years). It could be stated that climatic variation influences adaptation on a continental scale, whereas truly local adaptations are driven by habitat factors such as soil properties and underlying geology for example (Macel *et al.*, 2007).

4.1.8 Examples of plant adaptations

Literature references to plant adaptations are generally of two kinds. The first discuss the driving forces behind adaptations (i.e. the causative agents), while the second consider plant responses to the driving forces (i.e. the adaptations resulting in the individual plant or population).

The driving forces responsible for plant adaptations are factors such as climate, soil chemistry, soil type and soil structure, drought and water limitation, freezing events, competition and interaction with other species, presence of toxic compounds or toxic soils, the effects of symbionts, and occasional events, such as wildfires, that are hard to account for experimentally (Good *et al.*, 1985; Worrell, 1992; Keller and Kollman, 1999; Wilkinson, 2001; McKay *et al.*, 2005; Mylecraine *et al.*, 2005; Smith *et al.*, 2005; Bischoff *et al.*, 2006a; Macel *et al.*, 2007; and O'Brien *et al.*, 2007).

Responses of plants to stressing conditions or ecosystem components, can be seen in processes such as germination, establishment, early growth and survival, which can all be enhanced or reduced depending upon the causative agent; altered growth form, degree of branching and productivity; earlier or later bud burst; greater resistance to disturbance, climatic extremes and disease; change in root biomass and form; increased fecundity, decreased seedling mortality and changes in seed dormancy periods (Good *et al.,* 1985; Cundall *et al.,* 1998; Keller and Kollman, 1999; Jones *et al.,* 2001; Wilkinson, 2001; Petersen *et al.,* 2004; Mylecraine *et al.,* 2005; Renison *et al.,* 2005; Smith *et al.,* 2005; Bischoff *et al.,* 2006a; Bischoff *et al.,* 2006b; O'Brien *et al.,* 2007).

Several authors make specific reference to adaptations of plant populations to serpentine soils (McKay *et al.,* 2005; Macel *et al.,* 2007). It is this type of situation where, in the presence of an extreme soil type, strong selection pressures are exerted and it is likely that very localised and obvious plant adaptations will occur (i.e. increased survival in adapted individuals).

One provenance study of hawthorn (*Crataegus monogyna*) carried out by Jones *et al.* (2001) compared UK provenances with those from continental Europe. UK provenances were both thornier and more resistant to mildew. Both of these adaptations will increase survival potential through reducing grazing and susceptibility to disease.

Some adaptations might, at first, not appear to be particularly beneficial. Petersen *et al.* (2004) state that locally-sourced material may often exhibit slower establishment rates than non-local material. Yet these slower-establishing, local individuals often persist through extreme climatic periods and conditions far more successfully than non-local plants. This may be due to adaptive traits such as enhanced resource retention (e.g. the ability to survive with limited water and nutrient availability and the stockpiling of starch from reducing the rate of growth), which would enhance survival in the harshest of situations.

Many plants have adaptations to disturbance events characteristic of their home site; these events may be caused by seasonal wild fires or extreme fluxes in temperature, precipitation or wind speed. Regardless of type, these disturbance events could be seen as drivers of localised plant adaptation. It is reported that many plant species are adapted to periodic disturbance, which allows localised specialisation and increased survival (Knapp and Rice, 1994, cited in Smith *et al.*, 2005) in challenging habitats. Sometimes it may even require disturbance events or extreme episodes to define variation among populations and identify individuals that are able to cope with the selection pressures (Cooper, 1954, cited in Smith *et al.*, 2005).

McKay *et al.* (2005) state that there is also strong evidence of maladaptation in plant populations (no examples provided); plants showing maladaptations are poorly suited to local habitat or climatic conditions. Whether maladaptations are a direct result of evolution and response to selection, or merely reflect a transitory stage, is not made entirely clear.

McKay *et al.* (2005) also raise the point that, on disturbed sites, adaptation to local conditions can be less important than competitive ability. For example, the most competitive species, often non-natives (for instance *Rhododendron ponticum, Fallopia japonica* and *Impatiens glandulifera*), *are* able to exploit bare, disturbed ground of any type without any prior genetic or phenotypic adaptations. This is a result of phenotypic plasticity (McKay *et al.*, 2005) rather than local adaptation, whereby tolerance to a wide range of environmental conditions facilitates rapid development and dispersal, overwhelming less competitive native species.

4.1.9 Recommendations for use of local provenance in restoration projects

Various authors (Good *et al.,* 1985; Renison *et al.,* 2005; Smith *et al.,* 2005; Bischoff *et al.,* 2006a) make reference to observations of non-local populations out-performing local ones, contradicting the home-site advantage theory. Renison *et al.* (2005) suggest that this could be explained by a reduction in the quality of reproductive material (of local populations) as a result of ecosystem disturbance. Plants growing at restoration sites, for example mineral workings, will typically be subjected to both nutrient and water shortages, and their ability to produce healthy seeds is reduced as resources are needed for basic plant physiological functions. Reproductive material bought onto site to supplement disturbed native populations will therefore be more competitive as it will not have faced limiting factors during its generative stages.

Good *et al.* (1985) hypothesise that plants adapted to cope with low nutrient conditions have lower growth rates than individuals growing with readily available nutrient resources. The authors remark that the utilisation of such plants in restoration planting of disturbed, low-nutrient sites may not result in rapid vegetation cover, but the probability of achieving greater survival rates without the need for regular supplementation with fertilisers is higher, and the approach offers a low-input, low-maintenance option.

Smith *et al.* (2005) propose that results demonstrating superiority of non-local over local provenances may be deceptive if the former have not been subjected to important but infrequent environmental conditions such as severe droughts, frosts or heat waves. Local plants, growing on challenging sites, may have adapted to grow at reduced rates and to cope with extreme climatic events, whereas it might be that non-local specimens have no such adaptations. If extreme environmental conditions did occur they could result in high mortality in non-local populations.

O'Brien *et al.* (2007) state that sometimes the use of local planting stock specifically to conserve locally-adapted genotypes may be of little benefit on highly disturbed sites. The authors advocate comprehensive trials incorporating multiple sites and levels of disturbance to better understand the interaction of local adaptation and potential for restoration success (i.e. to pinpoint exactly what adaptations are found in particular populations and how they can be matched to particular sites). Sackville Hamilton (2001) agrees with these comments, stating that generally the use of local provenances is the safest option and should be normal practice, although there is no guarantee that it is the best solution. Expanding, the author remarks that in some situations it is likely to be beneficial if non-local genotypes are introduced to restoration sites, to increase genetic diversity and prevent inbreeding depression, for example. However, these non-local provenances should be chosen with caution and only when there is solid justification for doing so.

Seeds should be collected from a sufficient number of individual donor plants to reduce the chance of inbreeding depression. If only limited numbers of plants are sampled the likelihood of founder effects and genetic bottlenecks is increased (Hufford and Mazer, 2003).

Seed collections made from non-local "wild" populations of unknown history present potential stumbling blocks for restoration practitioners. Sources such as these may be products of hybridisation or indeed remnants of formerly disturbed populations; in these circumstances the potential donor plants may be of questionable genetic quality or reproductive fitness (Smith *et al.,* 2007). This highlights the many pitfalls associated with making assumptions about seed donors. For example, distance between populations is generally not a good indication of genetic similarity (Hufford and Mazer, 2003), neither is it true that locally collected seed is necessarily locally-adapted (Smith *et al.,* 2007). It would be prudent to develop a best practice procedure to take account of both ecological and geographic proximity between donor and receptor sites (Sackville Hamilton, 2001). Seed collection zones that have been tested in thorough research programmes should be delineated (Hufford and Mazer, 2003). Such practices would substantially reduce the potential for failure throughout seed collection and habitat restoration projects.

Several researchers (Cundall *et al.,* 1998; Cundall *et al.,* 2003; Smith *et al.,* 2005; Harris *et al.,* 2006) have suggested ways in which experimental practices can be modified so that major potential benefits to provenance testing can be achieved. For example, by using a series of common garden trials it should be possible to reproduce various regional conditions (Harris *et al.,* 2006) (e.g. rainfall, temperature, substrate pH etc.), identify desirable growth or survival traits (Cundall *et al.,* 1998), and distances from its source over which plant material can be successfully established (Cundall *et al.,* 2003). Data from these preliminary provenance tests can then be used to develop and carry out *in situ* large-scale experiments (Smith *et al.,* 2005).

Ultimately, despite all the best efforts of restoration practitioners, questions will always be asked about the worth and quality of restored habitats compared to the conservation of existing ones (Stevenson *et al.,* 1997; McKay *et al.,* 2005). However, if populations of self-sustaining plants can be introduced successfully to otherwise derelict disturbance sites, restoration should be considered successful and an important part of current conservation strategy, regardless of the genetic similarity of introduced populations to native pre-disturbance populations (Stevenson *et al.,* 1997; McKay *et al.,* 2005).

4.1.10 Research context and experimental brief

Restoration efforts have previously been attempted at Penrhyn Quarry but have met with limited success. There is a requirement to develop successful restoration strategies to satisfy conditions imposed by the local authority for future expansion of quarrying operations. Current restoration practice at Penrhyn Quarry involves the sourcing and collection of seed from local areas, but not specifically from within the confines of the quarry itself. Therefore, despite the maintenance of local genetic integrity, the possibility of enhancing the success of restoration projects through the incorporation of locally-adapted seed populations into planting strategies is being overlooked.

Given the harsh nature of both ground and climatic conditions at Penrhyn Quarry, it is considered possible that the numerous small populations of plants that have naturally colonised or regenerated, following disturbance by quarrying, may have specific adaptations to quarry conditions. If adaptations do exist within quarry plant populations that give improved growth and survival, future restoration planting should consider utilising on-site seed resources.

4.1.11 Research objectives

- To compare the survival, establishment and growth of quarry provenances (sourced from Penrhyn Quarry) and local non-quarry provenances of a range of plant species.

- To infer whether naturally-colonised or naturally-regenerated quarry populations of plant species have developed adaptations to quarry conditions.

- To recommend whether current seed sourcing practice should be replaced with one that uses collections from on-site quarry populations.

All experimental trials will be short-term and conducted under greenhouse and nursery conditions.

4.1.12 Hypotheses

Comparative growth bioassays with plants sourced from quarry and non-quarry populations will allow the following hypotheses will be tested:

- That plants sourced from different populations grow at different rates when planted into artificial slate waste tips.
- That plants from the population exhibiting the most rapid establishment and development over the duration of experimental observations will provide the most successful option for long-term quarry restoration practices.
- And, that it will be possible to identify whether any clear local/ecotypic adaptation is or is not present within quarry sourced plants.

4.2 Methods

4.2.1 Willow clones

4.2.1.1 Collection of vegetative material

It was decided to utilise the significant population of naturally regenerated willow plants present at Penrhyn Quarry as a source of plant material with which to conduct comparative growth studies. The species of willow present is a hybrid of *Salix caprea* and *Salix cinerea* known as *Salix caprea* × *Salix cinerea* = *Salix reichardtii*.

Vegetative cuttings from nine willow trees growing on dormant slate waste tips at Penrhyn Quarry ("quarry trees") were collected on May 3rd 2006. Cuttings were taken from straight sections of tree branch and measured at least 250 mm in length and more than 10 mm in diameter. Cuttings were taken using a pair of secateurs with a fine, sharp cutting edge, providing a clean-cut surface. The cuttings were cut flat at the basal end (i.e. cut at 90° to the direction of stem growth), and angled at the apical end (i.e. cut at 45° to the direction of stem growth). This cutting technique allowed easy identification of apical and basal ends for planting, and the angled apical surface minimised water accumulation and rot development. All cuttings from an individual donor tree were placed in a polythene sample bag lined with damp newspaper. A GPS point was recorded for the location of the donor tree.

Several non-quarry populations of willow, with phenotypes similar to those of trees at Penrhyn Quarry, were located in the surrounding area. Cuttings were collected in the same manner as utilised for quarry trees, from eight individual trees ("non-quarry trees") at three locations. Two trees growing on the periphery of rough grazing pastural land adjacent to a roadside several kilometres away from the settlement of Llanbedr-y-cennin in the Conwy valley were sampled on May 1st 2006. Four individual trees located in fairly scrubby woodland adjacent to the Welsh Highland railway line near Rhyd Ddu were sampled on May 2nd 2006. And two trees growing in a hedgerow on the grassy margin between rough grazing land and a road at Cwm-y-glo were sampled on May 4th 2006.

See table 4.1 for locations and details of individual quarry and non-quarry donor trees.

All vegetative cuttings were kept in fresh clean water (cuttings submerged basal end first to approximately half their length) for at least seven days prior to insertion into the experimental substrate.

Table 4.1 Willow sampling location data

Source location	Provenance	Tree ID	NGR	Altitude (m.a.s.l)	No. of cuttings	Used in Penrhyn trials	Used in Penyffridd trials
Penrhyn	Quarry	QT1	SH 62629,64813	279	35	yes	yes
Penrhyn	Quarry	QT2	SH 62616,64809	282	23	yes	
Penrhyn	Quarry	QT3	SH 62642,64796	277	26	yes	
Penrhyn	Quarry	QT4	SH 62676,64993	251	23	yes	
Penrhyn	Quarry	QT5	SH 62682,65007	250	45	yes	yes
Penrhyn	Quarry	QT6	SH 62715,64987	253	16		
Penrhyn	Quarry	QT7	SH 62427,65832	171	56	yes	yes
Penrhyn	Quarry	QT8	SH 62434,65835	175	22	yes	
Penrhyn	Quarry	QT9	SH 62434,65851	170	32	yes	yes
Conwy	Non-quarry	CT1	SH 74917,69678	263	42	yes	yes
Conwy	Non-quarry	CT2	SH 74664,69524	301	40	yes	
Rhyd Ddu	Non-quarry	RDT1	SH 57369,50309	170	33	yes	yes
Rhyd Ddu	Non-quarry	RDT2	SH 57447,50426	176	31	yes	yes
Rhyd Ddu	Non-quarry	RDT3	SH 57445,50431	172	28	yes	
Rhyd Ddu	Non-quarry	RDT4	SH 57450,50430	171	24	yes	
Cwm y glo	Non-quarry	CYGT1	SH 55850,62592	108	59	yes	yes
Cwm y glo	Non-quarry	CYGT2	SH 55680,62347	107	46	yes	

4.2.1.2 Clonal tests: stage 1

<u>*Artificial slate waste tips*</u>

Six box pallets, ordinarily employed to transport roofing slates from quarry to building site, were used to create artificial slate waste tips. The dimensions of the box pallets were 105 × 107 × 76 cm, providing a volume of 0.85 m^3. Durable builders plastic sheeting (polyethylene film, also known as Visqueen) material was used to line the box pallets, and prevent excessively rapid drainage. Sheeting was attached to the wooden box pallet with staple gun tacks, and approximately 20 drainage holes ≤10 mm diameter were drilled into the base of the pallets. Large fraction (≥30 cm diameter) pieces of slate waste were placed in the box pallets, with minimal additions (10-15 shovel loads, equivalent to approximately 20-30 litres) of slate processing fines (slate sand) (0-4 mm diameter particles) (plate 1). This process was repeated four times, resulting in a layered effect with many large pieces of slate, minimal amounts of fines material and a high proportion of interstitial void spaces. This was intended to reproduce the conditions present in the surface layers of slate waste tips at Penrhyn Quarry. A layer (approximately 125 mm deep) of slate sand was placed on the upper surface of the blocky slate layers in each box pallet surface, and willow cuttings were inserted into this.

Three of these artificial slate waste tips were made in the tree nursery at Penrhyn Quarry, and three within a greenhouse at Pen-y-ffridd research station, Bangor (NGR SH 56140,70325). The boxes at Pen-y-ffridd were intended as backups, should the outdoor experiment at the quarry fail.

The surface of each artificial slate waste tip was divided into 16 sections (each section measured 13.13 × 53.5 cm) using string tacked onto the box pallet edges. The sections provided treatment plots for the experiments described below.

Five planting wells, 125 mm deep, were created in each plot (i.e. 80 wells per box). Wells on the periphery of each section were positioned 25 mm from the pallet edge; successive wells were laid out 100 mm apart, providing a planting density equal to 100 plants m^{-2} (1, 000, 000 ha^{-1}).

Insertion of cuttings

Following the seven-day period of soaking, all willow cuttings were cut to precisely 250 mm in length, again with sharp, fine cutting secateurs, and inserted into the planting wells. Cuttings were inserted such that 100 mm of their length was inserted into the planting media, and 150 mm remained above the surface (plate 2). Slate sand surrounding willow cuttings was packed down firmly to eliminate void spaces in the rooting zone. Following this tamping down, all cuttings were thoroughly watered in.

Plate 1 Artificial slate waste tips at Penrhyn Quarry, part way through filling with large fraction pieces of slate waste (left box) and addition of slate sand (right box)

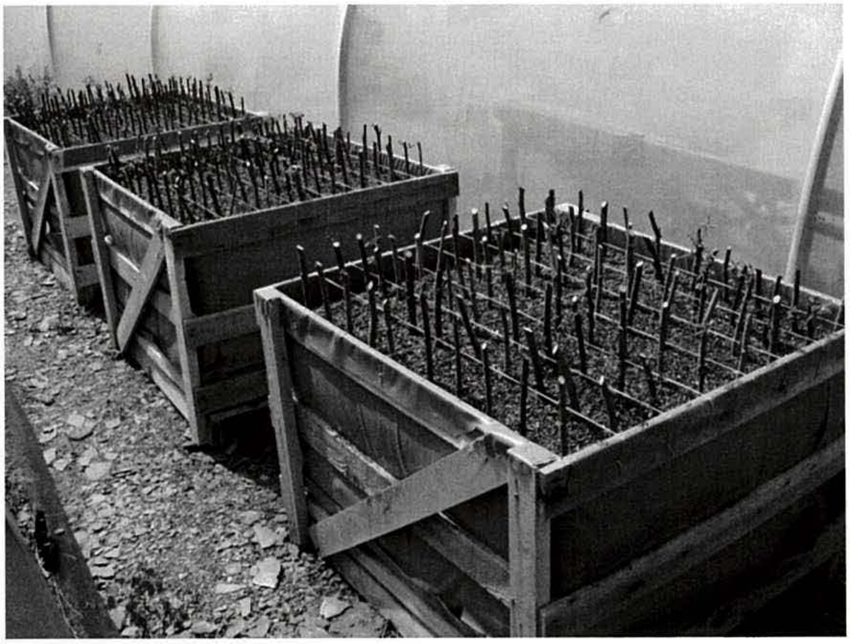


Plate 2 Artificial slate waste tips at Penrhyn Quarry following insertion of willow cuttings

Experimental design

The experimental design of the artificial slate waste tip planters was a randomised complete block design for planters located at both Penrhyn Quarry and Pen-y-ffridd research station. Planters at Penrhyn Quarry were planted with five cuttings from each of sixteen donor trees (8 × quarry (Q), 8 × non-quarry (NQ)) giving a total of 240 cuttings (3 boxes/blocks × 16 clones × 5 cuttings per clone). Planters at Pen-y-ffridd were planted with five cuttings from each of eight donor trees (4 × Q, 4 × NQ) giving a total of 120 cuttings (3 bloxes/blocks × 8 clones × 5 cuttings per clone).

Watering

During the period of experimental observations (from May 2006 until December 2007) no nutritional additions were made to the artificial slate waste tips. Regular watering was carried out through the summer months of 2006 (at both sites), using collected rainwater. Watering was carried out pragmatically, by observing physiological condition and general plant health, and by noting the moisture content of the slate sand growth medium.

4.2.1.3 Clonal tests: stage 2 (influence of drought stress)

Following 13 months of experimental observations, it was decided to introduce drought stress as an additional factor to the three artificial slate waste tip boxes at Penrhyn Quarry. In order to facilitate this, three hoods were made (one for each box) to exclude all atmospheric precipitation.

Hoods were constructed from roofing batten, speed-fit flexible polyethylene plumbing pipe, polyethylene pipe lagging, and a quadruple layer of transparent builders polythene sheeting, forming small poly-tunnel style shelters that fitted snugly on top of each box (plate 3). The maximum height of each hood (i.e. height at the apex of the arching hood) was 60 cm, therefore allowing sufficient room for continued shoot growth.

Watering regimes were calculated using climate change data provided by UKCIP (UKCIP, 2007), assuming a 2020, high-emissions scenario that predicted annual precipitation 10 % to 20 % below current ambient levels. It was therefore decided to use three watering regimes: one replicate artificial slate waste tip receiving ambient −20 %, one ambient +20 %, and the final replicate receiving ambient levels. Assuming 2472.2 mm year^{-1} precipitation (mean figure calculated from 1960 – 1999 data for Bethesda, Gwynedd (UKCIP, 2007)), the other watering regimes were calculated to be 1977.76 and 2966.64 mm year^{-1}. From monthly breakdowns of precipitation for the location and the surface area of the artificial slate waste tips, it was possible to calculate weekly amounts of precipitation, assuming three rainfall events per week, for each calendar month of the study (see table 4.2).

Plate 3 Hoods positioned on artificial slate waste tips at Penrhyn Quarry to exclude all atmospheric precipitation

Watering regimes were randomly allocated to the three artificial slate waste tip boxes using a random number generator (Anon, 2006b). This meant that watering regime and box (replicate) were completely confounded, but results prior to the start of the watering treatments had shown that there were no significant differences among blocks (see section 4.3).

On each of the thrice weekly watering occasions, the specified quantity of water, depending on month and watering regime (see table 4.2), was measured into a watering can, the hoods on the artificial slate waste tip boxes were propped open, and water was applied to the willow cuttings.

Table 4.2 Monthly watering rates assuming ambient (precipitation), ambient −20 %, and ambient +20 %

Watering regime	Month	Rainfall/week (mm)	Watering level (litres/week)	Water level (litres) (3 occasions/week)
Ambient	July	32.06	32.33	10.78
Ambient	August	45.01	45.39	15.13
Ambient	September	50.96	51.39	17.13
Ambient	October	59.08	59.58	19.86
Ambient	November	67.06	67.62	22.54
Ambient	December	65.87	66.42	22.14
Ambient -20%	July	25.65	25.86	8.62
Ambient -20%	August	36.01	36.31	12.10
Ambient -20%	September	40.77	41.11	13.70
Ambient -20%	October	47.26	47.66	15.89
Ambient -20%	November	53.65	54.10	18.03
Ambient -20%	December	52.70	53.14	17.71
Ambient +20%	July	38.47	38.80	12.93
Ambient +20%	August	54.01	54.47	18.16
Ambient +20%	September	61.15	61.67	20.56
Ambient +20%	October	70.90	71.49	23.83
Ambient +20%	November	80.47	81.15	27.05
Ambient +20%	December	79.04	79.71	26.57

4.2.1.4 Data collection

Data were collected monthly from June 2006 to December 2007. Shoot count, length of the longest individual shoot per clone (total shoot length, i.e. from the point of emergence from cutting to tip of shoot), total shoot diameter per clone (measured with digital vernier callipers at the point of emergence from the cutting, all shoots per cutting measured and totalled, e.g. if 5 shoots each measuring 2 mm in diameter were present, the total shoot diameter for that clone would be 10 mm), and survival were recorded. Plate 4 shows the 3-month (August 2006) shoot growth produced by willow cuttings at Penrhyn Quarry.

N.B. Data for the drought experiment were calculated as percentage changes in shoot count, length of longest shoot and total shoot diameter. Starting values were the last values recorded in the 13-month measurements in stage 1 of the clonal test, prior to the implementation of drought conditions.

Following the final collection of growth data in December 2007, all shoots produced from each cutting were harvested using secateurs, and placed into labelled paper bags. Total fresh shoot biomass per cutting was recorded. Following a minimum period of 120 hours drying in an oven set to 70°C, total dry shoot biomass per cutting was recorded. The willow cuttings still in the artificial slate waste tip boxes but now minus the shoots, were labelled using a permanent marker pen to write directly onto their bark. Using a combination of crowbar and hammer, the box pallet sides were removed. Slate blocks and sand were then carefully excavated allowing the mass of roots to be disentangled and cuttings to be carefully separated from each other. The roots of the separated cuttings were harvested, again using secateurs, rinsed under running cold water until all removable slate sand had been washed away, and then gently patted dry with paper towel. After a period to allow excess remaining water to evaporate, the roots were placed in labelled paper bags, weighed, dried in a 70°C oven for a minimum period of 120 hours and re-weighed to determine dry weight and allow calculation of moisture content.

Plate 4 Shoot growth from willow cuttings at Penrhyn Quarry three months after insertion (August 2006)

Phenological data were recorded throughout the duration of the experiment. Cuttings at both Penrhyn Quarry and Pen-y-ffridd showed clonal differences during bud opening and leaf emergence in spring 2007. Phenological development was recorded using scores suitable for analysis (see tables 4.3a and 4.3b for scoring criteria). N.B. Where the term "buds" is used here it is in reference to leaf buds only.

Table 4.3 Scoring criteria for Pen-y-ffridd (A) and Penrhyn (B) willow cutting phenological observations

A

Observation	Score
All new leaves	1
All old leaves	2
Old and new leaves	3
Leaf buds opening	4
No leaves- tight in bud	5
No leaves or buds- dead?	6
No leaves- in bud opening	7
Dead	8

B

Observation	Score
Lifeless/dead	1
Buds tight	2
Buds ready to open	3
Buds opening	4
Buds open	5
Leaves just forming	6
Leaves fully open	7

Cuttings at Pen-y-ffridd also showed clonal differences in leaf senescence, some leaves dropping early during autumn 2006, and some leaves remaining on shoots whilst buds were opening in 2007. A Minolta SPAD-502 chlorophyll meter (Konica Minolta, Tokyo, Japan) was used to record the range of leaf colouration during this period. Five leaves per cutting were analysed in this way. Recording of chlorophyll content was carried out once only.

4.2.2 Provenance trials

4.2.2.1 Seed collection

Seeds of beech (*Fagus sylvatica*), birch (*Betula pendula*), broom (*Cytisus scoparius*), gorse (*Ulex europaeus*), oak (*Quercus petraea*), rowan (*Sorbus aucuparia*) and sycamore (*Acer pseudoplatanus*) were collected in Penrhyn Quarry (i.e. from quarry populations) during autumn 2006. Substantial quantities of gorse, broom, birch, sycamore, rowan and beech were collected from plants and trees growing in blocky slate waste within the confines of the quarry complex. Due to the limited numbers of oak established upon the blocky slate waste tips at Penrhyn Quarry, and the stunted nature of individual specimens resulting from the harsh conditions presented by slate waste, very few seeds are produced by this species. As a result it was only possible to collect approximately 30 oak seeds (acorns), and full-scale experimental tests with this species were not carried out.

Non-quarry provenances of beech, sycamore, and gorse were sourced from locations within Bangor; rowan (Moel-y-ci), oak (Nant Ffrancon), and broom and birch (Newborough) were sourced from a selection of locations within the Forestry Commission 303 seed region.

All non-quarry provenance seeds were collected from areas considered to be non-limiting with regard to plant-available water and nutrients, therefore representing a contrast to growth conditions at Penrhyn Quarry.

See table 4.4 for details of quarry and non-quarry provenances.

Table 4.4 Details of quarry and non-quarry provenances

Species	Provenance	National grid reference	Date collected	Date sown	No. of donor plants
Sycamore	Quarry	SH 616,660	16.10.2006	16.01.2007	2
Rowan	Quarry	SH 625,654	23.08.2006	17.01.2007	2
Birch	Quarry	SH 620,659	01.11.2006	20.01.2007	4
Oak	Quarry	SH 616,660	01.11.2006	20.01.2007	6
Broom	Quarry	SH 626,650	31.07.2006	24.01.2007	10+
Gorse	Quarry	SH 626,650	01.08.2006	24.01.2007	10+
Beech	Quarry	SH 624,659	01.11.2006	N/A	4
Sycamore	Non-quarry	SH 577,727	03.11.2006	16.01.2007	3
Rowan	Non-quarry	SH 598,663	23.08.2006	17.01.2007	5
Birch	Non-quarry	SH 414,641	03.11.2006	20.01.2007	4
Oak	Non-quarry	SH 629,647	01.11.2006	20.01.2007	4
Broom	Non-quarry	SH 414,641	09.10.2005	24.01.2007	10+
Gorse	Non-quarry	SH 583,719	23.08.2006	24.01.2007	10+
Beech	Non-quarry	SH 591,718	03.11.2006	N/A	5

4.2.2.2 Seed processing and treatment

Seeds collected from gorse, broom, beech, rowan, and birch all required some degree of processing. For gorse, broom and beech this meant the splitting of pods and removal of the seeds. Rowan seed was extracted by splitting and squashing the fruit in a food processor and rinsing the seed into a sieve. Birch seed also required sieving to break down individual catkins and remove husk material.

Unfortunately almost all beech seed collected was non-viable. The majority of nutlets extracted from husks were merely empty shells; no embryo had formed within the testa. This was the case for both quarry and non-quarry provenances and, therefore, no further tests were conducted with this species.

After seed processing, the guidelines of Gordon and Rowe (1982) were followed to treat seed in order to break seed dormancy. Table 4.5 shows the treatments used for the seven test species.

Table 4.5 Guidelines for breaking seed dormancy in test species (Gordon and Rowe, 1982)

Species	Pretreatment to break seed dormancy
Birch	Naked seed, 4 weeks cold
Broom	Treat with boiling water/conc. sulphuric acid/mechanically scarify
Gorse	Treat with boiling water/conc. sulphuric acid/mechanically scarify
Beech	Naked seed, 4-20 weeks cold
Rowan	Extract seed from fruit, mix with compost, 2 weeks warm then 14-16 weeks cold
Sycamore	Naked seed, 6-12 weeks cold
Oak	Moist peat 4-8 weeks cold

Scarification of both gorse and broom seeds was carried out by using a sharp-tipped penknife blade to remove a small ($\leq 4\ mm^2$) amount of the testa of each individual seed. Ordinarily this process might be carried out using sand paper, for example, to scarify many seeds in one go, this, however, does not guarantee that every seed is scarified to the same degree.

Additional pre-treatment to that suggested by Gordon and Rowe (1982) was carried out on duplicate sets of broom and gorse seeds. This involved simply soaking the seeds in warm water for several days prior to sowing, a method removing the need for time-consuming scarification. This method was employed following the advice of restoration practitioners working at Penrhyn Quarry.

In order to manage the very small and mobile birch seeds, they were mixed with a small quantity of damp slate sand. Seed from the eight (4 × Q, 4 × NQ) trees sampled was kept separate, five grams of sieved seed (equivalent to 10510 seeds) from each tree was mixed with 300 g damp 0 – 4 mm slate sand. These mixtures of seed and slate were placed into labelled polythene bags, loosely sealed, and placed into cold storage (5°C) for a minimum period of 30 days.

4.2.2.3 Seed sowing

Early in January 2007 all seeds had undergone the necessary pre-treatment to break seed dormancy, and were therefore ready to sow (see table 4.4 for dates of sowing).

Seed trays, measuring 35 cm × 21 cm (internal dimensions), were filled with two litres of 0 – 4 mm slate sand. Four replicate seed trays for each provenance of each species were set up in this manner (i.e. eight seed trays per species). With the exception of birch, 100 seeds were sown in each seed tray (i.e. 400 seeds of each provenance of each species). Polythene bags containing birch seed and slate sand mixtures were emptied into the pre-prepared seed trays, thus sowing the pre-treated seed.

Seed trays were arranged on benches within a temperate greenhouse (see section 4.2.3 for greenhouse conditions). The experiment used a split plot design with four blocks, seed provenance (i.e. quarry and non-quarry) as main plots and species as sub-plots. There were seven sub-plots in each main plot (five test species plus two additional treatments for soaked broom and gorse).

4.2.2.4 Data collection

Regular observations (every 2 – 4 days) were carried out to record seed germination. When germination reached 50 % in an individual seed tray, 30 seedlings were transplanted, selecting those appearing to be of optimum condition. Germinant counts were continued until 1st April 2007, a maximum germination period of 75 days (first species sown 16th January 2007, last observation 1st April 2007). However, sampling points (1 – 13) were not equally spaced for all species as not all seeds were sown on the same day (see table 4.4 for sowing dates). However, provenances of the same species were sown on the same date.

N.B. No further tests were carried out with gorse or broom seed that underwent the soaking pre-treatment; this treatment was applied simply to evaluate its effect on germination.

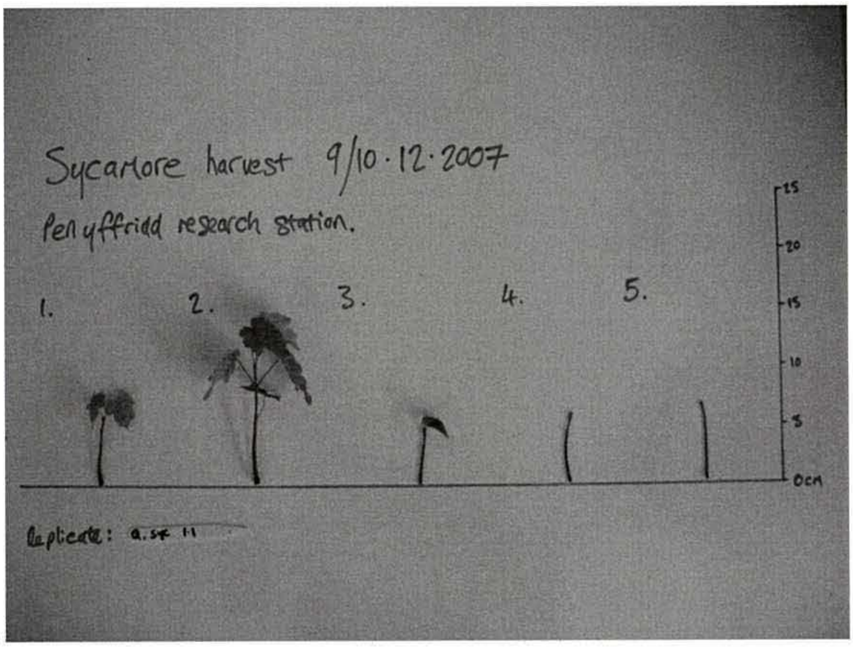
Seedlings removed from germination seed trays were transplanted into scaled-down versions of the artificial slate waste tips as described in section 4.2.1.2. Flowerpots of 5 litres volume were filled with mixtures of slate; 3 litres of roughly circular large-fraction slate aggregate (<100 mm diameter) was combined with minimal quantities of slate sand (0 – 4 mm diameter) in a stratified pattern, with a bulk slate sand layer on the surface of each pot. This mixture was intended to reproduce conditions found on slate waste tips at Penrhyn Quarry; i.e. blocky in nature, with minimal fines material, and a complete absence of organic matter. Five seedlings were transplanted into each 5 litre flowerpot, giving a total of 120 individual seedlings per provenance (4 germination seed trays $\times$ 30 seedlings transplanted from each tray). Plate 5 shows 11-month (December 2007) growth of broom, gorse and sycamore in the slate-filled flowerpots positioned on greenhouse benches at Pen-y-ffridd.

Transplanting of birch seedlings was carried out in a slightly different fashion. It was anticipated that birch, being a quick growing species of tree, would benefit if transplanted into larger containers. Birch seedlings were, therefore, transplanted into empty plots in the artificial slate waste tips at Pen-y-ffridd research station (see section 4.2.1.2). Ten seedlings from each donor tree (four quarry and four non-quarry) were transplanted into one of the eight plots in each of the three replicate artificial slate waste tips (therefore the total number of plants per provenance = 120). Donor trees were randomly allocated to plots.

Monthly records of survival and shoot length (total cumulative) were made for all species. For broom and gorse shoot length measurements were made on the longest individual shoot, recording length from the point of branching to shoot tip. For sycamore and birch seedlings the length of the main shoot, from soil (slate) surface to the tip of the stem (location of developing leaf bud), was measured.

Plate 5 Broom, gorse and sycamore 11-month (December 2007) growth in slate-filled flowerpots at Pen-y-ffridd

Measurements of shoot length continued until December 2007, at which point all vegetative matter was harvested for biomass determination. Each individual flowerpot was emptied carefully, allowing all roots of individual plants to be teased apart and separated. Roots were then rinsed under cold running tap water to remove all attached slate sand, and patted dry with paper towels. Shoots and roots were separated and placed into labelled paper bags for fresh weight determination. (Plate 6a and 6b show quarry and non-quarry sycamores following harvesting). Following weighing, all harvested biomass was dried in a 70°C oven for a minimum period of 120 hours, after which dry weights were measured, and calculations of moisture content were carried out.



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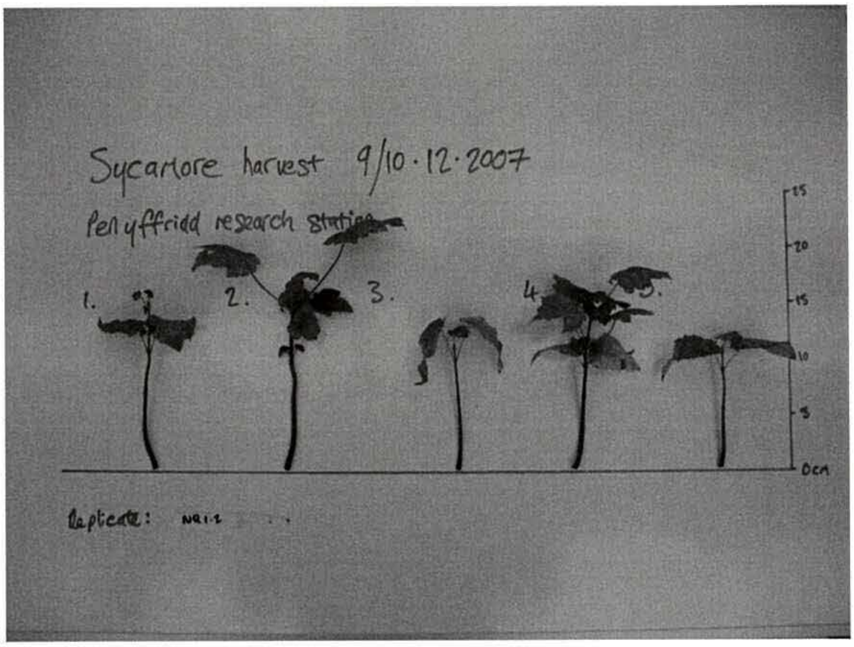


B

Plate 6 Quarry (A) and non-quarry (B) sycamore following harvesting (December 2007)

Only the above-ground part of birch seedlings was harvested; roots could not be harvested without disturbing adjacent rooted willow cuttings. Birch seedlings were cut at soil (slate) level using sharp fine tipped secateurs, placed into labelled paper bags, weighed (fresh weight), dried in a 70°C oven for 120 hours minimum, and re-weighed (dry weight); moisture content was then calculated.

Before weighing the gorse plants, their colour was determined by the utilisation of the Horticultural Colour Chart (part 2) issued by the British Colour Council in collaboration with the Royal Horticultural Society (Wilson, 1938).

4.2.3 Greenhouse environmental conditions and plant husbandry

The greenhouses at Pen-y-ffridd research station provided temperate growing conditions. Daytime (18°C) and night-time (15°C) temperatures were kept constant by adjustments of venting and utilisation of heat lamps (Osram vialox nav-T (son-T) 400W high pressure sodium lamps (Osram, Germany)). Time-controlled lighting was set, providing a photoperiod of 16 hours daylight and 8 hours darkness.

Watering was done manually, generally at least once daily, with the exception of some days during the winter months when watering was unnecessary. Water was sourced directly from the mains supply. No fertilisation or provision of nutrients was conducted.

4.2.4 Statistical analyses

Data were statistically analysed using SPSS 14.0 (SPSS Inc., 2007). All data were tested for normality using the Kolmogorov-Smirnov test, which demonstrated that the majority of data were not normally distributed. Additionally it was observed (by using Levene's test of equality of error variances) that data often demonstrated a lack of homogeneity. Due to regular violations of the assumptions of equality of variances and normality, parametric analysis with T-tests and analysis of variance (ANOVA) was not possible. All data were, therefore, analysed using Mann-Whitney and Kruskal-Wallis non-parametric methods.

Correlations between willow cutting volumes and growth of rooted cuttings were calculated using data from Penrhyn Quarry. Cutting volume (and the majority of growth variables) was found to be non-normally distributed; as a result Spearman's rank correlation was calculated.

4.3 Results

4.3.1 Willow clonal tests: stage 1 (Penrhyn Quarry)

4.3.1.1 Survival and growth

Survival was good for both quarry (66.7 %) and non-quarry (67.5 %) clones, with no significant differences between the two sources (p = 0.891). As demonstrated in figure 4.1, shoot count, longest shoot length, and total shoot diameter over the period of experimental observations (13 months) were all greater in non-quarry clones.

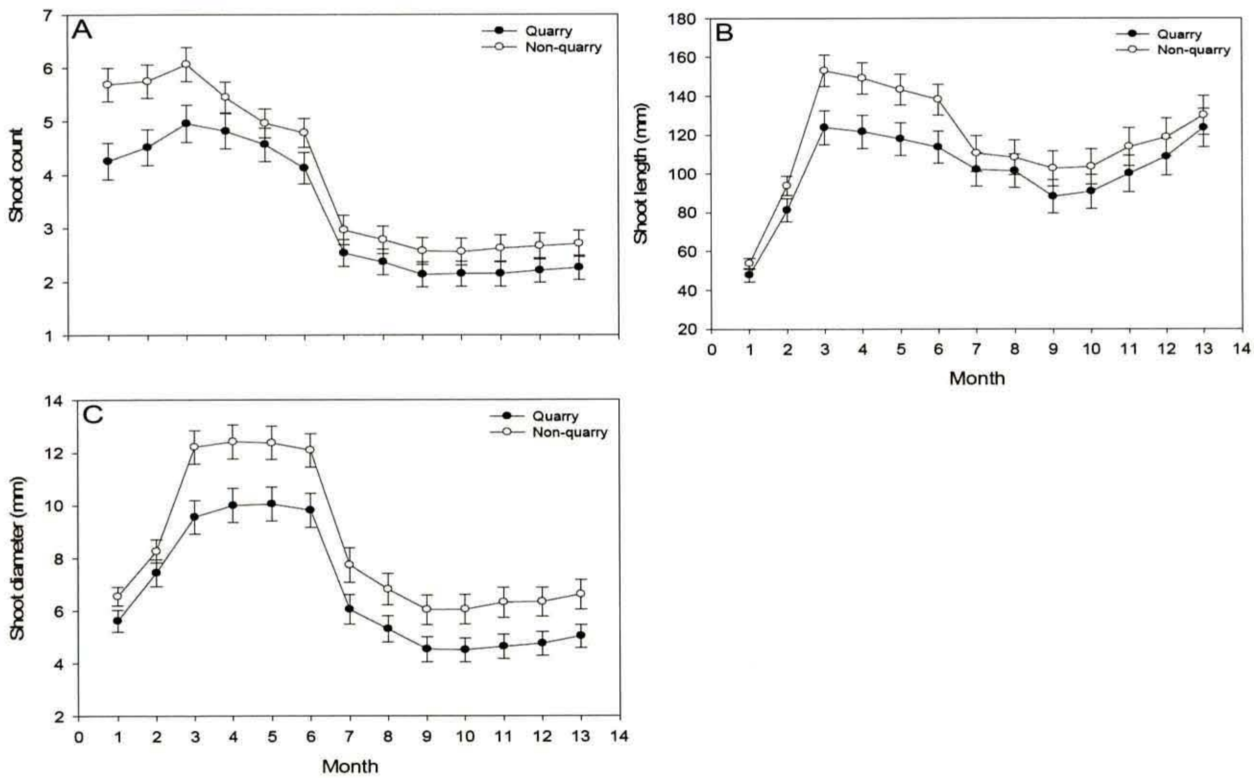


Figure 4.1 Mean shoot count (A), mean length of longest shoot (B), and mean total shoot diameter (C) of rooted cuttings of quarry and non-quarry willow clones (Penrhyn Quarry experiment) (bars = standard error of the mean)

Despite the superiority of non-quarry plants, significant differences were only observed during the initial six months of observations (table 4.6). Maximum mean shoot count for both quarry and non-quarry clones occurred at month 3 (August 2006) (Q = 4.96, NQ = 6.07); after this point, mean shoot count decreased every month until the start of 2007 (quarry = January, non-quarry = February) at which point observations stabilised. Similarly, mean length of the longest shoot was greatest in August 2006 (Q = 123.68 mm, NQ = 152.93 mm). The greatest mean total shoot diameter however, was recorded at observation point 4 (September 2006) in non-quarry clones (12.42 mm), and point 5 (October 2006) in quarry clones (10.05 mm).

Table 4.6 Significance of differences in shoot development of quarry and non-quarry clones of willow (Penrhyn Quarry experiment). Significant values shown in bold

Sample occasion	Shoot count	Shoot length	Shoot diameter
June 2006	**0.0010**	0.1647	0.0644
July 2006	**0.0044**	**0.0483**	0.1920
August 2006	**0.0126**	**0.0104**	**0.0036**
September 2006	0.1033	**0.0140**	**0.0092**
October 2006	0.2693	**0.0224**	**0.0146**
November 2006	0.0810	**0.0243**	**0.0168**
December 2006	0.3262	0.5107	0.1130
January 2007	0.3202	0.5538	0.1198
February 2007	0.2362	0.2502	0.0773
March 2007	0.3168	0.3648	0.0993
April 2007	0.1850	0.3089	0.0594
May 2007	0.2368	0.4573	0.0884
June 2007	0.2308	0.5790	0.0710

Analysis of development data (i.e. presence or absence of buds, the state of budburst, and the presence or absence of leaves) gathered from Penrhyn willows during April 2007 demonstrated no significant differences between quarry and non-quarry clones ($p = 0.426$).

4.3.1.2 Biomass and moisture content

There were no significant differences (e.g. aboveground FW $p = 0.104$ and belowground FW $p = 0.293$) in biomass between non-quarry and quarry clones (figure 4.2). Similarly, there were no significant differences (aboveground $p = 0.331$, belowground $p = 0.805$) in moisture content between quarry and non-quarry clones.

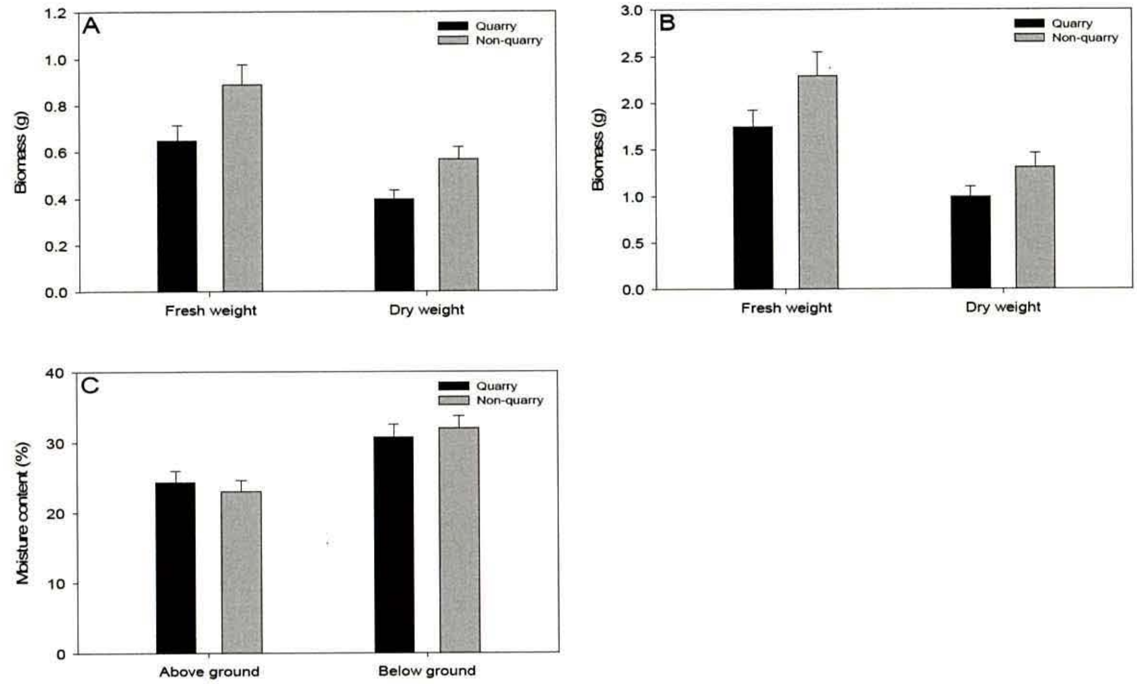


Figure 4.2 Aboveground (A) and belowground (B) biomass, and moisture content (C) of rooted cuttings of quarry and non-quarry willow clones (Penrhyn Quarry experiment) (bars = standard error of the mean)

4.3.2 Willow clonal tests: stage 1 (Pen-y-ffridd)

4.3.2.1 Survival and growth

Survival of willow cuttings at Pen-y-ffridd research station (combined mean survival = 85 %) was higher than that of cuttings at Penrhyn Quarry (combined mean survival = 67.1 %). Unlike the experiment at Penrhyn Quarry, however, there was a significant difference ($p = 0.002$) in survival between quarry (75.0 %) and non-quarry (95.0 %) clones. Growth closely mirrored that of clones at Penrhyn Quarry, as shown in figure 4.3. At all but two observation times (mean shoot count September 2006 (Q = 5.72, NQ = 5.65) and October 2006 (Q = 5.62, NQ = 5.43)), non-quarry clones had greater mean values of shoot count, longest shoot length, and total shoot diameter.

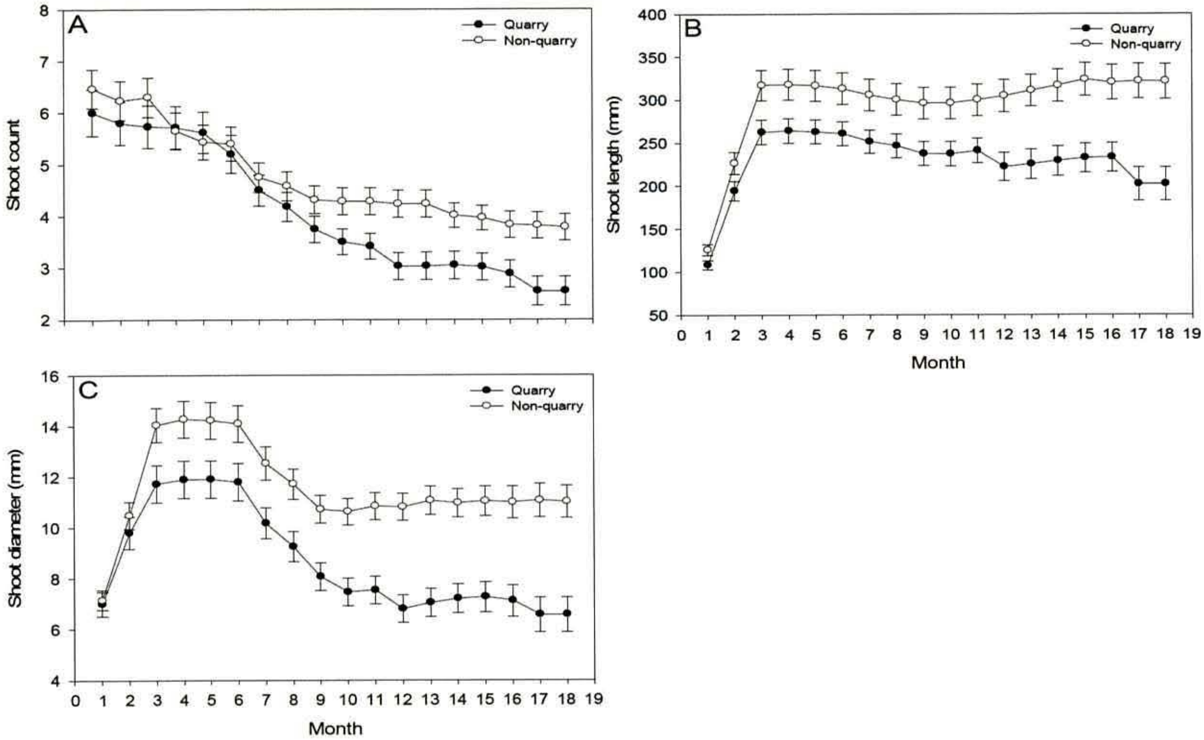


Figure 4.3 Mean shoot count (A), mean length of longest shoot (B), and mean total shoot diameter (C) of rooted cuttings of quarry and non-quarry willow clones (Pen-y-ffridd experiment) (bars = standard error of the mean)

Both quarry and non-quarry clones produced the greatest mean shoot count values (Q = 6.00, NQ = 6.47) during the first month (June 2006) after insertion. From this point onwards both showed steady and consistent declines (November 2007 mean shoot count – Q = 2.55, NQ = 3.78). A similar trend was observed for mean longest shoot length of quarry clones. After initial rapid growth, the maximum shoot length of 264.05 mm was achieved four months after insertion (September 2006), decreasing monthly until August 2007 (240.80 mm) and then fluctuating at values just above 200 mm until the end of observations in November 2007. Non-quarry clones showed similar initial growth spurts until October 2006, then a small decrease, and then an increase to a maximum mean longest shoot length of 323.38 mm in August 2007. As in the Penrhyn Quarry experiment, non-quarry clones achieved the maximum total shoot diameter (14.28 mm, September 2006) a month before the quarry clones (11.92 mm, October 2006).

Table 4.7 Significance of differences in shoot development of quarry and non-quarry clones of willow (Pen-y-ffridd experiment). Significant values shown in bold

Sample occasion	Shoot count	Shoot length	Shoot diameter
June 2006	0.3684	0.1974	0.7950
July 2006	0.3983	0.0585	0.3908
August 2006	0.2779	**0.0467**	**0.0247**
September 2006	0.9663	**0.0452**	**0.0230**
October 2006	0.7552	**0.0452**	**0.0284**
November 2006	0.6667	0.0512	**0.0324**
December 2006	0.6430	**0.0433**	**0.0123**
January 2007	0.3725	**0.0428**	**0.0047**
February 2007	0.1889	**0.0260**	**0.0004**
March 2007	**0.0426**	**0.0253**	**0.0000**
April 2007	**0.0241**	**0.0247**	**0.0000**
May 2007	**0.0032**	**0.0027**	**0.0000**
June 2007	**0.0030**	**0.0023**	**0.0000**
July 2007	**0.0095**	**0.0017**	**0.0000**
August 2007	**0.0102**	**0.0011**	**0.0000**
September 2007	**0.0097**	**0.0015**	**0.0000**
October 2007	**0.0013**	**0.0001**	**0.0000**
November 2007	**0.0017**	**0.0001**	**0.0000**

As demonstrated in table 4.7, differences between quarry and non-quarry clones were significant for much of the experiment. Mean values for shoot count were significantly different from month 10 (March 2007) until the end of experimental observations; mean values for both longest shoot length (with the exception of month 6 (November 2006, p = 0.051) and total shoot diameter were significantly different from month 3 (August 2006) until the end of observations.

Analysis of development data (i.e. presence or absence of new leaf growth, old leaf presence, and the state of budburst) gathered from Pen-y-ffridd willows during May 2007 demonstrated significant differences between quarry and non-quarry provenances (p = 0.034).

Chlorophyll content was significantly different (p = 0.002) between quarry and non-quarry clones (figure 4.4).

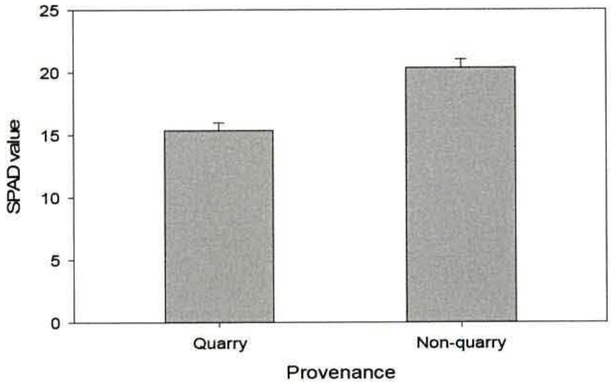


Figure 4.4 SPAD value (chlorophyll content) of quarry and non-quarry willow clones (Pen-y-ffridd experiment) (bars = standard error of the mean)

4.3.2.2 Biomass and moisture content

Biomass of non-quarry clones was greater than that of quarry clones (figure 4.5). For both fresh and dry weight, the differences were highly significant (all $p = 0.000$). No significant differences were observed in moisture content values of above ($p = 0.553$) or belowground ($p = 0.384$) biomass.

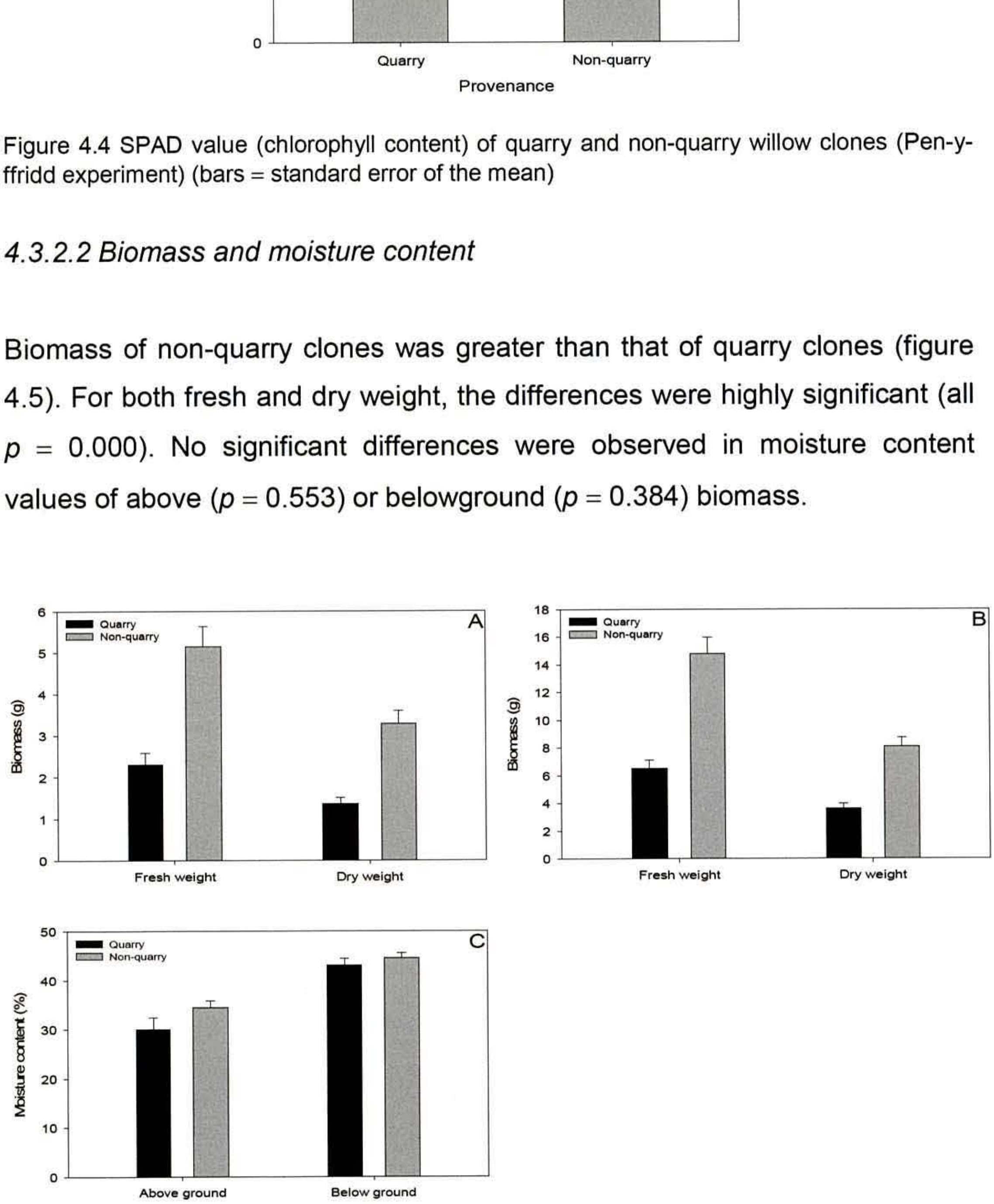


Figure 4.5 Aboveground (A) and belowground (B) biomass and mean moisture content (C) of rooted cuttings of quarry and non-quarry willow clones (Pen-y-ffridd experiment) (bars = standard error of the mean)

4.3.3 Correlations between cutting volume and growth traits

N.B. References to correlation strengths are based on Pallant's (2004) guidance that Spearman's rank order correlations of 0.1 – 0.29 indicate a weak relationship, 0.3 – 0.49 indicate a medium relationship and 0.5 – 1.0 indicate a strong relationship.

Correlations are shown in table 4.8. Above and belowground fresh and dry weight, and aboveground moisture content all showed significant weak correlations with willow cutting volume. The strongest correlation was between cutting volume and mean aboveground fresh weight ($r = 0.275$, $r^2 = 0.0756$).

Table 4.8 Significant correlations between cutting volume and three growth traits (shoot count, shoot length and shoot diameter), biomass (above ground fresh weight (AGFW) and dry weight (AGDW), and below ground fresh weight (BGFW) and dry weight (BGDW)) and moisture content (above ground moisture content (AGMC) and below ground moisture content (BGMC)) of willow clones (A- whole dataset, B- data split by clone source)

A

	June 2006	July 2006	Aug 2006	Sept 2006	Oct 2006	Nov 2006	Dec 2006	Jan 2007	Feb 2007	Mar 2007	Apr 2007	May 2007	June 2007
Shoot count	nc	nc	.162(*)	**.184(**)**	.167(*)	nc	nc	nc	nc	nc	nc	nc	nc
Shoot length	**.382(**)**	**.378(**)**	**.386(**)**	**.391(**)**	**.382(**)**	**.381(**)**	**.353(**)**	**.335(**)**	**.341(**)**	**.340(**)**	**.310(**)**	**.265(**)**	**.321(**)**
Shoot diameter	.151(*)	**.215(**)**	**.220(**)**	**.233(**)**	**.224(**)**	**.220(**)**	nc	nc	nc	nc	nc	nc	nc
Biomass	AG FW	BG FW	AG DW	BG DW	AG MC	BG MC							
	.275(**)	.166(*)	**.252(**)**	.158(*)	**.221(**)**	nc							

* Correlation is significant at 0.05 level (2-tailed)
** Correlation is significant at 0.01 level (2-tailed)
nc = no correlation
Medium strength correlation

B

	Prov.	June 2006	July 2006	Aug 2006	Sept 2006	Oct 2006	Nov 2006	Dec 2006	Jan 2007	Feb 2007	Mar 2007	Apr 2007	May 2007	June 2007
Shoot count	Q	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc	nc
	NQ	nc	nc	.215(*)	.220(*)	nc	nc	.234(*)	nc	nc	nc	nc	nc	nc
Shoot length	Q	**.337(**)**	**.412(**)**	**.418(**)**	**.427(**)**	**.428(**)**	**.401(**)**	**.329(**)**	**.325(**)**	.301(*)	**.316(**)**	**.320(**)**	nc	.252(*)
	NQ	**.384(**)**	**.379(**)**	**.420(**)**	**.430(**)**	**.424(**)**	**.417(**)**	**.519(**)**	**.481(**)**	**.499(**)**	**.504(**)**	**.395(**)**	**.408(**)**	**.451(**)**
Shoot diameter	Q		.200(*)	.237(*)	.247(*)	.247(*)	nc	nc	nc	nc	nc	nc	nc	nc
	NQ	.192(*)	.229(*)	**.276(**)**	**.295(**)**	**.279(**)**	**.295(**)**	**.361(**)**	.290(*)	nc	.240(*)	nc	nc	nc
Biomass		AG FW	BG FW	AG DW	BG DW	AG MC	BG MC							
	Q	.227(*)	nc	nc	nc	nc	nc							
	NQ	**.460(**)**	nc	**.467(**)**	nc	nc	nc							

* Correlation is significant at 0.05 level (2-tailed)
** Correlation is significant at 0.01 level (2-tailed)
nc = no correlation
Medium strength correlation
Strong correlation

Correlations between cutting volume and growth traits were significant but weak. The strongest correlation was between cutting volume and shoot count

(September 2006, $r = 0.184$, $r^2 = 0.0339$). The strongest correlation between cutting volume and total shoot diameter ($r = 0.233$, $r^2 = 0.0543$), and between cutting volume and shoot length ($r = 0.391$, $r^2 = 0.1529$) both occurred in September 2006.

When data were split by clone type (i.e. quarry or non-quarry) some stronger correlations were found (table 4.8). Quarry clones showed 17 significant correlations between cutting volume and other variables, while non-quarry clones showed 27. The strongest correlation was between cutting volume and shoot length of non-quarry clones in December 2006 ($r = 0.519$, $r^2 = 0.2694$).

4.3.4 Willow clonal tests: stage 2 (drought experiment)

No mortality was observed in any of the three differing watering regimes during this six-month drought study.

Table 4.9 Significance of differences in shoot count, shoot length and total shoot diameter between quarry and non-quarry willow clones growing under ambient, ambient −20 %, and ambient +20 % watering regimes. Significant values shown in bold

Measurement	Sample occasion	Ambient	-20%	+20%
Shoot count	July 2007	0.161	1.000	0.345
	August 2007	0.986	0.440	0.533
	September 2007	0.649	**0.023**	0.112
	October 2007	0.245	0.198	0.562
	November 2007	**0.024**	0.321	0.944
	December 2007	**0.005**	0.711	0.459
Shoot length	July 2007	0.088	0.936	0.504
	August 2007	0.202	0.246	0.310
	September 2007	0.292	0.638	0.695
	October 2007	0.120	0.667	0.803
	November 2007	0.114	1.000	0.392
	December 2007	0.088	0.810	0.240
Shoot diameter	July 2007	0.142	0.968	0.402
	August 2007	**0.022**	1.000	0.465
	September 2007	**0.031**	0.719	0.364
	October 2007	0.061	0.841	1.000
	November 2007	**0.003**	0.697	0.695
	December 2007	**0.000**	0.194	0.327

Figures 4.6 – 4.8 show that quarry clones growing in ambient watering conditions produced the greatest percentage increases in all measured variables. However, differences were only significant for shoot count during

November (p = 0.024) and December 2007 (p = 0.005), and total shoot diameter in August (p = 0.022), September (p = 0.031), November (p = 0.003) and December 2007 (p = 0.000) (table 4.9).

When exposed to increased (ambient +20 %) and decreased (ambient –20 %) watering levels the majority of observations demonstrate superiority of non-quarry sourced clones over their quarry counterparts. However, only one observation demonstrated a significant between sources difference (shoot count –20 %, September 2007) (table 4.9).

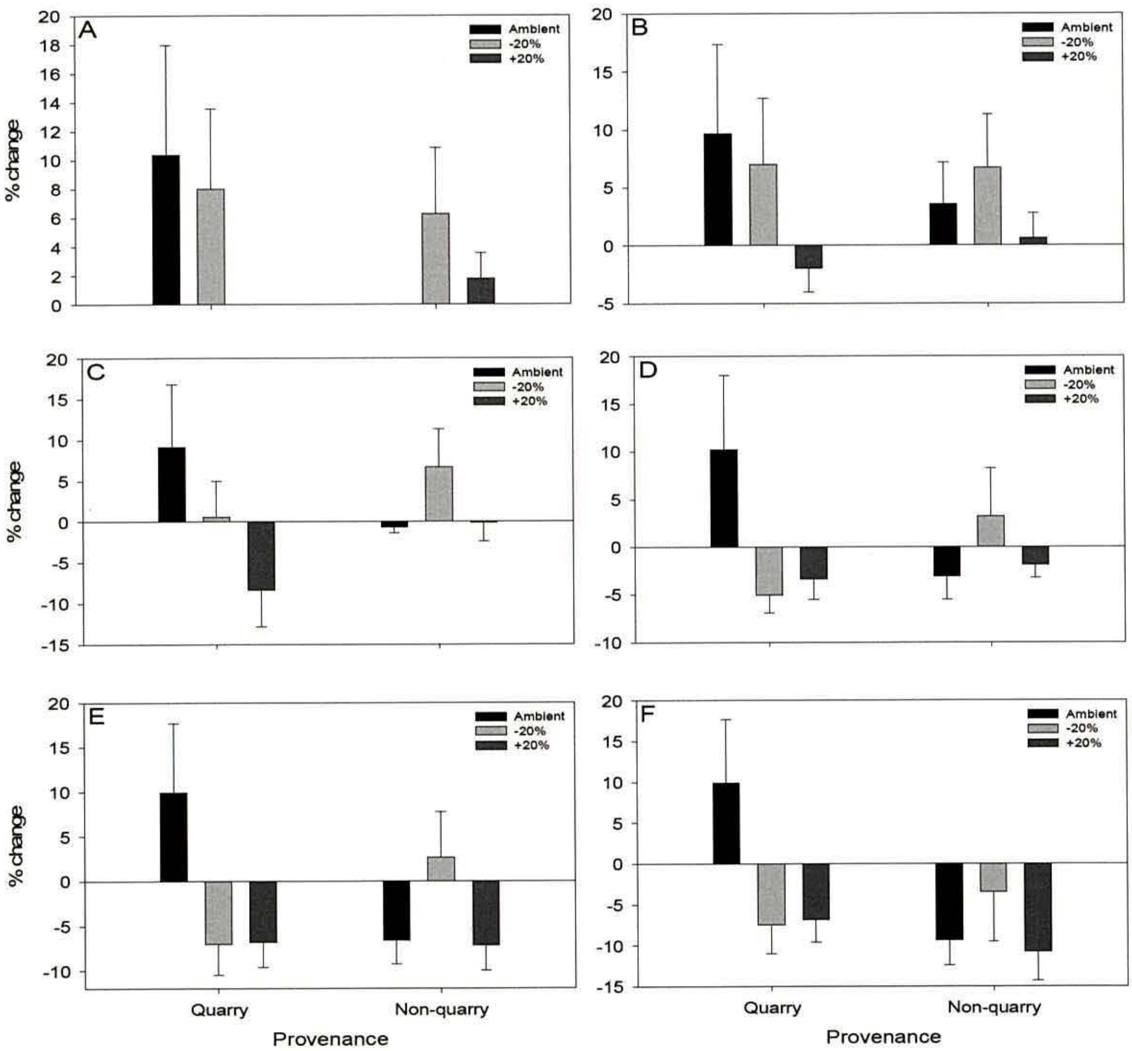


Figure 4.6 Percentage change in shoot count (A- July, B- August, C- September, D- October, E- November, F- December) of quarry and non-quarry willow clones growing under three watering regimes (bars = standard error of the mean)

N.B. No percentage changes in shoot count were recorded in July for quarry trees in the ambient +20 % watering regime or for non-quarry trees in the ambient watering regime (figure 4.6 A)

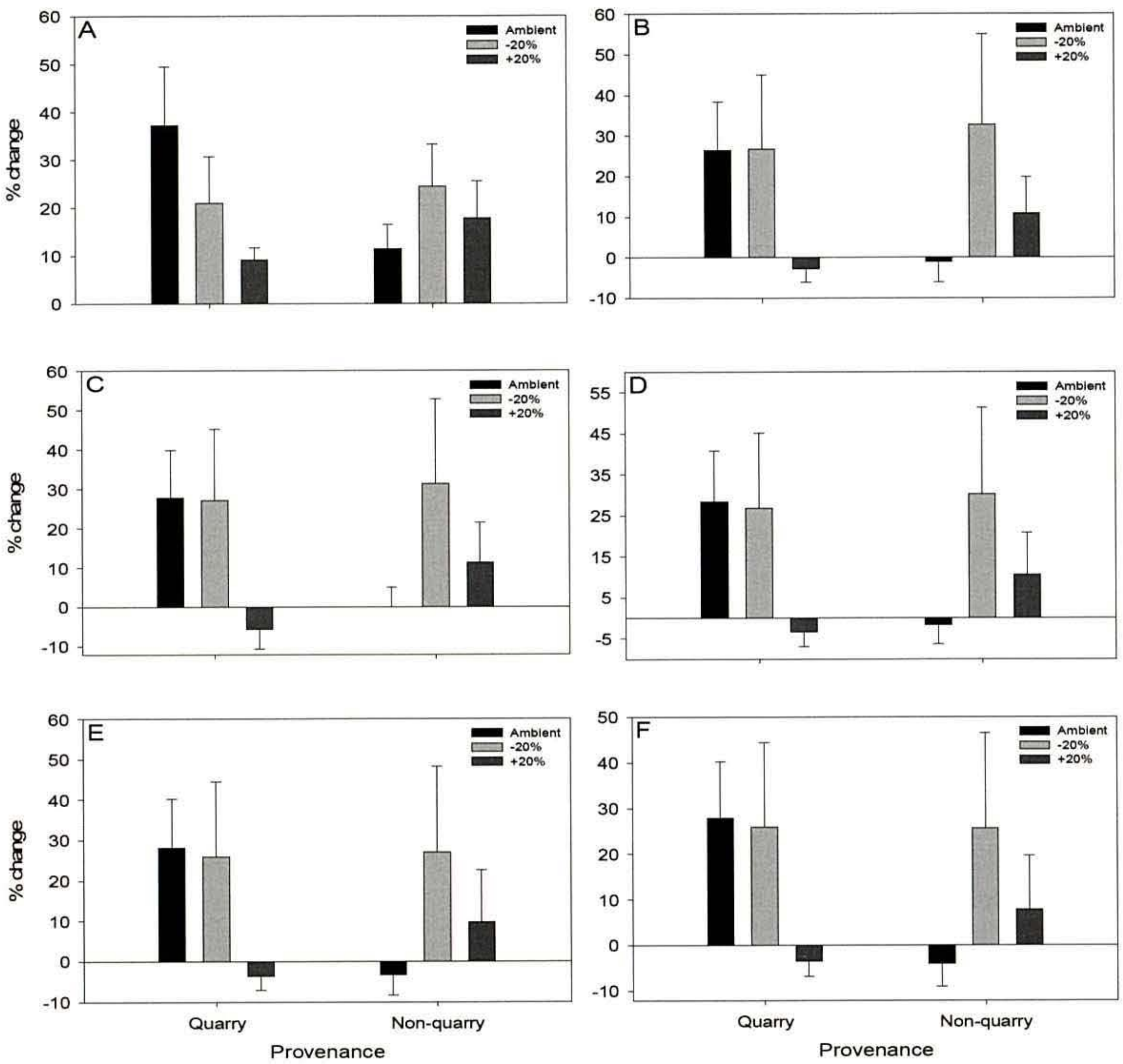


Figure 4.7 Percentage change in the length of the longest shoot (A- July, B- August, C- September, D- October, E- November, F- December) of quarry and non-quarry willow clones growing under three watering regimes (bars = standard error of the mean)

N.B. No percentage change in longest shoot length was recorded in September for non-quarry trees in the ambient watering regime (figure 4.7 C)

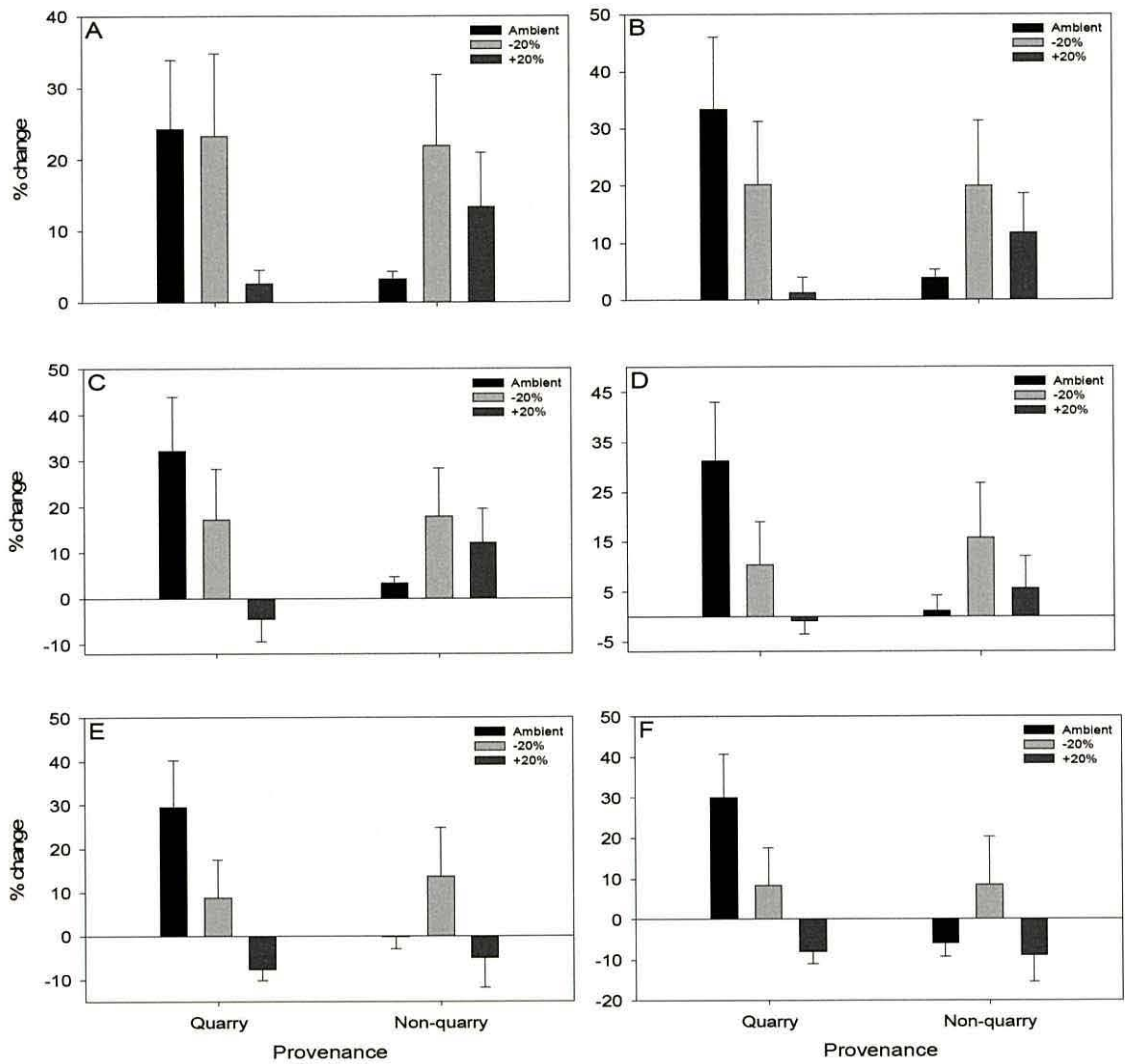


Figure 4.8 Percentage change in total shoot diameter (A- July, B- August, C- September, D- October, E- November, F- December) of quarry and non-quarry willow clones growing under three watering regimes (bars = standard error of the mean)

N.B. No percentage change in total shoot diameter was recorded in November for non-quarry trees in the ambient watering regime (figure 4.8 E)

When data from all watering regimes were combined, quarry clones performed better than non-quarry clones (figure 4.9). Differences were significant on four occasions (table 4.10).

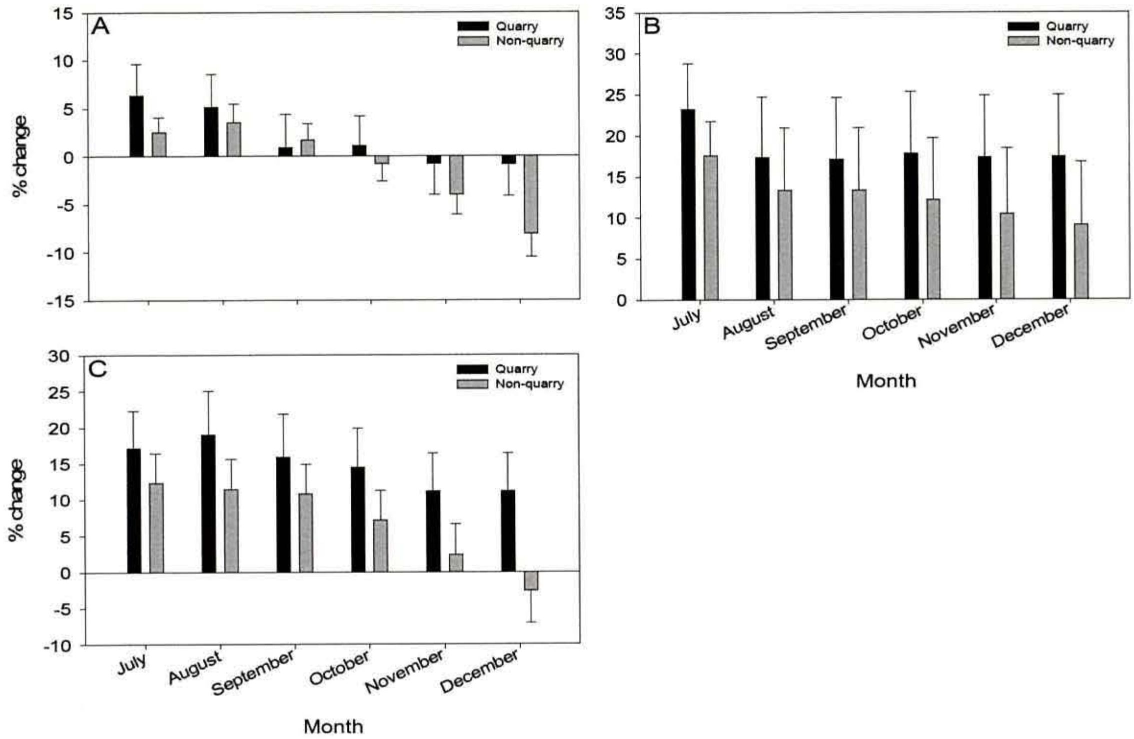


Figure 4.9 Percentage change in shoot count (A), shoot length (B) and shoot diameter (C) of quarry and non-quarry willow clones growing under three watering regimes (data combined over watering regimes) (bars = standard error of the mean)

Table 4.10 Significance of differences in shoot development between quarry and non-quarry willow clones growing under three watering regimes (data combined over watering regimes). Significant values shown in bold

Measurement	Sample occasion	p
Shoot count	July 2007	0.656
	August 2007	0.446
	September 2007	**0.033**
	October 2007	0.572
	November 2007	0.437
	December 2007	**0.027**
Shoot length	July 2007	0.360
	August 2007	0.931
	September 2007	0.702
	October 2007	0.435
	November 2007	0.143
	December 2007	0.070
Shoot diameter	July 2007	0.602
	August 2007	0.404
	September 2007	0.858
	October 2007	0.368
	November 2007	**0.040**
	December 2007	**0.000**

When data from quarry and non-quarry clones were combined there were significant differences among watering regimes in shoot length (for example December 2007; ambient = 12.257 %, ambient –20 % = 25.812 %, and ambient +20 % = 2.539 %) and total shoot diameter (for example December

2007; ambient = 12.349 %, ambient −20 % = 8.488 %, and ambient +20 % = −8.476 %), but not shoot count (tables 4.11 and 4.12).

Table 4.11 Mean percentage change in shoot development (shoot count (A), shoot length (B) and shoot diameter (C)) of quarry and non-quarry willow clones growing under three watering regimes (data combined over clones)

A

Water regime	Observation	N	Mean	Std. Error
Ambient	July 2007	57	5.263	3.895
	August 2007	57	6.667	4.258
	September 2007	57	4.269	3.959
	October 2007	57	3.684	4.183
	November 2007	57	1.796	4.272
	December 2007	57	0.422	4.376
-20%	July 2007	49	7.143	3.571
	August 2007	49	6.859	3.627
	September 2007	49	3.571	3.171
	October 2007	49	-1.006	2.710
	November 2007	49	-2.281	3.109
	December 2007	49	-5.507	3.418
+20%	July 2007	53	0.943	0.943
	August 2007	53	-0.629	1.487
	September 2007	53	-4.012	2.475
	October 2007	53	-2.597	1.243
	November 2007	53	-7.017	1.947
	December 2007	53	-8.904	2.258

B

Water regime	Observation	N	Mean	Std. Error
Ambient	July 2007	57	24.532	6.905
	August 2007	57	12.949	6.710
	September 2007	57	14.175	6.778
	October 2007	57	13.608	6.948
	November 2007	57	12.703	6.943
	December 2007	57	12.257	7.019
-20%	July 2007	49	22.688	6.523
	August 2007	49	29.705	14.171
	September 2007	49	29.181	13.867
	October 2007	49	28.506	13.737
	November 2007	49	26.446	13.878
	December 2007	49	25.812	13.752
+20%	July 2007	53	13.664	4.291
	August 2007	53	4.479	4.998
	September 2007	53	3.347	5.946
	October 2007	53	3.933	5.741
	November 2007	53	3.437	6.929
	December 2007	53	2.539	6.385

C

Water regime	Observation	N	Mean	Std. Error
Ambient	July 2007	57	13.898	5.079
	August 2007	57	18.819	6.726
	September 2007	57	18.009	6.266
	October 2007	57	16.478	6.439
	November 2007	57	14.963	5.950
	December 2007	57	12.349	6.170
-20%	July 2007	49	22.591	7.540
	August 2007	49	20.055	7.879
	September 2007	49	17.617	7.521
	October 2007	49	13.020	6.928
	November 2007	49	11.166	6.998
	December 2007	49	8.488	7.399
+20%	July 2007	53	8.237	4.153
	August 2007	53	6.730	3.898
	September 2007	53	4.309	4.774
	October 2007	53	2.593	3.561
	November 2007	53	-6.173	3.755
	December 2007	53	-8.476	3.672

Table 4.12 Significance of differences in shoot development between three watering regimes (data combined over willow clones). Significant values shown in bold

Measurement	Sample occasion	p
Shoot count	July 2007	0.282
	August 2007	0.198
	September 2007	0.271
	October 2007	0.231
	November 2007	0.400
	December 2007	0.416
Shoot length	July 2007	**0.034**
	August 2007	0.095
	September 2007	**0.020**
	October 2007	**0.049**
	November 2007	**0.024**
	December 2007	**0.043**
Shoot diameter	July 2007	**0.017**
	August 2007	**0.020**
	September 2007	**0.012**
	October 2007	**0.012**
	November 2007	**0.000**
	December 2007	**0.000**

4.3.5 Provenance trials

4.3.5.1 Germination

Germination was generally good for all test species (figure 4.10), with the notable exception of rowan, which showed almost complete failure (maximum mean germination - quarry = 0.75 %, non-quarry = 0.25 %).

Tests to examine the success of different pre-treatment methods for broom and gorse seeds showed that soaked seeds did not germinate as well as scarified seeds (broom – soaked Q = 10 %, NQ = 14.5 %, broom – scarified Q = 91.25 %, NQ = 91.5 %; gorse – soaked Q = 59.50 %, NQ = 60.75 %, gorse – scarified Q = 87.25 %, NQ = 92.75 %).

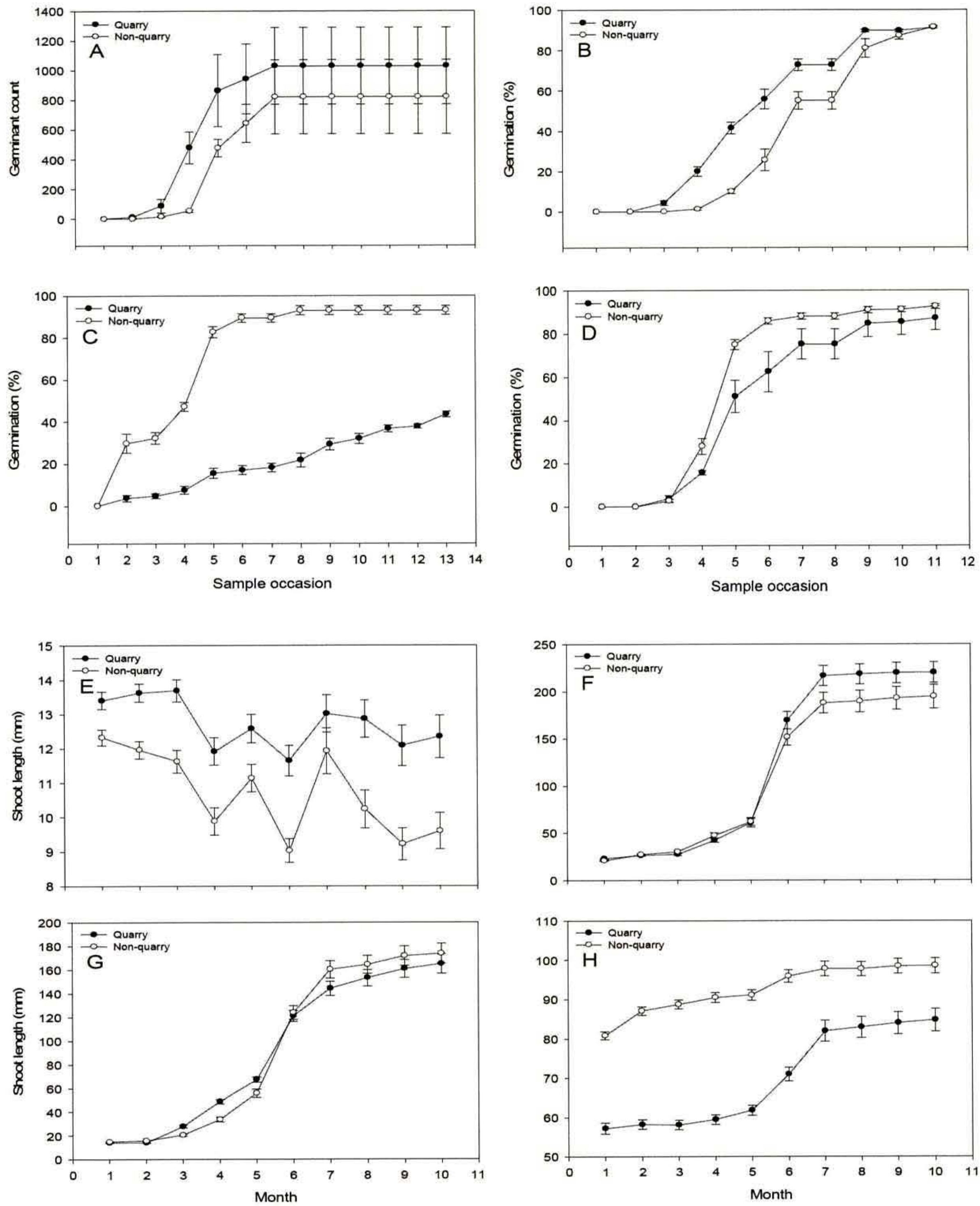


Figure 4.10 Germination of birch (A), broom (B), sycamore (C) and Gorse (D), and (total cumulative) longest shoot length of birch (E), broom (F), gorse (G) and Sycamore (H) (bars = standard error of the mean)

N.B. 4.10 E shows shoot length of birch seedlings, measured as with F- H, however, the noticeable variation apparent in this graph from the others results from delicate birch seedlings being damaged by watering, therefore the main stems often collapsed and the total cumulative longest shoot length changed monthly.

Table 4.13 Significant differences in germination between quarry and non-quarry provenances of birch, broom, gorse, sycamore and rowan seed. Significant values shown in bold; * = no germination

Sample occasion	Birch	Broom	Gorse	Sycamore	Rowan	Broom (soaked)	Gorse (soaked)
	p	p	p	p	p	p	p
1	1.0000	*	*	1.0000	1.0000	*	*
2	0.1081	*	*	**0.0202**	1.0000	*	*
3	0.1913	1.0000	1.0000	**0.0202**	1.0000	*	*
4	**0.0209**	1.0000	1.0000	**0.0209**	1.0000	1.0000	1.0000
5	0.0833	**0.0132**	0.7660	**0.0209**	0.1266	0.3173	1.0000
6	0.1489	**0.0194**	**0.0202**	**0.0209**	**0.0404**	1.0000	1.0000
7	0.1489	**0.0209**	**0.0209**	**0.0209**	0.1859	0.7389	0.0833
8	0.1489	**0.0209**	**0.0209**	**0.0202**	**0.0404**	0.8744	**0.0433**
9	0.1489	**0.0202**	**0.0433**	**0.0202**	0.1266	0.2944	0.3865
10	0.1489	**0.0202**	**0.0433**	**0.0202**	0.1266	0.2944	0.3865
11	0.1489	**0.0421**	0.5637	**0.0194**	0.1266	**0.0421**	0.7715
12	0.1489	0.2186	0.6552	**0.0194**	0.3173	0.0575	0.8845
13	0.1489	0.7645	0.5614	**0.0202**	0.3173	0.1913	0.7674

Sycamore and birch showed the biggest differences in germination between provenances. The quarry provenance of sycamore reached a maximum mean germination percentage of 43.50 %, significantly (table 4.13) lower than that of the non-quarry provenance, which reached 93 % germination. In contrast, the quarry provenance of birch germinated better than the non-quarry provenance, although the difference between provenances was only significant on one occasion (observation point 4, $p = 0.0209$, table 4.13).

4.3.5.2 Shoot growth

Results are shown in figure 4.10. The quarry provenance of birch showed significantly better shoot growth over the duration of experimental observations than the non-quarry provenance (figure 4.10, table 4.14). Maximum shoot length in quarry seedlings (13.69 mm) was achieved three months (April 2007) after sowing, whereas in non-quarry plants it peaked at 12.33 mm in the first month (February) after sowing.

Table 4.14 Significance of differences in mean longest shoot length between quarry and non-quarry provenances of birch, broom, gorse and sycamore. Significant values shown in bold

Sample occasion (all 2007)	Birch	Broom	Gorse	Sycamore
	p	p	p	p
February	**0.0104**	**0.0004**	0.0769	**0.0000**
March	**0.0000**	0.0948	**0.0004**	**0.0000**
April	**0.0000**	**0.0234**	**0.0000**	**0.0000**
May	**0.0006**	0.0570	**0.0000**	**0.0000**
June	**0.0108**	0.5375	**0.0000**	**0.0000**
July	**0.0000**	**0.0328**	0.9148	**0.0000**
August	**0.0391**	**0.0067**	0.1570	**0.0000**
September	**0.0002**	**0.0059**	0.3055	**0.0000**
October	**0.0002**	**0.0090**	0.3091	**0.0000**
November	**0.0006**	**0.0092**	0.3627	**0.0000**

In contrast, the non-quarry provenance of sycamore grew significantly more than the quarry provenance over the full experimental period (Figure 4.10, table 4.14). Both provenances achieved maximum mean longest shoot length (quarry = 84.72 mm, non-quarry = 98.53 mm) at the final observation point (November 2007), indicating sustained and continual divergence in growth throughout the duration of the whole experiment.

The quarry provenance of broom generally grew better than the non-quarry provenance (figure 4.10, table 4.14), having the greater shoot length in six of the ten monthly observations and at the final observation (Q = 219.91 mm, NQ = 194.47 mm). However, the non-quarry provenance of gorse grew better and had significantly longer shoots in eight of ten observations and at the final recording (NQ = 173.69 mm, Q = 164.96 mm).

4.3.5.3 Gorse foliage colour

There was no significant difference in colour between provenances (p = 0.056).

4.3.5.4 Biomass and moisture content

Non-quarry provenances had greater aboveground biomass in all species except broom (figure 4.11). In sycamore, differences were significant for all recorded variables (all $p = 0.0000$) except root moisture content (table 4.15). In gorse, most of the differences between provenances were not significant; however, the quarry provenance did have a significantly greater belowground biomass than the non-quarry provenance (table 4.15). Differences between quarry and non-quarry provenances of birch were not significant (see table 4.15). In broom, the quarry provenance had significantly (all $p \leq 0.0025$) greater values for every variable except aboveground moisture content (figure 4.12 B).

Table 4.15 Significance of differences in aboveground (AG) and belowground (BG) biomass and moisture content (MC) between quarry and non-quarry provenances of sycamore, broom, gorse and birch plants. Significant values shown in bold

Species	AG fresh biomass	BG fresh biomass	AG dry biomass	BG dry biomass	AG biomass MC	BG biomass MC
	p	p	p	p	p	p
Sycamore	**0.0000**	**0.0000**	**0.0000**	**0.0000**	**0.0000**	0.6514
Broom	**0.0025**	**0.0000**	**0.0001**	**0.0000**	**0.0000**	**0.0016**
Gorse	0.2232	0.2767	0.0659	0.1488	**0.0000**	0.5923
Birch	0.4658	*	0.4606	*	0.7105	*

4.3.5.5 Survival

With the exception of the non-quarry provenance of birch, survival was excellent (table 4.16). There were a total of 28 mortalities among 240 individual birch seedlings during the 11 months of this study. Most (23) mortalities occurred in the non-quarry birch provenance, and the between-provenance difference in survival of birch was significant ($p = 0.0000$). No other differences were significant.

Table 4.16 Survival of quarry and non-quarry provenances of birch, broom, gorse and sycamore from an 11-month growth study

Species	Mean survival (%)	
	Quarry	Non-quarry
Birch	99.17	80.83
Broom	98.33	100.00
Gorse	100.00	99.17
Sycamore	99.17	100.00

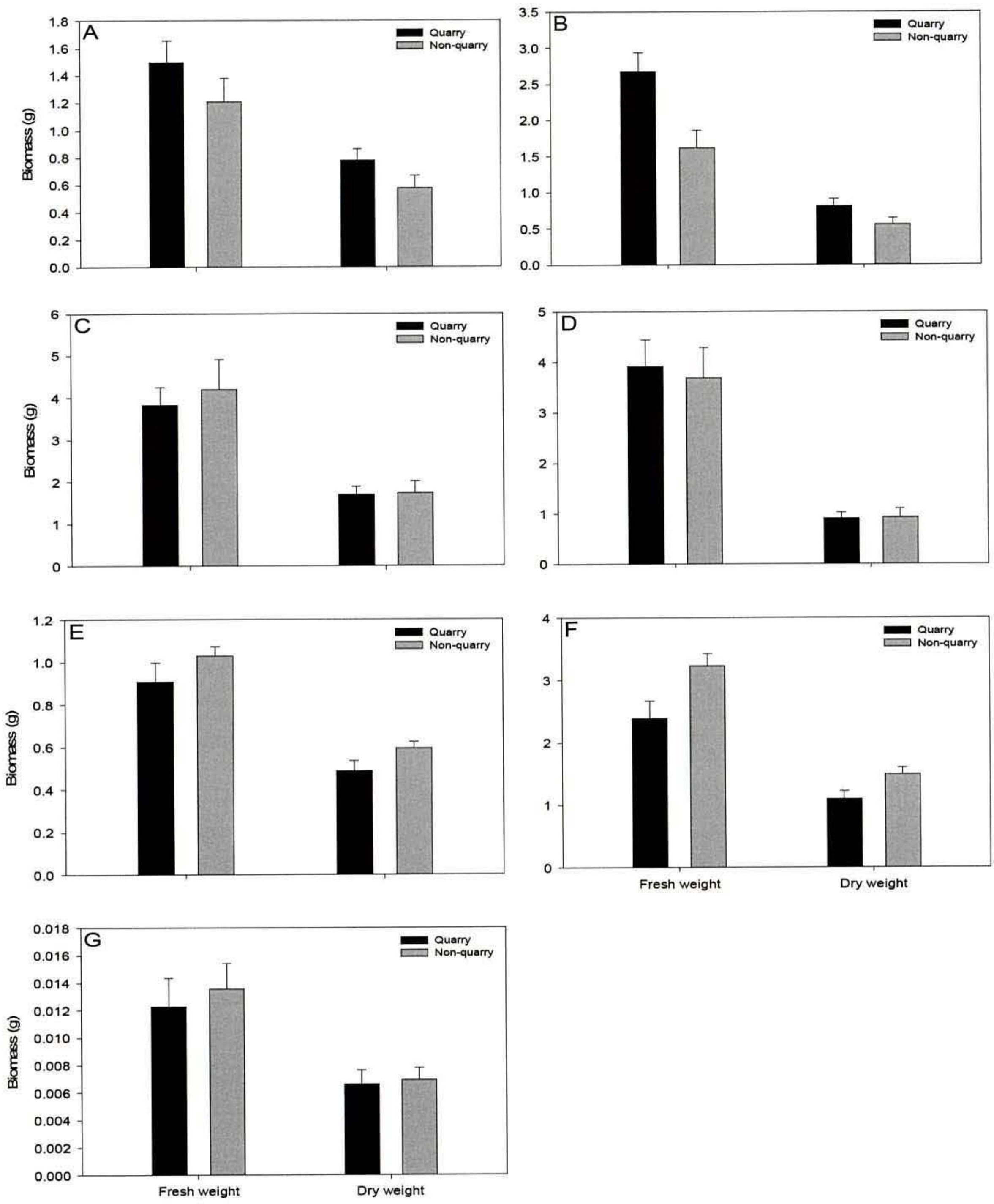


Figure 4.11 Aboveground (AG) and belowground (BG) biomass of quarry and non-quarry provenanced broom (A = AG, B = BG), gorse (C = AG, D = BG), sycamore (E = AG, F = BG) and birch (G = AG) (bars = standard error of the mean)

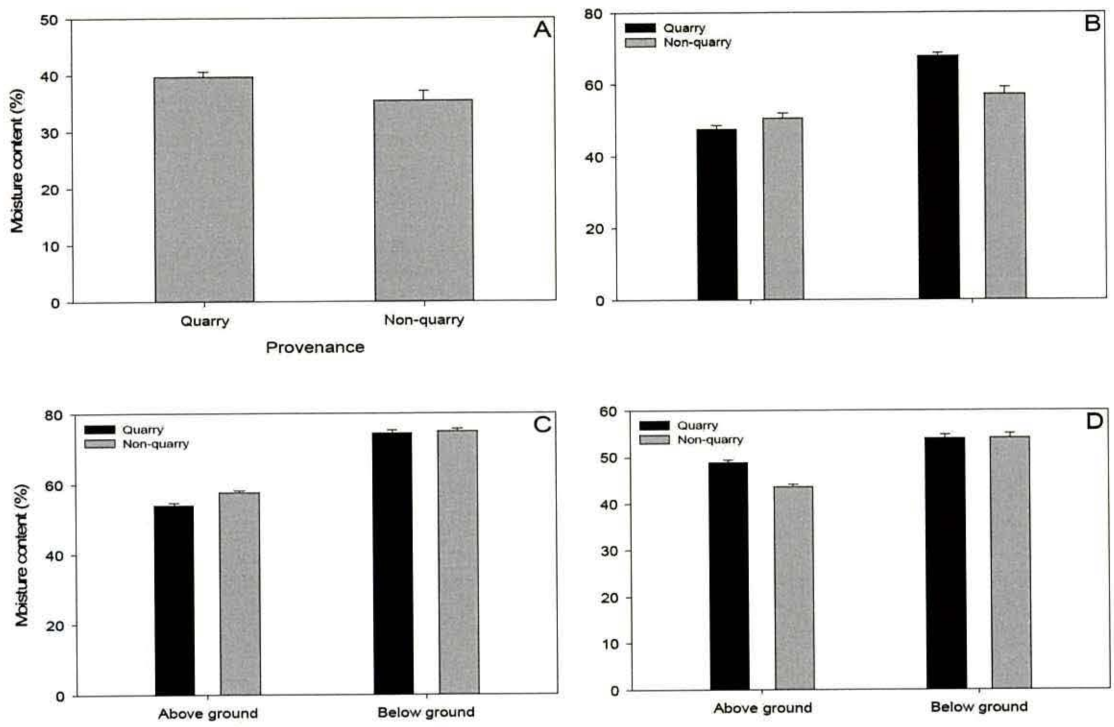


Figure 4.12 Moisture content (%) of aboveground (all species) and belowground (all species except birch) biomass of quarry and non-quarry provenances of birch (A), broom (B), gorse (C) and sycamore (D) (bars = standard error of the mean)

4.4 Discussion

In order to achieve an accurate picture of how plants survive and grow on slate waste tips at Penrhyn Quarry it was proposed that samples from all common, naturally-colonising quarry species would be collected. Much time was spent scouring the slate waste tips for vegetative material and seed from as many species as possible. Some of the seed collected had to be discounted prior to the onset of experimentation (oak, due to insufficient numbers of seeds, and beech due to a poor seed year and many non-viable seeds). However, experiments conducted with the remaining species provided some useful information on the plant populations growing in this habitat.

Survival in both quarry and non-quarry willow clones was found to be good, both exceeding 65 %. No significant between-source differences in survival were found. The observed mortalities were probably unavoidable, resulting from extreme climatic conditions, poor quality donor plants, or indeed poor vegetative samples. Renison *et al.* (2005) state that reproductive plant material derived from degraded or disturbed land is often less well developed than similar material derived from non-limiting habitats. Limited water and nutrient availability, or other stressing agents such as contamination or environmental extremes during seed set may impact upon the allocation of energy reserves and subsequent seed or cutting quality. Although the results of this experiment showed no differences in survival Smith *et al.* (2005) present results that suggest long-term survival may be better in locally-sourced plants, although they may be smaller and have lower fecundity than non-local sources. A longer period of observation than was practical in this study is needed to establish whether this is true of willow clones sourced from Penrhyn Quarry.

The development and growth of shoots was consistently better in non-quarry clones than quarry clones for the full duration (13 months) of experimental observations, invalidating both the null hypothesis (that no differences would exist between quarry and non-quarry shoot development) and the alternative hypothesis (that quarry clones would produce a greater number of shoots that were also of greater size, compared to shoot production by non-quarry clones, as a result of adaptation to localised conditions).

This finding is, however, consistent with the work of Schwanitz and Hahn (1954a and 1954b, cited in Antonovics and Bradshaw (1969)), who observed that across a range of species, plants tolerant of strong selective agents generally had smaller flowers, leaves and stems when compared to non-tolerant plants. Similarly, Antonovics and Bradshaw (1969), studying plant communities establishing on mine-site spoil heaps in north Wales, recorded significant between-population differences in plant height, leaf length and width, and culm (distance from inflorescence tip to first inter-node) length in tolerant and non-tolerant *Anthoxanthum odoratum.*

Explanations for the observed inferiority of quarry clones again centre on the probability that quarry donor plants were of overall lower quality (in terms of vegetative vigour) or health status than non-quarry plants. Smith *et al.* (2005) also found that locally-sourced plants were smaller and less productive than non-local plants. The authors point out that the initial success of non-local plants may be misleading and may not persist over long-term timeframes. Infrequent climatic events or periodic disturbances, for example, might favour local plants adapted to cope with these events. These are factors that are difficult to test experimentally. The better performance of non-quarry plants was only significant for the initial six months of the experiment, and the difference between quarry and non-quarry clones lessened with time (figure 4.1).

Records of leaf-drop during autumn/winter 2006 and budburst and leaf development during spring 2007 exhibited no differences between quarry and non-quarry clones. Bennington and McGraw (1995) report contrasting results; a population of *Impatiens pallida* growing in stressing conditions on a hillside site were found to produce flowers earlier and of smaller proportions than a non-stressed floodplain population. It was concluded that this mechanism was important for the hillside population, such that individuals could set seed prior to the onset of drought conditions.

Reporting on transplanted birch and willow establishment on opencast colliery spoil, Good *et al.* (1985) found that local (i.e. geographically local) coal-spoil provenance plants demonstrated ecotypic adaptation to conditions of low nutrient availability. These "coal-spoil" ecotypes showed reduced growth rates compared to non-local (i.e. geographically non-local), non-adapted plants. The authors suggest, therefore, that the ability to survive in severely nutrient-limited conditions may be strongly associated with slow rates of development to ensure long-term survival. Non-quarry willow clones, not being adapted to conditions found on quarry substrates, grow at a rate typical of non-limiting conditions as found in scrub-land and field margins, producing greater numbers of larger shoots. Over time this rapid growth coupled with the lack of available plant nutrients in slate waste causes a reduced rate of shoot development, as shown in figure 4.1. Initial growth would have been mostly fuelled from reserves held within the vegetative cutting. Over time this supply diminished and growth rates started to fall off, becoming dependent on the ability of the clone to survive in nutrient-poor conditions. However, as Good *et al.* (1985) state, despite the observed slow growth of locally-sourced ecotypes, sustained and long-term survival in low nutrient conditions, without the need for repeated fertilisation, is preferential to rapid growth, costly nutrient additions and ultimately, low survival. Successful restoration planting need not be rushed, as it is often assuming the role of post-glacial succession, a process occurring over thousands of years. Therefore, if locally-adapted ecotypes are chosen for restoration planting, and take a long time to become established, this should just be considered an integral part of the restoration process.

Shoots and roots harvested from each willow peg were analysed for total above and belowground biomass. As a direct consequence of having a greater mean shoot count and greater overall shoot size, non-quarry plants were found to be superior to quarry plants in both fresh and dried biomass. This pattern was repeated for root biomass. As discussed above, these findings are likely to be a reflection of how plants sourced from contrasting habitats are either adapted or non-adapted to cope in stressing conditions, such as those presented by blocky slate waste. To evaluate biomass production over time it would be necessary to conduct long-term experiments, using more cuttings and including several destructive samples at different times throughout the duration of the experiment.

Cuttings inserted at Pen-y-ffridd showed better survival than those established at Penrhyn Quarry, and there were significant differences in survival between non-quarry (95 %) and quarry (75 %) clones. This difference may be a reflection of the ability of non-quarry clones to take advantage of the freely-available water provided by regular watering, a situation rarely encountered on blocky slate waste tips. N.B. Penrhyn cuttings were watered sparingly, only providing enough water in periods of low precipitation to prevent them dying, whereas cuttings at Pen-y-ffridd were watered almost daily due to the high incidence of solar irradiation in the greenhouses in-turn causing high rates of evapo-transpiration.

Non-quarry willow clones growing at Pen-y-ffridd outperformed the quarry-sourced clones in all recorded growth variables. This again points toward the reduced growth rate of local (quarry) plants and the non-adapted rapid growth (typical of a pioneer species such as willow (Good *et al.*, 1985)) demonstrated by non-locally (non-quarry) sourced populations. These trends indicate, at the very least, that there is some form of adaptation/mechanism present within quarry clones acting to reduce or keep annual vegetative growth to a slow and controlled rate. Shoot and root biomass harvested at the end of this experiment similarly shows clear superiority of non-quarry sourced clones over those sourced from the quarry. All biomass observations were consistently and significantly greater in non-quarry clones, consistent with the superior shoot count and shoot dimensions recorded in these plants.

In contrast to the trends observed at Penrhyn Quarry, growth differences between quarry and non-quarry clones at Pen-y-ffridd increased over time (figure 4.3). This might be a reflection of the constant environmental conditions (and near optimal growth conditions) under which these clones were grown. These apparently increased the differences in growth between the two sources of willow clones, making them more apparent in the experimental timeframe than when grown in ambient environmental conditions. Antonovics and Bradshaw (1969) point out that some of the observed adaptations recorded between *Anthoxanthum odoratum* populations grown under "ideal" experimental conditions might not occur when grown under natural conditions.

Non-quarry clones had higher mean SPAD scores than quarry clones. SPAD score is a direct measure of chlorophyll content of sampled plants. Chlorophyll content is representative of a plant's health status (for instance, chlorophyll contains nitrogen, plants growing with non-limiting supplies of nitrogen will produce more chlorophyll than nitrogen-inhibited plants, and will consequently be healthier in terms of overall fitness (Purves *et al.,* 1998)), and this result, therefore suggests that non-quarry clones are healthier than quarry clones. This is likely to be a reflection of the more nutrient-rich conditions (i.e. pastural field margins etc.) the non-quarry clones were sampled from, in contrast to the nutrient-poor status of the substrates from which the quarry clones are derived. Significant differences were recorded in the timing of leaf drop, leaf generation and budburst between quarry and non-quarry clones. These observed differences might again indicate that quarry clones are adapted to grow at a reduced rate compared to non-quarry clones, therefore conserving limited resources of nutrients and water present within slate waste tips. Differences in the timing of flowering has the potential to bring about reproductive isolation (leading to ecotypic variation) between quarry and non-quarry populations, as discussed by Antonovics (2006).

Correlation analysis of cutting volume and cutting developmental data enabled identification of any relationships between initial cutting dimensions and the potential for successful establishment and development. All correlations between cutting volume and mean shoot counts, total shoot diameter and biomass were weak, accounting for $\leq$ 8.41% of the observed variation. The size of vegetative cuttings appears to have little influence upon the establishment success of these plants. All but one of the correlations between mean cutting volume and mean longest shoot length were of medium strength, but still accounted for ≤ 24.01% of the observed variation. When the data were split by source (i.e. quarry and non-quarry) several strong ($r \geq 0.50$) correlations between cutting volume and shoot length were shown by non-quarry sourced plants. This indicates that there is a closer relationship between the initial dimensions of vegetative cuttings sampled and shoot growth in non-quarry sourced plants than in quarry sourced plants.

Experiments carried out to explore the effect of drought stress on quarry and non-quarry sourced willow clones demonstrated differences in response between sources. Both increased and decreased watering rates had the greatest effects on non-quarry clones; all measures of shoot growth and development showed greater percentage changes than in quarry clones, though all but one of the differences were non-significant. The opposite was true under watering conditions mimicking ambient rainfall levels; all growth measures were superior in quarry clones, and more of the differences were significant.

With respect to these results the null hypothesis (that no difference between quarry and non-quarry trees exists in terms of their performance to increased or decreased watering regimes) is rejected, given that there are very evidently differences between the sources of willow cuttings in their ability to cope with watering regimes differing from ambient conditions. The general observed trend suggests that quarry-sourced plants are better able to respond to variable water availability conditions than non-quarry plants. This is the opposite of the findings of Steinger *et al.* (2002), who suggested that populations from fertile sites are generally better suited to respond to periods of nutrient abundance than populations from poorer, ruderal sites.

Over all clones, growth was better in ambient and ambient −20 % conditions. This is contrary to the common sense supposition that more water, i.e. ambient +20 %, would result in the greatest vegetative growth. There are several possible explanations for this result. Firstly, the increased levels of water may have a flushing effect, with any build up of organic matter or nutrients from wind blown matter or climatic sources is being flushed through the blocky, free-draining slate waste. Secondly, the high watering load may lead to localised water-logging. Because this was only a short-term experiment it is possible that the willow clones were not able to produce sufficient amounts of root aerenchyma to allow normal growth in the wet conditions, and were therefore out-performed by the clones growing under the two other watering regimes.

Smith *et al.* (2009) advocate the testing of a wide range of species when drawing up guidelines for sourcing plant material for restoration activities. Therefore, following the initial growth trials with willow clones, investigation of provenance effects was conducted with seed collected from several other species. Between-provenance differences in germination of birch, broom, gorse and sycamore seed when sown in slate-sand-filled seed trays was found to be variable, with no consistent across-species trends.

Final germination of non-quarry sycamore (93 %) was much greater than that of quarry sycamore (43.5 %). These observations reflect the characteristics of the collected seed; non-quarry seed was much larger than that collected from quarry donor plants. Other researchers (Renison *et al.,* 2005; Smith *et al.,* 2005) report that seed sourced from disturbed sites will typically be of reduced size, and therefore have fewer reserves available for germination. Keller and Kollmann (1999) highlight the potential problems resulting from different seed development processes. For instance, two samples of seed of one species, sampled from different populations and on different occasions, could undergo storage for dissimilar lengths of time; this might therefore cause variation in germination. However, all seed used in this study was collected within a short time period (see table 4.4) and therefore variation in germination cannot be accounted for by differences in storage, although development times may have differed.

Both broom and birch seed of quarry provenance germinated faster than non-quarry seed, significantly so for broom.

In broom and gorse, the differences in germination between seeds treated by scarification and by soaking are stark (see section 4.3.5.1). This suggests that best practice for restoration practitioners utilising these, or other similarly hard-coated seed species, is to scarify seed before sowing.

Rowan seed of both quarry and non-quarry provenance produced very few germinants. There are two possible reasons for this poor result: the pre-treatment process designed to break seed dormancy was not suitable or adequate; or, this species had a poor fruiting year and produced seed of low viability.

The greatest differences in between provenances shoot length were observed in birch and sycamore; in birch the quarry provenance grew significantly better (all measurements $p \leq 0.0391$) than the non-quarry provenance, while the reverse was true for sycamore, with non-quarry clones significantly (all measurements $p = 0.0000$) larger than quarry plants. Some early patterns in comparative provenance performance can be seen in these results (figure 4.10). For example, the difference in longest shoot length of birch seedlings during the final four observations grew increasingly greater, representing a steep decline in the performance of non-quarry plants. Whereas, despite the initial slow development of quarry-sourced sycamore and rapid development of non-quarry sycamore, the gap in longest shoot length was continuously closed month-by-month from May 2007 until the final observation in November 2007. Although it is not possible to know if this pattern would have continued over a longer experimental period, it appears that in both of these species continued seedling growth is more vigorous in quarry-sourced plants.

The between-provenance difference for broom was significant from July 2007 to November 2007, during which time the quarry provenance grew better than the non-quarry provenance. In gorse the only significant between-provenance differences were found from May 2007 to June 2007, when the non-quarry provenance grew better than the quarry provenance.

Substantial variation in foliage colouration was observed across the gorse plants grown in these studies. In addition to various other phenotypic differences, Aston and Bradshaw (1966) recorded colour differentiation in populations of *Agrostis stolonifera* growing along a transect ranging from a cliff edge to sheltered pastural fields. Two distinct categories were identified, "normal green" and "glaucous blue-green". However, despite the variation observed in these trials, no significant differences between provenances was found. Foliage colouration provides an indication of plant health; very pale, chlorotic plants being of poorer health than very dark green plants. However, as no provenance effect was detected, it cannot be concluded from this test that quarry plants were any healthier growing in nutrient-poor slate waste than plants from the non-quarry provenance.

Results of total biomass production were similar to results for both germination and shoot extension, with non-quarry sycamore and gorse, and quarry broom producing more biomass than the other provenance. Non-quarry birch also produced greater aboveground biomass than the quarry provenance.

Survival over the whole experimental period was very high, with both quarry and non-quarry provenances of all species, achieving $\geq$ 98 % survival, the only exception being non-quarry birch (80.83 %); only in birch was the between-provenance difference significant.

Had there been more time available to continue this research, reproductive material could have been collected from some of the experimental plants (i.e. broom and gorse). Smith *et al.* (2009) suggest that tests of seed viability represent a good marker of plant "fitness", therefore supplementing information gathered from growth, biomass and survival records.

The lack of clear evidence of ecotypic adaptation in the species studied in these experiments might be a consequence of the perennial nature of the plants or of the limited age range of the slate waste tips. For instance, the tree and shrub species used, although generally considered to be fast growing (e.g. birch and willow), take a number of years to reach sexual maturity, limiting the turnover of generations. This may have limited their potential to accumulate any mutations or adaptations capable of enhancing their survival potential on slate waste tips. The age of slate waste tips, despite being several hundred years in some parts of Penrhyn Quarry, is equivalent to a relatively low number of generations of birch, sycamore or willow, which may not be sufficient time for adaptations to appear. However, Wu *et al.* (1975) state that plant evolution can be rapid with significant changes able to occur within the space of a single generation. Research by these authors on sites heavily polluted with copper identified *Agrostis stolonifera* plants with heavy-metal tolerance establishing on sites as young as five years old, and forming almost complete cover on seventy-year-old sites. Antonovics (2006) supports these findings, reporting on the evolution of heavy-metal tolerance in *Anthoxanthum odoratum* populations growing on mine-site tailings that can be no more than 100 years old.

The selection process that all seeds go through upon being deposited on the slate waste tips at Penrhyn Quarry, regardless of the distribution mechanism, is an extremely powerful one, and only the best-adapted seeds will establish and develop into mature plants. It is possible that only a very small percentage of the total seed-rain falling onto slate waste tips ever develops beyond the seedling stage.

Ehrlich and Raven (1969, cited in Quinn, 1978) succinctly state, "that local populations will differentiate if they are subjected to different selective forces but will tend to remain similar if they are not". Aston and Bradshaw (1966) concur, adding that wherever there are sudden changes in environmental conditions it is likely that sudden changes within populations will occur. Some species might be unable to adapt to very localised and sudden environmental changes of this nature; however, others might possess the phenotypic plasticity/flexibility or genetic variability to allow their expansion into a range of local habitats. This was indeed found to be the case in the studies conducted by Aston and Bradshaw (1966) with *Agrostis stolonifera*; they concluded that this species was able to increase its range into a variety of habitats by evolving distinct and specialised populations. And Wu *et al.* (1975) surmise that some species can easily adapt to selection pressures, such as heavy-metal pollution or severe substrate conditions, while others cannot. This may explain the results from these trials.

There are, however, additional benefits arising from local/quarry seed collection, including the maintenance of local genetic integrity and limitation of negative effects, such as out-breeding depression resulting from hybridisation associated with the introduction of foreign genotypes (Hufford and Mazer, 2003; Jones, 2005).

Jones (2005) introduces a rather radical perspective into the debate over the sourcing of plants for use in restoration activities, suggesting that with artificial selection by plant breeders it is possible to select specific traits and increase resistance to stresses such as drought. It is proposed that increased stress tolerances of this type are probably desirable on severely disturbed sites. However, widespread planting of bred cultivars in this fashion would not be financially viable and would probably be rejected by regulatory authorities.

4.5 Conclusions

Seed collection for the purpose of producing planting stock to restore slate waste tips at Penrhyn Quarry currently involves sourcing desired species from seed zone 303 (west Wales). The 24 seed zones of Great Britain have been drawn to correspond with geographic and topographic regions and features, such as landforms and watersheds (Herbert *et al.*, 1999). Many different habitat types will be found within single seed zones, and using seed zones alone as a basis for seed collection ignores the potential benefits arising from very localised seed collection and habitat matching.

Results from the provenance trials reported here do not provide clear evidence of the superiority of local quarry plants nor the presence of ecotypic adaptation (table 4.17). Aston and Bradshaw (1966) state that the experimental assessment of ecological adaptation of this type is extremely difficult, while Smith *et al.* (2009) detail several studies that have failed to identify any significant intra-specific differences among populations. Bischoff *et al.* (2006a) comment that the failure to identify ecotypic adaptations during provenance trials may result from insufficient simulation of local environmental conditions. However, it is also possible that ecotypic adaptation might be expressed purely through genetic or reproductive traits and not through the vegetative growth assessed in these studies. For example, Lewontin (1974) discusses cryptic genetic change, which is not expressed phenotypically, but remains a form of divergence between populations.

Some of the species (i.e. broom and birch) tested in these short-term experiments demonstrated signs of home-site advantage. With these species, therefore, it could be argued that vegetative material or seed collection from donor plants within the confines of Penrhyn Quarry, or from other local slate waste tip habitats, could provide potential long-term benefits for the successful re-vegetation and habitat development on the severely-disturbed substrates resulting from slate quarrying. However, this certainly was not true of all species tested, and without further research, very localised collections cannot be recommended to meet these objectives.

Table 4.17 Summary of experimental results. X = provenance (quarry or non-quarry) showing the better performance

Test	Measurement	Quarry	Non-quarry
Willow boxes Penrhyn	Shoot count		X
	Shoot extension		X
	Shoot diameter		X
	Aboveground biomass FW		X
	Belowground biomass FW		X
	Aboveground biomass DW		X
	Belowground biomass DW		X
	Aboveground MC	X	
	Belowground MC		X
	Survival		X
Willow boxes Pen-y-ffridd	Shoot count		X
	Shoot extension		X
	Shoot diameter		X
	Aboveground biomass FW		X
	Belowground biomass FW		X
	Aboveground biomass DW		X
	Belowground biomass DW		X
	Aboveground MC		X
	Belowground MC		X
	Survival		X
	SPAD		X
Willow drought exp.	Shoot count (ambient)	X	
	Shoot extension (ambient)	X	
	Shoot diameter (ambient)	X	
Willow drought exp.	Shoot count (ambient -20%)		X
	Shoot extension (ambient -20%)		X
	Shoot diameter (ambient -20%)		X
Willow drought exp.	Shoot count (ambient +20%)	X	
	Shoot extension (ambient +20%)		X
	Shoot diameter (ambient +20%)		X
Willow drought exp.	Shoot count (combined watering regime)	X	
	Shoot extension (combined watering regime)	X	
	Shoot diameter (combined watering regime)	X	
Slate pots- Birch	Germination	X	
	Shoot extension	X	
	Aboveground biomass FW		X
	Belowground biomass FW	N/A	
	Aboveground biomass DW		X
	Belowground biomass DW	N/A	
	Aboveground MC	X	
	Belowground MC	N/A	
	Survival	X	
Slate pots- Broom	Germination	X	
	Shoot extension	X	
	Aboveground biomass FW	X	
	Belowground biomass FW	X	
	Aboveground biomass DW	X	
	Belowground biomass DW	X	
	Aboveground MC		X
	Belowground MC	X	
	Survival		X
Slate pots- Gorse	Germination		X
	Shoot extension		X
	Aboveground biomass FW		X
	Belowground biomass FW	X	
	Aboveground biomass DW		X
	Belowground biomass DW		X
	Aboveground MC		X
	Belowground MC		X
	Survival	X	
Slate pots- Sycamore	Germination		X
	Shoot extension		X
	Aboveground biomass FW		X
	Belowground biomass FW		X
	Aboveground biomass DW		X
	Belowground biomass DW		X
	Aboveground MC	X	
	Belowground MC		X
	Survival		X

4.6 Further work and recommendations

The obvious limitation of the experiments described in this chapter is the period over which experimental observations were carried out. The longest experiment lasted 18 months (combined duration of Penrhyn willow clone and drought experiments, and duration of Pen-y-ffridd willow clone observations). For all but annual herb and forb species this is not a long enough time to allow for anything other than establishment effects to be exhibited. Sustained observations of plant growth and survival over a period of, for example, five years is likely to yield much more information on population adaptations. Smith *et al.* (2009) concede that the information derived from short-term experiments of this nature is often limited, and state that reciprocal long-term field studies would be useful in revealing the actual degree of adaptation in local (geographical or ecological) populations.

With specific reference to the collection of vegetative cuttings of willow, the often-stunted nature of the donor plants sometimes meant that it was impossible to collect cuttings of adequate size and quality. Some of the willow cuttings used in these experiments fell outside of the recommended limits for propagation material. Future studies should sample only from donor plants capable of supplying the required number of healthy cuttings of recommended size.

Another major problem in the last of the studies outlined here (germination and growth of species in pots of slate waste) was the final number of species tested. Due to a poor oak mast from quarry trees, the failure of seed set in beech, and the failed germination of the collected rowan seed, experimental observations were limited to four species. It was intended to utilise a wider range of species to provide an extensive appreciation of species-specific adaptations to quarry conditions, but unfortunately this was not possible. Future work should test a greater range of species.

Considering the results presented here, views expressed in the literature, and of the time and resources required to identify populations expressing adaptations suitable for successfully restoring disturbed habitats, it is suggested that in particularly challenging circumstances, such as on slate waste tips, the importance of provenance might be of secondary importance. The main focus in such situations should be to establish vegetation cover of nurse crops, regardless of provenance, providing site stabilisation and conditions suitable for and capable of facilitating further plant recruitment and habitat development (addressed further in chapter 5). However, further research with a greater number of species and over a longer period of time would identify any definitive benefits, and indeed problems, associated with planting of particular provenances.

5 Material applications to post-industrial land for restorative purposes

Abstract

Water-holding capacity tests carried out on a range of super-absorbent polymers showed that granular polyacrylamide gel offered both very high water-holding ability (343.2296 g H_2O g^{-1}), and chemical and physical properties suitable for land restoration. This material was therefore included in artificial soil mixtures destined for restorative application to slate waste tips.

Bioassays of six green waste composts produced in north Wales, using four plant species (*Triticum aestivum*, *Agrostis capillaris*, *Cytisus scoparius*, and *Betula pendula*), together with chemical and physical analyses of composts, demonstrated clear differences in quality of compost among different producers. The compost produced by Conwy County Council was the most suitable for restorative application to slate waste tips.

Applications of restorative materials to slate waste tip surfaces were made during July 2006. Eight experimental treatments were applied in three randomised blocks on two sites located within Penrhyn slate quarry, Bethesda, Gwynedd, the largest slate quarry in the world (Anon, 2008a). Experimental treatments comprised green waste compost, slate processing fines, hydrated polyacrylamide gel, two-way mixtures of all materials and the three-way mixture of materials, providing seven treatments; the experimental control was the eighth treatment.

Number and frequency of germinants, species richness, total vegetation cover and biomass production were determined over an 18-month observation period. Green waste compost mixed with hydrated polyacrylamide gel was the most successful experimental treatment.

At the end of experimental observations, maximum total vegetation cover (95.2 %) was found on plots treated with green waste compost and

polyacrylamide mixtures. This was a >90 % increase over that present prior to applications of restoration materials 18 months earlier. Species richness averaged 22.3, indicating recruitment of 5.6 species over 18 months.

Significant differences (p = 0.0000) in biomass were found among all treatments. The green waste compost and polyacrylamide gel mixture was the most productive, with a dry biomass of 159.3521 g m^{-2}.

These results suggest that processed waste materials, when applied to slate waste tip surfaces, can achieve high levels of vegetation establishment and development within short time periods. Rapid development of vegetation upon denuded land resulting from industrial practice, provides conditions suitable and conducive to further plant recruitment and habitat development.

5.1 Introduction

5.1.1 Water-holding materials

5.1.1.1 Rationale

On post-industrial sites, the nature of the substrate very often dictates the potential for habitat restoration success (Holliman *et al.,* 2005). Mineral extraction and quarrying wastes exhibit many different traits, reflecting the properties of the exploited parent-material. Sand and china clay, for example, present far fewer problems of water-holding capacity than harder materials such as granite and slate. Discarded mineral tailings resulting from mining hard materials generally comprise large fraction waste particles. Depending on the management strategy employed to process waste material, considerable stratification may occur. For instance, if mineral waste is managed by tipping and piling, all small fraction material will pass to the bottom of the tip/pile, and the surface layers will therefore consist predominantly of large fraction particles. As a result, there are often very low levels of plant-available water in the upper layers of mineral extraction and quarry waste tips. Material additions that increase water-holding capacity therefore offer the potential to increase restoration success on such substrates.

5.1.1.2 Super-absorbent polymers (SAPs)

Due to the relatively limited use of SAPs in published research, and the inclusion of descriptions of specific branded products, web-based resources are frequently referenced in the following sections.

SAPs were originally developed in the 1960s (Anon, 2005a) to act as conditioning agents, providing much-needed hydration for plant establishment and development in drought-prone growth substrates or arid regions (Woodhouse *et al.,* 1991a). Early SAPs were often based on modified starch and cellulose structures, and on polymers such as poly-ethylene oxide and poly-vinyl alcohol (Elliot, 2004). Modern SAPs have been developed to withstand greater soil pressures (Anon, 2002) and absorb greater volumes of liquid, and are suitable for use in a greater range of products and applications.

5.1.1.3 Examples of SAPs

SAPs are long-chain molecules that bind water in huge volumes. Small changes in the molecular composition of these polymer chains, or in the way in which they are linked, allow SAPs to be specifically tailored to have particular characteristics (Grabosky and Basset, 1998). Such characteristics include water-binding strength, rapidity of hydration, total hydration potential and salt tolerance. Woodhouse *et al.* (1991a) state that three forms of SAPs are commonly available: starch-polyacrylonitrile graft co-polymers (starch co-polymers); vinyl alcohol-acrylic acid co-polymers (poly-vinyl alcohols); and acrylamide-alkali metal co-polymers (polyacrylamides). Because there are many possible molecular combinations, there is a great diversity of SAP brands available (Appendix 1).

5.1.1.4 SAP functionality

The chemical structure of SAPs is such that water molecules can attach to them in large numbers. The method by which water molecules attach varies among SAPs. Callaghan *et al.* (1988) describe polyacrylamide based SAPs as having cellular structures formed from a matrix of water-storing vacuoles connected by hexagonal bridges of cross-linked polyacrylamide. Gabrosky and Basset (1998) describe the reaction between water and non-hydrated SAPs as an explosion that creates a void within the molecular matrix, which allows water to be sucked in and made available to plant roots.

Cross-linking between the constituent monomers of SAPs is very important; it prevents ordinarily soluble monomers from dissolving and dispersing into the hydrating solvent, allowing water absorption, swelling, and the formation of a gel (Cook *et al.*, 1997; Kyritsis *et al.*, 1995). Cross-linkages also reduce the rate of water loss from the structure of SAPs (Callaghan *et al.*, 1988). Thus when SAPs are incorporated into planting media, water release is controlled and evapotranspiration is reduced. Water retention and recovery, and utilisation of water for plant growth and biomass production are therefore facilitated (Choudhary *et al.,* 1995).

5.1.1.5 Common uses of SAPs

Although SAPs were initially developed for improving water-holding capacity of plant growth media, their use has now greatly diversified. The single greatest use of SAPs is in disposable nappies for infants. The technology has also been adapted for use in other everyday, single-use disposable items such as adult incontinence pads, feminine hygiene products, surgical sponges, wound dressings, and even meat trays and door mats (Elliot, 2004; Cook *et al.,* 1997).

SAPs continue to be employed in horticulture as well as for agricultural and now, more commonly, landscaping and restorative purposes (Anon, 2002; Anon, 2008b). Woodhouse *et al.* (1991b) describe how SAPs are best used in horticulture, stating that they should be mixed when dry into the growing medium and then irrigated to full hydration to allow gel formation. These guidelines are echoed by Wilson *et al.* (1991) who state that polyacrylamide is best utilised by mixing with potting soil in planting holes.

5.1.1.6 Experimental trials using SAPs

Relatively limited numbers of researchers have carried out experimental trials with SAPs to evaluate their effectiveness in land application scenarios (i.e. under field conditions). Callaghan *et al.* (1988 and 1989), Woodhouse *et al.* (1991a and 1991b) and Choudhary *et al.* (1995) conducted studies focussing on the beneficial effects of SAPs on growth media properties, plant establishment and plant growth. These authors concur that the use of SAPs is advantageous for plants grown in challenging conditions such as those resulting in severe drought stress. Callaghan *et al.* (1988 and 1989) observed increased germination, growth and survival of economically-valuable trees in

Sudanese forest nurseries (open nurseries with potted tree seedlings) when polyacrylamide and poly-vinyl alcohol were used as soil conditioners. Increased water-holding capacity and decreased water loss by evaporation from amended nursery soils was also recorded (Callaghan *et al.*, 1988 and 1989; Choudhary *et al.*, 1995). Ten-day growth studies of barley sown on coarse sand mixed with SAPs showed that germination, establishment and dry weight were all higher in SAP treatments than in non-SAP controls (Woodhouse *et al.*, 1991a and 1991b).

5.1.1.7 SAP properties useful for land restoration

In terms of gross water absorption potential it is reported that SAPs can absorb into their structure between 20 and 1000 times their weight in water (Callaghan *et al.*, 1985, 1988 and 1989; Wilson *et al.*, 1991; Elliot, 2004). Combined with this huge potential water-holding capacity, effective SAPs show minimal high-tension binding of water molecules, providing reservoirs of plant-available water at soil-root interfaces (Callaghan *et al.*, 1988 and 1989).

Woodhouse *et al.* (1991b) suggest that mixing SAPs with long life spans into soils increases seed water uptake, germination rates and establishment due to SAP contact or close proximity to seed surfaces.

Cook *et al.* (1997) studied the effects of composting polyacrylamide (PAM) SAPs commonly found in municipal solid waste streams. Following 100 days of laboratory-scale composting, molecular size distribution and extractable organic fraction infrared analysis confirmed that PAM materials maintained structural integrity during composting. The composting process caused the SAP material to break down into residual high molecular weight polyacrylates. The authors suggest that this fraction has the potential to improve soil structure, whilst allowing the continuation of slow degradation without the production of any toxic residues. Thus, the recovery of SAPs from municipal waste streams and their subsequent utilisation for land reclamation is viable.

5.1.1.8 Potential for SAP use in quarry restoration

Holliman *et al.* (2005) studied the chemical and physical properties of cross-linked anionic (polyacrylamide) co-polymers to evaluate their suitability for addition to quarry residues in order to stimulate vegetation and habitat regeneration by enhancing levels of plant-available water. Studies carried out by Rowe *et al.* (2005) combined other lightweight soil-forming materials with SAP to minimize the heating and drying effects of sunlight to increase the survival and growth of trees planted into slate waste (see chapter 3).

5.1.1.9 Possible problems associated with SAP use in ecological restoration

Not all reports of SAP usage in land reclamation are positive. It is possible that high concentrations of SAP are detrimental to seed germination (Wilson *et al.*, 1991; Woodhouse *et al.*, 1991b). This may occur if SAPs absorb water away from germinating seeds, a risk if the material is not mixed fully into soils. Methods of SAP application may influence their effectiveness in land restoration practices. Callaghan *et al.* (1989) state that general surface application is inappropriate, while Woodhouse *et al.* (1991b) claim SAP materials may increase soil aridity.

Some problems may be more sinister than this. Highly complex synthetic polymer materials are made up of many constituent monomers; when these polymers break down in the environment there is potential for harmful toxic by-products to be liberated. Holliman *et al.* (2005) studied the environmental degradation of cross-linked polyacrylamide. Some microbial and physicochemical degradation resulted in the release of the toxic monomer acrylamide, a known neurotoxin and carcinogen. The European legal limit for free acrylamide (0.25 g kg^{-1}) (Holliman *et al.*, 2005) in polyacrylamide was exceeded only in newly-manufactured and field-conditioned gel when exposed to elevated temperatures (35°C). The authors established that although small amounts of acrylamide can be liberated during degradation, biodegradation by amidase enzymes produced by *Pseudomonas* species break down the monomer into non-toxic acrylic acid and ammonia within a few days. The authors therefore suggest that polyacrylamide gel is of low toxic threat, is reasonably environmentally stable and therefore suitable for land reclamation purposes.

5.1.2 Green waste compost (GWC)

5.1.2.1 Why is compost required for ecological restoration?

Much of the current research on compost production and utilisation is published on web sites; as a result there is extensive use of web-based citations in this section.

N.B. Where the term "quality" is used with reference to composts in the following passages, it reflects the general character of the material in terms of compost maturity, nutritional status, contaminant counts and physical nature. For example, a compost of "poor quality" may be nutrient deficient as a result of a poor composting process that prevented compost maturation; it may also contain many contaminants and large particles due to poor screening. In contrast, "high quality" composts are likely to be mature, nutritionally rich, contaminant free and of consistent texture, all as a result of more extensive processing.

Industrial activity often results in substantial disturbance of topsoil and subsoil (WRAP, 2003a), and indigenous soils and substrates are not often retained on the disturbed site. Denuded quarry sites, for example, therefore require substantial amendment with imported or artificial soils before vegetation can be established (Sellers *et al.*, 2001; Koolen and Rossignol, 1998; Curtis and Claassen, 2007a). Solutions for increasing the water-holding capacity of industrial waste substrates have been discussed; these substrates also suffer from severely diminished plant-available nutrient pools. There are numerous methods and options available for increasing nutrient levels on post-industrial sites; decisions on which method to employ depend upon site-specific conditions, restoration requirements, budget, and availability of materials or technologies. Curtis and Claassen (2007a), for example, state that topsoil re-application is a very effective method for facilitating re-vegetation. Despite its effectiveness, this method cannot be considered sustainable or cheap, and adequate topsoil resources are not often available. Soil amendment treatments and techniques able to provide conditions similar to those pre-disturbance, in terms, for example of root penetrability, water-holding capacity and adequate nutrient pools, are urgently required. Some post-industrial restoration projects carried out on transport embankments and abandoned quarries, for example, are increasingly using artificial soils (Koolen and Rossignol, 1998); Curtis and Claassen (2007a) describe such a project on road-cut slopes in northern California, USA. Low water-holding capacity and low plant-available nutrient concentrations are symptomatic of the barren disturbed soils on these road-cuts; they were treated with yard waste (green waste) compost mixed with decomposed granite substrates (6 %, 12 % and 24 % compost v/v with decomposed granite).

Previous restoration practices in the UK have sometimes involved the use of substantial quantities of peat (Sheldon and Bradshaw, 1975). There are many concerns regarding peat usage, primarily those of natural ecosystem destruction and the dispersal (and even loss) of rare plants, insects and birds. It could be claimed that green waste compost provides a local, natural and sustainable alternative to peat. Dawson and Probert (2007) indicate that composting initiatives have provided an alternative to peat-based growth media, in the form of green waste compost.

5.1.2.2 What is green waste compost?

The technical definition of composting is the controlled, biological decomposition and stabilisation of organic substrates derived from animal and plant sources, under conditions that are predominantly aerobic and allow the development of thermophilic temperatures as a result of biologically produced heat (Brady and Weil, 1999). The composting process results in the production of a sanitised and stabilised product that is high in humic substances and can be used as a soil improver, an ingredient in growing media, or blended to produce topsoil (MIRO, 2004; Anon, 2005b). Commonly composted materials are biodegradable municipal wastes such as paper, textiles, food and garden waste. Green waste compost is the specific fraction produced from source-segregated, plant-derived organic material, which may be composted commercially for resale as a recycled product (Dawson and Probert, 2007; Lungley, 2004).

In the UK, composting provides a method of processing organic waste material (Keeling *et al.*, 2003), and the resultant compost product has proved to be an important material for the landscaping industry (WRAP, 2003a). Substantial quantities of consistently high-quality, fertile media suitable even for agricultural use are being produced (N.B. higher quality restrictions are applicable for material applications to agricultural land). If modified to meet site-specific requirements, compost can be used as a medium for post-industrial land restoration.

5.1.2.3 Organic waste material: legislation and accreditation

Ever-increasing pressure is exerted upon the environment by mankind. Consumer-driven societies dominate most developed countries; this has many associated environmental impacts, not least the problem of waste and its disposal in the "throw away" culture driven by modern, busy lifestyles. The problem has been recognised as a major environmental issue and legislation has been put in place to make waste management efficient and sustainable (Dawson and Probert, 2007). European Union member states have agreed upon action plans to address waste management issues, producing legislation in the form of the European Landfill Directive (Council Directive 1999/31/EC on landfill of waste) (Anon, 1999). Legislative guidelines set out maximum disposal allowances on land-filling of biodegradable waste for all member states; these are intended to discourage outdated land-filling practices and encourage alternative waste management approaches. Targets set out in the Landfill Directive state that by 2010 member states must have reduced land-filled biodegradable municipal wastes to 75 % of those produced in 1995, with longer term targets of 35 % (of 1995 quantities) by 2020 (Probert *et al.*, 2005).

The Publicly Available Specification (PAS) 100 accreditation system was launched in 2002 for on-farm, community and centralised local authority compost producers. It was developed jointly between the composting association (TCA) (now known as the Association for Organics Recycling), the Waste and Resources Action Programme (WRAP) and the British Standards Institution (BSI), building upon existing TCA guidelines (WRAP, 2004). Included in the guidance notes of the PAS 100 are details of the minimum requirements for feedstock quality, composting processes, and marketing and labelling of final composted products. These details are intended to provide greater confidence in compost quality for end users. Some examples of composting process guidelines set out in the PAS 100 are:

- core temperature >55°C (in a 14-day windrow process)
- at least five turning events (during a 14-day windrow process)
- daily temperature checks (for every day of the composting process)
- absence of human pathogens in any 25 g sample
- potentially toxic element limits (e.g. lead ≤200 mg kg^{-1} dry matter)
- ≤5 weed propagules per litre sample

(WRAP, 2003b)

5.1.2.4 UK green waste resources

In 2001 a census of the UK was carried out, response data showing that there are approximately 21.7 million households in England and Wales (National Statistics, 2003). From these households approximately 30 million tonnes of domestic refuse is produced annually, of which 38 % is compostable organic matter, 21 % being green garden waste. Annual *per capita* waste production equates to approximately 105 kg of botanical waste (Anon, 2005b). Assuming the UK population to be approximately 60 million (National Statistics, 2007), it is calculated that 6.3 million tonnes of green garden waste are produced annually. In a review of green waste compost demand in the UK landscaping industry, Barnsley *et al.* (2005) claim that annual total organic matter requirements for 2007/08 would amount to 3.26-3.62 million m^3. The green waste compost fraction of this total is estimated to be 0.93 million m^3. Lungley (2004) estimates that the total annual amount of organic material used specifically for land restoration, topsoil manufacture and erosion control is approximately 270,000 m^3. Further data presented by Barnsley *et al.* (2005) estimate the annual market for green waste compost used in land restoration; this only amounts to 4000 m^3. It can therefore be assumed that green waste compost only makes up a fraction of the material used in land restoration processes.

Waste material that is not used in compost production and other sustainable re-uses is simply land-filled. Probert *el al.* (2005) state that in 2003-04 72 % of all municipal waste produced in England was disposed of via landfill. The authors do however state that compost production has increased, with a 56 % rise from 71,000 tonnes in 2002-03 to 111,000 tonnes in 2003-04. Despite this increasing trend, it is predicted that approximately 1.26 million tonnes of biodegradable municipal waste will continue to be land-filled in Wales alone in 2020 (Dawson and Probert, 2007). Breakdown of organic matter within landfills is the main source of methane production in England (Anon, 2005b). Methane (CH_4) is a greenhouse gas that remains in the atmosphere for up to 15 years; it is 20 times more effective at trapping atmospheric heat than carbon dioxide (CO_2) over a 100-year period (USEPA, 2007). Therefore, intercepting and recycling municipal biodegradable wastes instead of allowing them to enter landfills has many potential environmental benefits.

5.1.2.5 Properties and benefits of composted materials

Composted material, when produced to a high standard, has the ability to perform multiple physical, chemical and biological functions; these make it a useful material for addition to soil and application to land (Curtis and Claassen, 2007a; WRAP, 2003a). Effects such as reducing soil bulk density, improving soil structure and increasing water-holding capacity (Anon, 2005b) will have beneficial impacts upon plant growth. Composts are also acknowledged to contain high concentrations of plant-available nutrients.

Curtis and Claassen (2007a) state that yard-waste (green waste) compost has a large recalcitrant carbon content due to the lignin and cellulose-rich woody nature of the compost feedstock. The authors observed that yard-waste compost amelioration of road-cutting waste significantly increased soil carbon (by up to 2 %) and soil nitrogen (by approximately 0.2 %) in all experimental soils (6 %, 12 % and 24 % v/v yard-waste compost and decomposed granite). High carbon concentrations promote fungal growth, and provide large quantities of fungal feedstock for the first few years after compost application. Fungal-driven green waste decomposition is an important source of plant nutrient provision (Curtis and Claassen, 2007a). Compost generally contains high concentrations of plant-available macronutrients, for example phosphorus and potassium. Keeling *et al.* (2003) present data indicating the consistently better growth of wheat and oilseed rape in soil amended with green waste compost (compared to non-amended soil), including up to 40 % increases in plant fresh weight, and 18 % increases in root length. Other researchers have also noted the benefits of green waste ameliorants upon root biomass production. Seaker and Sopper (1988) observed an increase of root biomass (unspecified grass and legume species) in sludge-amended mine spoil, and Curtis and Claassen (2007a) found rooting density (in *Elymus multisetus* and *Nassella pulchura*) to increase substantially with compost addition (540 Mg (dry mass) ha^{-1} of unscreened, windrowed yard waste compost) to road cutting slopes. They suggested that this occurs because compost amendment increases water availability to plants, resulting in denser and deeper root growth. Essential macronutrients (for example calcium, magnesium and sulphur) and micronutrients (for example zinc, copper, manganese and boron) within composted materials are plant-available but present in a stable, slow-release form, so that the compost will continue to provide these nutrients for several growing seasons (WRAP, 2003a). A 50 mm surface application of typical green waste compost will provide approximately 200-400 kg ha^{-1} nitrogen, 380 kg ha^{-1} phosphate, and 1200 kg ha^{-1} potash (potassium oxide) in the year following application (WRAP, 2003a).

When used as a surface dressing or mulching amendment green waste compost offers a further range of benefits. It suppresses weed growth (Anon, 2005b), decreases overland flow by increasing infiltration rates and surface porosity, and gives soil surface protection in areas prone to severe storm disturbance. These effects can be considered especially important on sites where vegetation cover has yet to develop (Curtis and Claassen, 2007b).

Complex populations of micro-organisms constituting the biological fraction of composts are able to suppress plant pathogens such as *Phytophthora*, *Pythium*, and *Rhizoctonia* (WRAP, 2003a), further enhancing plant survival and productivity. Micro-organisms can also break down potentially toxic substances such as hydrocarbons and petrochemicals present within soils amended with green waste compost (WRAP, 2005).

5.1.2.6 Use of composts in restoration activities

Agriculture, land restoration and landfill engineering can be considered bulk outlets for poor quality, low-value composts. In landfill engineering compost is used to cover and restore landfill sites; this practice in particular probably has the lowest requirement for compost quality (Dawson and Probert, 2007). Demand for higher-quality compost products is greater in horticulture and related industries (Lungley, 2004), and where the compost is used for ecological restoration of industrially-disturbed sites.

Compost may be used as a stand-alone treatment or it may be incorporated with mineral parent materials at the site of restoration (Sellers *et al.,* 2001; WRAP, 2005). Combinations of minerals and green waste composts, for example, can be produced to form artificial soils that have similar properties to those of naturally-formed mineral or organic soils. Such artificial soils may be applied as permanent restoration substrates (Koolen and Rossignol, 1998). Generally the aim of amelioration work using applications of artificial soils should be to supplement or to introduce substrates to restoration sites to achieve a minimum of 5 % soil organic content (WRAP, 2003a).

Barnsley *et al.* (2005) reported findings of a substantial survey of the landscaping industry. Correspondence between the authors and UK coal producers such as Scottish Coal, HJ Banks, Tower, UK Coal and ATH Resources, who operate collieries and open cast mines, demonstrated the significant demand for organic matter, for example green waste compost, in large-scale brown-field reclamation projects. Indeed brown-field development is likely to become a significant part of the work of the landscaping industry. On brown-field sites soil absence, degradation or contamination are commonplace (WRAP, 2003a). Green waste compost may provide an effective low-cost product for sensitive landscaping applications on sites left by activities such as coal extraction in rural localities.

Sellers *et al.* (2001) presented results from a three-year growth study of oilseed rape and field beans sown on a landfill site in south Essex, England. Two methods, both involving a thick clay cap, were used to restore this site. One cap was pure clay; the other was supplemented with imported material. Indigenous topsoil, mineral fertiliser, sewage sludge cake and green waste compost were used as supplements. Although the experimental crops performed poorly (e.g. oilseed rape yield <0.2 t ha^{-1}) on clay amended with green waste compost, this provides another example of how composted products can be utilised for land reclamation schemes.

There are also less conventional methods of using composts for restorative purposes. Batty and Younger (2004) describe research on the use of organic materials, such as compost, in passive treatment systems for ameliorating toxic mine drainage waters. The biological and chemical properties of organic waste composts favour the development of sulfate-reducing bacteria, which facilitate acidity reduction and subsequent removal (by precipitation, for example) of elements such as iron and aluminium.

5.1.2.7 Limitations and problems of compost use

Despite all the perceived benefits of compost use, there are inevitably some potential problems and areas of concern. Variability of composting processes and consistency of the compost produced (Dawson and Probert, 2007) are significant issues and affect compost uptake for many purposes. Despite practitioners' best efforts, some contaminants will exist in the final compost product. Most large fraction contaminants are removed in a process of screening; however, some materials, glass for example, are very hard to remove with screens. The most contaminated green waste composts will probably be destined for landfill restoration projects (Anon, 2005b). Keeling *et al.* (2003) suggest that composts that fail to reach full maturity may contain phytotoxins, for example acetic acid. These authors also report that nitrogen immobilisation in immature compost may inhibit plant growth. These may have been the causes of reduced crop yields observed by Sellers *et al.* (2001), who suggest the compost utilised in growth studies was at fault.

Ward and Litterick (2004) state that the general assumption within the landscaping industry is that seasonal compost feedstock variability inevitably results in variability of compost quality. Waste materials are mainly soft, high-nitrogen wastes in the summer (May to August), woodier materials with lower amounts of macronutrients in winter and spring (January to April), and a balanced mixture during the remainder of the year. However, the authors challenge this assumption and claim that although there are clear seasonal differences in garden waste composition, variability of management practices could be as important in determining compost quality. Windrowing of organic wastes is a simple and common method of compost production in the UK, but there is great variability in the process and final product. For example: the windrow may remain uncovered for different periods during the composting process; the type of equipment used is site-specific; the number of turning events is operator-specific; water may or may not be added; various additives may or may not be utilised; the length of composting periods may fluctuate, and compost maturity may therefore not be achieved. Despite the introduction of the BSI PAS 100 certification scheme, these management differences may continue. The PAS 100 specifications focus upon issues such as contaminant, pathogen, toxin, and weed contents, more than on consistency of compost products.

Other factors also influence compost feedstock quality. Catchment area geography, such as population density or geology, may mean that adjacent composting units produce quite different quality composts at the same point in time (Krogmann, 1999). Bary *et al.* (2005), in a chemical analysis of compost from different processing sites, found variations from site to site and over time, the most significant being the among-sites variability.

Ward and Litterick (2004) report that final compost quality is strongly dependent upon the indicators used to determine whether compost has reached maturity. For instance, Rainbow and Wilson (1998) state that low pH (as a result of ammonium nitrification to nitrate) may be an indicator of compost maturity, and Brinton *et al.* (2001) claim that low electrical conductivity is also an effective indicator of maturity.

Variability of composts resulting from seasonality of feedstock could be minimised by diluting the final product with peat or coir, and/or monitoring the nitrogen content and making adjustments when necessary with nitrate or urea. Peat or coir dilution of composts may seem to contradict the ethos of production with green waste, although these materials could be sourced as waste products from other sectors, horticulture for example. However, more rigorous testing of composts, giving greater attention to physical and biological properties rather than relying wholly upon chemical data, may provide the best indicator of the quality of individual compost batches (Ward and Litterick, 2004).

5.1.2.8 Examples of green waste compost in UK post-industrial land restoration

In northwest England a partnership between WRAP and the Mersey Forest has carried out restorative works at many locations. More than 14,000 tonnes of compost has been used for cost-effective restoration of a former ordnance production site (Chorley) and a former airport (Manchester). These projects involved green waste compost incorporation into disturbed soils, followed by tree and shrub planting. Early results reported from these projects suggest successful (no figures provided) establishment of planting stock (Anon, 2005b; WRAP, 2006). Further restorative work involving WRAP was carried out at an abandoned coal mine in Kent. An ameliorative topsoil was created on site by mixing 15,000 m^3 green waste compost with indigenous shale. Subsequent establishment of willow whips and poplar saplings has reportedly been excellent (no figures provided) (WRAP, undated). Some more recent examples of restoration work using composted materials are described by a research team from Bangor University, UK. A European Commission "Life" funded project entitled "Treating wastes for restoring land sustainability (TWIRLS)" utilised in-vessel composting with a range of feedstock materials to treat a former steelworks (Flintshire, UK) and a slate quarry site (Gwynedd, UK). Feedstock materials were green waste, de-inking paper fibre (paper pulp), tertiary treated sewage sludge and slate mineral fines. Materials were mixed in various combinations and fed into the composting vessel (an EcoPod in-vessel composter) and composted for 80 days. Resultant composted materials were spread onto the brown-field sites (Nason *et al.*, 2007).

5.1.3 Methods of applying materials to land for restorative purposes

5.1.3.1 Rationale

As discussed in the preceding sections, various materials (for example, SAPs and green waste composts) can be applied to industrially disturbed land for restorative purposes. Other materials, such as pulverised fuel ash, quarry wastes, sewage sludge or wood residue, may also be used to aid soil reconstruction or management. All these materials are beneficial for their ability to fill interstitial voids or begin the formation of soil flocs and aggregates, thus reducing excess drainage, and (in rapidly draining substrates) allowing the development of a water table (Jim, 2001).

Subsequent to the formation, addition or amelioration of soils to disturbed sites, the establishment of vegetation is a critical phase of landscape rehabilitation (Jim, 2001; Paradelo *et al.*, 2007). Quarry and post-industrial wastes in general are by and large inhospitable to the establishment of vegetation. Thus techniques must be developed and employed to kick-start the process of "vegetalisation" (Merlin *et al.,* 1999). A "nurse" crop of plants can be useful in this process (Skousen and Zipper, 1996, cited in Estaun, *et al.,* 2007; Jim, 2001); nurse plants will improve soil quality and provide a platform from which plant community (and subsequently habitat) development can take place. Plants with the potential to act as nurse crops in restoration practices include grasses and legumes (Skousen and Zipper, 1996, cited in Estaun *et al.,* 2007). Grasses will develop widespread fibrous root systems, providing structure in developing soil, whereas legumes are nitrogen fixers, boosting the availability of plant nutrients (Estaun *et al.,* 2007). These early colonisation processes allow the soil-plant system to stabilise and develop, permitting subsequent introduction or reintroduction of other, including native, species and contributing to site ecological restoration (Merlin *et al.,* 1999). Brindle (2003) suggests that an efficient method of introducing a nurse crop to disturbed land is by seed projected from a hydro-seeder. The author attributes the advantages of hydro-application of seed to uniform and efficient distribution even in the challenging situations found at many post-industrial sites.

5.1.3.2 Hydro-applications of material to disturbed land

Hydro-seeding, hydro-mulching, hydro-spraying, and hydro-mechanical spreading all refer to the process of hydraulically distributing a homogenised slurry laced with plant seeds to inaccessible land by high pressure hoses or similar spraying equipment often mounted upon a truck-bed (Sarrailh and Ayrault, 2001; Cano *et al.,* 2002; Martinez-Ruiz *et al.,* 2007; Estaun *et al.,* 2007). Collectively these processes can be grouped under the term hydraulic- or hydro- application (i.e. the application by fluids or water). This technique is increasingly being utilised in ecological restoration (Estaun *et al.,* 2007) as it can overcome the problems encountered when agricultural seeding machinery is employed on steep inaccessible slopes created by mining and quarrying (Montoro *et al.,* 2000; Martinez-Ruiz *et al.,* 2007). In fact Carr and Marchant (1982) even report the successful re-vegetation of harvested and abandoned forestry land that had been hydro-seeded from a helicopter. The carrier slurry is highly variable and generally project specific, tailored for specific site characteristics or project outcomes. For example, Merlin *et al.* (1999) describe the carrier slurry as a colloid, and state that it must be designed for particular site characteristics to ensure successful re-vegetation. Constituent materials are usually solvents (generally water), tackifiers and fertilisers, into which the desired seed is mixed. Supplementary materials that may be added to hydro-seeding mixtures are biodegradable paints (Sarrailh and Ayrault, 2001) (so that areas treated can be visualised), vegetal mulches and humic acids (Montoro *et al.*, 2000), mycorrhizal inocula (Estaun *et al.*, 2007), cigarette filter tow and cotton linters (Griffith and Toy, 2001), shredded hardwood bark fines (Emanual, 1976), municipal treatment plant sewage sludge (Glazewski, 2003), and a range of fertilisers (Merlin *et al.*, 1999).

To meet site-specific requirements and project objectives additional practices may also be employed to maximise the potential for success when hydro-seeding. For instance, Bochet and Garcia-Fayos (2004) discuss bioengineering techniques such as the application of synthetic, grid-like materials to stabilise soil surfaces and improve the efficiency of hydro-seeding. Similar techniques are discussed by Muzzi *et al.* (1997), who used hydro-seeding in conjunction with jute mats and tillage.

Mitchley *et al.* (1996) report on work carried out on chalk-marl spoil produced from the Channel Tunnel workings. Nearly 4 million m^3 of spoil was produced; it has since been landscaped and now forms a 36-hectare platform (i.e. a landscaped spoil heap). This platform was successfully seeded in 1992 with *Lolium perenne* and wild flowers using a hydro-seeder, with the intention of establishing vegetation of amenity and conservation interest. Earlier work was carried out in the 1970s on the china clay wastes in the southwest of England. Allaby (1978) reports on the use of hydro-seeding to successfully reclaim waste tips within three years of abandonment. This demonstrates that the technology has been available and developing for more than three decades, and yet it is still overlooked in many situations. There is good evidence of hydro-seeding use throughout the 1980s, for example on roadside slopes in Vancouver, Canada (Carr and Ballard, 1980), on large areas of wasteland resulting from forest harvesting in north America (Carr and Marchant, 1982), on mine waste in southern Arizona, USA (Bengssen, 1985), and in habitat restoration for several species of butterfly in California, USA (Walsh, 1985).

Several authors also discuss the use of hydro-seeding for the establishment of plant cover on steep, denuded slopes resulting from human activity. For example, Bochet and Garcia-Fayos (2004) describe the reclamation of motorway slopes in the Valencia region of Spain, where hydro-applications of a mixture of herbaceous species resulted in rapid development of dense cover. Griffith and Toy (2001) describe the re-vegetation of iron ore mines in Minas Gerais state, Brazil, often using hydro-seeding methods, while Sarrailh and Ayrault (2001) state that restoration of nickel mines in New Caledonia may be reliant upon hydro-seeding as it is the most cost-efficient method of re-vegetation for the situation (the alternative being restoration planting with nursery-raised stock which the authors describe as being costly (no values specified).

Hydro-seeding can be used in conjunction with various other practices to achieve desired outcomes. These may be relatively simple (e.g. hydro-seeding followed by mulching with woodchips) (Petersen *et al.*, 2004), or more sophisticated (e.g. hydro-seeding following landform alteration, for instance on terraced slopes (Cassios, 1990), or on purposefully-created scree resulting from blasting (Wheater and Cullen, 1997; Cullen *et al.*, 1998)).

Cano *et al.* (2002) and Martinez-Ruiz *et al.* (2007) provide detailed information on hydro-application treatments, providing a useful technical insight and indicating the high cost of this practice. Cano *et al.* (2002) used a one-step application of short-fibre mulch, soluble chemical fertiliser, organic tackifier and a commercial seed mixture of grasses and herbaceous legumes at a rate of 10-35 g m^{-2}. Martinez-Ruiz *et al.* (2007) used 450 kg ha^{-1} short-fibre mulch, 300 kg ha^{-1} soluble chemical fertiliser, 400 kg ha^{-1} organic tackifier, 5000 million g^{-1} legume inoculum and 275 kg ha^{-1} commercial seed when re-vegetating uranium mine wastes in west-central Spain.

Studies of the effects of hydro-seeding on erosion report significantly reduced runoff and soil loss (Montoro *et al.,* 2000), both of which contribute to enhanced plant recruitment and establishment (Petersen *et al.,* 2004). Reporting upon restoration work carried out on uranium mine wastes in Spain, Martinez-Ruiz *et al.* (2007) found that plant cover and density were significantly increased within two years of hydroseeding with a commercial seed mixture. Floristic diversity was higher in plots treated with hydro-seeding than in non-hydro-seeded plots (no values reported). Findings from these authors are similar to those reported by Wheater and Cullen (1997), who commented that increased species richness on limestone scree probably occurred as a result of hydro-seeding. Even in highly disturbed and extreme environments, hydro-seeding has been shown to lead to vegetation establishment. Francis (1973) describes the hydro-seeding of a 200-acre area on mine tailings in Pennsylvania, USA. Despite the high alkalinity and metallic content, and complete absence of organic matter or any true soil on the site, luxuriant growth was obtained, and has continued to flourish for eight years without additional treatment.

5.1.3.3 Hydro-seeding problems and limitations

Both Sarrailh and Ayrault (2001) and Martinez-Ruiz *et al.* (2007) allude to the high costs of hydro-seeding. These authors attribute the high costs to the specialised nature of the technique, and to the fact that it is still developing. Estaun *et al.* (2007) state that hydro-seeding often results in low rates of plant establishment, and therefore recommend the use of seeds with high germination rates. Cano *et al.* (2002) and Martinez-Ruiz *et al.* (2007) identify two kinds of hydro-seeding problems, those associated with the hydro-mixtures and application processes, and those that are intrinsic to the site of application. Problems associated with the hydro-mixture include toxic effects created by fertilisers, inhibitory effects exerted by tackifiers and stabilisers, and even the quality and fecundity of the seed mix used. The rate, time of year, and prevailing weather conditions when application is carried out may have significant impacts on the outcome of hydro-seeding. Challenging intrinsic site conditions may limit the success of hydro-seeding projects; such conditions include slope angle, aspect, surface roughness and hardness. Cano *et al.* (2002) consider that site steepness is the major factor influencing plant establishment following hydro-seeding, and that technical factors relating to hydro-mixtures and spreading regimes tend to be less important than site characteristics. This view is supported by Andres *et al.* (1996), who state that hydro-seeding is not suitable for road-cut stabilisation on slopes near or above 45° (cited in Bochet and Garcia-Fayos, 2004).

Bochet and Garcia-Fayos (2004) state that in semi-arid Mediterranean Europe, long periods of drought and intense rainfall frequently cause hydro-seeding to fail, thus emphasising the importance of weather conditions at the time of spreading. Sarrailh and Ayrault (2001) identify the volume of seed that has to be included in spreading mixtures as a considerable problem, since up to 5000 seeds per square metre may be required. An observation made by Faucette *et al.* (2006), when comparing vegetation establishment and growth on land treated with either compost mulch or hydro-seeding mixtures, was that hydro-seeded plots were more likely to develop greater weed biomass. The authors recommended the use of compost mulching rather than hydro-seeding, if more rapid vegetation cover with less weed intrusion is desired. Although this may pose a problem in some circumstances, on many denuded sites some native non-invasive weed species may actually be welcomed, as they provide an important source of biomass. Cano *et al.* (2002) provide some advisory guidance on what can realistically be achieved when hydro-seeding is employed, and point out that hydro-seeding with seed of herbaceous plants on steep rock slopes is unlikely to achieve an objective of rapid establishment of dense vegetation cover. It is therefore strongly recommended that use of expensive techniques like hydro-seeding is only considered following detailed site investigations.

5.1.4 Relevance and rationale for experimental work

There is yet to be developed a method of ecologically restoring slate waste tips that can truly be considered successful, efficient, sustainable and cost effective. If islands of self-sustaining vegetation can be established upon slate waste tips, the foundations of succession will be laid.

If materials and practices can be identified that enable and facilitate vegetation establishment, even if only for a limited time period, a process similar to nurse cropping can take place. Transitional nurse crops could act to deposit organic matter, stabilise surface substrates, trap windblown matter, and change microclimatic conditions; thus, the potential for further plant recruitment, establishment and development is maximised and restoration potential is optimised.

5.1.5 Objectives

- To identify a super-absorbent polymer with water-holding capacity and physical properties suitable for application to slate waste tip surfaces to increase plant-available water and promote vegetation establishment.

- To identify a source of green waste compost produced in north Wales that has the physical and chemical properties to maximise plant growth.

- To determine whether fertiliser additions significantly increase plant productivity, and whether its inclusion in artificial soil mixtures for slate waste tip restoration is necessary.

- To identify the combination of super-absorbent polymer, green waste compost and slate sand (slate processing fines) that produces the greatest plant biomass, diversity and surface coverage on slate waste tip surfaces in field conditions with no prior site preparation.

5.1.6 Hypotheses

The following hypotheses will be tested:

- Differences in WHC of the SAPs will be recorded.
    - Different grades (i.e. particle sizes of SAPs) will have an effect on the WHC of SAPs.
    - An individual SAP type will be identified that possesses the properties most suited for use in slate waste tip restoration practices.
- GWCs produced in different counties of north Wales will vary significantly in their relative qualities for promoting plant establishment and growth.
    - A single source of GWC from those tested will be identified that possesses the chemical and physical properties most suited to the promotion of plant establishment, growth and survival, and habitat development for the ecological restoration of slate waste tips.
- A single combination of SAP and GWC will be identified (from a range of different combinations of SAP and GWC applied to slate waste tip surfaces) that significantly promotes plant development and survival over other combinations of these materials.
    - The successful combination of restoration materials (SAP and GWC), when applied to slate waste tip surfaces will effectively promote plant recruitment, establishment, development and survival, providing nurse crop cover and stabilising the slate waste tip surfaces.
        - The establishment of nurse crop cover will facilitate habitat development and lead to eventual ecological restoration of the industrially damaged landscape resulting from slate quarrying processes.

5.2 Methods

5.2.1 Water-holding materials

5.2.1.1 Water-holding capacity

Four materials were identified for possible inclusion in a soil-forming matrix. Following work by Rowe *et al.* (2005), polyacrylamide gel (PAM) was sourced from Biotechnica Services Limited (Stoke-on-Trent, UK) in two forms, granular and powdered. Sodium polyacrylate (Na-PA), a material similar to powdered PAM, was sourced from Arquest Limited (Flintshire, UK). Following dialogue with restoration practitioners at Penrhyn Quarry, a wallpaper paste (Polycell, Slough, UK) was also identified as a potentially useful water-holding material.

The technique of Keen and Raczkowski (1920) was used to determine the water-holding capacity (WHC) of these materials.

Following several pilot studies (see Appendix 2) packets were constructed from two types of fine nylon fabric mesh ($\leq$100 μm and ~200 μm mesh) (Fisher Scientific, Pennsylvania, USA). Strips (10 cm $\times$ 20 cm) of the nylon mesh were cut and folded in half; the edges were stapled together forming packets (10 cm $\times$ 10 cm). Silicone sealant was then used to seal the stapled edges.

Using the coarser (~200 μm) mesh packets for granular PAM, and the finer ($\leq$100 μm) mesh packets for powdered PAM, Na-PA and wallpaper paste, five replicate 0.1 g aliquots of each SAP were weighed into mesh packets. The packets were placed in 800 ml beakers containing 400 ml distilled water, and were left to hydrate for a period of 12 hours (sufficient to achieve constant weight). No evidence of SAP material loss from the mesh packets was observed. Following two hours of draining the hydrated SAP materials were weighed.

WHC was calculated as WHC = ((C-A) - (B-A)) / (B-A), where A is the weight of the mesh packet, B is the weight of the air-dry granular PAM and C is the weight of fully saturated granular PAM (Keen and Raczkowski, 1920).

5.2.1.2 Repeated drying effects on PAM

Based on WHC determination and documented (e.g. Holliman *et al.*, 2005; Rowe *et al.*, 2005) use of PAM in land restoration, it was decided to use only PAM in all further experiments. No wetting and drying tests were therefore carried out with Na-PA or wallpaper paste.

Wetting and drying tests were carried out to determine whether repeated cycles (of wetting and drying) caused loss of water-holding capacity. Five aliquots, each 0.1 g, of both granular PAM and powdered PAM were weighed into mesh packets (~200 μm and $\leq$100 μm mesh packets respectively); these were placed in 800 ml beakers containing 400 ml of distilled water. The PAM filled packets were allowed to hydrate for a period of 12 hours (sufficient to achieve constant weight); after this the distilled water was poured off, the hydrated PAM packets drained for two hours and weights of the saturated PAM were recorded. The hydrated PAM from each packet was then emptied into a labelled double paper bag and placed into a drying oven set at 80°C for 48 hours (sufficient to achieve constant weight), after which dry weight was recorded. The saturation and drying process was repeated twice.

Loss of WHC was calculated as a percentage change of the previous WHC value, determined as explained in section 5.2.1.1. For example, the change in WHC from wetting-drying cycle 1 (WHC1) to wetting-drying cycle 2 (WHC2) was calculated as 100 $\times$ (WHC1-WHC2) / WHC1.

5.2.2 Green waste composts

5.2.2.1 Triticum aestivum bioassay

Five local green waste composts (GWCs) (see table 5.1) and three commercially-available growing media (peat, topsoil, and neem coir) were compared in a 10-day bioassay using *Triticum aestivum* (wheat).

Table 5.1 Source of composts and other growing media used for bioassay studies

GWC / growing medium	Source
Anglesey (GWC)	Mr Gwyn Maple, Gaerwen
Conwy (GWC)	Conwy County Council, Dolgarrog
Denbighshire (GWC)	Rhug Organic Estate, Corwen
Flintshire (GWC)	Flintshire County Council, Shotton
Wrexham (GWC)	R. A. & C. E. Platt Ltd., Llay
Topsoil	B&Q plc., multipurpose topsoil
Peat	Humax growman multipurpose sphagnum mixture
Neem coir	O. G. P. Ltd., UK

Two litres of compost/growing medium were placed in a seed tray measuring 35 cm × 21 cm (internal dimensions), and 25 *Triticum aestivum* seeds sown in the filled tray. Four trays of each compost/growing medium were prepared and all seed trays (32) were laid out in completely randomised arrangement on a single bench in a temperate greenhouse (16-hour day at 18°C, 8-hour night at 15°C). Germinant counts and measurement of seedling height were made daily for ten consecutive days in September 2005. On the tenth day aboveground biomass was harvested; individual plants were placed into labelled paper bags, fresh weights (FW) were determined and samples were placed in a 65°C drying oven for 48 hours. Dry biomass was removed from the paper bags and dry weights (DW) were determined. Moisture content (MC) was calculated as MC = 100 × (FW-DW) / FW.

5.2.2.2 Restoration species bioassay

N.B. In addition to the five GWCs tested in section 5.2.2.1, an additional source was used for this experiment. The sixth GWC was sourced from Gwynedd and was supplied by Mr. Harri Parry of Pwllheli. Topsoil sourced from B&Q plc., as used in section 5.2.2.1, was excluded from this test.

These bioassays used species considered to be relevant to potential restoration activities at Penrhyn Quarry: *Agrostis capillaris*, *Cytisus scoparius* and *Betula pendula*. *Cytisus scoparius* and *Betula pendula* seed was collected from Newborough Forest, Anglesey (NGR SH41325,64450). *Agrostis capillaris* seed was sourced from the Snowdonia National Park, and supplied by Emorsgate Wild Seeds (Norfolk, UK). Prior to sowing, the seeds of all species were treated (table 5.2) following the recommendations of the International Seed Testing Association (ISTA, 1996); *Betula pendula* seed was also sown untreated. Half-sized (internal measurements 16 cm $\times$ 20 cm) seed trays were filled with one litre of compost/growing medium, into which seeds were sown. For *Cytisus scoparius* and *Agrostis capillaris*, 25 seeds were counted and sown. *Betula pendula* seeds were weighed into 0.1 g aliquots (c. 2000 seeds per gram fresh seed, therefore c. 200 seeds per tray) and sown on to the surface of the compost. Coarse washed sand was sieved onto the surface of seed trays containing *Agrostis capillaris* and *Betula pendula* seeds, to minimise seed disturbance by watering. The bioassay lasted two months (December 2005 – January 2006); during this period regular (daily for the first 33 days) counts of germination and survival were made. A count of weed plant germination was also made; weed seedlings were removed after counting. Shoot length of individual test plants was measured using a standard tape measure, and biomass chlorophyll content of two plants from each replicate for all test species were recorded using a Minolta SPAD-502 chlorophyll meter (Konica Minolta, Tokyo, Japan) prior to harvesting (13th January 2006). On day 66 (16th January 2006) all above-ground biomass was harvested, placed into labelled paper bags and weighed to record fresh weight. Following drying in a 65°C oven for 72 hours biomass was removed from the paper bags and re-weighed for dry weight determination and moisture content calculations (see section 5.2.2.1 for calculation).

Table 5.2 Pre-sowing seed treatment processes prescribed by the ISTA

Species	Treatment
Agrostis capillaris	Soak with 2 % potassium nitrate (KNO_3) solution
Betula pendula	21-days refrigeration (4°C)
Cytisus scoparius	Scarification of hard seed cases

This experiment was carried out in a greenhouse providing the same day-length and temperature conditions as for the *Triticum aestivum* bioassay (section 5.2.2.1). The experiment used a split plot design, with four blocks, eight main plots (composts) and four sub-plots (pre-treated *Agrostis capillaris*, pre-treated *Cytisus scoparius*, pre-treated *Betula pendula*, untreated *Betula pendula*).

5.2.2.3 Physical and chemical analysis

All GWCs used in this experiment were of the finest grade, as a result of being passed through a series of screens. There were, however, marked differences in particle size among and within composts. The particle size distribution of composts was determined by taking three 100 g samples per compost, weighing them and passing them through a series of ten soil-testing sieves (10 mm, 9 mm, 5 mm, 2 mm, 1 mm, 355 μm, 211 μm, 180 μm, 152 μm, and 106 μm). The 11 fractions produced were weighed.

Samples (approximately 10 g) of each GWC were weighed (FW) into labelled crucibles, oven-dried at 80°C for 36 hours and re-weighed (DW). Loss on ignition (LOI) analysis was then determined by placing the dried and weighed GWC samples into a 450°C muffle furnace for a period of 12 hours. The ashed GWC weights (ADW) were recorded, allowing calculation of LOI as $100 \times (DW\text{-}ADW) / DW$.

GWC pH and electrical conductivity (EC) were determined on water extracts (25 g fresh compost in 25 ml distilled water, stirred at 250 rpm on a reciprocating shaker for 30 minutes) using a Hanna Instruments pH 209 (Hanna Instruments Ltd., Leighton Buzzard, UK) pH meter and Jenway 4010 (Bibby scientific T/As Jenway, Essex, England) conductivity meter respectively.

Further water extracts were prepared by adding 20 g fresh GWC to 100 ml distilled water and agitating for an hour at 250 rpm on a reciprocating shaker to ensure thorough extraction of water soluble nutrients. Water extracts were then centrifuged at 8000 rpm and 5°C for 8 minutes; the resultant supernatant was filtered through 90 mm Whatman no. 41 filter papers into 20 ml sample pots, and subsequently frozen prior to chemical analysis.

After thawing, water extracts of GWCs were analysed colorimetrically for water-soluble ammonium (Mulvaney, 1996), nitrate (Downes, 1978) and phosphate (Murphy and Riley, 1962) using the microplate technique and measuring light absorbance at 667, 540, and 820 nm respectively by spectrophotometry in a Biotek PowerWave XS (BioTek Instruments, Inc., Vermont, USA). Samples were analysed against standard curves produced from standards of known concentration. Further analysis of these water extracts was carried out using a Sherwood flame photometer 410 (Sherwood Scientific, Cambridge, UK) to determine concentrations of water-soluble total sodium, potassium and calcium (Hald, 1947).

5.2.3 Soil-forming mixtures: greenhouse fertiliser trial

This trial compared the development of vegetation grown in soil-forming mixtures and their three constituent materials: granular PAM (Biotechnica Limited); grade 1 (10 mm screened) green waste compost (GWC), sourced from Conwy County Council, and 0 – 4 mm slate processing fines (slate sand), from Penrhyn Quarry.

Rowe *et al.* (2005) provides a detailed description of slate sand. Some noteworthy qualities are: texture equivalent to sandy loam; organic carbon content of 0.2 %; total nitrogen content of <0.01 %; and Olsen P phosphate of 2 mg kg^{-1}.

The eight treatments are listed below.

PAM
GWC
Slate sand
GWC + PAM
PAM + slate sand
GWC + slate sand
GWC + PAM + slate sand
Control (see below)

N.B. Slate sand is hereafter referred to as slate.

The control treatment comprised processed slate waste "slate-lets", which are roughly circular discs of slate measuring up to 100 mm in diameter. This was intended to represent conditions similar to those found on slate waste tip surfaces.

Mixtures of materials were made up as equal volume combinations, i.e. two constituent materials = 50:50 v/v, three constituent materials = 33:33:33 v/v.

Mixtures/constituents were placed into half-sized seed trays (internal measurements 16 cm × 20 cm). One litre of material was placed in each seed tray, providing a substrate depth of approximately 5 cm. Six seed trays of each mixture/constituent were set up; three received inorganic mineral fertiliser, and three received no fertiliser.

British Seed Houses Limited (Lincoln, UK) (BSH) recommended the use of fertiliser in conjunction with the restoration seed mix (see below). The specified fertiliser was a low nitrogen (6:9:6 ratio N:P:K) inorganic mineral granular form to be applied at a rate of 20 g m^{-2} (equivalent to 200 kg ha^{-1}).

Thus 0.64 g of fertiliser was required per seed tray. Individual particles of granular fertiliser weighed between 0.03 g and 0.05 g, and it was therefore necessary to grind the granular fertiliser with a pestle and mortar before weighing and applying it to the seed trays.

A seed mix specifically formulated to meet restoration requirements for Penrhyn Quarry was produced by the BSH, largely based upon a standard seed mix (see Appendix 3).

The seed mix used for this greenhouse trial and the field trial at Penrhyn Quarry, is given below.

35 % *Festuca rubra* ssp. *rubra* variety "Adinda" (strong creeping red fescue)
10 % *Lolium perenne* variety "Talgo" (perennial ryegrass)
20 % *Poa compressa* variety "Reubens" (flattened meadow grass)
10 % *Festuca rubra* ssp. *commutata* variety "Olivia" (chewings fescue)
7.5 % *Agrostis castellana* variety "Highland" (browntop bent)
2.5 % *Trifolium repens* variety "Aberace" (small leaved white clover)
5 % *Teucrium scorodonia* (wood sage)
2.5 % *Ulex europaeus* (western gorse)
2.5 % *Cytisus scoparius* (broom)
5 % *Digitalis purpurea* (foxglove)

The recommendation of BSH technical advisors was that this seed mix should be applied at a rate of 5 g m^{-2} (equivalent to 50 kg ha^{-1}) in contrast to the usual application rate (25 g m^{-2}), to reduce the chances of developing a monoculture of dominant grass species. Woody shrub species were included at a rate of approximately 50 seeds m^{-2}; this was intended to allow approximately 5 plants m^{-2} to develop, assuming 10 % survival. 0.16 g of the seed mix was sown in each tray.

Seed trays were positioned on a single bench in a temperate greenhouse (16-hour day at 18°C and 8-hour night at 15°C). The experiment was set up using a randomised block design, with three blocks and sixteen treatments (eight growth media, with and without fertiliser). Germinant counts were made daily for seven days, at which point the most successful trays were too densely populated to differentiate individual seedlings. Separate records were made of the numbers of dicotyledonous and monocotyledonous seedlings.

After two months (June, 2006) all above-ground biomass was harvested. Total biomass for each seed tray was weighed, placed into labelled paper bags and oven-dried for 72 hours (achieving constant weight) at 65°C. All biomass was then removed from the labelled bags and re-weighed to determine dry weight. Moisture content was calculated as described in section 5.2.2.1.

5.2.4 Soil-forming mixtures: quarry trial

5.2.4.1 Experimental sites

Sites had to have good vehicular access, ideally above and below the slope, be in a part of the quarry that would not be disturbed for at least 18 months after the establishment of the experiment, and be safe to work on. Two sites providing conditions representative of Penrhyn Quarry (as a whole) were located that satisfied most of these prerequisites.

The first (upper) site (referred to as site 1) was located in the eastern fringe of the quarry complex at NGR SH624,645, 333 m. a. s. l., with an aspect of 111° (~ESE), and a slope of approximately 45°. The slate waste was composed of a range of size fractions, from fine crushed slate to boulders of slate and igneous rock greater than 1 m in diameter. There were some well-established plants on site, for example several mature (5 - 10 year old) individuals of *Ulex* *europaeus, Acer pseudoplatanus* and *Pinus sylvestris*, and a sparse (<5 %) ground cover of grasses and forbs such as *Digitalis purpurea* and *Teucrium scorodonia*. Access both above and below the site slope was good as it was located at the apex of a hairpin bend on a haulage road. Plate 7 shows part of site 1.

The second (lower) site (referred to as site 2) was located at NGR SH625,652, 189 m. a. s. l., with an aspect of 322° (~NW) and a slope of approximately 45°. The surface substrate was composed predominantly of large (≥30 cm diameter) to very large (≥50 cm diameter) blocky slate waste material with frequent voids throughout the matrix. The site was very sparsely vegetated (<1 % surface cover) with few plants present on site at all, apart from several reasonably well-established *Salix caprea* individuals, and several small patches of *Teucrium scorodonia*. Access to this site was poor; it was located on a quarry gallery with a vertical drop at one end and a considerable amount of vertical over-burden slate waste at the top of the slope. Plate 8 shows a complete block at site 2.

Plate 7 Experimental plots being marked out at site 1 (May 2006)

Plate 8 Experimental plots at site 2 (June 2006)

5.2.4.2 Experimental layout and baseline survey

Experimental sites were measured out roughly using a tape measure to give an indication of where the first plot should be positioned. Plots were then marked onto the slate waste tip using line-marker spray paint in May 2006. A weighted guideline with calibration marks was used to mark all experimental plots and walkways. The plots were positioned to incorporate the maximum possible amount of slate waste slope, and were sized according to the amount of slope space available on each site. Most experimental plots were 1.5 m wide by 10 m long; two plots at site 1 were approximately 2.5 m $\times$ 6 m due to the site dimensions. A randomised block design with three blocks and eight treatments was used; treatments were the same as those used in the greenhouse trial (see section 5.2.3), with fertiliser added to all treatments. Figure 5.1 shows the layout of experiments at sites 1 and 2. Following consultation with restoration practitioners, an additional plot covered with straw (seeded and fertilised at the same rate as all other treatments) was added to each site. Therefore, the experimental dimensions were: plots = 15 m^2, blocks = 120 m^2; sites = 375 m^2 (including one straw plot/site); whole experiment (both sites) = 750 m^2.

Once both sites had been marked out, a survey of baseline conditions was carried out. Visual assessments were made to record percentage vegetation cover and number of plant species in each whole plot. The size of each established woody plant (height and basal stem diameter) occurring on each site, and invertebrate presence (two pitfall traps (non-baited, containing only saline solution to preserve trapped organisms until traps removed) per plot) were recorded. Soil samples were collected from 0 – 10 cm depth at three random points within every plot and were then combined to form composite samples of approximately 500 g for each plot (samples collected during June 2006).

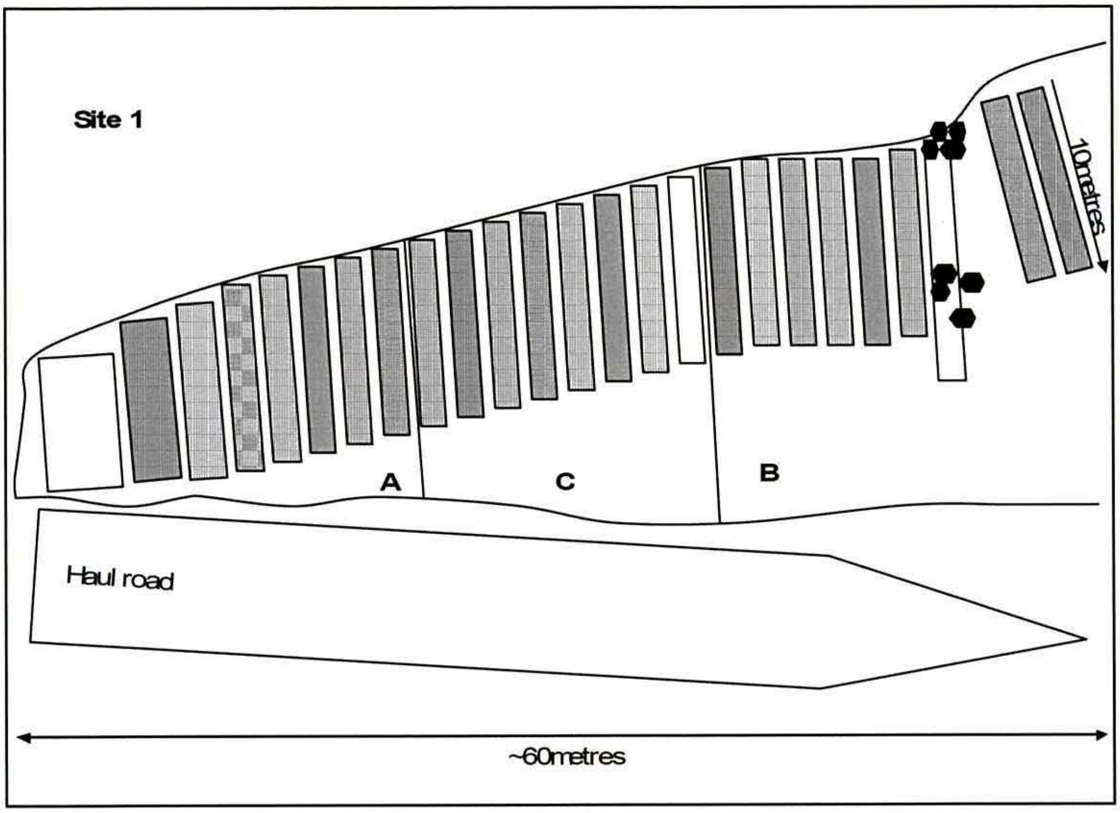
5.2.4.3 Application of treatments

All soil-forming mixtures and constituent materials were spread manually following the method described below.

1. A processing area was set up in the operational part of the quarry; this allowed all materials to be stockpiled in one area, and access to power and water supplies.

2. The restoration seed mixture was weighed into 50 $\times$ 75 g aliquots and the granular fertiliser into 50 $\times$ 300 g aliquots, based on recommendations for sowing and fertilising rates made by BSH advisors (5 g m^{-2} seed and 20 g m^{-2} fertiliser). The aliquots of seed and fertiliser were weighed into paper bags for later addition to spreading mixes.

3. PAM was applied to soil-forming mixes in a fully hydrated form; pre-mixing hydration was therefore required. This was done by adding granular PAM (approximately 3 kg anhydrous) to a plastic tank (40 gallon volume) filled with water and leaving it to hydrate for approximately 30 minutes.



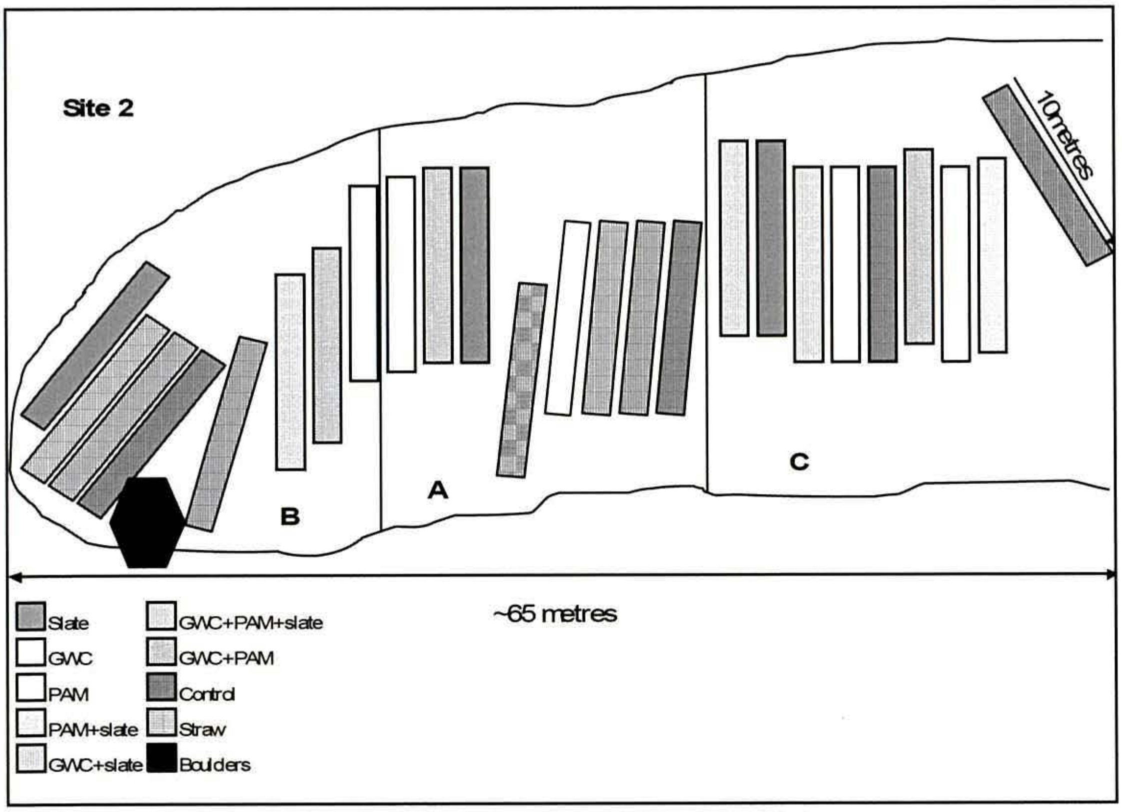


Figure 5.1 Experimental layout at site 1 and site 2

4. A 90-litre concrete mixer was used for processing. Thirteen concrete mixer drums full of mixture equated to 1170 litres and the required amount of mixture per experimental plot was 1125 litres; the drum, therefore, was not filled completely. Constituent materials were shovelled into the concrete mixer in even quantities. For example GWC + PAM + slate mixes required ten shovels GWC, ten shovels slate, and five small buckets of hydrated PAM per drum load.

5. Seed and fertiliser were added in small handfuls to each drum full of mixture until the full allocated seed or fertiliser aliquot was mixed, ensuring uniform distribution throughout the whole of each mixture.

6. Mixtures were processed in individual batches; for example GWC for plot 3, block 1, site 1, was mixed in its entirety at the processing area. Each drum load of mixture was emptied into a set of 40 litre buckets that were then emptied into the loading area of a 4×4 pick-up vehicle. Mixtures were then transported to the experimental sites for spreading. The 4×4 pick-up vehicle was positioned as close as possible to the recipient experimental plot. Mixtures were then shovelled into 40 litre buckets, carried and emptied onto experimental plots; 35 – 45 buckets per plot were required to give a 75 mm depth of soil-forming mixture.

For the straw treatment, buckets of straw were emptied on to the experimental plot area, over which seed and fertiliser had been distributed by hand. Once straw had been positioned and seed and fertiliser applied, a roll of hessian netting was placed over the plot and weighted down with blocks of slate, securing the straw in place (pilot tests with loose straw are described in Appendix 4).

Control plots received no soil-forming mixture applications, but the restoration seed mixture and granular fertiliser were distributed by hand over these plots at the same rates as for other treatments (5 g m^{-2} seed, 20 g m^{-2} fertiliser).

Further information on spreading of soil-forming mixtures is given in Appendix 5.

Spreading of soil-forming mixtures on both experimental sites was carried out during July 2006.

5.2.4.4 Vegetation establishment and development

Spreading of soil forming materials over both experimental sites took a month to complete; at maximum efficiency it was possible to apply material to three plots per day, but more commonly two plots were completed daily. Early post-spreading observations were therefore made at different times in different plots; observation day "X", for example, may be five days after spreading for some treatments and eight days after for others. The term "sample occasion" is therefore used in preference to "sample point", for example.

Three fixed-position 20 cm $\times$ 20 cm quadrats were set up randomly in each plot, and counts of germinants in these quadrats were made on six occasions. Counting stopped when seedling density was considered great enough to conduct more extensive survey techniques, for example, estimation of percentage vegetation cover. The last germinant count was made 53 days after spreading.

Germinant frequency was determined once a month for three months after spreading. A 50 cm $\times$ 50 cm quadrat divided into 25 (10 cm $\times$ 10 cm) sub-quadrats was used. Three random (different randomisation for each sample occasion) samples per plot were assessed using this quadrat. If a germinant from the restoration seed mix species was observed in a sub-quadrat, it was recorded; *Digitalis purpurea,* for example, could potentially be recorded a maximum of 25 times per sample. Non-restoration seed mix species were recorded as "other" species.

Total percentage vegetation cover and species richness were estimated using 1 m × 1 m quadrats randomly placed (different randomisation for each sample occasion) at two points in each experimental plot, starting one month after the final spreading. Surveys were conducted at monthly intervals for the initial five months of the experiment, and thereafter at three-monthly intervals for 12 months. Whole plot values for species richness were calculated from quadrat data.

Above-ground biomass harvests were carried out twice during the experiment. Two samples were taken in each plot using 20 cm × 20 cm quadrats, subjectively placed to sample areas of greatest vegetation cover within each plot, and thus providing maximum biomass production estimates for each treatment. All above-ground biomass within quadrats was harvested using fine-tip pruning secateurs, removing shoots and stems completely to the substrate surface. The first biomass harvest was carried out during December 2006, at the end of the first growing season after spreading. The second biomass harvest was carried out in May 2007, during the second growing season post-spreading. All biomass harvested was placed in labelled paper bags and taken to the laboratory for weighing after being oven-dried for a minimum of 48 hours (until constant weight) at 65°C. Biomass fresh weight was not recorded due to the time delay between harvesting and weighing, which would have varied significantly between the day's first and final samples.

Extensive photographic records were also made for the entirety of the experiment, recording the stark changes in vegetation development over time. For example, plate 7 shows site 1 prior to any soil-forming material applications (May 2006), plate 9 shows site 1 following completion of soil-forming material applications (July 2006), and plate 10 shows site 1 12-months after soil forming applications were made (August 2007).

Plate 9 Site 1 following application of all soil-forming material treatments (mid-July 2006)

Plate 10 Vegetation development at site 1 12 months after soil-forming material applications (August 2007)

5.2.4.5 Soil sampling and analysis

Soil samples were collected in June 2006 (before spreading, see section 5.2.4.2) and in November 2007 (17 months after spreading), from 0 – 10 cm depth at three random points in each plot and combined to form composite samples of approximately 500 g for each plot. Soils were weighed (approximately 10 g) into labelled crucibles and oven-dried at 80°C for 36 hours (to constant weight) prior to re-weighing to determine dry weight (DW) and calculate moisture content (MC). Samples were then sieved with a standard 2 mm soil sieve and weighed again; this removed the majority of large slate particles and left the virgin quarry soils for loss on ignition (LOI) analysis, conducted by placing weighed and sieved samples into a 450°C muffle furnace for a minimum period of 12 hours (to constant weight). Following ignition, ashed soil weights were recorded again and LOI calculated (for formula see section 5.2.2.3).

pH and electrical conductivity (EC) of quarry soils were determined on water-extracted slurries (25 g soil in 25 ml distilled water) following stirring for 30 minutes, using an Orion 410A pH (unknown source) meter and Jenway 4010 (Bibby scientific T/As Jenway, Essex, England) conductivity meter respectively.

Further water extracts for analysis of basic soil chemistry were prepared by the addition of 20 g of sieved quarry soil to 100 ml distilled water followed by an hour of subsequent agitation at 250 rpm on a reciprocating shaker ensuring thorough extraction of water soluble elements. Centrifugation of water extracts was then conducted at 8000 rpm and 5°C for 8 minutes; the resultant supernatant was subsequently filtered with 90 mm Whatman no. 41 filter papers into 20 ml sample pots and frozen prior to chemical analysis.

Frozen water extracts of quarry soil samples were analysed colourimetrically, once thawed, for water-soluble ammonium (Mulvaney, 1996), nitrate (Downes, 1978), and phosphate (Murphy and Riley, 1962) using the microplate technique, by measuring light absorbance at 667, 540, and 820 nm respectively by spectrophotometry (Biotek PowerWave XS) (BioTek Instruments, Inc., Vermont, USA). Samples were analysed against standard curves produced from standards of known concentration. Further analysis of these water extracts was carried out using a Sherwood flame photometer 410 (Sherwood Scientific, Cambridge, UK) instrument allowing water soluble sodium, potassium and calcium to be calculated (Hald, 1947).

Heavy metals analysis was conducted on GWC (pre-application samples) and soils collected from GWC plots in November 2007, to provide information on the passage of any potentially toxic elements on to site soils over the course of this experiment. Samples of GWC and soils from GWC plots were allowed to air dry for 14 days and were then ground using a T1-100 vibrating sample mill (Heiko Co. Ltd., Fukushima, Japan) equipped with tungsten grinding vessels. Sub-samples of the ground GWC and soils were then digested using concentrated HNO_3 (Havlin and Soltanpour, 1980). Determination of heavy metals was then carried out by inductively coupled plasma mass spectrometry (ICP-MS) using a Fissions PlasmaQUAD II Turbo ICP-MS.

5.2.4.6 Soil-forming mixture analysis

Analysis of pH, EC, nutrient (ammonium, nitrate, phosphate) and cation (calcium, potassium, sodium) concentrations of soil-forming mixtures were carried out as described in section 5.2.4.5. The three components of surface dressing mixtures were analysed separately and in all combinations as applied to field trial sites (PAM, GWC, slate, GWC + PAM, PAM + slate, GWC + slate, GWC + PAM + slate), in ratios of 0:100 v/v, 50:50 v/v, or 33:33:33 v/v. No replication of samples was done and no statistical analysis of data was carried out.

5.2.5 Statistical analysis

All data were analysed using SPSS 14.0 (SPSS Inc., 2007), after testing for normality using the Kolmogorov-Smirnov test. Most data were not normally distributed and also showed lack of homogeneity (Levene's test of equality of error variances). Assumptions underlying the use of parametric statistical tests, specifically T-tests and analysis of variance (ANOVA), were repeatedly violated. Therefore all statistical analyses were carried out using Mann-Whitney, Wilcoxon signed rank, or Kruskal-Wallis non-parametric tests. Some information derived from parametric post-hoc (Tukey) testing is presented for illustrative purposes only; these results should be treated with caution.

N.B. Data were analysed both with and without the inclusion of the straw treatment results and the conclusions for all other treatments did not change. Results presented hereafter include the straw treatment, but comparisons between the straw and other treatments should be treated with caution.

5.3 Results

5.3.1 Water-holding materials

5.3.1.1 Water-holding capacity

There were significant differences (p = 0.001) in water-holding capacity (WHC) among the four materials tested, as shown in table 5.3. Both wallpaper paste ($p = 0.000$) and powdered PAM ($p \leq 0.001$) had significantly lower WHC than Na-polyacrylate (Na-PA) and granular PAM, which were not significantly different from each other ($p = 0.907$).

Table 5.3 Water-holding capacity of super-absorbent polymers

Polymer	N	Mean WHC (g H_2O/g polymer (dry))	Std. Error
Na-polyacrylate	5	347.0426	4.9098
PAM (granular)	5	343.2296	5.5761
Wall paper paste	5	10.2680	1.0020
PAM (powdered)	5	313.8606	2.9686

5.3.1.2 Repeated drying effects on PAM

Both granular and powdered PAM showed reduced WHC after wetting and drying (see table 5.4), but the granular form was more affected, with significant losses of WHC, for example, by 9.039 % from drying cycle 1 to 2 (p = 0.0431) and 7.588 % from drying cycle 2 to 3 (p = 0.0431). The powdered form only showed a significant loss of WHC from drying cycle 1 to 2, losing 4.924 % (p = 0.0431). By the end of the third drying cycle there was no significant difference in WHC between the two polymers (p = 0.1745), whereas at the end of drying cycle 1 and 2 there were significant differences (cycle 1 $p = 0.0090$; cycle 2 $p = 0.0283$).

Table 5.4 Effect of wetting-drying cycles on the water-holding capacity of polyacrylamide (PAM)

Drying cycle	Polymer	N	Mean WHC (g H_2O/g polymer (dry))	Std. Error
1	PAM (granular)	5	378.6066	11.6267
	PAM (powdered)	5	313.8606	2.9686
2	PAM (granular)	5	344.3856	11.1006
	PAM (powdered)	5	298.4052	3.0897
3	PAM (granular)	5	318.2554	13.0417
	PAM (powdered)	5	290.9922	7.0215

5.3.2 Green waste composts

5.3.2.1 Triticum aestivum bioassay

No germination occurred during the first two days after sowing, but germination on days 3 to 9 was significantly different among the composts tested (table 5.5). Seeds sown in Conwy GWC demonstrated the fastest emergence and the highest total germination after ten days (mean= 24.75, table 5.6). From day 6 onwards, GWC sourced from Anglesey had significantly lower germination than all other composts in the trial (e.g. day 9 $p \leq 0.010$), and the lowest germination after ten days (table 5.6).

There were significant differences (p = 0.0000) in shoot length among all composts tested. Compost sourced from Anglesey again gave poor results, producing plants with an average shoot length of 9.93 cm, the shortest observed and significantly lower ($p \leq 0.0170$) than all other treatments. There were also significant differences in fresh weight (p = 0.0000), dry weight (p = 0.0000) and moisture content (p = 0.0000) among treatments. Peat-based compost produced significantly greater biomass (p = 0.0000; dry weight mean = 0.0324 g) than all other treatments and Anglesey GWC again performed most poorly (significantly different from Conwy, Wrexham, peat and neem-coir ($p \leq 0.0010$); dry weight mean= 0.0163 g). Survival was 100 % in all treatments. Figure 5.2 shows the performance of *Triticum aestivum* in the eight growing media.

Table 5.5 Results of Kruskal-Wallis test for differences in wheat germination in eight growing media. Figures in bold indicate significant differences among composts

Sample occasion (days)	p
1	1.000
2	1.000
3	**0.028**
4	**0.001**
5	**0.045**
6	**0.038**
7	**0.040**
8	**0.038**
9	**0.034**
10	0.052

Table 5.6 Mean number of wheat germinants per growing medium

Sample occasion (days)	Compost source	Mean	N	Std. Error
2	Conwy	0.00	4	0.00
	Flintshire	0.00	4	0.00
	Denbigh	0.00	4	0.00
	Anglesey	0.00	4	0.00
	Wrexham	0.00	4	0.00
	Neem-coir	0.00	4	0.00
	Peat	0.00	4	0.00
	Topsoil	0.00	4	0.00
4	Conwy	24.50	4	0.50
	Flintshire	6.00	4	1.29
	Denbigh	12.50	4	0.29
	Anglesey	9.75	4	1.44
	Wrexham	18.00	4	1.78
	Neem-coir	18.75	4	2.17
	Peat	19.00	4	2.68
	Topsoil	7.25	4	3.35
6	Conwy	24.75	4	0.25
	Flintshire	23.50	4	0.87
	Denbigh	23.50	4	0.65
	Anglesey	19.00	4	0.82
	Wrexham	23.75	4	0.48
	Neem-coir	24.00	4	0.41
	Peat	23.75	4	0.25
	Topsoil	22.50	4	1.04
8	Conwy	24.75	4	0.25
	Flintshire	23.75	4	0.63
	Denbigh	23.75	4	0.63
	Anglesey	19.50	4	0.96
	Wrexham	23.75	4	0.48
	Neem-coir	24.00	4	0.41
	Peat	23.75	4	0.25
	Topsoil	23.00	4	0.82
10	Conwy	24.75	4	0.25
	Flintshire	23.75	4	0.63
	Denbigh	24.25	4	0.48
	Anglesey	19.75	4	0.75
	Wrexham	23.75	4	0.48
	Neem-coir	24.00	4	0.41
	Peat	24.00	4	0.41
	Topsoil	23.25	4	0.85

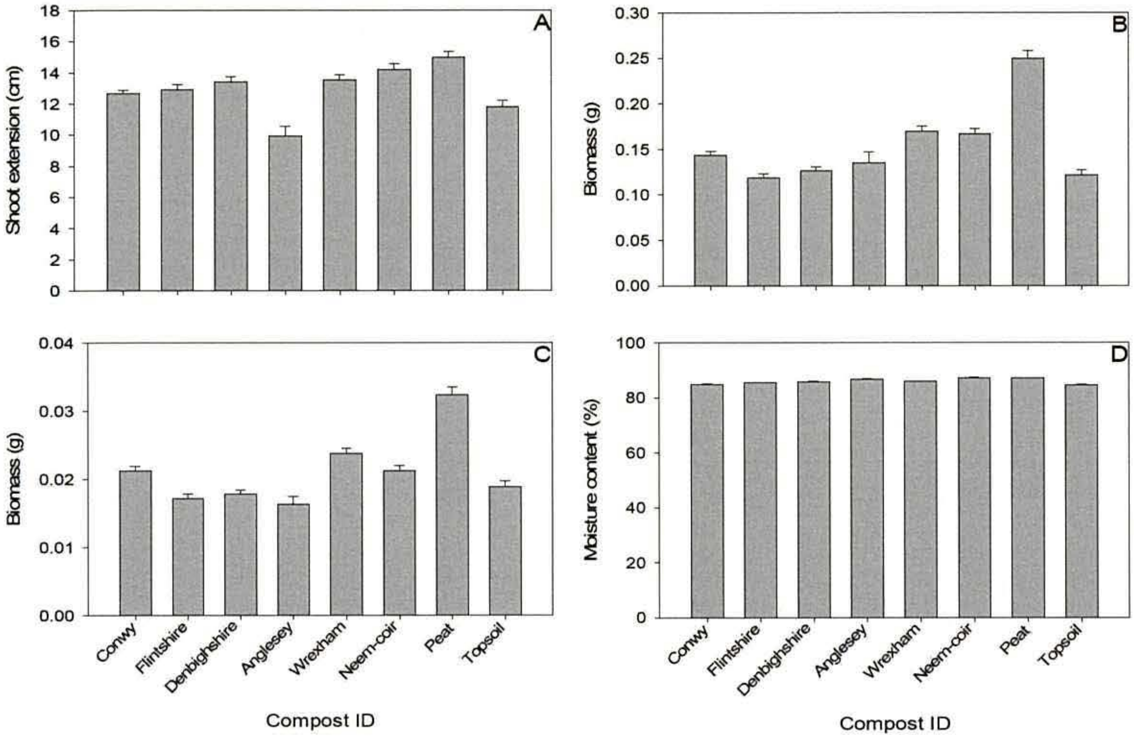


Figure 5.2 Wheat shoot length (A), fresh biomass (B), dry biomass (C), and moisture content (D) when grown in a range of composts (bars = standard error of the mean)

5.3.2.2 Restoration species bioassay

There were significant differences among the composts tested. Figure 5.3 shows that no germination occurred in any treatment until day five, when several germinants of *Agrostis capillaris* were observed. At day seven the first *Cytisus scoparius* germination was recorded, and not until day nine did any *Betula pendula* seeds germinate. At day 10 there were significant differences among composts sown with *Agrostis capillaris* ($p = 0.0039$). Compost sourced from Wrexham had the highest mean number of germinants (13.50), and compost sourced from Flintshire the lowest (2.50). At day 20 all species showed significant differences among treatments (*Agrostis capillaris* $p = 0.0021$; *Cytisus scoparius* $p = 0.0054$; *Betula pendula* $p = 0.0316$; and pre-treated *Betula pendula* $p = 0.0416$). Some patterns were evident at this stage. For example, the compost with the lowest mean number of germinants of both *Agrostis capillaris* and *Cytisus scoparius* was sourced from Flintshire (*Agrostis capillaris* = 4.25, *Cytisus scoparius* = 1.25), while both un-treated and pre-treated birch showed the greatest mean germination in peat-based compost and the lowest in the compost from Denbighshire (peat = 6.25 and 5.25 for un-treated and pre-treated seed respectively; Denbighshire = 0.75 and 0.50 for un-treated and pre-treated seed respectively). By the end of the bioassay (day 65), the only differences among treatments were in composts sown with *Agrostis capillaris* and *Cytisus scoparius* (*Agrostis capillaris* $p = 0.0030$ and *Cytisus scoparius* $p = 0.0202$).

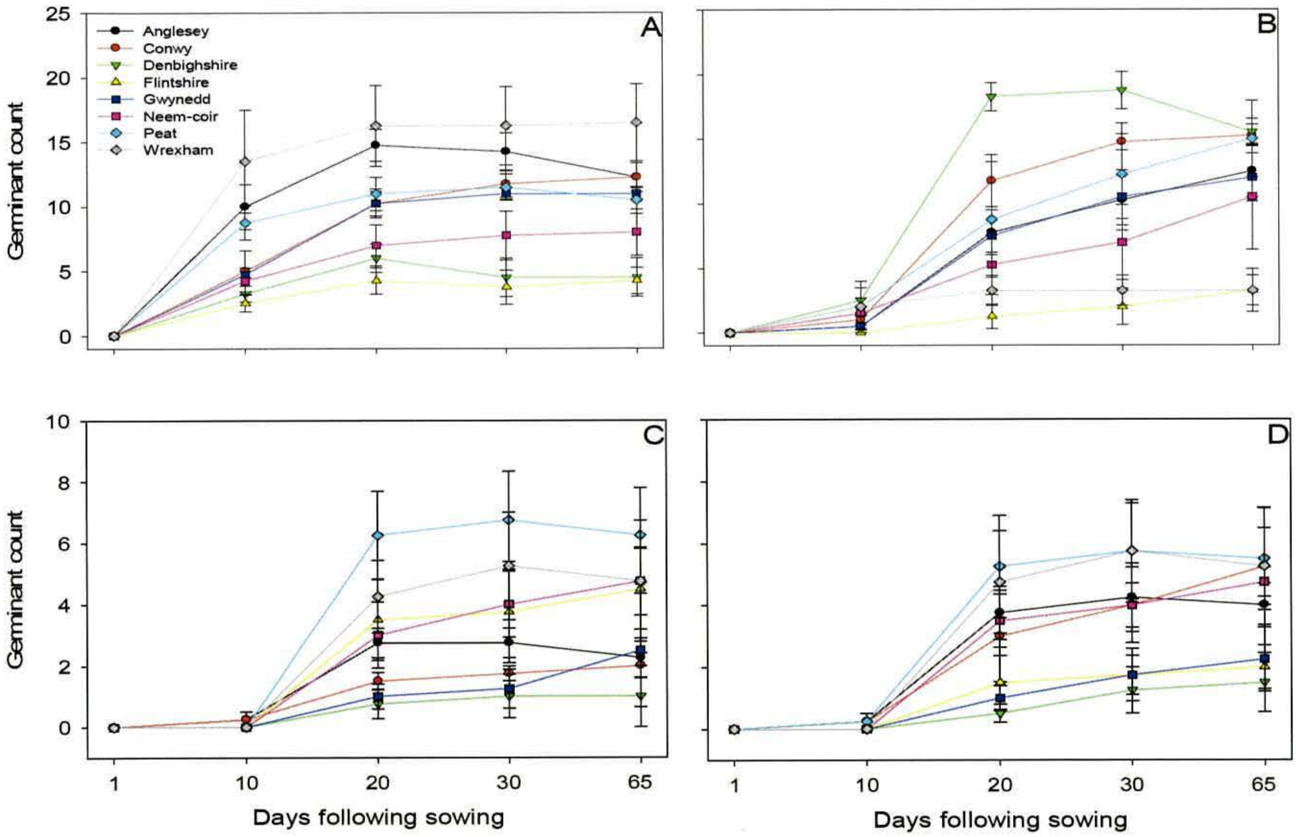


Figure 5.3 *Agrostis capillaris* (A), *Cytisus scoparius* (B), *Betula pendula* (C), and pre-treated *Betula pendula* (D) germination in a range of composts and growing media (bars = standard error of the mean)

There were significant differences among treatments in shoot length ($p = 0.0000$), fresh weight ($p = 0.0000$) and dry weight ($p = 0.0000$), moisture content ($p \leq 0.0005$), and SPAD value ($p \leq 0.0500$)) for all test species (figure 5.4). Peat-based compost produced plants with the greatest shoot length in all species, and Gwynedd GWC produced plants with the lowest shoot length in all species but *Cytisus scoparius*. This was similar to the pattern of fresh and dry weights; plants grown in peat-based compost had significantly greater dry weights than plants grown in most other composts, except for pre-treated *Betula pendula*, where plants in neem-coir compost had greater dry weights than those in peat-based compost. Plants grown in compost sourced from Gwynedd had the lowest dry weight in all species but *Cytisus scoparius*.

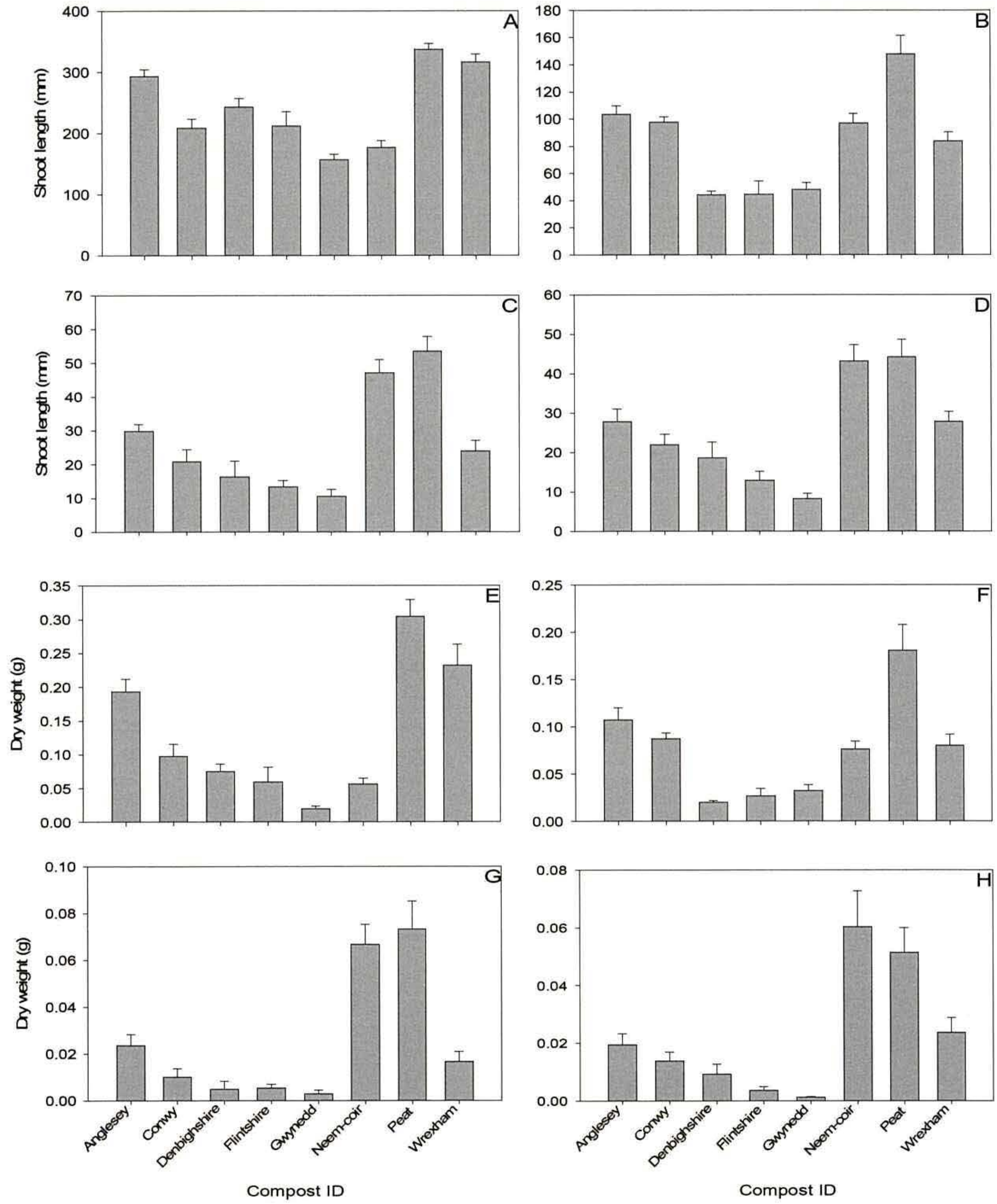


Figure 5.4 Shoot length of *Agrostis capillaris* (A), *Cytisus scoparius* (B), un-treated *Betula pendula* (C), and pre-treated *Betula pendula* (D) and dry weight of *Agrostis capillaris* (E), *Cytisus scoparius* (F), un-treated *Betula pendula* (G), and pre-treated *Betula pendula* (H), when grown in a range of green waste composts (bars = standard error of the mean)

There were significant differences in moisture content among treatments, but no consistent patterns across species. *Cytisus scoparius* (86.26 %) and pre-treated *Betula pendula* (80.09 %) plants had the greatest moisture content when grown in neem-coir compost; *Agrostis capillaris* plants had the greatest moisture content (86.74 %) when grown in Flintshire GWC, and un-treated *Betula pendula* the greatest (84.47 %) when grown in GWC from Denbighshire.

There were significant differences in SPAD values among treatments (figure 5.5). Specimens of *Agrostis capillaris* (mean= 35.34) and pre-treated *Betula pendula* (31.68) had the highest SPAD scores when grown in Anglesey GWC, while un-treated *Betula pendula* performed best in Denbighshire GWC (mean = 31.00), and *Cytisus scoparius* performed best in peat-based compost (mean= 40.05). Composts producing the least-green plants, and therefore those with the lowest SPAD scores, showed no particular pattern, although both *Cytisus scoparius* (mean = 13.76) and pre-treated *Betula pendula* (6.06) had the lowest scores when grown in Flintshire GWC.

Weed count was also significantly different among composts in three of the test species (*Agrostis capillaris* $p = 0.0006$; *Cytisus scoparius* $p = 0.0046$; pre-treated *Betula pendula* $p = 0.0397$). Compost sourced from Anglesey had the most weeds in seed trays sown with both *Agrostis capillaris* (mean = 3.25) and *Cytisus scoparius* (mean = 2.25) (figure 5.5), whereas in the trays sown with pre-treated *Betula pendula*, peat-based compost had the highest mean number of weeds (mean = 5.50). Some seed trays did not produce any weeds throughout the duration of the 65-day growth study. In trays sown with *Agrostis capillaris*, composts from Flintshire and Wrexham, and peat-based and neem-coir composts remained weed free; in trays sown with *Cytisus scoparius*, Denbigh and Flintshire GWCs did not develop any weeds; and in trays sown with pre-treated *Betula pendula*, Flintshire and Gwynedd GWC also remained weed free.

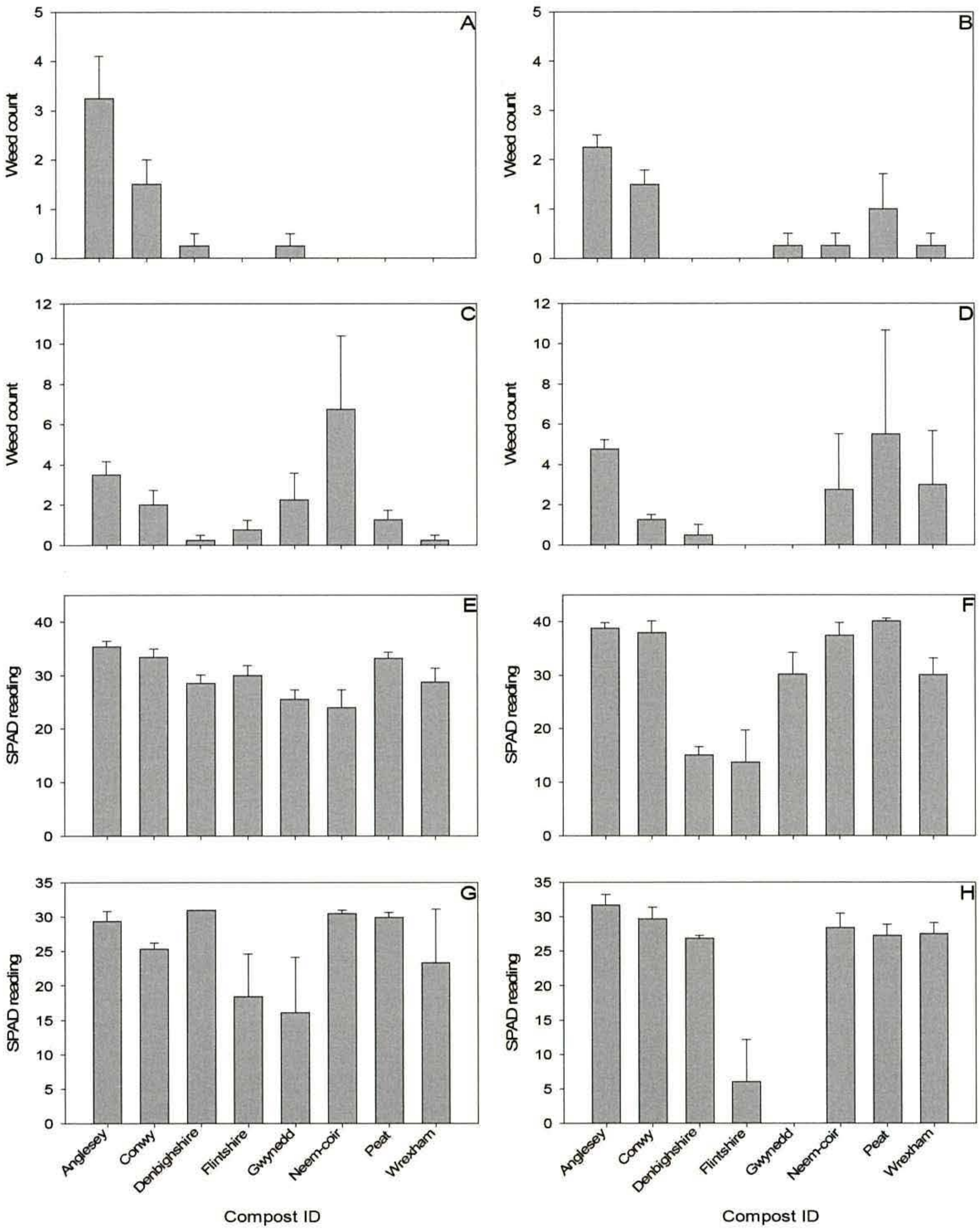


Figure 5.5 Weed count (number of individuals) in trays of *Agrostis capillaris* (A), *Cytisus scoparius* (B), un-treated *Betula pendula* (C), and pre-treated *Betula pendula* (D) and SPAD scores of *Agrostis capillaris* (E), *Cytisus scoparius* (F), un-treated *Betula pendula* (G), and pre-treated *Betula pendula* (H) when grown in a range of green waste composts (bars = standard error of the mean)

5.3.2.3 Physical and chemical analysis

Figure 5.6 shows the particle size distribution of the six GWCs used in the two bioassays. There were significant differences among composts. The percentages of all particle size classes, with the exception of the >9 mm (p = 0.0526) and >106 μm (p = 0.0521) classes, were significantly different among GWCs. Conwy GWC was notably different from all other composts; it contained no particles in the size classes >10 mm and >9 mm.

N.B. Size classes are mutually exclusive by definition due to sieving with the largest sieves first, e.g. the >10 mm fraction is removed prior to sieving with the >9 mm sieve, and particles that are 11 mm in diameter, for example, despite also being >9 mm, will have previously been removed in the >10 mm fraction.

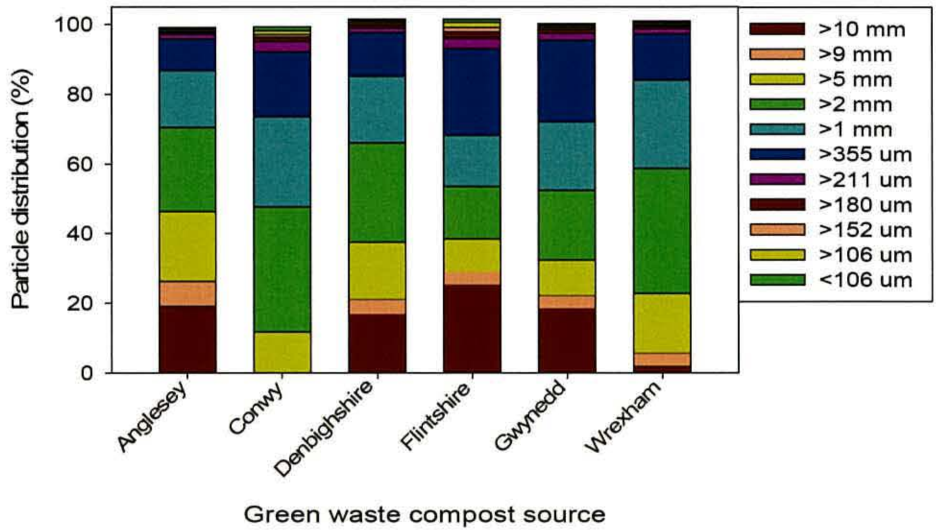


Figure 5.6 Particle size distribution of green waste composts

There was also variation among GWCs in terms of their chemical composition. Only one sample of each GWC was analysed, and no statistical analysis of the results was possible.

There were some clear differences in loss on ignition (LOI) between the composts, as shown in figure 5.7 (D). For example, composts sourced from Anglesey and Conwy both had LOI values greater than 50 % (53.81 % and 50.25 % respectively), whereas composts sourced from Gwynedd and Wrexham had LOI values of 24.81 % and 22.00 % respectively.

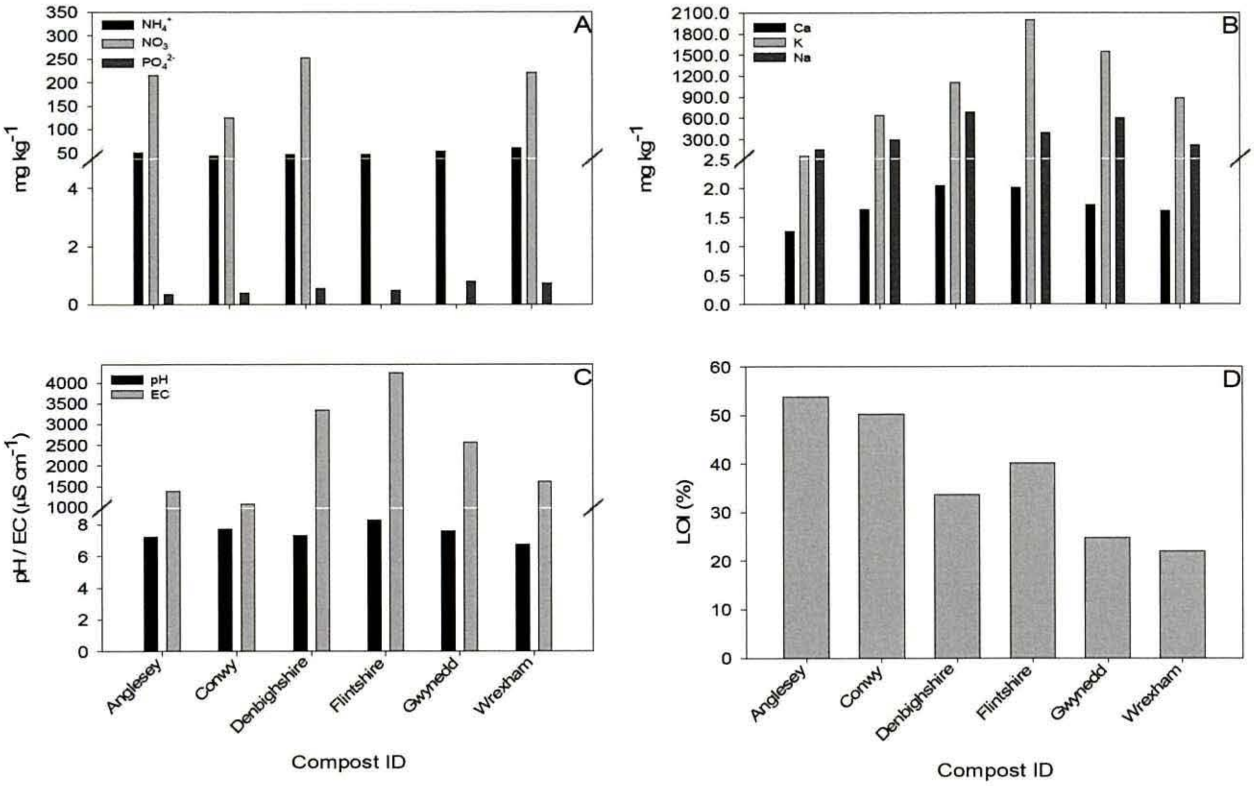


Figure 5.7 Ammonium, nitrate, and phosphate concentration (A); calcium, potassium, and sodium concentration (B); pH and EC (C); and loss on ignition (D) of green waste composts used in bioassay studies

N.B. Flintshire and Gwynedd GWCs had nitrate concentrations below the minimum detection limit

All composts were roughly pH neutral, or slightly acidic or alkaline (figure 5.7 (C)). The greatest departure from neutrality was found in Flintshire GWC, which had a pH of 8.30.

All electrical conductivity (EC) values of composts were relatively similar, as shown in figure 5.7 (C), falling within the range 1000 - 5000 μS cm^{-1}. The compost extract with the greatest concentration of dissolved salts was sourced from Flintshire, which had an EC of 4240 μS cm^{-1}, in contrast with Conwy GWC, which had the lowest EC of 1080 μS cm^{-1}.

Water extractable ammonium and phosphate showed little variation among GWCs, but nitrate concentrations did differ appreciably among GWC sources (see figure 5.7 (A)). Two composts (Flintshire and Gwynedd) had nitrate levels below the minimum detection limit. Conwy-sourced GWC had a nitrate concentration (124.38 mg kg^{-1} N) appreciably lower than the remaining three sources, which had concentrations ranging from 215.92 mg kg^{-1} N to 252.57 mg kg^{-1} N.

Water extractable cation concentrations showed some variation among GWCs, as shown in figure 5.7 (B). Calcium concentration showed little variation, but both potassium and sodium concentrations differed quite substantially among composts. Anglesey GWC had the lowest potassium concentration of 60.30 mg kg^{-1}; this is some 30 times less than the highest concentration of potassium in Flintshire GWC (1996.94 mg kg^{-1}). Anglesey GWC also had the lowest concentration of sodium (152.53 mg kg^{-1}); the highest sodium concentration (687.36 mg kg^{-1}) was in the Denbighshire GWC.

5.3.2.4 Composite comparison of GWCs

By ranking the GWCs by results of all tests, a composite score was generated for the six GWC sources included in all tests, enabling comparisons of their relative overall quality in terms of their nutrient content, physical properties and general suitability for plant growth. Ranking was conducted by giving a score of 1 to the GWC with the best result for an individual test and 6 to the GWC with the worst result (e.g. for wheat germination, Conwy GWC produced the greatest number of germinants after 10 days and was given a score of 1; whereas Anglesey GWC produced the lowest number of germinants and was scored 5). (N.B. in the wheat experiments the lowest score given was 5 because only five GWCs were tested in this study). Commercially-produced growth media were excluded from this ranking (i.e. peat, peat-free topsoil and neem coir were not ranked). The total of ranks is shown in table 5.7 and the overall ranking is based on this total, with the best rank being granted to the GWC with the lowest total (i.e. the greatest number of first place ranks is equivalent to the lowest total score).

Table 5.7 Composite comparisons of GWCs

GWC source	Test				Total	Rank
	Wheat trial	Multi-species trials	Physical characteristics	Chemical characteristics		
Anglesey	5	2	2	5	14	5
Conwy	2	3	1	1=	7	1=
Denbighshire	3	4	3	1=	11	3
Flintshire	4	5	5=	6	20	6
Gwynedd	*	6	5=	1=	12	4
Wrexham	1	1	4	1=	7	1=

N.B. Wheat trial comprises results of germination, shoot length and fresh weight; multi-species trial comprises results of germination, shoot length, fresh weight, SPAD and weed count for *Agrostis capillaris*, *Cytisus scoparius*, un-treated *Betula pendula* and pre-treated *Betula pendula*; physical characteristics comprises results of particle size analysis and loss on ignition; chemical analysis comprises results of nitrate, ammonium, phosphate, calcium, potassium, sodium and electrical conductivity determination (pH was not included).

5.3.3 Soil-forming mixtures: greenhouse fertiliser trial

5.3.3.1 Germination

There were significant differences in germination of both monocotyledonous and dicotyledonous species between fertilised and un-fertilised treatments. No germination was recorded on the first day after sowing. The first germination occurred on day two, when differences in monocotyledonous germinants between fertiliser treatments were found in the GWC (p = 0.0495) and GWC + slate (p = 0.0369) mixtures. At the end of germination recording (seven days after sowing) there were significant differences between fertiliser treatments in monocotyledon counts in GWC (p = 0.0495), GWC + slate (p = 0.0495) and GWC + PAM + slate (p = 0.0495) mixtures, and in dicotyledon counts in the GWC + PAM + slate (p = 0.0495) mixture (figure 5.8).

5.3.3.2 Biomass

There were significant differences in fresh weight between fertiliser treatments (all p = 0.0495) in all but one of the soil-forming mixtures (GWC + PAM + slate) and the control. The result was similar for dry weight (in this case there was also no significant difference between fertiliser treatments in the GWC +

PAM mixture). There were no significant differences in moisture content between fertiliser treatments (figure 5.9).

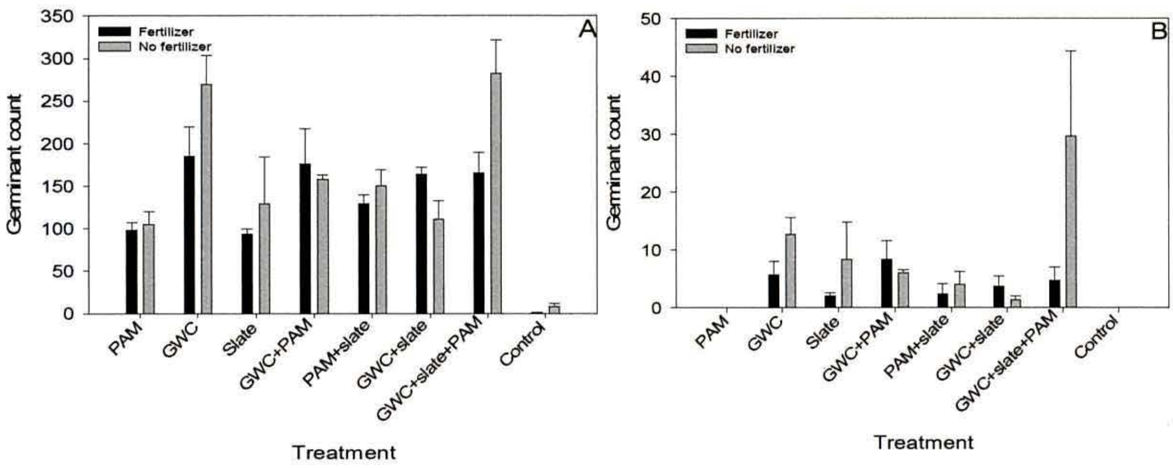


Figure 5.8 Effect of fertiliser upon day 7 germinant counts of monocotyledons (A) and dicotyledons (B) in soil-forming mixtures (bars = standard error of the mean)

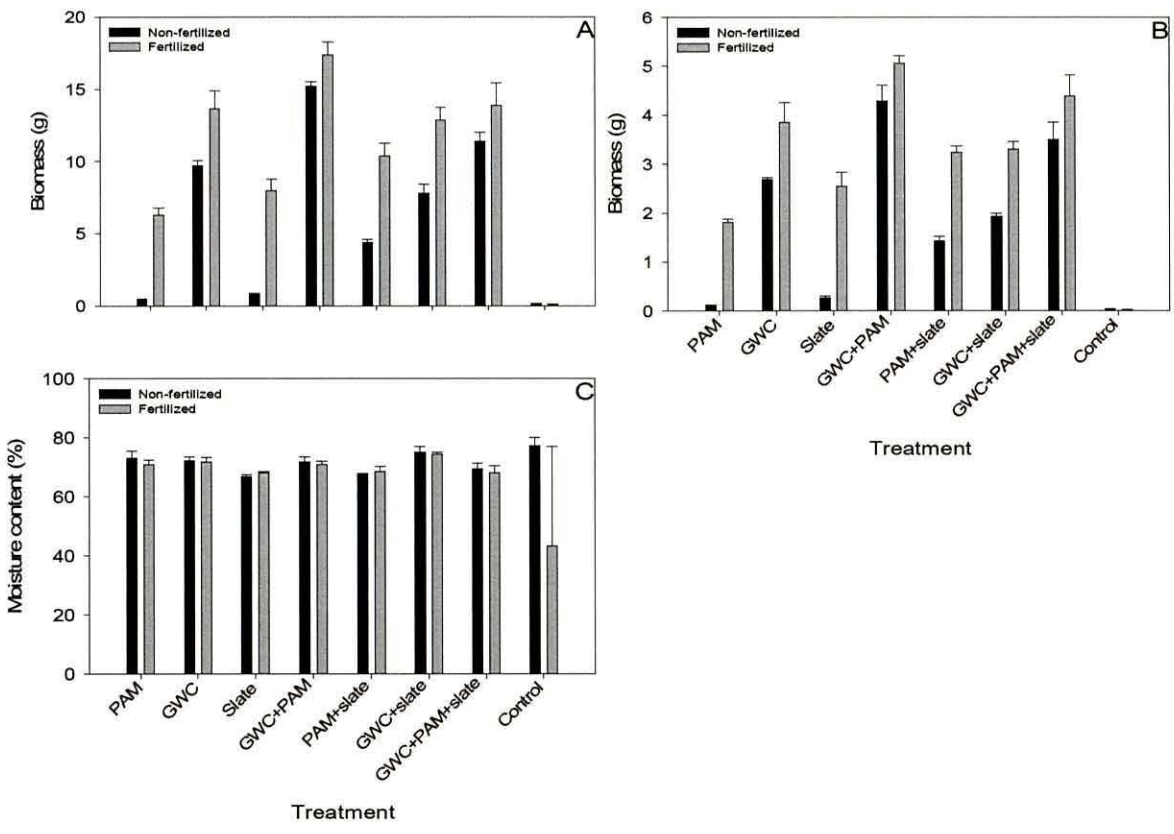


Figure 5.9 Effect of fertiliser on fresh weight (A), dry weight (B) and moisture content (C) of vegetation grown in soil-forming mixtures (bars = standard error of the mean)

5.3.4 Soil-forming mixtures: quarry trial

5.3.4.1 Germination

There were significant differences (all p = 0.000) in germination among treatments on every sampling occasion at both site 1 and site 2. As shown in table 5.8, the earliest germination at site 1 was in slate plots. On sample occasion 3 (roughly 20 days after spreading), GWC plots had the greatest number of germinants per 400 cm^{-2} quadrat (mean = 26.7). This continued to be the situation until sample occasion 6 (approximately 50 days after spreading) when GWC + PAM + slate plots had the greatest number of germinants per quadrat (mean = 44.4). Germination at site 2 showed a similar pattern to that at site 1, although germination was best in GWC plots until sample occasion 5 (roughly 35 days after spreading) when GWC + PAM + slate plots had the same number of germinants per quadrat (mean = 32.6); on sample occasion 6 (roughly 45 days after spreading) the tri-material mixture had more germinants per quadrat (mean= 42.6) than GWC plots.

Table 5.8 Germinant counts per 400 cm^{-2} quadrat at site 1 and site 2 (* indicates that on this sample occasion these soil forming mixtures had not yet been spread)

Site	Treatment	N	Sample occasion											
			1		2		3		4		5		6	
			Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.
1	GWC	9	1.8	0.4	20.1	1.0	26.7	1.1	30.0	0.9	35.0	0.7	43.2	0.8
	Slate	9	6.9	1.3	22.1	0.8	24.4	0.6	26.0	0.4	27.2	0.8	24.1	0.7
	PAM	9	0.0	0.0	2.4	0.5	4.2	0.3	7.0	0.4	8.8	0.5	11.0	0.4
	GWC+slate	9	0.0	0.0	20.8	0.9	24.0	0.9	26.1	0.6	30.8	0.4	37.2	0.6
	GWC+PAM	9	0.0	0.0	3.1	0.4	12.1	0.5	22.3	0.6	33.0	0.7	41.9	0.4
	Slate+PAM	9	0.0	0.0	2.4	0.4	8.4	1.1	10.8	0.8	21.8	1.0	34.6	1.1
	GWC+slate+PAM	9	0.0	0.0	6.0	0.5	17.1	0.4	25.9	0.8	34.1	0.9	44.4	0.5
	Control	9	0.0	0.0	0.0	0.0	2.6	0.6	7.2	0.8	9.1	0.8	11.1	0.6
	Straw	3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
2	GWC	9	1.4	0.2	7.1	0.3	16.8	0.7	24.1	0.6	32.6	0.6	41.6	0.2
	Slate	9	0	0	3.4	0.4	10.8	0.8	17.8	0.7	24.2	0.7	23.8	0.4
	PAM	9	*	*	0.0	0.0	0.0	0.0	4.0	0.2	7.9	0.3	10.6	0.3
	GWC+slate	9	*	*	0.0	0.0	0.0	0.0	20.0	0.4	29.1	0.5	33.2	0.6
	GWC+PAM	9	0	0	3.4	0.4	5.9	0.4	19.7	0.4	30.8	0.4	41.3	0.7
	Slate+PAM	9	0	0	0.7	0.3	2.2	0.3	9.4	0.6	18.8	0.6	29.6	0.4
	GWC+slate+PAM	9	0	0	6.8	0.4	9.9	0.2	19.2	0.4	32.6	0.5	42.6	0.3
	Control	9	*	*	0.0	0.0	0.0	0.0	0.2	0.1	0.2	0.1	0.0	0.0
	Straw	3	*	*	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.3

There were significant differences among treatments in the germination of grasses, clover and foxglove/woodsage on each sample occasion at both site 1 and site 2 (all $p = 0.000$). Analysis of broom/gorse germinant frequency data was complicated by the fact that even by sample occasion 3, three months after spreading, few germinants had emerged (figures 5.10 and 5.11). At site 1 (figure 5.10) differences in germination of non-seed mix species among the treatments were significant (p = 0.017) only on the first sampling occasion (August 2006, one month after spreading). However, at site 2 (figure 5.11) there were significant differences in germination of non-seed-mix species among treatments on each of the sampling occasions (August p = 0.040; September $p = 0.008$; October $p = 0.010$).

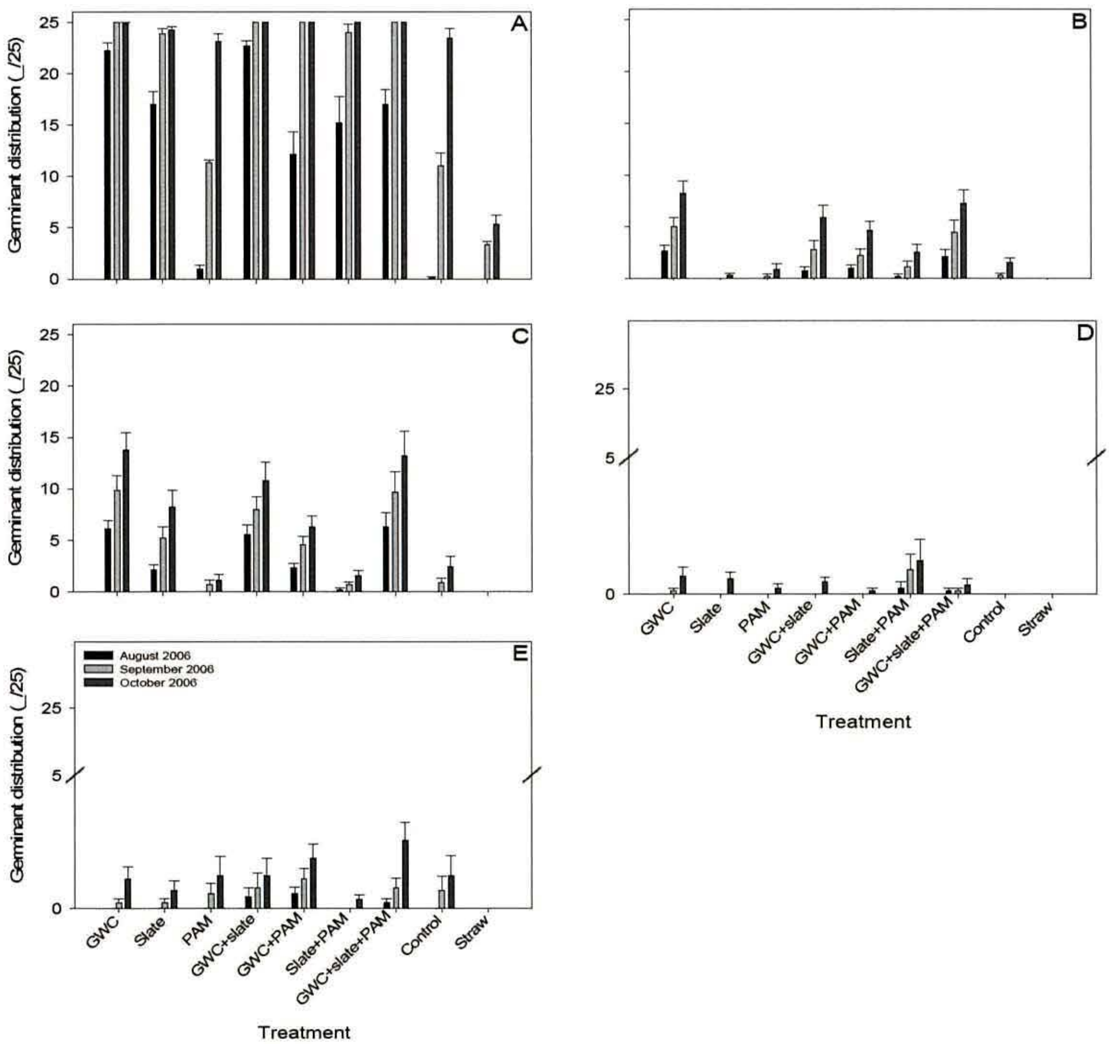


Figure 5.10 Germination of grasses (A), clover (B), foxglove/woodsage (C), broom/gorse (D) and other species (E) at site 1 (bars = standard error of the mean)

Generally germination was better in soil-forming mixtures containing GWC. The only real exception was broom/gorse germination; at site 1, for instance, on all three sampling occasions the treatment with the highest mean germination was slate + PAM, and the same was true at site 2 on sample occasions one and three. It appears that broom and gorse germinate most successfully in this mixture.

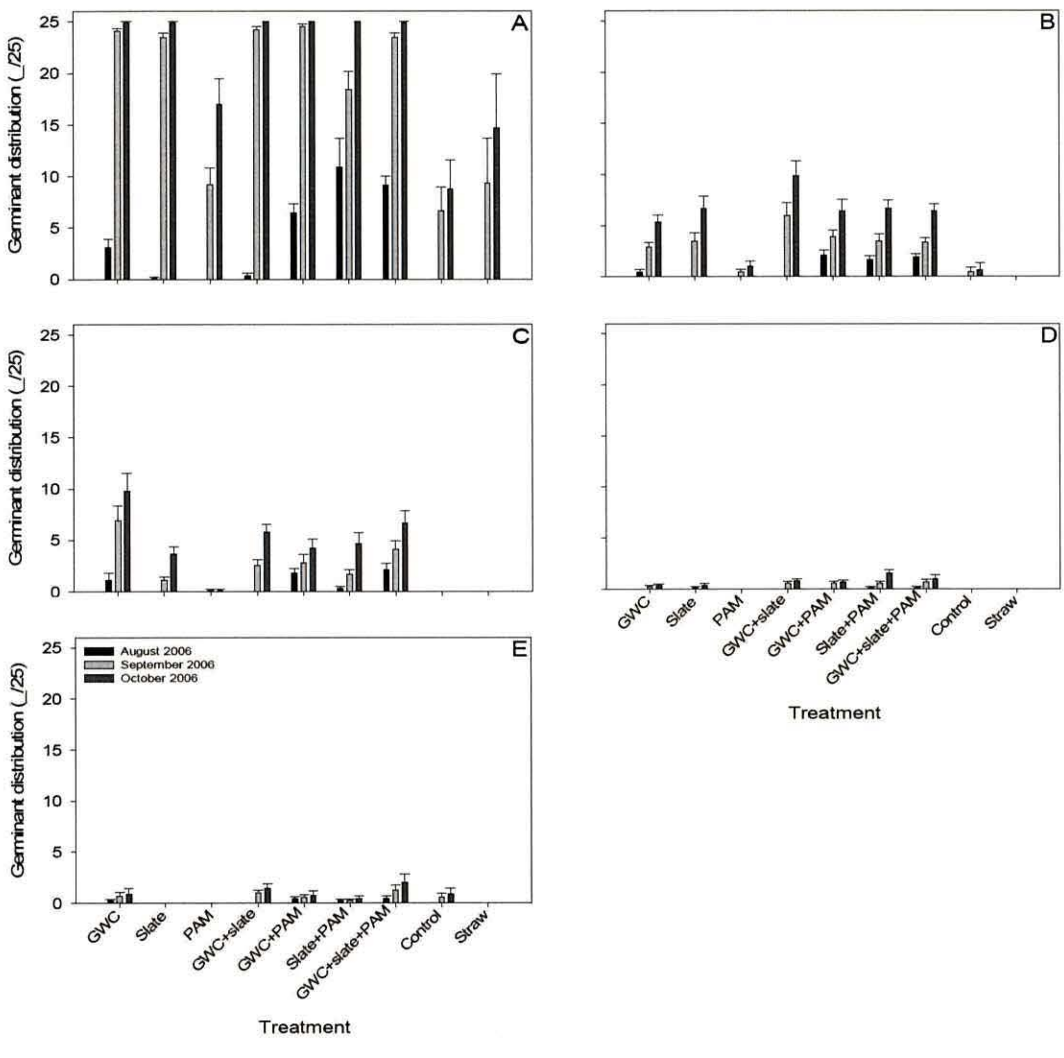


Figure 5.11 Germination of grasses (A), clover (B), foxglove/woodsage (C), broom/gorse (D) and other species (E) at site 2 (bars = standard error of the mean)

5.3.4.2 Species richness

There were clear among-treatment differences in species richness, as shown in figure 5.12. On all but two sampling occasions at site 1, and on all eight sampling occasions at site 2, differences among treatments were significant (site 1 = all $p \leq 0.028$; site 2 = all $p \leq 0.022$). The most pronounced differences at site 1 occurred on sample occasion 5 (March 2007), when the GWC + PAM soil-forming mixture had the greatest species richness (mean = 14.33); this was significantly greater than species richness in all other treatments (all $p \leq 0.046$). All species recorded across both experimental sites and all treatments are listed in table 5.9.

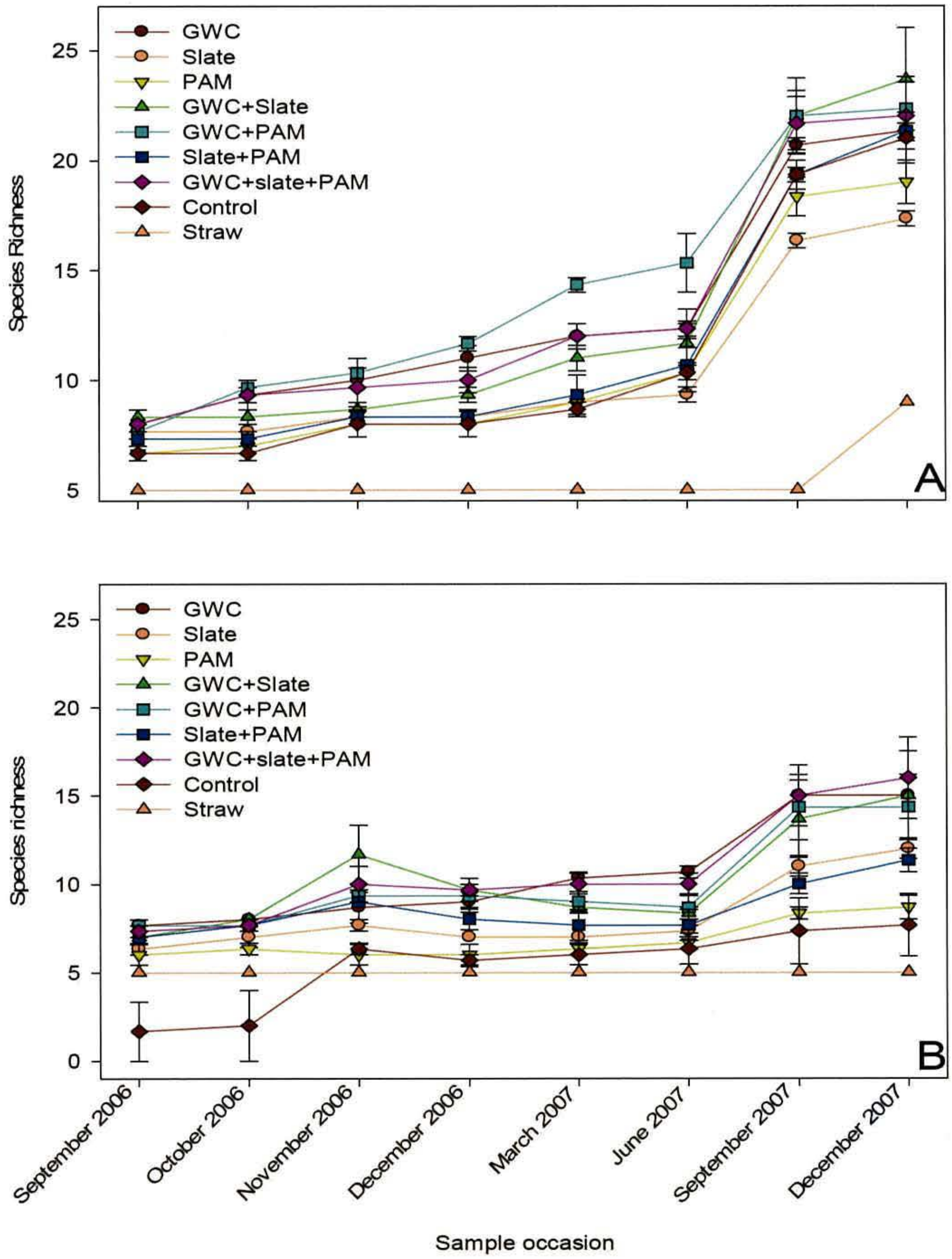


Figure 5.12 Species richness development over duration of field observations at site 1 (A) and site 2 (B) (bars = standard error of the mean)

Figure 5.13 shows the species richness recorded in the baseline survey carried out prior to soil-forming mixture applications in June 2006, and species richness 17 months after applications (November 2007). It can be clearly seen that all soil-forming treatments (excluding straw at site 1) had a beneficial impact on species richness at both sites.

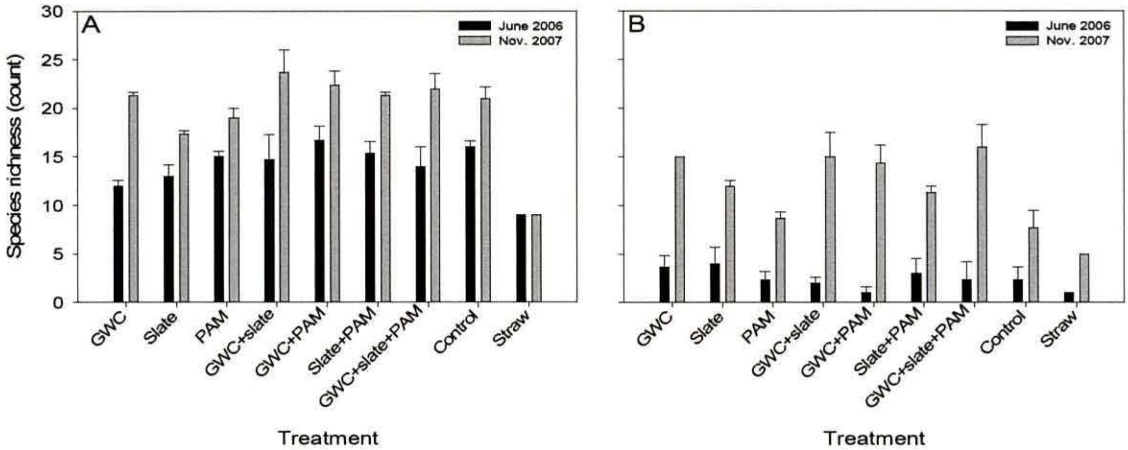


Figure 5.13 Comparison of species richness at site 1 (A) and site 2 (B) before applying soil-forming mixtures and 17 months after application (bars = standard error of the mean)

5.3.4.3 Vegetation cover

Five months after spreading (December 2006), as demonstrated in figure 5.14, plots treated with GWC had the greatest vegetation cover at site 1 (mean = 28.8 %), whereas GWC + PAM plots had the greatest vegetation cover at site 2 (mean = 56.7 %). It is noteworthy that on this sampling occasion, the most successful treatment at site 2 had markedly greater vegetation cover than the most successful treatment at site 1.

By sample occasion 6 (March 2007), vegetation at site 2 had developed further, and the most successful treatment at site 2 was GWC + PAM (vegetation cover mean = 64.2 %). At site 1, the best performing treatment on this sample occasion was GWC + slate + PAM (mean = 30.0 %). As shown in figure 5.14 (B), a substantial decline in vegetation cover occurred between March and June 2007 at site 2; on the later sample occasion, although the most successful treatment at site 2 was still GWC + PAM, it had a mean vegetation cover of 51.7 %, 12.5 % less than in March 2007. However, at site 1, the most successful treatment (GWC + slate + PAM) had a mean vegetation cover of 57.5 % in June 2007, nearly double that recorded in March 2007.

Table 5.9 Species recorded in experimental plots at sites 1 and 2

Species	Common name
Acer pseudoplatanus	Sycamore
Agrostis capillaris	Common bent
Alnus glutinosa	Alder
Arenaria serpyllifolia	Thyme leaved sandwort
Asplenium scolopendrium	Harts tongue
Asplenium trichomanes	Maidenhair spleenwort
Betula pendula	Birch
Buddleja davidii	Buddleia
Calluna vulgaris	Heather (ling)
Campanula rotundifolia	Harebell
Cardamine hirsuta	Hairy bittercress
Cerastium holosteoides	Common mouse-ear (chickweed)
Cirsium arvense	Thistle
Cryptogramma crispa	Parsley fern
Cystopteris dickieana	Dickies bladder fern
Cytisus scoparius	Broom
Digitalis purpurea	Foxglove
Dryopteris filix-mas	Male fern
Dryopteris oreades	Dwarf male fern
Dryopteris villarii	Rigid buckler fern
Epilobium augustifolium	Rosebay willowherb
Epilobium penduncularе	New Zealand willowherb
Erica cinerea	Bell heather
Festuca rubra	Strong creeping red fescue
Festuca rubra ssp. rubra	Chewings fescue
Geranium robertianum	Herb robert
Hieracium perpropinquum	Hawkweed
Huperzia selago	Fir clubmoss
Hydrocotyle vulgaris	Marsh pennywort
Lolium perenne	Perennial ryegrass
Pinus sylvestris	Scots pine
Plantago major	Ratstail plantain
Poa compressa	Flattened meadow grass
Polytrichum sp.	Moss
Rubus fruticosus	Bramble
Rumex obtusifolius	Broadleaf dock
Salix reichardtii	Willow
Sedum acre	Biting stonecrop
Sedum anglicum	English stonecrop
Senecio jacobaea	Common ragwort
Sorbus aucuparia	Rowan
Taraxacum officinale	Dandelion
Teucrium scorodonia	Wood-sage
Trifolium repens	Small leaved white clover
Ulex europaeus	Gorse
Urtica dioica	Stinging nettle
Vaccinium myrtillus	Bilberry
Veronica hederifolia	Ivy speedwell

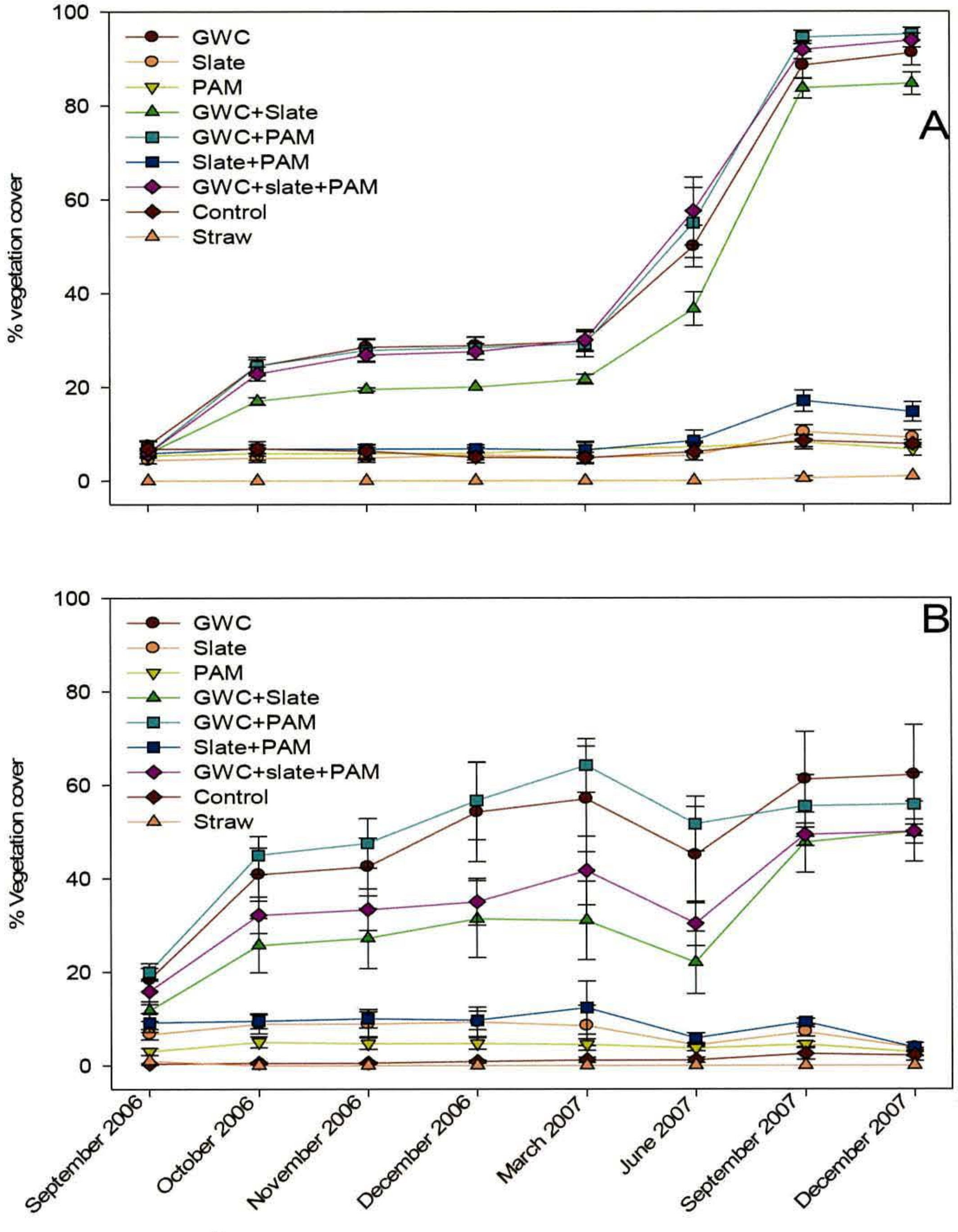


Figure 5.14 Vegetation cover development over the duration of field observations at site 1 (A) and site 2 (B) (bars = standard error of the mean)

Following the decline in vegetation cover at site 2 between March and June 2007 most treatments recovered to levels comparable, but generally slightly lower than, those found in March 2007. On the final sample occasion in December 2007 the most successful treatment at site 2 was GWC (mean =

62.2 %), and all treatments containing GWC at site 2 were significantly more successful than the control (all $p = 0.000$).

Unlike site 2, vegetation cover at site 1 increased steadily over the duration of the experiment. On the final sample occasion (December 2007) the most successful treatment was GWC + PAM with a mean vegetation cover (95.2 %) that was significantly (all $p < 0.005$) greater than that of all but two of the other eight treatments (GWC; GWC + slate + PAM). It was also an increase of >90 % in vegetation cover compared with that present prior to applications of soil-forming mixtures, within a period of less than 18 months, and 74.2 % more vegetation cover than that developed on the control plots (mean = 21.0 %).

At site 1 there were significant differences among treatments (all $p = 0.000$) on all but the first sample occasion (September 2006). At site 2 there were significant differences (all $p = 0.000$) among treatments on every sample occasion.

5.3.4.4 Biomass

There were significant differences in biomass among treatments at both sites and on both sample occasions (all $p = 0.000$).

The GWC + PAM soil-forming mixture consistently produced the greatest mean biomass of all treatments at both sites and on both sampling occasions, as shown in figure 5.15. Samples collected from GWC + PAM plots in May 2007 (sample occasion 2), had a mean dry weight of 159.35 g m^{-2} (site 1) and 114.26 g m^{-2} (site 2). Both figures are significantly higher (both $p = 0.000$) than the dry weight of biomass from control plots (site 1 mean = 25.06 g m^{-2}; site 2 mean = 11.24 g m^{-2}).

Excluding control treatments and straw additions, the lowest dry weight on both sample occasions came from PAM plots at site 1 (sample 2 mean = 20.24 g m^{-2}), and slate + PAM plots at site 2 (sample 2 mean = 26.49 g m^{-2}).

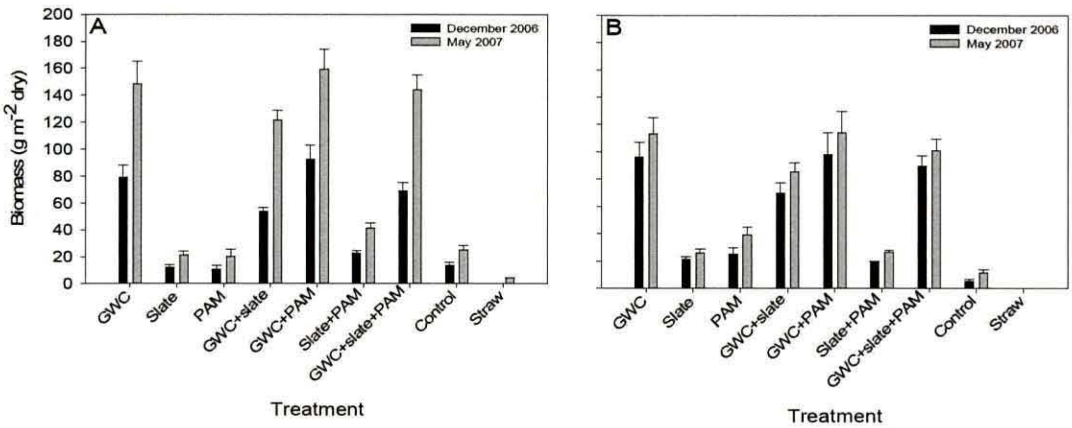


Figure 5.15 Dry weight of biomass at site 1 (A) and site 2 (B) harvested from experimental plots in December 2006 and May 2007 (bars = standard error of the mean)

5.3.4.5 Soil analysis

Information presented in this section is for site 1 only; it was not possible to collect adequate soil samples at site 2. Results are shown in figures 5.16 – 5.17.

There were no significant differences between pre- and post-application values of any of the soil properties measured.

In November 2007 there were significant differences among soil-forming mixture treatments in pH ($p = 0.013$) and calcium concentration ($p = 0.023$).

As table 5.10 demonstrates, soils sampled in November 2007 from plots treated with GWC had higher concentrations of all heavy metals except nickel than in the freshly-produced GWC.

Table 5.10 Heavy metals analysis of GWC (pre-application) and soils sampled in November 2007 from plots treated with GWC 17 months previously

Element	GWC		Soil	
	mg/kg	S.E.	mg/kg	S.E.
Cr	0.866	0.310	1.502	0.087
Ni	1.155	0.454	1.068	0.061
Cu	1.019	0.447	3.406	0.582
Zn	1.079	0.370	12.637	8.897
As	0.227	0.091	0.230	0.100
Cd	0.002	0.000	0.227	0.199
Pb	0.505	0.282	4.558	0.669

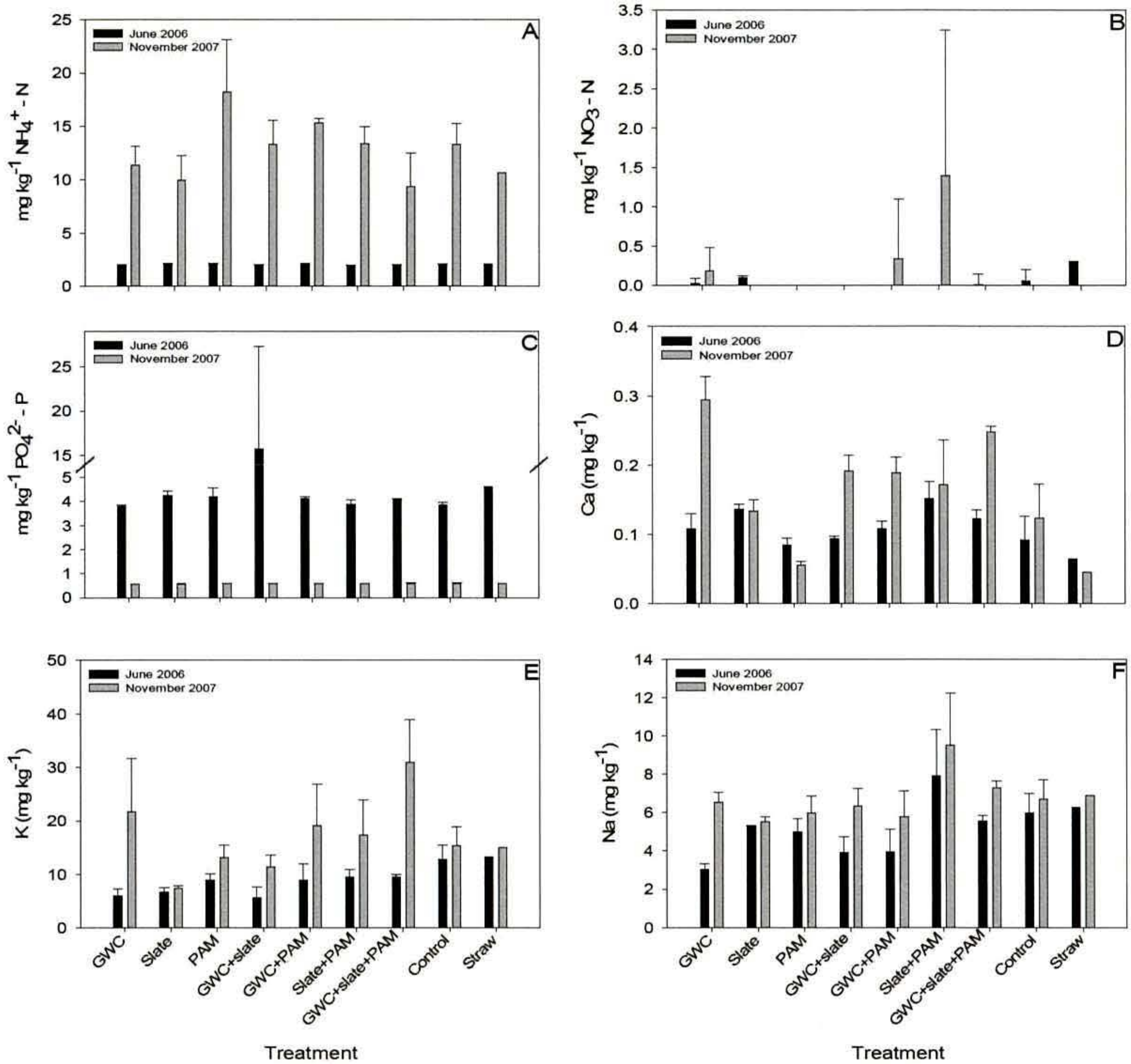


Figure 5.16 Water extractable ammonium (A), nitrate (B), phosphate (C) and cation (calcium (D), potassium (E), and sodium (F)) concentrations at site 1 prior to, and 18 months after, soil-forming mixture applications (bars = standard error of the mean)

5.3.4.6 Soil-forming mixture analysis

There were differences in pH among the three component materials; GWC was fairly neutral (pH 7.73), whereas both slate (pH 9.69) and hydrated PAM (pH 8.50) were quite alkaline. Mixtures of these constituent materials had a pH range of 8.31 – 8.84 (figure 5.18 (A)). In contrast, EC was variable across the soil-forming mixtures, ranging from 2.69 $\mu S\ cm^{-1}$ (PAM + slate) and 1270.00 $\mu S\ cm^{-1}$ (GWC + PAM).

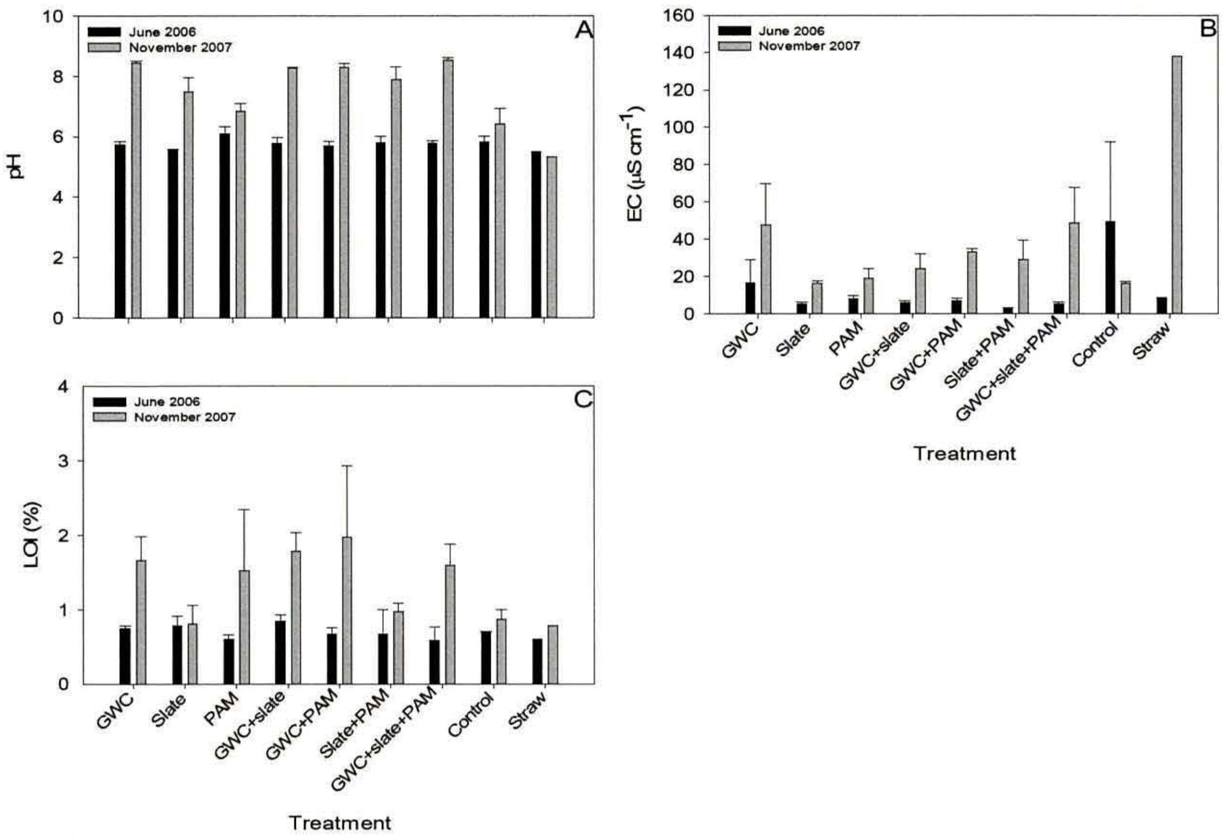


Figure 5.17 pH (A), EC (B) and LOI (C) at site 1 prior to and following soil-forming mixture applications (bars = standard error of the mean)

Neither slate nor PAM (and therefore also slate + PAM) contained any ammonium or nitrate (i.e. levels were below minimum detection limits); in contrast, GWC contained 43.78 mg kg^{-1} N ammonium and 124.38 mg kg^{-1} N nitrate (figure 5.18 (B)). The resulting soil-forming mixtures contained very low concentrations of these nutrients, GWC + slate containing most ammonium (1.89 mg kg^{-1} N) and nitrate (1.89 mg kg^{-1} N). All materials and mixtures contained low concentrations of phosphate, the greatest concentration was found in PAM (0.56 mg kg^{-1} P).

Water extractable calcium concentrations in all soil-forming mixtures were very low (figure 5.18 (C)). Concentrations of both potassium (637.38 mg kg^{-1}) and sodium (290.95 mg kg^{-1}) in GWC were much higher than in both slate and PAM. However, PAM did contain an appreciable quantity of potassium (94.62 mg kg^{-1}); as a result, mixtures containing these two materials had medium-high potassium concentrations (figure 5.18 (C)).

Loss on ignition data for GWC is presented in section 5.3.2.3 (figure 5.6 D), and slate and PAM are inorganic compounds.

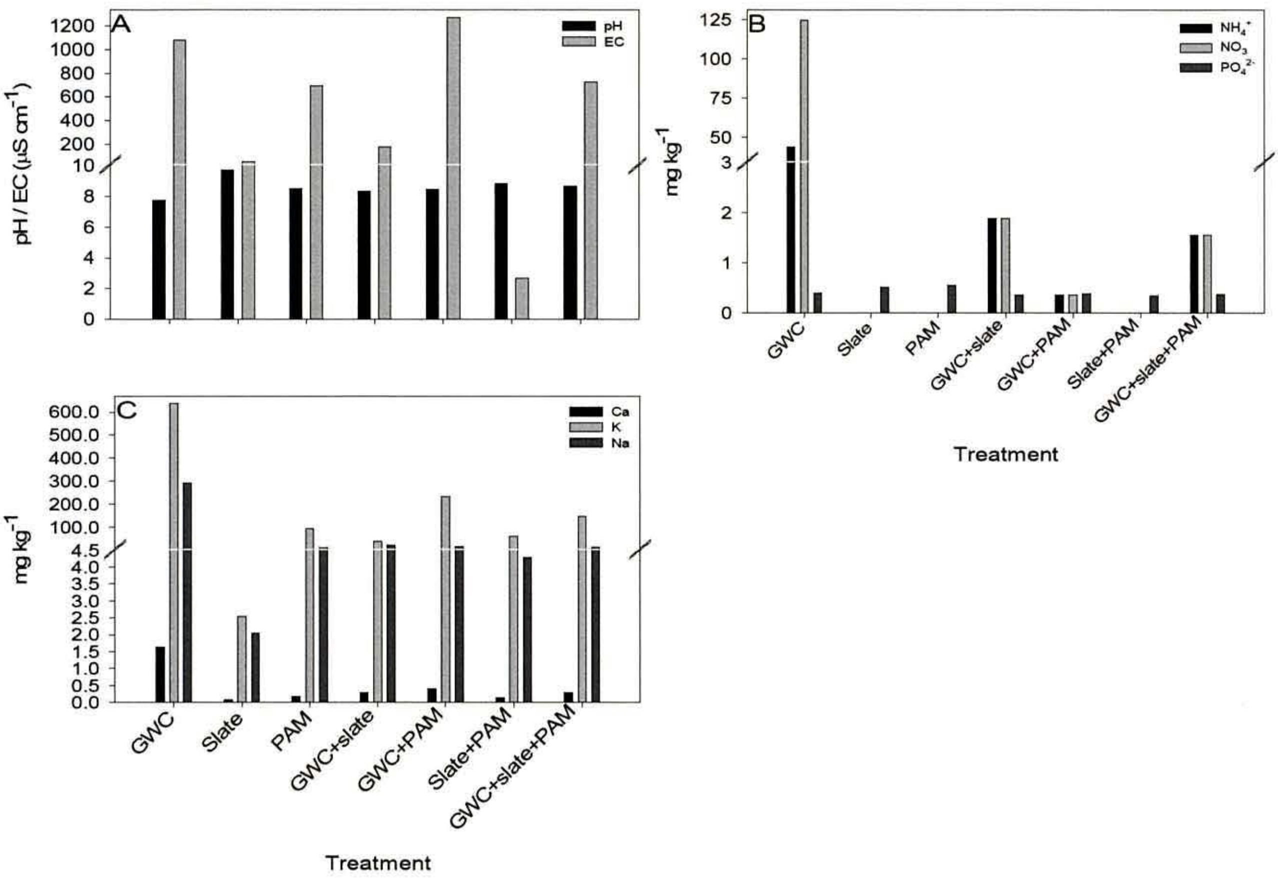


Figure 5.18 pH and EC (A), water extractable ammonium, nitrate, and phosphate concentration (B), and water extractable calcium, potassium, and sodium concentration (C) of soil-forming mixtures prior to field application

N.B. Slate, PAM and Slate+PAM soil-forming mixtures contained concentrations of ammonium and nitrate below the minimum detection limits

5.4 Discussion

Unlike work carried out by Williamson *et al.* (2009) for example, where the objective was to restore slate waste to woodland, which is considered to be the climax community type for some of the land surrounding Penrhyn Quarry, this set of experiments focussed on providing conditions that would facilitate rapid and sustained vegetation cover to act as a nurse crop and stabiliser for later colonisation. This will provide the building blocks for habitat development and progression to the climax community, as opposed to recreating that community immediately, as in the work of Williamson *et al.* (2009). Although it is recognised that trees are able to act as primary colonists on slate waste (Sheldon, 1975; Sheldon and Bradshaw, 1977; Rowe *et al.*, 2005; Rowe *et al.*, 2006), it will still be many years before the established trees are able to influence their immediate surroundings to such a degree that further plant growth is benefited. Additionally, unless significant amendments are utilised (e.g. the provision of a 750 mm layer of boulder clay by Williamson *et al.* (2009)), the likelihood of achieving successful tree development is low.

To achieve any degree of vegetation cover on the slate waste tips at Penrhyn Quarry, several layers of functionality first needed to be restored, namely water-holding capacity and a nutrient pool.

Sheldon (1975) states that slate waste tip surfaces predominantly experience drought conditions despite being located in mountainous regions typically receiving in excess of 2000 mm of precipitation *per annum*. Toy (1979, cited in Jim, 2001) states that water shortage is a major cause of re-vegetation failure. Therefore, to effectively and successfully carryout the ecological restoration of slate waste tips, a method of increasing the availability of water is required. Following the work carried out by Rowe *et al.* (2005) and Holliman *et al.* (2005) the use of super absorbent polymers (SAPs) and similar materials was investigated.

Of the materials tested for their water-holding capacity (WHC), the super-absorbent polymers performed much better than the wallpaper paste.

Powdered sodium-polyacrylate (Na-PA) was found to have the greatest WHC, followed by granular polyacrylamide (PAM) and then powdered PAM. However, the decision was made to focus on PAM in later experiments. If Na-PA had been available in both powdered and granular forms, this would have been the SAP on which further experimental tests would have been conducted. However, because it was not available in a granular form it was decided to use PAM and compare its properties in granular and powdered forms.

It was considered important that PAM in both of its forms (powdered and granular) was tested, because all previous work at Penrhyn Quarry has used granular PAM, and problems have been encountered with the material. For example, Rowe *et al.* (2005) reported that UV exposure and extremes of temperature caused granular PAM to be ineffective in promoting vegetation survival or growth; also adding that finer-grade PAM might suffer less from drying and shrinking effects, and that mixing either grade of PAM with other soil-forming materials might alleviate previously encountered problems. Therefore, it was hypothesised that using a different form of the same material might solve these problems.

Despite the better performance of powdered PAM in some respects (it maintains WHC ability better over several wetting-drying cycles, and becomes hydrated much faster due to smaller particle size of dry material), it was decided to use granular PAM in the soil-forming mixtures. This decision was not solely based on the greater WHC of granular PAM; potential problems arising from *in situ* quarry conditions were also considered. It was hypothesised that if powdered PAM were applied to the quarry surfaces in hydrated form, drying by wind and solar irradiation, would mean that the material would rapidly return to its dry, powdered state and be susceptible to wind-blown distribution, which would be undesirable in terms of a long-term field study or when applied to a specific location for ecological restoration. It was considered that granular PAM would be less susceptible to wind-blown movement and therefore more suitable for applying to slate waste tips for restoration purposes.

In addition to considerations of the *in situ* behaviour of PAM, a greater pool of knowledge of using PAM in land restoration was available, for example, work carried out at Penrhyn Quarry by Rowe *et al.* (2005) and Holliman *et al.* (2005). These publications include information on the environmental degradation and toxicity potential of PAM; these details were not readily available for Na-PA, and their determination experimentally was not possible within the timeframe of this research.

The second major limiting factor to vegetation development on slate waste tips is the lack of nutrient availability. This fact is acknowledged in the literature; Williamson *et al.* (2009) state that tree establishment on slate waste is limited by nutrient availability, and Rowe *et al.* (2006) found that by improving the nutrient availability in slate waste tip surfaces, both plant cover and biomass accumulation increased. Therefore, experiments were carried out to identify a suitable material for improving nutrient availability on slate waste. Past restoration work (e.g. Sheldon and Bradshaw, 1975) used large quantities of peat to add this nutrition, however this is no longer environmentally acceptable and current research (e.g. Curtis and Claassen, 2007a) is more focused on using sustainable materials such as composts made from waste organic material, referred to as green waste compost (GWC).

Six GWCs were sourced from north Wales, and an extensive range of bioassays, physical and chemical tests were carried out on them to determine their relative suitability for addition to slate waste tips to promote vegetation establishment and development. Table 5.7 shows how the GWCs scored in relation to one-another across all tests carried out. Conwy and Wrexham sourced GWCs were jointly ranked first, out-performing all other GWCs tested.

Some key attributes demonstrated by Conwy GWC explain why it was the joint best-performing GWC of the range tested. For instance, where Flintshire GWC contained more than 25 % (by mass) particles >10 mm diameter, and those sourced from Anglesey, Denbighshire, and Gwynedd, all contained more than 15 % (by mass) particles >10 mm diameter, Conwy GWC contained no particles >9 mm. High percentages of large particles indicate either poor quality-control when screening green waste or the use of large-grate screens, and can result in large numbers of contaminants in the GWC (this was observed in this study, where composts sourced from Conwy and Wrexham contained far fewer contaminants than the other sources (data not presented)).

In addition to its good textural quality, Conwy GWC exhibited both high loss on ignition (LOI) and low electrical conductivity (EC). LOI is a measure of the organic content of soils and composts, and high values represent high levels of organic materials. Conwy GWC had a LOI value in excess of 50 %, much higher than that of all other GWCs tested (e.g. Wrexham LOI 22.00 %). Organic-rich materials offer both high water-holding capacity and good long-term plant nutrition, both of which are traits required for slate waste amendment. In contrast to LOI, high EC values are not considered to be beneficial, as they indicate high concentrations of total dissolved salts or ions. The highest EC recorded was in the Flintshire GWC (4240 μS cm^{-1}) and the lowest in the Conwy GWC (1080 μS cm^{-1}). High concentrations of salts, as found in the Flintshire GWC, are likely to cause high rates of seedling mortality and negatively affect long-term plant health and growth.

It was decided to use Conwy GWC rather than the equally-ranked Wrexham GWC because Wrexham GWC is produced and sold as a commercial product by R. A. & C. E. Platt Ltd., whereas Conwy GWC is produced by a local authority and is available for collection free of charge as a sustainable, recycled material. Thus Conwy GWC is locally-sourced, available in large quantities and is economic, all of which are key attributes for large-scale land restoration practices. Williamson *et al.* (2009) advocate the use of locally available materials as a low "carbon footprint" approach to restoration ecology.

The effect of fertiliser inclusion into soil-forming mixtures was also examined using bioassays with a restoration seed mix (see section 5.2.3). Germination showed some unexpected results; the general trend (true in all but two of the eight treatments (GWC + slate, and GWC + PAM)) was that mean germinant counts were greater in non-fertilised treatments than in the fertilised counterparts. This suggests that fertiliser inhibits seed germination, an observation that has previously been recorded (Sheldon and Bradshaw, 1977; Roberts and Bradshaw, 1985). It is unclear whether or not the suppression of emergence was merely short-term, as observations were only made for a period of seven days following sowing.

Despite the greater germination in non-fertilised treatments, results presented in figure 5.9 show clearly that all fertilised soil-forming mixtures produced greater biomass than non-fertilised treatments; only in the control did the non-fertilised treatments outperform the fertilised. It may be that, despite initial suppression of germination, fertilised treatments did achieve a germinant count equal to that in non-fertilised treatments. Alternatively, it could be that the higher nutrient levels (0.64 g granular, slow-release, mineral fertiliser per seed tray) in fertilised treatments allowed those plants present to develop greater biomass than in non-fertilised treatments.

It was therefore considered that provision of small quantities (20 g m^{-2} (equivalent to 57.143 kg N ha^{-1}, 85.714 kg P ha^{-1}, and 57.143 kg K ha^{-1})) of granular, slow-release, mineral fertiliser were sufficiently beneficial to vegetation establishment and development that it should be included in all soil-forming mixtures for field scale experimentation. Walker *et al.* (2004b) discuss the use of mycorrhizal inoculants as an alternative to using chemical fertilisers in restoration schemes. This approach may be better suited to restoration work taking place in areas that are sensitive to nutrient additions, for example, close to watercourses that are susceptible to becoming eutrophic.

Applications of soil-forming mixtures to slate waste tip surfaces at Penrhyn Quarry were carried out in July 2006. The application of soil-forming materials was intended to be carried out with a purpose-made application chute (see appendix 6 for full details), but this method failed and applications were made by hand. At both sites the highest germinant counts were present in mixtures including GWC. This is in contrast to results presented by Sellers *et al.* (2001), where the authors report that green waste compost amelioration might have suppressed oilseed rape and field bean emergence.

Early vegetation cover development was greatest at site 2 despite soil-forming material applications at site 1 occurring approximately 2 – 3 weeks earlier than at site 2, and the fact that the surface substrates had a greater proportion of fines material at site 1. This difference may reflect the different aspect and altitude of the two sites. Site 2 had an aspect of 322° (~NW) and altitude of 189 m, while site 1 had an aspect of 111° (~ESE) and an elevation of 333 m. As a result site 2 received far less direct solar radiation than site 1, an important microclimatic factor in slope restoration (Cano *et al.,* 2002), and resultantly stayed cooler and damper for longer; these are conditions more conducive to efficient biomass production. This result is similar to the findings of Bochet and Garcia-Fayos (2004) and Paschke *et al.* (2000), who found that north-facing road cut slopes achieved significantly greater vegetation cover than south-facing slopes.

It was observed that where hydrated PAM was applied as an individual restoration treatment, movement of PAM into interstitial spaces in the surface layers of slate waste tips occurred. This indicates that a degree of connectivity between surface layers and the main matrix of slate waste tips was being created. Connectivity between the surface and the inner portions of slate waste tips is important in the restoration of vegetation and habitats; without it no root penetration will occur and vegetation will remain as a blanket layer on the tip surface, susceptible to atmospheric conditions and grazing pressures. It is likely that similar movement of soil-forming materials took place with other mixtures, although it was not as easily observed.

Vegetation establishment at site 2 suffered during April 2007 from consistently high air temperatures and low levels of precipitation. Site 2, unlike site 1, received soil-forming material applications onto large blocky slate waste with minimal fines content. The very warm, dry weather conditions caused the relatively thin (75 mm) soil-forming applications to dry out considerably from both above and below. Site 1 was not affected nearly as badly, because the greater percentage of fines prevented excess desiccation. As demonstrated by figure 5.14, total vegetation cover at site 2 suffered badly and because of dieback, decreased in coverage. By November 2007 the greatest total vegetation cover at site 2 had developed in GWC treatments (mean = 62.2 %). Although this figure does not represent what may have been achieved if there had been more frequent precipitation events, it does represent significantly (p = 0.000) greater vegetation cover than that present in the experimental control (mean = 7.7 %).

Species richness recorded in the control treatment at site 1 was higher than in the slate and PAM treatments. The control plots were not covered with a 75 mm depth of soil-forming mixture, and all species *in situ* prior to applications were still visible and could be recorded. In the other treatments these plants were completely covered by the soil-forming mixtures. Species richness in control plots was, in essence, instantly increased by the addition of the ten species in the applied restoration seed mixture. Site 1 had a small number of established plants prior to soil-forming applications; in contrast, site 2 had very few established plants and more clearly shows the success of species recruitment in each of the experimental treatments. It can be seen from figure 5.12 (B) that all soil-forming material treatments (except straw) showed more species recruitment than the control at each sample time from December 2006 until the final sampling in November 2007. This suggests that any application of restorative materials to un-vegetated, blocky slate waste is beneficial for species recruitment.

Except for the straw treatment at site 1, all treatments at both sites exhibited much greater species richness at the final sampling (November 2007) than in the baseline survey carried out in June 2006 (figure 5.13). This demonstrates the success of soil-forming material applications for increasing biodiversity on slate waste tips.

Observations made in August 2008 of vegetation on experimental plots at both site 1 and site 2 demonstrated that vegetation development is continuing without any signs of senescence or dieback (data not presented) (plate 11). This represents continued vegetation cover for a period of 25 months. It was also observed that the vegetation was becoming self-sustaining; for example, grasses growing in experimental treatments were observed to be flowering and seeding during surveys in 2007 and when re-visited during 2008 (data not presented). This is in contrast to the situation reported by Jochimsen (2001), where grasses and legumes typically used in landscape restoration activities, quickly become moribund and ineffectual as a nurse crop.

Plate 11 Established vegetation at site 1 two years after soil-forming material applications

Additional observations made in August 2008 showed that the surface of the slate waste tip had been substantially stabilised. Prior to soil-forming material applications on both experimental sites, the surface of the slate waste tips was very mobile when walked on. However, by August 2008 it was possible to walk upright, both up and down slope, without causing any disturbance of site substrates, as a result of the established vegetation. This stabilisation of substrates is considered to be a significant benefit; it has not only improved site safety, locking in loose potentially hazardous slate waste, but has also stabilised the site so that it is suitable for secondary colonisers.

Biomass harvest data demonstrate how productive each soil-forming material is. If materials applied for restorative purposes fail to produce significant amounts of biomass, the turnover of organic matter will be low and there may be nutrient leaching and erosion of the restorative soil-forming materials. In combination these factors will reduce the likelihood of further plant/species recruitment, as site stabilisation by pioneer or nurse crop plants will not have taken place, and secondary colonisers may fail to establish. However, Gretarsdottir *et al.* (2004) point out that even deteriorating vegetation established from restoration seeding can act as a nurse crop, providing safe micro-sites, soil stabilisation and increased moisture, therefore facilitating colonisation by native plants.

The most productive soil-forming treatments were those containing GWC. In the absence of GWC, productivity was poor at both experimental sites; for example, PAM and slate applied individually at site 1 were less productive than the experimental control when sampled in December 2006 and May 2007. This indicates that PAM and slate additions onto slate waste tips with a high proportion of fines material is not only unnecessary but in fact suppresses biomass production. This is in accordance with results presented by Wilson *et al.* (1991), who found that a cross-linked polyacrylamide was slightly detrimental to *Acacia tortilis*, *Prosopis juliflora*, *Terminalia brownii* and *Terminalia prunoides* growth.

At site 2, all soil-forming material applications except straw, showed greater biomass production than the control plots, providing evidence that the addition of any soil-forming materials to blocky slate waste lacking any fines content is beneficial for vegetation establishment.

Analysis of heavy metals was carried out on GWC (as received direct from the composting plant) and on quarry soils collected from under GWC soil-forming treatment plots (i.e. sample taken from 0-10 cm depth of quarry soil surface beneath applied soil-forming GWC treatment) 17 months after applications. These were the only samples analysed because the only concerns involving heavy metal presence were with the use of GWC. If significant concentrations of heavy metals were present in the GWC it would be reflected in the quarry soils to which it had been applied. The results show that of all heavy metals tested for, only nickel concentrations were higher in the GWC than in the quarry soils (1.155 mg/kg compared to 1.068 mg/kg). Concentrations of all other metals tested for (As, Cd, Cr, Cu, Pb and Zn) were higher in quarry soils than in the GWC; therefore it is unlikely that the introduction of soil-forming material mixtures containing GWC have any associated risks of metal toxicity or contamination. However, all heavy metal concentrations recorded in the GWC provided by Conwy County Council are lower than legislative limits set out in the PAS 100 (WRAP, 2003b), and as concluded by Tandy *et al.* (2009), this is unlikely to cause any negative effects if used in land restoration projects.

No formal surveys of faunal presence or behaviour were carried out during this experiment, though observations were recorded. Table 5.11 shows a list of birds and mammals recorded on site throughout the period of experimentation. The addition of the soil-forming mixtures provided islands of soft substrate inside the typically hard-surfaced quarry complex; this appeared to be favourable for certain bird species that forage for soil-borne invertebrates, for example chough (*Pyrrhocorax pyrrhocorax*) and ring ouzel (*Turdus torquatus*), which were observed foraging on experimental plots at site 1 on several occasions.

Several regurgitated pellets from kestrels (*Falco tinnunculus*) and buzzards (*Buteo buteo*) were collected at both site 1 and 2. Fur and bones of unidentified small mammals were present in the pellets; it might be, therefore, that the vegetation developing on the experimental sites was attracting small mammals to feed on the increased abundance of seed and invertebrates, which in turn might be accompanied by increased presence of raptors such as kestrels.

Table 5.11 Fauna observed at Penrhyn Quarry (June 2006 – November 2007)

Class	Species	Common name
Aves (bird)	*Accipiter nisus*	Sparrow hawk
Aves (bird)	*Buteo buteo*	Buzzard
Aves (bird)	*Falco columbarius*	Merlin
Aves (bird)	*Falco pereginus*	Peregrin falcon
Aves (bird)	*Falco tinnunculus*	Kestrel
Aves (bird)	*Turdus torquatus*	Ring ouzel
Aves (bird)	*Saxicola rubetra*	Winchat
Aves (bird)	*Saxicola torquata*	Stonechat
Aves (bird)	*Oenanthe oenanthe*	Wheatear
Aves (bird)	*Troglodytes troglodytes*	Wren
Aves (bird)	*Erithacus rubecula*	Robin
Aves (bird)	*Motacilla alba*	Pied wagtail
Aves (bird)	*Motacilla cinerea*	Grey wagtail
Aves (bird)	*Pyrrocorax pyrrocorax*	Chough
Aves (bird)	*Pica pica*	Magpie
Aves (bird)	*Corvus corax*	Raven
Aves (bird)	*Garrulus glandarius*	Jay
Aves (bird)	*Sylvia borin*	Garden warbler
Aves (bird)	*Apus apus*	Swift
Aves (bird)	*Delichon urbica*	House martin
Aves (bird)	*Hirundo rustica*	Swallow
Aves (bird)	*Larus argentatus*	Herring gull
Aves (bird)	*Columba palumbus*	Wood pigeon
Aves (bird)	*Scolopax rusticola*	Woodcock
Aves (bird)	*Cuculus canorus*	Cuckoo
Aves (bird)	*Ardea cinerea*	Heron
Mammalia (Mammal)	*Oryctolagus cuniculus*	Rabbit
Mammalia (Mammal)	*Vulpes vulpes*	Fox

In a similar manner, the islands of vegetation developing at experimental sites 1 and 2 also influenced large mammal activity. For example, significant quantities of rabbit (*Oryctolagus cuniculus*) faecal matter were deposited on the experimental plots and perhaps as a result of the concentrated nature of this rabbit grazing subsequent carnivore presence increased. For example, numerous fox (*Vulpes vulpes*) scats were identified on or very close to both site 1 and site 2.

Unlike previous work carried out at Penrhyn Quarry that relied upon extensive electrified boundary fencing (Rowe *et al.,* 2005) or a combination of fencing materials to actively exclude both sheep and rabbits (Williamson *et al.,* 2009), this research was intended to be low-cost (see Appendix 7 for economic assessment of soil-forming material applications) and low-intervention. Fencing introduces additional and substantial expense to restoration projects, and therefore no fencing was used. The results have demonstrated that restoration success can be achieved under these unfenced conditions.

As a result of the abundant herbivory taking place on the experimental plots it appears that a feedback mechanism has been instated. The substantial levels of faecal matter deposited by both sheep and rabbits, introduces organic matter and nutrients, benefiting plant development, which subsequently attracts more herbivory. Sheldon (1975) observed this, noting that colonising plants produced localised dense patches of vegetation where rabbit droppings occurred, and pointing out that this "indicates the severe lack of nutrients in the slate". This herbivory therefore plays an important role in the development of a self-sustaining habitat.

The methods developed during this research study could be easily adapted and incorporated into several current quarrying practices. Firstly, for example, at present quarry vehicles travelling from rock-blasting faces towards processing sheds are fully laden with slate; however, on the return journey to the blasting faces, these same vehicles carry no load. With minimal intervention, this system could be adapted to transport restoration materials to the slate waste tips. Material such as green waste compost could be loaded into vehicles returning to the blasting faces; a short detour would then allow this material to be tipped down slate waste tip slopes, providing a shallow surface covering of organic material. If this were co-tipped (i.e. one load of slate waste followed by one load of compost), organic material would begin to accumulate throughout the structure of slate waste tips, and connectivity between the surface and inner of waste tips would be achieved, offering greater potential for root intrusion. Secondly, during summer months or extended periods of dry conditions, a large vehicle patrols quarry roads discharging huge quantities of water; this is intended to reduce airborne dust levels. The potential exists to incorporate restoration materials as identified in these studies with this machinery to efficiently distribute soil-forming mixtures.

For example, on days when dust suppression is not required, the water-spraying nozzle mounted on the vehicle could be swapped for a more robust attachment, and it might then be possible to spread liquids containing solids, such as GWCs or SAPs. The vehicle could therefore continue patrolling the quarry roads whilst discharging restoration materials down slate waste tip slopes in an efficient manner.

5.5 Conclusions

Current restoration planting at Penrhyn Quarry is not considered to be achieving acceptable levels of success; various authors (Rowe *et al.*, 2005; Rowe *et al.*, 2006; Nason *et al.*, 2007; Williamson *et al.*, 2009) have tested different restoration methods, but all have involved significant levels of landscape manipulation. By using a range of materials diverted from waste-streams destined for land-filling, considerable levels of vegetation development have been achieved on slate waste tip surfaces within 18 months, using low input, sustainable methods.

Granular polyacrylamide (PAM) gel supplied by Biotechnica Ltd. and green waste compost (GWC) produced by Conwy County Council were identified as the most suitable materials to add water-holding capacity and plant-available nutrition to slate waste tips. These materials were tested in a number of combinations to find the optimum mixture for promoting vegetation establishment and development in field conditions on slate waste tips at Penrhyn Quarry. A 1:1 (v/v) mixture of hydrated PAM and GWC, supplemented with 5 g m^{-2} granular inorganic slow release fertiliser, was found to be the most successful combination of soil-forming materials.

Measurements of vegetation cover, species richness, biomass production and soil chemical and physical qualities in plots amended with this soil-forming mixture demonstrated consistent improvements over the control treatment.

It was found that vegetation at site 2 developed more quickly than at site 1, probably in response to the different site aspects. To achieve the quickest and most dense vegetation coverage at restoration sites it is recommended, where possible, to focus on areas with a northerly aspect and a high proportion of fines material in the surface layers. This will rapidly create islands of vegetation and a degree of habitat connectivity across the restoration site. Even in the most difficult situations within a quarry complex for example, where no fines material is present and slope aspect is less than favourable, vegetation can be successfully established.

The maximum vegetation cover recorded was 95.2 % (GWC + PAM, site 1, November 2007); the greatest amount of biomass produced was 159.35 g m^{-2} dry biomass (GWC + PAM, site 1, May 2007); and the highest species diversity recorded was 22.3 (GWC + PAM, site 1, November 2007). These figures demonstrate considerable and significant increases in vegetation presence over that recorded prior to soil-forming mixture applications.

The methods developed during these studies demonstrate that ecological restoration of slate waste tips can successfully be achieved in short time periods, with limited resources and budget, and minimal intervention. The established vegetation, albeit not representative of target habitat types (NVC H8, H9, H10 and W17), is self-sustaining and provides conditions conducive to subsequent vegetation colonisation and the development of habitat of high conservation value.

5.6 Further work and recommendations

Ideally WHC tests should have been done with a greater variety of SAP materials. This would have provided more data on the most suitable material for water provision in habitat restoration activities. Similarly, several additional rounds of wetting and drying cycles could have been carried out to demonstrate how a SAP reacts to conditions encountered when applied to land and exposed to atmospheric conditions.

Germination rates of *Betula pendula* were much lower than expected in the GWC bioassays carried out in 2005 / 2006. Thus data were gathered from very small populations and offer a limited representation of the growth response of this species to the GWCs tested. It is suggested, therefore, that more appropriate species, greater seed quantities or more rigorous seed pre-treatment or pre-test checks of seed viability should be used in future trials of this nature.

The common perception of GWC is one of overall low inherent quality. This may have once been true, but evidence from these experiments suggests that it is no longer valid. Green waste compost production invariably suffers from seasonality due to feedstock type and availability. However, discussions with composting operators suggest that seasonality does not necessarily play a significant role in GWC quality. Compost producers who are capable of supplying the public, produce large quantities of material; for example, Conwy County Council's site at Dolgarrog receives more than 100 tonnes of green waste per month. The turnover period for the finished GWC product is approximately 13 – 20 weeks. In such situations GWC feedstock will rarely consist primarily of one vegetation type, and mature, attentively-processed compost will be produced from heterogeneous source material. To fully assess this heterogeneity, germination and growth bioassays could have been carried out on several occasions throughout the year; unfortunately time constraints did not allow this.

The decision to include slate in experimental trials in this chapter was a pragmatic one, based on material availability, the theory that the slate would buffer the soil-forming mixtures, closely matching them to the surrounding environmental chemistry, and because of the use of slate in past restoration activities at Penrhyn Quarry (Rowe *et al.,* 2005). With hindsight it is clear that the same level of initial experimentation was not carried out with this material as with SAPs or GWCs. But given the volume of waste slate produced annually it was considered that if a beneficial end use could be developed for this material rather than depleting materials sourced from other natural resources outside the local area, it would be desirable.

Manual applications of soil-forming mixtures to 50 experimental plots required one whole calendar month to complete, this affected later germination surveys. Because one treatment might have been applied on 1st July 2006 and another on 8th July (for example), at the first germination survey on 10th July the first treatment would have been *in situ* for a total of ten days, while the second treatment would have only been *in situ* for two days. Records of application dates were made, and considered when analysing the data; days since spreading was used as a covariate in univariate analyses of variance. However, normality testing showed that the germination dataset was non-normally distributed; non-parametric analysis was carried out and it was no longer possible to include data on days since spreading. If time had not been a limiting factor during soil-forming material applications, all germination surveys would have been carried out at the same times after the start of treatments.

Soil samples were collected on only two occasions: for analysis of soil chemical properties prior to soil-forming material applications, and 18 months after applications. This was intended to demonstrate the nutritional influence of restorative treatments upon site soil chemistry. In hindsight, perhaps analysis of spreading materials should have been carried out at both the onset and the termination of experimental observations, as this might have therefore provided evidence of nutrient movement from restoration materials into site substrates. More regular sampling of soils would have provided better interpretation of restoration material behaviour and influence upon site soil chemistry.

However, soil sampling at the experimental sites was problematic due to the fact that samples had to be collected from beneath the soil-forming materials (i.e. samples of the *in situ* substrates). Sampling therefore caused significant disturbance to experimental treatments, the more so because not every sampling point provided sufficient fines material for soil analysis to be carried out upon. Inspection pits had to be dug through soil-forming mixture layers to determine if a sample was available; if not, another pit was dug. If, for example, monthly soil samples had been collected, damage to the developing vegetation would have been substantial and would have negatively impacted the overall results of the experiment. Additionally, regular movement up and down the slate waste tips at sites 1 and 2 in the months directly following material applications caused significant disturbance of site substrates.

One possible flaw in soil analysis is the seasonality of sampling. Ideally, soil samples should be collected at similar times in the year. The influence of weather conditions at sampling times may impact upon the properties of each soil sample. If it had been possible to carry out experimental observations for a period of two whole years, for example, this sampling regime would have been done.

If this experiment were to be repeated, it is recommended that foliar analysis of the biomass produced in each experimental treatment be carried out. This data would supplement information from soils samples and vegetation sampling would be less disruptive than soil sampling. It is also recommended that future studies should include assessment of soil microbial populations. This might provide a valuable insight into the maturation and stabilisation of developing soils at restoration sites.

Pitfall trapping was carried out during June 2006; two traps were set per experimental block, at both site 1 and site 2 ($n = 6$ per site) for 10-day periods.

This was intended to provide a baseline set of data on invertebrate populations prior to soil-forming mixture applications. Substantial numbers of invertebrates were trapped and subsequently transferred into vials of 80 % ethanol for storage until identification. Unfortunately, during storage and before identification had been carried out, pop-top lids on several of the storage vials became dislodged and the alcohol evaporated from the samples; consequently the invertebrate samples decomposed. The decision was therefore taken to abandon invertebrate sampling for these experiments. Pitfall trapping is however recommended; visual observations on both experimental sites appeared to show that the invertebrate populations increased with time.

At the onset of this experiment, dialogue with quarry managers confirmed that site meteorological records would be available. This was considered important information because of the severely challenging conditions on site, which are either lessened or made increasingly severe by the prevailing weather situation. However, it later became apparent that these meteorological data were not available, which was a considerable disappointment. Data provided by First Hydro (FHC, undated) were used in place of on site data, but they were for Llanberis, which is located approximately 6 km away (although it fairly similar micro-climatic conditions).

If circumstances had allowed, earlier application of soil-forming mixtures (at the start of the growing season of 2006, for example) would have been desirable. However, by chance the climatic conditions were sympathetic following the experimental applications. July 2006 was extremely hot and dry (air temperature in direct sunlight on experimental site 2 recorded at 52°C), whereas August 2006 was dominated by frequent, intense rainfall events, providing higher than average precipitation. As a result, germination and establishment within experimental treatments were successful.

If this experiment had been established earlier in 2006 it would have provided a longer time period over which experimental records could have been made.

Ideally, this type of experiment would benefit from repeated observations over a period of 5 – 10 years, because the plant community established during the period of experimental observations is essentially a nurse crop, and it is the succession of the plant communities on the experimental treatments that truly indicates the success of restoration work.

It was intended to develop an effective and efficient method for applying restoration materials to slate waste tips (discussed further in Appendix 8). Due to a series of problems with compost delivery and pump procurement, followed by pump failure and reaching the end of the research period and budget, this work could not be completed. This work might have provided an insight into how practicable, efficient and economically viable the restoration of disturbed land using the materials identified throughout these studies could be on a larger scale.

6 General discussion and conclusions

The restoration of post-industrial land is generally a complex practice, typical complications can include the presence of contaminants such as heavy metals and hydrocarbons, or by other industrial legacies such as damaged or missing substrates, or accumulations of waste and spoil. Slate waste tips are accumulations of a natural material, and fortunately do not suffer from the added complication of contaminants. However, despite their relative chemical innocuousness, they have inherent properties that act as obstacles to both natural colonisation and to conventional methods of ecological restoration.

Current climate change models (UKCIP, 2009) indicate that global warming will bring about mean annual air temperature rises of 1-3 °C by 2020 and up to 6 °C by 2080 in the north Wales region. If these temperature increases do occur, conditions on the slate waste tips at Penrhyn Quarry, for example, will become ever more extreme and challenging to plant colonisation and development. Such conditions are likely to severely impact current restoration practices such as tree planting, with sapling mortality expected to increase from current levels of approximately 25 – 50 % to perhaps 75 – 100 %. This increases the need for alternative restoration practices to be devised and put into place, so that any early vegetation development that does take place is provided with adequate water and is protected from over-heating and desiccation.

The challenge that needs to be overcome in order to reinstate vegetation on slate waste tips is the provision of substrates that have adequate water-holding ability and plant nutrient levels. If the cost (both financially and ecologically) of restoration projects was not a limiting factor, this could be done without difficulty; large amounts of topsoil, peat or other materials could be imported and methods such as hydro-application or direct tipping could be used to establish a topsoil layer conducive to successful plant establishment and development. However, given the nature of industry and commerce, the focus is mainly on profit margins, and costly restoration schemes are not viewed with much enthusiasm.

This raises some questions. For example, what price or cost should be placed upon the restoration of damaged landscapes? Is it justifiable to destroy or damage ecosystems in the process of exploiting natural resources, and then skimp on making good the resulting damage? Can a price tag be placed on a non-profitable ecosystem or habitat? Is the conservation or ecological value of a site in itself sufficient to justify a particular level of costly aftercare? Addressing these questions is vital to the potential outcome and success of any restoration project, and without at least some degree of consideration, industry will continue to negatively impact on the natural world.

Similarly, is it reasonable or acceptable to allow industry to carry out restoration works that merely represent a minimum token gesture towards habitat restoration? In terms of its ecological value, a poorly thought-out restored habitat or ecosystem is in no way analogous to that present prior to disturbance. However, it is not feasible to reinstate thousands of years of ecosystem development in a restoration scheme, even if it is thorough, well funded and followed through in the long-term. There is a need to clarify the level of effort that is acceptable in contributing to the repair of industrial damage.

An example of what can be achieved in terms of post-industrial restoration when resources and finances are not a limiting factor is the Eden Project near St. Austell, Cornwall, UK. This is an extreme example of how post-industrial site reclamation can be carried out. Although this is not an example of ecological restoration (despite introducing much plant life to the abandoned china clay pit in which it is located), it is nonetheless a very good example of successful quarry restoration. It also effectively demonstrates the stark contrast between what is achievable when the primary aim of restoration is a commercial enterprise and what is attainable in ordinary ecological restoration schemes; the difference in availability of finances and resources is substantial.

In his presentation at the 2005 Society for Ecological Restoration (SER) World Conference in Zaragoza, Spain, Robert Costanza made some interesting points regarding the contribution and value of natural ecosystems to human society. He argued that natural ecosystems add value to the global economy, in that they add to "the sustainable well being of people". Costanza went on to state that there is substantial and growing evidence that natural systems make a major contribution to human well-being; Costanza *et al.* (1997) estimate that the global, annual, non-market value of the earth's ecosystem services is $33 trillion, a figure substantially greater than the global GDP. In assessing the "real" economy, it is stated (Costanza, 2005) that everything which contributes to real, sustainable, human welfare, not simply the "market" economy, must be measured, including all non-marketed contributions, one of which is nature. Non-market contributions of this sort can be grouped as capital types, for example natural capital, and in the same way as the impact of the aforementioned Eden Project, this natural capital can help to support the real, human-welfare-producing economy in terms of its education value and effect on well being. Therefore, the protection and indeed enhancement of natural capital should be a key concern and consideration for industry.

The social impact of restoration projects should also be a significant consideration at the planning stage. In the case of the slate quarrying industry in north Wales, for example, many generations of the same families have worked in and died while working for the industry. The landscape that remains serves as a reminder of the lives of these workers, and many small towns and villages would not exist today had it not been for the slate industry. It is not always socially acceptable to introduce rapid restoration schemes to green over and cover up these perceived landscape scars. "Ethno-restoration" (Balaguer, pers. comm., 2005) is a concept that was discussed at the 2005 SER World conference; it is a term that promotes the idea of consulting people local to proposed restoration schemes, in particular older generations, and incorporating their thoughts and ideas of how, if at all, components of restoration projects should be developed or put into place. This should be a key focus for many restoration projects, especially given that, by their very nature, these projects often take place within environmentally sensitive areas (Cripps *et al.*, 2007).

There are additional and often unappreciated (and indeed previously unaccounted for) benefits of natural ecosystems to human societies that are far reaching in their positive effects, not only in the sense of their value as described by Costanza (2005), but also in terms of buffering the negative impacts of human-kind as a whole. Current government environmental focus is being placed on reducing carbon emissions to meet limits and agreements set out in the Kyoto Protocol; as a result emphasis is being placed on "carbon sinks", and frequent references are made to the "carbon footprint" of products and practices, with the overall aim of becoming as close to "carbon neutral" as possible. These targets indirectly promote the concept of post-industrial restoration; damaged land of low or no ecological value and with little capacity for carbon sequestration is restored to a functioning habitat and vegetation capable of actively locking atmospheric carbon dioxide into "carbon sinks". Material recovery from waste streams destined for landfill can play a central role in transforming post-industrial land into vegetated habitat; this has a synergistic effect on carbon footprints (i.e. the vegetation is a carbon sink, and preventing organic waste materials ending up in landfills and decomposing prevents the liberation of greenhouse gases such as methane (see section 5.2.1.4)).

The research presented in this thesis demonstrates that using an integrative approach to post-industrial ecological site restoration (e.g. the evaluation of past restoration techniques and methods (chapter 3); observations of how and where natural colonisation is taking place; detailed investigations of what planting stock is most successful in the substrates destined for restoration (chapter 4); and the utilisation of restoration materials of proven quality and sustainable nature (chapter 5)), slate waste tips can be re-vegetated in a short period of time, with limited associated costs and minimal levels of intervention or disturbance. This can provide areas of nurse-crop cover susceptible to colonisation from local populations depositing propagules as seed-rain. In turn, this will allow continued habitat development and further stabilisation.

Thus the simple methods developed and presented here provide the first step towards habitat stabilisation, development and self-sufficiency, enabling succession to lead to the establishment of the target habitat types of upland heath-land and woodland assemblages.

The methods developed in this research have built on and adapted past techniques (see chapter 3) to provide a low-cost, low-tech, sustainable and rapid form of restoration that has exhibited levels of success previously unachieved, despite many attempts, in the extremely harsh conditions present at Penrhyn Quarry.

Carrying out restoration projects of this type and reinstating vegetation cover to denuded areas created by industrial processes will help to contribute to both the total natural capital and to the sequestration of atmospheric carbon concentrations. However, it is considered that there is scope for developing these methods further to increase their efficacy. For example, research carried out by Landcare Research (Ross *et al.,* 2003) and Hortresearch (Stanley *et al.,* 2000) in New Zealand has focussed on developing techniques that include application of bryophytes as inocula of spores or distribution of small moss fragments as part of the restoration process. The inclusion of such methods into the techniques developed and used in this thesis have the potential to increase both the rate of substrate stabilisation and the rate and/or success of vegetation establishment, and are therefore recommended for testing in future experimentation. Additionally, it was initially intended to focus a part of this research on social surveys to address the question of public input into restoration projects, the influence the public can have, and the public interpretation of quarry restoration as a whole. However, due to the time demands associated with setting up large-scale field experiments this was not accomplished. It is, however, recommended that future restoration projects of this type consult the public, as their views are integral to the efficiency and success of such projects.

Despite the restoration success that has been achieved and demonstrated in this thesis, an alternative option for post-industrial restoration exists, to not carry out any restorative practices at all. Dr. E. Rowe (pers. comm., 2006) suggested that in some instances (such as those presented by Penrhyn Quarry) where habitats such as inland cliffs and acid-scree have been created that are not widespread, perhaps the main emphasis should be on preserving them for their interesting mix of early colonists, successional assemblages, nesting sites and general high ecological value, instead of simply replacing them with restored habitats of low (initially at least) ecological value to satisfy planning demands placed upon industries by local authorities to enable the continued working of the land and further material exploitation. Restoration choices are important ones that should not be rushed, because even after the initial habitat damage or destruction has taken place, the resulting land-formation or spontaneously-arising vegetation assemblages can be of high ecological value or significance, and as such should not then be destroyed by implementing a poorly-thought-out restoration scheme.

It is apparent that many options (e.g. abandonment and natural regeneration, rapid greening to conceal landscape scars, or carefully planned and tested ecological restoration schemes) are available for land affected by industrial processes such as slate quarrying, and that in even the most difficult of situations successful restoration can be achieved. Each option has inherent associated costs and benefits, and these should be carefully assessed and evaluated in order to achieve restoration outcomes that are acceptable to all concerned parties. Where human activities, such as quarrying, have deliberately destroyed natural habitats, the assumption should be that restoration practices are carried out following the cessation of extractive and damaging processes unless it can be shown that the newly created post-industrial habitat has either an ecological value or a social or cultural value that exceeds that of the proposed restored habitat.

The key findings of this research are:

- Chapter 3-
    - The greatest levels of long-term restoration success are achieved only by introducing large quantities of restoration materials to slate waste tip surfaces.
    - Nutritional amelioration appears to have a greater effect on vegetation development when growing on slate waste than water-holding amendments.
    - Organic forms of nutritional amelioration are more effective than inorganic/mineral sources in promoting vegetation development on slate waste tips.

- Chapter 4 –
    - Differences in growth and survival between quarry and non-quarry populations of tree and shrub species grown in slate waste tip conditions were recorded. However, no consistent trend was observed and as such it cannot be recommended that future restoration planting uses only very locally collected plant material.

- Chapter 5 –
    - Restoration applications of granular PAM and Conwy County Council GWC (50:50 mixture) to slate waste tip surfaces achieved significantly greater levels of vegetation development when compared to the experimental control.

7 Appendices

Appendix 1 Example of available branded SAPs

Table A.1 Example of available branded SAPs

SAP classification	SAP brand name	Reference
Polyacrylamide	Broadleaf P4	Choudhary *et al.*, 1995
	Aquasorb	Choudhary *et al.*, 1995
	Aquastore	Anon, 2008
	Luquasorb	Elliot, 2004
	Hydropol	DTI, 2006
	Hydrosorb	Anon, 2002
Polyacrylate	Hysorb	Elliot, 2004
	Agrihope	Choudhary *et al.*, 1995
Acrylates, co-polymers, and other SAP materials	Hydrogel	Choudhary *et al.*, 1995
	Terracottem	Edwards, 2006
	Gelscape	Gabrosky, 1998

Appendix 2 Preliminary WHC trials

To record the maximum WHC of the SAPs a vessel that could contain the material whilst dry and un-hydrated, yet allow water to freely surround the SAP to facilitate complete hydration was needed. Using a sharp knife, the tapered end of a 50 ml centrifuge tube was removed. To the flat end of the centrifuge tube a small piece of fine plastic mesh (mesh size ~200 μm) was attached using a rubber elastic band and silicone sealant, this was repeated to produce a set of these vessels. A small amount of the SAP material (an amount equal to approximately 10 ml of the centrifuge tube (equal to 7 – 10 g dry weight of PAM and Na-PA, and 4- 6 g wall paper paste) was weighed into these vessels, they were placed into a test tube rack and submersed in a tank containing approximately 3000 ml of distilled water. After 5 hours saturating the Na-PA filled vessels had absorbed a large quantity of the distilled water to the point where the material had erupted out of the vessel and into the tank. Both the PAM and wallpaper paste had absorbed a small quantity of distilled water and formed an impervious plug in the bottom of the vessel, therefore preventing any subsequent water uptake. At this point the experiment was aborted. It was noted that although no definite WHC measurement for Na-PA could be made, the contents of the five replicate vessels ($5 \times \leq 10$ g dry Na-PA) had absorbed the whole content of distilled water present within the tank (~3000 ml).

The design of the SAP containing vessels was subsequently developed to allow greater movement of water in and around the SAP material, with a greater volume than that present in the centrifuge tube design. Fine plastic mesh as used previously was cut into strips, folded in half and the edges were stapled together to form a packet. Silicone sealant was then applied to seal the stapled edges forming a mesh packet with no peripheral gaps. Into these mesh packets 0.5000 g SAP was weighed, the packets were then placed inside 800 ml glass beakers to which approximately 200 ml distilled water was added, ensuring the packets were fully submerged. After approximately 5 hours the distilled water was removed and the packets left to drain. In the packets containing Na-PA a small fraction of the SAP had passed through the mesh into the beaker, therefore a smaller grade mesh was required. Both PAM and Na-PA had hydrated and swollen greatly and had filled the volume of the packets and began overflowing into the beakers, therefore 0.5000 g dry SAP was too great an amount for use in these packets. Weighing to determine the WHC for the SAPs was carried out after approximately 1.5 hours of draining.

Appendix 3 Restoration seed mixture criteria

Seed inclusion into a soil-forming matrix was an important criterion for this experiment. Its inclusion was primarily aimed at developing a rapid, efficient and economically viable method for restoring slate waste tips. It was therefore imperative that a mix of seed both suited to growing in hostile conditions and representative of local flora was formulated and utilised. For guidance regarding seed requirements the British Seed Houses Limited (BSH) were consulted. BSH have significant knowledge with regard to site-specific seed mixtures as they represent the largest privately owned seed company in the UK. Having worked closely with IGER (Institute of Grassland and Environmental Research) since 1987 in development of the largest UK based grass-breeding programme they were the ideal candidates to help develop a quarry restoration seed mixture.

A standard amenity seed mixture produced by BSH was suggested as a starting block for the quarry restoration mix, this formula is marketed as "A15-reclamation landfill sites". The key properties of this seed mix are its suitability for growth in a wide range of soil types and qualities, ability to withstand herbivorous grazing pressure, and rapidity of ground cover. The seed mix consists of-

35 % *Festuca rubra* ssp. *rubra* variety "Adinda" (strong creeping red fescue)
30 % *Lolium perenne* variety "Talgo" (perennial ryegrass)
15 % *Poa compressa* variety "Reubens" (flattened meadow grass)
10 % *Festuca rubra* ssp. *commutata* variety "Olivia" (chewings fescue)
7.5 % *Agrostis capillaris / Agrostis tenuis* variety "Highland" (browntop bent)
2.5 % *Trifolium repens* variety "Aberace" (small leaved white clover)

-and is recommended for sowing at 25 g m^{-2} (equivalent to 250 kg ha^{-1}). Species such as *Poa compressa* are utilised in this mixture, offering early colonising potential on disturbed sites and resilience to wide pH ranges and toxin presence. This species is often used in areas such as gravel pits or roadside developments, for example verges and cuttings (Sykes, 2000) therefore offers potential for successful development on slate waste tips.

Species such as *Purpurea digitalis, Teucrium scorodonia, Ulex europaeus,* and *Cytisus scoparius* are prevalent throughout the quarry complex at Penrhyn. It was therefore decided these species are obviously well suited to growth in conditions presented by slate waste and should be included in the restoration seed mix. Inclusion of *Ulex europaeus* and *Cytisus scoparius* was considered important as it provided a chance to observe the establishment rates and development of woody plant species when included in restoration activities on slate waste tips. Ideally a mixture of locally relevant tree species seed would have been included into the restoration seed mix, but due to time constraints associated with this limited study period it was considered to be of no value with regard to the potential for tree growth from seed in an 18 month period.

Therefore the BSH "A15" seed mix was slightly modified to incorporate site specific requirements.

The greatest alteration to the marketed seed mix was the significant reduction of *Lolium perenne* content, it was hoped that its reduction would reduce the potential out-competition of the included shrub and wild flower species. Recommendation by BSH technical advisors was that this seed mix should be applied at a rate of 5 g m^{-2} in contrast to the usual application rate (25 g m^{-2}), therefore preventing a monoculture of dominant grass species to establish, allowing wild flowers and others species to compete successfully. Woody shrub species were included at a rate of approximately 50 seeds m^{-2}; this was intended to allow approximately 5 plants m^{-2} to develop assuming 10 % survival.

BSH recommend the use of fertiliser in conjunction with marketed seed mixes. A range of pre and post seeding fertilisers are marketed, specifically associated to individual seed mixtures produced. Two fertiliser mixes are recommended for use with seed mix "A15", with 10:15:10 and 16:16:16 N:P:K ratios respectively, and both are recommended for spreading at 70 g m^{-2}. The nutritional characteristics of habitats present within and surrounding the quarry are generally nutrient poor. As such introduction of high quantities of nutrition by fertilisation within a site adjacent to environmentally important and sensitive habitats would have associated potential negative impacts. Also nutrient application to un-vegetated successional land may result in opportunist weed species to develop and flourish presenting implications for subsequent eradication or removal. With regard to these considerations it was proposed that fertiliser, pending results of subsequent greenhouse growth studies, be used in conjunction with a restoration seed mix, but in a low nitrogen form (6:9:6 ratio N:P:K) and at a greatly reduced rate of 20 g m^{-2}.

Appendix 4 Application of straw as a restorative material for slate waste

Following consultation with restoration practitioners it was decided to add a supplementary plot to each experimental site. This plot was set aside to test restoration potential of applying spent straw to slate waste tip surfaces. The additional straw plot was not to be included within the block structure of the experimental design, merely acting as a bolt-on treatment.

Once all straw, seed and fertiliser had been positioned a roll of hessian netting was placed over the plot and weighted down with blocks of slate, therefore securing straw from wind disturbance. It was felt this was an important measure to carryout as when applied the straw was dry and potentially mobile in moderate winds. To test the impact of wind disturbance on uncovered straw two 2×2 m plots were marked out adjacent to each experimental site, onto which an amount of straw equivalent to that spread on experimental plots was positioned and left uncovered. Soon after positioning the straw upon the two test plots a substantial rainfall event occurred, subsequent to this very little straw movement was observed, only small disturbances caused by inquisitive herbivores.

Appendix 5 Information regarding soil-forming material spreading

Soil forming materials were applied to a depth of 75 mm depth on all experimental plots, (i.e. 1.125 m^3 of mixture per plot) (15 (m^2) $\times$ 0.075 (m)) material per plot. If materials from all treatments, for example GWC, for both experimental sites are combined (assuming 50:50 % split in 2 material plots, and 33:33:33 % split in 3 material plots) it equates to each material covering 14 experimental plots. The total area per material is equivalent to 210 m^2 (14 $\times$ 15 m^2), which is therefore equates to 15.75 m^3 (14 $\times$ 1.125 m^3) per material. These calculations account for control plots totalling 90 m^2 (6 $\times$ 15 m^2) requiring no material application, and 30 m^2 (2 $\times$ 15 m^2) straw plots requiring 2.25 m^3 straw. Accumulated all materials required for application to waste tip surfaces in this experiment totalled 49.5 m^3.

Materials destined for use in these trials were quickly assessed for bulk mass prior to spreading. Three one litre replicates for both slate and GWC were measured out and weighed. Mean slate bulk mass equated to 1600 g $litre^{-1}$, and GWC 860 g $litre^{-1}$, therefore equivalent to 1.6 tonnes m^{-3} and 860 kg m^{-3} respectively.

Appendix 6 Development of an application chute for spreading soil-forming materials on to slate waste tips

Much consideration was focussed upon the development of an efficient material spreading method for this set of experimental trials. This was an important factor to contemplate because of the sheer practicality of spreading 49.5 m^3 upon 750 m^2 of slate scree angled at 45°. Availability of mini-diggers, excavators or machinery in general was not a possibility due to site access and stability, and the accuracy required for these experimental trials. If therefore, materials were to be hand spread it would be a phenomenally labour intensive exercise. It was therefore proposed that a material application chute could be constructed. This would allow material to be applied to waste tip slopes directly from the 4×4 pick-up vehicle via the application chute. Construction of the application chute was principally made up from a chain of connected dustbins, of which the bottoms had been removed by sawing off.

Trials testing the efficiency of the material application chute were carried out (plate 12). The chute was connected to the tailgate of the 4×4 pick-up vehicle and laid out on the waste tip slope in the respective experimental plot, in this instance GWC (plot 3, block 1, site 1). Shovelling of material into the chute commenced, initially it appeared that the GWC was travelling down the application chute quite successfully. Quickly it became apparent that the material within the chute was not exiting it rapidly enough onto the experimental plot. This was causing the GWC to back-up, physical agitation of the chute failed to alleviate this problem. Work to try and move the blockage was extremely challenging, because the chute now contained approximately one cubic metre of GWC equivalent to approximately 860 kg. It was apparent that this method could not be employed for the experimental application of material to waste tip slopes at the two designated experimental sites. It is considered that the main crux of this technique was that the slope angle was slightly too shallow, if it were just several degrees steeper the material would have flowed quickly enough through the chute to prevent blockage.

Failure of the material application chute technique, dictated that all material for this set of experimental field trials, was required to be hand distributed.

Plate 12 Material application chute connected to the pick-up tailgate

Appendix 7 Economic assessment of soil-forming material spreading

Total man-hours spent applying restoration materials = 300.5

Total area covered = 750 m^2

Therefore the area of slate waste tip restored/treated man-hour^{-1} = 2.496 m^2

Table A.2 Estimated costs associated to hand applied experimental restoration materials

Equipment	Cost (£)	Unit	Relative cost m^{-2} (£)
4×4 vehicle	117.50	Week	1.2553
Concrete mixer	37.60	Week	0.4017
Compost haulage	100	N/A	0.1333
Seed mix	99.88	kg	0.4994
Fertilizer	27.02	25 kg	0.2162
PAM	138.06	25 kg	0.4143
Slate	10	Tonne^{-1}	1.2004
Green waste compost	Free	N/A	Nil
Total	**530.06**	**N/A**	**4.1206**

N.B. Estimates assuming 37.5 hour week^{-1}, and 2.496 m^2 hour^{-1}.

Costs not included into above estimates for this experiment, are labour, fuel, and consumables.

Appendix 8 Novel method of applying soil-forming materials to slate waste tips

Development of restoration material application method

Following experimental work to develop a successful soil forming material for restorative application to slate waste tips, trials were carried out to develop a viable method for applying this material. This was an important consideration, as manual application of materials by hand, as was conducted throughout prior experimental trials, can be considered neither practical, efficient nor economically viable for large-scale restoration projects. Development of an application method had to incorporate a degree of flexibility, allowing easy transport in challenging terrain, as presented in the quarry complex, and the ability to apply material to slate waste tip surfaces, both up and down slope, and amongst large blocky waste boulder fields.

Initial considerations were focussed upon modifying established methods for applying vegetative matter to land, such as hydro-seeding. This method utilises a pressurized tank of water into which grass seed and tackifiers have been added. Material is then sprayed through a hose-like nozzle onto rock faces, roadside cuttings and embankments *et cetera*. This technique is limited to the use of seed in solution, with no, or limited inclusion of solids, therefore offering no potential for use with mixtures as developed for vegetation establishment in the preceding material trials. As such, a broader range of equipment and machinery, suitable for movement of liquids containing solids, slurry, and other wet mixtures, was investigated. Initial optimism was placed upon a range of five pumps supplied by Jewson limited tool hire, of which two stated their capability to pass solids up to 25 mm diameter. These were soon ruled out following technical advise stating that the intended pump would not be suitable to pump thick slurry.

Eventually a more suitable range of pumping equipment was sourced from Selwood Pump Hire and Sales Group Limited. Following extensive consultation with a technical advisor, and inspection of the material required to be pumped, it was recommended that a PD75 “Spate” general-purpose diaphragm pump, would be the most suitable piece of equipment for the proposed spreading activities. The technical specification for this pump states capabilities of passing 6 mm diameter solids, 30 m^3 $hour^{-1}$ flow rate, and 30.5 m total head.

Experimental set up for pumping trial

To carryout experimental applications of soil forming materials to slate waste tip surfaces; using the suggested pumping equipment, the following procedures were followed. Compost, again sourced from Conwy, was mixed in a 50:50 ratio with hydrated granular polyacrylamide (the successful mixture determined in prior experimental observations). Mixing was carried out as previously used for experimental material applications, via addition of constituent materials, restoration seed mix, and slow release granular fertiliser, into a 90 litre concrete mixer. Mixed material was emptied from the concrete mixer into a storage tank. When enough material had been mixed to fill the storage tank, it was pumped into a 1000 litre intermediate bulk container (IBC) mounted in a 4 × 4 vehicle. At the point when the IBC was full, it was transported to the restoration site via the 4 × 4.

At this point the process failed due to pump technicalities. However, the following passage describes the envisaged method of continuing the process, and applying the restorative materials to slate waste tip surfaces.

Upon reaching the site within the quarry complex due for restorative material applications, the pump was unloaded from the vehicle, hoses attached and placed, one end within the stock IBC, and one end on slope, at the point of material application. Once running, the pump delivers all material through the hose system to an operator positioned on the slate waste tip, material would therefore be efficiently delivered up or down-slope, and effectively manoeuvred around until required objectives are fulfilled.

However, this process was unable to be fulfilled. At the point of transferring the mixed artificial soil material from the storage vessel, to the 4 × 4 mounted IBC, the pump actually failed to deliver any material. As such, the pump and all hoses were cleared of any blockages, and the process was initiated once again, albeit with a substantially watered down artificial soil mixture. Unfortunately, the pump continually failed to pass the mixture of materials.

Following consultation with an advisor from Selwood Pumps Ltd., a sample mixture of the artificial soil materials was provided to company operatives, such that a range of pumps could be tested with it. However, following trials with a solids handling pump ("Seltorque" S100), equipment that can pass solids of 75 mm diameter, the material again failed to be pumped.

Due to time constraints, it was decided to terminate this set of trials. It was suggested however, that a hydraulic screw pump, equipment that is submersed within the material required for pumping, would probably fulfil the requirements of this activity. Failing this, utilisation of agricultural equipment, such as pressurized slurry tankers, would also potentially delivery this material.

If the initial pumping equipment had worked successfully, it is proposed that this method would provide an effective, efficient, and economic technique to apply restorative materials to slate waste tips. Material application rates utilised in experiments described in section 5.2, consisted of 75000 cm^3 m^{-2}, and pumping capabilities of the "Spate" pump are stated as 30 m^3 $hour^{-1}$. Therefore, it can be assumed that theoretically it would be possible to apply restorative material to 400 m^2 $hour^{-1}$. Costs associated to this spreading process are minimal. All costs involved in this demonstration are presented in table A.3 (minus operator rates, consumables, and fuel for pump, 4×4 vehicle, and concrete mixer).

Table A.3 Costs associated to application of restorative materials to slate waste tips

Equipment	Cost (£)	Unit	Relative cost m^{-2} (£)
Selwood PD75 "Spate" pump	50	Week	0.0033
4×4 vehicle	117.50	Week	0.0078
Concrete mixer	37.60	Week	0.0025
Compost haulage	100	N/A	0.0066
Seed mix	25	kg	0.1250
Fertiliser	27.02	25 kg	0.2162
PAM	138.06	25 kg	0.4143
Total	**495.18**	**N/A**	**0.7757**

N.B. Relative cost estimates based on pump capability of 30 m^3 $hour^{-1}$, 7.5 hour working day, and 5 day working week, therefore assuming restoration of 15000 m^2 $week^{-1}$.

Despite problems encountered during this material application trial, the described method is considered to offer great potential in terms of efficient and cost effective application of materials for restoration of challenging sites. Enough evidence of the potential of this method was demonstrated during the initial stages of the trial, to suggest that it would successfully work if a suitable piece of pumping equipment were identified. If some assumptions are made, the cost of this restoration technique can be estimated. If for example the restoration operator rate was equivalent to £20 $hour^{-1}$, with £10 day^{-1} consumables charge (fuel etc.), and materials were applied at the optimum rate of deliver for one week, the total cost (labour and materials) for application of restorative materials to 15000 m^2 would total £12435.50 (£8290.33 ha^{-1}). Factoring in data collected from vegetation surveys carried out in section 5.3, it can be assumed that, within 18 months following application, up to 95.2 % total vegetation cover could be achieved by utilising this material and method.

Current restoration practice carried out by practitioners employed at Penrhyn Quarry, involves transplantation of nursery raised tree stock. Involved in this process are many hidden costs. For example, only locally collected seed stock is raised, therefore time is required for collection; time is required to process collected seed i.e. de-fleshing, scarifying, soaking, chilling, to break seed dormancy; sowing of seed; transplanting germinants to root trainers;

transplanting year old saplings into slate waste. Throughout the process of raising this planting stock, time and resources are also required for watering and feeding. Materials and equipment are also used throughout this process, including seed compost, seed trays and root training cells. Upon transplantation of nursery-raised stock to slate waste tips, further costs are incurred, firstly operator time, and utilisation of tree shelter tubes and stakes. All processes combined contribute to sizeable periods of practitioner time, and consumption of resources. Table A.4 shows an estimate of costs associated in raising and transplanting a year-old tree sapling of one species.

Table A.4 An estimate of costs associated with nursery raised tree stock

Activity	Time involved (minutes)	Estimated cost per plant (£)
Seed collection	180	0.108
Seed processing	120	0.072
Sowing seeds	30	0.018
Pricking out germinants	450	0.270
Husbandry (watering etc.) year^{-1}	2600	1.560
Transplantation to slate waste	30	9.00
Total	**3410**	**11.028**

N.B. Assuming practitioner rate of £18 hour^{-1}, and total number of saplings raised from one seed batch to be 500.

If therefore, saplings raised in a manner as described, were planted out at a density of 3000 ha^{-1}, the total cost associated would be equivalent to £33084 ha^{-1} slate waste tip restored, representing an excess cost of £24793.67 over that proposed via utilisation of the pumped material application technique. It should also be considered a high percentage of transplanted tree saplings present within slate waste tip surfaces suffer from severely retarded growth, seed check, and high mortality rates, therefore no estimate of vegetation cover development can be proposed.

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